INTERGOVERNMENTAL PANEL ON climate change

Climate Change and Land

An IPCC Special Report on climate change, desertification, land degradation, sustainable land management, food security, and greenhouse gas fluxes in terrestrial ecosystems

Summary for Policymakers

WG I WG II WG III

WMO UNEP

IPCC Special Report on Climate Change, Desertification, Land Degradation, Sustainable Land Management, Food Security, and Greenhouse gas fluxes in Terrestrial Ecosystems

Summary for Policymakers
Approved Draft

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Introduction

This Special Report on Climate Change and Land[1] responds to the Panel decision in 2016 to prepare three Special Reports[2] during the Sixth Assessment cycle, taking account of proposals from governments and observer organizations[3]. This report addresses greenhouse gas (GHG) fluxes in land-based ecosystems , land use and sustainable land management[4] in relation to climate change adaptation and mitigation, desertification[5], land degradation[6] and food security[7]. This report follows the publication of other recent reports, including the IPCC *Special Report on Global Warming of 1.5°C* (SR15), the thematic assessment of the Intergovernmental Science Policy Platform on Biodiversity and Ecosystem Services (IPBES) on Land Degradation and Restoration, the IPBES Global Assessment Report on Biodiversity and Ecosystem Services, and the Global Land Outlook of the UN Convention to Combat Desertification (UNCCD). This report provides an updated assessment of the current state of knowledge[8] while striving for coherence and complementarity with other recent reports.

This Summary for Policymakers (SPM) is structured in four parts: *A) People, land and climate in a warming world; B) Adaptation and mitigation response options; C) Enabling response options; and D) Action in the near-term.*

Confidence in key findings is indicated using the IPCC calibrated language[9]; the underlying scientific basis of each key finding is indicated by references to the main report.

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[1] The terrestrial portion of the biosphere that comprises the natural resources (soil, near-surface air, vegetation and other biota, and water), the ecological processes, topography, and human settlements and infrastructure that operate within that system.
[2] The three Special reports are: "Global Warming of 1.5°C. An IPCC special report on the impacts of global warming of 1.5°C above pre-industrial levels and related global greenhouse gas emission pathways, in the context of strengthening the global response to the threat of climate change, sustainable development, and efforts to eradicate poverty."; "Climate Change and Land: an IPCC Special Report on Climate Change, Desertification, Land Degradation, Sustainable Land Management, Food Security, and Greenhouse gas fluxes in Terrestrial Ecosystems"; "The Ocean and Cryosphere in a Changing Climate"
[3] related proposals were: climate change and desertification; desertification with regional aspects; land degradation – an assessment of the interlinkages and integrated strategies for mitigation and adaptation; agriculture, foresty and other landuse; food and agriculture; and food security and climate change.
[4] Sustainable Land Management is defined in this report as "the stewardship and use of land resources, including soils, water, animals and plants, to meet changing human needs, while simultaneously ensuring the long-term productive potential of these resources and the maintenance of their environmental functions".
[5] Desertification is defined in this report as 'land degradation in arid, semi-arid, and dry sub-humid areas resulting from many factors, including climatic variations and human activities'.
[6] Land degradation is defined in this report as 'a negative trend in land condition, caused by direct or indirect human induced processes, including anthropogenic climate change, expressed as long-term reduction and as loss of at least one of the following: biological productivity, ecological integrity, or value to humans'.
[7] Food security is defined in this report as 'a situation that exists when all people, at all times, have physical, social, and economic access to sufficient, safe and nutritious food that meets their dietary needs and food preferences for an active and healthy life'.
[8] The assessment covers literature accepted for publication by 7th April 2019.
[9] Each finding is grounded in an evaluation of underlying evidence and agreement. A level of confidence is expressed using five qualifiers: very low, low, medium, high and very high, and typeset in italics, for example, medium

A. People, land and climate in a warming world

A 1. Land provides the principal basis for human livelihoods and well-being including the supply of food, freshwater and multiple other ecosystem services, as well as biodiversity. Human use directly affects more than 70% (*likely* 69-76%) of the global, ice-free land surface (*high confidence*). Land also plays an important role in the climate system. {1.1, 1.2, 2.3, 2.4, Figure SPM.1}

A1.1. People currently use one quarter to one third of land's potential net primary production[10] for food, feed, fibre, timber and energy. Land provides the basis for many other ecosystem functions and services[11], including cultural and regulating services, that are essential for humanity (*high confidence*). In one economic approach, the world's terrestrial ecosystem services have been valued on an annual basis to be approximately equivalent to the annual global Gross Domestic Product[12] (*medium confidence*). {1.1, 1.2, 3.2, 4.1, 5.1, 5.5, Figure SPM.1}

A1.2. Land is both a source and a sink of greenhouse gases (GHGs) and plays a key role in the exchange of energy, water and aerosols between the land surface and atmosphere. Land ecosystems and biodiversity are vulnerable to ongoing climate change and weather and climate extremes, to different extents. Sustainable land management can contribute to reducing the negative impacts of multiple stressors, including climate change, on ecosystems and societies (*high confidence*). {1.1, 1.2, 3.2, 4.1, 5.1, 5.5, Figure SPM.1}

A1.3. Data available since 1961[13] show that global population growth and changes in per capita consumption of food, feed, fibre, timber and energy have caused unprecedented rates of land and freshwater use (*very high confidence*) with agriculture currently accounting for ca. 70% of global fresh-water use (*medium confidence*). Expansion of areas under agriculture and forestry, including commercial production, and enhanced agriculture and forestry productivity have supported consumption and food availability for a growing population *(high confidence).* With

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confidence. The following terms have been used to indicate the assessed likelihood of an outcome or a result: virtually certain 99–100% probability, very likely 90–100%, likely 66–100%, about as likely as not 33–66%, unlikely 0–33%, very unlikely 0–10%, exceptionally unlikely 0–1%. Additional terms (extremely likely 95–100%, more likely than not >50–100%, more unlikely than likely 0–<50%, extremely unlikely 0–5%) may also be used when appropriate. Assessed likelihood is typeset in italics, for example, very likely. This is consistent with IPCC AR5.

[10] Land's potential net primary production (NPP) is defined in this report as the amount of carbon accumulated through photosynthesis minus the amount lost by plant respiration over a specified time period that would prevail in the absence of land use.

[11] In its conceptual framework, IPBES uses "nature's contribution to people" in which it includes ecosystem goods and services.

[12] i.e. estimated at $75 trillion for 2011, based on US dollars for 2007.

[13] This statement is based on the most comprehensive data from national statistics available within FAOSTAT, which starts in 1961. This does not imply that the changes started in 1961. Land use changes have been taking place from well before the pre-industrial period to the present.

large regional variation, these changes have contributed to increasing net GHG emissions (*very high confidence*), loss of natural ecosystems (e.g. forests, savannahs, natural grasslands and wetlands) and declining biodiversity (*high confidence*). {1.1, 1.3, 5.1, 5.5, Figure SPM.1}

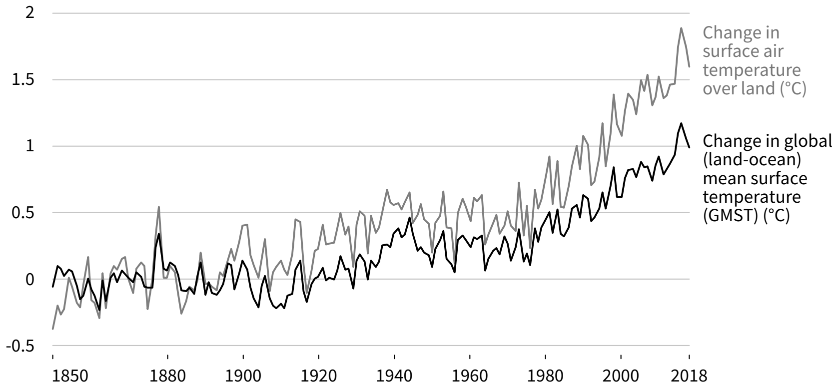
A1.4. Data available since 1961 shows the per capita supply of vegetable oils and meat has more than doubled and the supply of food calories per capita has increased by about one third (*high confidence*). Currently, 25-30% of total food produced is lost or wasted (*medium confidence*). These factors are associated with additional GHG emissions (*high confidence*). Changes in consumption patterns have contributed to about 2 billion adults now being overweight or obese (*high confidence*). An estimated 821 million people are still undernourished (*high confidence*). {1.1, 1.3, 5.1, 5.5, Figure SPM.1}

A1.5. About a quarter of the Earth's ice-free land area is subject to human-induced degradation (*medium confidence*). Soil erosion from agricultural fields is estimated to be currently 10 to 20 times (no tillage) to more than 100 times (conventional tillage) higher than the soil formation rate (*medium confidence*). Climate change exacerbates land degradation, particularly in low-lying coastal areas, river deltas, drylands and in permafrost areas (*high confidence*). Over the period 1961-2013, the annual area of drylands in drought has increased, on average by slightly more than 1% per year, with large inter-annual variability. In 2015, about 500 (380-620) million people lived within areas which experienced desertification between the 1980s and 2000s. The highest numbers of people affected are in South and East Asia, the circum Sahara region including North Africa, and the Middle East including the Arabian peninsula (*low confidence*). Other dryland regions have also experienced desertification. People living in already degraded or desertified areas are increasingly negatively affected by climate change (*high confidence*). {1.1, 1.2, 3.1, 3.2, 4.1, 4.2, 4.3, Figure SPM.1}

Land use and observed climate change

A. Observed temperature change relative to 1850-1900
Since the pre-industrial period (1850-1900) the observed mean land surface air temperature has risen considerably more than the global mean surface (land and ocean) temperature (GMST).

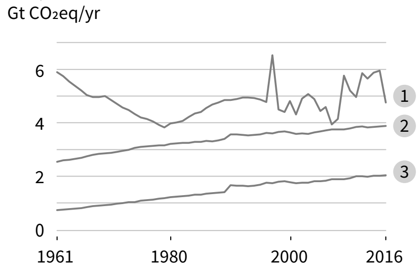
CHANGE in TEMPERATURE rel. to 1850-1900 (°C)

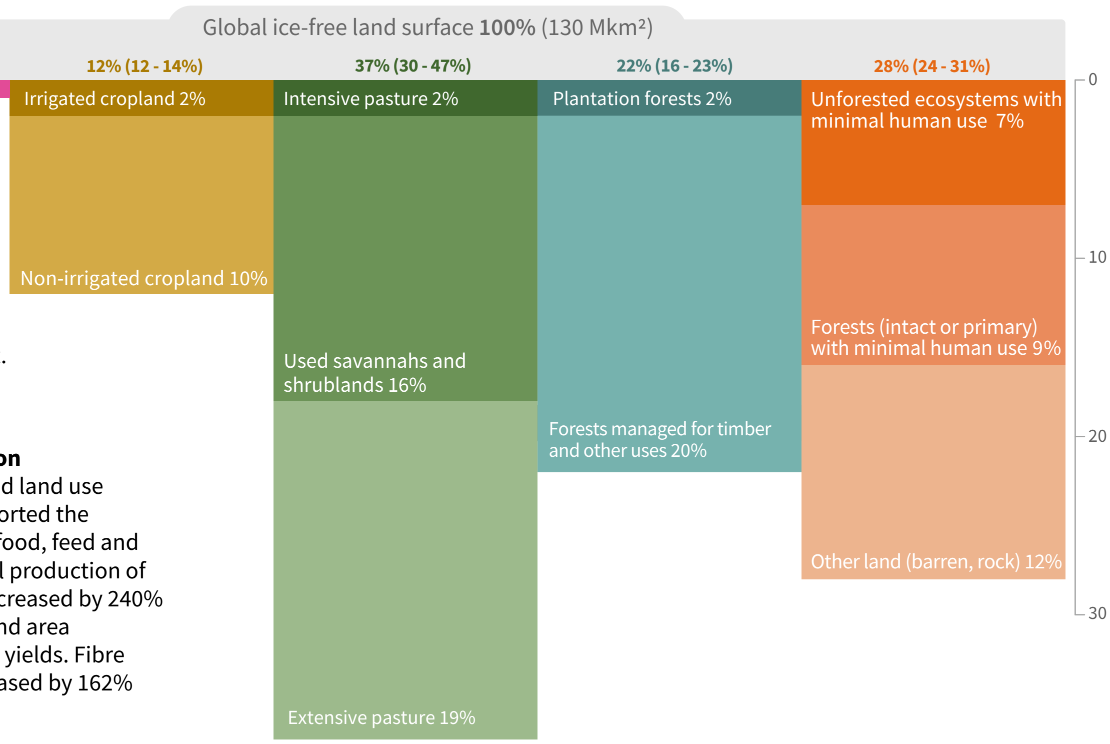


B. GHG emissions
An estimated 23% of total anthropogenic greenhouse gas emissions (2007-2016) derive from Agriculture, Forestry and Other Land Use (AFOLU).

CHANGE in emissions rel. to 1961

1. Net CO_2 emissions from FOLU (Gt CO_2/yr)
2. CH_4 emissions from Agriculture (Gt CO_2eq/yr)
3. N_2O emissions from Agriculture (Gt CO_2eq/yr)



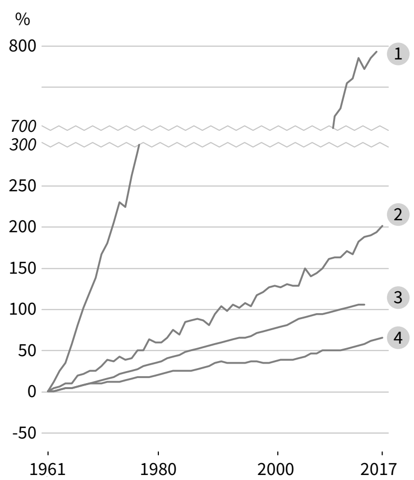


C. Global land use in circa 2015
The barchart depicts shares of different uses of the global, ice-free land area. Bars are ordered along a gradient of decreasing land-use intensity from left to right.

D. Agricultural production
Land use change and rapid land use intensification have supported the increasing production of food, feed and fibre. Since 1961, the total production of food (cereal crops) has increased by 240% (until 2017) because of land area expansion and increasing yields. Fibre production (cotton) increased by 162% (until 2013).

CHANGE in % rel. to 1961

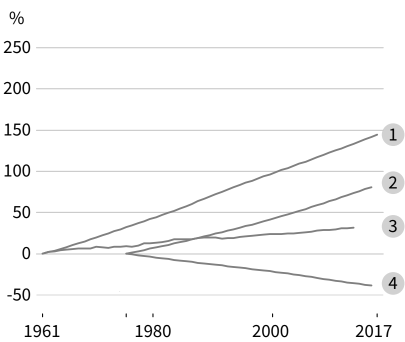
1. Inorganic N fertiliser use
2. Cereal yields
3. Irrigation water volume
4. Total number of ruminant livestock



E. Food demand
Increases in production are linked to consumption changes.

CHANGE in % rel. to 1961 and 1975

1. Population
2. Prevalence of overweight + obese
3. Total calories per capita
4. Prevalence of underweight



F. Desertification and land degradation
Land-use change, land-use intensification and climate change have contributed to desertification and land degradation.

CHANGE in % rel. to 1961 and 1970

1. Population in areas experiencing desertification
2. Dryland areas in drought annually
3. Inland wetland extent

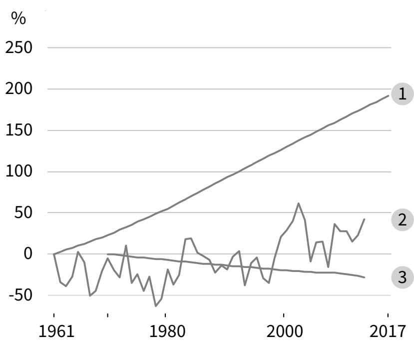


Figure SPM.1: Land use and observed climate change

A representation of the land use and observed climate change covered in this assessment report. Panels A-F show the status and trends in selected land use and climate variables that represent many of the core topics covered in this report. The annual time series in B and D-F are based on the most comprehensive, available data from national statistics, in most cases from FAOSTAT which starts in 1961. Y-axes in panels D-F are expressed relative to the starting year of the time series (rebased to zero). Data sources and notes: **A:** The warming curves are averages of four datasets {2.1; Figure 2.2; Table 2.1} **B:** N_2O and CH_4 from agriculture are from FAOSTAT; Net CO_2 emissions from FOLU using the mean of two bookkeeping models (including emissions from peatland fires since 1997). All values expressed in units of CO_2-eq are based on AR5 100 year Global Warming Potential values without climate-carbon feedbacks (N_2O=265; CH_4=28). {see Table SPM.1, 1.1, 2.3} **C:** Depicts shares of different uses of the global, ice-free land area for approximately the year 2015, ordered along a gradient of decreasing land-use intensity from left to right. Each bar represents a broad land cover category; the numbers on top are the total % of the ice-free area covered, with uncertainty ranges in brackets. Intensive pasture is defined as having a livestock density greater than 100 animals/km². The area of 'forest managed for timber and other uses' was calculated as total forest area minus 'primary/intact' forest area. {1.2, Table 1.1, Figure 1.3} **D:** Note that fertiliser use is shown on a split axis. The large percentage change in fertiliser use reflects the low level of use in 1961 and relates to both increasing fertiliser input per area as well as the expansion of fertilised cropland and grassland to increase food production. {1.1, Figure 1.3} **E:** Overweight population is defined as having a body mass index (BMI) > 25 kg m^{-2}; underweight is defined as BMI < 18.5 kg m^{-2}. {5.1, 5.2} **F:** Dryland areas were estimated using TerraClimate precipitation and potential evapotranspiration (1980-2015) to identify areas where the Aridity Index is below 0.65. Population data are from the HYDE3.2 database. Areas in drought are based on the 12-month accumulation Global Precipitation Climatology Centre Drought Index. The inland wetland extent (including peatlands) is based on aggregated data from more than 2000 time series that report changes in local wetland area over time. {3.1, 4.2, 4.6}

A 2. Since the pre-industrial period, the land surface air temperature has risen nearly twice as much as the global average temperature *(high confidence*). Climate change, including increases in frequency and intensity of extremes, has adversely impacted food security and terrestrial ecosystems as well as contributed to desertification and land degradation in many regions (*high confidence*). {2.2, 3.2, 4.2, 4.3, 4.4, 5.1, 5.2, Executive Summary Chapter 7, 7.2}

A2.1. Since the pre-industrial period (1850-1900) the observed mean land surface air temperature has risen considerably more than the global mean surface (land and ocean) temperature (GMST) (high confidence). From 1850-1900 to 2006-2015 mean land surface air temperature has increased by 1.53°C (very likely range from 1.38°C to 1.68°C) while GMST increased by 0.87°C (likely range from 0.75°C to 0.99°C). {2.2.1, Figure SPM.1}

A2.2. Warming has resulted in an increased frequency, intensity and duration of heat-related events, including heat waves[14] in most land regions (*high confidence*). Frequency and intensity of droughts has increased in some regions (including the Mediterranean, west Asia, many parts of South America, much of Africa, and north-eastern Asia) (*medium confidence*) and there

[14] A heatwave is defined in this report as 'a period of abnormally hot weather. Heatwaves and warm spells have various and in some cases overlapping definitions'.

has been an increase in the intensity of heavy precipitation events at a global scale (*medium confidence*). {2.2.5, 4.2.3, 5.2}

A2.3. Satellite observations[15] have shown vegetation greening[16] over the last three decades in parts of Asia, Europe, South America, central North America, and southeast Australia. Causes of greening include combinations of an extended growing season, nitrogen deposition, CO_2 fertilisation[17], and land management (*high confidence*). Vegetation browning[18] has been observed in some regions including northern Eurasia, parts of North America, Central Asia and the Congo Basin, largely as a result of water stress (*medium confidence*). Globally, vegetation greening has occurred over a larger area than vegetation browning (*high confidence*). {2.2.3, Box 2.3, 2.2.4, 3.2.1, 3.2.2, 4.3.1, 4.3.2, 4.6.2, 5.2.2}

A2.4. The frequency and intensity of dust storms have increased over the last few decades due to land use and land cover changes and climate-related factors in many dryland areas resulting in increasing negative impacts on human health, in regions such as the Arabian Peninsula and broader Middle East, Central Asia (*high confidence*)[19]. {2.4.1, 3.4.2}

A2.5. In some dryland areas, increased land surface air temperature and evapotranspiration and decreased precipitation amount, in interaction with climate variability and human activities, have contributed to desertification. These areas include Sub-Saharan Africa, parts of East and Central Asia, and Australia. (medium confidence) {2.2, 3.2.2, 4.4.1}

A2.6. Global warming has led to shifts of climate zones in many world regions, including expansion of arid climate zones and contraction of polar climate zones (*high confidence*). As a consequence, many plant and animal species have experienced changes in their ranges, abundances, and shifts in their seasonal activities (*high confidence*). {2.2, 3.2.2, 4.4.1}

A2.7. Climate change can exacerbate land degradation processes (*high confidence*) including through increases in rainfall intensity, flooding, drought frequency and severity, heat stress, dry spells, wind, sea-level rise and wave action, permafrost thaw with outcomes being

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[15] The interpretation of satellite observations can be affected by insufficient ground validation and sensor calibration. In addition their spatial resolution can make it difficult to resolve small-scale changes.

[16] Vegetation greening is defined in this report as an increase in photosynthetically active plant biomass which is inferred from satellite observations.

[17] CO2 fertilization is defined in this report as the enhancement of plant growth as a result of increased atmospheric carbon dioxide (CO2) concentration. The magnitude of CO2 fertilization depends on nutrients and water availability.

[18] Vegetation browning is defined in this report as a decrease in photosynthetically active plant biomass which is inferred from satellite observations.

[19] Evidence relative to such trends in dust storms and health impacts in other regions is limited in the literature assessed in this report.

modulated by land management. Ongoing coastal erosion is intensifying and impinging on more regions with sea level rise adding to land use pressure in some regions (*medium confidence*). {4.2.1, 4.2.2, 4.2.3, 4.4.1, 4.4.2, 4.9.6, Table 4.1, 7.2.1, 7.2.2}

A2.8. Climate change has already affected food security due to warming, changing precipitation patterns, and greater frequency of some extreme events (*high confidence*). In many lower-latitude regions, yields of some crops (e.g., maize and wheat) have declined, while in many higher-latitude regions, yields of some crops (e.g., maize, wheat and sugar beets) have increased over recent decades *(high confidence)*. Climate change has resulted in lower animal growth rates and productivity in pastoral systems in Africa (*high confidence)*. There is robust evidence that agricultural pests and diseases have already responded to climate change resulting in both increases and decreases of infestations (*high confidence*). Based on indigenous and local knowledge, climate change is affecting food security in drylands, particularly those in Africa, and high mountain regions of Asia and South America[20]. {5.2.1, 5.2.2, 7.2.2}

A 3. Agriculture, Forestry and Other Land Use (AFOLU) activities accounted for around 13% of CO_2, 44% of methane (CH_4), and 82% of nitrous oxide (N_2O) emissions from human activities globally during 2007-2016, representing 23% (12.0 +/- 3.0 $GtCO_2e\ yr^{-1}$) of total net anthropogenic emissions of GHGs[21] (*medium confidence*). The natural response of land to human-induced environmental change caused a net sink of around 11.2 $GtCO_2\ yr^{-1}$ during 2007-2016 (equivalent to 29% of total CO_2 emissions) (*medium confidence*); the persistence of the sink is uncertain due to climate change (*high confidence*). If emissions associated with pre- and post-production activities in the global food system[22] are included, the emissions are estimated to be 21-37% of total net anthropogenic GHG emissions (*medium confidence*). {2.3, Table 2.2, 5.4}.

A3.1. Land is simultaneously a source and a sink of CO_2 due to both anthropogenic and natural drivers, making it hard to separate anthropogenic from natural fluxes (*very high confidence*). Global models estimate net CO_2 emissions of $5.2 \pm 2.6\ GtCO_2\ yr^{-1}$ (*likely* range) from land use and land-use change during 2007-16. These net emissions are mostly due to deforestation, partly offset by afforestation/reforestation, and emissions and removals by other land use activities

[20] The assessment covered literature whose methodologies included interviews and surveys with indigenous peoples and local communities.
[21] This assessment only includes CO2, CH4 and N2O.
[22] Global food system in this report is defined as 'all the elements (environment, people, inputs, processes, infrastructures, institutions, etc.) and activities that relate to the production, processing, distribution, preparation and consumption of food, and the output of these activities, including socioeconomic and environmental outcomes at the global level'. These emissions data are not directly comparable to the national inventories prepared according to the 2006 IPCC Guidelines for National Greenhouse Gas.

(*very high confidence*) (Table SPM.1)[23]. There is no clear trend in annual emissions since 1990 (*medium confidence*) (Figure SPM.1). {1.1, 2.3, Table 2.2, Table 2.3}

A3.2. The natural response of land to human-induced environmental changes such as increasing atmospheric CO_2 concentration, nitrogen deposition, and climate change, resulted in global net removals of 11.2 +/– 2.6 Gt CO_2 yr^{-1} (*likely* range) during 2007-2016 (Table SPM.1). The sum of the net removals due to this response and the AFOLU net emissions gives a total net land-atmosphere flux that removed 6.0+/-2.6 $GtCO_2$ yr^{-1} during 2007-2016 (*likely* range). Future net increases in CO_2 emissions from vegetation and soils due to climate change are projected to counteract increased removals due to CO_2 fertilisation and longer growing seasons (*high confidence*). The balance between these processes is a key source of uncertainty for determining the future of the land carbon sink. Projected thawing of permafrost is expected to increase the loss of soil carbon (*high confidence*). During the 21st century, vegetation growth in those areas may compensate in part for this loss (*low confidence*). {Box 2.3, 2.3.1, 2.5.3, 2.7; Table 2.3}

A3.3. Global models and national GHG inventories use different methods to estimate anthropogenic CO_2 emissions and removals for the land sector. Both produce estimates that are in close agreement for land-use change involving forest (e.g., deforestation, afforestation), and differ for managed forest. Global models consider as managed forest those lands that were subject to harvest whereas, consistent with IPCC guidelines, national GHG inventories define managed forest more broadly. On this larger area, inventories can also consider the natural response of land to human-induced environmental changes as anthropogenic, while the global model approach {Table SPM.1} treats this response as part of the non-anthropogenic sink. For illustration, from 2005 to 2014, the sum of the national GHG inventories net emission estimates is 0.1±1.0 $GtCO_2yr^{-1}$, while the mean of two global bookkeeping models is 5.1±2.6 $GtCO_2yr^{-1}$ (*likely* range). Consideration of differences in methods can enhance understanding of land sector net emission estimates and their applications.

[23] The net anthropogenic flux of CO2 from "bookkeeping" or "carbon accounting" models is composed of two opposing gross fluxes: gross emissions (about 20 GtCO2 yr-1) are from deforestation, cultivation of soils, and oxidation of wood products; gross removals (about 14 GtCO2 yr-1) are largely from forest growth following wood harvest and agricultural abandonment (medium confidence).

Table SPM1. Net anthropogenic emissions due to Agriculture, Forestry, and other Land Use (AFOLU) and non-AFOLU (Panel 1) and global food systems (average for 2007-2016)[1] (Panel 2). Positive value represents emissions; negative value represents removals.

		Direct Anthropogenic							
		Net anthropogenic emissions due to Agriculture, Forestry, and Other Land Use (AFOLU)			Non-AFOLU anthropogenic GHG emissions[6]	Total net anthropogenic emissions (AFOLU + non-AFOLU) by gas	AFOLU as a % of total net anthropogenic emissions, by gas	Natural response of land to human-induced environmental change[7]	Net land – atmosphere flux from all lands
Panel 1: Contribution of AFOLU									
		FOLU	Agriculture	Total					
		A	B	C = B + A	D	**E = C + D**	F = (C/E)*100	G	A + G
CO_2[2]	Gt CO_2 y^{-1}	5.2 ± 2.6	--[11]	5.2 ± 2.6	33.9 ± 1.8	**39.1 ± 3.2**	~13%	-11.2 ± 2.6	-6.0 ± 2.0
CH_4[3,8]	Mt CH_4 y^{-1}	19 ± 6	142 ± 43	162 ± 48.6	201 ± 100	**363 ± 111**			
	Gt CO_2e y^{-1}	0.5 ± 0.2	4.0 ± 1.2	4.5 ± 1.4	5.6 ± 2.8	**10.1 ± 3.1**	~44%		
N_2O[3,8]	Mt N_2O y^{-1}	0.3 ± 0.1	8 ±2	8.3 ± 2.5	2.0 ± 1.0	**10.4 ± 2.7**			
	Gt CO_2e y^{-1}	0.09 ± 0.03	2.2 ± 0.7	2.3 ± 0.7	0.5 ± 0.3	**2.8 ± 0.7**	~82%		
Total (GHG)	**Gt CO_2e y^{-1}**	**5.8 ± 2.6**	**6.2 ± 1.4**	**12.0 ± 3.0**	**40.0 ± 3.4**	**52.0 ± 4.5**	**~23%**		
Panel 2: Contribution of global food system									
		Land-use change	Agriculture		Non-AFOLU[5] other sectors pre- to post-production	Total global food system emissions			
CO_2[4] Land-use change	Gt CO_2 y^{-1}	4.9 ± 2.5							
CH_4[3,8,9] Agriculture	Gt CO_2e y^{-1}		4.0 ± 1.2						
N_2O[3,8,9] Agriculture	Gt CO_2e y^{-1}		2.2 ± 0.7						
CO_2 other sectors	Gt CO_2 y^{-1}				2.4 – 4.8				
Total (CO2e)[10]	**Gt CO_2e y^{-1}**	**4.9 ± 2.5**	**6.2 ± 1.4**		**2.4 – 4.8**	**10.7 – 19.1**			

Data sources and notes:

1 Estimates are only given until 2016 as this is the latest date when data are available for all gases.

2 Net anthropogenic flux of CO_2 due to land cover change such as deforestation and afforestation, and land management including wood harvest and regrowth, as well as peatland burning, based on two bookkeeping models as used in the Global Carbon Budget and for AR5. Agricultural soil carbon stock change under the same land use is not considered in these models. {2.3.1.2.1, Table 2.2, Box 2.2}

3 Estimates show the mean and assessed uncertainty of two databases, FAOSTAT and USEPA 2012 {2.3; Table 2.2}

4 Based on FAOSTAT. Categories included in this value are "net forest conversion" (net deforestation), drainage of organic soils (cropland and grassland), biomass burning (humid tropical forests, other forests, organic soils). It excludes "forest land" (forest management plus net forest expansion), which is primarily a sink due to afforestation. Note: total FOLU emissions from FAOSTAT are 2.8 (±1.4) Gt CO_2 yr^{-1} for the period 2007-2016. {Table 2.2, Table 5.4}

5 CO_2 emissions induced by activities not included in the AFOLU sector, mainly from energy (e.g. grain drying), transport (e.g. international trade), and industry (e.g. synthesis of inorganic fertilizers) part of food systems, including agricultural production activities (e.g. heating in greenhouses), pre-production (e.g. manufacturing of farm inputs) and post-production (e.g. agri-food processing) activities. This estimate is land based and hence excludes emissions from fisheries. It includes emissions from fibre and other non-food agricultural products since these are not separated from food use in data bases. The CO_2 emissions related to food system in other sectors than AFOLU are 6-13% of total anthropogenic CO_2 emissions. These emissions are typically low in smallholder subsistence farming. When added to AFOLU emissions, the estimated share of food systems in global anthropogenic emissions is 21-37%. {5.4.5, Table 5.4}

6 Total non-AFOLU emissions were calculated as the sum of total CO_2e emissions values for energy, industrial sources, waste and other emissions with data from the Global Carbon Project for CO_2, including international aviation and shipping and from the PRIMAP database for CH_4 and N_2O averaged over 2007-2014 only as that was the period for which data were available {2.3; Table 2.2}.

7 The natural response of land to human-induced environmental changes is the response of vegetation and soils to environmental changes such as increasing atmospheric CO_2 concentration, nitrogen deposition, and climate change. The estimate shown represents the average from Dynamic Global Vegetation Models {2.3.1.2.4, Box 2.2, Table 2.3}

8 All values expressed in units of CO_2e are based on AR5 100 year Global Warming Potential (GWP) values without climate-carbon feedbacks (N_2O = 265; CH_4 = 28). Note that the GWP has been used across fossil fuel and biogenic sources of methane. If a higher GWP for fossil fuel CH4 (30 per AR5), then total anthropogenic CH4 emissions expressed in CO2e would be 2% greater.

9 This estimate is land based and hence excludes emissions from fisheries and emissions from aquaculture (except emissions from feed produced on land and used in aquaculture), and also includes non-food use (e.g. fibre and bioenergy) since these are not separated from food use in databases. It excludes non-CO_2 emissions associated with land use change (FOLU category) since these are from fires in forests and peatlands.

10 Emissions associated with food loss and waste are included implicitly, since emissions from food system are related to food produced, including food consumed for nutrition and to food loss and waste. The latter is estimated at 8-10% of total anthropogenic emissions in CO2e. {5.5.2.5}

11 No global data are available for agricultural CO_2 emissions

A3.4. Global AFOLU emissions of methane in the period 2007-2016 were 162 ± 49 Mt CH_4 yr-1 (4.5 ± 1.4 $GtCO_2eq$ yr-1) (*medium confidence*). The globally averaged atmospheric concentration of methane shows a steady increase between the mid-1980s and early 1990s, slower growth thereafter until 1999, a period of no growth between 1999-2006, followed by a resumption of growth in 2007 (*high confidence*). Biogenic sources make up a larger proportion of emissions than they did before 2000 (*high confidence*). Ruminants and the expansion of rice cultivation are important contributors to the rising concentration (*high confidence*). {Table 2.2, 2.3.2, 5.4.2, 5.4.3, Figure SPM.1}.

A3.5. Anthropogenic AFOLU N_2O emissions are rising, and were 8.3 ± 2.5 MtN_2O yr^{-1} (2.3 ± 0.7 GtCO_2eq yr^{-1}) during the period 2007-2016. Anthropogenic N_2O emissions (Figure SPM.1, Table SPM.1) from soils are primarily due to nitrogen application including inefficiencies (over-application or poorly synchronised with crop demand timings) (*high confidence*). Cropland soils emitted around 3 Mt N_2O yr^{-1} (around 795 MtCO_2-eq yr^{-1}) during the period 2007-2016 (*medium confidence*). There has been a major growth in emissions from managed pastures due to increased manure deposition (*medium confidence*). Livestock on managed pastures and rangelands accounted for more than one half of total anthropogenic N_2O emissions from agriculture in 2014 (*medium confidence*). {Table 2.1, 2.3.3, 5.4.2, 5.4.3}

A3.6. Total net GHG emissions from agriculture, forestry, and other land use (AFOLU) emissions represent 12.0 +/- 3.0 GtCO_2eq yr^{-1} during 2007-2016. This represents 23% of total net anthropogenic emissions[24] (Table SPM.1). Other approaches, such as global food system, include agricultural emissions and land use change (i.e., deforestation and peatland degradation), as well as outside farm gate emissions from energy, transport and industry sectors for food production. Emissions within farm gate and from agricultural land expansion contributing to the global food system represent 16-27% of total anthropogenic emissions (*medium confidence*). Emissions outside the farm gate represent 5-10% of total anthropogenic emissions (*medium confidence*). Given the diversity of food systems, there are large regional differences in the contributions from different components of the food system (*very high confidence*). Emissions from agricultural production are projected to increase (*high confidence*), driven by population and income growth and changes in consumption patterns (*medium confidence*). {5.5, Table 5.4}

A 4. Changes in land conditions[25], either from land-use or climate change, affect global and regional climate (*high confidence*). At the regional scale, changing land conditions can reduce or accentuate warming and affect the intensity, frequency and duration of extreme events. The magnitude and direction of these changes vary with location and season (*high confidence*). {Executive Summary Chapter 2, 2.3, 2.4, 2.5, 3.3}

A4.1. Since the pre-industrial period, changes in land cover due to human activities have led to both a net release of CO_2 contributing to global warming (*high confidence*), and an increase in global land albedo[26] causing surface cooling (*medium confidence*). Over the historical period, the resulting net effect on globally averaged surface temperature is estimated to be small (*medium confidence*). {2.4, 2.6.1, 2.6.2}

[24] This assessment only includes CO2, CH4 and N2O.
[25] Land conditions encompass changes in land cover (e.g. deforestation, afforestation, urbanisation), in land use (e.g. irrigation), and in land state (e.g. degree of wetness, degree of greening, amount of snow, amount of permafrost)
[26] Land with high albedo reflects more incoming solar radiation than land with low albedo.

A4.2. The likelihood, intensity and duration of many extreme events can be significantly modified by changes in land conditions, including heat related events such as heat waves (*high confidence*) and heavy precipitation events (*medium confidence*). Changes in land conditions can affect temperature and rainfall in regions as far as hundreds of kilometres away (*high confidence*). {2.5.1, 2.5.2, 2.5.4, 3.3; Cross-Chapter Box 4 in Chapter 2}

A4.3. Climate change is projected to alter land conditions with feedbacks on regional climate. In those boreal regions where the treeline migrates northward and/or the growing season lengthens, winter warming will be enhanced due to decreased snow cover and albedo while warming will be reduced during the growing season because of increased evapotranspiration (*high confidence*). In those tropical areas where increased rainfall is projected, increased vegetation growth will reduce regional warming (*medium confidence*). Drier soil conditions resulting from climate change can increase the severity of heat waves, while wetter soil conditions have the opposite effect (*high confidence*). {2.5.2, 2.5.3}

A4.4. Desertification amplifies global warming through the release of CO_2 linked with the decrease in vegetation cover (*high confidence*). This decrease in vegetation cover tends to increase local albedo, leading to surface cooling (*high confidence*). {3.3}

A4.5. Changes in forest cover for example from afforestation, reforestation and deforestation, directly affect regional surface temperature through exchanges of water and energy[27] (*high confidence*). Where forest cover increases in tropical regions cooling results from enhanced evapotranspiration (*high confidence*). Increased evapotranspiration can result in cooler days during the growing season (*high confidence*) and can reduce the amplitude of heat related events (*medium confidence*). In regions with seasonal snow cover, such as boreal and some temperate, increased tree and shrub cover also has a wintertime warming influence due to reduced surface albedo[28] (*high confidence*). {2.3, 2.4.3, 2.5.1, 2.5.2, 2.5.4}

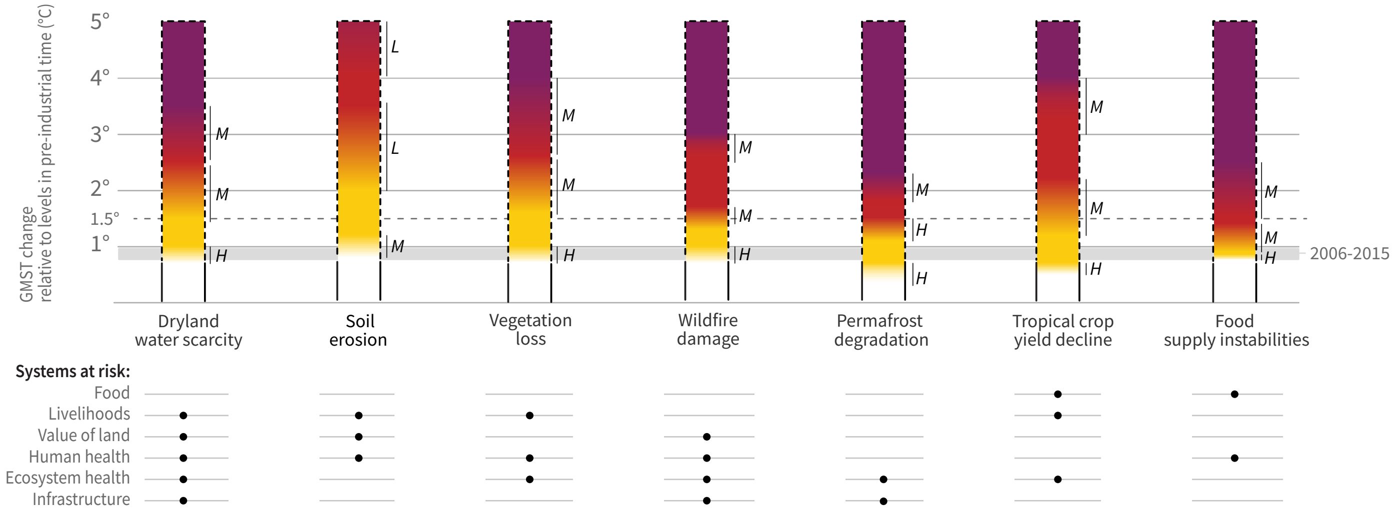
A4.6. Both global warming and urbanisation can enhance warming in cities and their surroundings (heat island effect), especially during heat related events, including heat waves (*high confidence*). Night-time temperatures are more affected by this effect than daytime temperatures (*high confidence*). Increased urbanisation can also intensify extreme rainfall events over the city or downwind of urban areas (*medium confidence*). {2.5.1, 2.5.2, 2.5.3, 4.9.1, Cross-Chapter Box 4 in Chapter 2}

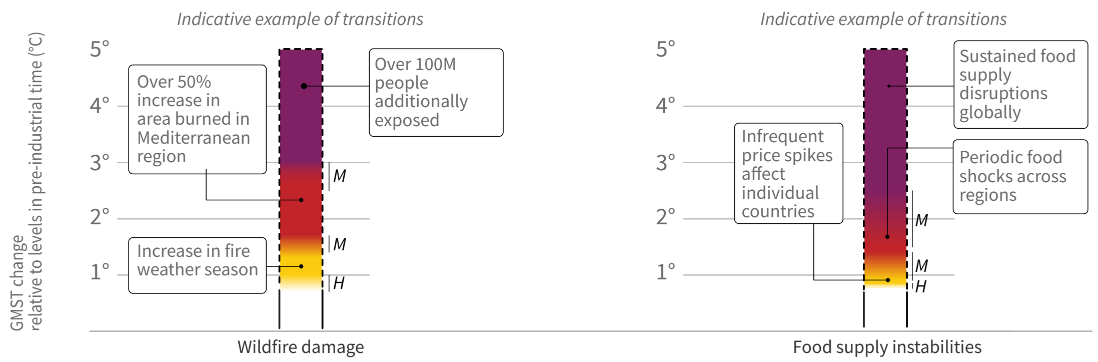
[27] The literature indicates that forest cover changes can also affect climate through changes in emissions of reactive gases and aerosols {2.4, 2.5}.

[28] Emerging literature shows that boreal forest-related aerosols may counteract at least partly the warming effect of surface albedo {2.4.3}.

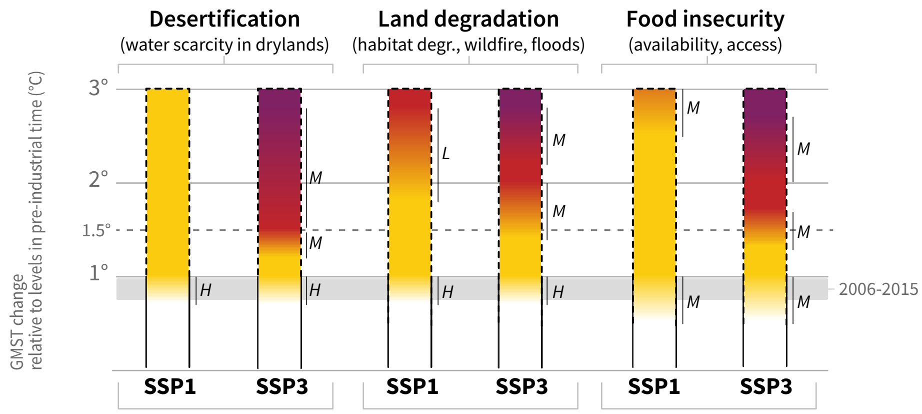
A. Risks to humans and ecosystems from changes in land-based processes as a result of climate change

Increases in global mean surface temperature (GMST), relative to pre-industrial levels, affect processes involved in **desertification** (water scarcity), **land degradation** (soil erosion, vegetation loss, wildfire, permafrost thaw) and **food security** (crop yield and food supply instabilities). Changes in these processes drive risks to food systems, livelihoods, infrastructure, the value of land, and human and ecosystem health. Changes in one process (e.g. wildfire or water scarcity) may result in compound risks. Risks are location-specific and differ by region.





B. Different socioeconomic pathways affect levels of climate related risks



Socio-economic choices can reduce or exacerbate climate related risks as well as influence the rate of temperature increase. The **SSP1** pathway illustrates a world with low population growth, high income and reduced inequalities, food produced in low GHG emission systems, effective land use regulation and high adaptive capacity. The **SSP3** pathway has the opposite trends. Risks are lower in SSP1 compared with SSP3 given the same level of GMST increase.

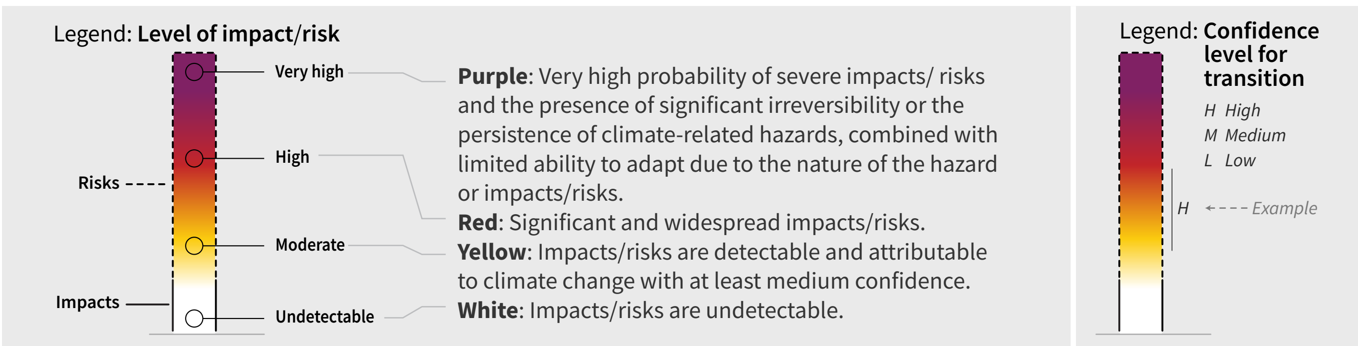


Figure SPM. 2 Risks to land-related human systems and ecosystems from global climate change, socio-economic development and mitigation choices in terrestrial ecosystems.

As in previous IPCC reports the literature was used to make expert judgements to assess the levels of global warming at which levels of risk are undetectable, moderate, high or very high, as described further in Chapter 7 and other parts of the underlying report. The figure indicates assessed risks at approximate warming levels which may be influenced by a variety of factors, including adaptation responses. The assessment considers adaptive capacity consistent with the SSP pathways as described below. **Panel A:** Risks to selected elements of the land system as a function of global mean surface temperature {2.1; Box 2.1; 3.5; 3.7.1.1; 4.4.1.1; 4.4.1.2; 4.4.1.3; 5.2.2; 5.2.3; 5.2.4; 5.2.5; 7.2;7.3, Table SM7.1}. Links to broader systems are illustrative and not intended to be comprehensive. Risk levels are estimated assuming medium exposure and vulnerability driven by moderate trends in socioeconomic conditions broadly consistent with an SSP2 pathway. {Table SM7.4}. **Panel B:** Risks associated with desertification, land degradation and food security due to climate change and patterns of socio-economic development. Increasing risks associated with desertification include population exposed and vulnerable to water scarcity in drylands. Risks related to land degradation include increased habitat degradation, population exposed to wildfire and floods and costs of floods. Risks to food security include availability and access to food, including population at risk of hunger, food price increases and increases in disability adjusted life years attributable due to childhood underweight. Risks are assessed for two contrasted socio-economic pathways (SSP1 and SSP3 {SPM Box 1}) excluding the effects of targeted mitigation policies {3.5; 4.2.1.2; 5.2.2; 5.2.3; 5.2.4; 5.2.5; 6.1.4; 7.2, Table SM7.5}. Risks are not indicated beyond 3°C because SSP1 does not exceed this level of temperature change. **All panels**: As part of the assessment, literature was compiled and data extracted into a summary table. A formal expert elicitation protocol (based on modified-Delphi technique and the Sheffield Elicitation Framework), was followed to identify risk transition thresholds. This included a multi-round elicitation process with two rounds of independent anonymous threshold judgement, and a final consensus discussion. Further information on methods and underlying literature can be found in Chapter 7 Supplementary Material.

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BOX SPM.1: Shared Socioeconomic Pathways (SSPs)

In this report the implications of future socio-economic development on climate change mitigation, adaptation and land-use are explored using shared socio-economic pathways (SSPs). The SSPs span a range of challenges to climate change mitigation and adaptation.

- SSP1 includes a peak and decline in population (~7 billion in 2100), high income and reduced inequalities, effective land-use regulation, less resource intensive consumption, including food produced in low-GHG emission systems and lower food waste, free trade and environmentally-friendly technologies and lifestyles. Relative to other pathways, SSP1 has low challenges to mitigation and low challenges to adaptation (i.e., high adaptive capacity).
- SSP2 includes medium population growth (~9 billion in 2100), medium income; technological progress, production and consumption patterns are a continuation of past trends, and only gradual reduction in inequality occurs. Relative to other pathways, SSP2 has medium challenges to mitigation and medium challenges to adaptation (i.e., medium adaptive capacity).

- SSP3 includes high population (~13 billion in 2100), low income and continued inequalities, material-intensive consumption and production, barriers to trade, and slow rates of technological change. Relative to other pathways, SSP3 has high challenges to mitigation and high challenges to adaptation (i.e., low adaptive capacity).
- SSP4 includes medium population growth (~9 billion in 2100), medium income, but significant inequality within and across regions. Relative to other pathways, SSP4 has low challenges to mitigation, but high challenges to adaptation (i.e., low adaptive capacity).
- SSP5 includes a peak and decline in population (~7 billion in 2100), high income, reduced inequalities, and free trade. This pathway includes resource-intensive production, consumption and lifestyles. Relative to other pathways, SSP5 has high challenges to mitigation, but low challenges to adaptation (i.e., high adaptive capacity).

The SSPs can be combined with Representative Concentration Pathways (RCPs) which imply different levels of mitigation, with implications for adaptation. Therefore, SSPs can be consistent with different levels of global mean surface temperature rise as projected by different SSP-RCP combinations. However, some SSP-RCP combinations are not possible; for instance RCP2.6 and lower levels of future global mean surface temperature rise (e.g., 1.5ºC) are not possible in SSP3 in modelled pathways. {1.2.2, Cross-Chapter Box 1 in Chapter 1, 6.1.4, Cross-Chapter Box 9 in Chapter 6}

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A 5. Climate change creates additional stresses on land, exacerbating existing risks to livelihoods, biodiversity, human and ecosystem health, infrastructure, and food systems (*high confidence*). Increasing impacts on land are projected under all future GHG emission scenarios (*high confidence*). Some regions will face higher risks, while some regions will face risks previously not anticipated (*high confidence*). Cascading risks with impacts on multiple systems and sectors also vary across regions (*high confidence*). {2.2, 3.5, 4.2, 4.4, 4.7, 5.1, 5.2, 5.8, 6.1, 7.2, 7.3, Cross-Chapter Box 9 in Chapter 6, Figure SPM.2}

A5.1. With increasing warming, the frequency, intensity and duration of heat related events including heat waves are projected to continue to increase through the 21st century (*high confidence*). The frequency and intensity of droughts are projected to increase particularly in the Mediterranean region and southern Africa (*medium confidence*). The frequency and intensity of extreme rainfall events are projected to increase in many regions (*high confidence*). {2.2.5, 3.5.1, 4.2.3, 5.2}

A5.2. With increasing warming, climate zones are projected to further shift poleward in the middle and high latitudes (*high confidence*). In high-latitude regions, warming is projected to increase disturbance in boreal forests, including drought, wildfire, and pest outbreaks (*high confidence*). In tropical regions, under medium and high GHG emissions scenarios, warming is projected to result in the emergence of unprecedented[29] climatic conditions by the mid to late 21st century (*medium confidence*). {2.2.4, 2.2.5, 2.5.3, 4.3.2}

A5.3. Current levels of global warming are associated with moderate risks from increased dryland water scarcity, soil erosion, vegetation loss, wildfire damage, permafrost thawing, coastal degradation and tropical crop yield decline (*high confidence*). Risks, including cascading risks, are projected to become increasingly severe with increasing temperatures. At around 1.5°C of global warming the risks from dryland water scarcity, wildfire damage, permafrost degradation and food supply instabilities are projected to be high (*medium confidence*). At around 2°C of global warming the risk from permafrost degradation and food supply instabilities are projected to be very high (*medium confidence*). Additionally, at around 3°C of global warming risk from vegetation loss, wildfire damage, and dryland water scarcity are also projected to be very high (*medium confidence*). Risks from droughts, water stress, heat related events such as heatwaves and habitat degradation simultaneously increase between 1.5°C and 3°C warming (*low confidence*). {Figure SPM.2, 7.2.2, Cross-Chapter Box 9 in Chapter 6, Chapter 7 supplementary material}

A5.4. The stability of food supply[30] is projected to decrease as the magnitude and frequency of extreme weather events that disrupt food chains increases (*high confidence*). Increased atmospheric CO_2 levels can also lower the nutritional quality of crops (*high confidence*). In SSP2, global crop and economic models project a median increase of 7.6% (range of 1 to 23%) in cereal prices in 2050 due to climate change (RCP6.0), leading to higher food prices and increased risk of food insecurity and hunger (*medium confidence*). The most vulnerable people will be more severely affected (*high confidence*). {5.2.3, 5.2.4, 5.2.5, 5.8.1, 7.2.2.2, 7.3.1}

A5.5. In drylands, climate change and desertification are projected to cause reductions in crop and livestock productivity (*high confidence*), modify the plant species mix and reduce biodiversity (*medium confidence*). Under SSP2, the dryland population vulnerable to water stress, drought intensity and habitat degradation is projected to reach 178 million people by 2050 at 1.5°C warming, increasing to 220 million people at 2°C warming, and 277 million people at 3°C warming (*low confidence*). {3.5.1, 3.5.2, 3.7.3}

[29] Unprecedented climatic conditions are defined in this report as not having occurred anywhere during the 20th century. They are characterized by high temperature with strong seasonality and shifts in precipitation. In the literature assessed, the effect of climatic variables other than temperature and precipitation were not considered.
[30] The supply of food is defined in this report as encompassing availability and access (including price). Food supply instability refers to variability that influences food security through reducing access.

A5.6. Asia and Africa[31] are projected to have the highest number of people vulnerable to increased desertification. North America, South America, Mediterranean, southern Africa and central Asia may be increasingly affected by wildfire. The tropics and subtropics are projected to be most vulnerable to crop yield decline. Land degradation resulting from the combination of sea level rise and more intense cyclones is projected to jeopardise lives and livelihoods in cyclone prone areas (*very high confidence*). Within populations, women, the very young, elderly and poor are most at risk (*high confidence*). {3.5.1, 3.5.2, 4.4, Table 4.1, 5.2.2, 7.2.2, Cross-Chapter Box 3 in Chapter 2}

A5.7. Changes in climate can amplify environmentally induced migration both within countries and across borders (*medium confidence*), reflecting multiple drivers of mobility and available adaptation measures (*high confidence*). Extreme weather and climate or slow-onset events may lead to increased displacement, disrupted food chains, threatened livelihoods (*high confidence*), and contribute to exacerbated stresses for conflict (*medium confidence*). {3.4.2, 4.7.3, 5.2.3, 5.2.4, 5.2.5, 5.8.2, 7.2.2, 7.3.1}

A5.8 Unsustainable land management has led to negative economic impacts (*high confidence*). Climate change is projected to exacerbate these negative economic impacts (*high confidence*). {4.3.1, 4.4.1, 4.7, 4.8.5, 4.8.6, 4.9.6, 4.9.7, 4.9.8, 5.2, 5.8.1, 7.3.4, 7.6.1, Cross-Chapter Box 10 in Chapter 7}

A6. The level of risk posed by climate change depends both on the level of warming and on how population, consumption, production, technological development, and land management patterns evolve (*high confidence*). Pathways with higher demand for food, feed, and water, more resource-intensive consumption and production, and more limited technological improvements in agriculture yields result in higher risks from water scarcity in drylands, land degradation, and food insecurity (*high confidence*). {5.1.4, 5.2.3, 6.1.4, 7.2, Cross-Chapter Box 9 in Chapter 6, Figure SPM.2b}

A6.1. Projected increases in population and income, combined with changes in consumption patterns, result in increased demand for food, feed, and water in 2050 in all SSPs (*high confidence*). These changes, combined with land management practices, have implications for land-use change, food insecurity, water scarcity, terrestrial GHG emissions, carbon sequestration potential, and biodiversity (*high confidence*). Development pathways in which incomes increase and the demand for land conversion is reduced, either through reduced

[31] West Africa has a high number of people vulnerable to increased desertification and yield decline. North Africa is vulnerable to water scarcity.

agricultural demand or improved productivity, can lead to reductions in food insecurity (*high confidence*). All assessed future socio-economic pathways result in increases in water demand and water scarcity (*high confidence*). SSPs with greater cropland expansion result in larger declines in biodiversity (*high confidence*). {6.1.4}

A6.2. Risks related to water scarcity in drylands are lower in pathways with low population growth, less increase in water demand, and high adaptive capacity, as in Shared Socio-economic Pathway 1 (SSP1) (See BOX SPM.1). In these scenarios the risk from water scarcity in drylands is moderate even at global warming of 3°C (*low confidence*). By contrast, risks related to water scarcity in drylands are greater for pathways with high population growth, high vulnerability, higher water demand, and low adaptive capacity, such as SSP3. In SSP3 the transition from moderate to high risk occurs between 1.2°C and 1.5°C (*medium confidence*). {7.2, Figure SPM.2b, BOX SPM.1}

A6.3. Risks related to climate change driven land degradation are higher in pathways with a higher population, increased land-use change, low adaptive capacity and other barriers to adaptation (e.g., SSP3). These scenarios result in more people exposed to ecosystem degradation, fire, and coastal flooding (*medium confidence*). For land degradation, the projected transition from moderate to high risk occurs for global warming between 1.8°C and 2.8°C in SSP1 (*low confidence*) and between 1.4°C and 2°C in SSP3 (*medium confidence*). The projected transition from high to very high risk occurs between 2.2°C and 2.8°C for SSP3 (*medium confidence*). {4.4, 7.2, Figure SPM.2b}

A6.4. Risks related to food security are greater in pathways with lower income, increased food demand, increased food prices resulting from competition for land, more limited trade, and other challenges to adaptation (e.g., SSP3) (*high confidence*). For food security, the transition from moderate to high risk occurs for global warming between 2.5°C and 3.5°C in SSP1 (*medium confidence*) and between 1.3°C and 1.7°C in SSP3 (*medium confidence*). The transition from high to very high risk occurs between 2°C and 2.7°C for SSP3 (*medium confidence*). {7.2, Figure SPM.2b}

A6.5 Urban expansion is projected to lead to conversion of cropland leading to losses in food production (*high confidence*). This can result in additional risks to the food system. Strategies for reducing these impacts can include urban and peri-urban food production and management of urban expansion, as well as urban green infrastructure that can reduce climate risks in cities[32] (*high confidence*). {4.9.1, 5.5, 5.6, 6.3, 6.4, 7.5.6} (Figure SPM3)

[32] The land systems considered in this report do not include urban ecosystem dynamics in detail. Urban areas, urban expansion, and other urban processes and their relation to land-related processes are extensive, dynamic, and complex.

B. Adaptation and mitigation response options

B 1. Many land-related responses that contribute to climate change adaptation and mitigation can also combat desertification and land degradation and enhance food security. The potential for land-related responses and the relative emphasis on adaptation and mitigation is context specific, including the adaptive capacities of communities and regions. While land-related response options can make important contributions to adaptation and mitigation, there are some barriers to adaptation and limits to their contribution to global mitigation. (*very high confidence*) {2.6, 4.8, 5.6, 6.1, 6.3, 6.4, Figure SPM.3}

B1.1. Some land-related actions are already being taken that contribute to climate change adaptation, mitigation and sustainable development. The response options were assessed across adaptation, mitigation, combating desertification and land degradation, food security and sustainable development, and a select set of options deliver across all of these challenges. These options include, but are not limited to, sustainable food production, improved and sustainable forest management, soil organic carbon management, ecosystem conservation and land restoration, reduced deforestation and degradation, and reduced food loss and waste (*high confidence*). These response options require integration of biophysical, socioeconomic and other enabling factors. {6.3, 6.4.5; Cross-Chapter Box 10 in Chapter 7}

B1.2. While some response options have immediate impact, others take decades to deliver measurable results. Examples of response options with immediate impacts include the conservation of high-carbon ecosystems such as peatlands, wetlands, rangelands, mangroves and forests. Examples that provide multiple ecosystem services and functions, but take more time to deliver, include afforestation and reforestation as well as the restoration of high-carbon ecosystems, agroforestry, and the reclamation of degraded soils (*high confidence*). {6.4.5; Cross-Chapter Box 10 in Chapter 7}

B1.3. The successful implementation of response options depends on consideration of local environmental and socio-economic conditions. Some options such as soil carbon management are potentially applicable across a broad range of land use types, whereas the efficacy of land management practices relating to organic soils, peatlands and wetlands, and those linked to freshwater resources, depends on specific agro-ecological conditions (*high confidence*). Given

Several issues addressed in this report such as population, growth, incomes, food production and consumption, food security, and diets have close relationships with these urban processes. Urban areas are also the setting of many processes related to land-use change dynamics, including loss of ecosystem functions and services, that can lead to increased disaster risk. Some specific urban issues are assessed in this report.

the site-specific nature of climate change impacts on food system components and wide variations in agroecosystems, adaptation and mitigation options and their barriers are linked to environmental and cultural context at regional and local levels *(high confidence).* Achieving land degradation neutrality depends on the integration of multiple responses across local, regional and national scales, multiple sectors including agriculture, pasture, forest and water (*high confidence*). {4.8, 6.2, 6.3, 6.4.4}

B1.4. Land based options that deliver carbon sequestration in soil or vegetation, such as afforestation, reforestation, agroforestry, soil carbon management on mineral soils, or carbon storage in harvested wood products do not continue to sequester carbon indefinitely (*high confidence*). Peatlands, however, can continue to sequester carbon for centuries (*high confidence*). When vegetation matures or when vegetation and soil carbon reservoirs reach saturation, the annual removal of CO_2 from the atmosphere declines towards zero, while carbon stocks can be maintained (*high confidence*). However, accumulated carbon in vegetation and soils is at risk from future loss (or sink reversal) triggered by disturbances such as flood, drought, fire, or pest outbreaks, or future poor management (*high confidence*). {6.4.1}

B 2. Most of the response options assessed contribute positively to sustainable development and other societal goals (*high confidence*). Many response options can be applied without competing for land and have the potential to provide multiple co-benefits (*high confidence*). A further set of response options has the potential to reduce demand for land, thereby enhancing the potential for other response options to deliver across each of climate change adaptation and mitigation, combating desertification and land degradation, and enhancing food security (*high confidence*). {4.8, 6.2, 6.3.6, 6.4.3; Figure SPM.3}

B2.1. A number of land management options, such as improved management of cropland and grazing lands, improved and sustainable forest management, and increased soil organic carbon content, do not require land use change and do not create demand for more land conversion (*high confidence*). Further, a number of response options such as increased food productivity, dietary choices and food losses and waste reduction, can reduce demand for land conversion, thereby potentially freeing land and creating opportunities for enhanced implementation of other response options (*high confidence*). Response options that reduce competition for land are possible and are applicable at different scales, from farm to regional (*high confidence*). {4.8, 6.3.6, 6.4; Figure SPM.3}

B2.2. A wide range of adaptation and mitigation responses, e.g. preserving and restoring natural ecosystems such as peatland, coastal lands and forests, biodiversity conservation, reducing competition for land, fire management, soil management, and most risk management options (e.g. use of local seeds, disaster risk management, risk sharing instruments) have the potential to make positive contributions to sustainable development, enhancement of ecosystem functions and services and other societal goals (*medium confidence*). Ecosystem-based adaptation can, in some contexts, promote nature conservation while alleviating poverty and even provide co-benefits by removing greenhouse gases and protecting livelihoods (e.g. mangroves) (*medium confidence*). {6.4.3, 7.4.6.2}

B2.3. Most of the land management-based response options that do not increase competition for land, and almost all options based on value chain management (e.g. dietary choices, reduced post-harvest losses, reduced food waste) and risk management, can contribute to eradicating poverty and eliminating hunger while promoting good health and wellbeing, clean water and sanitation, climate action, and life on land (*medium confidence*). {6.4.3}

B 3. Although most response options can be applied without competing for available land, some can increase demand for land conversion (*high confidence*). At the deployment scale of several $GtCO_2yr^{-1}$, this increased demand for land conversion could lead to adverse side effects for adaptation, desertification, land degradation and food security (*high confidence*). If applied on a limited share of total land and integrated into sustainably managed landscapes, there will be fewer adverse side-effects and some positive co-benefits can be realised (*high confidence*). {4.5, 6.2, 6.4; Cross-Chapter Box 7 in Chapter 6; Figure SPM.3}

B3.1. If applied at scales necessary to remove CO_2 from the atmosphere at the level of several $GtCO_2yr^{-1}$, afforestation, reforestation and the use of land to provide feedstock for bioenergy with or without carbon capture and storage, or for biochar, could greatly increase demand for land conversion (*high confidence*). Integration into sustainably managed landscapes at appropriate scale can ameliorate adverse impacts (*medium confidence*). Reduced grassland conversion to croplands, restoration and reduced conversion of peatlands, and restoration and reduced conversion of coastal wetlands affect smaller land areas globally, and the impacts on land use change of these options are smaller or more variable (*high confidence*). {Cross-Chapter Box 7 in Chapter 6; 6.4; Figure SPM.3}

B3.2. While land can make a valuable contribution to climate change mitigation, there are limits to the deployment of land-based mitigation measures such as bioenergy crops or afforestation. Widespread use at the scale of several millions of km^2 globally could increase risks for desertification, land degradation, food security and sustainable development (*medium confidence*). Applied on a limited share of total land, land-based mitigation measures that displace other land uses have fewer adverse side-effects and can have positive co-benefits for adaptation, desertification, land degradation or food security. (*high confidence*) {4.2, 4.5, 6.4; Cross-Chapter Box 7 in Chapter 6, Figure SPM3}

B3.3 The production and use of biomass for bioenergy can have co-benefits, adverse side effects, and risks for land degradation, food insecurity, GHG emissions and other environmental and sustainable development goals (*high confidence*). These impacts are context specific and depend on the scale of deployment, initial land use, land type, bioenergy feedstock, initial carbon stocks, climatic region and management regime, and other land-demanding response options can have a similar range of consequences (*high confidence*). The use of residues and organic waste as bioenergy feedstock can mitigate land use change pressures associated with bioenergy deployment, but residues are limited and the removal of residues that would otherwise be left on the soil could lead to soil degradation (*high confidence*). {2.6.1.5; Cross-Chapter Box 7 in Chapter 6; Figure SPM3}

B3.4. For projected socioeconomic pathways with low population, effective land-use regulation, food produced in low-GHG emission systems and lower food loss and waste (SSP1), the transition from low to moderate risk to food security, land degradation and water scarcity in dry lands occur between 1 and 4 million km^2 of bioenergy or BECCS (*medium confidence*). By contrast, in pathways with high population, low income and slow rates of technological change (SSP3), the transition from low to moderate risk occurs between 0.1 and 1 million km^2 (*medium confidence*). {6.4; Cross-Chapter Box 7 in Chapter 6; Table SM7.6; Box SPM1}

B 4. Many activities for combating desertification can contribute to climate change adaptation with mitigation co-benefits, as well as to halting biodiversity loss with sustainable development co-benefits to society (*high confidence*). Avoiding, reducing and reversing desertification would enhance soil fertility, increase carbon storage in soils and biomass, while benefitting agricultural productivity and food security (*high confidence*). Preventing desertification is preferable to attempting to restore degraded land due to the potential for residual risks and maladaptive outcomes (*high confidence*). {3.6.1, 3.6.2, 3.6.3, 3.6.4, 3.7.1, 3.7.2}

B4.1. Solutions that help adapt to and mitigate climate change while contributing to combating desertification are site and regionally specific and include *inter alia*: water harvesting and micro-irrigation, restoring degraded lands using drought-resilient ecologically appropriate plants; agroforestry and other agroecological and ecosystem-based adaptation practices (*high confidence*). {3.3, 3.6.1, 3.7.2, 3.7.5, 5.2, 5.6}

B4.2. Reducing dust and sand storms and sand dune movement can lessen the negative effects of wind erosion and improve air quality and health (*high confidence*). Depending on water availability and soil conditions, afforestation, tree planting and ecosystem restoration programs, which aim for the creation of windbreaks in the form of "green walls", and "green dams" using native and other climate resilient tree species with low water needs, can reduce sand storms, avert wind erosion, and contribute to carbon sinks, while improving micro-climates, soil nutrients and water retention (*high confidence*). {3.3, 3.6.1, 3.7.2, 3.7.5}

B4.3. Measures to combat desertification can promote soil carbon sequestration (*high confidence*). Natural vegetation restoration and tree planting on degraded land enriches, in the long term, carbon in the topsoil and subsoil (*medium confidence*). Modelled rates of carbon sequestration following the adoption of conservation agriculture practices in drylands depend on local conditions (*medium confidence*). If soil carbon is lost, it may take a prolonged period of time for carbon stocks to recover. {3.1.4, 3.3, 3.6.1, 3.6.3, 3.7.1, 3.7.2}

B4.4 Eradicating poverty and ensuring food security can benefit from applying measures promoting land degradation neutrality (including avoiding, reducing and reversing land degradation) in rangelands, croplands and forests, which contribute to combating desertification, while mitigating and adapting to climate change within the framework of sustainable development. Such measures include avoiding deforestation and locally suitable practices including management of rangeland and forest fires (*high confidence*). {3.4.2, 3.6.1, 3.6.2, 3.6.3, 4.8.5}.

B4.5 Currently there is a lack of knowledge of adaptation limits and potential maladaptation to combined effects of climate change and desertification. In the absence of new or enhanced adaptation options, the potential for residual risks and maladaptive outcomes is high (*high confidence*). Even when solutions are available, social, economic and institutional constraints could pose barriers to their implementation (*medium confidence*). Some adaptation options can become maladaptive due to their environmental impacts, such as irrigation causing soil salinisation or over extraction leading to ground-water depletion (*medium confidence*). Extreme forms of desertification can lead to the complete loss of land productivity, limiting adaptation options or reaching the limits to adaptation (*high confidence*). {Executive Summary Chapter 3, 3.6.4, 3.7.5, 7.4.9}

B4.6. Developing, enabling and promoting access to cleaner energy sources and technologies can contribute to adaptation and mitigating climate change and combating desertification and forest degradation through decreasing the use of traditional biomass for energy while increasing the diversity of energy supply (*medium confidence*). This can have socioeconomic and health benefits, especially for women and children. (*high confidence*). The efficiency of wind and solar energy infrastructures is recognized; the efficiency can be affected in some regions by dust and sand storms (*high confidence*). {3.5.3, 3.5.4, 4.4.4, 7.5.2, Cross-Chapter Box 12 in Chapter 7}

B 5. Sustainable land management[33], including sustainable forest management[34], can prevent and reduce land degradation, maintain land productivity, and sometimes reverse the adverse impacts of climate change on land degradation (*very high confidence*). It can also contribute to mitigation and adaptation (*high confidence*). Reducing and reversing land degradation, at scales from individual farms to entire watersheds, can provide cost effective, immediate, and long-term benefits to communities and support several Sustainable Development Goals (SDGs) with co-benefits for adaptation (*very high confidence*) and mitigation (*high confidence*). Even with implementation of sustainable land management, limits to adaptation can be exceeded in some situations (*medium confidence*). {1.3.2, 4.1.5, 4.8, Table 4.2}

B5.1. Land degradation in agriculture systems can be addressed through sustainable land management, with an ecological and socioeconomic focus, with co-benefits for climate change adaptation. Management options that reduce vulnerability to soil erosion and nutrient loss include growing green manure crops and cover crops, crop residue retention, reduced/zero tillage, and maintenance of ground cover through improved grazing management (*very high confidence*). {4.8}

B5.2. The following options also have mitigation co-benefits. Farming systems such as agroforestry, perennial pasture phases and use of perennial grains, can substantially reduce erosion and nutrient leaching while building soil carbon (*high confidence*). The global sequestration potential of cover crops would be about 0.44 +/- 0.11 $GtCO_2\ yr^{-1}$ if applied to 25% of global cropland (*high confidence*). The application of certain biochars can sequester carbon (*high confidence*), and improve soil conditions in some soil types/climates (*medium confidence*). {4.8.1.1, 4.8.1.3, 4.9.2, 4.9.5, 5.5.1, 5.5.4; Cross-Chapter Box 6 in Chapter 5}

B5.3. Reducing deforestation and forest degradation lowers GHG emissions (*high confidence*), with an estimated technical mitigation potential of 0.4–5.8 $GtCO_2\ yr^{-1}$. By providing long-term livelihoods for communities, sustainable forest management can reduce the extent of

[33] Sustainable land management is defined in this report as the stewardship and use of land resources, including soils, water, animals and plants, to meet changing human needs, while simultaneously ensuring the long-term productive potential of these resources and the maintenance of their environmental functions. Examples of options include inter alia agroecology (including agroforestry), conservation agriculture and forestry practices, crop and forest species diversity, appropriate crop and forest rotations, organic farming, integrated pest management, the conservation of pollinators, rain water harvesting, range and pasture management, and precision agriculture systems.

[34] Sustainable forest management is defined in this report as the stewardship and use of forests and forest lands in a way, and at a rate, that maintains their biodiversity, productivity, regeneration capacity, vitality, and their potential to fulfill now and in the future, relevant ecological, economic and social functions at local, national and global levels and that does not cause damage to other ecosystems.

forest conversion to non-forest uses (e.g., cropland or settlements) (*high confidence*). Sustainable forest management aimed at providing timber, fibre, biomass, non-timber resources and other ecosystem functions and services, can lower GHG emissions and can contribute to adaptation. (*high confidence*). {2.6.1.2, 4.1.5, 4.3.2, 4.5.3, 4.8.1.3, 4.8.3, 4.8.4}

B5.4. Sustainable forest management can maintain or enhance forest carbon stocks, and can maintain forest carbon sinks, including by transferring carbon to wood products, thus addressing the issue of sink saturation (*high confidence*). Where wood carbon is transferred to harvested wood products, these can store carbon over the long-term and can substitute for emissions-intensive materials reducing emissions in other sectors (*high confidence*). Where biomass is used for energy, e.g., as a mitigation strategy, the carbon is released back into the atmosphere more quickly (*high confidence*). {2.6.1, 2.7, 4.1.5, 4.8.4, 6.4.1, Figure SPM.3, Cross-Chapter Box 7 in Chapter 6}

B5.5. Climate change can lead to land degradation, even with the implementation of measures intended to avoid, reduce or reverse land degradation (*high confidence*). Such limits to adaptation are dynamic, site specific and are determined through the interaction of biophysical changes with social and institutional conditions (*very high confidence*). In some situations, exceeding the limits of adaptation can trigger escalating losses or result inundesirable transformational changes (*medium confidence)*, such as forced migration (*low confidence*), conflicts (*low confidence*) or poverty (*medium confidence*). Examples of climate change induced land degradation that may exceed limits to adaptation include coastal erosion exacerbated by sea level rise where land disappears (*high confidence*), thawing of permafrost affecting infrastructure and livelihoods (*medium confidence*), and extreme soil erosion causing loss of productive capacity (*medium confidence*). {4.7, 4.8.5, 4.8.6, 4.9.6, 4.9.7, 4.9.8}

B 6. Response options throughout the food system, from production to consumption, including food loss and waste, can be deployed and scaled up to advance adaptation and mitigation (*high confidence*). The total technical mitigation potential from crop and livestock activities, and agroforestry is estimated as 2.3-9.6 $GtCO_2e.yr^{-1}$ by 2050 (*medium confidence)*. The total technical mitigation potential of dietary changes is estimated as 0.7-8 $GtCO_2e$.yr-1 by 2050 (*medium confidence*). {5.3, 5.5, 5.6}

B6.1. Practices that contribute to climate change adaptation and mitigation in cropland include increasing soil organic matter, erosion control, improved fertiliser management, improved crop management, for example, paddy rice management, and use ofvarieties and genetic improvements for heat and drought tolerance. For livestock, options include better grazing land management, improved manure management, higher-quality feed, and use of breeds and genetic improvement. Different farming and pastoral systems can achieve reductions in the emissions intensity of livestock products. Depending on the farming and pastoral systems and level of development, reductions in the emissions intensity of livestock products may lead to absolute reductions in GHG emissions (*medium confidence*). Many livestock related options can enhance the adaptive capacity of rural communities, in particular, of smallholders and pastoralists. Significant synergies exist between adaptation and mitigation, for example through sustainable land management approaches (*high confidence*). {4.8, 5.3.3, 5.5.1, 5.6}

B6.2. Diversification in the food system (e.g., implementation of integrated production systems, broad-based genetic resources, and diets) can reduce risks from climate change (*medium confidence*). Balanced diets, featuring plant-based foods, such as those based on coarse grains, legumes, fruits and vegetables, nuts and seeds, and animal-sourced food produced in resilient, sustainable and low-GHG emission systems, present major opportunities for adaptation and mitigation while generating significant co-benefits in terms of human health (*high confidence*). By 2050, dietary changes could free several Mkm^2 (*medium confidence*) of land and provide a technical mitigation potential of 0.7 to 8.0 $GtCO_2e$ yr^{-1}, relative to business as usual projections (*high confidence*). Transitions towards low-GHG emission diets may be influenced by local production practices, technical and financial barriers and associated livelihoods and cultural habits (*high confidence*). {5.3, 5.5.2, 5.5, 5.6}

B6.3. Reduction of food loss and waste can lower GHG emissions and contribute to adaptation through reduction in the land area needed for food production (*medium confidence*). During 2010-2016, global food loss and waste contributed 8-10% of total anthropogenic GHG emissions (*medium confidence*). Currently, 25-30% of total food produced is lost or wasted (*medium confidence*). Technical options such as improved harvesting techniques, on-farm storage, infrastructure, transport, packaging, retail and education can reduce food loss and waste across the supply chain. Causes of food loss and waste differ substantially between developed and developing countries, as well as between regions (*medium confidence*). {5.5.2} By 2050, reduced food loss and waste can free several Mkm^2 of land (*low confidence).* {6.3.6}

B 7. Future land use depends, in part, on the desired climate outcome and the portfolio of response options deployed (*high confidence*). All assessed modelled pathways that limit warming to 1.5ºC or well below 2°C require land-based mitigation and land-use change, with most including different combinations of reforestation, afforestation, reduced deforestation, and bioenergy (*high confidence*). A small number of modelled pathways achieve 1.5ºC with reduced land conversion (*high confidence*) and, thus, reduced consequences for desertification, land degradation, and food security (*medium confidence*). {2.6, 6.4, 7.4, 7.6; Cross-Chapter Box 9 in Chapter 6; Figure SPM.4}

B7.1. Modelled pathways limiting global warming to 1.5ºC[35] include more land-based mitigation than higher warming level pathways (*high confidence*), but the impacts of climate change on land systems in these pathways are less severe (*medium confidence*). {2.6, 6.4, 7.4, Cross-Chapter Box 9 in Chapter 6, Figure SPM.2, Figure SPM.4}

B7.2. Modelled pathways limiting global warming to 1.5°C and 2ºC project a 2 million km^2 reduction to a 12 million km^2 increase in forest area in 2050 relative to 2010 (*medium confidence*). 3ºC pathways project lower forest areas, ranging from a 4 million km^2 reduction to a 6 million km^2 increase (*medium confidence*). {2.5, 6.3, 7.3, 7.5; Cross-Chapter Box 9 in Chapter 6; Figure SPM.3, Figure SPM.4}

B7.3. The land area needed for bioenergy in modelled pathways varies significantly depending on the socioeconomic pathway, the warming level, and the feedstock and production system used (*high confidence*). Modelled pathways limiting global warming to 1.5°C use up to 7 million km^2 for bioenergy in 2050; bioenergy land area is smaller in 2°C (0.4 to 5 million km^2) and 3°C pathways (0.1 to 3 million km^2) (*medium confidence*). Pathways with large levels of land conversion may imply adverse side-effects impacting water scarcity, biodiversity, land degradation, desertification, and food security, if not adequately and carefully managed, whereas best practice implementation at appropriate scales can have co-benefits, such as management of dryland salinity, enhanced biocontrol and biodiversity and enhancing soil carbon sequestration (*high confidence*). {2.6, 6.1, 6.4, 7.2; Cross-Chapter Box 7 in Chapter 6, Figure SPM.3}

B7.4. Most mitigation pathways include substantial deployment of bioenergy technologies. A small number of modelled pathways limit warming to 1.5ºC with reduced dependence on bioenergy and BECCS (land area below <1 million km^2 in 2050) and other carbon dioxide removal (CDR) options (*high confidence*). These pathways have even more reliance on rapid and far-reaching transitions in energy, land, urban systems and infrastructure, and on behavioural and lifestyle changes compared to other 1.5°C pathways. {2.6.2, 5.5.1, 6.4, Cross-Chapter Box 7 in Chapter 6

B7.5. These modelled pathways do not consider the effects of climate change on land or CO_2 fertilisation. In addition, these pathways include only a subset of the response options assessed in this report (*high confidence*); the inclusion of additional response options in models could reduce the projected need for bioenergy or CDR that increases the demand for land. {6.4.4, Cross-Chapter Box 9 in Chapter 6}

[35] In this report references to pathways limiting global warming to a particular level are based on a 66% probability of staying below that temperature level in 2100 using the MAGICC model.

Potential global contribution of response options to mitigation, adaptation, combating desertification and land degradation, and enhancing food security

Panel A shows response options that can be implemented without or with limited competition for land, including some that have the potential to reduce the demand for land. Co-benefits and adverse side effects are shown quantitatively based on the high end of the range of potentials assessed. Magnitudes of contributions are categorised using thresholds for positive or negative impacts. Letters within the cells indicate confidence in the magnitude of the impact relative to the thresholds used (see legend). Confidence in the direction of change is generally higher.

Response options based on land management

	Response option	Mitigation	Adaptation	Desertification	Land Degradation	Food Security	Cost
Agriculture	Increased food productivity	L	M	L	M	H	—
	Agro-forestry	M	M	M	M	L	●
	Improved cropland management	M	L	L	L	L	●●
	Improved livestock management	M	L	L	L	L	●●●
	Agricultural diversification	L	L	L	M	L	●
	Improved grazing land management	M	L	L	L	L	—
	Integrated water management	L	L	L	L	L	●●
	Reduced grassland conversion to cropland	L	—	L	L	- L	●
Forests	Forest management	M	L	L	L	L	●●
	Reduced deforestation and forest degradation	H	L	L	L	L	●●
Soils	Increased soil organic carbon content	H	L	M	M	L	●●
	Reduced soil erosion	↔ L	L	M	M	L	●●
	Reduced soil salinization	—	L	L	L	L	●●
	Reduced soil compaction	—	L	—	L	L	●
Other ecosystems	Fire management	M	M	M	M	L	●
	Reduced landslides and natural hazards	L	L	L	L	L	—
	Reduced pollution including acidification	↔ M	M	L	L	L	—
	Restoration & reduced conversion of coastal wetlands	M	L	M	M	↔ L	—
	Restoration & reduced conversion of peatlands	M	—	na	M	- L	●

Response options based on value chain management

	Response option	Mitigation	Adaptation	Desertification	Land Degradation	Food Security	Cost
Demand	Reduced post-harvest losses	H	M	L	L	H	—
	Dietary change	H	—	L	H	H	—
	Reduced food waste (consumer or retailer)	H	—	L	M	M	—
Supply	Sustainable sourcing	—	L	—	L	L	—
	Improved food processing and retailing	L	L	—	—	L	—
	Improved energy use in food systems	L	L	—	—	L	—

Response options based on risk management

	Response option	Mitigation	Adaptation	Desertification	Land Degradation	Food Security	Cost
Risk	Livelihood diversification	—	L	—	L	L	—
	Management of urban sprawl	—	L	L	M	L	—
	Risk sharing instruments	↔ L	L	—	↔ L	L	●●

Options shown are those for which data are available to assess global potential for three or more land challenges. The magnitudes are assessed independently for each option and are not additive.

Key for criteria used to define magnitude of impact of each integrated response option

		Mitigation *Gt CO_2-eq yr^{-1}*	Adaptation *Million people*	Desertification *Million km^2*	Land Degradation *Million km^2*	Food Security *Million people*
Positive	Large	More than 3	Positive for more than 25	Positive for more than 3	Positive for more than 3	Positive for more than 100
	Moderate	0.3 to 3	1 to 25	0.5 to 3	0.5 to 3	1 to 100
	Small	Less than 0.3	Less than 1	Less than 0.5	Less than 0.5	Less than 1
	Negligible	No effect	No effect	No effect	No effect	No effect
Negative	Small	Less than -0.3	Less than 1	Less than 0.5	Less than 0.5	Less than 1
	Moderate	-0.3 to -3	1 to 25	0.5 to 3	0.5 to 3	1 to 100
	Large	More than -3	Negative for more than 25	Negative for more than 3	Negative for more than 3	Negative for more than 100

↔ **Variable:** Can be positive or negative — no data na not applicable

Confidence level

Indicates confidence in the estimate of magnitude category.

H High confidence
M Medium confidence
L Low confidence

Cost range

See technical caption for cost ranges in US\$ tCO_2e^{-1} or US\$ ha^{-1}.

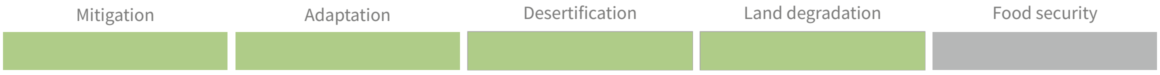
●●● High cost
●● Medium cost
● Low cost
— no data

Potential global contribution of response options to mitigation, adaptation, combating desertification and land degradation, and enhancing food security

Panel B shows response options that rely on additional land-use change and could have implications across three or more land challenges under different implementation contexts. For each option, the first row (high level implementation) shows a quantitative assessment (as in Panel A) of implications for global implementation at scales delivering CO_2 removals of more than 3 $GtCO_2$ yr^{-1} using the magnitude thresholds shown in Panel A. The red hatched cells indicate an increasing pressure but unquantified impact. For each option, the second row (best practice implementation) shows qualitative estimates of impact if implemented using best practices in appropriately managed landscape systems that allow for efficient and sustainable resource use and supported by appropriate governance mechanisms. In these qualitative assessments, green indicates a positive impact, grey indicates a neutral interaction.

Bioenergy and BECCS

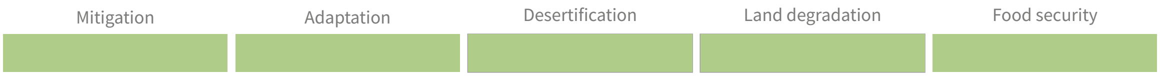
High level: Impacts on adaptation, desertification, land degradation and food security are maximum potential impacts, assuming carbon dioxide removal by BECCS at a scale of 11.3 $GtCO_2$ yr^{-1} in 2050, and noting that bioenergy without CCS can also achieve emissions reductions of up to several $GtCO_2$ yr^{-1} when it is a low carbon energy source {2.7.1.5; 6.4.1.1.5}. Studies linking bioenergy to food security estimate an increase in the population at risk of hunger to up to 150 million people at this level of implementation {6.4.5.1.5}. The red hatched cells for desertification and land degradation indicate that while up to 15 million km_2 of additional land is required in 2100 in 2°C scenarios which will increase pressure for desertification and land degradation, the actual area affected by this additional pressure is not easily quantified {6.4.3.1.5; 6.4.4.1.5}.



Best practice: The sign and magnitude of the effects of bioenergy and BECCS depends on the scale of deployment, the type of bioenergy feedstock, which other response options are included, and where bioenergy is grown (including prior land use and indirect land use change emissions). For example, limiting bioenergy production to marginal lands or abandoned cropland would have negligible effects on biodiversity, food security, and potentially co-benefits for land degradation; however, the benefits for mitigation could also be smaller. {Table 6.58}

Reforestation and forest restoration

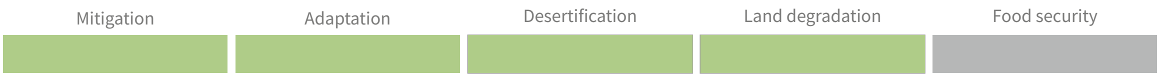
High level: Impacts on adaptation, desertification, land degradation and food security are maximum potential impacts assuming implementation of reforestation and forest restoration (partly overlapping with afforestation) at a scale of 10.1 $GtCO_2$ yr^{-1} removal {6.4.1.1.2}. Large-scale afforestation could cause increases in food prices of 80% by 2050, and more general mitigation measures in the AFOLU sector can translate into a rise in undernourishment of 80–300 million people; the impact of reforestation is lower {6.4.5.1.2}.



Best practice: There are co-benefits of reforestation and forest restoration in previously forested areas, assuming small scale deployment using native species and involving local stakeholders to provide a safety net for food security. Examples of sustainable implementation include, but are not limited to, reducing illegal logging and halting illegal forest loss in protected areas, reforesting and restoring forests in degraded and desertified lands {Box6.1C; Table 6.6}.

Afforestation

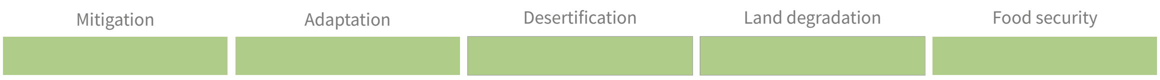
High level: Impacts on adaptation, desertification, land degradation and food security are maximum potential impacts assuming implementation of afforestation (partly overlapping with reforestation and forest restoration) at a scale of 8.9 $GtCO_2$ yr^{-1} removal {6.4.1.1.2}. Large-scale afforestation could cause increases in food prices of 80% by 2050, and more general mitigation measures in the AFOLU sector can translate into a rise in undernourishment of 80–300 million people {6.4.5.1.2}.



Best practice: Afforestation is used to prevent desertification and to tackle land degradation. Forested land also offers benefits in terms of food supply, especially when forest is established on degraded land, mangroves, and other land that cannot be used for agriculture. For example, food from forests represents a safety-net during times of food and income insecurity {6.4.5.1.2}.

Biochar addition to soil

High level: Impacts on adaptation, desertification, land degradation and food security are maximum potential impacts assuming implementation of afforestation at a scale of 6.6 $GtCO_2$ yr^{-1} removal {6.4.1.1.3}. Dedicated energy crops required for feedstock production could occupy 0.4–2.6 Mkm^2 of land, equivalent to around 20% of the global cropland area, which could potentially have a large effect on food security for up to 100 million people {6.4.5.1.3}.



Best practice: When applied to land, biochar could provide moderate benefits for food security by improving yields by 25% in the tropics, but with more limited impacts in temperate regions, or through improved water holding capacity and nutrient use efficiency. Abandoned cropland could be used to supply biomass for biochar, thus avoiding competition with food production; 5-9 Mkm^2 of land is estimated to be available for biomass production without compromising food security and biodiversity, considering marginal and degraded land and land released by pasture intensification {6.4.5.1.3}.

Figure SPM.3 Potential global contribution of response options to mitigation, adaptation, combating desertification and land degradation, and enhancing food security.

This Figure is based on an aggregation of information from studies with a wide variety of assumptions about how response options are implemented and the contexts in which they occur. Response options implemented differently at local to global scales could lead to different outcomes. **Magnitude of potential:** For panel A, magnitudes are for the technical potential of response options globally. For each land challenge, magnitudes are set relative to a marker level as follows. For mitigation, potentials are set relative to the approximate potentials for the response options with the largest individual impacts (~3 $GtCO_2$-eq yr^{-1}). The threshold for the "large" magnitude category is set at this level. For adaptation, magnitudes are set relative to the 100 million lives estimated to be affected by climate change and a carbon-based economy between 2010 and 2030. The threshold for the "large" magnitude category represents 25% of this total. For desertification and land degradation, magnitudes are set relative to the lower end of current estimates of degraded land, 10-60 million km^2. The threshold for the "large" magnitude category represents 30% of the lower estimate. For food security, magnitudes are set relative to the approximately 800 million people who are currently undernourished. The threshold for the "large" magnitude category represents 12.5% of this total. For panel B, for the first row (high level implementation) for each response option, the magnitude and thresholds are as defined for panel A. In the second row (best practice implementation) for each response option, the qualitative assessments that are green denote potential positive impacts, and those shown in grey indicate neutral interactions. Increased food production is assumed to be achieved through sustainable intensification rather than through injudicious application of additional external inputs such as agrochemicals. **Levels of confidence:** Confidence in the magnitude category (high, medium or low) into which each option falls for mitigation, adaptation, combating desertification and land degradation, and enhancing food security. *High confidence* means that there is a high level of agreement and evidence in the literature to support the categorisation as high, medium or low magnitude. *Low confidence* denotes that the categorisation of magnitude is based on few studies. *Medium confidence* reflects medium evidence and agreement in the magnitude of response. **Cost ranges:** Cost estimates are based on aggregation of often regional studies and vary in the components of costs that are included. In panel B, cost estimates are not provided for best practice implementation. One coin indicates low cost (<USD10 tCO_2-eq^{-1} or <USD20 ha^{-1}), two coins indicate medium cost (USD10-USD100 tCO_2-eq^{-1} or USD20-USD200 ha^{-1}), and three coins indicate high cost (>USD100 tCO_2-eq^{-1} or USD200 ha^{-1}). Thresholds in USD ha^{-1} are chosen to be comparable, but precise conversions will depend on the response option. **Supporting evidence:** Supporting evidence for the magnitude of the quantitative potential for land management-based response options can be found as follows: for mitigation tables 6.13 to 6.20, with further evidence in Section 2.7.1; for adaptation tables 6.21 to 6.28; for combating desertification tables 6.29 to 6.36, with further evidence in Chapter 3; for combating degradation tables 6.37 to 6.44, with further evidence in Chapter 4; for enhancing food security tables 6.45 to 6.52, with further evidence in Chapter 5. Other synergies and trade-offs not shown here are discussed in Chapter 6. Additional supporting evidence for the qualitative assessments in the second row for each option in panel B can be found in the tables 6.6, 6.55, 6.56 and 6.58, section 6.3.5.1.3, and Box 6.1c.

C. Enabling response options

C 1. Appropriate design of policies, institutions and governance systems at all scales can contribute to land-related adaptation and mitigation while facilitating the pursuit of climate-adaptive development pathways (*high confidence*). Mutually supportive climate and land policies have the potential to save resources, amplify social resilience, support ecological restoration, and foster engagement and collaboration between multiple stakeholders (*high confidence*). {Figure SPM.1, Figure SPM.2, Figure SPM.3; 3.6.2, 3.6.3, 4.8, 4.9.4, 5.7, 6.3, 6.4, 7.2.2, 7.3, 7.4, 7.4.7, 7.4.8, 7.5, 7.5.5, 7.5.6, 7.6.6; Cross-Chapter Box 10 in Chapter 7}

C1.1. Land-use zoning, spatial planning, integrated landscape planning, regulations, incentives (such as payment for ecosystem services), and voluntary or persuasive instruments (such as environmental farm planning, standards and certification for sustainable production, use of scientific, local and indigenous knowledge and collective action), can achieve positive adaptation and mitigation outcomes (*medium confidence*). They can also contribute revenue and provide incentive to rehabilitate degraded lands and adapt to and mitigate climate change in certain contexts (*medium confidence*). Policies promoting the target of land degradation neutrality can also support food security, human wellbeing and climate change adaptation and mitigation (*high confidence*). {Figure SPM.2; 3.4.2, 4.1.6, 4.7, 4.8.5, 5.1.2, 5.7.3, 7.3, 7.4.6, 7.4.7, 7.5}

C1.2. Insecure land tenure affects the ability of people, communities and organisations to make changes to land that can advance adaptation and mitigation (*medium confidence*). Limited recognition of customary access to land and ownership of land can result in increased vulnerability and decreased adaptive capacity (*medium confidence*). Land policies (including recognition of customary tenure, community mapping, redistribution, decentralisation, co-management, regulation of rental markets) can provide both security and flexibility response to climate change (*medium confidence*). {3.6.1, 3.6.2, 5.3, 7.2.4, 7.6.4, Cross-Chapter Box 6 in Chapter 5}

C1.3. Achieving land degradation neutrality will involve a balance of measures that avoid and reduce land degradation, through adoption of sustainable land management, and measures to reverse degradation through rehabilitation and restoration of degraded land. Many interventions to achieve land degradation neutrality commonly also deliver climate change adaptation and mitigation benefits. The pursuit of land degradation neutrality provides impetus to address land degradation and climate change simultaneously (*high confidence*). {4.5.3, 4.8.5, 4.8.7, 7.4.5}

C1.4. Due to the complexity of challenges and the diversity of actors involved in addressing land challenges, a mix of policies, rather than single policy approaches, can deliver improved results in addressing the complex challenges of sustainable land management and climate change (*high confidence*). Policy mixes can strongly reduce the vulnerability and exposure of human and natural systems to climate change (*high confidence*). Elements of such policy mixes may include weather and health insurance, social protection and adaptive safety nets, contingent finance and reserve funds, universal access to early warning systems combined with effective contingency plans (*high confidence*). {1.2, 4.8, 4.9.2, 5.3.2, 5.6, 5.6.6, 5.7.2, 7.3.2, 7.4, 7.4.2, 7.4.6, 7.4.7, 7.4.8, 7.5.5, 7.5.6, 7.6.4, Figure SPM.4}

C2. Policies that operate across the food system, including those that reduce food loss and waste and influence dietary choices, enable more sustainable land-use management, enhanced food security and low emissions trajectories (*high confidence*). Such policies can contribute to climate change adaptation and mitigation, reduce land degradation, desertification and poverty as well as improve public health (*high confidence*). The adoption of sustainable land management and poverty eradication can be enabled by improving access to markets, securing land tenure, factoring environmental costs into food, making payments for ecosystem services, and enhancing local and community collective action *(high confidence)*. {1.1.2, 1.2.1, 3.6.3, 4.7.1, 4.7.2, 4.8, 5.5, 6.4, 7.4.6, 7.6.5}

C2.1. Policies that enable and incentivise sustainable land management for climate change adaptation and mitigation include improved access to markets for inputs, outputs and financial services, empowering women and indigenous peoples, enhancing local and community collective action, reforming subsidies and promoting an enabling trade system (*high confidence*). Land restoration and rehabilitation efforts can be more effective when policies support local management of natural resources, while strengthening cooperation between actors and institutions, including at the international level. {3.6.3, 4.1.6, 4.5.4, 4.8.2, 4.8.4, 5.7, 7.2}

C2.2. Reflecting the environmental costs of land-degrading agricultural practices can incentivise more sustainable land management *(high confidence)*. Barriers to the reflection of environmental costs arise from technical difficulties in estimating these costs and those embodied in foods. {3.6.3, 5.5.1, 5.5.2, 5.6.6, 5.7, 7.4.4, Cross-Chapter Box 10 in Chapter 7}

C2.3. Adaptation and enhanced resilience to extreme events impacting food systems can be facilitated by comprehensive risk management, including risk sharing and transfer mechanisms *(high confidence)*. Agricultural diversification, expansion of market access, and preparation for increasing supply chain disruption can support the scaling up of adaptation in food systems (*high confidence)*. {5.3.2, 5.3.3, 5.3.5}

C2.4. Public health policies to improve nutrition, such as increasing the diversity of food sources in public procurement, health insurance, financial incentives, and awareness-raising campaigns, can potentially influence food demand, reduce healthcare costs, contribute to lower GHG emissions and enhance adaptive capacity *(high confidence)*. Influencing demand for food, through promoting diets based on public health guidelines, can enable more sustainable land management and contribute to achieving multiple SDGs *(high confidence)*. {3.4.2, 4.7.2, 5.1, 5.7, 6.3, 6.4}

C 3. Acknowledging co-benefits and trade-offs when designing land and food policies can overcome barriers to implementation (*medium confidence*). Strengthened multilevel, hybrid and cross-sectoral governance, as well as policies developed and adopted in an iterative, coherent, adaptive and flexible manner can maximise co-benefits and minimise trade-offs, given that land management decisions are made from farm level to national scales, and both climate and land policies often range across multiple sectors, departments and agencies (*high confidence*). {Figure SPM.3; 4.8.5, 4.9, 5.6, 6.4, 7.3, 7.4.6, 7.4.8, 7.4.9, 7.5.6, 7.6.2}

C3.1. Addressing desertification, land degradation, and food security in an integrated, coordinated and coherent manner can assist climate resilient development and provides numerous potential co-benefits (*high confidence*). {3.7.5, 4.8, 5.6, 5.7, 6.4, 7.2.2, 7.3.1, 7.3.4, 7.4.7, 7.4.8, 7.5.6, 7.5.5}

C3.2. Technological, biophysical, socio-economic, financial and cultural barriers can limit the adoption of many land-based response options, as can uncertainty about benefits (*high confidence*). Many sustainable land management practices are not widely adopted due to insecure land tenure, lack of access to resources and agricultural advisory services, insufficient and unequal private and public incentives, and lack of knowledge and practical experience (*high confidence*). Public discourse, carefully designed policy interventions, incorporating social learning and market changes can together help reduce barriers to implementation (*medium confidence*). {3.6.1, 3.6.2, 5.3.5, 5.5.2, 5.6, 6.2, 6.4, 7.4, 7.5, 7.6}

C3.3. The land and food sectors face particular challenges of institutional fragmentation and often suffer from a lack of engagement between stakeholders at different scales and narrowly focused policy objectives (*medium confidence*). Coordination with other sectors, such as public health, transportation, environment, water, energy and infrastructure, can increase co-benefits, such as risk reduction and improved health (*medium confidence*). {5.6.3, 5.7, 6.2, 6.4.4, 7.1, 7.3, 7.4.8, 7.6.2, 7.6.3}

C3.4. Some response options and policies may result in trade-offs, including social impacts, ecosystem functions and services damage, water depletion, or high costs, that cannot be well-managed, even with institutional best practices (*medium confidence*). Addressing such trade-offs helps avoid maladaptation (*medium confidence).* Anticipation and evaluation of potential trade-offs and knowledge gaps supports evidence-based policymaking to weigh the costs and benefits of specific responses for different stakeholders (*medium confidence*). Successful management of trade-offs often includes maximising stakeholder input with structured feedback processes, particularly in community-based models, use of innovative fora like facilitated dialogues or spatially explicit mapping, and iterative adaptive management that allows for continuous readjustments in policy as new evidence comes to light (*medium confidence*). {5.3.5, 6.4.2, 6.4.4, 6.4.5, 7.5.6; Cross-Chapter Box 13 in Chapter 7}

C 4. The effectiveness of decision-making and governance is enhanced by the involvement of local stakeholders (particularly those most vulnerable to climate change including indigenous peoples and local communities, women, and the poor and marginalised) in the selection, evaluation, implementation and monitoring of policy instruments for land-based climate change adaptation and mitigation (*high confidence*). Integration across sectors and scales increases the chance of maximising co-benefits and minimising trade-offs (*medium confidence*). {1.4, 3.1, 3.6, 3.7, 4.8, 4.9, 5.1.3, Box 5.1, 7.4, 7.6}

C4.1. Successful implementation of sustainable land management practices requires accounting for local environmental and socio-economic conditions (*very high confidence*). Sustainable land management in the context of climate change is typically advanced by involving all relevant stakeholders in identifying land-use pressures and impacts (such as biodiversity decline, soil loss, over-extraction of groundwater, habitat loss, land-use change in agriculture, food production and forestry) as well as preventing, reducing and restoring degraded land (*medium confidence*). {1.4.1, 4.1.6, 4.8.7, 5.2.5, 7.2.4, 7.6.2, 7.6.4}

C4.2. Inclusiveness in the measurement, reporting and verification of the performance of policy instruments can support sustainable land management (*medium confidence*). Involving stakeholders in the selection of indicators, collection of climate data, land modelling and land-use planning, mediates and facilitates integrated landscape planning and choice of policy (*medium confidence*). {3.7.5, 5.7.4, 7.4.1, 7.4.4, 7.5.3, 7.5.4, 7.5.5, 7.6.4, 7.6.6}

C4.3. Agricultural practices that include indigenous and local knowledge can contribute to overcoming the combined challenges of climate change, food security, biodiversity conservation, and combating desertification and land degradation (*high confidence*). Coordinated action across a range of actors including businesses, producers, consumers, land managers and policymakers in partnership with indigenous peoples and local communities enable conditions for the adoption of response options (*high confidence*) {3.1.3, 3.6.1, 3.6.2, 4.8.2, 5.5.1, 5.6.4, 5.7.1, 5.7.4, 6.2, 7.3, 7.4.6, 7.6.4}

C4.4. Empowering women can bring synergies and co-benefits to household food security and sustainable land management (*high confidence*). Due to women's disproportionate vulnerability to climate change impacts, their inclusion in land management and tenure is constrained. Policies that can address land rights and barriers to women's participation in sustainable land management include financial transfers to women under the auspices of anti-poverty programmes, spending on health, education, training and capacity building for women, subsidised credit and program dissemination through existing women's community-based organisations (*medium confidence*). {1.4.1, 4.8.2, 5.1.3, Box 5.1, Cross-Chapter Box 11 in Chapter 7}.

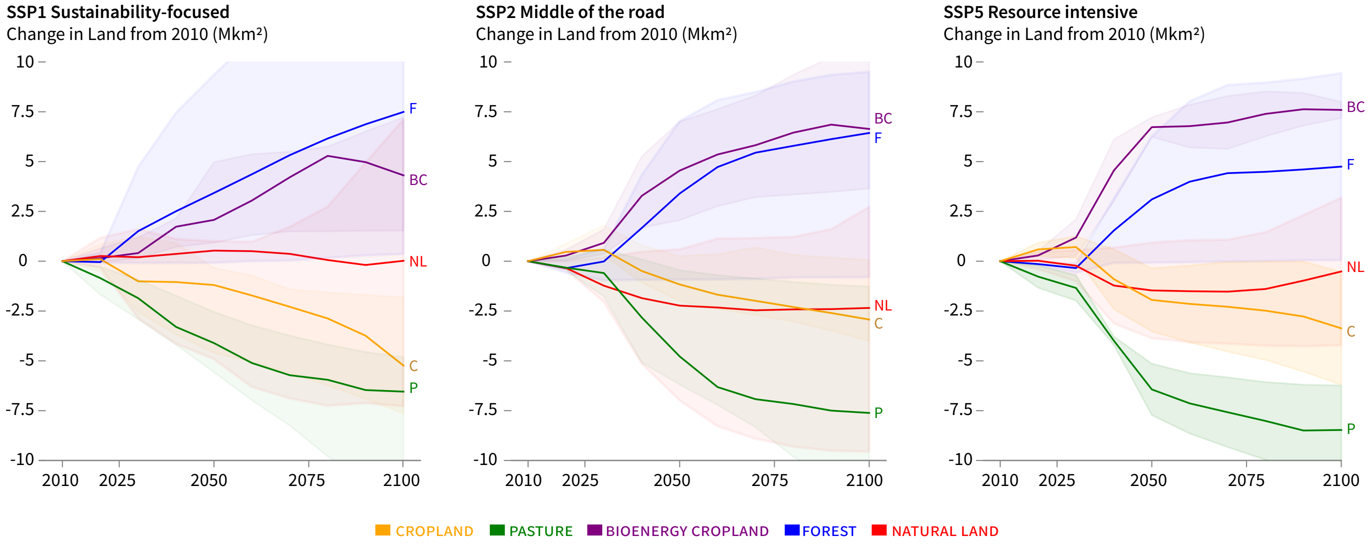
A. Pathways linking socioeconomic development, mitigation responses and land

Socioeconomic development and land management influence the evolution of the land system including the relative amount of land allocated to **CROPLAND**, **PASTURE**, **BIOENERGY CROPLAND**, **FOREST**, and **NATURAL LAND**. The lines show the median across Integrated Assessment Models (IAMs) for three alternative shared socioeconomic pathways (**SSP1**, **SSP2** and **SSP5** at **RCP1.9**); shaded areas show the range across models. Note that pathways illustrate the effects of climate change mitigation but not those of climate change impacts or adaptation.

A. Sustainability-focused (SSP1)
Sustainability in land management, agricultural intensification, production and consumption patterns result in reduced need for agricultural land, despite increases in per capita food consumption. This land can instead be used for reforestation, afforestation, and bioenergy.

B. Middle of the road (SSP2)
Societal as well as technological development follows historical patterns. Increased demand for land mitigation options such as bioenergy, reduced deforestation or afforestation decreases availability of agricultural land for food, feed and fibre.

C. Resource intensive (SSP5)
Resource-intensive production and consumption patterns, results in high baseline emissions. Mitigation focuses on technological solutions including substantial bioenergy and BECCS . Intensification and competing land uses contribute to declines in agricultural land.



B. Land use and land cover change in the SSPs

	Quantitative indicators for the SSPs	*Count of models included**	*Change in Natural Land from 2010 Mkm²*	*Change in Bioenergy Cropland from 2010 Mkm²*	*Change in Cropland from 2010 Mkm²*	*Change in Forest from 2010 Mkm²*	*Change in Pasture from 2010 Mkm²*
SSP1	**RCP1.9** in 2050	5/5	0.5 (-4.9, 1)	2.1 (0.9, 5)	-1.2 (-4.6, -0.3)	3.4 (-0.1, 9.4)	-4.1 (-5.6, -2.5)
	↳ 2100		0 (-7.3, 7.1)	4.3 (1.5, 7.2)	-5.2 (-7.6, -1.8)	7.5 (0.4, 15.8)	-6.5 (-12.2, -4.8)
	RCP2.6 in 2050	5/5	-0.9 (-2.2, 1.5)	1.3 (0.4, 1.9)	-1 (-4.7, 1)	2.6 (-0.1, 8.4)	-3 (-4, -2.4)
	↳ 2100		0.2 (-3.5, 1.1)	5.1 (1.6, 6.3)	-3.2 (-7.7, -1.8)	6.6 (-0.1, 10.5)	-5.5 (-9.9, -4.2)
	RCP4.5 in 2050	5/5	0.5 (-1, 1.7)	0.8 (0.5, 1.3)	0.1 (-3.2, 1.5)	0.6 (-0.7, 4.2)	-2.4 (-3.3, -0.9)
	↳ 2100		1.8 (-1.7, 6)	1.9 (1.4, 3.7)	-2.3 (-6.4, -1.6)	3.9 (0.2, 8.8)	-4.6 (-7.3, -2.7)
	Baseline in 2050	5/5	0.3 (-1.1, 1.8)	0.5 (0.2, 1.4)	0.2 (-1.6, 1.9)	-0.1 (-0.8, 1.1)	-1.5 (-2.9, -0.2)
	↳ 2100		3.3 (-0.3, 5.9)	1.8 (1.4, 2.4)	-1.5 (-5.7, -0.9)	0.9 (0.3, 3)	-2.1 (-7, 0)
SSP2	**RCP1.9** in 2050	4/5	-2.2 (-7, 0.6)	4.5 (2.1, 7)	-1.2 (-2, 0.3)	3.4 (-0.9, 7)	-4.8 (-6.2, -0.4)
	↳ 2100		-2.3 (-9.6, 2.7)	6.6 (3.6, 11)	-2.9 (-4, 0.1)	6.4 (-0.8, 9.5)	-7.6 (-11.7, -1.3)
	RCP2.6 in 2050	5/5	-3.2 (-4.2, 0.1)	2.2 (1.7, 4.7)	0.6 (-1.9, 1.9)	1.6 (-0.9, 4.2)	-1.4 (-3.7, 0.4)
	↳ 2100		-5.2 (-7.2, 0.5)	6.9 (2.3, 10.8)	-1.4 (-4, 0.8)	5.6 (-0.9, 5.9)	-7.2 (-8, 0.5)
	RCP4.5 in 2050	5/5	-2.2 (-2.2, 0.7)	1.5 (0.1, 2.1)	1.2 (-0.9, 2.7)	-0.9 (-2.5, 2.9)	-0.1 (-2.5, 1.6)
	↳ 2100		-3.4 (-4.7, 1.5)	4.1 (0.4, 6.3)	0.7 (-2.6, 3.1)	-0.5 (-3.1, 5.9)	-2.8 (-5.3, 1.9)
	Baseline in 2050	5/5	-1.5 (-2.6, -0.2)	0.7 (0, 1.5)	1.3 (1, 2.7)	-1.3 (-2.5, -0.4)	-0.1 (-1.2, 1.6)
	↳ 2100		-2.1 (-5.9, 0.3)	1.2 (0.1, 2.4)	1.9 (0.8, 2.8)	-1.3 (-2.7, -0.2)	-0.2 (-1.9, 2.1)
SSP3	**RCP1.9** in 2050	*Infeasible in all assessed models*		-	-	-	-
	↳ 2100			-	-	-	-
	RCP2.6 in 2050	*Infeasible in all assessed models*		-	-	-	-
	↳ 2100			-	-	-	-
	RCP4.5 in 2050	3/3	-3.4 (-4.4, -2)	1.3 (1.3, 2)	2.3 (1.2, 3)	-2.4 (-4, -1)	2.1 (-0.1, 3.8)
	↳ 2100		-6.2 (-6.8, -5.4)	4.6 (1.5, 7.1)	3.4 (1.9, 4.5)	-3.1 (-5.5, -0.3)	2 (-2.5, 4.4)
	Baseline in 2050	4/4	-3 (-4.6, -1.7)	1 (0.2, 1.5)	2.5 (1.5, 3)	-2.5 (-4, -1.5)	2.4 (0.6, 3.8)
	↳ 2100		-5 (-7.1, -4.2)	1.1 (0.9, 2.5)	5.1 (3.8, 6.1)	-5.3 (-6, -2.6)	3.4 (0.9, 6.4)
SSP4	**RCP1.9** in 2050	*Infeasible in all assessed models***		-	-	-	-
	↳ 2100			-	-	-	-
	RCP2.6 in 2050	3/3	-4.5 (-6, -2.1)	3.3 (1.5, 4.5)	0.5 (-0.1, 0.9)	0.7 (-0.3, 2.2)	-0.6 (-0.7, 0.1)
	↳ 2100		-5.8 (-10.2, -4.7)	2.5 (2.3, 15.2)	-0.8 (-0.8, 1.8)	1.4 (-1.7, 4.1)	-1.2 (-2.5, -0.2)
	RCP4.5 in 2050	3/3	-2.7 (-4.4, -0.4)	1.7 (1, 1.9)	1.1 (-0.1, 1.7)	-1.8 (-2.3, 2.1)	0.8 (-0.5, 1.5)
	↳ 2100		-2.8 (-7.8, -2)	2.7 (2.3, 4.7)	1.1 (0.2, 1.2)	-0.7 (-2.6, 1)	1.4 (-1, 1.8)
	Baseline in 2050	3/3	-2.8 (-2.9, -0.2)	1.1 (0.7, 2)	1.1 (0.7, 1.8)	-1.8 (-2.3, -1)	1.5 (-0.5, 2.1)
	↳ 2100		-2.4 (-5, -1)	1.7 (1.4, 2.6)	1.2 (1.2, 1.9)	-2.4 (-2.5, -2)	1.3 (-1, 4.4)
SSP5	**RCP1.9** in 2050	2/4	-1.5 (-3.9, 0.9)	6.7 (6.2, 7.2)	-1.9 (-3.5, -0.4)	3.1 (-0.1, 6.3)	-6.4 (-7.7, -5.1)
	↳ 2100		-0.5 (-4.2, 3.2)	7.6 (7.2, 8)	-3.4 (-6.2, -0.5)	4.7 (0.1, 9.4)	-8.5 (-10.7, -6.2)
	RCP2.6 in 2050	4/4	-3.4 (-6.9, 0.3)	4.8 (3.8, 5.1)	-2.1 (-4, 1)	3.9 (-0.1, 6.7)	-4.4 (-5, 0.2)
	↳ 2100		-4.3 (-8.4, 0.5)	9.1 (7.7, 9.2)	-3.3 (-6.5, -0.5)	3.9 (-0.1, 9.3)	-6.3 (-9.1, -1.4)
	RCP4.5 in 2050	4/4	-2.5 (-3.7, 0.2)	1.7 (0.6, 2.9)	0.6 (-3.3, 1.9)	-0.1 (-1.7, 6)	-1.2 (-2.6, 2.3)
	↳ 2100		-4.1 (-4.6, 0.7)	4.8 (2, 8)	-1 (-5.5, 1)	-0.2 (-1.4, 9.1)	-3 (-5.2, 2.1)
	Baseline in 2050	4/4	-0.6 (-3.8, 0.4)	0.8 (0, 2.1)	1.5 (-0.7, 3.3)	-1.9 (-3.4, 0.5)	-0.1 (-1.5, 2.9)
	↳ 2100		-0.2 (-2.4, 1.8)	1 (0.2, 2.3)	1 (-2, 2.5)	-2.1 (-3.4, 1.1)	-0.4 (-2.4, 2.8)

** Count of models included / Count of models attempted. One model did not provide land data and is excluded from all entries.*

*** One model could reach RCP1.9 with SSP4, but did not provide land data*

Figure SPM.4 Pathways linking socioeconomic development, mitigation responses and land

Future scenarios provide a framework for understanding the implications of mitigation and socioeconomics on land. The Shared Socioeconomic Pathways (SSPs) span a range of different socioeconomic assumptions (Box SPM.1). They are combined with Representative Concentration Pathways (RCPs)[36]which imply different levels of mitigation. The changes in cropland, pasture, bioenergy cropland, forest, and natural land from 2010 are shown. For this figure: Cropland includes all land in food, feed, and fodder crops, as well as other arable land (cultivated area). This category includes 1st generation non-forest bioenergy crops (e.g. corn for ethanol, sugar cane for ethanol, soybeans for biodiesel), but excludes 2nd generation bioenergy crops. Pasture includes categories of pasture land, not only high quality rangeland, and is based on FAO definition of "permanent meadows and pastures". Bioenergy cropland includes land dedicated to 2nd generation energy crops (e.g., switchgrass, miscanthus, fast-growing wood species). Forest includes managed and unmanaged forest. Natural land includes other grassland, savannah, and shrubland. **Panel A:** This panel shows integrated assessment model (IAM)[37] results for SSP1, SSP2 and SSP5 at RCP1.9[38]. For each pathway, the shaded areas show the range across all IAMs; the line indicates the median across models. For RCP1.9, SSP1, SSP2 and SSP5 include results from five, four and two IAMs respectively. **Panel B**: Land use and land cover change are indicated for various SSP-RCP combinations, showing multi-model median and range (min, max). {Box SPM.1, 1.3.2, Cross-Chapter Box 1 in Chapter 1, 2.7.2, Cross-Chapter Box 9 in Chapter 6, 6.1, 6.4.4, 7.4.2, 7.4.4, 7.4.5, 7.4.6, 7.4.7, 7.4.8, 7.5.3, 7.5.6; Cross-Chapter Box 9 in Chapter 6}

D. Action in the near-term

D 1. Actions can be taken in the near-term, based on existing knowledge, to address desertification, land degradation and food security while supporting longer-term responses that enable adaptation and mitigation to climate change. These include actions to build individual and institutional capacity, accelerate knowledge transfer, enhance technology transfer and deployment, enable financial mechanisms, implement early warning systems, undertake risk management and address gaps in implementation and upscaling (*high confidence*). {3.6.1, 3.6.2, 3.7.2, 4.8, 5.3.3, 5.5, 5.6.4, 5.7, 6.2, 6.4, 7.3, 7.4.9, 7.6; Cross-Chapter Box 10 in Chapter 7}

D1.1. Near-term capacity-building, technology transfer and deployment, and enabling financial mechanisms can strengthen adaptation and mitigation in the land sector. Knowledge and technology transfer can help enhance the sustainable use of natural resources for food security under a changing climate (*medium confidence*). Raising awareness, capacity building and education about sustainable land management practices, agricultural extension and advisory

[36] Representative Concentration Pathways (RCPs) are scenarios that include timeseries of emissions and concentrations of the full suite of greenhouse gases (GHGs) and aerosols and chemically active gases, as well as land use/land cover".

[37] Integrated Assessment Models (IAMs) integrate knowledge from two or more domains into a single framework. In this figure, IAMs are used to assess linkages between economic, social and technological development and the evolution of the climate system.

[38] The RCP1.9 pathways assessed in this report have a 66% chance of limiting warming to 1.5C in 2100, but some of these pathways overshoot 1.5C of warming during the 21st century by >0.1C.

services, and expansion of access to agricultural services to producers and land users can effectively address land degradation (*medium confidence*). {3.1, 5.7.4, 7.2, 7.3.4, 7.5.4}

D1.2. Measuring and monitoring land use change including land degradation and desertification is supported by the expanded use of new information and communication technologies (cellphone based applications, cloud-based services, ground sensors, drone imagery), use of climate services, and remotely sensed land and climate information on land resources (*medium confidence*). Early warning systems for extreme weather and climate events are critical for protecting lives and property and enhancing disaster risk reduction and management (*high confidence*). Seasonal forecasts and early warning systems are critical for food security (famine) and biodiversity monitoring including pests and diseases and adaptive climate risk management (*high confidence*). There are high returns on investments in human and institutional capacities. These investments include access to observation and early warning systems, and other services derived from in-situ hydro-meteorological and remote sensing-based monitoring systems and data, field observation, inventory and survey, and expanded use of digital technologies (*high confidence*). {1.2, 3.6.2, 4.2.2, 4.2.4, 5.3.1, 5.3.6, 6.4, 7.3.4, 7.4.3, 7.5.4, 7.5.5, 7.6.4; Cross-Chapter Box 5 in Chapter 3}

D1.3. Framing land management in terms of risk management, specific to land, can play an important role in adaptation through landscape approaches, biological control of outbreaks of pests and diseases, and improving risk sharing and transfer mechanisms (*high confidence*). Providing information on climate-related risk can improve the capacity of land managers and enable timely decision making (*high confidence*). {5.3.2, 5.3.5, 5.6.2, 5.6.3; Cross-Chapter Box 6 in Chapter 5; 5.6.5, 5.7.1, 5.7.2, 7.2.4}

D1.4. Sustainable land management can be improved by increasing the availability and accessibility of data and information relating to the effectiveness, co-benefits and risks of emerging response options and increasing the efficiency of land use (*high confidence*). Some response options (e.g., improved soil carbon management) have been implemented only at small-scale demonstration facilities and knowledge, financial, and institutional gaps and challenges exist with upscaling and the widespread deployment of these options (*medium confidence*). {4.8, 5.5.1, 5.5.2, 5.6.1, 5.6.5, 5.7.5, 6.2, 6.4,}

D 2. Near-term action to address climate change adaptation and mitigation, desertification, land degradation and food security can bring social, ecological, economic and development co-benefits (*high confidence*). Co-benefits can contribute to poverty eradication and more resilient livelihoods for those who are vulnerable (*high confidence*). {3.4.2, 5.7, 7.5}

D2.1. Near-term actions to promote sustainable land management will help reduce land and food-related vulnerabilities, and can create more resilient livelihoods, reduce land degradation and desertification, and loss of biodiversity (*high confidence*). There are synergies between sustainable land management, poverty eradication efforts, access to market, non-market mechanisms and the elimination of low-productivity practices. Maximising these synergies can lead to adaptation, mitigation, and development co-benefits through preserving ecosystem functions and services (*medium confidence*). {3.4.2, 3.6.3, Table 4.2, 4.7, 4.9, 4.10, 5.6, 5.7, 7.3, 7.4, 7.5, 7.6; Cross-Chapter Box 12 in Chapter 7}

D2.2. Investments in land restoration can result in global benefits and in drylands can have benefit-cost ratios of between three and six in terms of the estimated economic value of restored ecosystem services (*medium confidence*). Many sustainable land management technologies and practices are profitable within three to 10 years (*medium confidence*). While they can require upfront investment, actions to ensure sustainable land management can improve crop yields and the economic value of pasture. Land restoration and rehabilitation measures improve livelihood systems and provide both short-term positive economic returns and longer-term benefits in terms of climate change adaptation and mitigation, biodiversity and enhanced ecosystem functions and services (*high confidence*). {3.6.1, 3.6.3, 4.8.1, 7.2.4, 7.2.3, 7.3.1, 7.4.6, Cross-Chapter Box 10 in Chapter 7}

D2.3. Upfront investments in sustainable land management practices and technologies can range from about USD 20 ha^{-1} to USD 5000 ha^{-1}, with a median estimated to be around USD 500 ha^{-1}. Government support and improved access to credit can help overcome barriers to adoption, especially those faced by poor smallholder farmers (*high confidence*). Near-term change to balanced diets (see B6.2) can reduce the pressure on land and provide significant health co-benefits through improving nutrition (*medium confidence)*. {3.6.3, 4.8, 5.3, 5.5, 5.6, 5.7, 6.4, 7.4.7, 7.5.5; Cross-Chapter Box 9 in Chapter 6}

D 3. Rapid reductions in anthropogenic GHG emissions across all sectors following ambitious mitigation pathways reduce negative impacts of climate change on land ecosystems and food systems (*medium confidence*). Delaying climate mitigation and adaptation responses across sectors would lead to increasingly negative impacts on land and reduce the prospect of sustainable development (*medium confidence*). {Box SPM.1, Figure SPM.2, 2.5, 2.7, 5.2, 6.2, 6.4, 7.2, 7.3.1, 7.4.7, 7.4.8, 7.5.6; Cross-Chapter Box 9 in Chapter 6, Cross-Chapter Box 10 in Chapter 7}

D3.1. Delayed action across sectors leads to an increasing need for widespread deployment of land-based adaptation and mitigation options and can result in a decreasing potential for the array of these options in most regions of the world and limit their current and future effectiveness (*high confidence*). Acting now may avert or reduce risks and losses, and generate benefits to society (*medium confidence*). Prompt action on climate mitigation and adaptation aligned with sustainable land management and sustainable development depending on the region could reduce the risk to millions of people from climate extremes, desertification, land degradation and food and livelihood insecurity (*high confidence*). {1.3.5, 3.4.2, 3.5.2, 4.1.6, 4.7.1, 4.7.2, 5.2.3, 5.3.1, 6.3, 6.5, 7.3.1}

D3.2. In future scenarios, deferral of GHG emissions reductions implies trade-offs leading to significantly higher costs and risks associated with rising temperatures (*medium confidence*). The potential for some response options, such as increasing soil organic carbon, decreases as climate change intensifies, as soils have reduced capacity to act as sinks for carbon sequestration at higher temperatures (*high confidence*). Delays in avoiding or reducing land degradation and promoting positive ecosystem restoration risk long-term impacts including rapid declines in productivity of agriculture and rangelands, permafrost degradation and difficulties in peatland rewetting (*medium confidence*). {1.3.1, 3.6.2, 4.8, 4.9, 4.9.1, 5.5.2, 6.3, 6.4, 7.2, 7.3; Cross-Chapter Box 10 in Chapter 7}

D3.3. Deferral of GHG emissions reductions from all sectors implies trade-offs including irreversible loss in land ecosystem functions and services required for food, health, habitable settlements and production, leading to increasingly significant economic impacts on many countries in many regions of the world (*high confidence*). Delaying action as is assumed in high emissions scenarios could result in some irreversible impacts on some ecosystems, which in the longer-term has the potential to lead to substantial additional GHG emissions from ecosystems that would accelerate global warming (*medium confidence*). {1.3.1, 2.5.3, 2.7, 3.6.2, 4.9, 4.10.1, 5.4.2.4, 6.3, 6.4, 7.2, 7.3; Cross-Chapter Box 9 in Chapter 6, Cross-Chapter Box 10 in Chapter 7}

Chapter 1: Framing and Context

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Executive summary

Land, including its water bodies, provides the basis for human livelihoods and well-being through primary productivity, the supply of food, freshwater, and multiple other ecosystem services (*high confidence*). Neither our individual or societal identities, nor the World's economy would exist without the multiple resources, services and livelihood systems provided by land ecosystems and biodiversity. The annual value of the World's total terrestrial ecosystem services has been estimated at 75–85 trillion USD in 2011 (based on USD 2007 values) (*low confidence*). This substantially exceeds the annual World GDP (*high confidence*). Land and its biodiversity also represent essential, intangible benefits to humans, such as cognitive and spiritual enrichment, sense of belonging and aesthetic and recreational values. Valuing ecosystem services with monetary methods often overlooks these intangible services that shape societies, cultures and quality of life and the intrinsic value of biodiversity. The Earth's land area is finite. Using land resources sustainably is fundamental for human well-being (*high confidence*). {1.1.1}

The current geographic spread of the use of land, the large appropriation of multiple ecosystem services and the loss of biodiversity are unprecedented in human history (*high confidence*). By 2015, about three-quarters of the global ice-free land surface was affected by human use. Humans appropriate one quarter to one third of global terrestrial potential net primary production (*high confidence*). Croplands cover 12–14% of the global ice-free surface. Since 1961, the supply of global per capita food calories increased by about one third, with the consumption of vegetable oils and meat more than doubling. At the same time, the use of inorganic nitrogen fertiliser increased by nearly 9-fold, and the use of irrigation water roughly doubled (*high confidence*). Human use, at varying intensities, affects about 60–85% of forests and 70–90% of other natural ecosystems (e.g., savannahs, natural grasslands) (*high confidence*). Land use caused global biodiversity to decrease by around 11–14% (*medium confidence*). {1.1.2}

Warming over land has occurred at a faster rate than the global mean and this has had observable impacts on the land system (*high confidence*). The average temperature over land for the period 1999–2018 was 1.41°C higher than for the period 1881–1900, and 0.54°C larger than the equivalent global mean temperature change. These warmer temperatures (with changing precipitation patterns) have altered the start and end of growing seasons, contributed to regional crop yield reductions, reduced freshwater availability, and put biodiversity under further stress and increased tree mortality (*high confidence*). Increasing levels of atmospheric CO_2, have contributed to observed increases in plant growth as well as to increases in woody plant cover in grasslands and savannahs (*medium confidence*). {1.1.2}

Urgent action to stop and reverse the over-exploitation of land resources would buffer the negative impacts of multiple pressures, including climate change, on ecosystems and society (*high confidence*). Socio-economic drivers of land use change such as technological development, population growth and increasing per capita demand for multiple ecosystem services are projected to continue into the future (*high confidence*). These and other drivers can amplify existing environmental and societal challenges, such as the conversion of natural ecosystems into managed land, rapid urbanisation, pollution from the intensification of land management and equitable access to land resources (*high confidence*). Climate change will add to these challenges through direct, negative impacts on ecosystems and the services they provide (*high confidence*). Acting immediately and simultaneously on these multiple drivers would enhance food, fibre and water security, alleviate desertification, and reverse land degradation, without compromising the non-material or regulating benefits from land (*high confidence*). {1.1.2, 1.2.1, 1.3.2-1.3.6, Cross-Chapter Box 1, Chapter 1}

Rapid reductions in anthropogenic greenhouse gas emissions that restrict warming to *"well-below"* 2°C would greatly reduce the negative impacts of climate change on land ecosystems (*high confidence*). In the absence of rapid emissions reductions, reliance on large-scale, land-based, climate change mitigation is projected to increase, which would aggravate existing pressures on land (*high confidence*). Climate change mitigation efforts that require large land areas (e.g., bioenergy and afforestation/reforestation) are projected to compete with existing uses of land (*high confidence*). The competition for land could increase food prices and lead to further intensification (e.g., fertiliser and water use) with implications for water and air pollution, and the further loss of biodiversity (*medium confidence*). Such consequences would jeopardise societies' capacity to achieve many sustainable development goals that depend on land (*high confidence*). {1.3.1, Cross-Chapter Box 2 in Chapter 1}

Nonetheless, there are many land-related climate change mitigation options that do not increase the competition for land (*high confidence*). Many of these options have co-benefits for climate change adaptation (*medium confidence*). Land use contributes about one quarter of global greenhouse gas emissions, notably CO_2 emissions from deforestation, CH_4 emissions from rice and ruminant livestock and N_2O emissions from fertiliser use (*high confidence*). Land ecosystems also take up large amounts of carbon (*high confidence*). Many land management options exist to both reduce the magnitude of emissions and enhance carbon uptake. These options enhance crop productivity, soil nutrient status, microclimate or biodiversity, and thus, support adaptation to climate change (*high confidence*). In addition, changes in consumer behaviour, such as reducing the over-consumption of food and energy would benefit the reduction of GHG emissions from land (*high confidence*). The barriers to the implementation of mitigation and adaptation options include skills deficit, financial and institutional barriers, absence of incentives, access to relevant technologies, consumer awareness and the limited spatial scale at which the success of these practices and methods have been demonstrated. {1.2.1, 1.3.2, 1.3.3, 1.3.4, 1.3.5, 1.3.6}

Sustainable food supply and food consumption, based on nutritionally balanced and diverse diets, would enhance food security under climate and socio-economic changes (*high confidence*). Improving food access, utilisation, quality and safety to enhance nutrition, and promoting globally equitable diets compatible with lower emissions have demonstrable positive impacts on land use and food security (*high confidence*). Food security is also negatively affected by food loss and waste (estimated as more than 30% of harvested materials) (*high confidence*). Barriers to improved food security include economic drivers (prices, availability and stability of supply) and traditional, social and cultural norms around food eating practices. Climate change is expected to increase variability in food production and prices globally (*high confidence*), but the trade in food commodities can buffer these effects. Trade can provide embodied flows of water, land and nutrients (*medium confidence*). Food trade can also have negative environmental impacts by displacing the effects of overconsumption (*medium confidence*). Future food systems and trade patterns will be shaped as much by policies as by economics (*medium confidence*). {1.2.1, 1.3.3}

A gender inclusive approach offers opportunities to enhance the sustainable management of land (*medium confidence*). Women play a significant role in agriculture and rural economies globally. In many World regions, laws, cultural restrictions, patriarchy and social structures such as discriminatory customary laws and norms reduce women's capacity in supporting the sustainable use of land resources (*medium confidence*). Therefore, acknowledging women's land rights and bringing women's land management knowledge into land-related decision-making would support the alleviation of land degradation, and facilitate the take-up of integrated adaptation and mitigation measures (*medium confidence*). {1.4.1, 1.4.2}

Regional and country specific contexts affect the capacity to respond to climate change and its impacts, through adaptation and mitigation (*high confidence*). There is large variability in the availability and use of land resources between regions, countries and land-management systems. In addition, differences in socio-economic conditions, such as wealth, degree of industrialisation, institutions and governance, affect the capacity to respond to climate change, food insecurity, land degradation and desertification. The capacity to respond is also strongly affected by local land ownership. Hence, climate change will affect regions and communities differently (*high confidence*). {1.3, 1.4}

Cross-scale, cross-sectoral and inclusive governance can enable coordinated policy that supports effective adaptation and mitigation (*high confidence*). There is a lack of coordination across governance levels, for example, local, national, transboundary and international, in addressing climate change and sustainable land management challenges. Policy design and formulation is often strongly sectoral, which poses further barriers when integrating international decisions into relevant (sub)national policies. A portfolio of policy instruments that are inclusive of the diversity of governance actors would enable responses to complex land and climate challenges (*high confidence*). Inclusive governance that considers women's and indigenous people's rights to access and use land enhances the equitable sharing of land resources, fosters food security and increases the existing knowledge about land use, which can increase opportunities for adaptation and mitigation (*medium confidence*). {1.3.5, 1.4.1, 1.4.2, 1.4.3}

Scenarios and models are important tools to explore the trade-offs and co-benefits of land management decisions under uncertain futures (*high confidence*). Participatory, co-creation processes with stakeholders can facilitate the use of scenarios in designing future sustainable development strategies (*medium confidence*). In addition to qualitative approaches, models are critical in quantifying scenarios, but uncertainties in models arise from, for example, differences in baseline datasets, land cover classes and modelling paradigms (*medium confidence*). Current scenario approaches are limited in quantifying time-dependent, policy and management decisions that can lead from today to desirable futures or visions. Advances in scenario analysis and modelling are needed to better account for full environmental costs and non-monetary values as part of human decision-making processes. {1.2.2, Cross Chapter Box 1 in Chapter 1}

1.1 Introduction and scope of the report

1.1.1 Objectives and scope of the assessment

Land, including its water bodies, provides the basis for our livelihoods through basic processes such as net primary production that fundamentally sustain the supply of food, bioenergy and freshwater, and the delivery of multiple other ecosystem services and biodiversity (Hoekstra and Wiedmann 2014; Mace et al. 2012; Newbold et al. 2015; Runting et al. 2017; Isbell et al. 2017)(see Cross-Chapter Box 8: Ecosystem Services, Chapter 6). The annual value of the world's total terrestrial ecosystem services has been estimated to be about USD 75–85 trillion (in 2011 based on USD 2007 values)(Costanza et al. 2014). This equates approximately to the world's average GDP over the last 5 years (IMF 2018). Land also supports non-material ecosystem services such as cognitive and spiritual enrichment and aesthetic values (Hernández-Morcillo et al. 2013; Fish et al. 2016), intangible services that shape societies, cultures and human well-being. Exposure of people living in cities to (semi-)natural environments has been found to decrease mortality, cardiovascular disease and depression (Rook 2013; Terraube et al. 2017). Non-material and regulating ecosystem services have been found to decline globally and rapidly, often at the expense of increasing material services (Fischer et al. 2018; IPBES 2018a). Climate change will exacerbate diminishing land and freshwater resources, increase biodiversity loss, and will intensify societal vulnerabilities, especially in regions where economies are highly dependent on natural resources. Enhancing food security and reducing malnutrition, whilst also halting and reversing desertification and land degradation, are fundamental societal challenges that are increasingly aggravated by the need to both adapt to and mitigate climate change impacts without compromising the non-material benefits of land (Kongsager et al. 2016; FAO et al. 2018).

Annual emissions of greenhouse gases (GHGs) and other climate forcers continue to increase unabatedly. *Confidence* is *very high* that the window of opportunity, the period when significant change can be made, for limiting climate change within tolerable boundaries is rapidly narrowing (Schaeffer et al. 2015; Bertram et al. 2015; Riahi et al. 2015; Millar et al. 2017; Rogelj et al. 2018a). The Paris Agreement formulates the goal of limiting global warming this century well below 2^{o}C above pre-industrial levels, for which rapid actions are required across the energy, transport, infrastructure and agricultural sectors, while factoring in the need for these sectors to accommodate a growing human population (Wynes and Nicholas 2017; Le Quere et al. 2018). Conversion of natural land, and land management, are significant net contributors to GHG emissions and climate change, but land ecosystems are also a GHG sink (Smith et al. 2014; Tubiello et al. 2015; Le Quere et al. 2018; Ciais et al. 2013a). It is not surprising, therefore, that land plays a prominent role in many of the Nationally Determined Contributions (NDCs) of the parties to the Paris Agreement (Rogelj et al. 2018a,b; Grassi et al. 2017; Forsell et al. 2016), and land-measures will be part of the NDC review by 2023.

A range of different climate change mitigation and adaptation options on land exist, which differ in terms of their environmental and societal implications (Meyfroidt 2018; Bonsch et al. 2016; Crist et al. 2017; Humpenoder et al. 2014; Harvey and Pilgrim 2011; Mouratiadou et al. 2016; Zhang et al. 2015; Sanz-Sanchez et al. 2017; Pereira et al. 2010; Griscom et al. 2017; Rogelj et al. 2018a)(see Chapters 4-6). The Special Report on climate change, desertification, land degradation, sustainable land management, food security, and GHG fluxes in terrestrial ecosystems (SRCCL) synthesises the current state of scientific knowledge on the issues specified in the report's title (see Figure 1.1, Figure 1.2). This knowledge is assessed in the context of the Paris Agreement, but many of the SRCCL issues concern other international conventions such as the United Nations Convention on Biodiversity (UNCBD), the UN Convention to Combat Desertification (UNCCD), the UN Sendai Framework for Disaster Risk Reduction (UNISDR) and the UN Agenda 2030 and its Sustainable Development Goals (SDGs). The SRCCL is the first report in which land is the central focus since the IPCC Special Report on land use, land-use change and forestry (Watson et al. 2000)(see Box 1.1). The main objectives of the SRCCL are to:

1) Assess the current state of the scientific knowledge on the impacts of socio-economic drivers and their interactions with climate change on land, including degradation, desertification and food security;
2) Evaluate the feasibility of different land-based response options to GHG mitigation, and assess the potential synergies and trade-offs with ecosystem services and sustainable development;
3) Examine adaptation options under a changing climate to tackle land degradation and desertification and to build resilient food systems, as well as evaluating the synergies and trade-offs between mitigation and adaptation; and
4) Delineate the policy, governance and other enabling conditions to support climate mitigation, land ecosystem resilience and food security in the context of risks, uncertainties and remaining knowledge gaps.

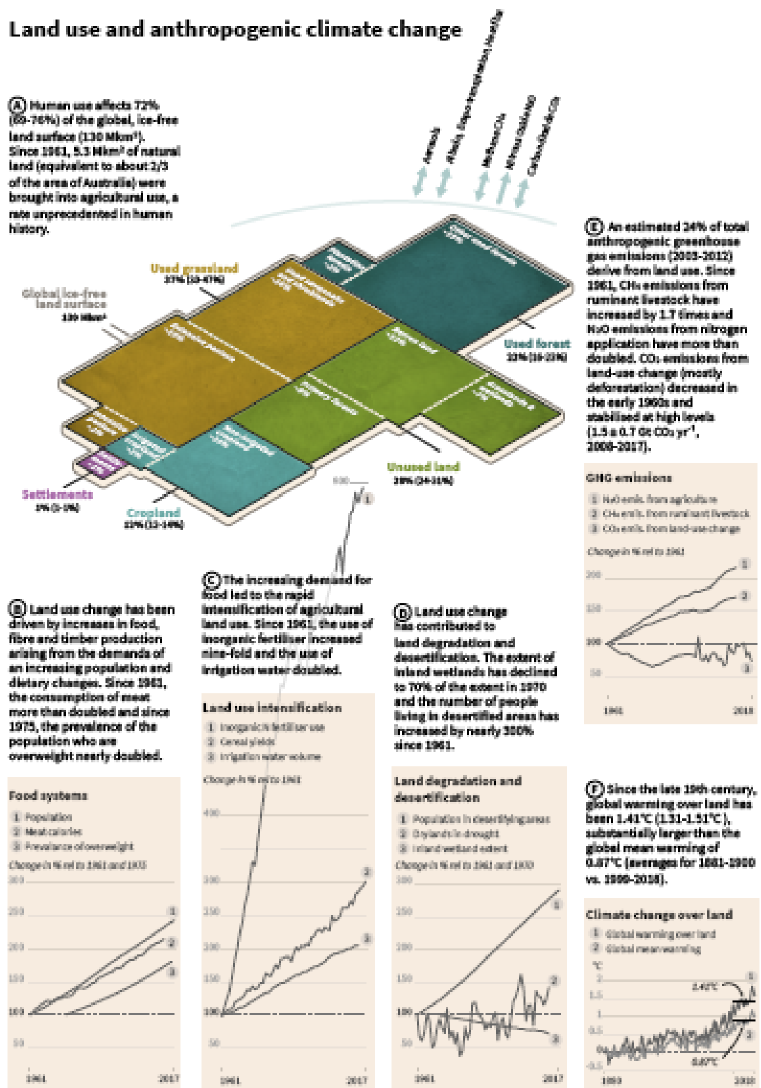

Figure 1.1 A representation of the principal land challenges and land-climate system processes covered in this assessment report. A. The tiles show the current extent (in about 2015) of the human use of the land surface, aggregated into five broad land use and land cover categories with uncertainty ranges. Colour shading indicates different intensities of human use (Table 1.1**)**. B. Agricultural areas have increased to supply the increasing demand for food arising from population growth, income growth and increasing consumption of animal-sourced products. The proportion of the global population that is overweight (body mass index > 25 kg/m^2) has increased markedly (section 5.1.2). Population density *(Source: United Nations, Department of Economic and Social Affairs 2017)*. Meat calories supplied (*Source:* **FAOSTAT 2018** Prevalence of people overweight (*Source: Abarca-Gómez et al. 2017)*(5.1.2). C. Increasing food production has led to rapid land use intensification, including increases in the use of nitrogen fertiliser and irrigation water that have supported the growth in cereal yields (section 1.1). Change in cereal yield and irrigation water use (Source: **FAOSTAT 2018***)*; Change in total inorganic nitrogen fertiliser consumption (*Source: International Fertiliser Industry Association, https://www.ifastat.org/databases*). Note that the very large percentage change in fertiliser use reflects the very low use in 1961. The increase relates to both increasing fertiliser input per area as well as the expansion of fertilised cropland and grassland. D. Land use change has led to substantial losses in the extent of inland wetlands (section 4.2.1, 4.6.1). Dryland areas are under increasing pressures both from the increasing number of people living in these areas and from the increase in droughts (section 3.1.1). The inland wetland extent trends (WET) index was developed by aggregating data from 2130 time series that report changes in local wetland area over time (Dixon et al. 2016; Darrah et al. 2019). Dryland areas were defined using TerraClimate precipitation and potential evapotranspiration (1980-2015) (Abatzoglou et al. 2018) to identify areas

where the Aridity Index is below 0.65. Areas undergoing human caused desertification, after accounting for precipitation variability and CO_2 fertilisation, are identified in (Le et al. 2016). Population data for these areas were extracted from the gridded historical population database HYDE3.2 (Goldewijk et al. 2017). The 12-month accumulation Global Precipitation Climatology Centre Drought Index (Ziese et al. 2014) was extracted for drylands. The area in drought was calculated for each month (Drought Index below -1), and the mean over the year was used to calculate the percentage of drylands in drought that year. E. Land use change and intensification have contributed to CH_4 emissions from ruminant livestock, agricultural N_2O emissions and CO_2 emissions from net deforestation {2.3}. Sources: N_2O from agricultural activities and CH_4 from enteric fermentation: Edgar database (*http://edgar.jrc.ec.europa.eu/overview.php?v=42FT2012*) from 1970. From 1970 back to 1961, CH_4 and N_2O were extrapolated using a regression with time, taken for the years 1970-1979 from Edgar. Net-land use change emissions of CO_2 are from the annual Global Carbon Budget, using the mean of two bookkeeping models (Le Quéré et al. 2018). Chapter 2 (Section 2.2, 2.3) and Chapter 5 (Section 5.4) provides a discussion of uncertainties and other emissions estimates. The various exchanges between the land surface and the atmosphere, including the emission and uptake of greenhouse gases, exchanges related to the land-surface energy balance and aerosols are indicated by arrows (section 2.1, 2.3, 2.4). Warming over land is more rapid than the global mean temperature change (section 2.2). Future climate change will exacerbate the already considerable challenges faced by land systems. The warming curves are averages of four historical estimates, and described in Section 2.1.

The SRCCL identifies and assesses land-related challenges and response-options in an integrative way, aiming to be policy relevant across sectors. Chapter 1 provides a synopsis of the main issues addressed in this report, which are explored in more detail in Chapters 2–7. Chapter 1 also introduces important concepts and definitions and highlights discrepancies with previous reports that arise from different objectives (a full set of definitions is provided in the Glossary). Chapter 2 focuses on the natural system dynamics, assessing recent progress towards understanding the impacts of climate change on land, and the feedbacks arising from altered biogeochemical and biophysical exchange fluxes (Figure 1.2).

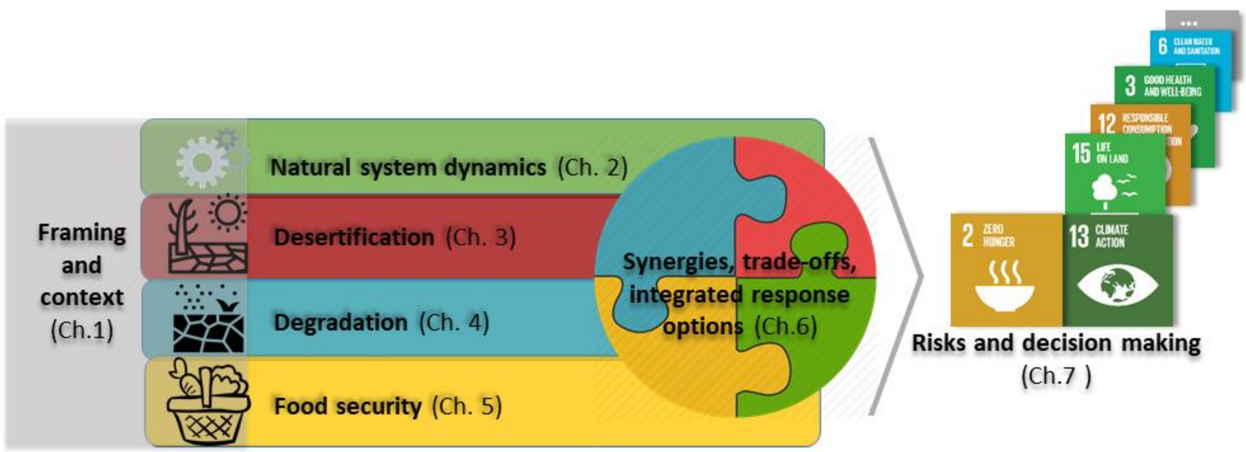

Figure 1.2 Overview over the SRCCL

Chapter 3 examines how the world's dryland populations are uniquely vulnerable to desertification and climate change, but also have significant knowledge in adapting to climate variability and addressing desertification. Chapter 4 assesses the urgency of tackling land degradation across all land ecosystems. Despite accelerating trends of land degradation, reversing these trends is attainable through restoration efforts and proper implementation of sustainable land management (SLM), which is expected to improve resilience to climate change, mitigate climate change, and ensure food security for generations to come. Food security is the focus of Chapter 5, with an assessment of the risks and opportunities that climate change presents to food systems, considering how mitigation and adaptation can contribute to both human and planetary health.

Chapters 6 focuses on the response options within the land system that deal with trade-offs and increase benefits in an integrated way in support of the SDGs. Chapter 7 highlights these aspects further, by assessing the opportunities, decision making and policy responses to risks in the climate-land-human system.

Box 1.1 Land in previous IPCC and other relevant reports

Previous IPCC reports have made reference to land and its role in the climate system. Threats to agriculture forestry and other ecosystems, but also the role of land and forest management in climate change, have been documented since the IPCC Second Assessment Report, especially so in the Special report on land use, land-use change and forestry (Watson et al. 2000). The IPCC Special Report on Extreme events (SREX) discussed sustainable land management, including land use planning, and ecosystem management and restoration among the potential low-regret measures that provide benefits under current climate and a range of future, climate change scenarios. Low-regret measures are defined in the report as those with the potential to offer benefits now and lay the foundation for tackling future, projected change. Compared to previous IPCC reports, the SRCCL offers a more integrated analysis of the land system as it embraces multiple direct and indirect drivers of natural resource management (related to food, water and energy securities), which have not previously been addressed to a similar depth (Field et al. 2014a; Edenhofer et al. 2014).

The recent IPCC Special Report on Global Warming of 1.5°C (SR15) targeted specifically the Paris Agreement, without exploring the possibility of future global warming trajectories above 2°C (IPCC 2018). Limiting global warming to 1.5°C compared to 2°C is projected to lower the impacts on terrestrial, freshwater and coastal ecosystems and to retain more of their services for people. In many scenarios proposed in this report, large-scale land use features as a mitigation measure. In the reports of the Food and Agriculture Organisation (FAO), land degradation is discussed in relation to ecosystem goods and services, principally from a food security perspective (FAO and ITPS 2015). The UNCCD report (2014) discusses land degradation through the prism of desertification. It devotes due attention to how land management can contribute to reversing the negative impacts of desertification and land degradation. The IPBES assessments (2018a,b,c,d,e) focuses on biodiversity drivers, including a focus on land degradation and desertification, with poverty as a limiting factor. The reports draw attention to a world in peril in which resource scarcity conspires with drivers of biophysical and social vulnerability to derail the attainment of sustainable development goals. As discussed in chapter 4 of the SRCCL, different definitions of degradation have been applied in the IPBES degradation assessment (IPBES 2018b), which potentially can lead to different conclusions for restoration and ecosystem management.

The SRCCL complements and adds to previous assessments, whilst keeping the IPCC-specific “climate perspective”. It includes a focussed assessment of risks arising from maladaptation and land-based mitigation (i.e. not only restricted to direct risks from climate change impacts) and the co-benefits and trade-offs with sustainable development objectives. As the SRCCL cuts across different policy sectors it provides the opportunity to address a number of challenges in an integrative way at the same time, and it progresses beyond other IPCC reports in having a much more comprehensive perspective on land.

1.1.2 Status and dynamics of the (global) land system

1.1.2.1 Land ecosystems and climate change

Land ecosystems play a key role in the climate system, due to their large carbon pools and carbon exchange fluxes with the atmosphere (Ciais et al. 2013b). Land use, the total of arrangements, activities and inputs applied to a parcel of land (such as agriculture, grazing, timber extraction, conservation or city dwelling; see glossary), and land management (sum of land-use practices that take place within broader land-use categories, see glossary) considerably alter terrestrial ecosystems and play a key role in the global climate system. An estimated one quarter of total anthropogenic GHG emissions arise mainly from deforestation, ruminant livestock and fertiliser application (Smith et al. 2014; Tubiello et al. 2015; Le Quere et al. 2018; Ciais et al. 2013a), and especially methane and nitrous oxide emissions from agriculture have been rapidly increasing over the last decades (Hoesly et al. 2018; Tian et al. 2019)(see Figure 1.1, see Section 2.3.2, 2.3.3).

Globally, land also serves as a large carbon dioxide sink, which was estimated for the period 2008–2017 to be nearly 30% of total anthropogenic emissions (Le Quere et al. 2015; Canadell and Schulze 2014; Ciais et al. 2013a; Zhu et al. 2016)(see Section 2.3.1). This sink has been attributed to increasing atmospheric CO_2 concentration, a prolonged growing season in cool environments, or forest regrowth (Le Quéré et al. 2013; Pugh et al. 2019; Le Quéré et al. 2018; Ciais et al. 2013a; Zhu et al. 2016). Whether or not this sink will persist into the future is one of the largest uncertainties in carbon cycle and climate modelling (Ciais et al. 2013a; Bloom et al. 2016; Friend et al. 2014; Le Quere et al. 2018). In addition, changes in vegetation cover caused by land use (such as conversion of forest to cropland or grassland, and vice versa) can result in regional cooling or warming through altered energy and momentum transfer between ecosystems and the atmosphere. Regional impacts can be substantial, but whether the effect leads to warming or cooling depends on the local context (Lee et al. 2011; Zhang et al. 2014; Alkama and Cescatti 2016; see Section 2.6). Due to the current magnitude of GHG emissions and carbon dioxide removal in land ecosystems, there is *high confidence* that greenhouse-gas reduction measures in agriculture, livestock management and forestry would have substantial climate change mitigation potential with co-benefits for biodiversity and ecosystem services (Smith and Gregory 2013; Smith et al. 2014; Griscom et al. 2017; see Section 2.6, Section 6.3).

The mean temperature increase over land has been substantially larger than the global mean (land and ocean), averaging 1.41°C vs. 0.87°C for the years 1999–2018 compared with 1881–1900 (see Section 2.2). Climate change affects land ecosystems in various ways (see Section 7.2). Growing seasons and natural biome boundaries shift in response to warming or changes in precipitation (Gonzalez et al. 2010; Wärlind et al. 2014; Davies-Barnard et al. 2015; Nakamura et al. 2017). Atmospheric CO_2 increases have been attributed to underlie, at least partially, observed woody plant cover increase in grasslands and savannahs (Donohue et al. 2013). Climate change-induced shifts in habitats, together with warmer temperatures, causes pressure on plants and animals (Pimm et al. 2014; Urban et al. 2016). National cereal crop losses of nearly 10% have been estimated for the period 1964–2007 as a consequence of heat and drought weather extremes (Deryng et al. 2014; Lesk et al. 2016). Climate change is expected to reduce yields in areas that are already under heat and water stress (Schlenker and Lobell 2010; Lobell et al. 2011,2012; Challinor et al. 2014; see Section 5.2.2). At the same time, warmer temperatures can increase productivity in cooler regions (Moore and Lobell 2015) and might open opportunities for crop area expansion, but any overall benefits might be counterbalanced by reduced suitability in warmer regions (Pugh et al. 2016; Di Paola et al. 2018). Increasing atmospheric CO_2 is expected to increase productivity and water use efficiency in crops and in forests (Muller et al. 2015; Nakamura et al. 2017; Kimball 2016). The increasing number of extreme weather events linked to climate change is also expected to result in forest losses; heat waves and droughts foster wildfires (Seidl et al. 2017; Fasullo et al. 2018; see Cross-Chapter Box 3: Fire and Climate Change, Chapter 2). Episodes of observed enhanced tree mortality across many world regions have been attributed to heat and drought stress (Allen et al. 2010; Anderegg et al. 2012), whilst weather extremes also impact local infrastructure and hence transportation and trade in land-related goods (Schweikert et al. 2014; Chappin and van der Lei 2014). Thus, adaptation is a key challenge to reduce adverse impacts on land systems (see Section 1.3.6).

1.1.2.2 Current patterns of land use and land cover

Around three quarters of the global ice-free land, and most of the highly-productive land area, are by now under some form of land use (Erb et al. 2016a; Luyssaert et al. 2014; Venter et al. 2016; see Table 1.1). One third of used land is associated with changed land cover. Grazing land is the single largest land-use category, followed by used forestland and cropland. The total land area used to raise livestock is notable: it includes all grazing land and an estimated additional one fifth of cropland for feed production (Foley et al. 2011). Globally, 60–85% of the total forested area is used, at different levels of intensity, but information on management practices globally are scarce (Erb et al. 2016a). Large areas of unused (primary) forests remain only in the tropics and northern boreal zones (Luyssaert et al. 2014; Birdsey and Pan 2015; Morales-Hidalgo et al. 2015; Potapov et al. 2017; Erb et al. 2017), while 73–89% of other, non-forested natural ecosystems (natural grasslands, savannas, etc.) are used. Large uncertainties relate to the extent of forest (32.0–42.5 million km²) and grazing land (39–62 million km²), due to discrepancies in definitions and observation methods (Luyssaert et al. 2014; Erb et al. 2017; Putz and Redford 2010; Schepaschenko et al. 2015; Birdsey and Pan 2015; FAO 2015a; Chazdon et al. 2016a; FAO 2018a). Infrastructure areas (including settlements, transportation and mining), while being almost negligible in terms of extent, represent particularly pervasive land-use activities, with far-reaching ecological, social and economic implications (Cherlet et al. 2018; Laurance et al. 2014).

The large imprint of humans on the land surface has led to the definition of anthromes, i.e. large-scale ecological patterns created by the sustained interactions between social and ecological drivers. The dynamics of these ‘anthropogenic biomes’ are key for land-use impacts as well as for the design of integrated response options (Ellis and Ramankutty 2008; Ellis et al. 2010; Cherlet et al. 2018; Ellis et al. 2010, see Chapter 6).

The intensity of land use varies hugely within and among different land use types and regions. Averaged globally, around 10% of the ice-free land surface was estimated to be intensively managed (such as tree plantations, high livestock density grazing, large agricultural inputs), two thirds moderately and the remainder at low intensities (Erb et al. 2016a). Practically all cropland is fertilised, with large regional variations. Irrigation is responsible for 70% of ground- or surface-water withdrawals by humans (Wisser et al. 2008; Chaturvedi et al. 2015; Siebert et al. 2015; FAOSTAT 2018). Humans appropriate one quarter to one third of the total potential net primary production, i.e. the NPP that would prevail in the absence of land use (estimated at about 60 GtC yr^{-1}; Bajželj et al. 2014; Haberl et al. 2014), about equally through biomass harvest and changes in NPP due to land management. The current total of agricultural (cropland and grazing) biomass harvest is estimated at about 6 GtC yr^{-1}, around 50–60% of this is consumed by livestock. Forestry harvest for timber and wood fuel amounts to about 1 GtC yr^{-1} (Alexander et al. 2017; Bodirsky and Müller 2014; Lassaletta et al. 2014, 2016; Mottet et al. 2017; Haberl et al. 2014; Smith et al. 2014; Bais et al. 2015; Bajželj et al. 2014)(see Cross-Chapter Box 7: Bioenergy and BECCS, Chapter 6).

Table 1.1 Extent of global land use and management around the year 2015

	Best guess	Range		Range	Type	Ref.
	[million km²]		[% of total]			
Total	130.4		100%			
USED LAND	92.6	90.0-99.3	71%	69-76%		
Infrastructure (Settlements, mining, etc.)	**1.4**	**1.2-1.9**	**1%**		**LCC**	**1,2,3,4,5,6**

Cropland	**15.9**	**15.9-18.8**	**12%**	**12-14%**		**1,7**
irrigated cropland	3.1		2%		LCC	8
non-irrigated cropland	12.8	12.8-15.7	10%		LCC	8
Grazing land	**48.0**	**38.8-61.9**	**37%**	**30-47%**		
Permanent pastures	**27.1**	**22.8-32.8**	**21%**	**17-25%**		**5,7,8**
Intensive permanent pastures*	2.6		2%		LCC	8,9
Extensive perm. pastures, on potential forest sites**	8.7		7%		LCC	9
Extensive perm. pastures, on natural grasslands**	15.8	11.5-21.56	12%	9-16%	LM	
Non-forested, used land, multiple uses[§]	**20.1**	**6.1-39.1**	**16%**	**5-30%**	**LM**	
Used forests[#]	**28.1**	**20.3-30.5**	**22%**	**16-23%**		**10,11,12**
Planted forests	2.9		2%		LCC	12
Managed for timber and other uses	25.2	17.4-27.6	19%	13-21%	LM	12
UNUSED LAND	37.0	31.1-40.4	28%	24-31%		5,11,13
Unused, unforested ecosystems, including grasslands and wetlands	9.4	5.9-10.4	7%	5-8%		1,13
Unused forests (intact or primary forests)	12.0	11.7-12.0	9%			11,12
Other land (barren wilderness, rocks, etc.)	15.6	13.5-18.0	12%	10-14%		4,5,13,14
Land-cover conversions (sum of LCC)	31.5	31.3-34.9	24%	24-27%		
Land-use occurring within natural land-cover types (sum of LM)	61.1	55.1-68.0	47%	42-52%		

*>100 animals/km^2

**<100 animals/km^2, residual category within permanent pastures

[§] Calculated as residual category. Contains land not classified as forests or cropland, such as savanna and tundra used as rangelands, with extensive uses like seasonal, rough grazing, hunting, fuelwood collection outside forests, wild products harvesting, etc.

[#] used forest calculated as total forest minus unused forests

Note: This table is based on data and approaches described in Lambin and Meyfroidt (2011,2014); Luyssaert et al. (2014); Erb et al. (2016a), and references below. The target year for data is 2015, but proportions of some subcategories are from 2000 (the year with still most reconciled datasets available) and their relative extent was applied to some broad land use categories for 2015. Sources: Settlements (1): (Luyssaert et al. 2014); (2) (Lambin and Meyfroidt 2014); (3) Global Human Settlements dataset, https://ghsl.jrc.ec.europa.eu/. Total infrastrucure including transportation (4) (Erb et al. 2007); (5) (Stadler et al. 2018); mining (6) (Cherlet et al. 2018) ; (7) (FAOSTAT 2018) ; (8) proportions from (Erb et al. 2016a); (9) (Ramankutty et al. 2008) extrapolated from 2000 to 2010 trend for permanent pastures from (7); (9) (Erb et al. 2017); (10) (Schepaschenko et al. 2015); (11) (Potapov et al. 2017); (12) (FAO 2015a); (13) (Venter et al. 2016); (14) (Ellis et al. 2010)

1.1.2.3 Past and ongoing trends

Globally, cropland area changed by +15% and the area of permanent pastures by +8% since the early 1960s (FAOSTAT 2018), with strong regional differences (Figure 1.3). In contrast, cropland production since 1961 increased by about 3.5 times, the production of animal products by 2.5 times, and forestry by 1.5 times; in parallel with strong yield (production per unit area) increases (FAOSTAT 2018; Figure 1.3). Per capita calorie supply increased by 17% (since 1970; Kastner et al. 2012), and diet composition changed markedly, tightly associated with economic development and lifestyle: Since the early 1960s, per capita dairy product consumption increased by a factor 1.2, and meat and vegetable oil consumption more than doubled (FAO 2017, 2018b; Tilman and Clark 2014; Marques et al. 2019). Population and livestock production represent key drivers of the global expansion of cropland for food production, only partly compensated by yield increases at the global level (Alexander et al. 2015). A number of studies have reported reduced growth rates or stagnation in yields in some regions in the last decades (*medium evidence, high agreement*; Lin and Huybers 2012; Ray et al. 2012; Elbehri, Aziz, Joshua Elliott 2015; see Section 5.2.2).

The past increases in agricultural production have been associated with strong increases in agricultural inputs (Foley et al. 2011; Siebert et al. 2015; Lassaletta et al. 2016; Figure 1.1, Figure 1.3). Irrigation area doubled, total nitrogen fertiliser use increased 9 times (FAOSTAT 2018; IFASTAT 2018) since the early 1960s. Biomass trade volumes grew by a factor of nine (in tons dry matter yr^{-1}) in this period, which is much stronger than production (FAOSTAT 2018), resulting in a growing spatial disconnect between regions of production and consumption (Friis et al. 2016; Friis and Nielsen 2017; Schröter et al. 2018; Liu et al. 2013; Krausmann and Langthaler 2019). Urban and other infrastructure areas expanded by a factor 2 since 1960 (Krausmann et al. 2013), resulting in disproportionally large losses of highly-fertile cropland (Seto and Reenberg 2014; Martellozzo et al. 2015; Bren d'Amour et al. 2016; Seto and Ramankutty 2016; van Vliet et al. 2017). World regions show distinct patterns of change (Figure 1.3).

While most pastureland expansion replaced natural grasslands, cropland expansion replaced mainly forests (Ramankutty et al. 2018; Ordway et al. 2017; Richards and Friess 2016). Noteworthy large conversions occurred in tropical dry woodlands and savannahs, for example, in the Brazilian Cerrado (Lehmann and Parr 2016; Strassburg et al. 2017), the South-American Caatinga and Chaco regions (Parr et al. 2014; Lehmann and Parr 2016) or African savannahs (Ryan et al. 2016). More than half of the original 4.3–12.6 million km² global wetlands (Erb et al. 2016a; Davidson 2014; Dixon et al. 2016) have been drained; since 1970 the wetland extend index, developed by aggregating data field-site time series that report changes in local wetland area indicate a decline by > 30% (Figure 1.1, see Section 4.2.1, Darrah et al. 2019). Likewise, one third of the estimated global area that in a non-used state would be covered in forests (Erb et al. 2017) has been converted to agriculture.

Global forest area declined by 3% since 1990 (about -5% since 1960) and continues to do so (FAO 2015a; Keenan et al. 2015; MacDicken et al. 2015; FAO 1963; Figure 1.1), but uncertainties are large. *Low agreement* relates to the concomitant trend of global tree-cover. Some remote-sensing based assessments show global net-losses of forest or tree cover (Li et al. 2016; Nowosad et al. 2018; Hansen et al. 2013), others indicate a net gain (Song et al. 2018). Tree-cover gains would be in line with observed and modelled increases in photosynthetic active tissues ("greening"; Chen et al. 2019; Zhu et al. 2016; Zhao et al. 2018; de Jong et al. 2013; Pugh et al. 2019; De Kauwe et al. 2016; Kolby Smith et al. 2015; see Box 2.3 in Chapter 2), but *confidence* remains *low* whether gross forest or tree cover gains are as large, or larger, than losses. This uncertainty, together with poor information on forest management, affects estimates and attribution of the land carbon sink (see Section 2.3, 4.3, 4.6). Discrepancies are caused by different classification schemes and applied thresholds (e.g., minimum tree height and tree cover thresholds used to define a forest), the divergence of forest and tree cover, and differences in methods and spatiotemporal resolution (Keenan et al. 2015; Schepaschenko et al. 2015; Bastin et al. 2017; Sloan and Sayer 2015; Chazdon et al. 2016a; Achard et al. 2014). However, there is *robust evidence and high agreement* that a net loss of forest and tree cover prevails in the tropics and a net-gain, mainly of secondary, semi-natural and planted, forests, in the temperate and boreal zones.

The observed regional and global historical land-use trends result in regionally distinct patterns of C fluxes between land and the atmosphere (Figure 1.3B). They are also associated with declines in biodiversity, far above background rates (Ceballos et al. 2015; De Vos et al. 2015; Pimm et al. 2014; Newbold et al. 2015; Maxwell et al. 2016; Marques et al. 2019). Biodiversity losses from past global land-use change have been estimated to be about 8–14%, depending on the biodiversity indicator applied (Newbold et al. 2015; Wilting et al. 2017; Gossner et al. 2016; Newbold et al. 2018; Paillet et al. 2010). In future, climate warming has been projected to accelerate losses of species diversity rapidly (Settele et al. 2014; Urban et al. 2016; Scholes et al. 2018; Fischer et al. 2018; Hoegh-Guldberg et al. 2018). The concomitance of land-use and climate-change pressures render ecosystem restoration a key challenge (Anderson-Teixeira 2018; Yang et al. 2019; see Section 4.8, 4.9).

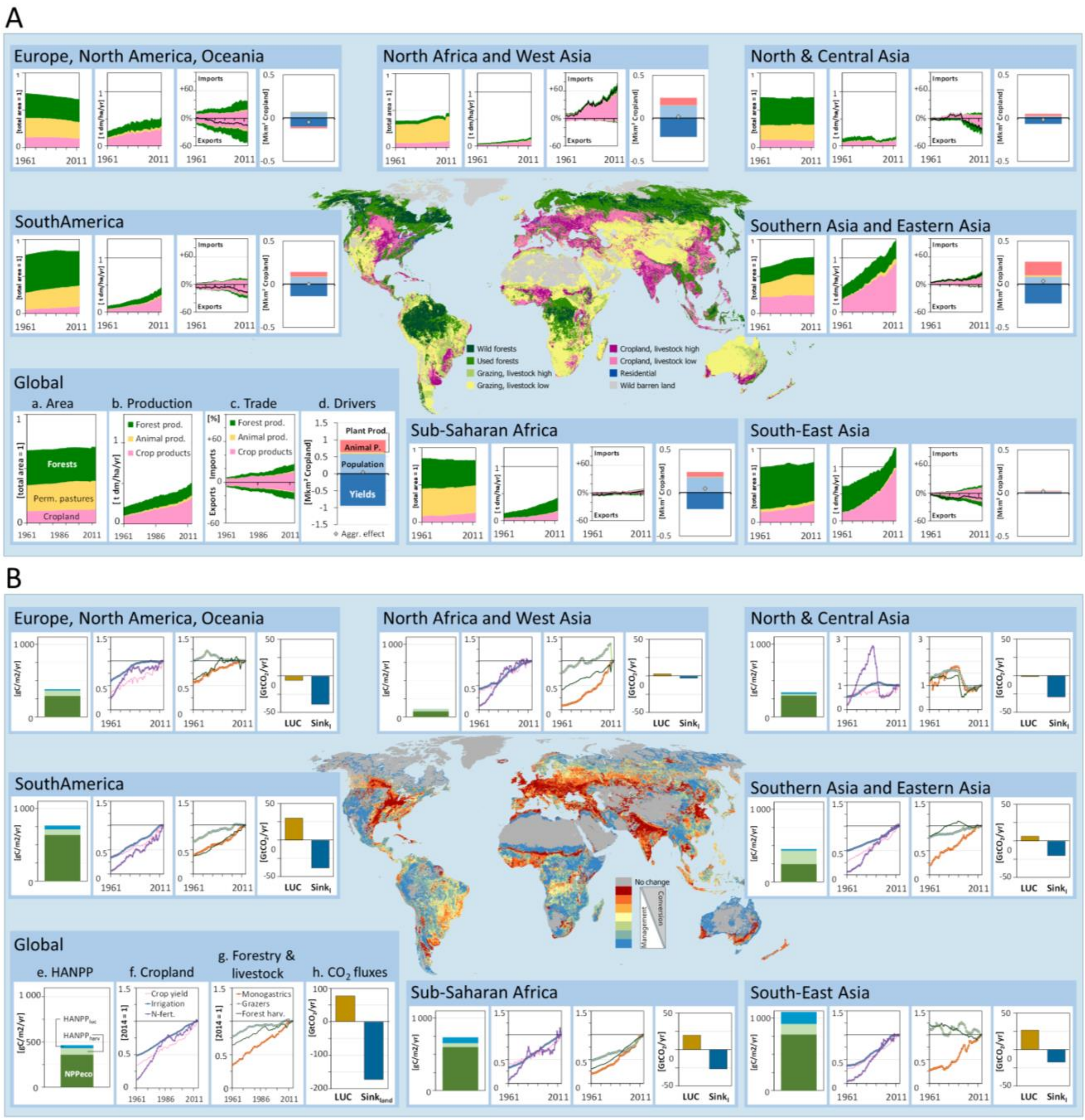

Figure 1.3 Status and trends in the global land system. A. Trends in area, production and trade, and drivers of change. The map shows the global pattern of land systems (combination of maps Nachtergaele (2008); Ellis et al. (2010); Potapov et al. (2017); FAO's Animal Production and Health Division (2018); livestock low/high relates to low or high livestock density, respectively). The inlay figures show, for the globe and 7 world regions, from left to right: (a) Cropland, permanent pastures and forest (used and unused) areas,

standardised to total land area, (b) production in dry matter per year per total land area, (c) trade in dry matter in percent of total domestic production, all for 1961 to 2014 (data from FAOSTAT (2018) and FAO (1963) for forest area 1961). (d) drivers of cropland for food production between 1994 and 2011 (Alexander et al. 2015). See panel "global" for legend. "Plant Produc., Animal P.": changes in consumption of plant-based products and animal-products, respectively. B. Selected land-use pressures and impacts. The map shows the ratio between impacts on biomass stocks of land cover conversions and of land management (changes that occur with land cover types; only changes larger than 30 gCm-2 displayed; Erb et al. 2017), compared to the biomass stocks of the potential vegetation (vegetation that would prevail in the absence of land use, but with current climate). The inlay figures show, from left to right (e) the global Human Appropriation of Net Primary production (HANPP) in the year 2005, in $gCm^{-2}yr^{-1}$ (Krausmann et al. 2013). The sum of the three components represents the NPP of the potential vegetation and consist of: (i) NPP_{eco}, i.e. the amount of NPP remaining in ecosystem after harvest, (ii) $HANPP_{harv}$, i.e. NPP harvested or killed during harvest, and (iii) $HANPP_{luc}$, i.e. NPP foregone due to land-use change. The sum of NPP_{eco} and $HANPP_{harv}$ is the NPP of the actual vegetation (Haberl et al. 2014; Krausmann et al. 2013). The two central inlay figures show changes in land-use intensity, standardised to 2014, related to (f) cropland (yields, fertilisation, irrigated area) and (g) forestry harvest per forest area, and grazers and monogastric livestock density per agricultural area (FAOSTAT 2018). (h) Cumulative CO_2 fluxes between land and the atmosphere between 2000 and 2014. LUC: annual CO_2 land use flux due to changes in land cover and forest management; $Sink_{land}$: the annual CO_2 land sink caused mainly by the indirect anthropogenic effects of environmental change (e.g, climate change and the fertilising effects of rising CO_2 and N concentrations), excluding impacts of land-use change (Le Quéré et al. 2018; see Section 2.3).

1.2 Key challenges related to land use change

1.2.1 Land system change, land degradation, desertification and food security

1.2.1.1 Future trends in the global land system

Human population is projected to increase to nearly 9.8 (± 1) billion people by 2050 and 11.2 billion by 2100 (United Nations 2018). More people, a growing global middle class (Crist et al. 2017), economic growth, and continued urbanisation (Jiang and O'Neill 2017) increase the pressures on expanding crop and pasture area and intensifying land management. Changes in diets, efficiency and technology could reduce these pressures (Billen et al. 2015; Popp et al. 2016; Muller et al. 2017; Alexander et al. 2015; Springmann et al. 2018; Myers et al. 2017; Erb et al. 2016c; FAO 2018b; see Section 5.3, Section 6.2.2).

Given the large uncertainties underlying the many drivers of land use, as well as their complex relation to climate change and other biophysical constraints, future trends in the global land system are explored in scenarios and models that seek to span across these uncertainties (see Cross-Chapter Box 1: Scenarios, in this Chapter). Generally, these scenarios indicate a continued increase in global food demand, owing to population growth and increasing wealth. The associated land area needs are a key uncertainty, a function of the interplay between production, consumption, yields, and production efficiency (in particular for livestock and waste)(FAO 2018b; van Vuuren et al. 2017; Springmann et al. 2018; Riahi et al. 2017; Prestele et al. 2016; Ramankutty et al. 2018; Erb et al. 2016b; Popp et al. 2016; see 1.3 and Cross-Chapter Box 1: Scenarios, in this Chapter). Many factors, such as climate change, local contexts, education, human and social capital, policy-making, economic framework conditions, energy availability, degradation, and many more, affect this interplay, as discussed in all chapters of this report.

Global telecouplings in the land system, the distal connections and multidirectional flows between regions and land systems, are expected to increase, due to urbanisation (Seto et al. 2012; van Vliet et al. 2017; Jiang and O'Neill 2017; Friis et al. 2016), and international trade (Konar et al. 2016; Erb et al. 2016b; Billen et al. 2015; Lassaletta et al. 2016). Telecoupling can support efficiency gains in production, but can also lead to complex cause-effect chains and indirect effects such as land competition or leakage (displacement of the environmental impacts, see glossary), with governance challenges (Baldos and

Hertel 2015; Kastner et al. 2014; Liu et al. 2013; Wood et al. 2018; Schröter et al. 2018; Lapola et al. 2010; Jadin et al. 2016; Erb et al. 2016b; Billen et al. 2015; Chaudhary and Kastner 2016; Marques et al. 2019; Seto and Ramankutty 2016; see Section 1.2.1.5). Furthermore, urban growth is anticipated to occur at the expense of fertile (crop)land, posing a food security challenge, in particular in regions of high population density and agrarian-dominated economies, with limited capacity to compensate for these losses (Seto et al. 2012; Güneralp et al. 2013; Aronson et al. 2014; Martellozzo et al. 2015; Bren d'Amour et al. 2016; Seto and Ramankutty 2016; van Vliet et al. 2017).

Future climate change and increasing atmospheric CO_2 concentration are expected to accentuate existing challenges by, for example, shifting biomes or affecting crop yields (Baldos and Hertel 2015; Schlenker and Lobell 2010; Lipper et al. 2014; Challinor et al. 2014; Myers et al. 2017; see Section 5.2.2), as well as through land-based, climate change mitigation. There is *high confidence* that large-scale implementation of bioenergy or afforestation can further exacerbate existing challenges(Smith et al. 2016; see also Section 1.3.1 and Cross-chapter box 7 on bioenergy in Chapter 6).

1.2.1.2 Land Degradation

As discussed in Chapter 4, the concept of land degradation, including its definition, has been used in different ways in different communities and in previous assessments (such as the IPBES Land degradation and restoration assessment). In the SRCCL, land degradation is defined as a *negative trend in land condition, caused by direct or indirect human-induced processes including anthropogenic climate change, expressed as long-term reduction or loss of at least one of the following: biological productivity, ecological integrity or value to humans*. This definition applies to forest and non-forest land (see Chapter 4 and Glossary).

Land degradation is a critical issue for ecosystems around the world due to the loss of actual or potential productivity or utility (Ravi et al. 2010; Mirzabaev et al. 2015; FAO and ITPS 2015; Cerretelli et al. 2018). Land degradation is driven to a large degree by unsustainable agriculture and forestry, socioeconomic pressures, such as rapid urbanisation and population growth, and unsustainable production practices in combination with climatic factors (Field et al. 2014b; Lal 2009; Beinroth , F. H., Eswaran, H., Reich, P. F. and Van Den Berg 1994; Abu Hammad and Tumeizi 2012; Ferreira et al. 2018; Franco and Giannini 2005; Abahussain et al. 2002).

Global estimates of the total degraded area (excluding deserted area) vary from less than 10 million km^2 to over 60 million km^2, with additionally large disagreement regarding the spatial distribution (Gibbs and Salmon 2015; see Section 4.3). The annual increase in the degraded land area has been estimated as 50,000–10,000 million km^2 yr^{-1} (Stavi and Lal 2015), and the loss of total ecosystem services equivalent to about 10% of the world's GDP in the year 2010 (Sutton et al. 2016). Although land degradation is a common risk across the globe, poor countries remain most vulnerable to its impacts. Soil degradation is of particular concern, due to the long period necessary to restore soils (Lal 2009; Stockmann et al. 2013; Lal 2015), as well as the rapid degradation of primary forests through fragmentation (Haddad et al. 2015). Among the most vulnerable ecosystems to degradation are high carbon stock wetlands (including peatlands). Drainage of natural wetlands for use in agriculture leads to high CO_2 emissions and degradation (*high confidence*) (Strack 2008; Limpens et al. 2008; Aich et al. 2014; Murdiyarso et al. 2015; Kauffman et al. 2016; Dohong et al. 2017; Arifanti et al. 2018; Evans et al. 2019). Land degradation is an important factor contributing to uncertainties in the mitigation potential of land-based ecosystems (Smith et al. 2014). Furthermore, degradation that reduces forest (and agricultural) biomass and soil organic carbon leads to higher rates of runoff (*high confidence*) (Molina et al. 2007; Valentin et al. 2008; Mateos et al. 2017; Noordwijk et al. 2017) and hence to increasing flood risk (*low confidence*) (Bradshaw et al. 2007; Laurance 2007; van Dijk et al. 2009).

1.2.1.3 Desertification

The SRCCL adopts the definition of the UNCCD of desertification being land degradation in arid, semi-arid and dry sub-humid areas (drylands) (see glossary, and Section 3.1.1). Desertification results from various factors, including climate variations and human activities, and is not limited to irreversible forms of land degradation (Tal 2010)(Bai et al. 2008). A critical challenge in the assessment of desertification is to identify a "non-desertified" reference state (Bestelmeyer et al. 2015). While climatic trends and variability can change the intensity of desertification processes, some authors exclude climate effects, arguing that desertification is a purely human-induced process of land degradation with different levels of severity and consequences (Sivakumar 2007).

As a consequence of varying definitions and different methodologies, the area of desertification varies widely (see (D'Odorico et al. 2013; Bestelmeyer et al. 2015), and references therein). Arid regions of the world cover up to about 46% of the total terrestrial surface (about 60 million km^2; Pravalie 2016; Koutroulis 2019). Around 3 billion people reside in dryland regions (D'Odorico et al. 2013; Maestre et al. 2016; see Section 3.1.1), and the number of people living in areas affected by desertification has been estimated as > 630 million, compared to 211 million in the early 1960s (see Fig. 1.1, see Section 3.1.1). The combination of low rainfall with frequently infertile soils renders these regions, and the people who rely on them, vulnerable to both climate change, and unsustainable land management (*high confidence*). In spite of the national, regional and international efforts to combat desertification, it remains one of the major environmental problems (Abahussain et al. 2002; Cherlet et al. 2018).

1.2.1.4 Food security, food systems and linkages to land-based ecosystems

The High Level Panel of Experts of the Committee on Food Security define the food system as to "*gather all the elements (environment, people, inputs, processes, infrastructures, institutions, etc.) and activities that relate to the production, processing, distribution, preparation and consumption of food, and the output of these activities, including socio-economic and environmental outcomes*" (HLPE 2017). Likewise, food security has been defined as "*a situation that exists when all people, at all times, have physical, social and economic access to sufficient, safe and nutritious food that meets their dietary needs and food preferences for an active and healthy life* " (FAO 2017). By this definition, food security is characterised by food availability, economic and physical access to food, food utilisation and food stability over time. Food and nutrition security is one of the key outcomes of the food system (FAO 2018b; Figure 1.4).

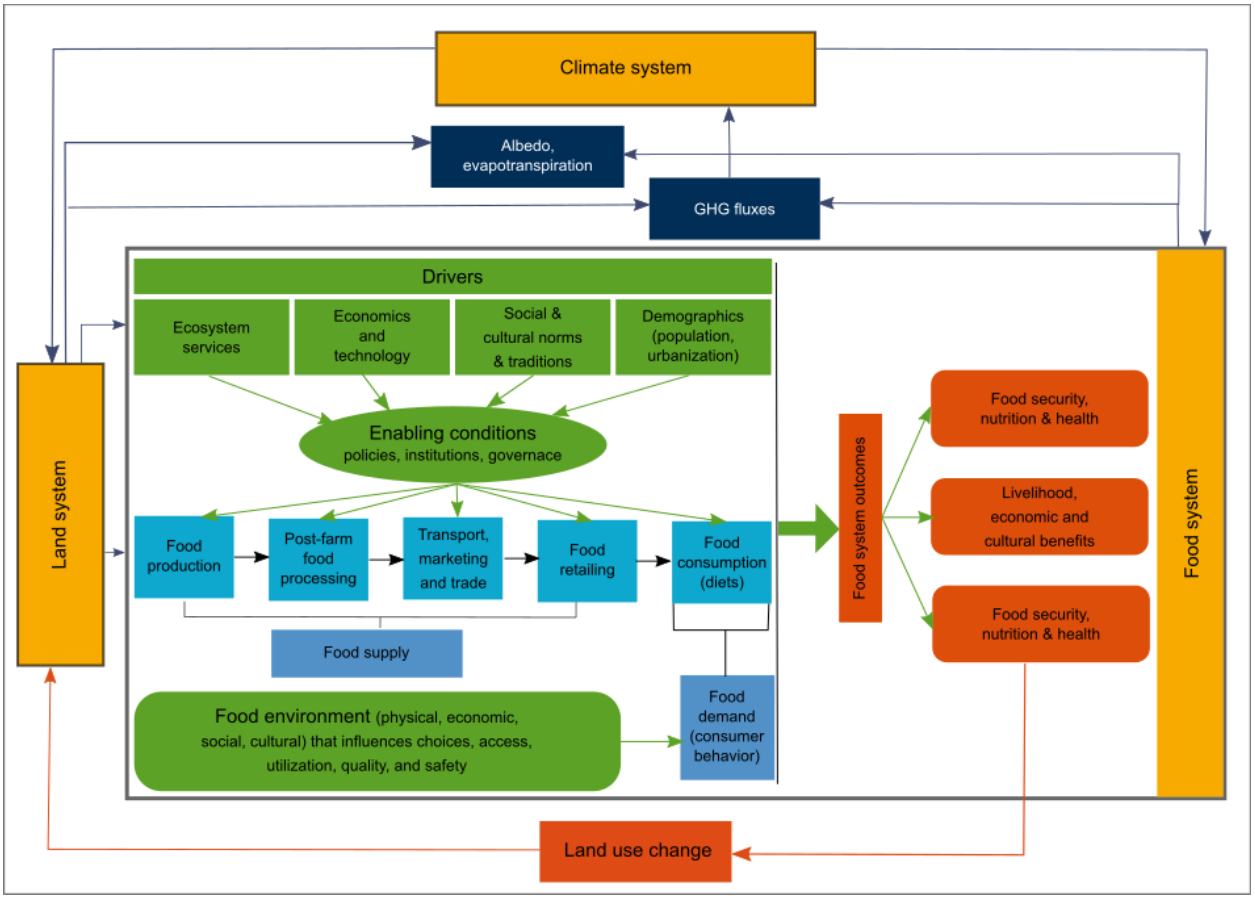

Figure 1.4 Food system (and its relations to land and climate): The food system is conceptualised through supply (production, processing, marketing and retailing) and demand (consumption and diets) that are shaped by physical, economic, social and cultural determinants influencing choices, access, utilisation, quality, safety and waste. Food system drivers (ecosystem services, economics and technology, social and cultural norms and traditions, and demographics) combine with the enabling conditions (policies, institutions and governance) to affect food system outcomes including food security, nutrition and health, livelihoods, economic and cultural benefits as well as environmental outcomes or side-effects (nutrient and soil loss, water use and quality, GHG emissions and other pollutants). Climate and climate change has direct impact on the food system (productivity, variability, nutritional quality) while the latter contribute to local climate (albedo, evapotranspiration) and global warming (GHGs). The land system (function, structures, and processes) affect the food system directly (food production) and indirectly (ecosystem services) while food demand and supply processes affect land (land use change) and land-related processes (e.g., land degradation, desertification) (see chapter 5).

After a prolonged decline, world hunger appears to be on the rise again with the number of undernourished people having increased to an estimated 821 million in 2017, up from 804 million in 2016 and 784 million in 2015, although still below the 900 million reported in 2000 (FAO et al. 2018; see Section 5.1.2). Of the total undernourished in 2018, lived, for example, 256.5 million in Africa, and 515.1 million in Asia (excluding Japan). The same report also states that child undernourishment continues to decline, but levels of overweight populations and obesity are increasing. The total number of overweight children in 2017 was 38-40 million worldwide, and globally up to around two billion adults are by now overweight (see Section 5.1.2). FAO also estimated that close to 2000 million people suffer from micronutrient malnutrition (FAO 2018b).

Food insecurity most notably occurs in situations of conflict and conflict combined with droughts or floods (Cafiero et al. 2018; Smith et al. 2017). The close parallel between food insecurity prevalence and poverty means that tackling development priorities would enhance sustainable land use options for climate mitigation.

Climate change affects the food system as changes in trends and variability in rainfall and temperature variability impact crop and livestock productivity and total production (Osborne and Wheeler 2013; Tigchelaar et al. 2018; Iizumi and Ramankutty 2015), the nutritional quality of food (Loladze 2014; Myers et al., 2014; Ziska et al. 2016; Medek et al., 2017), water supply (Nkhonjera 2017), and incidence of pests and diseases (Curtis et al. 2018). These factors also impact on human health and increase morbidity and affect human ability to process ingested food (Franchini and Mannucci 2015; Wu et al. 2016; Raiten and Aimone 2017). At the same time, the food system generates negative externalities (the environmental effects of production and consumption) in the form of GHG emissions (Section 1.1.2, Section 2.3), pollution (van Noordwijk and Brussaard 2014; Thyberg and Tonjes 2016; Borsato et al. 2018; Kibler et al. 2018), water quality (Malone et al. 2014; Norse and Ju 2015), and ecosystem services loss (Schipper et al. 2014; Eeraerts et al. 2017) with direct and indirect impacts on climate change and reduced resilience to climate variability. As food systems are assessed in relation to their contribution to global warming and/or to land degradation (e.g., livestock systems) it is critical to evaluate their contribution to food security and livelihoods and to consider alternatives, especially for developing countries where food insecurity is prevalent (Röös et al. 2017; Salmon et al. 2018).

1.2.1.5 Challenges arising from land governance

Land use change has both positive and negative effects: it can lead to economic growth, but it can become a source of tension and social unrest leading to elite capture, and competition (Haberl 2015). Competition for land plays out continuously among different use types (cropland, pastureland, forests, urban spaces, and conservation and protected lands) and between different users within the same land use category (subsistence vs. commercial farmers)(Dell'Angelo et al. 2017b). Competition is mediated through economic and market forces (expressed through land rental and purchases, as well as trade and investments). In the context of such transactions, power relations often disfavour disadvantaged groups such as small scale farmers, indigenous communities or women (Doss et al. 2015; Ravnborg et al. 2016). These drivers are influenced to a large degree by policies, institutions and governance structures. Land governance determines not only who can access the land, but also the role of land ownership (legal, formal, customary or collective) which influences land use, land use change and the resulting land competition (Moroni 2018).

Globally, there is competition for land because it is a finite resource and because most of the highly-productive land is already exploited by humans (Lambin and Meyfroidt 2011; Lambin 2012; Venter et al. 2016). Driven by growing population, urbanisation, demand for food and energy, as well as land degradation, competition for land is expected to accentuate land scarcity in the future(Tilman et al. 2011; Foley et al. 2011; Lambin 2012; Popp et al. 2016)(*robust evidence, high agreement*). Climate change influences land use both directly and indirectly, as climate policies can also a play a role in increasing land competition via forest conservation policies, afforestation, or energy crop production (see Section 1.3.1), with the potential for implications for food security (Hussein et al. 2013) and local land-ownership.

An example of large-scale change in land ownership is the much-debated large-scale land acquisition (LSLA) by investors which peaked in 2008 during the food price crisis, the financial crisis, and has also been linked to the search for biofuel investments (Dell'Angelo et al. 2017a). Since 2000, almost 50 million hectares of land, have been acquired, and there are no signs of stagnation in the foreseeable future

(Land Matrix 2018). The LSLA phenomenon, which largely targets agriculture, is widespread, including Sub-Saharan Africa, Southeast Asia, Eastern Europe and Latin America (Rulli et al. 2012; Nolte et al. 2016; Constantin et al. 2017). LSLAs are promoted by investors and host governments on economic grounds (infrastructure, employment, market development)(Deininger et al. 2011), but their social and environmental impacts can be negative and significant (Dell'Angelo et al. 2017a).

Much of the criticism of LSLA focuses on their social impacts, especially the threat to local communities' land rights (especially indigenous people and women) (Anseeuw et al. 2011) and displaced communities creating secondary land expansion (Messerli et al. 2014; Davis et al. 2015). The promises that LSLAs would develop efficient agriculture on non-forested, unused land (Deininger et al. 2011) has so far not been fulfilled. However, LSLAs is not the only outcome of weak land governance structures (Wang et al. 2016), other forms of inequitable or irregular land acquisition can also be home-grown pitting one community against a more vulnerable group (Xu 2018) or land capture by urban elites (McDonnell 2017). As demands on land are increasing, building governance capacity and securing land tenure becomes essential to attain sustainable land use, which has the potential to mitigate climate change, promote food security, and potentially reduce risks of climate-induced migration and associated risks of conflicts (see Section 7.6).

1.2.2 Progress in dealing with uncertainties in assessing land processes in the climate system

1.2.2.1 Concepts related to risk, uncertainty and confidence

In context of the SRCCL, risk refers to the potential for the adverse consequences for human or (land-based) ecological systems, arising from climate change or responses to climate change. Risk related to climate change impacts integrates across the hazard itself, the time of exposure and the vulnerability of the system; the assessment of all three of these components, their interactions, and outcomes are uncertain (see glossary for expanded definition and Section 7.1.2). For instance, a risk to human society is the continued loss of productive land which might arise from climate change, mismanagement, or a combination of both factors. However, risk can also arise from the potential for adverse consequences from responses to climate change, such as widespread deployment of bioenergy which is intended to reduce greenhouse gas emissions and thus limit climate change, but can present its own risks to food security (see chapters 5, 6 and 7).

Demonstrating with some statistical certainty that the climate or the land system affected by climate or land use has changed (detection), and evaluating the relative contributions of multiple causal factors to that change (with a formal assessment of confidence; attribution. See glossary) remain challenging aspects in both observations and models (Rosenzweig and Neofotis 2013; Gillett et al. 2016; Lean 2018). Uncertainties arising for example, from missing or imprecise data, ambiguous terminology, incomplete process representation in models, or human decision making contribute to these challenges, and some examples are provided in this subsection. In order to reflect various sources of uncertainties in the state of scientific understanding, IPCC assessment reports provide estimates of confidence (Mastrandrea et al. 2011). This confidence language is also used in the SRCCL (Figure 1.5):

Evaluation →	Confidence language →	Likelihood language (if possible)
Assess evidence based on numerous sources such as observations, model output, experiments.	Assign level of confidence based on (i) type and quantity of evidence and (ii) degree of agreement.	Assign likelihood language, if results can be placed into statistically defined brackets.

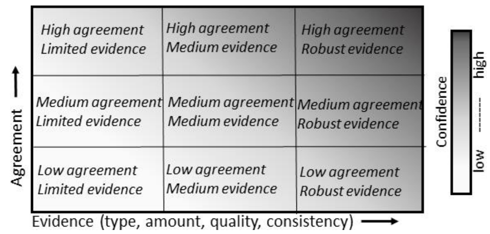

Figure 1.5 Use of confidence language

1.2.2.2 Nature and scope of uncertainties related to land use

Identification and communication of uncertainties is crucial to support decision making towards sustainable land management. Providing a robust, and comprehensive understanding of uncertainties in observations, models and scenarios is a fundamental first step in the IPCC confidence framework (see above). This will remain a challenge in future, but some important progress has been made over recent years.

Uncertainties in observations

The detection of changes in vegetation cover and structural properties underpins the assessment of land-use change, degradation and desertification. It is continuously improving by enhanced Earth observation capacity (Hansen et al. 2013; He et al. 2018; Ardö et al. 2018; Spennemann et al. 2018) (see also Table SM. 1.1 in Supplementary Materials). Likewise, the picture of how soil organic carbon, and GHG and water fluxes respond to land-use change and land management continues to improve through advances in methodologies and sensors (Kostyanovsky et al. 2018; Brümmer et al. 2017; Iwata et al. 2017; Valayamkunnath et al. 2018). In both cases, the relative shortness of the record, data gaps, data treatment algorithms and –for remote sensing- differences in the definitions of major vegetation cover classes limits the detection of trends (Alexander et al. 2016a; Chen et al. 2014; Yu et al. 2014; Lacaze et al. 2015; Song 2018; Peterson et al. 2017). In many developing countries, the cost of satellite remote sensing remains a challenge, although technological advances are starting to overcome this problem (Santilli et al. 2018), while ground-based observations networks are often not available.

Integration of multiple data sources in model and data assimilation schemes reduces uncertainties (Li et al. 2017; Clark et al. 2017; Lees et al. 2018), which might be important for the advancement of early warning systems. Early warning systems are a key feature of short-term (i.e. seasonal) decision support systems and are becoming increasingly important for sustainable land management and food security (Shtienberg 2013; Jarroudi et al. 2015; see Section 6.2.3, 7.4.3). Early warning systems can help to optimise fertiliser and water use, aid disease suppression, and/or increase the economic benefit by enabling strategic farming decisions on when and what to plant (Caffi et al. 2012; Watmuff et al. 2013; Jarroudi et al. 2015; Chipanshi et al. 2015). Their suitability depends on the capability of the methods to accurately predict crop or pest developments, which in turn depends on expert agricultural knowledge, and the accuracy of the weather data used to run phenological models (Caffi et al. 2012; Shtienberg 2013).

Uncertainties in models

Model intercomparison is a widely used approach to quantify some sources of uncertainty in climate change, land-use change and ecosystem modelling, often associated with the calculation of model-ensemble medians or means (see e.g., Section 2.2; Section 5.2). Even models of broadly similar structure differ in their projected outcome for the same input, as seen for instance in the spread in climate change projections from Earth System Models (ESMs) to similar future anthropogenic GHG emissions (Parker 2013; Stocker et al. 2013a). These uncertainties arise, for instance, from different parameter values, different processes represented in models, or how these processes are mathematically described. If the output of ESM simulations are used as input to impact models, these uncertainties can propagate to projected impacts (Ahlstrom et al. 2013).

Thus, the increased quantification of model performance in benchmarking exercises (the repeated confrontation of models with observations to establish a track-record of model developments and performance) is an important development to support the design and the interpretation of the outcomes of model ensemble studies (Randerson et al. 2009; Luo et al. 2012; Kelley et al. 2013). Since observational data sets in themselves are uncertain, benchmarking benefits from transparent information on the observations that are used, and the inclusion of multiple, regularly updated data sources (Luo et al. 2012; Kelley et al. 2013). Improved benchmarking approaches and the associated scoring of models may support weighted model means contingent on model performance. This could be an important step forward when calculating ensemble means across a range of models (Buisson et al. 2009; Parker 2013; Prestele et al. 2016).

Uncertainties arising from unknown futures

Large differences exist in projections of future land cover change, both between and within scenario projections (Fuchs et al. 2015; Eitelberg et al. 2016; Popp et al. 2016; Krause et al. 2017; Alexander et al. 2016a). These differences reflect the uncertainties associated with baseline data, thematic classifications, different model structures and model parameter estimation (Alexander et al. 2017a; Prestele et al. 2016; Cross-Chapter Box 1: Scenarios, in this Chapter). Likewise, projections of future land-use change are also highly uncertain, reflecting –among other factors- the absence of important crop, pasture and management processes in Integrated Assessment Models (Cross-Chapter Box 1: Scenarios, in this Chapter; Rose 2014) and in models of the terrestrial carbon cycle (Arneth et al. 2017). These processes have been shown to have large impacts on carbon stock changes (Arneth et al. 2017). Common scenario frameworks are used to capture the range of future uncertainties in scenarios. The most commonly used recent framework in climate change studies is based on the Representative Concentration Pathways (RCPs) and the Shared Socio-economic Pathways (SSPs)(Popp et al. 2016; Riahi et al. 2017). The RCPs prescribe levels of radiative forcing (Wm^{-2}) arising from different atmospheric concentrations of GHGs that lead to different levels of climate change. For example, RCP2.6 (2.6 Wm^{-2}) is projected to lead to global mean temperature changes of about 0.9°C–2.3°C, and RCP8.5 (8.5 Wm^{-2}) to global mean temperature changes of about 3.2°C–5.4°C (van Vuuren et al 2014).

The SSPs describe alternative trajectories of future socio-economic development with a focus on challenges to climate mitigation and challenges to climate adaptation (O'Neill et al. 2014). SSP1 represents a sustainable and co-operative society with a low carbon economy and high capacity to adapt to climate change. SSP3 has social inequality that entrenches reliance on fossil fuels and limits adaptive capacity. SSP4 has large differences in income within and across world regions that facilitates low carbon economies in places, but limits adaptive capacity everywhere. SSP5 is a technologically advanced world with a strong economy that is heavily dependent on fossil fuels, but with high adaptive capacity. SSP2 is an intermediate case between SSP1 and SSP3 (O'Neill et al. 2014). The SSPs are commonly used with models to project future land use change (Cross-Chapter Box 1: Scenarios, in this Chapter).

The SSPs map onto the RCPs through shared assumptions. For example, a higher level of climate change (RCP8.5) is associated with higher challenges for climate change mitigation (SSP5). Not all SSPs are, however, associated with all RCPs. For example, an SSP5 world is committed to high fossil fuel use, associated GHG emissions, and this is not commensurate with lower levels of climate change (e.g., RCP2.6). (Engstrom et al. 2016) took this approach further by ascribing levels of probability that associate an SSP with an RCP, contingent on the SSP scenario assumptions (see Cross-Chapter Box 1: Scenarios, in this Chapter).

Cross-Chapter Box 1: Scenarios and other methods to characterise the future of land

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About this box

The land-climate system is complex and future changes are uncertain, but methods exist (collectively known as *futures analysis*) to help decision makers in navigating through this uncertainty. Futures analysis comprises a number of different and widely used methods, such as scenario analysis (Rounsevell and Metzger 2010), envisioning or target setting (Kok et al. 2018), pathways analysis[1] (IPBES 2016; IPCC 2018), and conditional probabilistic futures (Vuuren et al. 2018; Engstrom et al. 2016; Henry et al. 2018)(see Cross-Chapter Box 1, Table 1). Scenarios and other methods to characterise the future can support a discourse with decision makers about the sustainable development options that are available to them. All chapters of this assessment draw conclusions from futures analysis and so, the purpose of this box is to outline the principal methods used, their application domains, their uncertainties and their limitations.

Exploratory scenario analysis

Many exploratory scenarios are reported in climate and land system studies on climate change (Dokken 2014), land-based, climate-change mitigation for example, reforestation/afforestation, avoided deforestation and bioenergy (Kraxner et al. 2013; Humpenoder et al. 2014; Krause et al. 2017) and climate change impacts and adaptation (Warszawski et al. 2014). There are global-scale scenarios of food security (Foley et al. 2011; Pradhan et al. 2013, 2014), but fewer scenarios of desertification, land degradation and restoration (Wolff et al. 2018). Exploratory scenarios combine qualitative 'storylines' or descriptive narratives of the underlying causes (or drivers) of change (Nakicenovic and Swart 2000; Rounsevell and Metzger 2010; O'Neill et al. 2014) with quantitative projections from computer models. Different types of models are used for this purpose based on very different modelling paradigms, baseline data and underlying assumptions (Alexander et al. 2016a; Prestele et al. 2016). Cross-Chapter Box 1, Figure 1 outlines how a combination of models can quantify these components as well as the interactions between them.

Exploratory scenarios often show that socio-economic drivers have a larger effect on land use change than climate drivers (Harrison et al. 2014, 2016). Of these, technological development is critical in affecting the production potential (yields) of food and bioenergy and the feed conversion efficiency of livestock (Rounsevell et al. 2006; Wise et al. 2014; Kreidenweis et al. 2018), as well as the area of land needed for food production (Foley et al. 2011; Weindl et al. 2017; Kreidenweis et al. 2018). Trends in consumption, for example, diets, waste reduction, are also fundamental in affecting land use change (Pradhan et al. 2013; Alexander et al. 2016b; Weindl et al. 2017; Alexander et al. 2017; Vuuren et al.

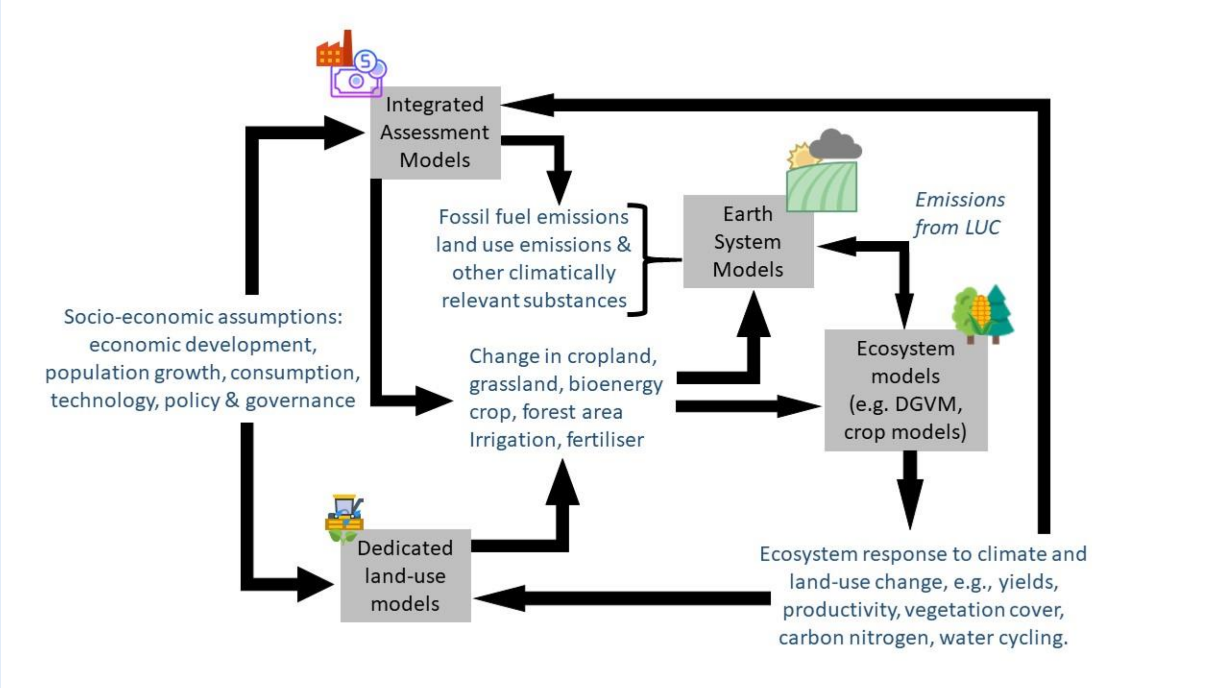
[1] FOOTNOTE: Different communities have a different understanding of the concept of pathways, as noted in the Cross-Chapter Box 1on scenarios in (IPCC 2018). Here, we refer to pathways as a description of the time-dependent actions required to move from today's world to a set of future visions (IPCC 2018). However, the term pathways is commonly used in the climate change literature as a synonym for projections or trajectories (e.g. Shared socio-economic pathways).

2018; Bajželj et al. 2014). Scenarios of land-based mitigation through large-scale bioenergy production and afforestation often lead to negative trade-offs with food security (food prices), water resources and biodiversity (cross chapter box on bioenergy, Ch6).

Cross-Chapter Box 1, Table 1 Description of the principal methods used in land and climate futures analysis

Futures method	Description and subtypes	Application domain	Time horizon	Examples in this assessment
Exploratory scenarios. Trajectories of change in system components from the present to contrasting, alternative futures based on plausible and internally consistent assumptions about the underlying drivers of change	*Long-term projections* quantified with models	Climate system, land system and other components of the environment (e.g., biodiversity, ecosystem functioning, water resources and quality), for example the SSPs	10-100 years	2.3, 2.6.2, 5.2.3, 6.1.4, 6.4.4, 7.2
	Business-as-usual scenarios (including 'outlooks')	A continuation into the future of current trends in key drivers to explore the consequences of these in the near-term	5-10 years, 20-30 years for outlooks	1.2.1, 2.6.2, 5.3.4, 6.1.4
	Policy & planning scenarios (including business planning)	Ex Ante analysis of the consequences of alternative policies or decisions based on known policy options or already implemented policy and planning measures	5-30 years	2.6.3, 5.5.2, 5.6.2, 6.4.4
	Stylised scenarios (with single and multiple options)	Afforestation/reforestation areas, bioenergy areas, protected areas for conservation, consumption patterns (e.g., diets, food waste)	10-100 years	2.6.1, 5.5.1, 5.5.2, 5.6.1, 5.6.2, 6.4.4, 7.2
	Shock scenarios (high impact single events)	Food supply chain collapses, cyberattacks, pandemic diseases (humans, crops and livestock)	Near-term events (up to 10 years) leading to long-term impacts (10-100 years)	5.8.1
	Conditional probabilistic futures ascribe probabilities to uncertain drivers that are conditional on scenario assumptions	Where some knowledge is known about driver uncertainties, for example, population, economic growth, land use change	10-100 years	1.2
Normative scenarios. Desired futures or outcomes that are aspirational and how to achieve them	*Visions, goal-seeking or target-seeking scenarios*	Environmental quality, societal development, human well-being, the Representative Concentration Pathways (RCPs,) 1.5 °C scenarios	5-10 years to 10-100 years	2.6.2, 6.4.4, 7.2, 5.5.2

	Pathways as alternative sets of choices, actions or behaviours that lead to a future vision (goal or target)	Socio-economic systems, governance and policy actions	5-10 years to 10-100 years	5.5.2, 6.4.4, 7.2



Cross-Chapter Box 1, Figure 1 Interactions between land and climate system components and models in scenario analysis. The blue text describes selected model inputs and outputs.

Many exploratory scenarios are based on common frameworks such as the Shared Socio-economic Pathways (SSPs) (Popp et al. 2016; Riahi et al. 2017; Doelman et al. 2018) (see section 1.2). However, other methods are used. *Stylised scenarios* prescribe assumptions about climate and land use change solutions for example, dietary change, food waste reduction, afforestation areas (Pradhan et al. 2013, 2014; Kreidenweis et al. 2016; Rogelj et al. 2018b; Seneviratne et al. 2018; Vuuren et al. 2018). These scenarios provide useful thought experiments, but the feasibility of achieving the stylised assumptions is often unknown. *Shock scenarios* explore the consequences of low probability, high impact events such as pandemic diseases, cyberattacks and failures in food supply chains (Challinor et al. 2018) often in food security studies. Because of the diversity of exploratory scenarios, attempts have been made to categorise them into ‘archetypes’ based on the similarity between their assumptions in order to facilitate communication (IPBES 2018a).

Conditional probabilistic futures explore the consequences of model parameter uncertainty in which these uncertainties are conditional on scenario assumptions (Neill 2004). Only a few studies have applied the conditional probabilistic approach to land use futures (Brown et al. 2014; Engstrom et al. 2016; Henry et al. 2018). By accounting for uncertainties in key drivers these studies show large ranges in land use change, for example, global cropland areas of 893–2380 Mha by the end of the 21st Century (Engstrom et al. 2016). They also find that land-use targets may not be achieved, even across a wide range of scenario parameter settings, because of trade-offs arising from the competition for land (Henry et al. 2018; Heck et al. 2018). Accounting for uncertainties across scenario assumptions can lead to convergent outcomes for land use change, which implies that certain outcomes are more robust across a wide range of uncertain scenario assumptions (Brown et al. 2014).

In addition to global scale scenario studies, sub-national studies demonstrate that regional climate change impacts on the land system are highly variable geographically because of differences in the spatial patterns of both climate and socio-economic change (Harrison et al. 2014). Moreover, the capacity to adapt to these impacts is strongly dependent on the regional, socio-economic context and coping capacity (Dunford et al. 2014); processes that are difficult to capture in global scale scenarios. Regional scenarios are often co-created with stakeholders through participatory approaches (Kok et al. 2014), which is powerful in reflecting diverse worldviews and stakeholder values. Stakeholder participatory methods provide additional richness and context to storylines, as well as providing saliency and legitimacy for local stakeholders (Kok et al. 2014).

Normative scenarios: visions and pathways analysis

Normative scenarios reflect a desired or target-seeking future. Pathways analysis is important in moving beyond the '*what if?'* perspective of exploratory scenarios to evaluate how normative futures might be achieved in practice, recognising that multiple pathways may achieve the same future vision. Pathways analysis focuses on consumption and behavioural changes through transitions and transformative solutions (IPBES 2018a). Pathways analysis is highly relevant in support of policy, since it outlines sets of time-dependent actions and decisions to achieve future targets, especially with respect to sustainable development goals, as well as highlighting trade-offs and co-benefits (IPBES 2018a). Multiple, alternative pathways have been shown to exist that mitigate trade-offs whilst achieving the priorities for future sustainable development outlined by governments and societal actors. Of these alternatives, the most promising focus on long-term societal transformations through education, awareness raising, knowledge sharing and participatory decision-making (IPBES 2018a).

What are the limitations of land use scenarios?

Applying a common scenario framework (e.g., RCPs/SSPs) supports the comparison and integration of climate and land system scenarios, but a 'climate-centric' perspective can limit the capacity of these scenarios to account for a wider range of land-relevant drivers (Rosa et al. 2017). For example, in climate mitigation scenarios it is important to assess the impact of mitigation actions on the broader environment for example, biodiversity, ecosystem functioning, air quality, food security, desertification/degradation and water cycles (Rosa et al. 2017). This implies the need for a more encompassing and flexible approach to creating scenarios that considers other environmental aspects, not only as a part of impact assessment, but also during the process of creating the scenarios themselves.

A limited number of models can quantify global scale, land use change scenarios, and there is large variance in the outcomes of these models (Alexander et al. 2016a; Prestele et al. 2016). In some cases, there is greater variability between the models themselves than between the scenarios that they are quantifying, and these differences vary geographically (Prestele et al. 2016). These differences arise from variations in baseline datasets, thematic classes and modelling paradigms (Alexander et al. 2016a; Popp et al. 2016; Prestele et al. 2016). Model evaluation is critical in establishing confidence in the outcomes of modelled futures (Ahlstrom et al. 2012; Kelley et al. 2013). Some, but not all, land use models are evaluated against observational data and model evaluation is rarely reported. Hence, there is a need for more transparency in land use modelling, especially in evaluation and testing, as well as making model code available with complete sets of scenario outputs (e.g., Dietrich et al. 2018).

There is a small, but growing literature on quantitative pathways to achieve normative visions and their associated trade-offs (IPBES 2018a). Whilst the visions themselves may be clearly articulated, the societal choices, behaviours and transitions needed to attain them, are not. Better accounting for human behaviour and decision-making processes in global scale, land-use models would improve the capacity to quantify pathways to sustainable futures (Rounsevell et al. 2014; Arneth et al. 2014; Calvin and Bond-Lamberty 2018). It is, however, difficult to understand and represent human behaviour and social interaction processes at global scales. Decision-making in global models is commonly represented through economic processes (Arneth et al. 2014). Other important human processes for land systems including equity, fairness, land tenure and the role of institutions and governance, receive less attention, and this limits the use of global models to quantify transformative pathways, adaptation and mitigation (Arneth et al. 2014; Rounsevell et al. 2014; Wang et al. 2016). No model exists at present to represent complex human behaviours at the global scale, although the need has been highlighted (Rounsevell et al. 2014; Arneth et al. 2014; Robinson et al. 2017; Brown et al. 2017; Calvin and Bond-Lamberty 2018).

1.2.2.3 Uncertainties in decision making

Decision makers develop and implement policy in the face of many uncertainties (Rosenzweig and Neofotis 2013; Anav et al. 2013; Ciais et al. 2013a; Stocker et al. 2013b; see Section 7.5).In context of climate change, the term *deep uncertainty* is frequently used to denote situations in which either the analysis of a situation is inconclusive, or parties to a decision cannot agree on a number of criteria that would help to rank model results in terms of likelihood (e.g., Hallegatte and Mach 2016; Maier et al. 2016) (see Section 7.1, 7.5, and Supplementary Material Table SM. 1.2). However, existing uncertainty does not support societal and political inaction.

The many ways of dealing with uncertainty in decision making can be summarised by two decision approaches: (economic) cost-benefit analyses, and the precautionary approach. A typical variant of cost benefit analysis is the minimisation of negative consequences. This approach needs reliable probability estimates (Gleckler et al. 2016; Parker 2013) and tends to focus on the short-term. The precautionary approach does not take account of probability estimates (cf. Raffensperger and Tickner 1999), but instead focuses on avoiding the worst outcome (Gardiner 2006).

Between these two extremes, various decision approaches seek to address uncertainties in a more reflective manner that avoids the limitations of cost-benefit analysis and the precautionary approach. Climate-informed decision analysis combines various approaches to explore options and the vulnerabilities and sensitivities of certain decisions. Such an approach includes stakeholder involvement (e.g., elicitation methods), and can be combined with, for example, analysis of climate or land-use change modelling (Hallegatte and Rentschler 2015; Luedeling and Shepherd 2016).

Flexibility is facilitated by political decisions that are not set in stone and can change over time (Walker et al. 2013; Hallegatte and Rentschler 2015). Generally, within the research community that investigates deep uncertainty a paradigm is emerging that requires to develop a strategic vision of the long- or mid-term future, while committing to short-term actions and establishing a framework to guide future actions including revisions and flexible adjustment of decisions (Haasnoot 2013; see Section 7.5).

1.3 Response options to the key challenges

A number of response options underpin solutions to the challenges arising from GHG emissions from land, and the loss of productivity arising from degradation and desertification. These options are discussed in Sections 2.5, 6.2 and rely on a) land management, b) value chain management and c) risk management (see Table 1.2). None of these response options are mutually exclusive, and it is their combination in a regionally, context-specific manner that is most likely to achieve co-benefits between climate change mitigation, adaptation and other environmental challenges in a cost- effective way (Griscom et al. 2017; Kok et al. 2018). Sustainable solutions affecting both demand and supply are expected to yield most co-benefits if these rely not only on the carbon footprint, but are extended to other vital ecosystems such as water, nutrients and biodiversity footprints (van Noordwijk and Brussaard 2014; Cremasch 2016). As an entry-point to the discussion in Chapter 6, we introduce here a selected number of examples that cut across climate change mitigation, food security, desertification, and degradation issues, including potential trade-offs and co-benefits.

Table 1.2 Broad categorisation of response options into three main classes and eight sub-classes. For illustration, the table includes examples of individual response options. A complete list and description is provided in Chapter 6.

Response options based on land management	
in agriculture	Improved management of: cropland, grazing land, livestock; Agro-forestry; Avoidance of conversion of grassland to cropland; Integrated water management
in forests	Improved management of forests and forest restoration; Reduced deforestation and degradation; Afforestation
of soils	Increased soil organic carbon content; Reduced soil erosion; Reduced soil salinisation
across all/other ecosystems	Reduced landslides and natural hazards; Reduced pollution including acidification; Biodiversity conservation; Restoration and reduced conversion of peatlands
specifically for carbon dioxide removal	Enhanced weathering of minerals; Bioenergy and BECCS
Response options based on value chain management	
through demand management	Dietary change; Reduced post-harvest losses; Reduced food waste
through supply management	Sustainable sourcing; Improved energy use in food systems; Improved food processing and retailing
Response options based on risk management	
risk management	Risk sharing instruments; Use of local seeds; Disaster risk management

1.3.1 Targeted decarbonisation relying on large land-area need

Most global future scenarios that aim to achieve global warming of 2°C or well below rely on bioenergy (BE; with or without carbon capture and storage, BECCS; see Cross-Chapter Box 7 in Chapter 6) or afforestation and reforestation (Cross-Chapter Box 2 in this Chapter)(de Coninck et al. 2018; Rogelj et al. 2018b,a; Anderson and Peters 2016; Popp et al. 2016; Smith et al. 2016). In addition to the very large area requirements projected for 2050 or 2100, several other aspects of these scenarios have also been criticised. For instance, they simulate very rapid technological and societal uptake rates for the land-related mitigation measures, when compared with historical observations (Turner et al. 2018; Brown et al. 2019; Vaughan and Gough 2016). Furthermore, confidence in the projected bioenergy or BECCS net carbon uptake potential is *low*, because of many diverging assumptions. This includes assumptions about bioenergy crop yields, the possibly large energy demand for CCS, which diminishes the net-GHG-saving of bioenergy systems, or the incomplete accounting for ecosystem processes and of the cumulative carbon-loss arising from natural vegetation clearance for bioenergy crops or bioenergy forests and subsequent harvest regimes (Anderson and Peters 2016; Bentsen 2017; Searchinger et al. 2017; Bayer et al. 2017; Fuchs et al. 2017; Pingoud et al. 2018; Schlesinger 2018). Bioenergy provision under politically unstable conditions may also be a problem (Erb et al. 2012; Searle and Malins 2015).

Large-scale bioenergy plantations and forests may compete for the same land area (Harper et al. 2018). Both potentially have adverse side effects on biodiversity and ecosystem services, as well as socio-economic trade-offs such as higher food prices due to land area competition(Shi et al. 2013; Bárcena et al. 2014; Fernandez-Martinez et al. 2014; Searchinger et al. 2015; Bonsch et al. 2016; Creutzig et al. 2015; Kreidenweis et al. 2016; Santangeli et al. 2016; Williamson 2016; Graham et al. 2017; Krause et al. 2017; Hasegawa et al. 2018; Humpenoeder et al. 2018). Although forest-based mitigation could have co-benefits for biodiversity and many ecosystem services, this depends on the type of forest planted and the vegetation cover it replaces (Popp et al. 2014; Searchinger et al. 2015)(see also Cross-Chapter Box 2 in this Chapter).

There is *high confidence* that scenarios with large land requirements for climate change mitigation may not achieve sustainable development goals, such as no poverty, zero hunger and life on land, if competition for land and the need for agricultural intensification are greatly enhanced (Creutzig et al. 2016; Dooley and Kartha 2018; Hasegawa et al. 2015; Hof et al. 2018; Roy et al. 2018; Santangeli et al. 2016; Boysen et al. 2017; Henry et al. 2018; Kreidenweis et al. 2016; UN 2015). This does not mean that smaller-scale land-based climate mitigation can have positive outcomes for then achieving these goals (see e.g., Sections 6.2, 4.5, cross chapter box 7 in Chapter 6).

Cross-Chapter Box 2: Implications of large-scale conversion from non-forest to forest land

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Efforts to increase forest area

While deforestation continues in many world regions, especially in the tropics, large expansion of mostly managed forest area has taken place in some countries. In the IPCC context, reforestation (conversion to forest of land that previously contained forests but has been converted to some other use) is distinguished from afforestation (conversion to forest of land that historically has not contained forests (see glossary)). Past expansion of managed forest area occurred in many world-regions for a variety of reasons, from meeting needs for wood fuel or timber (Vadell et al. 2016; Joshi et al. 2011; Zaloumis and Bond 2015; Payn et al. 2015; Shoyama 2008; Miyamoto et al. 2011) to restoration-driven efforts, with the aim of enhancing ecological function (Filoso et al. 2017; Salvati and Carlucci 2014; Ogle et al. 2018; Crouzeilles et al. 2016; FAO 2016)(see Section 3.7, 4.9).

In many regions, net forest area increase includes deforestation (often of native forests) alongside increasing forest area (often managed forest, but also more natural forest restoration efforts; (Heilmayr et al. 2016; Scheidel and Work 2018; Hua et al. 2018; Crouzeilles et al. 2016; Chazdon et al. 2016b). China and India have seen the largest net forest area increase, aiming to alleviate soil erosion, desertification and overgrazing (Ahrends et al. 2017; Cao et al. 2016; Deng et al. 2015; Chen et al. 2019)(see Section 3.7, 4.9) but uncertainties in exact forest area changes remain large, mostly due to differences in methodology and forest classification (FAO 2015a; Song et al. 2018; Hansen et al. 2013; MacDicken et al. 2015)(Section 1.1.2).

What are the implications for ecosystems?

1) Implications for biogeochemical and biophysical processes

There is *robust evidence* and *medium agreement* that whilst forest area expansion increases ecosystem carbon storage, the magnitude of the increased stock depends on the type and length of former land-use, forest type planted, and climatic regions (Bárcena et al. 2014; Poeplau et al. 2011; Shi et al. 2013; Li et al. 2012)(see Section 4.3). While, reforestation of former croplands increases net ecosystem carbon storage (Bernal et al. 2018; Lamb 2018), afforestation on native grassland results in reduction of soil carbon stocks, which can reduce or negate the net carbon benefits which are dominated by increases in biomass, dead wood and litter carbon pools (Veldman et al. 2015, 2017).

Forest vs. non-forest lands differ in land surface reflectiveness of short-wave radiation and evapotranspiration (Anderson et al. 2011; Perugini et al. 2017)(see Section 2.4). Evapotranspiration from forests during the growing season regionally cools the land surface and enhances cloud cover that reduces short wave radiation reaching the land, an impact that is especially pronounced in the tropics. However, dark evergreen conifer-dominated forests have low surface reflectance, and tend to cause warming of the near surface atmosphere compared to non-forest land, especially when snow cover is present such as in boreal regions (Duveiller et al. 2018; Alkama and Cescatti 2016; Perugini et al. 2017)(*medium evidence, high agreement*).

2) Implications for water balance

Evapotranspiration by forests reduces surface runoff and erosion of soil and nutrients (Salvati et al.

2014). Planting of fast-growing species in semi-arid regions or replacing natural grasslands with forest plantations can divert soil water resources to evapotranspiration from groundwater recharge (Silveira et al. 2016; Zheng et al. 2016; Cao et al. 2016). Multiple cases are reported from China where afforestation programs, some with irrigation, without having tailored to local precipitation conditions, resulted in water shortages and tree mortality (Cao et al. 2016; Yang et al. 2014; Li et al. 2014; Feng et al. 2016). Water shortages may create long-term water conflicts (Zheng et al. 2016). However, reforestation (in particular for restoration) is also associated with improved water filtration, groundwater recharge (Ellison et al. 2017) and can reduce risk of soil erosion, flooding, and associated disasters (Lee et al. 2018; see Section 4.9).

3) Implications for biodiversity

Impacts of forest area expansion on biodiversity depend mostly on the vegetation cover that is replaced: afforestation on natural non-tree dominated ecosystems can have negative impacts on biodiversity (Abreu et al. 2017; Griffith et al. 2017; Veldman et al. 2015; Parr et al. 2014; Wilson et al. 2017; Hua et al. 2016)(see also IPCC 1.5° report (2018). Reforestation with monocultures of fast growing, non-native trees has little benefit to biodiversity (Shimamoto et al. 2018; Hua et al. 2016). There are also concerns regarding the impacts of some commonly used plantation species (e.g., Acacia and Pinus species) to become invasive (Padmanaba and Corlett 2014; Cunningham et al. 2015b).

Reforestation with mixes of native species, especially in areas that retain fragments of native forest, can support ecosystem-services and biodiversity recovery, with positive social and environmental co-benefits (Cunningham et al. 2015a; Dendy et al. 2015; Chaudhary and Kastner 2016; Huang et al. 2018; Locatelli et al. 2015b)(see Section 4.5). Even though species diversity in re-growing forests is typically lower than in primary forests, planting native or mixed species can have positive effects on biodiversity (Brockerhoff et al. 2013; Pawson et al. 2013; Thompson et al. 2014). Reforestation has been shown to improve links among existing remnant forest patches, increasing species movement, and fostering gene flow between otherwise isolated populations (Gilbert-Norton et al. 2010; Barlow et al. 2007; Lindenmayer and Hobbs 2004).

4) Implications for other ecosystem services and societies

Forest area expansion could benefit recreation and health, preservation of cultural heritage and local values and knowledge, livelihood support (via reduced resource conflicts, restoration of local resources). These social benefits could be most successfully achieved if local communities' concerns are considered (Le et al. 2012). However, these co-benefits have rarely been assessed due to a lack of suitable frameworks and evaluation tools (Baral et al. 2016).

Industrial forest management can be in conflict with needs of forest-dependent people and community-based forest management over access to natural resources (Gerber 2011; Baral et al. 2016) and/or loss of customary rights over land use (Malkamäki et al. 2018; Cotula et al. 2014). A common result is out-migration from rural areas and diminishing local uses of ecosystems (Gerber 2011). Policies promoting large-scale tree plantations gain if these are reappraised in view of potential co-benefits with several ecosystem services and local societies (Bull et al. 2006; Le et al. 2012).

Scenarios of forest-area expansion for land-based climate change mitigation

Conversion of non-forest to forest land has been discussed as a relatively cost-effective climate change mitigation option when compared to options in the energy and transport sectors (*medium evidence, medium agreement*) (de Coninck et al. 2018; Griscom et al. 2017; Fuss et al. 2018), and can have co-benefits with adaptation.

Sequestration of CO_2 from the atmosphere through forest area expansion has become a fundamental part of stringent climate change mitigation scenarios (Rogelj et al. 2018a; Fuss et al. 2018)(see e.g., Sections 2.5, 4.5, 6.2). The estimated mitigation potential ranges from about 0.5 to 10 Gt $CO_2 yr^{-1}$ (*robust evidence, medium agreement*), and depends on assumptions regarding available land and forest carbon uptake potential (Houghton 2013; Houghton and Nassikas 2017; Griscom et al. 2017; Lenton 2014; Fuss et al. 2018; Smith 2016) (see Section 2.5.1). In climate change mitigation scenarios, typically, no differentiation is made between reforestation and afforestation despite different overall environmental impacts between these two measures. Likewise, biodiversity conservation, impacts on water balances, other ecosystem services, or land-ownership as constraints when simulating forest area expansion (see Cross-Chapter Box 1 in this Chapter) tend not to be included as constraints when simulating forest area expansion.

Projected forest area increases, relative to today's forest area, range from approximately 25% in 2050 and increase to nearly 50% by 2100 (Rogelj et al. 2018a; Kreidenweis et al. 2016; Humpenoder et al. 2014). Potential adverse side-effects of such large-scale measures, especially for low-income countries, could be increasing food prices from the increased competition for land (Kreidenweis et al. 2016; Hasegawa et al. 2015, 2018; Boysen et al. 2017)(see Section 5.5). Forests also emit large amounts of biogenic volatile compounds that under some conditions contribute to the formation of atmospherically short-lived climate forcing compounds, which are also detrimental to health (Ashworth et al. 2013; Harrison et al. 2013). Recent analyses argued for an upper limit of about 5 million km^2 of land globally available for climate change mitigation through reforestation, mostly in the tropics (Houghton 2013) – with potential regional co-benefits.

Since forest growth competes for land with bioenergy crops (Harper et al. 2018)(Cross-Chapter Box 7: Bioenergy and BECCS, Chapter 6), global area estimates need to be assessed in light of alternative mitigation measures at a given location. In all forest-based mitigation efforts, the sequestration potential will eventually saturate unless the area keeps expanding, or harvested wood is either used for long-term storage products or for carbon capture and storage (Fuss et al. 2018; Houghton et al. 2015)(see Section 2.5.1). Considerable uncertainty in forest carbon uptake estimates is further introduced by potential forest losses from fire or pest outbreaks (Allen et al. 2010; Anderegg et al. 2015)(Cross-Chapter Box 3: Fire and climate change, Chapter 2). And like all land-based mitigation measures, benefits may be diminshed by land-use displacement, through trade of land-based products, especially in poor countries that experience forest loss (e.g., Africa) (Bhojvaid et al. 2016; Jadin et al. 2016).

Conclusion

Reforestation is a mitigation measure with potential co-benefits for conservation and adaptation, including biodiversity habitat, air and water filtration, flood control, enhanced soil fertility and reversal of land degradation. Potential adverse side-effects of forest area expansion depend largely on the state of the land it displaces as well as tree species selections. Active governance and planning contribute to maximising co-benefits while minimising adverse side-effects (Laestadius et al. 2011; Dinerstein et al. 2015; Veldman et al. 2017)(see Section 4.8 and Chapter 7). At large spatial scales, forest expansion is expected to lead to increased competition for land, with potentially undesirable impacts on food prices, biodiversity, non-forest ecosystems and water availability (Bryan and Crossman 2013; Boysen et al. 2017; Kreidenweis et al. 2016; Egginton et al. 2014; Cao et al. 2016; Locatelli et al. 2015a; Smith et al.

2013)

1.3.2 Land Management

1.3.2.1 Agricultural, forest and soil management

Sustainable land management (SLM) describes "*the stewardship and use of land resources, including soils, water, animals and plants, to meet changing human needs while simultaneously assuring the long-term productive potential of these resources and the maintenance of their environmental functions*" (Alemu 2016, Altieri and Nicholls 2017)(see e.g., Section 4.1.5), and includes ecological, technological and governance aspects.

The choice of SLM strategy is a function of regional context and land use types, with *high agreement* on (a combination of) choices such as agroecology (including agroforestry), conservation agriculture and forestry practices, crop and forest species diversity, appropriate crop and forest rotations, organic farming, integrated pest management, the preservation and protection of pollination services, rain water harvesting, range and pasture management, and precision agriculture systems (Stockmann et al. 2013; Ebert, 2014; Schulte et al. 2014; Zhang et al. 2015; Sunil and Pandravada 2015; Poeplau and Don 2015; Agus et al. 2015; Keenan 2015; MacDicken et al. 2015; Abberton et al. 2016). Conservation agriculture and forestry uses management practises with minimal soil disturbance such as no tillage or minimum tillage, permanent soil cover with mulch combined with rotations to ensure a permanent soil surface, or rapid regeneration of forest following harvest (Hobbs et al. 2008; Friedrich et al. 2012). Vegetation and soils in forests and woodland ecosystems play a crucial role in regulating critical ecosystem processes, therefore reduced deforestation together with sustainable forest management are integral to SLM (FAO 2015b; see Section 4.8). In some circumstances, increased demand for forest products can also lead to increased management of carbon storage in forests (Favero and Mendelsohn 2014). Precision agriculture is characterised by a "management system that is information and technology based, is site specific and uses one or more of the following sources of data: soils, crops, nutrients, pests, moisture, or yield, for optimum profitability, sustainability, and protection of the environment" (USDA 2007)(see also Cross-Chapter Box 6: Agricultural intensification, Chapter 5). The management of protected areas that reduce deforestation also plays an important role in climate change mitigation and adaptation while delivering numerous ecosystem services and sustainable development benefits (Bebber and Butt 2017). Similarly, when managed in an integrated and sustainable way, peatlands are also known to provide numerous ecosystem services, as well as socio-economic and mitigation and adaptation benefits (Ziadat et al. 2018).

Biochar is an organic compound used as soil amendment and is believed to be potentially an important global resource for mitigation. Enhancing the carbon content of soil and/or use of biochar (see Chapter 4) have become increasingly important as a climate change mitigation option with possibly large co-benefits for other ecosystem services. Enhancing soil carbon storage and the addition of biochar can be practised with limited competition for land, provided no productivity/yield loss and abundant unused biomass, but evidence is limited and impacts of large scale application of biochar on the full GHG balance of soils, or human health are yet to be explored (Gurwick et al. 2013; Lorenz and Lal 2014; Smith 2016).

1.3.3 Value chain management

1.3.3.1 Supply management

Food losses from harvest to retailer. Approximately one third of losses and waste in the food system occurs between crop production and food consumption, increasing substantially if losses in livestock production and overeating are included (Gustavsson et al. 2011; Alexander et al. 2017). This includes on-farm losses, farm to retailer losses, as well retailer and consumer losses (see Section 1.3.3.2).

Post-harvest food loss on farm and from farm to retailer is a widespread problem, especially in developing countries (Xue et al. 2017), but are challenging to quantify. For instance, averaged for eastern and southern Africa an estimated 10–17% of annual grain production is lost (Zorya et al. 2011). Across 84 countries and different time periods, annual median losses in the supply chain before retailing were estimated at about 28 kg per capita for cereals or about 12 kg per capita for eggs and dairy products (Xue et al. 2017). For the year 2013, losses prior to the reaching retailers were estimated at 20% (dry weight) of the production amount (22% wet weight) (Gustavsson et al. 2011; Alexander et al. 2017). While losses of food cannot be realistically reduced to zero, advancing harvesting technologies (Bradford et al. 2018; Affognon et al. 2015), storage capacity (Chegere 2018) and efficient transportation could all contribute to reducing these losses with co-benefits for food availability, the land area needed for food production and related GHG emissions.

Stability of food supply, transport and distribution. Increased climate variability enhances fluctuations in world food supply and price variability (Warren 2014; Challinor et al. 2015; Elbehri et al. 2017). "Food price shocks" need to be understood regarding their transmission across sectors and borders and impacts on poor and food insecure populations, including urban poor subject to food deserts and inadequate food accessibility (Widener et al. 2017; Lehmann et al. 2013; LE 2016; FAO 2015b). Trade can play an important stabilising role in food supply, especially for regions with agro-ecological limits to production, including water scarce regions, as well as regions that experience short term production variability due to climate, conflicts or other economic shocks (Gilmont 2015; Marchand et al. 2016). Food trade can either increase or reduce the overall environmental impacts of agriculture (Kastner et al. 2014). Embedded in trade are virtual transfers of water, land area, productivity, ecosystem services, biodiversity, or nutrients (Marques et al. 2019; Wiedmann and Lenzen 2018; Chaudhary and Kastner 2016) with either positive or negative implications(Chen et al. 2018; Yu et al. 2013). Detrimental consequences in countries in which trade dependency may accentuate the risk of food shortages from foreign production shocks could be reduced by increasing domestic reserves or importing food from a diversity of suppliers (Gilmont 2015; Marchand et al. 2016).

Climate mitigation policies could create new trade opportunities (e.g., biomass) (Favero and Massetti 2014) or alter existing trade patterns. The transportation GHG-footprints of supply chains may be causing a differentiation between short and long supply chains (Schmidt et al. 2017) that may be influenced by both economics and policy measures (see Section 5.4). In the absence of sustainable practices and when the ecological footprint is not valued through the market system, trade can also exacerbate resource exploitation and environmental leakages, thus weakening trade mitigation contributions (Dalin and Rodríguez-Iturbe 2016; Mosnier et al. 2014; Elbehri et al. 2017). Ensuring stable food supply while pursuing climate mitigation and adaptation will benefit from evolving trade rules and policies that allow internalisation of the cost of carbon (and costs of other vital resources such as water, nutrients). Likewise, future climate change mitigation policies would gain from measures designed to internalise the environmental costs of resources and the benefits of ecosystem services (Elbehri et al. 2017; Brown et al., 2007).

1.3.3.2 Demand management

Dietary change. Demand-side solutions to climate mitigation are an essential complement to supply-side, technology and productivity driven solutions (Creutzig et al. 2016; Bajželj et al. 2014; Erb et al. 2016b; Creutzig et al. 2018)(see Sections 5.5.1, 5.5.2)(*high confidence*). The environmental impacts of the animal-rich “western diets” are being examined critically in the scientific literature (Hallström et al. 2015; Alexander et al. 2016b; Alexander et al. 2015; Tilman and Clark 2014; Aleksandrowicz et al. 2016; Poore and Nemecek 2018)(see Section 5.4.6). For example, if the average diet of each country were consumed globally, the agricultural land area needed to supply these diets would vary 14-fold, due to country differences in ruminant protein and calorific intake (-55% to +178% compared to existing cropland areas). Given the important role enteric fermentation plays in methane (CH4) emissions, a number of studies have examined the implications of lower animal diets (Swain et al. 2018; Röös et al. 2017; Rao et al. 2018). Reduction of animal protein intake has been estimated to reduce global green water (from precipitation) use by 11% and blue water (from rivers, lakes, groundwater) use by 6% (Jalava et al. 2014). By avoiding meat from producers with above-median GHG emissions and halving animal-product intake, consumption change could free-up 21 million km^2 of agricultural land and reduce GHG emissions by nearly 5 Gt CO_2-eq yr^{-1} or up to 10.4 Gt CO_2-eq yr^{-1} when vegetation carbon uptake is considered on the previously agricultural land (Poore and Nemecek 2018, 2019).

Diets can be location and community specific, are rooted in culture and traditions while responding to changing lifestyles driven for instance by urbanisation and changing income. Changing dietary and consumption habits would require a combination of non-price (government procurement, regulations, education and awareness raising) and price (Juhl and Jensen 2014) incentives to induce consumer behavioural change with potential synergies between climate, health and equity (addressing growing global nutrition imbalances that emerge as undernutrition, malnutrition, and obesity) (FAO 2018b).

Reduced waste and losses in the food demand system. Global averaged per capita food waste and loss (FWL) have increased by 44% between 1961 and 2011 (Porter et al. 2016) and are now around 25–30% of global food produced (Kummu et al. 2012)(Alexander et al. 2017). Food waste occurs at all stages of the food supply chain from the household to the marketplace (Parfitt et al. 2010) and is found to be larger at household than at supply chain levels. A meta-analysis of 55 studies showed that the highest share of food waste was at the consumer stage (43.9% of total) with waste increasing with per capita GDP for high income countries until a plateaux at about 100 kg cap^{-1} yr^{-1} (around 16% of food consumption) above about 70 000 USD cap^{-1} (van der Werf and Gilliland 2017; Xue et al. 2017). Food loss from supply chains tends to be more prevalent in less developed countries where inadequate technologies, limited infrastructure, and imperfect markets combine to raise the share of the food production lost before use.

There are several causes behind food waste including economics (cheap food), food policies (subsidies) as well as individual behaviour (Schanes et al. 2018). Household level food waste arises from overeating or overbuying (Thyberg and Tonjes 2016). Globally, overconsumption was found to waste 9–10% of food bought (Alexander et al. 2017).

Solutions to FWL thus need to address technical and economic aspects. Such solutions would benefit from more accurate data on the loss-source, -magnitude and -causes along the food supply chain. In the long run, internalising the cost of food waste into the product price would more likely induce a shift in consumer behaviour towards less waste and more nutritious, or alternative, food intake (FAO 2018b). Reducing FWL would bring a range of benefits for health, reducing pressures on land, water and nutrients, lowering emissions and safeguarding food security. Reducing food waste by 50% would generate net emissions reductions in the range of 20 to 30% of total food-sourced GHGs (Bajželj et al. 2014). The SDG 12 ("Ensure sustainable consumption and production patterns") calls for per capita global food waste to be reduced by one half at the retail and consumer level, and reducing food losses along production and supply chains by 2030.

1.3.4 Risk management

Risk management refers to plans, actions, strategies or policies to reduce the likelihood and/or magnitude of adverse potential consequences, based on assessed or perceived risks' Insurance and early warning systems are examples of risk management, but risk can also be reduced (or resilience enhanced) through a broad set of options ranging from seed sovereignty, livelihood diversification, to reducing land loss through urban sprawl. Early warning systems support farmer decision making on management strategies (see Section 1.2) and are a good example of an adaptation measure with mitigation co-benefits such as reducing carbon losses (see Section 1.3.6). Primarily designed to avoid yield losses, early warning systems also support fire management strategies in forest ecosystems, which prevents financial as well as carbon losses (de Groot et al. 2015). Given that over recent decades on average around 10% of cereal production was lost through extreme weather events (Lesk et al. 2016), where available and affordable, insurance can buffer farmers and foresters against the financial losses incurred through such weather and other (fire, pests) extremes (Falco et al. 2014)(see Section 7.2, 7.4). Decisions to take up insurance are influenced by a range of factors such as the removal of subsidies or targeted education (Falco et al. 2014). Enhancing access and affordability of insurance in low-income countries is a specific objective of the UNFCCC (Linnerooth-Bayer and Mechler 2006). A global mitigation co-benefit of insurance schemes may also include incentives for future risk reduction (Surminski and Oramas-Dorta 2014).

1.3.5 Economics of land-based mitigation pathways: Costs versus benefits of early action under uncertainty

The overarching societal costs associated with GHG emissions and the potential implications of mitigation activities can be measured by various metrics (cost-benefit analysis, cost effectiveness analysis) at different scales (project, technology, sector or the economy) (IPCC 2018; section 1.4). The Social Cost of Carbon (SCC), measures the total net damages of an extra metric ton of CO_2 emissions due to the associated climate change (Nordhaus 2014; Pizer et al. 2014). Both negative and positive impacts are monetised and discounted to arrive at the net value of consumption loss. As the SCC depends on discount rate assumptions and value judgements (e.g., relative weight given to current vs. future generations), it is not a straightforward policy tool to compare alternative options. At the sectoral level, marginal abatement cost curves (MACCs) are widely used for the assessment of costs related to GHG emissions reduction. MACCs measure the cost of reducing one more GHG unit and are either expert-based or model-derived and offer a range of approaches and assumptions on discount rates or available abatement technologies (Kesicki 2013). In land-based sectors, Gillingham and Stock (2018) reported short term static abatement costs for afforestation of between 1 and 10 USD 2017/tCO_2, soil management at 57 and livestock management at 71 USD 2017/tCO_2. MACCs are more reliable when used to rank alternative options compared to a baseline (or business as usual) rather than offering absolute numerical measures (Huang et al. 2016). The economics of land-based mitigation options encompass also the "costs of inaction" that arise either from the economic damages due to continued accumulation of GHGs in the atmosphere and from the diminution in value of ecosystem services or the cost of their restoration where feasible (Rodriguez-Labajos 2013; Ricke et al. 2018). Overall, it remains challenging to estimate the costs of alternative mitigation options owing to the context- and scale specific interplay between multiple drivers (technological, economic, and socio-cultural) and enabling policies and institutions (IPCC 2018)(section 1.4).

The costs associated with mitigation (both project-linked such as capital costs or land rental rates or sometimes social costs) generally increase with stringent mitigation targets and over time. Sources of uncertainty include the future availability, cost and performance of technologies (Rosen and Guenther

2015; Chen et al. 2016) or lags in decision making, which have been demonstrated by the uptake of land use and land utilisation policies (Alexander et al. 2013; Hull et al. 2015; Brown et al. 2018b). There is growing evidence of significant mitigation gains through conservation, restoration and improved land management practices (Griscom et al. 2017; Kindermann et al. 2008; Golub et al. 2013; Favero et al. 2017)(see Chapter 4 and Chapter 6), but the mitigation cost efficiency can vary according to region and specific ecosystem (Albanito et al. 2016). Recent model developments that treat process-based, human-environment interactions have recognised feedbacks that reinforce or dampen the original stimulus for land use change (Robinson et al. 2017; Walters and Scholes 2017). For instance, land mitigation interventions that rely on large-scale, land use change (i.e., afforestation) would need to account for the rebound effect (which dampens initial impacts due to feedbacks) in which raising land prices also raises the cost of land-based mitigation (Vivanco et al. 2016). Although there are few direct estimates, indirect assessments strongly point to much higher costs if action is delayed or limited in scope (*medium confidence*). Quicker response options are also needed to avoid loss of high-carbon ecosystems and other vital ecosystem services that provide multiple services that are difficult to replace (peatlands, wetlands, mangroves, forests) (Yirdaw et al. 2017; Pedrozo-Acuña et al. 2015). Delayed action would raise relative costs in the future or could make response options less feasible (Goldstein et al. 2019; Butler et al. 2014)(*medium confidence*).

1.3.6 Adaptation measures and scope for co-benefits with mitigation

Adaptation and mitigation have generally been treated as two separate discourses, both in policy and practice with mitigation addressing cause and adaptation dealing with the consequences of climate change (Hennessey et al. 2017). While adaptation (e.g., reducing flood risks) and mitigation (e.g., reducing non-CO_2 emissions from agriculture) may have different objectives and operate at different scales, they can also generate joint outcomes (Locatelli et al. 2015b) with adaptation generating mitigation co-benefits. Seeking to integrate strategies for achieving adaptation and mitigation goals is attractive in order to reduce competition for limited resources and trade-offs (Lobell et al. 2013; Berry et al. 2015; Kongsager and Corbera 2015). Moreover, determinants that can foster adaptation and mitigation practices are similar. These tend to include available technology and resources, and credible information for policy makers to act on (Yohe 2001).

Four sets of mitigation-adaptation interrelationships can be distinguished: 1) mitigation actions that can result in adaptation benefits; 2) adaptation actions that have mitigation benefits; 3) processes that have implications for both adaptation and mitigation; 4) strategies and policy processes that seek to promote an integrated set of responses for both adaptation and mitigation (Klein et al. 2007). A high level of adaptive capacity is a key ingredient to developing successful mitigation policy. Implementing mitigation action can result in increasing resilience especially if it is able to reduce risks. Yet, mitigation and adaptation objectives, scale of implementation, sector and even metrics to identify impacts tend to differ (Ayers and Huq 2009), and institutional setting, often does not enable an environment where synergies are sought (Kongsager et al. 2016). Trade-offs between adaptation and mitigation exist as well and need to be understood (and avoided) to establish win-win situations (Porter et al. 2014; Kongsager et al. 2016).

Forestry and agriculture offer a wide range of lessons for the integration of adaptation and mitigation actions given the vulnerability of forest ecosystems or cropland to climate variability and change (Keenan 2015; Gaba et al. 2015)(see Section 5.6, 4.8). Increasing adaptive capacity in forested areas has the potential to prevent deforestation and forest degradation (Locatelli et al. 2011). Reforestation projects, if well managed, can increase community economic opportunities that encourage conservation (Nelson and de Jong 2003), build capacity through training of farmers and installation of multifunctional plantations with income generation (Reyer et al. 2009), strengthen local institutions (Locatelli et al. 2015a) and increase cash-flow to local forest stakeholders from foreign donors (West 2016). A forest plantation that sequesters carbon for mitigation can also reduce water availability to downstream populations and heighten their vulnerability to drought. Inversely, not recognising mitigation in adaptation projects may yield adaptation measures that increase greenhouse gas emissions, a prime example of 'maladaptation'. Analogously, 'mal-mitigation' would result in reducing greenhouse gas emissions, but increasing vulnerability (Barnett and O'Neill 2010; Porter et al. 2014). For instance, the cost of pursuing large scale adaptation and mitigation projects has been associated with higher failure risks, onerous transactions costs and the complexity of managing big projects (Swart and Raes 2007).

Adaptation encompasses both biophysical and socio-economic vulnerability and underlying causes (informational, capacity, financial, institutional, and technological; Huq et al. 2014) and it is increasingly linked to resilience and to broader development goals (Huq et al. 2014). Adaptation measures can increase performance of mitigation projects under climate change and legitimise mitigation measures through the more immediately felt effects of adaptation (Locatelli et al. 2011; Campbell et al. 2014; Locatelli et al. 2015b). Effective climate policy integration in the land sector is expected to gain from 1) internal policy coherence between adaptation and mitigation objectives, 2) external climate coherence between climate change and development objectives, 3) policy integration that favours vertical governance structures to foster effective mainstreaming of climate change into sectoral policies, and 4) horizontal policy integration through overarching governance structures to enable cross-sectoral co-ordination (see Sections 1.4, 7.4).

1.4 Enabling the response

Climate change and sustainable development are challenges to society that require action at local, national, transboundary and global scales. Different time-perspectives are also important in decision making, ranging from immediate actions to long-term planning and investment. Acknowledging the systemic link between food production and consumption, and land-resources more broadly is expected to enhance the success of actions (Bazilian et al. 2011; Hussey and Pittock 2012). Because of the complexity of challenges and the diversity of actors involved in addressing these challenges, decision making would benefit from a portfolio of policy instruments. Decision making would also be facilitated by overcoming barriers such as inadequate education and funding mechanisms, as well as integrating international decisions into all relevant (sub)national sectoral policies (see Section 7.4).

'Nexus thinking' emerged as an alternative to the sector-specific governance of natural resource use to achieve global securities of water (D'Odorico et al. 2018), food and energy (Hoff 2011; Allan et al. 2015), and also to address biodiversity concerns (Fischer et al. 2017). Yet, there is no agreed definition of "nexus" nor a uniform framework to approach the concept, which may be land-focused (Howells et al. 2013), water-focused (Hoff 2011) or food-centred (Ringler and Lawford 2013; Biggs et al. 2015). Significant barriers remain to establish nexus approaches as part of a wider repertoire of responses to global environmental change, including challenges to cross-disciplinary collaboration, complexity, political economy and the incompatibility of current institutional structures (Hayley et al. 2015; Wichelns 2017)(see Section 7.5.6, 7.6.2).

1.4.1 Governance to enable the response

Governance includes the processes, structures, rules and traditions applied by formal and informal actors including governments, markets, organisations, and their interactions with people. Land governance actors include those affecting policies and markets, and those directly changing land use (Hersperger et al.

2010). The former includes governments and administrative entities, large companies investing in land, non-governmental institutions and international institutions. It also includes UN agencies that are working at the interface between climate change and land management, such as the FAO and the World Food Programme that have *inter alia* worked on advancing knowledge to support food security through the improvement of techniques and strategies for more resilient farm systems. Farmers and foresters directly act on land (actors in proximate causes) (Hersperger et al. 2010)(see also Chapter 7.).

Policy design and formulation has often been strongly sectoral. For example, agricultural policy might be concerned with food security, but have little concern for environmental protection or human health. As food, energy and water security and the conservation of biodiversity rank highly on the Agenda 2030 for Sustainable Development, the promotion of synergies between and across sectoral policies is important (IPBES 2018a). This can also reduce the risks of anthropogenic climate forcing through mitigation, and bring greater collaboration between scientists, policy makers, the private sector and land managers in adapting to climate change (FAO 2015a). Polycentric governance (see Section 7.6) has emerged as an appropriate way of handling resource management problems, in which the decision- making centers take account of one another in competitive and cooperative relationships and have recourse to conflict resolution mechanisms (Carlisle and Gruby 2017). Polycentric governance is also multi-scale and allows the interaction between actors at different levels (local, regional, national, and global) in managing common pool resources such as forests or aquifers.

Implementation of systemic, nexus approaches has been achieved through socio-ecological systems (SES) frameworks that emerged from studies of how institutions affect human incentives, actions and outcomes (Ostrom and Cox 2010). Recognition of the importance of SES laid the basis for alternative formulations to tackle the sustainable management of land resources focusing specifically on institutional and governance outcomes (Lebel et al. 2006; Bodin 2017). The SES approach also addresses the multiple scales in which the social and ecological dimensions interact (Veldkamp et al. 2011; Myers et al. 2016; Azizi et al. 2017) (see Section 6.1).

Adaptation or resilience pathways within the SES frameworks require several attributes, including indigenous and local knowledge (ILK) and trust building for deliberative decision making and effective collective action, polycentric and multi-layered institutions and responsible authorities that pursue just distributions of benefits to enhance the adaptive capacity of vulnerable groups and communities (Lebel et al. 2006). The nature, source, and mode of knowledge generation are critical to ensure that sustainable solutions are community-owned and fully integrated within the local context (Mistry and Berardi 2016; Schneider and Buser 2018). Integrating ILK with scientific information is a prerequisite for such community-owned solutions (see Cross-Chapter Box 13: ILK, Chapter 7). ILK is context-specific, transmitted orally or through imitation and demonstration, adaptive to changing environments, collectivised through a shared social memory (Mistry and Berardi 2016). ILK is also holistic since indigenous people do not seek solutions aimed at adapting to climate change alone, but instead look for solutions to increase their resilience to a wide range of shocks and stresses (Mistry and Berardi 2016). ILK can be deployed in the practice of climate governance especially at the local level where actions are informed by the principles of decentralisation and autonomy (Chanza and de Wit 2016). ILK need not be viewed as needing confirmation or disapproval by formal science, but rather it can complement scientific knowledge (Klein et al. 2014).

The capacity to apply individual policy instruments and policy mixes is influenced by governance modes. These modes include hierarchical governance that is centralised and imposes policy through top-down measures, decentralised governance in which public policy is devolved to regional or local government, public-private partnerships that aim for mutual benefits for the public and private sectors and self or private governance that involves decisions beyond the realms of the public sector (IPBES 2018a). These governance modes provide both constraints and opportunities for key actors that affect the effectiveness, efficiency and equity of policy implementation.

1.4.2 Gender agency as a critical factor in climate and land sustainability outcomes

Environmental resource management is not gender neutral. Gender is an essential variable in shaping ecological processes and change, building better prospects for livelihoods and sustainable development (Resurrección 2013)(see Cross-Chapter Box 11: Gender, Chapter 7). Entrenched legal and social structures and power relations constitute additional stressors that render women's experience of natural resources disproportionately negative than men. Socio-economic drivers and entrenched gender inequalities affect land-based management (Agarwal 2010). The intersections between climate change, gender and climate adaptation takes place at multiple scales: household, national, international, and adaptive capacities are shaped through power and knowledge.

Germaine to the gender inequities are the unequal access to land-based resources. Women play a significant role in agriculture (Boserup 1989; Darity 1980) and rural economies globally (FAO 2011), but are well below their share of labour in agriculture globally (FAO 2011). In 59% of 161 surveyed countries, customary, traditional and religious practices hinder women land rights (OECD 2014). Moreover, women typically shoulder disproportionate responsibility for unpaid domestic work including care-giving activities (Beuchelt and Badstue 2013) and the provision of water and firewood (UNEP 2016). Exposure to violence restricts in large regions their mobility for capacity-building activities and productive work outside the home (Day et al. 2005; UNEP 2016). Large-scale development projects can erode rights, and lead to over-exploitation of natural resources. Hence, there are cases where reforms related to land-based management, instead of enhancing food security, have tended to increase the vulnerability of both women and men and reduce their ability to adapt to climate change (Pham et al. 2016). Access to, and control over, land and land-based resources is essential in taking concrete action to land based mitigation, and inadequate access can affect women's rights and participation in land governance and management of productive assets.

Timely information, such as from early warning systems, is critical in managing risks, disasters, and land degradation, and in enabling land-based adaptation. Gender, household resources and social status, are all determinants that influence the adoption of land-based strategies (Theriault et al. 2017). Climate change is not a lone driver in the marginalisation of women, their ability to respond swiftly to its impacts will depend on other socio-economic drivers that may help or hinder action towards adaptive governance. Empowering women and removing gender-based inequities constitutes a mechanism for greater participation in the adoption of sustainable practices of land management (Mello and Schmink 2017). Improving women's access to land (Arora-Jonsson 2014) and other resources (water) and means of economic livelihoods (such as credit and finance) are the prerequisites to enable women to participate in governance and decision-making structures (Namubiru-Mwaura 2014). Still women are not a homogenous group, and distinctions through elements of ethnicity, class, age and social status, require a more nuanced approach and not a uniform treatment through vulnerability lenses only. An intersectional approach that accounts for various social identifiers under different situation of power (Rao 2017) is considered suitable to integrate gender into climate change research and helps to recognise overlapping and interdependent systems of power (Djoudi et al. 2016; Kaijser and Kronsell 2014; Moosa and Tuana 2014; Thompson-Hall et al. 2016).

1.4.3 Policy Instruments

Policy instruments enable governance actors to respond to environmental and societal challenges through policy action. Examples of the range of policy instruments available to public policy-makers is discussed below based on four categories of instruments: legal and regulatory instruments, rights-based instruments and customary norms, economic and financial instruments and social and cultural instruments.

1.4.3.1 Legal & regulatory instruments

Legal and regulatory instruments deal with all aspects of intervention by public policy organisations to correct market failures, expand market reach, or intervene in socially relevant areas with inexistent markets. Such instruments can include legislation to limit the impacts of intensive land management, for example, protecting areas that are susceptible to nitrate pollution or soil erosion. Such instruments can also set standards or threshold values, for example, mandated water quality limits, organic production standards, or geographically defined regional food products. Legal and regulatory instruments may also define liability rules, for example, where environmental standards are not met, as well as establishing long-term agreements for land resource protection with land owners and land users.

1.4.3.2 Economic and financial instruments

Economic (such as taxes, subsidies) and financial (weather-index insurance) instruments deal with the many ways in which public policy organisations can intervene in markets. A number of instruments are available to support climate mitigation actions including public provision, environmental regulations, creating property rights and markets, using markets (Sterner 2003). Market-based policies such as carbon taxes, fuel taxes, cap and trade systems or green payments have been promoted (mostly in industrial economies) to encourage markets and businesses to contribute to climate mitigation, but their effectiveness to date has not always matched expectations (Grolleau et al. 2016) (see Section 7.4.4). Market-based instruments in ecosystem services generate both positive (incentives for conservation), but also negative environmental impacts, and also push food prices up or increase price instability (Gómez-Baggethun and Muradian 2015; Farley and Voinov 2016). Footprint labels can be an effective means of shifting consumer behaviour. However, private labels focusing on a single metric (e.g., carbon) may give misleading signals if they target a portion of the life cycle (e.g., transport) (Appleton 2009) or ignore other ecological indicators (water, nutrients, biodiversity)(van Noordwijk and Brussaard 2014).

Effective and durable, market-led responses for climate mitigation depend on business models that internalise the cost of emissions into economic calculations. Such "business transformation" would itself require integrated policies and strategies that aim to account for emissions in economic activities (Biagini and Miller 2013; Weitzman 2014; Eidelwein et al. 2018). International initiatives such as REDD+ and agricultural commodity roundtables (beef, soybeans, palm oil, sugar) are expanding the scope of private sector participation in climate mitigation (Nepstad et al. 2013), but their impacts have not always been effective (Denis et al. 2014). Payments for environmental services (PES) defined as "*voluntary transactions between service users and service providers that are conditional on agreed rules of natural resource management for generating offsite services*" (Wunder 2015) have not been widely adopted and have not yet been demonstrated to deliver as effectively as originally hoped (Börner et al. 2017)(see Sections 7.4, 7.5). PES in forestry were shown to be effective only when coupled with appropriate regulatory measures (Alix-Garcia and Wolff 2014). Better designed and expanded PES schemes would encourage integrated soil-water-nutrient management packages (Stavi et al. 2016), services for pollinator protection (Nicole 2015), water use governance under scarcity and engage both public and private actors (Loch et al. 2013). Effective PES also requires better economic metrics to account for human-directed losses in terrestrial ecosystems and to food potential, and to address market failures or externalities unaccounted for in market valuation of ecosystem services.

Resilient strategies for climate adaptation can rely on the construction of markets through social networks as in the case of livestock systems (Denis et al. 2014) or when market signals encourage adaptation through land markets or supply chain incentives for sustainable land management practices (Anderson et al. 2018). Adequate policy (through regulations, investments in research and development or support to social capabilities) can support private initiatives for effective solutions to restore degraded lands (Reed and Stringer 2015), or mitigate against risk and to avoid shifting risks to the public (Biagini and Miller 2013). Governments, private business, and community groups could also partner to develop sustainable production codes (Chartres and Noble 2015), and in co-managing land-based resources (Baker and Chapin 2018), while private-public partnerships can be effective mechanisms in deploying infrastructure to cope with climatic events (floods) and for climate-indexed insurance (Kunreuther 2015). Private initiatives that depend on trade for climate adaptation and mitigation require reliable trading systems that do not impede climate mitigation objectives (Elbehri et al 2015; Mathews 2017).

1.4.3.3 Rights-based instruments and customary norms

Rights-based instruments and customary norms deal with the equitable and fair management of land resources for all people (IPBES 2018a). These instruments emphasise the rights in particular of indigenous peoples and local communities, including for example, recognition of the rights embedded in the access to, and use of, common land. Common land includes situations without legal ownership (e.g., hunter-gathering communities in south America or Africa and bushmeat), where the legal ownership is distinct from usage rights (Mediterranean transhumance grazing systems), or mixed ownership-common grazing systems (e.g., Crofting in Scotland). A lack of formal (legal) ownership has often led to the loss of access rights to land, where these rights were also not formally enshrined in law, which especially effects indigenous communities, for example, deforestation in the Amazon basin. Overcoming the constraints associated with common-pool resources (forestry, fisheries, water) are often of economic and institutional nature (Hinkel et al. 2014) and require tackling the absence or poor functioning of institutions and the structural constraints that they engender through access and control levers using policies and markets and other mechanisms (Schut et al. 2016). Other examples of rights-based instruments include the protection of heritage sites, sacred sites and peace parks (IPBES 2018a). Rights-based instruments and customary norms are consistent with the aims of international and national human rights, and the critical issue of liability in the climate change problem.

1.4.3.4 Social and cultural norms

Social and cultural instruments are concerned with the communication of knowledge about conscious consumption patterns and resource-effective ways of life through awareness raising, education and communication of the quality and the provenance of land-based products. Examples of the latter include consumption choices aided by ecolabelling (see 1.4.3.2) and certification. Cultural indicators (such as social capital, cooperation, gender equity, women's knowledge, socio-ecological mobility) contribute to the resilience of social-ecological systems (Sterling et al. 2017). Indigenous communities (such as the Inuit and Tsleil Waututh Nation in Canada) that continue to maintain traditional foods exhibit greater dietary quality and adequacy (Sheehy et al. 2015). Social and cultural instruments also include approaches to self-regulation and voluntary agreements, especially with respect to environmental management and land resource use. This is becoming especially important in the increasingly important domain of corporate social responsibility (Halkos and Skouloudis 2016).

1.5 The interdisciplinary nature of the SRCCL

Assessing the land system in view of the multiple challenges that are covered by the SRCCL requires a broad, inter-disciplinary perspective. Methods, core concepts and definitions are used differently in different sectors, geographic regions, and across academic communities addressing land systems, and these concepts and approaches to research are also undergoing a change in their interpretation through time. These differences reflect varying perspectives, in nuances or emphasis, on land as component of the climate and socio-economic systems. Because of its inter-disciplinary nature, the SRCCL can take advantage of these varying perspectives and the diverse methods that accompany them. That way, the report aims to support decision makers across sectors and world regions in the interpretation of its main findings and support the implementation of solutions.

Frequently Asked Questions

FAQ 1.1 What are the approaches to study the interactions between land and climate?

Climate changes shapes the way land is able to support supply of food and water for humans. At the same time the land surface interacts with the overlying atmosphere, thus human modifications of land use, land cover and urbanisation affect global, regional and local climate. The complexity of the land-climate interactions requires multiple study approaches embracing different spatial and temporal scales. Observations of land atmospheric exchanges, such as of carbon, water, nutrients and energy can be carried out at leaf level and soil with gas exchange systems, or at canopy scale by means of micrometeorological techniques (i.e. eddy covariance). At regional scale, atmospheric measurements by tall towers, aircraft and satellites can be combined with atmospheric transport models to obtain spatial explicit maps of relevant greenhouse gases fluxes. At longer temporal scale (> 10 years) other approaches are more effective such as tree ring chronologies, satellite records, population and vegetation dynamics and isotopic studies. Models are important to bring information from measurement together and to extend the knowledge in space and time, including the exploration of scenarios of future climate-land interactions.

FAQ 1.2 How region-specific are the impact of different land-based adaptation and mitigation options?

Land based adaptation and mitigation options are closely related to regional specific features for several reasons. Climate change has a definite regional pattern with some regions already suffering from enhanced climate extremes and others being impacted little, or even benefiting. From this point of view increasing confidence in regional climate change scenarios is becoming a critical step forward towards the implementation of adaptation and mitigation options. Biophysical and socio-economic impacts of climate change depend on the exposures of natural ecosystems and economic sectors, which are again specific to a region, reflecting regional sensitivities due to governance. The overall responses in terms of adaptation or mitigation capacities to avoid and reduce vulnerabilities and enhance adaptive capacity, depend on institutional arrangements, socio-economic conditions, and implementation of policies, many of them having definite regional features. However global drivers, such as agricultural demand, food prices, changing dietary habits associated with rapid social transformations (i.e. urban versus rural, meat versus vegetarian) may interfere with regional specific policies for mitigation and adaptation options and require the global level to be addressed.

FAQ 1.3 What is the difference between desertification and land degradation? And where are they happening?

The difference between land degradation and desertification is geographic. Land degradation is a general term used to describe a negative trend in land condition caused by direct or indirect human-induced processes (including anthropogenic climate change). Degradation can be identified by the long-term reduction or loss in biological productivity, ecological integrity or value to humans. Desertification is land degradation when it occurs in arid, semi-arid, and dry sub-humid areas, which are also called drylands. Contrary to some perceptions, desertification is not the same as the expansion of deserts. Desertification is also not limited to irreversible forms of land degradation.

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Supplementary Material

Table SM.1.1 Observations related to variables indicative of land management, and their uncertainties

LM-related process	Observations methodology	Scale of observations (space and time)	Uncertainties[2]	Pros and cons	Select literature
GHG emissions	Micrometeorological fluxes (CO_2) Micrometeorological fluxes (CH_4) Micrometeorological fluxes (N_2O)	1-10 ha 0.5hr- >10 y	5-15% 10-40% 20-50%	Pros Larger footprints Continuous monitoring Less disturbance on monitored system Detailed protocols Cons Limitations by fetch and turbulence scale Not all trace gases	(Richardson et al. 2006; Luyssaert et al. 2007; Foken and Napo 2008; Mauder et al. 2013; Peltola et al. 2014; Wang et al. 2015; Rannik et al. 2015; Campioli et al. 2016; Rannik et al. 2016; Wang et al. 2017a; Brown and Wagner-Riddle 2017; Desjardins et al. 2018)
	Soil chambers (CO_2) Soil chambers (CH_4) Soil chambers (N_2O)	0.01-1 ha 0.5hr - 1 y	5%-15% 5%- 25% 53%- 100%[3]	Pros Relatively inexpensive Possibility of manipulation experiments Large range of trace gases Cons Smaller footprint Complicate upscaling Static pressure interference	(Vargas and Allen 2008; Lavoie et al. 2015; Barton et al. 2015; Dossa et al. 2015; Ogle et al. 2016; Pirk et al. 2016; Morin et al. 2017; Lammirato et al. 2018)
	Atmospheric inversions (CO_2) Atmospheric inversions	Regional 1->10 y	50% 3-8%	Pros Integration on large scale Attribution detection (with	(Wang et al. 2017b) (Pison et al. 2018)

[2] FOOTNOTE: Uncertainty here is defined as the coefficient of variation CV. In the case of micrometeorological fluxes they refer to random errors and CV of daily average

[3] FOOTNOTE: > 100 for fluxes less than 5g N_2O-N ha^{-1} d^{-1}

	(CH_4)			14C) Rigorously derived uncertainty <u>Cons</u> Not suited at farm scale Large high precision observation network required	
Carbon balance	Soil carbon point measurements	0.01ha-1ha >5 y	5-20%	<u>Pros</u> Easy protocol Well established analytics <u>Cons</u> Need high number of samples for upscaling Detection limit is high	(Chiti et al. 2018; Castaldi et al. 2018; Chen et al. 2018; Deng et al. 2018)
	Biomass measurements	0.01ha – 1ha 1-5 y	2-8%	<u>Pros</u> Well established allometric equations High accuracy at plot level <u>Cons</u> Difficult to scale up Labour intensive	(Pelletier et al. 2012; Henry et al. 2015; Vanguelova et al. 2016; Djomo et al. 2016; Forrester et al. 2017; Xu et al. 2017Marziliano et al. 2017; Clark et al. 2017; Disney et al. 2018; Urbazaev et al. 2018; Paul et al. 2018)
Water balance	Soil moisture (IoT sensors, Cosmic rays, Thermo-optical sensing etc.)	0.01ha – regional 0.5hr-<1y	3-5% vol	<u>Pros</u> New technology Big data analytics Relatively inexpensive <u>Cons</u> Scaling problems	(Yu et al. 2013; Zhang and Zhou 2016; Iwata et al. 2017; McJannet et al. 2017; Karthikeyan et al. 2017; Iwata et al. 2017; Cao et al. 2018;Amaral et al. 2018; Moradizadeh and Saradjian 2018; Strati et al. 2018)
	Evapotranspiration	0.01ha – Regional 0.5hr->10y	10-20%	<u>Pros</u> Well established methods Easy integration in models and DSS <u>Cons</u> Partition of fluxes need additional measurements	(Zhang et al. 2017; Papadimitriou et al. 2017; Kaushal et al. 2017; Valayamkunnath et al. 2018; Valayamkunnath et al. 2018; Tie et al. 2018; Wang et al. 2018)
Soil Erosion	Sediment transport	1 ha – Regional 1d - >10y	-21-34%	<u>Pros</u> Long history of	(Efthimiou 2018; García-Barrón et al.

				methods Integrative tools Cons Validation is lacking Labour intensive	2018; Fiener et al. 2018)
Land cover	Satellite	0.01ha – Regional 1d - >10y	16 - 100%	Pros Increasing platforms available Consolidated algorithms Cons Need validation Lack of common Land Use definitions	(Olofsson et al. 2014; Liu et al. 2018; Yang et al. 2018)

Table SM. 1.2 Possible uncertainties decision making faces (following (Hansson and Hadorn 2016)

Type	Knowledge gaps	Understanding the uncertainties
Uncertainty of consequences	Do the model(s) adequately represent the target system? What are the numerical values of input parameters, boundary conditions, or initial conditions? What are all potential events that we would take into account if we were aware of them? Will future events relevant for our decisions, including expected impacts from these decisions, in fact take place?	Ensemble approaches; downscaling Benchmarking, sensitivity analyses Scenario approaches
Moral uncertainty	How to (ethically) evaluate the decisions? What values to base the decision on (→ often unreliable ranking of values not doing justice to the range of values at stake, cp. Sen 1992), including choice of discount rate, risk attitude (risk aversion, risk neutral, …) Which ethical principles? (i.e. utilitarian, deontic, virtue, or other?)	Possibly scenario analysis Identification of lock-in effects and path-dependency (e.g. Kinsley et al 2016)
Uncertainty of demarcation	What are the options that we can actually choose between? (not fully known because "decision costs" may be high, or certain options are not "seen" as they are outside current ideologies). How can the mass of decisions divided into individual decisions? e.g. how this influences international negotiations and the question who does what and when (cp. Hammond et al. 1999).	Possibly scenario analysis
Uncertainty of consequences & uncertainty of demarcation	What effects does a decision have when combined with the decision of others? (e.g. other countries may follow the inspiring example in climate reduction of country X, or they use it solely in their own economic interest)	Games
Uncertainty of demarcation & moral uncertainty	How would we decide in the future? (Spohn 1977; Rabinowicz 2002)	

References SM1

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Chapter 2: Land-Climate Interactions

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Executive Summary

Land and climate interact in complex ways through changes in forcing and multiple biophysical and biogeochemical feedbacks across different spatial and temporal scales. This chapter assesses climate impacts on land and land impacts on climate, the human contributions to these changes, as well as land-based adaptation and mitigation response options to combat projected climate changes.

Implications of climate change, variability, and extremes for land systems

It is certain that globally averaged land surface air temperature (LSAT) has risen faster than the global mean surface temperature (i.e., combined LSAT and sea surface temperature) from preindustrial (1850–1900) to present day (1999–2018). According to the single longest and most extensive dataset, the LSAT increase between the preindustrial period and present day was 1.52°C (the *very likely* range of 1.39°C to 1.66°C). For the 1880–2018 period, when four independently produced datasets exist, the LSAT increase was 1.41°C (1.31°C–1.51°C), where the range represents the spread in the datasets' median estimates. Analyses of paleo records, historical observations, model simulations, and underlying physical principles are all in agreement that LSATs are increasing at a higher rate than SST as a result of differences in evaporation, land-climate feedbacks, and changes in the aerosol forcing over land (*very high confidence*). For the 2000–2016 period, the land-to-ocean warming ratio (about 1.6) is in close agreement between different observational records and the CMIP5 climate model simulations (the *likely* range of 1.54 to 1.81). {2.2.1}

Anthropogenic warming has resulted in shifts of climate zones, primarily as an increase in dry climates and decrease of polar climates (*high confidence*). Ongoing warming is projected to result in new, hot climates in tropical regions and to shift climate zones poleward in the mid- to high latitudes and upward in regions of higher elevation (*high confidence*). Ecosystems in these regions will become increasingly exposed to temperature and rainfall extremes beyond climate regimes they are currently adapted to (*high confidence*), which can alter their structure, composition and functioning. Additionally, high-latitude warming is projected to accelerate permafrost thawing and increase disturbance in boreal forests through abiotic (e.g., drought, fire) and biotic (e.g., pests, disease) agents (*high confidence*). {2.2.1, 2.2.2, 2.5.3}

Globally, greening trends (trends of increased photosynthetic activity in vegetation) have increased over the last 2-3 decades by 22–33%, particularly over China, India, many parts of Europe, central North America, southeast Brazil and southeast Australia (*high confidence*). This results from a combination of direct (i.e., land use and management, forest conservation and expansion) and indirect factors (i.e., CO_2 fertilisation, extended growing season, global warming, nitrogen deposition, increase of diffuse radiation) linked to human activities (*high confidence*). Browning trends (trends of decreasing photosynthetic activity) are projected in many regions where increases in drought and heat waves are projected in a warmer climate. There is *low confidence* in the projections of global greening and browning trends. {2.2.4, Cross-Chapter Box 4: Climate change and urbanisation, in this chapter}

The frequency and intensity of some extreme weather and climate events have increased as a consequence of global warming and will continue to increase under medium and high emission scenarios (*high confidence*). Recent heat-related events, e.g., heat waves, have been made more frequent or intense due to anthropogenic greenhouse gas emissions in most land regions and the frequency and intensity of drought has increased in Amazonia, north-eastern Brazil, the Mediterranean, Patagonia, most of Africa and north-eastern China (*medium confidence*). Heat waves are projected to increase in frequency, intensity and duration in most parts of the world (*high confidence*) and drought frequency and intensity is projected to increase in some regions that are already drought prone, predominantly in the Mediterranean, central Europe, the southern Amazon and southern Africa (*medium confidence*). These changes will impact ecosystems, food security and land processes including greenhouse gas (GHG) fluxes (*high confidence*). {2.2.5}

Climate change is playing an increasing role in determining wildfire regimes along-side human activity (*medium confidence*), with future climate variability expected to enhance the risk and severity of wildfires in many biomes such as tropical rainforests (*high confidence*). Fire weather seasons have lengthened globally between 1979 and 2013 (*low confidence*). Global land area burned has declined in recent decades, mainly due to less burning in grasslands and savannahs (*high confidence*). While drought remains the dominant driver of fire emissions, there has recently been increased fire activity in some tropical and temperate regions during normal to wetter than average years due to warmer temperatures that increase vegetation flammability (*medium confidence*). The boreal zone is also experiencing larger and more frequent fires, and this may increase under a warmer climate (*medium confidence*). {Cross-Chapter Box 4: Climate change and urbanisation, in this chapter}

Terrestrial greenhouse gas fluxes on unmanaged and managed lands

Agriculture, Forestry and Other Land Use (AFOLU) is a significant net source of GHG emissions (*high confidence*), contributing to about 22% of anthropogenic emissions of carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O) combined as CO_2 equivalents in 2007 to 2016 (*medium confidence*). AFOLU results in both emissions and removals of CO_2, CH_4, and N_2O to and from the atmosphere (*high confidence*). These fluxes are affected simultaneously by natural and human drivers, making it difficult to separate natural from anthropogenic fluxes (*very high confidence*). {2.3}

The total net land-atmosphere flux of CO_2 on both managed and unmanaged lands *very likely* provided a global net removal from 2008 to 2017 according to models, (-6.2 ± 3.7 $GtCO_2$ yr^{-1}, *medium confidence*). This net removal is comprised of two major components: i) modelled net anthropogenic emissions from AFOLU are *likely* 5.5 ± 2.6 $GtCO_2$ yr^{-1} driven by land cover change, including deforestation and afforestation/reforestation, and wood harvesting (accounting for about 13% of total net anthropogenic emissions of CO_2) (*medium confidence*); and ii) modelled net removals due to non-anthropogenic processes are *likely* 11.7 ± 2.6 $GtCO_2$ yr^{-1} on managed and unmanaged lands, driven by environmental changes such as increasing CO_2, nitrogen deposition, and changes in climate (accounting for a removal of 29% of the CO_2 emitted from all anthropogenic activities (fossil fuel, industry and AFOLU) (*medium confidence*). {2.3.1}

The anthropogenic emissions of CO_2 from AFOLU reported in countries' GHG inventories were 0.1 ± 1.0 $GtCO_2$ yr^{-1} globally during 2005 to 2014 (*low confidence*), much lower than emission estimates from global models of 5.1 ± 2.6 $GtCO_2$ yr^{-1} over the same time period. Reconciling these differences can support consistency and transparency in assessing global progress towards meeting modelled mitigation pathway such as under the Paris Agreement's global stocktake (*medium confidence*). This discrepancy is consistent with understanding of the different approaches used to defining anthropogenic fluxes. Inventories consider larger areas of forested lands as managed than models do, and report all fluxes on managed lands as anthropogenic, including a large net sink due to the indirect effects of changing environmental conditions (e.g., climate change, and change in atmospheric CO_2 and N). In contrast, the models assign part of this indirect forest sink to the non-anthropogenic sink on unmanaged lands. {2.3.1}

The gross emissions from AFOLU (one third of total global emissions) are more indicative of mitigation potential of reduced deforestation than the global net emissions (13% of total global emissions), which include compensating deforestation and afforestation fluxes (*high confidence*). The net flux of CO_2 from AFOLU is composed of two opposing gross fluxes: gross emissions (20 $GtCO_2$ yr^{-1}) from deforestation, cultivation of soils, and oxidation of wood products; and gross removals (-14 $GtCO_2$ yr^{-1}) largely from forest growth following wood harvest and agricultural abandonment (*medium confidence*). {2.3.1}

Land is a net source of CH_4, accounting for 61% of anthropogenic CH_4 emissions for the 2005–2015

period (*medium confidence*). The pause in the rise of atmospheric CH_4 concentrations between 2000 and 2006 and the subsequent renewed increase appear to be partially associated with land use and land use change. The recent depletion trend of the ^{13}C isotope in the atmosphere indicates that higher biogenic sources explain part of the current CH_4 increase and that biogenic sources make up a larger proportion of the source mix than they did before 2000 (*high confidence*). In agreement with the findings of AR5, tropical wetlands and peatlands continue to be important drivers of inter-annual variability and current CH_4 concentration increases (*medium evidence, high agreement*). Ruminants and the expansion of rice cultivation are also important contributors to the current trend (*medium evidence, high agreement*). There is significant and ongoing accumulation of CH_4 in the atmosphere (*very high confidence*). {2.3.2}

AFOLU is the main anthropogenic sources of N_2O primarily due to nitrogen (N) application to soils (*high confidence*). In croplands, the main driver of N_2O emissions is a lack of synchronisation between crop N demand and soil N supply, with approximately 50% of the N applied to agricultural land not taken up by the crop. Cropland soils emit over 3 Mt N_2O-N yr^{-1} (*medium confidence*). Because the response of N_2O emissions to fertiliser application rates is non-linear, in regions of the World where low N application rates dominate, such as sub-Saharan Africa and parts of Eastern Europe, increases in N fertiliser use would generate relatively small increases in agricultural N_2O emissions. Decreases in application rates in regions where application rates are high and exceed crop demand for parts of the growing season will have very large effects on emissions reductions (*medium evidence, high agreement*). {2.3.3}

While managed pastures make up only one-quarter of grazing lands, they contributed more than three-quarters of N_2O emissions from grazing lands between 1961 and 2014 with rapid recent increases of N inputs resulting in disproportionate growth in emissions from these lands (*medium confidence*). Grazing lands (pastures and rangelands) are responsible for more than one-third of total anthropogenic N_2O emissions or more than one-half of agricultural emissions (*high confidence*). Emissions are largely from North America, Europe, East Asia, and South Asia, but hotspots are shifting from Europe to southern Asia (*medium confidence*). {2.3.3}

Increased emissions from vegetation and soils due to climate change in the future are expected to counteract potential sinks due to CO_2 fertilisation *(low confidence)*. Responses of vegetation and soil organic carbon (SOC) to rising atmospheric CO_2 concentration and climate change are not well constrained by observations *(medium confidence)*. Nutrient (e.g., nitrogen, phosphorus) availability can limit future plant growth and carbon storage under rising CO_2 (*high confidence*). However, new evidence suggests that ecosystem adaptation through plant-microbe symbioses could alleviate some nitrogen limitation (*medium evidence, high agreement*). Warming of soils and increased litter inputs will accelerate carbon losses through microbial respiration (*high confidence*). Thawing of high-latitude/altitude permafrost will increase rates of SOC loss and change the balance between CO_2 and CH_4 emissions (*medium confidence*). The balance between increased respiration in warmer climates and carbon uptake from enhanced plant growth is a key uncertainty for the size of the future land carbon sink *(medium confidence*). {2.3.1, 2.7.2, Box 2.3}

Biophysical and biogeochemical land forcing and feedbacks to the climate system

Changes in land conditions from human use or climate change in turn affect regional and global climate *(high confidence)*. On the global scale, this is driven by changes in emissions or removals of CO_2, CH_4, and N_2O by land (biogeochemical effects) and by changes in the surface albedo (*very high confidence*). Any local land changes that redistribute energy and water vapour between the land and the atmosphere influence regional climate (biophysical effects; *high confidence*). However, there is *no confidence* in whether such biophisical effects influence global climate. {2.1, 2.3, 2.5.1, 2.5.2}

Changes in land conditions modulate the likelihood, intensity and duration of many extreme events including heat waves *(high confidence)* and heavy precipitation events *(medium confidence)*. Dry soil conditions favour or strengthen summer heat wave conditions through reduced evapotranspiration and increased sensible heat. By contrast wet soil conditions, for example from irrigation, or crop management practices that maintain a cover crop all year round, can dampen extreme warm events through increased evapotranspiration and reduced sensible heat. Droughts can be intensified by poor land management. Urbanisation increases extreme rainfall events over or downwind of cities (*medium confidence*). {2.5.1, 2.5.2, 2.5.3}

Historical changes in anthropogenic land cover have resulted in a mean annual global warming of surface air from biogeochemical effects *(very high confidence)*, dampened by a cooling from biophysical effects (*medium confidence*). Biogeochemical warming results from increased emissions of GHGs by land, with model-based estimates of +0.20±0.05°C (global climate models) and +0.24±0.12°C (dynamic global vegetation models, DGVMs) as well as an observation-based estimate of +0.25±0.10°C. A net biophysical cooling of –0.10±0.14°C has been derived from global climate models in response to the increased surface albedo and decreased turbulent heat fluxes, but it is smaller than the warming effect from land-based emissions. However when both biogeochemical and biophysical effects are accounted for within the same global climate model, the models do not agree on the sign of the net change in mean annual surface air temperature. {2.3, 2.5.1, Box 2.1}

The future projected changes in anthropogenic land cover that have been examined for AR5 would result in a biogeochemical warming and a biophysical cooling whose magnitudes depend on the scenario *(high confidence)*. Biogeochemical warming has been projected for RCP8.5 by both global climate models (+0.20±0.15°C) and DGVMs (+0.28±0.11°C) (*high confidence*). A global biophysical cooling of 0.10±0.14°C is estimated from global climate models, and projected to dampen the land-based warming (*low confidence*). For RCP4.5 the biogeochemical warming estimated from global climate models (+0.12±0.17°C) is stronger than the warming estimated by DGVMs (+0.01±0.04°C) but based on *limited evidence*, as is the biophysical cooling (–0.10±0.21°C). {2.5.2}

Regional climate change can be dampened or enhanced by changes in local land cover and land use (*high confidence*) but this depends on the location and the season (*high confidence*). In boreal regions, for example, where projected climate change will migrate treeline northward, increase the growing season length and thaw permafrost, regional winter warming will be enhanced by decreased surface albedo and snow, whereas warming will be dampened during the growing season due to larger evapotranspiration (*high confidence*). In the tropics, wherever climate change will increase rainfall, vegetation growth and associated increase in evapotranspiration will result in a dampening effect on regional warming (*medium confidence*). {2.5.2, 2.5.3}

According to model-based studies, changes in local land cover or available water from irrigation affect climate in regions as far as few hundreds of kilometres downwind (*high confidence*). The local redistribution of water and energy following the changes on land affect the horizontal and vertical gradients of temperature, pressure and moisture, thus alter regional winds and consequently moisture and temperature advection and convection, and this affects precipitation. {2.5.2, 2.5.4, Cross-Chapter Box 4: Climate Change and Urbanisation}

Future increases in both climate change and urbanisation will enhance warming in cities and their surroundings (urban heat island), especially during heat waves (*high confidence*). Urban and peri-urban agriculture, and more generally urban greening, can contribute to mitigation (*medium confidence*) as well as to adaptation (*high confidence*), with co-benefits for food security and reduced soil-water-air pollution. {Cross-Chapter Box 4: Climate Change and Urbanisation}

Regional climate is strongly affected by natural land aerosols *(medium confidence)* (e.g., mineral dust, black, brown and organic carbon), but there is *low confidence* in historical trends, interannual and

decadal variability, and future changes. Forest cover affects climate through emissions of biogenic volatile organic compounds (BVOC) and aerosols (*low confidence).* The decrease in the emissions of BVOC resulting from the historical conversion of forests to cropland has resulted in a positive radiative forcing through direct and indirect aerosol effects, a negative radiative forcing through the reduction in the atmospheric lifetime of methane and it has contributed to increased ozone concentrations in different regions (*low confidence*). {2.4, 2.5}

Consequences for the climate system of land-based adaptation and mitigation options, including carbon dioxide removal (negative emissions)

About one quarter of the 2030 mitigation pledged by countries in their initial Nationally Determined Contributions (NDCs) under the Paris Agreement is expected to come from land-based mitigation options *(medium confidence)*. Most of the Nationally Determined Contributions (NDCs) submitted by countries include land-based mitigation, although many lack details. Several refer explicitly to reduced deforestation and forest sinks, while a few include soil carbon sequestration, agricultural management and bioenergy. Full implementation of NDCs (submitted by February 2016) is expected to result in net removals of 0.4–1.3 $GtCO_2$ y^{-1} in 2030 compared to the net flux in 2010, where the range represents low to high mitigation ambition in pledges, not uncertainty in estimates (*medium confidence*). {2.6.3}

Several mitigation response options have technical potential for >3 $GtCO_2$-eq yr^{-1} by 2050 through reduced emissions and Carbon Dioxide Removal (CDR) *(high confidence)*, some of which compete for land and other resources, while others may reduce the demand for land *(high confidence)*. Estimates of the technical potential of individual response options are not necessarily additive. The largest potential for reducing AFOLU emissions are through reduced deforestation and forest degradation (0.4–5.8 $GtCO_2$-eq yr^{-1}) *(high confidence),* a shift towards plant-based diets (0.7–8.0 $GtCO_2$-eq yr^{-1}) *(high confidence)* and reduced food and agricultural waste (0.8–4.5 CO_2-eq yr^{-1}) *(high confidence)*. Agriculture measures combined could mitigate 0.3–3.4 $GtCO_2$-eq yr^{-1} (*medium confidence).* The options with largest potential for CDR are afforestation/reforestation (0.5–10.1 CO_2-eq yr^{-1}) *(medium confidence),* soil carbon sequestration in croplands and grasslands (0.4–8.6 CO_2-eq yr^{-1}) *(high confidence)* and Bioenergy with Carbon Capture and Storage (BECCS) (0.4–11.3 CO_2-eq yr^{-1}) *(medium confidence)*. While some estimates include sustainability and cost considerations, most do not include socio-economic barriers, the impacts of future climate change or non-GHG climate forcings. {2.6.1}

Response options intended to mitigate global warming will also affect the climate locally and regionally through biophysical effects *(high confidence)*. Expansion of forest area, for example, typically removes CO_2 from the atmosphere and thus dampens global warming (biogeochemical effect, *high confidence*), but the biophysical effects can dampen or enhance regional warming depending on location, season and time of day**.** During the growing season, afforestation generally brings cooler days from increased evapotranspiration, and warmer nights (*high confidence*). During the dormant season, forests are warmer than any other land cover, especially in snow-covered areas where forest cover reduces albedo (*high confidence*). At the global level, the temperature effects of boreal afforestation/reforestation run counter to GHG effects, while in the tropics they enhance GHG effects. In addition, trees locally dampen the amplitude of heat extremes (*medium confidence*). {2.5.2, 2.5.4, 2.7, Cross-Chapter Box 4: Climate Change and Urbanisation}

Mitigation response options related to land use are a key element of most modelled scenarios that provide strong mitigation, alongside emissions reduction in other sectors (*high confidence*). More stringent climate targets rely more heavily on land-based mitigation options, in particular, CDR (*high confidence*). Across a range of scenarios in 2100, CDR is delivered by both afforestation (median values of -1.3, -1.7 and -2.4 $GtCO_2yr^{-1}$ for scenarios RCP4.5, RCP2.6 and RCP1.9 respectively) and bioenergy with carbon capture and storage (BECCS) (-6.5, -11 and -14.9 $GtCO_2$ yr^{-1}). Emissions of CH_4

and N_2O are reduced through improved agricultural and livestock management as well as dietary shifts away from emission-intensive livestock products by 133.2, 108.4 and 73.5 $MtCH_4\ yr^{-1}$; and 7.4, 6.1 and 4.5 $MtN_2O\ yr^{-1}$ for the same set of scenarios in 2100 (*high confidence*). High levels of bioenergy crop production can result in increased N_2O emissions due to fertiliser use. The Integrated Assessment Models that produce these scenarios mostly neglect the biophysical effects of land-use son global and regional warming. {2.5, 2.6.2}

Large-scale implementation of mitigation response options that limit warming to 1.5 or 2°C would require conversion of large areas of land for afforestation/reforestation and bioenergy crops, which could lead to short-term carbon losses *(high confidence)*. The change of global forest area in mitigation pathways ranges from about -0.2 to +7.2 Mkm^2 between 2010 and 2100 (median values across a range of models and scenarios: RCP4.5, RCP2.6, RCP1.9), and the land demand for bioenergy crops ranges from about 3.2–6.6 Mkm^2 in 2100 (*high confidence*). Large-scale land-based CDR is associated with multiple feasibility and sustainability constraints (Chapters 6, 7). In high carbon lands such as forests and peatlands, the carbon benefits of land protection are greater in the short-term than converting land to bioenergy crops for BECCS, which can take several harvest cycles to 'pay-back' the carbon emitted during conversion (carbon-debt), from decades to over a century *(medium confidence)*. {2.6.2, Chapters 6, 7}

It is possible to achieve climate change targets with low need for land-demanding CDR such as BECCS, but such scenarios rely more on rapidly reduced emissions or CDR from forests, agriculture and other sectors. Terrestrial CDR has the technical potential to balance emissions that are difficult to eliminate with current technologies (including food production). Scenarios that achieve climate change targets with less need for terrestrial CDR rely on agricultural demand-side changes (diet change, waste reduction), and changes in agricultural production such as agricultural intensification. Such pathways that minimise land use for bioenergy and BECCS are characterised by rapid and early reduction of GHG emissions in all sectors, as well as earlier CDR in through afforestation. In contrast, delayed mitigation action would increase reliance on land-based CDR (*high confidence*). {2.6.2}

2.1 Introduction: Land – climate interactions

This chapter assesses the literature on two-way interactions between climate and land, with focus on scientific findings published since AR5 and some aspects of the land-climate interactions that were not assessed in previous IPCC reports. Previous IPCC assessments recognised that climate affects land cover and land surface processes, which in turn affect climate. However, previous assessments mostly focused on the contribution of land to global climate change via its role in emitting and absorbing greenhouse gases (GHGs) and short-lived climate forcers (SLCFs), or via implications of changes in surface reflective properties (i.e., albedo) for solar radiation absorbed by the surface. This chapter examines scientific advances in understanding the interactive changes of climate and land, including impacts of climate change, variability and extremes on managed and unmanaged lands. It assesses climate forcing of land changes from direct (e.g., land use change and land management) and indirect (e.g., increasing atmospheric CO_2 concentration and nitrogen deposition) effects at local, regional, and global scale.

2.1.1 Recap of previous IPCC and other relevant reports as baselines

The evidence that land cover matters for the climate system have long been known, especially from early paleoclimate modelling studies and impacts of human-induced deforestation at the margin of deserts (de Noblet et al. 1996; Kageyama et al. 2004). The understanding of how land use activities impact climate has been put forward by the pioneering work of (Charney 1975) who examined the role of overgrazing-induced desertification on the Sahelian climate.

Since then there have been many modelling studies that reported impacts of idealised or simplified land cover changes on weather patterns (e.g., Pielke et al. 2011). The number of studies dealing with such issues has increased significantly over the past 10 years, with more studies that address realistic past or projected land changes. However, very few studies have addressed the impacts of land cover changes on climate as very few land surface models embedded within climate models (whether global or regional), include a representation of land management. Observation-based evidence of land-induced climate impacts emerged even more recently (e.g., Alkama and Cescatti 2016; Bright et al. 2017; Lee et al. 2011; Li et al. 2015; Duveiller et al. 2018; Forzieri et al. 2017) and the literature is therefore limited.

In previous IPCC reports, the interactions between climate change and land were covered separately by three working groups. AR5 WGI assessed the role of land use change in radiative forcing, land-based GHGs source and sink, and water cycle changes that focused on changes of evapotranspiration, snow and ice, runoff, and humidity. AR5 WGII examined impacts of climate change on land, including terrestrial and freshwater ecosystems, managed ecosystems, and cities and settlements. AR5 WGIII assessed land-based climate change mitigation goals and pathways in the AFOLU. Here, this chapter assess land-climate interactions from all three working groups. It also builds on previous special reports such as the Special Report on Global Warming of 1.5°C (SR15). It links to the IPCC Guidelines on National Greenhouse Gas Inventories in the land sector. Importantly, this chapter assesses knowledge that has never been reported in any of those previous reports. Finally, the chapter also tries to reconcile the possible inconsistencies across the various IPCC reports.

Land-based water cycle changes: AR5 reported an increase in global evapotranspiration from the early 1980s to 2000s, but a constraint on further increases from low of soil moisture availability. Rising CO_2 concentration limits stomatal opening and thus also reduces transpiration, a component of evapotranspiration. Increasing aerosol levels, and declining surface wind speeds and levels of solar radiation reaching the ground are additional regional causes of the decrease in evapotranspiration.

Land area precipitation change: Averaged over the mid-latitude land areas of the Northern Hemisphere, precipitation has increased since 1901 (*medium confidence* before and *high confidence* after 1951). For other latitudes, area-averaged long-term positive or negative trends have *low confidence*. There are *likely* more land regions where the number of heavy precipitation events has increased than where it has decreased. Extreme precipitation events over most of the mid-latitude land masses and over wet tropical regions will very likely become more intense and more frequent (IPCC 2013a).

Land-based GHGs: AR5 reported that annual net CO_2 emissions from anthropogenic land use change were 0.9 [0.1–1.7] GtC yr^{-1} on average during 2002 to 2011 (*medium confidence*). From 1750 to 2011, CO_2 emissions from fossil fuel combustion have released an estimated 375 [345–405] GtC to the atmosphere, while deforestation and other land use change have released an estimated 180 [100–260] GtC. Of these cumulative anthropogenic CO_2 emissions, 240 [230–250] GtC have accumulated in the atmosphere, 155 [125–185] GtC have been taken up by the ocean and 160 [70–250] GtC have accumulated in terrestrial ecosystems (i.e., the cumulative residual land sink) (Ciais et al. 2013a). Updated assessment and knowledge gaps are covered in Section 2.3.

Future terrestrial carbon source/sink: AR5 projected with *high confidence* that tropical ecosystems will uptake less carbon and with *medium confidence* that at high latitudes, land carbon sink will increase in a warmer climate. Thawing permafrost in the high latitudes is potentially a large carbon source at warmer climate, but the magnitude of CO_2 and CH_4 emissions due to permafrost thawing is still uncertain. The SR15 further indicates that constraining warming to 1.5°C would prevent the melting of an estimated permafrost area of 2 million km^2 over the next centuries compared to 2°C. Updates to these assessments are found in Sections 2.3.

Land use change altered albedo: AR5 stated with *high confidence* that anthropogenic land use change has increased the land surface albedo, which has led to a RF of –0.15 ± 0.10 W m^{-2}. However, it also underlined that the sources of the large spread across independent estimates was caused by differences in assumptions for the albedo of natural and managed surfaces and for the fraction of land use change before 1750. Generally, our understanding of albedo changes from land use change has been enhanced from AR4 to AR5, with a narrower range of estimates and a higher confidence level. The radiative forcing from changes in albedo induced by land use changes was estimated in AR5 at -0.15 W m^{-2} (-0.25 to about -0.05), with *medium confidence* in AR5 (Shindell et al. 2013). This was an improvement over AR4 in which it was estimated at -0.2 W m^{-2} (-0.4 to about 0), with *low to medium confidence* (Forster et al. 2007). Section 2.5 shows that albedo is not the only source of biophysical land-based climate forcing to be considered.

Hydrological feedback to climate: Land use changes also affect surface temperatures through non-radiative processes, and particularly through the hydrological cycle. These processes are less well known and are difficult to quantify, but tend to offset the impact of albedo changes. As a consequence, there is low agreement on the sign of the net change in global mean temperature as a result of land use change (Hartmann et al. 2013a). An updated assessment on these points is covered in Section 2.5 and 2.2

Climate-related extremes on land: AR5 reported that impacts from recent climate-related extremes reveal significant vulnerability and exposure of some ecosystems to current climate variability. Impacts of such climate-related extremes include alteration of ecosystems, disruption of food production and water supply, damage to infrastructure and settlements, morbidity and mortality, and consequences for mental health and human well-being (Burkett et al. 2014). The SR15 further indicates that limiting global warming to 1.5°C limits the risks of increases in heavy precipitation events in several regions (*high confidence*). In urban areas climate change is projected to increase risks for people, assets, economies and ecosystems (*very high confidence*). These risks are amplified for those lacking essential infrastructure and services or living in exposed areas. Updated assessment and knowledge gap for this chapter are covered in Section 2.2 and Cross-Chapter Box 4: Climate Change and Urbanisation.

Land-based *climate change adaptation and mitigation:* AR5 reported that adaptation and mitigation choices in the near-term will affect the risks related to climate change throughout the 21st century (Burkett et al. 2014). Agriculture, forestry and other land use (AFOLU) are responsible for about 10–12 $GtCO_2eq\ yr^{-1}$ anthropogenic greenhouse gas emissions mainly from deforestation and agricultural production. Global CO_2 emissions from forestry and other land use have declined since AR4, largely due to increased afforestation. The SR15 further indicates that afforestation and bioenergy with carbon capture and storage (BECCS) are important land-based carbon dioxide removal (CDR) options. It also states that land use and land-use change emerge as a critical feature of virtually all mitigation pathways that seek to limit global warming to 1.5°C. Climate Change 2014 Synthesis Report concluded that co-benefits and adverse side effects of mitigation could affect achievement of other objectives such as those related to human health, food security, biodiversity, local environmental quality, energy access, livelihoods and equitable sustainable development. Updated assessment and knowledge gaps are covered in Section 2.6 and Chapter 7.

Overall, sustainable land management is largely constrained by climate change and extremes, but also puts bounds on the capacity of land to effectively adapt to climate change and mitigate its impacts. Scientific knowledge has advanced on how to optimise our adaptation and mitigation efforts while coordinating sustainable land management across sectors and stakeholder. Details are assessed in subsequent sections.

2.1.2 Introduction to the chapter structure

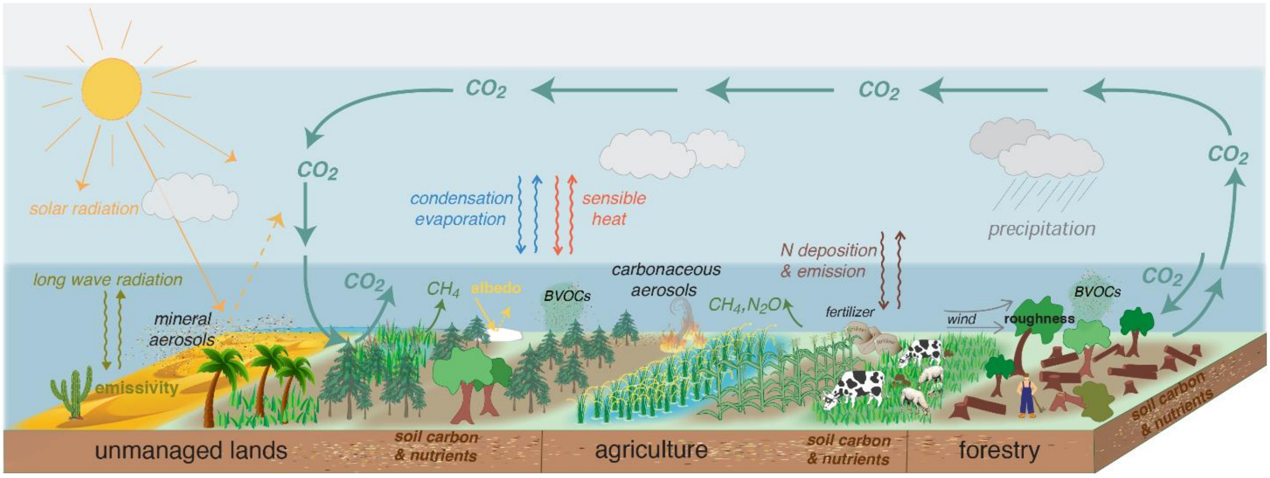
This chapter assess the consequences of changes in land cover and functioning, resulting from both land use and climate change, to global and regional climates. The chapter starts by an assessment of the historical and projected responses of land-based and processes to climate change and extremes (Section 2.2). Subsequently, the chapter assesses historical and future changes in terrestrial GHG fluxes (Section 2.3), non-GHG fluxes and precursors of SLCFs (Section 2.4. Section 2.4 focuses on how historical and future changes in land use and land cover influence climate change/variability through biophysical and biogeochemical forcing and feedbacks, how specific land management affects climate, and how in turn climate-induced land changes feedback to climate. Section 2.6 assesses consequences of land-based adaptation and mitigation options for the climate system in GHG and non-GHG exchanges. Sections 2.3 and 2.6 addresses implications of the Paris Agreement for land-climate interactions, and the scientific evidence base for ongoing negotiations around the Paris rulebook, the Global Stocktake, and credibility in measuring, reporting and verifying the climate impacts of anthropogenic activities on land. The chapter also examines how land use and management practices may affect climate change through biophysical feedbacks and radiative forcing (Section 2.5), and assesses policy relevant projected land use changes and sustainable land management for mitigation and adaptation (Section 2.6). Finally, the chapter concludes with a brief assessment of advances in the understanding of ecological and biogeochemical processes underlying land-climate interactions (Section 2.7).

The chapter includes three chapter boxes providing general overview of (i) processes underlying land-climate interactions (Box 2.1:); (ii) methodological approaches for estimating anthropogenic land carbon fluxes from national to global scales (Box 2.2:); (iii) CO_2 fertilisation and enhanced terrestrial uptake of carbon (Box 2.3). In addition this chapter includes two cross-chapter boxes on climate change and fire (Cross-Chapter Box 3); and on urbanisation and climate change (Cross-Chapter Box 4).

In summary, the chapter assesses scientific understanding related to: 1) how a changing climate affects terrestrial ecosystems, including those on managed lands; 2) how land affects climate through biophysical and biogeochemical feedbacks; and 3) how land use or cover change and land management play an important and complex role in the climate system. This chapter also pays special attention to advances in understanding cross-scale interactions, emerging issues, heterogeneity, and teleconnections.

Box 2.1: Processes underlying land-climate interactions

Land continuously interacts with the atmosphere through exchanges of, for instance, greenhouse gases (e.g., CO_2, CH_4, N_2O), water, energy, or precursors of short lived-climate forcers (e.g., biogenic volatile organic compounds, dust, black carbon). The terrestrial biosphere also interacts with oceans through processes such as the influx of freshwater, nutrients, carbon and particles. These interactions affect where and when rain falls and thus irrigation needs for crops, frequency and intensity of heat waves, and air quality. They are modified by global and regional climate change, decadal, interannual and seasonal climatic variations, and weather extremes, as well as human actions on land (e.g., crop and forest management, afforestation and deforestation). This in turn affects atmospheric composition, surface temperature, hydrological cycle and thus local, regional and global climate. This box introduces some of the fundamental land processes governing biophysical and biogeochemical effects and feedbacks to the climate (Box 2.1 Figure 1)



Box 2.1 Figure 1 The structure and functioning of managed and unmanaged ecosystems that affect local, regional, and global climate. Land surface characteristics such as albedo and emissivity determine the amount of solar and long-wave radiation absorbed by land and reflected or emitted to the atmosphere. Surface roughness influences turbulent exchanges of momentum, energy, water, and biogeochemical tracers. Land ecosystems modulate the atmospheric composition through emissions and removals of many GHGs and precursors of SLCFs, including biogenic volatile organic compounds (BVOCs) and mineral dust. Atmospheric aerosols formed from these precursors affect regional climate by altering amounts of precipitation and of radiation reaching land surfaces through their role in clouds physics.

'Biophysical interactions' are exchanges of water and energy between the land and the atmosphere (Section 2.5). Land warms up from absorbing solar and long-wave radiation; it cools down through transfers of sensible heat (*via* conduction and convection) and latent heat (energy associated with water evapotranspiration) to the atmosphere and through longwave radiation emission from the land surface (Box 2.1 Figure 1). These interactions between the land and the atmosphere depend on the land surface characteristics, including reflectivity of short-wave radiation (albedo), emissivity of long wave radiation by vegetation and soils, surface roughness, and soil water access by vegetation, which depends on both soil characteristics and amounts of roots. Over seasonal, interannual and decadal time scales, these characteristics vary among different land cover and land-use types and are affected by both natural processes and land management (Anderson et al. 2011). A dense vegetation with high leaf area index, like forests, may absorb more energy than nearby herbaceous vegetation partly due to differences in surface albedo (especially when snow is on the ground). However, denser vegetation also sends more energy back to the atmosphere in the form of evapotranspiration (Bonan, 2008; Burakowski et al., 2018; Ellison et al., 2017; Section 2.5.2) and this contributes to changes in atmospheric water vapour content, affecting and rainfall.

Particularly in extra-tropical regions, these characteristics exhibit strong seasonal patterns with the development and senescence of the vegetation (e.g., leaf colour change and drop). For example, in deciduous forests, seasonal growth increases albedo by 20–50% from the spring minima to growing season maxima, followed by rapid decrease during leaf fall, whereas in grasslands, spring greening causes albedo decreases and only increases with vegetation browning (Hollinger et al. 2010). The seasonal patterns of sensible and latent heat fluxes are also driven by the cycle of leaf development and senescence in temperate deciduous forests: sensible heat fluxes peak in spring and autumn and latent heat fluxes peak in mid-summer (Moore et al. 1996; Richardson et al. 2013).

Exchanges of greenhouse gases between the land and the atmosphere are referred to as 'biogeochemical interactions' (Section 2.3), which are driven mainly by the balance between photosynthesis and respiration by plants, and by the decomposition of soil organic matter by microbes. The conversion of atmospheric carbon dioxide into organic compounds by plant photosynthesis, known as terrestrial net primary productivity, is the source of plant growth, food for human and other organisms, and soil organic carbon. Due to strong seasonal patterns of growth, northern hemisphere terrestrial ecosystems are largely responsible for the seasonal variations in global atmospheric CO_2 concentrations. In addition to CO_2, soils emit methane (CH_4) and nitrous oxide (N_2O) (Section 2.3). Soil temperature and moisture strongly affect microbial activities and resulting fluxes of these three greenhouse gases.

Much like fossil fuel emissions, GHG emissions from anthropogenic land cover change and land management are 'forcers' on the climate system. Other land-based changes to climate are described as 'feedbacks' to the climate system - a process by which climate change influences some property of land, which in turn diminishes (negative feedback) or amplifies (positive feedback) climate change. Examples of feedbacks include the changes in the strength of land carbon sinks or sources, soil moisture and plant phenology (Section 2.5.3).

Incorporating these land-climate processes into climate projections allows for increased understanding of the land's response to climate change (Section 2.2), and to better quantify the potential of land-based response options for climate change mitigation (Section 2.6). However, to date Earth system models (ESMs) incorporate some combined biophysical and biogeochemical processes only to limited extent and many relevant processes about how plants and soils interactively respond to climate changes are still to be included. (Section 2.7). And even within this class of models, the spread in ESM projections is large, in part because of their varying ability to represent land-climate processes (Hoffman et al. 2014). Significant progress in understanding of these processes has nevertheless been made since AR5.

2.2 The effect of climate variability and change on land

2.2.1 Overview of climate impacts on land

2.2.1.1 Climate drivers of land form and function

Energy is redistributed from the warm equator to the colder poles through large-scale atmospheric and oceanic processes driving the Earth’s weather and climate (Oort and Peixóto 1983; Carissimo et al. 1985; Yang et al. 2015a). Subsequently, a number of global climate zones have been classified ranging from large-scale primary climate zones (tropical, sub-tropical, temperate, sub-polar, polar) to much higher-resolution, regional climate zones (e.g., the Köppen-Geiger classification, Kottek et al. 2006). Biomes are adapted to regional climates (Figure 2.1) and may shift as climate, land surface characteristics (e.g., geomorphology, hydrology), CO_2 fertilisation, and fire interact. These biomes and processes therein are subject to modes of natural variability in the ocean-atmosphere system that result in regionally wetter/dryer or hotter/cooler periods having temporal scales from weeks to months (e.g., Southern Annular Mode), months to seasons (e.g., Madden-Julian Oscillation), years (e.g., El Niño Southern Oscillation) and decades (e.g. Pacific Decadal Oscillation). Furthermore, climate and weather extremes (such as drought, heat waves, very heavy rainfall, strong winds), whose frequency, intensity and duration are often a function of large-scale modes of variability, impact ecosystems at various space and time scales.

It is *very likely* that changes to natural climate variability as a result of global warming has and will continue to impact terrestrial ecosystems with subsequent impacts on land processes (Hulme et al. 1999; Parmesan and Yohe 2003; Di Lorenzo et al. 2008; Kløve et al. 2014; Berg et al. 2015; Lemordant et al. 2016; Pecl et al. 2017). This chapter assesses climate variability and change, particularly extreme weather and climate, in the context of desertification, land degradation, food security and terrestrial ecosystems more generally. This section does specifically assess the impacts of climate variability and climate change on desertification, land degradation and food security as these impacts are assessed respectively in Chapters 3, 4 and 5. This chapter begins with an assessment of observed warming on land.

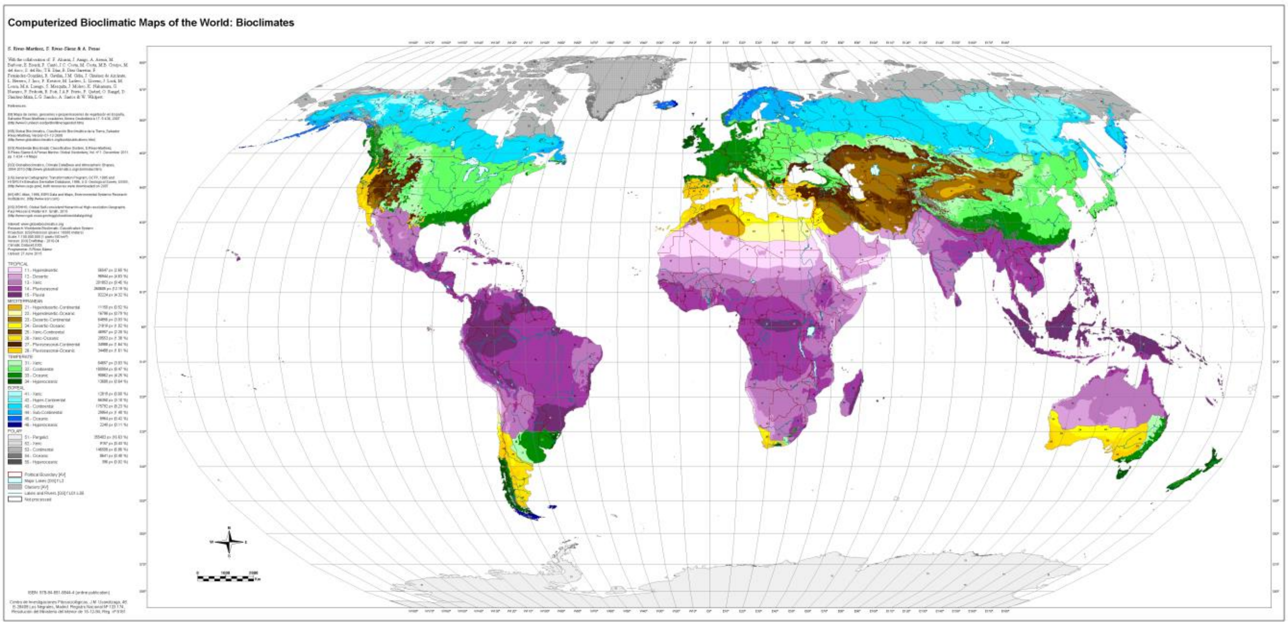


Figure 2.1 Worldwide Bioclimatic Classification System, 1996-2018. After (Rivas-Martinez et al. 2011). Online at http://www.globalbioclimatics.org.

2.2.1.2 Changes in global land surface air temperature

Based on analysis of several global and regional land surface air temperature (LSAT) datasets, AR5 concluded that the global LSAT had increased over the instrumental period of record, with the warming rate approximately double that reported over the oceans since 1979 and that “it is certain that globally averaged LSAT has risen since the late 19th century and that this warming has been particularly marked since the

1970s". Warming found in the global land datasets is also in a broad agreement with station observations (Hartmann et al. 2013a).

Since AR5, LSAT datasets have been improved and extended. The National Center for Environmental Information, which is a part of the US National Oceanic and Atmospheric Administration (NOAA), developed a new version of the Global Historical Climatology Network (GHCNm, version 4) dataset. The dataset provides an expanded set of station temperature records with more than 25,000 total monthly temperature stations compared to 7200 in versions v2 and v3 (Menne et al. 2018). Goddard Institute for Space Studies, which is a part of the US National Aeronautics and Space Administration, (NASA/GISS) provides estimate of land and ocean temperature anomalies (GISTEMP). The GISTEMP land temperature anomalies are based upon primarily NOAA/GHCN version 3 dataset (Lawrimore et al. 2011) and account for urban effects through nightlight adjustments (Hansen et al. 2010). The Climatic Research Unit of the University of East Anglia, UK (CRUTEM) dataset, now version CRUTEM4.6, incorporates additional stations (Jones et al. 2012). Finally, the Berkeley Earth Surface Temperature (BEST) dataset provides LSAT from 1750 to present based on almost 46,000 time series and has the longest temporal coverage of the four datasets (Rohde et al. 2013). This dataset was derived with methods distinct from those used for development of the NOAA and NASA datasets and the CRU dataset.

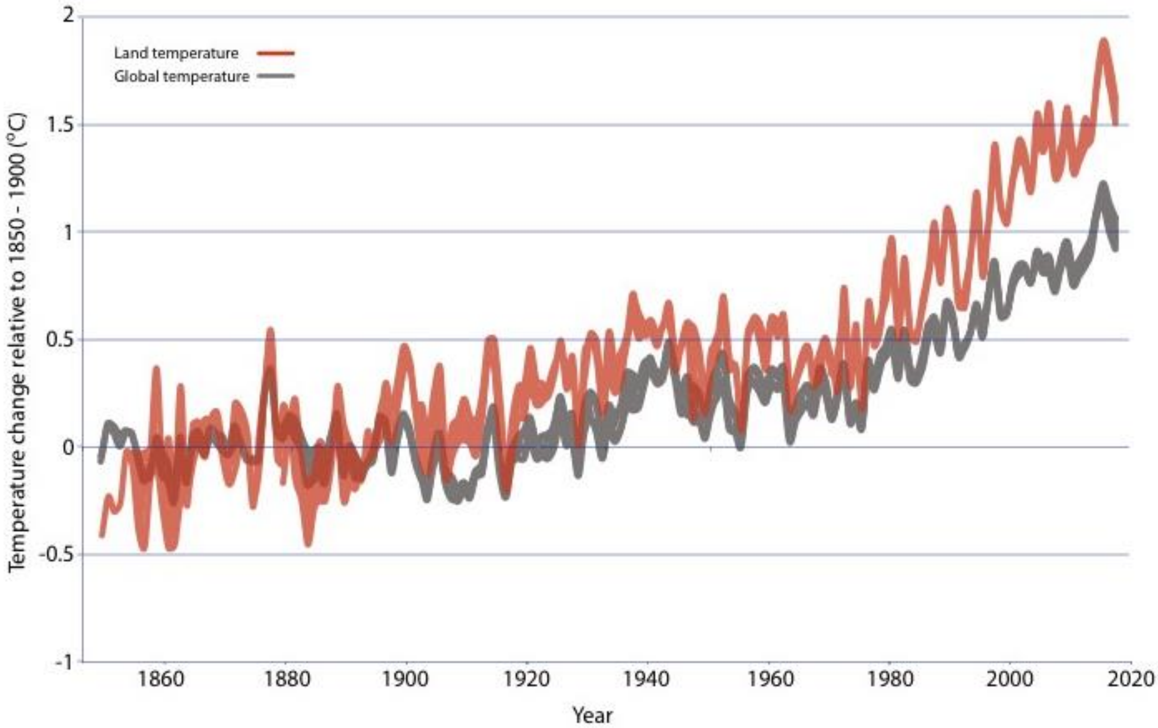


Figure 2.2 Evolution of land surface air temperature (LSAT) and global mean surface temperature (GMST) over the period of instrumental observations. Red line shows annual mean LSAT in the Berkeley, CRUTEM4, GHCNv4 and GISTEMP datasets, expressed as departures from global average LSAT in 1850–1900, with the red line thickness indicating inter-dataset range. Gray shaded line shows annual mean Global Mean Surface Temperature (GMST) in the HadCRUT4, NOAAGlobal Temp, GISTEMP and Cowtan&Way datasets (monthly values of which were reported in the Special Report on Global Warming of 1.5°C (Allen et al. 2018).

According to the available observations in the four datasets, the globally averaged LSAT increased by 1.44°C from the preindustrial period (1850–1900) to present (1999–2018). The warming from the late 19th century (1881–1900) to present (1999–2018) was 1.41°C (1.31°C–1.51°C) (Table 2.1). The 1.31°C–1.51°C range represents the spread in median estimates from the four available land datasets and does not reflect uncertainty in data coverage or methods used. Based on the Berkeley dataset (the longest dataset with the most extensive land coverage) the total increase in LSAT between the average of the 1850–1900 period and

the 1999–2018 period was 1.52°C, (1.39°C–1.66°C; 95% confidence).
The extended and improved land datasets reaffirmed the AR5 conclusion that it is certain that globally averaged LSAT has risen since the preindustrial period and that this warming has been particularly marked since the 1970s (Figure 2.2).

Table 2.1 Increases in land surface air temperature (LSAT) from preindustrial period and the late 19th century to present (1999–2018).

		Dataset of LSAT increase (C°)		
Reference period	**Berkeley**	**CRUTEM4**	**GHCNm, v4**	**GISTEMP**
Preindustrial 1850–1900	1.52 1.39–1.66 (95% confidence)	1.31	NA	NA
Late 19th century 1881–1900	1.51 1.40–1.63 (95% confidence)	1.31	1.37	1.45

Recent analyses of LSAT and sea surface temperature (SST) observations as well as analyses of climate model simulations have refined our understanding of underlying mechanisms responsible for a faster rate of warming over land than over oceans. Analyses of paleo records, historical observations, model simulations, and underlying physical principles are all in agreement that that land is warming faster than the oceans as a result of differences in evaporation, land-climate feedbacks (Section 2.5), and changes in the aerosol forcing over land (Braconnot et al. 2012; Joshi et al. 2013; Sejas et al. 2014; Byrne and O'Gorman 2013, 2015; Wallace and Joshi 2018; Allen et al. 2019) (*very high confidence*). There is also *high confidence* that difference in land and ocean heat capacity is not the primary reason for a faster land than ocean warming. For the recent period the land-to-ocean warming ratio is in close agreement between different observational records (about 1.6) and the CMIP5 climate model simulations (the *likely* range of 1.54°C to 1.81°C). Earlier studies analysing slab ocean models (models in which it is assumed that the deep ocean has equilibrated) produced a higher land temperature increases than sea surface temperature (Manabe et al. 1991; Sutton et al. 2007).

It is certain that globally averaged LSAT has risen faster than GMST from preindustrial (1850–1900) to present day (1999–2018). This is because the warming rate of the land compared to the ocean is substantially higher over the historical period (by approximately 60%) and because Earth surface is approximately one third land and two thirds ocean. This enhanced land warming impacts land processes with implications for desertification (Section 2.2.2, Chapter 3), food security (Section 2.2.3, Chapter 5), terrestrial ecosystems (Section 2.2.4), and GHG and non-GHG fluxes between the land and climate (Section 2.3, Section 2.4). Future changes in land characteristics through adaptation and mitigation processes and associated land-climate feedbacks can dampen warming in some regions and enhance warming in others (Section 2.5).

2.2.2 Climate driven changes in aridity

Desertification is defined and discussed at length in Chapter 3 of this report and is a function of both human activity and climate variability and change. There are uncertainties in distinguishing between historical climate-caused aridification and desertification and also future projections of aridity as different measurement methods of aridity do not agree on historical or projected changes (3.1.1, 3.2.1). However, warming trends over drylands are twice the global average (Lickley and Solomon 2018) some temperate drylands are projected to convert to subtropical drylands as a result of an increased drought frequency causing reduced soil moisture availability in the growing season (Engelbrecht et al. 2015; Schlaepfer et al. 2017). We therefore assess with *medium confidence* that a warming climate will result in regional increases in the spatial extent of drylands under mid- and high emission scenarios and that these regions will warm faster than the global average warming rate.

2.2.3 The influence of climate change on food security

Food security and the various components thereof is addressed in depth in Chapter 5. Climate variables relevant to food security and food systems are predominantly temperature and precipitation-related, but also include integrated metrics that combine these and other variables (like solar radiation, wind, humidity) and extreme weather and climate events including storm surge (see 5.2.1). The impact of climate change through changes in these variables is projected to negatively impact all aspects of food security (food availability, access, utilisation and stability), leading to complex impacts on global food security (Chapter 5) (Table 5.1) (*high confidence*).

Climate change will have regionally distributed impacts, even under aggressive mitigation scenarios (Howden et al. 2007; Rosenzweig et al. 2013; Challinor et al. 2014; Parry et al. 2005; Lobell and Tebaldi 2014; Wheeler and Von Braun 2013). For example, in the northern hemisphere the northward expansion of warmer temperatures in the middle and higher latitudes will lengthen the growing season (Gregory and Marshall 2012; Yang et al. 2015b) which may benefit crop productivity (Parry et al. 2004; Rosenzweig et al., 2014; Deryng et al. 2016). However, continued rising temperatures are expected to impact global wheat yields by about 4–6% reductions for every degree of temperature rise (Liu et al. 2016a; Asseng et al. 2015) and across both mid- and low latitude regions, rising temperatures are also expected to be a constraining factor for maize productivity by the end of the century (Bassu et al. 2014; Zhao et al. 2017). Although there has been a general reduction in frost occurrence during winter and spring and a lengthening of the frost free season in response to growing concentrations of greenhouse gases (Fischer and Knutti 2014; Wypych et al. 2017), there are regions where the frost season length has increased e.g. southern Australia (Crimp et al. 2016). Despite the general reduced frost season length, late spring frosts may increase risk of damage to warming induced precocious vegetation growth and flowering (Meier et al. 2018). Observed and projected warmer minimum temperatures have and will continue to reduce the number of winter chill units required by particularly fruit crops (Luedeling 2012). Crop yields are impacted negatively by increases of seasonal rainfall variability in the tropics, sub-tropics, water-limited and high elevation environments, drought severity and growing season temperatures have a negative impact on crop yield (IFPRI 2009; Schlenker and Lobell 2010; Müller et al. 2017; Parry et al. 2004; Wheeler and Von Braun 2013; Challinor et al. 2014).

Changes in extreme weather and climate (Section 2.2.5) have negative impacts on food security through regional reductions of crop yields. A recent study shows that between 18-43% of the explained yield variance of four crops (maize, soybeans, rice and spring wheat) is attributable to extremes of temperature and rainfall, depending on the crop type (Vogel et al. 2019). Climate shocks, particularly severe drought impact low-income small-holder producers disproportionately (Vermeulen et al. 2012b; Rivera Ferre 2014). Extremes also compromise critical food supply chain infrastructure, making the transport and access to harvested food more difficult (Brown et al. 2015; Fanzo et al. 2018). There is *high confidence* that the impacts of enhanced climate extremes, together with non-climate factors such as nutrient limitation, soil health and competitive plant species, generally outweighs the regionally positive impacts of warming (Lobell et al. 2011; Leakey et al. 2012; Porter et al. 2014; Gray et al. 2016; Pugh et al. 2016; Wheeler and Von Braun 2013; Beer 2018).

2.2.4 Climate-driven changes in terrestrial ecosystems

Previously, the IPCC AR5 reported high confidence that the Earth's biota composition and ecosystem processes have been strongly affected by past changes in global climate, but the rates of the historic climate change are lower than those projected for the 21st century under high warming scenarios like RCP8.5

(Settele et al. 2015a). There is *high confidence* that as a result of climate changes over recent decades many plant and animal species have experienced range size and location changes, altered abundances, and shifts in seasonal activities (Urban 2015a; Ernakovich et al. 2014; Elsen and Tingley 2015; Hatfield and Prueger 2015; Urban 2015b; Savage and Vellend 2015; Yin et al. 2016; Pecl et al. 2017; Gonsamo et al. 2017; Fadrique et al. 2018; Laurance et al. 2018). There is high confidence that climate zones have already shifted in many parts of the world primarily as an increase of dry, arid climates accompanied by a decrease of polar climates (Chan and Wu 2015; Chen and Chen 2013; Spinoni et al. 2015b). Regional climate zones shifts have been observed over the Asian monsoon region (Son and Bae 2015), Europe (Jylhä et al. 2010), China (Yin et al. 2019), Pakistan (Adnan et al. 2017), the Alps (Rubel et al. 2017) and North-Eastern Brazil, Southern Argentina, the Sahel, Zambia and Zimbabwe, the Mediterranean area, Alaska, Canada and North-Eastern Russia (Spinoni et al. 2015b).

There is *high confidence* that bioclimates zones will further shift as the climate warms (Williams et al. 2007; Rubel and Kottek 2010; Garcia et al. 2016; Mahony et al. 2017; Law et al. 2018). There is also high confidence that novel, unprecedented climates (climate conditions with no analog in the observational record) will emerge, particularly the tropics (Williams and Jackson 2007; Colwell et al. 2008a; Mora et al. 2013, 2014; Hawkins et al. 2014; Mahony et al. 2017; Maule et al. 2017). It is *very likely* that terrestrial ecosystems and land processes will be exposed to disturbances beyond the range of current natural variability as a result of global warming, even under low- to medium-range warming scenarios, and these disturbances will alter the structure, composition and functioning of the system (Settele et al. 2015b; Gauthier et al. 2015; Seddon et al. 2016).

In a warming climate many species will be unable to track their climate niche as it moves, especially those in extensive flat landscapes with low dispersal capacity and in the tropics whose thermal optimum is already near current temperature (Diffenbaugh and Field 2013; Warszawski et al. 2013). Range expansion in higher latitudes and elevations as a result of warming often, but not exclusively occurs in abandoned lands (Harsch et al. 2009; Landhäusser et al. 2010; Gottfried et al. 2012; Boisvert-Marsh et al. 2014; Bryn and Potthoff 2018; Rumpf et al. 2018; Buitenwerf et al. 2018; Steinbauer et al. 2018). This expansion typically favours thermophilic species at the expense of cold adapted species as climate becomes suitable for lower latitude/altitude species (Rumpf et al. 2018). In temperate drylands, however, range expansion can be countered by intense and frequent drought conditions which result in accelerated rates of taxonomic change and spatial heterogeneity in an ecotone (Tietjen et al. 2017).

Since the advent of satellite observation platforms, a global increase in vegetation photosynthetic activity (i.e. greening) as evidenced through remotely sensed indices such as leaf area index (LAI) and normalised difference vegetation index (NDVI). Three satellite-based leaf area index (GIMMS3g, GLASS and GLOMAP) records imply increased growing season LAI (greening) over 25–50% and browning over less than 4% of the global vegetated area, resulting in greening trend of $0.068 \pm 0.045\ m^2\ m^{-2}\ yr^{-1}$ over 1982–2009 (Cao et al. 2016). Greening has been observed in southern Amazonia, southern Australia, the Sahel and central Africa, India, eastern China and the northern extratropical latitudes (Myneni et al. 1997; de Jong et al. 2012; Los 2013; Piao et al. 2015; Mao et al. 2016; Zhu et al. 2016; Carlson et al. 2017; Forzieri et al. 2017; Pan et al. 2018; Chen et al. 2019). Greening has been attributed to direct factors, namely human land use management and indirect factors such as CO_2 fertilisation, climate change, nitrogen deposition (Donohue et al. 2013; Keenan et al. 2016; Zhu et al. 2016). Indirect factors have been used to explain most greening trends primarily through CO_2 fertilisation in the tropics and through an extended growing season and increased growing season temperatures as a result of climate change in the high latitudes (Fensholt et al. 2012; Zhu et al. 2016). The extension of the growing season in high latitudes has occurred together with an earlier spring greenup (the time at which plants begin to produce leaves in northern mid- and high-latitude ecosystems) (Goetz et al. 2015; Xu et al. 2016a, 2018) with subsequent earlier spring carbon uptake (2.3 days per decade) and gross primary productivity (GPP) (Pulliainen et al. 2017). The role of direct factors of greening are being increasingly investigated and a recent study has attributed over a third of observed global greening between 2000 and 2017 to direct factors, namely afforestation and croplands, in China and India (Chen et al. 2019).

It should be noted that as measured greening is a product of satellite-derived radiance data, and as such does not provide information on ecosystem health indicators such as species composition and richness, homeostasis, absence of disease, vigor, system resilience and the different components of ecosystems (Jørgensen et al. 2016). For example, a regional greening attributable to croplands expansion or intensification might occur at the expense of ecosystem biodiversity.

Within the global greening trend are also detected regional decreases in vegetation photosynthetic activity (i.e. browning) in northern Eurasia, the southwestern USA, boreal forests in North America, Inner Asia and the Congo Basin, largely as a result of intensified drought stress. Since the late-1990s rates and extents of browning have exceeded those of greening in some regions, the collective result of which has been a slowdown of the global greening rate (de Jong et al. 2012; Pan et al. 2018). Within these long-term trends, interannual variability of regional greening and browning is attributable to regional climate variability, responses to extremes such as drought, disease and insect infestation and large-scale teleconnective controls such as ENSO and the Atlantic Multi-decadal Organization (Verbyla 2008; Revadekar et al. 2012; Epstein et al. 2018; Zhao et al. 2018).

Projected increases in drought conditions in many regions suggest long-term global vegetation greening trends are at risk of reversal to browning in a warmer climate (de Jong et al. 2012; Pan et al. 2018; Pausas and Millán 2018). On the other hand, in higher latitudes vegetation productivity is projected to increase as a result of higher atmospheric CO_2 concentrations and longer growing periods as a result of warming here (Ito et al. 2016)(Section 2.3, Box 2.3). Additionally, climate-driven transitions of ecosystems, particularly range changes, can take years to decades for the equilibrium state to be realised and the rates of these “committed ecosystem changes” (Jones et al. 2009) vary between low and high latitudes (Jones et al. 2010). Furthermore, as direct factors are poorly integrated into Earth systems models (ESMs) uncertainties in projected trends of greening and browning are further compounded (Buitenwerf et al. 2018; Chen et al. 2019). Therefore, there is *low confidence* in the projection of global greening and browning trends.

Increased atmospheric CO_2 concentrations have both direct and indirect effects on terrestrial ecosystems (see Sections 2.2.2 and 2.2.3, Box 2.3). The direct effect is primarily through increased vegetation photosynthetic activity as described above. Indirect effects include decreased evapotranspiration that may offset the projected impact of drought in some water-stressed plants through improved water use efficiency in temperate regions suggesting that some rain-fed cropping systems and grasslands will benefit from elevated atmospheric CO_2 concentrations (Roy et al. 2016a; Milly and Dunne 2016; Swann et al. 2016; Chang et al. 2017; Zhu et al. 2017). In tropical regions increased flowering activity is associated primarily with increasing atmospheric CO_2 suggesting a long-term increase in flowering activity may persist in some vegetation, particularly mid-story trees and tropical shrubs, and enhance reproduction levels until limited by nutrient availability or climate factors like drought frequency, rising temperatures and reduced insolation (Pau et al. 2018).

2.2.5 Climate extremes and their impact on land functioning

Extreme weather events are generally defined as the upper or lower statistical tails of the observed range of values of climate variables or climate indicators (e.g., temperature/rainfall or drought/aridity indices respectively). Previous IPCC reports have reported with *high confidence* on the increase of many types of observed extreme temperature events (Seneviratne et al. 2012b; Hartmann et al. 2013b; Hoegh-Guldberg et al. 2018). However, as a result of observational constraints, increases in precipitation extremes are less confident, except in observation rich regions with dense, long-lived station networks such as Europe and North America where there have been likely increases in the frequency or intensity of heavy rainfall.

Extreme events occur across a wide range of time and space scales (Figure 2.3) and may include individual, relatively short-lived weather events (e.g., extreme thunderstorms storms) or a combination or accumulation of non-extreme events (Colwell et al. 2008b; Kundzewicz and Germany 2012) e.g., moderate rainfall in a saturated catchment having the flood peak at mean high tide (Leonard et al. 2014). Combinatory processes leading to a significant impact are referred to as a compound event and are a function of the nature and number of physical climate and land variables, biological agents such as pests and disease, the range of spatial and temporal scales, the strength of dependence between processes, and the perspective of the stakeholder who defines the impact (Leonard et al. 2014; Millar and Stephenson 2015). Current *confidence* in the impact of compound events on land is *low* as the multi-disciplinary approaches needed to address the problem are few (Zscheischler et al. 2018) and the rarity of compound extreme climatic events renders the analysis of impacts difficult.

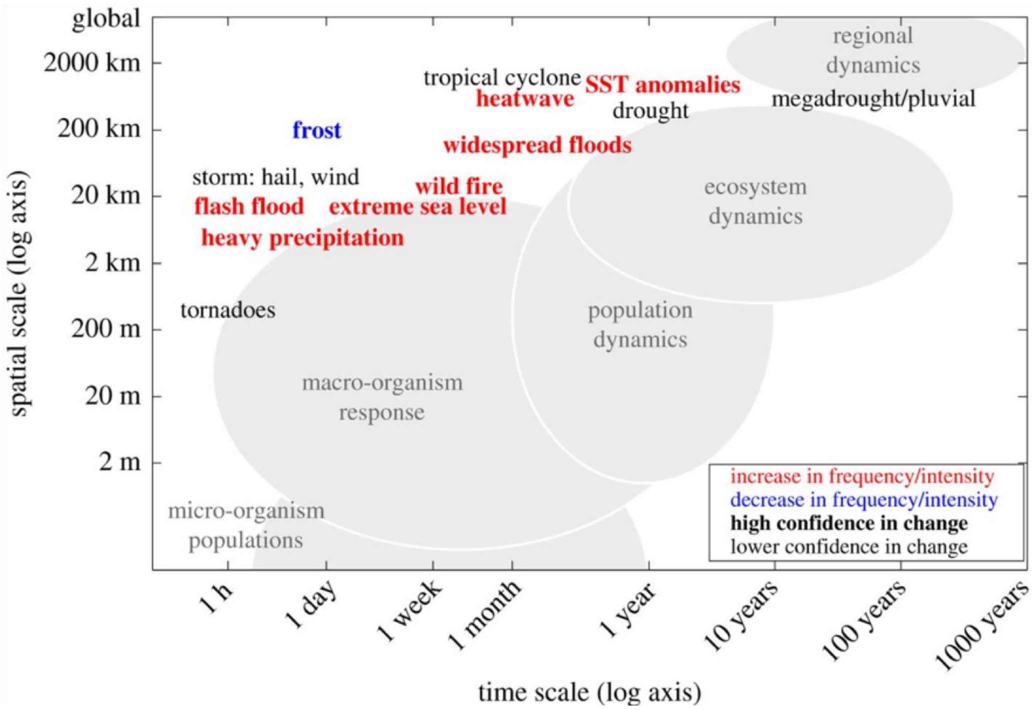


Figure 2.3 Spatial and temporal scales of typical extreme weather and climate events and the biological systems they impact (shaded grey). Individuals, populations and ecosystems within these space-time ranges respond to relevant climate stressors. Red (blue) labels indicate an increase (decrease) in the frequency or intensity of the event, with bold font reflecting confidence in the change. Non-bold black labels indicate low confidence in observed changes in frequency or intensity of these events. Each event type indicated in the figure is likely to affect biological systems at all temporal and spatial scales located to the left and below the specific event position in the figure. From Ummenhofer and Meehl (2017).

2.2.5.1 Changes in extreme temperatures, heat waves and drought

It is *very likely* that most land areas have experienced a decrease in the number of cold days and nights, and an increase in the number of warm days and unusually hot nights (Orlowsky and Seneviratne 2012; Seneviratne et al. 2012; Mishra et al. 2015; Ye et al. 2018). Although there is no consensus definition of heat waves as some heat wave indices have relative thresholds and others absolute thresholds, trends between indices of the same type show that recent heat-related events have been made more frequent or more intense due to anthropogenic greenhouse gas emissions in most land regions (Lewis and Karoly 2013; Smith et al. 2013b; Scherer and Diffenbaugh 2014; Fischer and Knutti 2015; Ceccherini et al. 2016; King et al. 2016;

Bador et al. 2016; Stott et al. 2016; King 2017; Hoegh-Guldberg et al. 2018). Globally, 50–80 % of the land fraction is projected to experience significantly more intense hot extremes than historically recorded (Fischer and Knutti 2014; Diffenbaugh et al. 2015; Seneviratne et al. 2016). There is *high confidence* that heat waves will increase in frequency, intensity and duration into the 21st century (Russo et al. 2016; Ceccherini et al. 2017; Herrera-Estrada and Sheffield 2017) and under high emission scenarios heat waves by the end of the century may become extremely long (more than 60 consecutive days) and frequent (once every two years) in Europe, North America, South America, Africa, Indonesia, the Middle East, south and south east Asia and Australia (Rusticucci 2012; Cowan et al. 2014; Russo et al. 2014; Scherer and Diffenbaugh 2014; Pal and Eltahir 2016; Rusticucci et al. 2016; Schär 2016; Teng et al. 2016; Dosio 2017; Mora et al. 2017; Dosio et al. 2018; Lehner et al. 2018; Lhotka et al. 2018; Lopez et al. 2018; Tabari and Willems 2018). Furthermore, unusual heat wave conditions today will occur regularly by 2040 under the RCP 8.5 scenario (Russo et al. 2016). The intensity of heat events may be modulated by the land cover and soil characteristics (Miralles et al. 2014; Lemordant et al. 2016; Ramarao et al. 2016). Where temperature increase results in decreased soil moisture, latent heat flux is reduced while sensible heat fluxes is increased allowing surface air temperature to rise further. However, this feedback may be diminished if the land surface is irrigated through the enhanced evapotranspiration (Mueller et al. 2015; Siebert et al. 2017)(Section 2.5.2.2).

Drought (Qin et al. 2013), including megadroughts of the last century, e.g., the Dustbowl drought (Hegerl et al. 2018), Chapter 5), is a normal component of climate variability (Hoerling et al. 2010; Dai 2011) and may be seasonal, multi-year (Van Dijk et al. 2013) or multi-decadal (Hulme 2001) with increasing degrees of impact on the regional activities. This interannual variability is controlled particularity through remote sea surface temperature (SST) forcings such as the Inter-decadal Pacific Oscillation (IPO) and the Atlantic Multi-decadal Oscillation (AMO), El Niño/Southern Oscillation (ENSO) and Indian Ocean Dipole (IOD), that cause drought as a result of reduced rainfall (Kelley et al. 2015; Dai 2011; Hoell et al. 2017; Espinoza et al. 2018). In some cases however, large scale SST modes do not fully explain the severity of drought some recent event attribution studies have identified a climate change fingerprint in several regional droughts, e.g., the western Amazon (Erfanian et al. 2017), southern Africa (Funk et al. 2018; Yuan et al. 2018), southern Europe and the Mediterranean including North Africa (Kelley et al. 2015; Wilcox et al. 2018), parts of North America (Williams et al. 2015; Mote et al. 2016), Russia (Otto et al. 2012), India (Ramarao et al. 2015) and Australia (Lewis and Karoly 2013).

Long-term global trends in drought are difficult to determine because of this natural variability, potential deficiencies in drought indices (especially in how evapotranspiration is treated) and the quality and availability of precipitation data (Sheffield et al. 2012; Dai 2013; Trenberth et al. 2014; Nicholls and Seneviratne 2015; Mukherjee et al. 2018). However, regional trends in frequency and intensity of drought are evident in several parts of the world, particularly in low latitude land areas, such as the Mediterranean, North Africa and Middle East (Vicente-Serrano et al. 2014; Spinoni et al. 2015a; Dai and Zhao 2017; Páscoa et al. 2017), many regions of sub-Saharan Africa (Masih et al. 2014; Dai and Zhao 2017), Central China (Wang et al. 2017e), the southern Amazon (Fu et al. 2013; Espinoza et al. 2018), India (Ramarao et al. 2016), east and south Asia, parts of North America and eastern Australia (Dai and Zhao 2017). A recent analysis of 4500 meteorological droughts globally found increased drought frequency over the U.S. East Coast, Amazonia and north-eastern Brazil, Patagonia, the Mediterranean region, most of Africa and north-eastern China with decreased drought frequency over northern Argentina, Uruguay and northern Europe (Spinoni et al. 2019). The study also found drought intensity has become more severe over north-western U.S., parts of Patagonia and southern Chile, the Sahel, the Congo River basin, southern Europe, north-eastern China, and south-eastern Australia, whereas the eastern U.S., south-eastern Brazil, northern Europe, and central-northern Australia experienced less severe droughts. In addition to the IPCC SR15 assessment of medium confidence in increased drying over the Mediterranean region (Hoegh-Guldberg et al. 2018), It is further assessed with *medium confidence* an increased frequency and intensity of drought in Amazonia and north-eastern Brazil, Patagonia, most of Africa and north-eastern China.

There is *low confidence* on ow large-scale modes of variability will respond to a warming climate (Deser et al. 2012; Liu 2012; Christensen et al. 2013; Hegerl et al. 2015; Newman et al. 2016). Although, there is evidence for an increased frequency of extreme ENSO events, such as the 1997/98 El Niño and 1988/89 La Niña (Cai et al. 2014a, 2015) and extreme positive phases of the IOD (Christensen et al. 2013; Cai et al. 2014b). However, The assessment by the SR15 was retained on an increased regional drought risk (*medium confidence)*, specifically over the Mediterranean and South Africa at both 1.5°C and 2°C warming levels compared to present day, with drought risk at 2°C being significantly higher than at 1.5°C (Hoegh-Guldberg et al. 2018).

2.2.5.2 Impacts of heat extremes and drought on land

There is *high confidence* that heat extremes such as unusually hot nights, extremely high daytime temperatures, heat waves and drought are damaging to crop production (Chapter 5). Extreme heat events impact a wide variety of tree functions including reduced photosynthesis, increased photooxidative stress, leaves abscise, a decreased growth rate of remaining leaves and decreased growth of the whole tree (Teskey et al. 2015). Although trees are more resilient to heat stress than grasslands (Teuling et al. 2010), it has been observed that different types of forest (e.g., needleleaf vs. broadleaf) respond differently to drought and heat waves (Babst et al. 2012). For example, in the Turkish Anatolian forests net primary productivity (NPP) generally decreased during drought and heat waves events between 2000 and 2010 but in a few other regions, NPP of needle leaf forests increased (Erşahin et al. 2016). However, forests may become less resilient to heat stress in future due to the long recovery period required to replace lost biomass and the projected increased frequency of heat and drought events (Frank et al. 2015a; McDowell and Allen 2015; Johnstone et al. 2016; Stevens-Rumann et al. 2018). Additionally, widespread regional tree mortality may be triggered directly by drought and heat stress (including warm winters) and exacerbated by insect outbreak and fire (Neuvonen et al. 1999; Breshears et al. 2005; Berg et al. 2006; Soja et al. 2007; Kurz et al. 2008b; Allen et al. 2010) .

Gross primary production (GPP) and soil respiration form the first and second largest carbon fluxes from terrestrial ecosystems to the atmosphere in the global carbon cycle (Beer et al. 2010; Bond-Lamberty and Thomson 2010). Heat extremes impact the carbon cycle through altering these and change ecosystem-atmosphere CO_2 fluxes and the ecosystem carbon balance. Compound heat and drought events result in a stronger carbon sink reduction compared to single-factor extremes as GPP is strongly reduced and ecosystem respiration less so (Reichstein et al. 2013; Von Buttlar et al. 2018). In forest biomes, however, GPP may increase temporarily as a result of increased insolation and photosynthetic activity as was seen during the 2015-2016 ENSO related drought over Amazonia (Zhu et al. 2018). Longer extreme events (heat wave or drought or both) result in a greater reduction in carbon sequestration and may also reverse long-term carbon sinks (Ciais et al. 2005; Phillips et al. 2009; Wolf et al. 2016b; Ummenhofer and Meehl 2017; Von Buttlar et al. 2018; Reichstein et al. 2013). Furthermore, extreme heat events may impact the carbon cycle beyond the lifetime of the event. These lagged effects can slow down or accelerate the carbon cycle: it will slow down if reduced vegetation productivity and/or widespread mortality after an extreme drought are not compensated by regeneration, or speed up if productive tree and shrub seedlings cause rapid regrowth after windthrow or fire (Frank et al. 2015a). Although some ecosystems may demonstrate resilience to a single heat climate stressor like drought, e.g. forests, compound effects of, e.g., deforestation, fire and drought potentially can result in changes to regional precipitation patterns and river discharge, losses of carbon storage and a transition to a disturbance-dominated regime (Davidson et al. 2012). Additionally, adaptation to seasonal drought may be overwhelmed by multi-year drought and their legacy effects (Brando et al. 2008; da Costa et al. 2010).

Under medium and high emission scenarios, global warming will exacerbate heat stress thereby amplifying deficits in soil moisture and runoff despite uncertain precipitation changes (Ficklin and Novick 2017; Berg and Sheffield 2018; Cook et al. 2018; Dai et al. 2018; Engelbrecht et al. 2015; Ramarao et al. 2015; Grillakis

2019). This will increase the rate of drying causing drought to set in quicker, become more intense and widespread, last longer and could result in an increased global aridity (Dai 2011; Prudhomme et al. 2014).

The projected changes in the frequency and intensity of extreme temperatures and drought is expected to result in decreased carbon sequestration by ecosystems and degradation of ecosystems health and loss of resilience (Trumbore et al. 2015). Also affected are many aspects of land functioning and type including agricultural productivity (Lesk et al. 2016a), hydrology (Mosley 2015; Van Loon and Laaha 2015), vegetation productivity and distribution (Xu et al. 2011; Zhou et al. 2014), carbon fluxes and stocks and other biogeochemical cycles (Frank et al. 2015b; Doughty et al. 2015; Schlesinger et al. 2016). Carbon stocks are particularly vulnerable to extreme events due to their large carbon pools and fluxes, potentially large lagged impacts and long recovery times to regain lost stocks (Frank et al. 2015a)(Section 2.2).

2.2.5.3 Changes in heavy precipitation

A large number of extreme rainfall events have been documented over the past decades (Coumou and Rahmstorf 2012; Seneviratne et al. 2012a; Trenberth 2012; Westra et al. 2013; Espinoza et al. 2014; Guhathakurta et al. 2017; Taylor et al. 2017; Thompson et al. 2017; Zilli et al. 2017). The observed shift in the trend distribution of precipitation extremes is more distinct than for annual mean precipitation and the global land fraction experiencing more intense precipitation events is larger than expected from internal variability (Fischer and Knutti 2014; Espinoza et al. 2018; Fischer et al. 2013) . As a result of global warming the number of record-breaking rainfall events globally has increased significantly by 12% during the period 1981 to 2010 compared to those expected due to natural multi-decadal climate variability (Lehmann et al. 2015) and the IPCC SR15 reports robust increases in observed precipitation extremes for annual maximum 1-day precipitation (RX1day) and consecutive 5-day precipitation (RX5day) (Hoegh-Guldberg et al. 2018; Schleussner et al. 2017). A number of extreme rainfall events have been attributed to human influence (Min et al. 2011; Pall et al. 2011; Sippel and Otto 2014; Trenberth et al. 2015; Krishnan et al. 2016) and the largest fraction of anthropogenic influence is evident in the most rare and extreme events (Fischer and Knutti 2014).

A warming climate is expected to intensify the hydrological cycle as a warmer climate facilitates more water vapour in the atmosphere, as approximated by the Clausius-Clapeyron (C-C) relationship, with subsequent effects on regional extreme precipitation events (Christensen and Christensen 2003; Pall et al. 2007; Berg et al. 2013; Wu et al. 2013; Guhathakurta et al. 2017; Thompson et al. 2017; Taylor et al. 2017; Zilli et al. 2017)(Manola et al. 2018). Furthermore, changes to the dynamics of the atmosphere amplify or weaken future precipitation extremes at the regional scale (O’Gorman 2015; Pfahl et al. 2017). Continued anthropogenic warming is very likely to increase the frequency and intensity of extreme rainfall in many regions of the globe (Seneviratne et al. 2012a; Mohan and Rajeevan 2017; Prein et al. 2017; Stott et al. 2016) although many GCMs underestimate observed increased trends in heavy precipitation suggesting a substantially stronger intensification of future heavy rainfall than the multi-model mean (Borodina et al. 2017; Min et al. 2011). Furthermore, the response of extreme convective precipitation to warming remains uncertain because GCMs and regional climate models (RCMs) are unable to explicitly simulate sub-grid scale processes such as convection, the hydrological cycle and surface fluxes and have to rely on parameterisation schemes for this (Crétat et al. 2012; Rossow et al. 2013; Wehner 2013; Kooperman et al. 2014; O’Gorman 2015; Larsen et al. 2016; Chawla et al. 2018; Kooperman et al. 2018; Maher et al. 2018; Rowell and Chadwick 2018). High-resolution regional climate models that explicitly resolve convection have a better representation of extreme precipitation but are dependent on the GCM to capture large scale environment in which the extreme event may occur (Ban et al. 2015; Prein et al. 2015; Kendon et al. 2017) . Interannual variability of precipitation extremes in the convective tropics are not well captured by global models (Allan and Liu 2018).

There is low confidence in the detection of long-term observed and projected seasonal and daily trends of extreme snowfall. The narrow rain–snow transition temperature range at which extreme snowfall can occur that is relatively insensitive to climate warming and subsequent large interdecadal variability (Kunkel et al. 2013; O'Gorman 2014, 2015).

2.2.5.4 Impacts of precipitation extremes on different land cover types

More intense rainfall leads to water redistribution between surface and ground water in catchments as water storage in the soil decreases (green water) and runoff and reservoir inflow increases (blue water) (Liu and Yang 2010; Eekhout et al. 2018). This results in increased surface flooding and soil erosion, increased plant water stress and reduced water security, which in terms of agriculture means an increased dependency on irrigation and reservoir storage (Nainggolan et al. 2012; Favis-Mortlock and Mullen 2011; García-Ruiz et al. 2011; Li and Fang 2016; Chagas and Chaffe 2018). As there is high confidence of a positive correlation between global warming and future flood risk, land cover and processes are likely to be negatively impacted, particularly near rivers and in floodplains (Kundzewicz et al. 2014; Alfieri et al. 2016; Winsemius et al. 2016; Arnell and Gosling 2016; Alfieri et al. 2017; Wobus et al. 2017).

In agricultural systems heavy precipitation and inundation can delay planting, increases soil compaction, and causes crop losses through anoxia and root diseases (Posthumus et al. 2009). In tropical regions flooding associated with tropical cyclones can lead to crop failure from both rainfall and storm surge. In some cases flooding can affect yield more than drought, particularly in tropical regions (e.g. India) and in some mid/high latitude regions such as China and central and northern Europe (Zampieri et al. 2017). Waterlogging of croplands and soil erosion also negatively affect farm operations and block important transport routes (Vogel and Meyer 2018; Kundzewicz and Germany 2012). Flooding can be beneficial in drylands if the floodwaters infiltrate and recharge alluvial aquifers along ephemeral river pathways, extending water availability into dry seasons and drought years and support riparian systems and human communities (Kundzewicz and Germany 2012; Guan et al. 2015). Globally, the impact of rainfall extremes on agriculture is less than that of temperature extremes and drought, although in some regions and for some crops, extreme precipitation explains a greater component of yield variability, e.g. of maize in the Mid-Western USA and southern Africa (Ray et al. 2015; Lesk et al. 2016b; Vogel et al. 2019) .

Although many soils on floodplains regularly suffer from inundation, the increases in the magnitude of flood events means that new areas with no recent history of flooding are now becoming severely affected (Yellen et al. 2014). Surface flooding and associated soil saturation often results in decreased soil quality through nutrient loss, reduced plant productivity, stimulates microbial growth and microbial community composition, negatively impacts soil redox and increases greenhouse gas emissions (Bossio and Scow 1998; Niu et al. 2014; Barnes et al. 2018; Sánchez-Rodríguez et al. 2019). The impact of flooding on soil quality is influenced by management systems that may mitigate or exacerbate the impact. Although soils tend to recover quickly after floodwater removal, the impact of repeated extreme flood events over longer timescales on soil quality and function is unclear (Sánchez-Rodríguez et al. 2017).

Flooding in ecosystems may be detrimental through erosion or permanent habitat loss, or beneficial, as a flood pulse brings nutrients to downstream regions (Kundzewicz et al. 2014). Riparian forests can be damaged through flooding; however, increased flooding may also be of benefit to forests where upstream water demand has lowered stream flow, but this is difficult to assess and the effect of flooding on forests is not well studied (Kramer et al. 2008; Pawson et al. 2013). Forests may mitigate flooding, however flood mitigation potential is limited by soil saturation and rainfall intensity (Pilaš et al. 2011; Ellison et al. 2017b). Some grassland species under heavy rainfall and soil saturated conditions responded negatively with decreased reproductive biomass and germination rates (Gellesch et al. 2017), however overall productivity in grasslands remains constant in response to heavy rainfall (Grant et al. 2014).

Extreme rainfall alters responses of soil CO_2 fluxes and CO_2 uptake by plants within ecosystems and therefore result in changes in ecosystem carbon cycling (Fay et al. 2008; Frank et al. 2015a). Extreme rainfall and flooding limits oxygen in soil which may suppress the activities of soil microbes and plant roots and lower soil respiration and therefore carbon cycling (Knapp et al. 2008; Rich and Watt 2013; Philben et al. 2015). However, the impact of extreme rainfall on carbon fluxes in different biomes differs. For example, extreme rainfall in mesic biomes reduces soil CO_2 flux to the atmosphere and GPP whereas in xeric biomes the opposite is true, largely as a result of increased soil water availability (Knapp and Smith 2001; Heisler and Knapp 2008; Heisler-White et al. 2009; Zeppel et al. 2014; Xu and Wang 2016; Liu et al. 2017b; Connor and Hawkes 2018).

As shown above greenhouse gas fluxes between the land and atmosphere are affected by climate. The next section assesses these fluxes in greater detail and the potential for land as a carbon sink.

Cross-Chapter Box 3: Fire and Climate Change

Raman Sukumar (India), Almut Arneth (Germany), Werner Kurz (Canada), Andrey Sirin (Russian Federation), Louis Verchot (Colombia/The United States of America)

Fires have been a natural part of Earth's geological past and its biological evolution since at least the late Silurian, about 400 million years ago (Scott 2000). Presently, roughly 3% of the Earth's land surface burns annually which affects both energy and matter exchanges between the land and atmosphere (Stanne et al. 2009). Climate is a major determinant of fire regimes through its control of fire weather, as well as through its interaction with vegetation productivity (fuel availability) and structure (fuel distribution and flammability) (Archibald et al. 2013) at the global (Krawchuk and Moritz 2011), regional (Pausas and Paula 2012) and local landscape (Mondal and Sukumar 2016) scales. Presently, humans are the main cause of fire ignition with lightning playing a lesser role globally (Bowman et al. 2017; Harris et al. 2016), although the latter factor has been predominantly responsible for large fires in regions such as the North American boreal forests (Veraverbeke et al. 2017). Humans also influence fires by actively extinguishing them, reducing spread and managing fuels.

Historical trends and drivers in land area burnt

While precipitation has been the major influence on wildfire regimes in pre-Industrial times, human activities have become the dominant drivers since then. There was less biomass burning during the 20th century than at any time during the past two millennia as inferred from charcoal sedimentary records (Doerr and Santín 2016), though there has been an increase in the most recent decades (Marlon et al. 2016). Trends in land area burnt have varied regionally (Giglio et al. 2013). Northern Hemisphere Africa has experienced a fire decrease of 1.7 Mha yr^{-1} (–1.4% yr^{-1}) since 2000, while Southern Hemisphere Africa saw an increase of 2.3 Mha yr^{-1} (+1.8% yr^{-1}) during the same period. Southeast Asia witnessed a small increase of 0.2 Mha yr^{-1} (+2.5% yr^{-1}) since 1997, while Australia experienced a sharp decrease of about 5.5 Mha yr^{-1} (–10.7% yr^{-1}) during 2001–2011, followed by an upsurge in 2011 that exceeded the annual area burned in the previous 14 years. A recent analysis using the Global Fire Emissions Database v.4 (GFED4s) that includes small fires concluded that the net reduction in land area burnt globally during 1998–2015 was –24.3±8.8% (-1.35±0.49% yr^{-1}) (Andela et al. 2017). However, from the point of fire emissions it is important to consider the land cover types which have experienced changes in area burned; in this instance, most of the declines have come from grasslands, savannas and other non-forest land cover types (Andela et al. 2017). Significant increases in forest area burned (with higher fuel consumption per unit area) have been recorded in western and boreal North America (Abatzoglou and Williams 2016; Ansmann et al. 2018) and in boreal Siberia (Ponomarev et al. 2016) in recent times. The 2017 and 2018 fires in British Columbia, Canada, were the largest ever recorded since the 1950s with 1.2 Mha and 1.4 Mha of forest burnt, respectively (Hanes et al. 2018) and smoke from these fires reaching the stratosphere over central Europe (Ansmann et al. 2018).

Climate variability and extreme climatic events such as severe drought, especially those associated with the

El Niño Southern Oscillation (ENSO), play a major role in fire upsurges as in equatorial Asia (Huijnen et al. 2016). Fire emissions in tropical forests increased by 133% on average during and following six El Niño years compared to six La Niña years during 1997–2016, due to reductions in precipitation and terrestrial water storage (Chen et al. 2017). The expansion of agriculture and deforestation in the humid tropics has also made these regions more vulnerable to drought-driven fires (Davidson et al. 2012; Brando et al. 2014). Even when deforestation rates were overall declining, as in the Brazilian Amazon during 2003–2015, the incidence of fire increased by 36% during the drought of 2015 (Aragão et al. 2018).

GHG emissions from fires

Emissions from wildfires and biomass burning are a significant source of greenhouse gases (CO_2, CH_4, N_2O), carbon monoxide (CO), carbonaceous aerosols, and an array of other gases including non-methane volatile organic compounds (NMVOC) (Akagi et al. 2011; Van Der Werf et al. 2010). GFED4s has updated fire-related carbon emission estimates biome-wise, regionally and globally, using higher resolution input data gridded at 0.25˚, a new burned area dataset with small fires, improved fire emission factors (Akagi et al. 2011; Urbanski 2014) and better fire severity characterisation of boreal forests (van der Werf et al. 2017). The estimates for the period 1997–2016 are 2.2 GtC yr^{-1}, being highest in the 1997 El Nino (3.0 GtC yr^{-1}) and lowest in 2013 (1.8 GtC yr^{-1}). Furthermore, fire emissions during 1997–2016 were dominated by savanna (65.3%), followed by tropical forest (15.1%), boreal forest (7.4%), temperate forest (2.3%), peatland (3.7%) and agricultural waste burning (6.3%) (van der Werf et al. 2017).

Fires not only transfer carbon from land to the atmosphere but also between different terrestrial pools: from live to dead biomass to soil, including partially charred biomass, charcoal and soot constituting 0.12–0.39 GtC yr^{-1} or 0.2–0.6% of annual terrestrial NPP (Doerr and Santín 2016). Carbon from the atmosphere is sequestered back into regrowing vegetation at rates specific to the type of vegetation and other environmental variables (Loehman et al. 2014). Fire emissions are thus not necessarily a net source of carbon into the atmosphere, as post-fire recovery of vegetation can sequester a roughly equivalent amount back into biomass over a time period of one to a few years (in grasslands and agricultural lands) to decades (in forests) (Landry and Matthews 2016). Fires from deforestation (for land use change) and on peatlands (which store more carbon than terrestrial vegetation) obviously are a net source of carbon from the land to the atmosphere (Turetsky et al. 2014); these types of fires were estimated to emit 0.4 GtC yr^{-1} in recent decades (van der Werf et al. 2017). Peatland fires dominated by smouldering combustion under low temperatures and high moisture conditions can burn for long periods (Turetsky et al. 2014).

Fires, land degradation/desertification and land-atmosphere exchanges

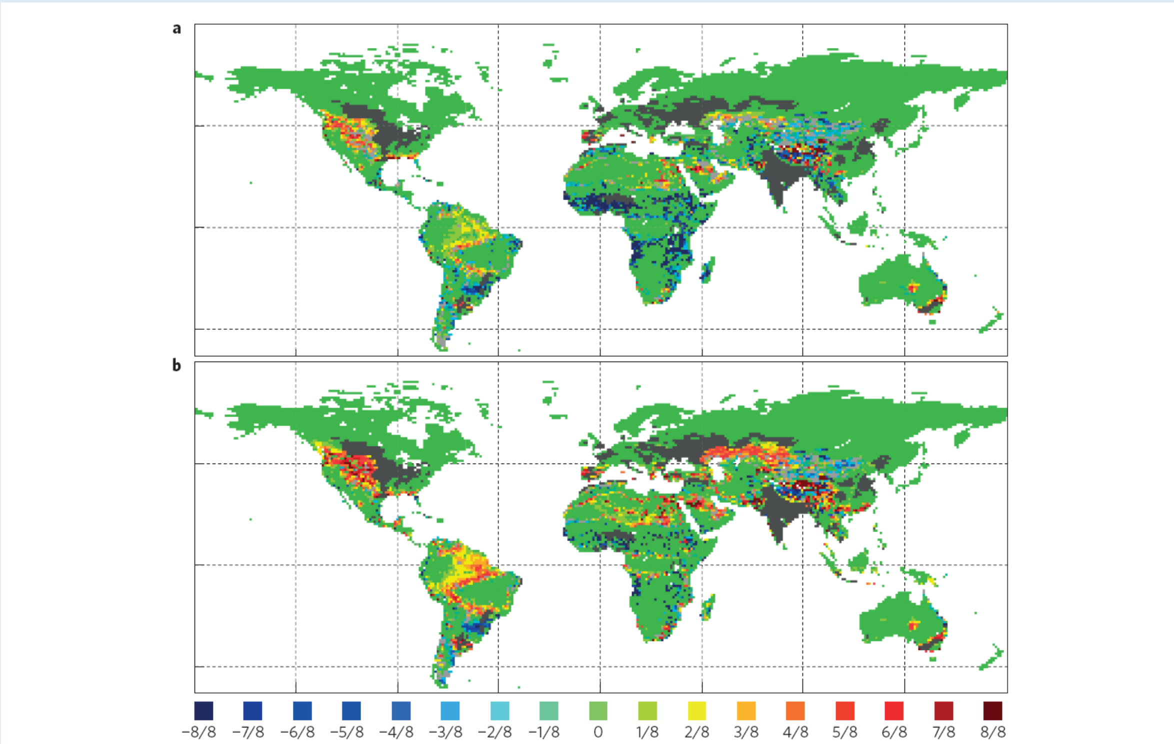
Flammable ecosystems are generally adapted to their specific fire regimes (Bond et al. 2005). A fire regime shift alters vegetation and soil properties in complex ways, both in the short- and the long-term, with consequences for carbon stock changes, albedo, fire-atmosphere-vegetation feedbacks and the ultimate biological capacity of the burnt land (Bond et al. 2004; Bremer and Ham 1999; MacDermott et al. 2016; Tepley et al. 2018; Moody et al. 2013; Veraverbeke et al. 2012) A fire-driven shift in vegetation from a forested state to an alternative stable state such as a grassland (Fletcher et al. 2014; Moritz 2015) with much less carbon stock is a distinct possibility. Fires cause soil erosion through action of wind and water (Moody et al. 2013) thus resulting in land degradation (see Chapter 4) and eventually desertification (see Chapter 3). Fires also affect carbon exchange between land and atmosphere through ozone (retards photosynthesis) and aerosol (slightly increases diffuse radiation) emissions; the net effect on global GPP during 2002-2011 is estimated to be –0.86±0.74 GtC yr^{-1} (Yue and Unger 2018).

Fires under future climate change

Temperature increase and precipitation decline would be the major driver of fire regimes under future climates as evapotranspiration increases and soil moisture decreases (Pechony and Shindell 2010; Aldersley et al. 2011; Abatzoglou and Williams 2016; Fernandes et al. 2017). The risk of wildfires in future could be expected to change, increasing significantly in North America, South America, central Asia, southern

Europe, southern Africa, and Australia (Liu et al. 2010). There is emerging evidence that recent regional surges in wildland fires are being driven by changing weather extremes, thereby signalling geographical shifts in fire proneness (Jolly et al. 2015). Fire weather season has already lengthened by 18.7% globally between 1979 and 2013, with statistically significant increases across 25.3% but decreases only across 10.7% of Earth's land surface covered with vegetation; even sharper changes have been observed during the second half of this period (Jolly et al. 2015). Correspondingly, the global area experiencing long fire weather season (defined as experiencing fire weather season greater than one standard deviation (SD) from the mean global value) has increased by 3.1% per annum or 108.1% during 1979–2013. Fire frequencies under 2050 conditions are projected to increase by approximately 27% globally, relative to the 2000 levels, with changes in future fire meteorology playing the most important role in enhancing global wildfires, followed by land cover changes, lightning activities and land use, while changes in population density exhibit the opposite effects (Huang et al. 2014).

However, climate is only one driver of a complex set of environmental, ecological and human factors in influencing fire regimes (Bowman et al. 2011a). While these factors lead to complex projections of future burnt area and fire emissions (Knorr et al. 2016a,b), human exposure to wildland fires could still increase because of population expansion into areas already under high risk of fires (Knorr et al. 2016a,b). There are still major challenges in projecting future fire regimes, and how climate, vegetation and socio/economic factors will interact (Hantson et al. 2016; Harris et al. 2016). There is also need for integrating various fire management strategies, such as fuel-reduction treatments in natural and planted forests, with other environmental and societal considerations to achieve the goals of carbon emissions reductions, maintain water quality, biodiversity conservation and human safety (Moritz et al. 2014; Gharun et al. 2017).



Cross-Chapter Box 3, Figure 1: The probability of low-fire regions becoming fire prone (positive values), or of fire-prone areas changing to a low-fire state (negative values) between 1971–2000 and 2017–2100 based on eight-Earth system model (ESM) ensembles, two Shared Socio-economic Pathways (SSPs; see (Jiang 2014)) and two Representative Concentration Pathways (RCPs). Light grey: areas where at least one ensemble simulation predicts a positive and one a negative change (lack of agreement). Dark grey: area with >50% past or future cropland. Fire-prone areas are defined as having a fire frequency of >0.01 yr^{-1} a RCP4.5 emissions with SSP3 demographics. b, RCP8.5 emissions with SSP5 demographics (Knorr et al. 2016a)

In summary, climate change is playing an increasing role in determining wildfire regimes along-side human activity (*medium confidence*), with future climate variability expected to enhance the risk and severity of wildfires in many biomes such as tropical rainforests (*high confidence*). Fire weather seasons have lengthened globally between 1979 and 2013 (*low confidence*). Global land area burned has declined in recent decades, mainly due to less burning in grasslands and savannas (*high confidence*). While drought remains the dominant driver of fire emissions, there has recently been increased fire activity in some tropical and temperate regions during normal to wetter than average years due to warmer temperatures that increase vegetation flammability (*medium confidence*). The boreal zone is also experiencing larger and more frequent fires, and this may increase under a warmer climate (*medium confidence*)

2.3 Greenhouse gas fluxes between land and atmosphere

Land is simultaneously a source and sink for several greenhouse gases (GHGs). Moreover, both natural and anthropogenic processes determine fluxes of GHGs, making it difficult to separate “anthropogenic” and “non-anthropogenic” emissions and removals. A meeting report by the (IPCC 2010a) divided the processes responsible for fluxes from land into three categories: (1) the *direct effects* of anthropogenic activity due to changing land cover and land management; (2) the *indirect effects* of anthropogenic environmental change, such as climate change, carbon dioxide (CO_2) fertilisation, nitrogen deposition; and (3) *natural* climate variability and natural disturbances (e.g. wildfires, windrow, disease). The meeting report (IPCC 2010a) noted that it was impossible with any direct observation to separate direct anthropogenic effects from non-anthropogenic (indirect and natural) effects in the land sector. As a result, different approaches and methods for estimating the anthropogenic fluxes have been developed by different communities to suit their individual purposes, tools and data availability.

The major GHGs exchanged between land and the atmosphere discussed in this chapter are CO_2 (2.3.1), methane (CH_4, Section 2.3.2) and nitrous oxide (N_2O, Section 2.3.3). We estimate the total emissions from Agriculture, Forestry, and Other Land Use (AFOLU) to be responsible for approximately 22% of global anthropogenic GHG emissions over the period 2003-2012 (Smith et al. 2013a; Ciais et al. 2013a) (Table 2.2). The estimate is similar to that reported in AR5 (*high confidence*), with slightly more than half these emissions coming as non-CO_2 GHGs from agriculture. Emissions from AFOLU have remained relatively constant since AR4, although their relative contribution to anthropogenic emissions has decreased due to increases in emissions from the energy sector.

Table 2.2 Summary of average annual land use fluxes aggregated over the decades 2001 to 2010 and 2007 to 2016. We present averages to smooth the effects of inter-annual variability.

Land use emissions	Mt CH_4 or N_2O	Gt CO_2e[1]	Mt CH_4 or N_2O	Gt CO_2e
	2001-2010		2007-2016	
Land use CO_2				
Bookkeeping model average		**4.69**		**4.84**
DGVM average		4.75		4.70
FAOSTAT		3.48		2.81
Non-CO_2 GHGs				

Agricultural CH_4				
FAOSTAT	130.44	3.65	135.42	3.79
USEPA[2]	141.64	3.97	147.83	4.14
EDGAR[3]	145.43	4.07	152.68	4.28
Average		**3.90**		**3.97**
Agricultural N_2O				
FAOSTAT	6.83	1.81	7.34	1.95
USEPA	8.69	2.30	9.32	2.47
EDGAR	5.92	1.57	6.31	1.67
Average		**1.89**		**2.21**
Total emissions from land use[4]		**10.48**		**11.02**
Total emissions from all sources[5]		45.29		50.72
Land use emissions: Total emissions		**23%**		**22%**

Data sources: Bookkeeping models: Hansis et al. 2015; Houghton & Nassikas 2017; DGVM average: Le Quéré et al. 2018; FAOSTAT: Tubiello et al. 2013; USEPA: USEPA 2012; EDGAR: Janssens-Maenhout et al. 2017.

[1] All values expressed in units of CO_2-eq are based on AR5 100 year Global Warming Potential values without climate-carbon feedbacks (N_2O = 265; CH_4 = 28).

[2] USEPA data are calculated from country data using IPCC Tier 1 methods (IPCC 2003) at 5-year increments through 2005 and are projected estimates from 2010 through 2030

[3] EDGAR data are complete only through 2012; 2007-2012 averages were computed in the second data column, for comparison purposes only. They were not used in the calculation of the average fluxes for the 2007-2016 period.

[4] Total land use emissions were calculated as the sum of the average emissions from the bookkeeping models and the average non-CO_2 GHG data from the different data sources (values in bold).

[5] Total emissions from all sources were calculated as the sum of total CO_2e emissions values for energy, industrial sources, waste and other emissions from the PRIMAP database (Gütschow et al. 2016). They do not include emissions from international bunkers and shipping.

2.3.1 Carbon Dioxide

This section is divided into four sub-sections (**Figure 2.4**): (1) the total net flux of CO_2 between land and atmosphere, (2) the contributions of AFOLU[1] fluxes and the non-AFOLU land sink to that total net CO_2 flux, (3) the gross emissions and removals comprising the net AFOLU flux, and (4) the gross emissions and removals comprising the land sink. Emissions to the atmosphere are positive; removals from the atmosphere are negative.

2.3.1.1 The total net flux of CO_2 between land and atmosphere

The net effects of all anthropogenic and non-anthropogenic processes on managed and unmanaged land result in a net removal of CO_2 from the atmosphere (*high confidence*). This total net land-atmosphere removal (defined here as *the total net land flux*) is estimated to have averaged 6.2 ± 2.6 $GtCO_2\ yr^{-1}$ from 2007 to 2016 (**Table 2.3**.) The estimate is determined from summing the AFOLU and non-AFOLU fluxes fluxes due to transient climate change, CO_2 fertilisation, nitrogen deposition calculated by models in the global carbon budget (Le Quéré et al. 2018) and is consistent with inverse modelling techniques based on atmospheric CO_2 concentrations and air transport (range: 5.1–8.8 $GtCO_2\ yr^{-1}$) (Peylin et al. 2013; Van Der Laan-Luijkx et al. 2017; Saeki and Patra 2017; Le Quéré et al. 2018) (See Box 2.2: for methods). A recent inverse analysis, considering carbon transport in rivers and oceans, found a net flux of CO_2 for land within this range, but a lower source from southern lands and a lower sink in northern lands (Resplandy et al. 2018).

The net removal of CO_2 by land has generally increased over the last 60 years in proportion to total

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emissions of CO_2 (*high confidence*). Although land has been a net sink for CO_2 since around the middle of last century, it was a net source to the atmosphere before that time, primarily as a result of emissions from AFOLU (Le Quéré et al. 2018).

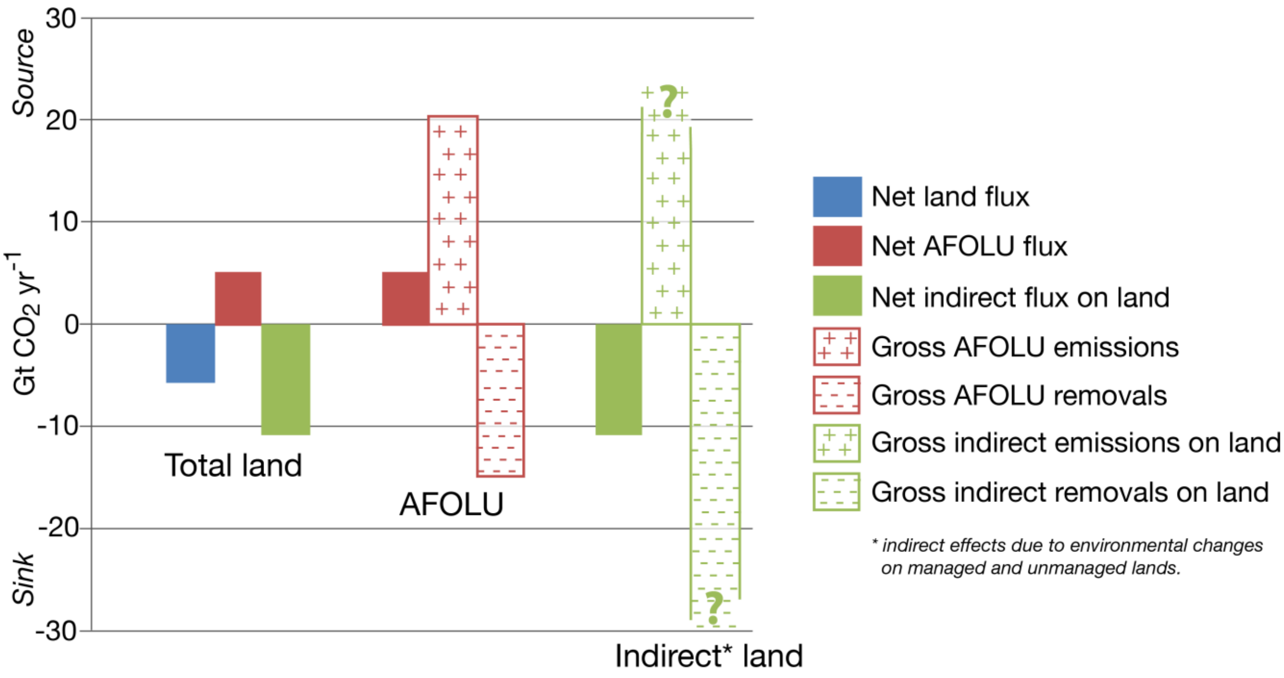

Figure 2.4 Net and gross fluxes of CO_2 from land (annual averages for 2008-2017). [Left]: The total net flux of CO_2 between land and atmosphere (blue) is shown with its two component fluxes: Net AFOLU emissions (red) and the net land sink (green) due to indirect environmental effects and natural effects on managed and unmanaged lands. [Middle]: The gross emissions and removals contributing to the net AFOLU flux. [Right]: The gross emissions and removals contributing to the land sink.

2.3.1.2 Separation of the total net land flux into AFOLU fluxes and the land sink

The total net flux of carbon between land and the atmosphere can be divided into fluxes due to direct human activities (i.e., AFOLU) and fluxes due to indirect anthropogenic and natural effects (i.e., the land sink) (**Table 2.3**). These two components are less certain than their sums, the total net flux of CO_2 between atmosphere and land. The land sink, estimated with DGVMs, is least certain (**Figure 2.5**).

Table 2.3 Perturbation of the global carbon cycle caused by anthropogenic activities ($GtCO_2\ yr^{-1}$) (from (Le Quéré et al. 2018).

	CO_2 flux ($GtCO_2\ yr^{-1}$), 10-year mean					
	1960–1969	1970–1979	1980–1989	1990–1999	2000–2009	2008–2017
Emissions						
Fossil CO_2 emissions	11.4 ± 0.7	17.2 ± 0.7	19.8 ± 1.1	23.1 ± 1.1	28.6 ± 1.5	34. ± 1.8
AFOLU net emissions	5.5 ± 2.6	4.4 ± 2.6	4.4 ± 2.6	5.1 ± 2.6	4.8 ± 2.6	5.5 ± 2.6
Partitioning						
Growth in atmosphere	6.2 ± 0.3	10.3 ± 0.3	12.5 ± 0.07	11.4 ± 0.07	14.7 ± 0.07	17.2 ± 0.07
Ocean sink	3.7 ± 1.8	4.8 ± 1.8	6.2 ± 1.8	7.3 ± 1.8	7.7 ± 1.8	8.8 ± 1.8
Land sink	4.4 ± 1.8	7.7 ± 1.5	6.6 ± 2.2	8.8 ± 1.8	9.9 ± 2.6	11.7 ± 2.6

(non-AFOLU)						
Budget imbalance	2.2	-1.1	-1.1	0.7	0.7	1.8
Total net land flux (AFOLU - Land sink)	+1.1 ± 3.2	–3.3 ± 3.0	–2.2 ± 3.4	–3.7 ± 2.2	–5.1 ± 3.2	–6.2 ± 3.7

2.3.1.2.1 Fluxes attributed to AFOLU

The modelled AFOLU flux was a net emission of 5.5 ± 2.6 $GtCO_2$ yr^{-1} for 2008–2017, approximately 14% of total anthropogenic CO_2 emissions (Le Quéré et al. 2018, **Table 2.3**). This net flux was due to direct anthropogenic activities, predominately tropical deforestation, but also afforestation/reforestation, and fluxes due to forest management (e.g. wood harvest) and other types of land management, including agriculture, grasslands and scrub. The AFOLU flux is the mean of two estimates from bookkeeping models (Hansis et al. 2015; Houghton and Nassikas 2017), and this estimated mean is consistent with the mean obtained from an assemblage of Dynamic Global Vegetation Models (DGVMs) (Le Quéré et al. 2018, Box 2.2:, **Figure 2.5**), although not all individual DGMVs include the same types of land use. Net CO_2 emissions from AFOLU have been relatively constant since 1900. AFOLU emissions were the dominant anthropogenic emissions until around the middle of the last century when fossil fuel emissions became dominant (Le Quéré et al. 2018). AFOLU activities have resulted in emissions of CO_2 over recent decades *(robust evidence, high agreement)* although there is a wide range of estimates from different methods and approaches (Smith et al. 2014; Houghton et al. 2012; Gasser and Ciais 2013; Pongratz et al. 2014; Tubiello et al. 2015; Grassi et al. 2018) (see Methods Box 2.2:, **Figure 2.5** and **Figure 2.7**).

DGVMs and one bookkeeping model (Hansis et al. 2015) used spatially explicit, harmonised land-use change data (LUH2) (Hurtt et al. 2017) based on HYDE 3.2. The HYDE data, in turn, are based on changes in the areas of croplands and pastures. In contrast, the Houghton bookkeeping approach (Houghton and Nassikas 2017) used primarily changes in forest area from the FAO Forest Resource Assessment (FAO 2015a) and FAOSTAT to determine changes in land use. To the extent that forests are cleared for land uses other than crops and pastures, estimates from Houghton and Nassikas (2017, 2018) are higher than estimates from DGMVs. In addition, both bookkeeping models (Hansis et al. 2015; Houghton and Nassikas 2017) included estimates of carbon emissions in SE Asia from peat burning from the Global Fire Emissions Database (GFED version 4, (Randerson et al. 2015)) and from peat drainage (Hooijer et al. 2010).

Satellite-based estimates of CO_2 emissions from loss of tropical forests during 2000-2010 corroborate the modelled emissions but are quite variable: 4.8 $GtCO_2$ yr^{-1} (Tyukavina et al. 2015), 3.0 $GtCO_2$ yr^{-1} (Harris et al. 2015), 3.2 $GtCO_2$ yr^{-1} (Achard et al. 2014) and 1.6 $GtCO_2$ yr^{-1} (Baccini et al. 2017). Differences in estimates can be explained to a large extent by the different approaches used. For example, the analysis by (Tyukavina et al. 2015) led to a higher estimate because they used a finer spatial resolution. Three of the estimates considered losses in forest area and ignored degradation and regrowth of forests. Baccini et al. (2017) in contrast, included both losses and gains in forest area and losses and gains of carbon within forests (i.e., forest degradation and growth). The four remote sensing studies cited above also reported committed emissions; i.e., all of the carbon lost from deforestation was assumed to be released to the atmosphere in the year of deforestation. In reality, only some of the carbon in trees is not released immediately to the atmosphere at the time of deforestation. The unburned portion is transferred to woody debris and wood products. Both bookkeeping models and DGVMs account for the delayed emissions in growth and decomposition. Finally, the satellite-based estimates do not include changes in soil carbon.

In addition to differences in land-cover data sets between models and satellites, there are many other methodological reasons for differences (See Box 2.2:) (Houghton et al. 2012; Gasser and Ciais 2013; Pongratz et al. 2014; Tubiello et al. 2015). There are different definitions of land-cover type, including forest (e.g. FAO uses a tree cover threshold for forests of 10%; Tyukavina et al. (2017) used 25%), different estimates of biomass and soil carbon density (Mg C ha^{-1}), different approaches to tracking emissions through time (legacy effects), and different types of activity included (e.g. forest harvest, peatland drainage and fires). Most DGVMS only recently (since AR5) included forest management processes, such as tree harvesting and land clearing for shifting cultivation, leading to larger estimates of CO_2 emissions than when these processes are not considered (Arneth et al. 2017; Erb et al. 2018). Grazing management has likewise been found to have large effects (Sanderman et al. 2017), and is not included in most DGVMs (Pugh et al. 2015; Pongratz et al., 2018).

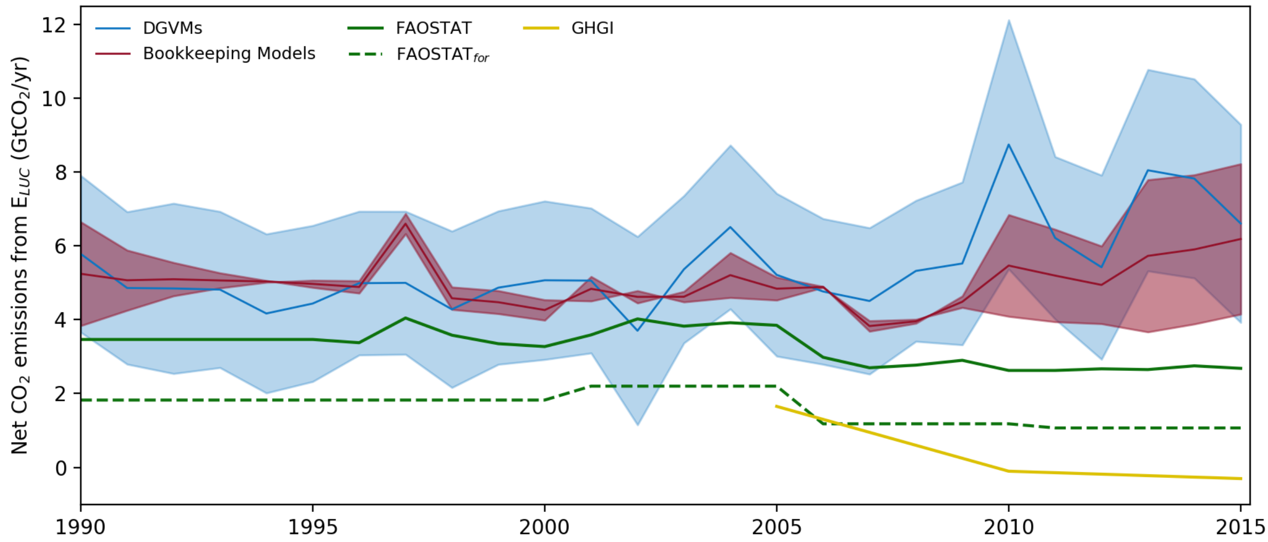

Figure 2.5 Global net CO_2 emissions due to AFOLU from different approaches (in $GtCO_2$ yr^{-1}). Red line: the mean and individual estimates from two bookkeeping models (Houghton and Nassikas 2017; Hansis et al. 2015). Blue line: the mean from DGVMs run with the same driving data with the pale blue shading showing the ±1 standard deviation range. Green line: data downloaded from FAOSTAT website (Tubiello et al. 2013); the dashed line is primarily forest-related emissions, while the solid green line also includes emissions from peat fires and peat draining. Yellow line: Greenhouse Gas Inventories (GHGI) based on country reports to UNFCCC (Grassi et al. 2018), data are shown only from 2005 because reporting in many developing countries became more consistent/reliable after this date. For more details on methods see Box 2.2:

2.3.1.2.2 Nationally reported Greenhouse Gas Inventories (GHGI) values versus global model estimates

There are large differences globally (**Figure 2.5**), between estimates of net anthropogenic land-atmosphere fluxes of CO_2 from national GHGIs and from global models, and the same is true in many regions (**Figure 2.5**). Fluxes reported to the UNFCCC through country GHGIs were noted as about 4.3 $GtCO_2$ yr^{-1} lower (Grassi et al. 2018) than estimates from the bookkeeping model (Houghton et al. 2012a) used in the carbon budget for AR5 (Conway 2012; Ciais et al. 2013a). The anthropogenic emissions of CO_2 from AFOLU reported in countries' GHG inventories were 0.1 ± 1.0 $GtCO_2$ yr^{-1} globally during 2005 to 2014 (*low confidence*) (Grassi et al. 2018) much lower than emission estimates from the two global bookeeping models of 5.1 ± 2.6 $GtCO_2$ yr^{-1} over the same time period (Quéré et al. 2018). Transparency and comparability in estimates can support measuring, reporting and verifying GHG fluxes under the UNFCCC, and also the global stocktake, which will assess globally the progress towards achieving the long-term goals of the Paris Agreement. These differences can be reconciled largely by taking account of the different approaches to defining anthropogenic in terms of different areas of land and treatment of indirect environmental change (Grassi et al. 2018).

To date there has been one study that quantitatively reconciles the global model estimates with GHGIs (Grassi et al. 2018). The separation of anthropogenic from non-anthropogenic effects is impossible with direct observation (IPCC 2010a). The different approaches of models and GHGIs to estimating anthropogenic emissions and removals are shown in (**Figure 2.6**). The difficulty is that *indirect* effects of environmental changes (e.g. climate change and rising atmospheric CO_2) affect both manged and unmanaged lands, and some approaches treat these as anthropogenic while others do not. Bookkeeping models (e.g. Houghton and Nassikas 2017) attempt to estimate the fluxes of CO_2 driven by direct anthropogenic effects alone. DGVMs model the *indirect* environmental effects of climate and CO_2. If the indirect effects happen on land experiencing anthropogenic land cover change or management (harvest and regrowth), DGVMs treat this as anthropogenic. Country GHGIs separately report fluxes due to land conversion (e.g. forests to croplands) and fluxes due to land management (e.g. forest land remaining forest land). The "managed land proxy" is used as a pragmatic approach to estimate anthropogenic fluxes on managed lands, whereby

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countries define the areas they consider managed, and include all of the emission and removals that occur on those lands. Emissions and removals are caused simultaneously by direct, indirect and natural drivers and are captured in the reporting, which often relies on inventories.

Grassi et al. (2018) demonstrated that estimates of CO_2 emissions from global models and from nationally reported GHGIs were similar for deforestation and afforestation, but different for managed forests. Countries generally reported larger areas of managed forests than the models, and the carbon removals by these managed forests were also larger. The flux due to indirect effects on managed lands was quantified using post-processing of results from DGVMs, looking at the *indirect* effects of CO_2 and climate change on secondary forest areas. The derived DGVM *indirect* managed forest flux was found to account for most of the difference between the bookkeeping models and the inventories.

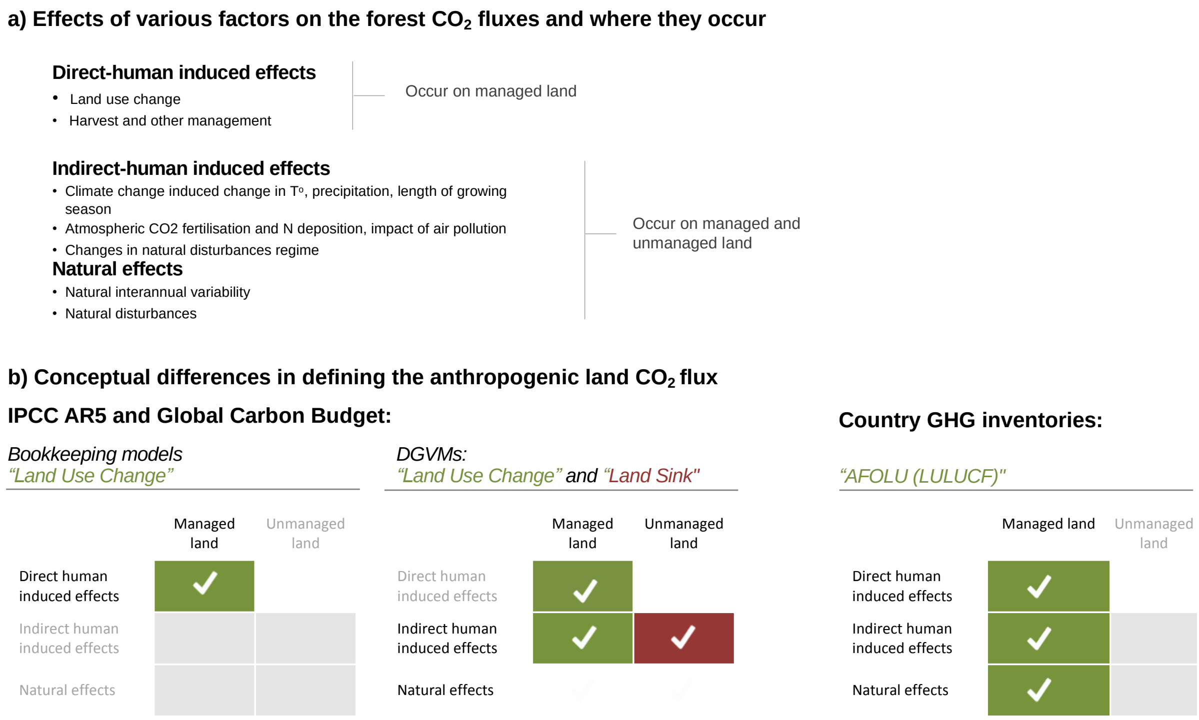

Figure 2.6 Summary of the main conceptual differences between GHG Inventories and global models in considering what is the "anthropogenic land CO2 flux". Adapted from Grassi et al. (2018): a) Effects of key processes on the land flux as defined by IPCC (2010) including where these effects occur (in managed and/or unmanaged lands); How these effects are captured in '(a) bookkeeping models that do not explicitly model the effects of environmental change, although some is implicitly captured in data on carbon densities and growth and decay rates; (b) Dynamic Global Vegetation Models (DGVMs) that include the effects of environmental change on all lands, and run the models with and without land use change to diagnose "land use change", the "land sink" is then conceptually assumed to be a natural response of land to the anthropogenic perturbation of environmental change, models include the effects of inter-annual climate variability, and some include fires but no other natural disturbances (c) GHG Inventories reported by countries to the UNFCCC that report all fluxes in areas the countries define as "managed land" but do not report unmanaged land. This is the CO_2 flux due to Land Use Land Use Change and Forestry (LULUCF) which is a part of the overall AFOLU (Agriculture, Forestry and Other Land Use) flux. The area of land considered as managed in the inventories is greater than that considered as subject to direct management activities (harvest and regrowth) in the models.

2.3.1.2.3 Regional differences

Figure 2.7 shows regional differences in emissions due to AFOLU. Recent increases in deforestation rates in some tropical countries have been partially balanced by increases in forest area in India, China, the USA and

Europe (FAO-FRA 2015). The trend in emissions from AFOLU since the 1990s is *uncertain* because some data suggest a declining rate of deforestation (FAO-FRA 2015), while data from satellites suggest an increasing rate (Kim 2014; Hansen et al. 2012). The disagreement results in part from differences in the definition of forest and approaches to estimating deforestation. The FAO defines deforestation as the conversion of forest to another land use (FAO-FRA 2015), while the measurement of forest loss by satellite may include wood harvests (forests remaining forests) and natural disturbances that are not directly caused by anthropogenic activity (e.g., forest mortality from droughts and fires). Trends in anthropogenic and natural disturbances may be in opposite directions. For example, recent drought-induced fires in the Amazon have increased the emissions from wildfires at the same time that emissions from anthropogenic deforestation have declined (Aragão et al. 2018). Furthermore, there have been advances since AR5 in estimating the GHG effects of different types of forest management (e.g. (Valade et al. 2017). Overall, there is *robust evidence and high agreement* for a net loss of forest area and tree cover in the tropics and a net gain, mainly of secondary forests and sustainably managed forests, in the temperate and boreal zones (Chapter 1).

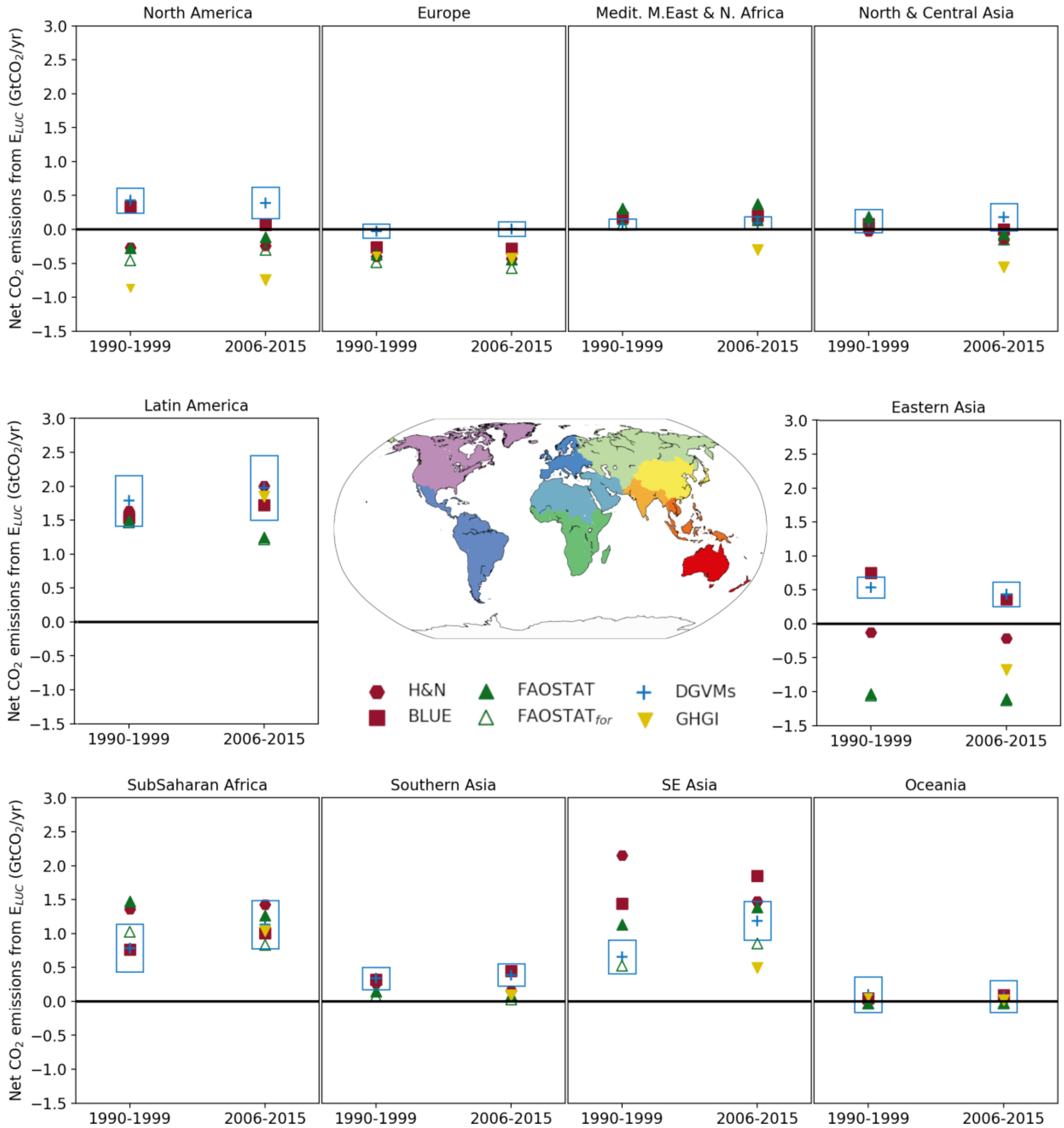

Figure 2.7 Regional trends in net anthropogenic land-atmosphere CO_2 flux from a range of different approaches (in $GtCO_2$ yr^{-1}). Red symbols: bookkeeping models (hexagon - Houghton and Nassikas 2017;

square - Hansis et al. 2015). Blue cross: the mean from DGMVs with the box showing the 1 standard deviation range. Green triangles: downloaded from FAOSTAT website; the open triangle is primarily forest-related emissions, while the closed triangle includes emission from peat fires and peat drainage. Yellow inverted triangle: Greenhouse Gas Inventories (GHGI) LULUCF flux (Land Use Land Use Change and Forestry – part of AFOLU) based on country reports to UNFCCC (Grassi et al. 2018) – data for developing countries is only shown for 2006-2015 because reporting in many developing countries became more consistent/reliable after 2005. For more details on methods see Box 2.2:.

2.3.1.2.4 Processes responsible for the land sink

Just over half of total net anthropogenic CO_2 emissions (AFOLU and fossil fuels) were taken up by oceanic and land sinks (Table 2.3) *(robust evidence, high agreement)*. The land sink was referred to in AR5 as the “residual terrestrial flux,” as it was not estimated directly, but calculated by difference from the other directly estimated fluxes in the budget (Table 2.3). In the 2018 budget (Le Quéré et al. 2018), the land sink term was instead estimated directly by DGVMs, leaving a budget imbalance of 2.2 $GtCO_2$ yr^{-1} (sources overestimated or sinks underestimated). The budget imbalance may result from variations in oceanic uptake, or from uncertainties in fossil fuel or AFOLU emissions, as well as from land processes not included in DGVMs.

The land sink is thought to be driven largely by the indirect effects of environmental change (e.g., climate change, increased atmospheric CO_2 concentration nitrogen deposition) on unmanaged and managed lands (*robust evidence*, *high agreement*). The land sink has generally increased since 1900 and was a net sink of 11.7 ± 3.7 $GtCO_2$ yr^{-1} during the period 2008 to 2017 (Table 2.3), absorbing 29% of global anthropogenic emissions of CO_2. The land sink has slowed the rise in global land-surface air temperature by 0.09 ± 0.02°C since 1982 (*medium confidence*) (Zeng et al. 2017).

The rate of CO_2 removal by land accelerated from -0.026 ± 0.24 $GtCO_2$ yr^{-1} during the warming period (1982 to 1998) to -0.436 ± 0.260 $GtCO_2$ yr^{-1} during the warming hiatus (1998-2012). One explanation is that respiration rates were lower during the warming hiatus (Ballantyne et al. 2017). However, the lower rate of growth in atmospheric CO_2 during the warming hiatus may have resulted, not from lower rates of respiration, but from declining emissions from AFOLU (lower rates of tropical deforestation and increased forest growth in northern mid-latitudes (Piao et al. 2018). Changes in the growth rate of atmospheric CO_2, by themselves, do not identify the processes responsible, and the cause of the variation is uncertain.

While year-to-year variability in the indirect land sink is high in response to climate variability, DGVM fluxes are far more influenced on decadal time scales by CO_2 fertilisation. A DGVM intercomparison (Sitch et al. 2015) for 1990 to 2009 found that CO_2 fertilisation alone contributed a mean global removal of –10.54 ± 3.68 $GtCO_2$ yr^{-1} (trend –0.444 ± 0.202 $GtCO_2$ yr^{-1}). Data from forest inventories around the world corroborate the modelled land sink (Pan et al. 2011a). The geographic distribution of the non-AFOLU land sink is less certain. While it seems to be distributed globally, its distribution between the tropics and non-tropics is estimated to be between 1:1 (Pan et al. 2011a) and 1:2 (Houghton et al. 2018).

As described in Box 2.3, rising CO_2 concentrations have a fertilising effect on land, while climate has mixed effects; e.g., rising temperature increases respiration rates and may enhance or reduce photosynthesis depending on location and season, while longer growing seasons might allow for higher carbon uptake. However, these processes are not included in DGVMs, which may account for at least some of the land sink. For example, a decline in the global area burned by fires each year (Andela et al. 2017) accounts for an estimated net sink (and/or reduced emissions) of 0.5 $GtCO_2$ yr^{-1} (Arora and Melton 2018) (*limited evidence, medium agreement*) (boreal forests represent an exception to this decline (Kelly et al. 2013)). The reduction in burning not only reduces emissions; it also allows more growth of recovering forests. There is also an estimated net carbon sink of about the same magnitude (0.5 $GtCO_2$ yr^{-1}) as a result of soil erosion from agricultural lands and redeposition in anaerobic environments where respiration is reduced (Wang et al. 2017d) (*limited evidence, low agreement*). A recent study attributes an increase in land carbon to a longer-term (1860-2005) aerosol-induced cooling (Zhang et al. 2019). Recent evidence also suggests that DGVMs and Earth System Models underestimate the effects of drought on CO_2 emissions (Humphrey et al. 2018; Green et al. 2019; Kolus et al. 2019).

2.3.1.3 Gross emissions and removals contributing to AFOLU emissions

The modelled AFOLU flux of 5.5 ± 3.7 $GtCO_2$ yr^{-1} over the period 2008 to 2017 represents a net value. It consists of both gross emissions of CO_2 from deforestation, forest degradation, and the oxidation of wood products, as well as gross removals of CO_2 in forests and soils recovering from harvests and agricultural abandonment (Figure 2.4). The uncertainty of these gross fluxes is high because few studies report gross fluxes from AFOLU. Houghton and Nassikas (2017) estimated gross emissions to be as high as 20.2 $GtCO_2$ yr^{-1} (*limited evidence, low agreement*) (Figure 2.4), and even this may be an underestimate because the land-use change data used from FAOSTAT (Tubiello et al. 2013) is itself a net of all changes within a country.

Gross emissions and removals of CO_2 result from rotational uses of land, such as wood harvest and shifting cultivation, including regrowth. These gross fluxes are more informative for assessing the timing and potential for mitigation than estimates of net fluxes, because the gross fluxes include a more complete accounting of individual activities. Gross emissions from rotational land use in the tropics are approximately 37% of total CO_2 emissions, rather than 14%, as suggested by net AFOLU emissions (Houghton and Nassikas 2018). Further, if the forest is replanted or allowed to regrow, gross removals of nearly the same magnitude would be expected to continue for decades.

2.3.1.4 Gross emissions and removals contributing to the non-anthropogenic land sink

The *net* land sink averaged 11.7 $GtCO_2$ yr^{-1} over 2008-2017 (*robust evidence, medium agreement*) (Table 2.3.2), but its gross components have not been estimated at the global level. There are many studies that suggest increasing emissions of carbon due to indirect environmental effects and natural disturbance, for example temperature-induced increases in respiration rates (Bond-Lamberty et al. 2018); increased tree mortality (Brienen et al. 2015; Berdanier and Clark 2016; McDowell et al. 2018); and thawing permafrost (Schuur et al. 2015). The global carbon budget indicates that land and ocean sinks have *increased* over the last six decades in proportion to total CO_2 emissions (Le Quéré et al. 2018) (*robust evidence, high agreement*). That means that any emissions must have been balanced by even larger removals (likely driven by CO_2 fertilisation, climate change, nitrogen deposition, erosion and redeposition of soil carbon, a reduction in areas burned, aerosol-induced cooling, and changes in natural disturbances, Box 2.3)

Climate change is expected to impact terrestrial biogeochemical cycles via an array of complex feedback mechanisms that will act to either enhance or decrease future CO_2 emissions from land. Because the gross emissions and removals from environmental changes are not constrained at present, the balance of future positive and negative feedbacks remains uncertain. Estimates from climate models included in AR5, CMIP5 (Coupled Model Intercomparison Project, 5), exhibit large differences for different carbon and nitrogen cycle feedbacks and how they change in a warming climate (Anav et al. 2013; Friedlingstein et al. 2006; Friedlingstein, et al. 2014) . The differences are in large part due to the uncertainty regarding how primary productivity and soil respiration will respond to environmental changes, with many of the models not even agreeing on the sign of change. Furthermore, many models do not include a nitrogen cycle, which may limit the CO_2 fertilisation effect in the future (see Box 2.3). There is an increasing amount of observational data available and methods to constrain models (e.g. Cox et al. 2013; Prentice, et al., 2015) which can reduce uncertainty.

2.3.1.5 Potential impact of mitigation on atmospheric CO_2 concentrations

If CO_2 concentrations decline in the future as a result of low emissions and large negative emissions, the global land and ocean sinks are expected to weaken (or even reverse). The oceans are expected to release CO_2 back to the atmosphere when the concentration declines (Ciais et al. 2013a; Jones et al. 2016). This means that to maintain atmospheric CO_2 and temperature at low levels, both the excess CO_2 from the atmosphere and the CO_2 progressively outgassed from the ocean and land sinks will need to be removed. This outgassing from the land and ocean sinks is called the “rebound effect” of the global carbon cycle (Ciais et al. 2013a). It will reduce the effectiveness of negative emissions and increase the deployment level needed to achieve a climate stabilisation target (Jackson et al. 2017; Jones et al. 2016) (*limited evidence, high agreement*).

2.3.2 Methane

2.3.2.1 Atmospheric trends

In 2017, the globally averaged atmospheric concentration of CH_4 was 1850 ± 1 ppbv (Figure 2.8A). Systematic measurements of atmospheric CH_4 concentrations began in the mid-1980s and trends show a steady increase between the mid-1980s and early-1990s, slower growth thereafter until 1999, a period of no growth between 1999 and 2006, followed by a resumption of growth in 2007. The growth rates show very high inter-annual variability with a negative trend from the beginning of the measurement period until about 2006, followed by a rapid recovery and continued high inter-annual variability through 2017 (Figure 2.8B). The growth rate has been higher over the past 4 years (*high confidence*) (Nisbet et al. 2019). The trend in $\delta^{13}C$-CH_4 prior to 2000 with less depleted ratios indicated that the increase in atmospheric concentrations was due to thermogenic (fossil) CH_4 emissions; the reversal of this trend after 2007 indicates a shift to biogenic sources (Figure 2.8C).

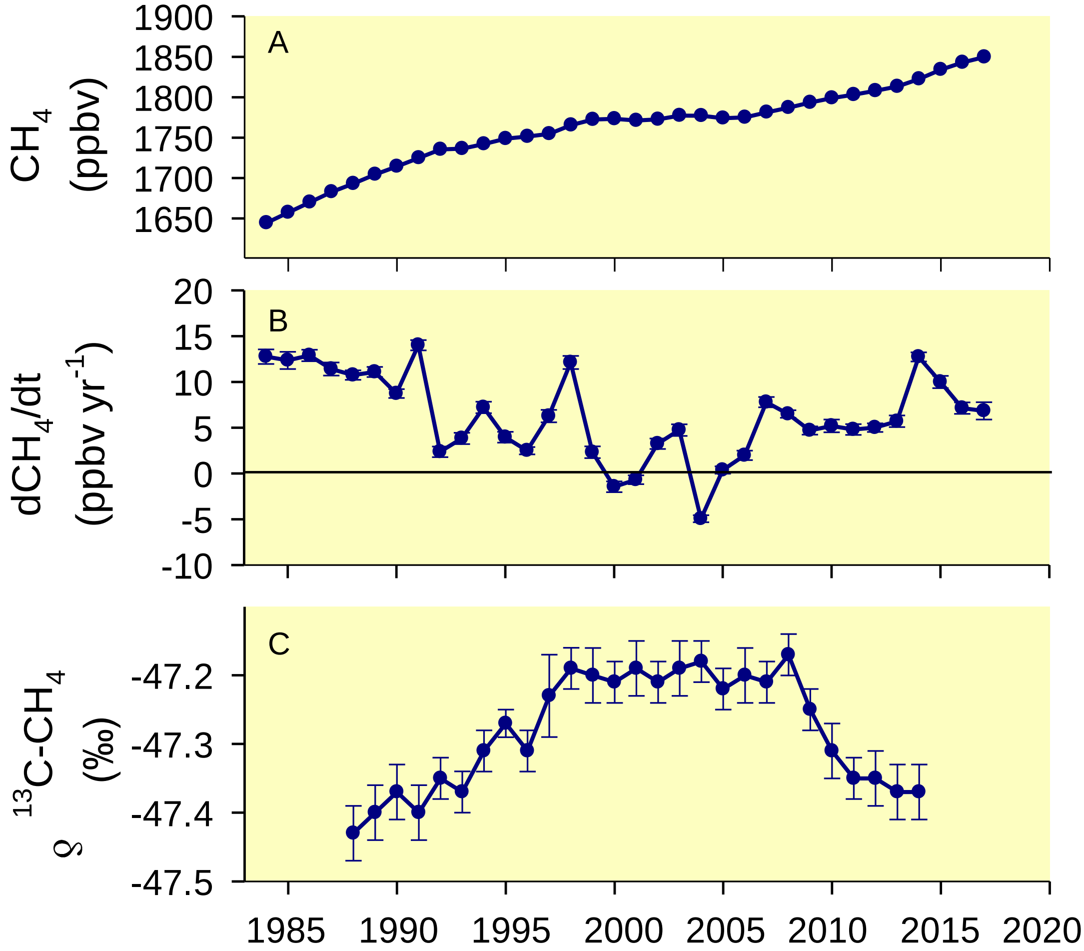

Figure 2.8 Globally averaged atmospheric CH_4 mixing ratios (Frame A) and instantaneous rates of change (Frame B) and C isotope /variation (Frame C) Data sources: NOAA/ESRL (www.esrl.noaa.gov/gmd/ccgg/trends_ch4/)(Dlugokencky et al. 1994) and Schaefer et al. (2016).

Understanding the underlying causes of temporal variation in atmospheric CH_4 concentrations is an active area of research. Several studies concluded that inter-annual variability of CH_4 growth was driven by variations in natural emissions from wetlands (Rice et al. 2016; Bousquet et al. 2006; Bousquet et al. 2011; Bousquet et al. 2011b). These modelling efforts concluded that tropical wetlands were responsible for between 50 and 100% of the inter-annual fluctuations and the renewed growth in atmospheric concentrations after 2007. However, results were inconsistent for the magnitude and geographic distribution of the wetland sources between the models. Pison et al. (2013) used two atmospheric inversion models and the ORCHIDEE model and found greater uncertainty in the role of wetlands in inter-annual variability between 1990 and 2009 and during the 1999-2006 pause. Poulter et al. (2017) used several of biogeochemical models and inventory-based wetland area data to show that wetland CH_4 emissions increases in the boreal zone have been offset by decreases in the tropics and concluded that wetlands have not contributed significantly to renewed atmospheric CH_4 growth.

The models cited above assumed that atmospheric hydroxyl radical (OH) sink over the period analysed did not vary. OH reacts with CH_4 as the first step toward oxidation to CO_2. In global CH_4 budgets, the atmospheric OH sink has been difficult to quantify because its short lifetime (~1 second) and its distribution is controlled by precursor species that have non-linear interactions (Taraborrelli et al., 2012; Prather et al., 2017). Understanding of the atmospheric OH sink has evolved recently. The development of credible time series of methyl chloroform (MCF: CH_3CCl_3) observations offered a way understand temporal dynamics of OH abundance and applying this to global budgets further weakened the argument for the role of wetlands in determining temporal trends since 1990. Several authors used the MCF approach and concluded that changes in the atmospheric OH sink explained a large portion of the suppression in global CH_4 concentrations relative to the pre-1999 trend (Turner et al. 2017; Rigby et al. 2013; McNorton et al. 2016). These studies could not reject the null hypothesis that OH has remained constant in recent decades and they did not suggest a mechanism for the inferred OH concentration changes (Nisbet et al. 2019). Nicely et al. (2018) used a mechanistic approach and demonstrated that variation in atmospheric OH was much lower than what MCF studies and found that positive trends in OH due to the effects of water vapour, nitrogen oxides (NO_x), tropospheric ozone, and expansion of the tropical Hadley cells offsets the decrease in OH that is expected from increasing atmospheric CH_4 concentrations.

The depletion of $\delta^{13}C_{atm}$ beginning in 2009 could be due to changes in several sources. Decreased fire emissions combined with increased tropical wetland emissions compared to earlier years could explain the $\delta^{13}C$ perturbations to atmospheric CH_4 sources (Worden et al. 2017; Schaefer et al. 2016). However, because tropical wetland emissions are higher in the Southern Hemisphere, and the remote sensing observations show that CH_4 emissions increases are largely in the north tropics (Bergamaschi et al. 2013; Melton et al. 2013; Houweling et al. 2014), an increased wetland source does not fit well with the southern hemisphere $\delta^{13}C$ observations. New evidence shows that tropical wetland CH_4 emissions are significantly underestimated, perhaps by a factor of 2, because estimates do not account for release by tree stems (Pangala et al. 2017). Several authors have concluded that agriculture is a more probable source of increased emissions, and particularly from rice and livestock in the tropics, which is consistent with inventory data (Wolf et al. 2017; Patra et al. 2016; Schaefer et al. 2016).

The importance of fugitive emissions in the global atmospheric accumulation rate is growing (*medium evidence, high agreement*). The increased production of natural gas in the US from the mid 2000's is of particular interest because it coincides with renewed atmospheric CH_4 growth (Rice et al. 2016; Hausmann et al. 2015). Reconciling increased fugitive emissions with increased isotopic depletion of atmospheric CH_4 indicates that there are *likely* multiple changes in emissions and sinks that affect atmospheric accumulation (*medium confidence*).

With respect to atmospheric CH_4 growth rates, we conclude that there is significant and ongoing accumulation of CH_4 in the atmosphere (*very high confidence*). The reason for the pause in growth rates and subsequent renewed growth is at least partially associated with land use and land use change. Evidence that variation in the atmospheric OH sink plays a role in the year to year variation of the CH_4 is accumulating, but results are contradictory (*medium evidence, low agreement*) and refining this evidence is constrained by lack of long-term isotopic measurements at remote sites, particularly in the tropics. Fugitive emissions *likely* contribute to the renewed growth after 2006 (*medium evidence, high agreement*). Additionally, the recent depletion trend of ^{13}C isotope in the atmosphere indicates that growth in biogenic sources explains part of the current growth and that biogenic sources make up a larger proportion of the source mix compared to the period before 1997 (*robust evidence, high agreement*). In agreement with the findings of AR5, we conclude that wetlands are important drivers of inter-annual variability and current growth rates (*medium evidence, high agreement*). Ruminants and the expansion of rice cultivation are also important contributors to the current growth trend (*medium evidence, high agreement*).

2.3.2.2 Land use effects

Agricultural emissions are predominantly from enteric fermentation and rice, with manure management and waste burning contributing small amounts (Figure 2.9). Since 2000, livestock production has been responsible for 33% of total global emissions and 66% of agricultural emissions (Source: EDGAR 4.3.2 database, accessed May 2018, (USEPA 2012; Tubiello et al. 2014; Janssens-Maenhout et al. 2017b). Asia has the largest livestock emissions (37%) and emissions in the region have been growing by around 2% per year over the same period. North America is responsible for 26% and emissions are stable; Europe is responsible for around 8% of emissions, and these are decreasing slightly (<1% per year). Africa is responsible for 14%, but emissions are growing fastest in this region at around 2.5% y^{-1}. In Latin America and the Caribbean, livestock emissions are decreasing at around 1.6% per year and the region makes up 16% of emissions. Rice emissions are responsible for about 24% of agricultural emissions, and 89% of these are from Asia. Rice emissions are increasing by 0.9% per year in that region. These trends are predicted to continue through 2030 (USEPA 2013).

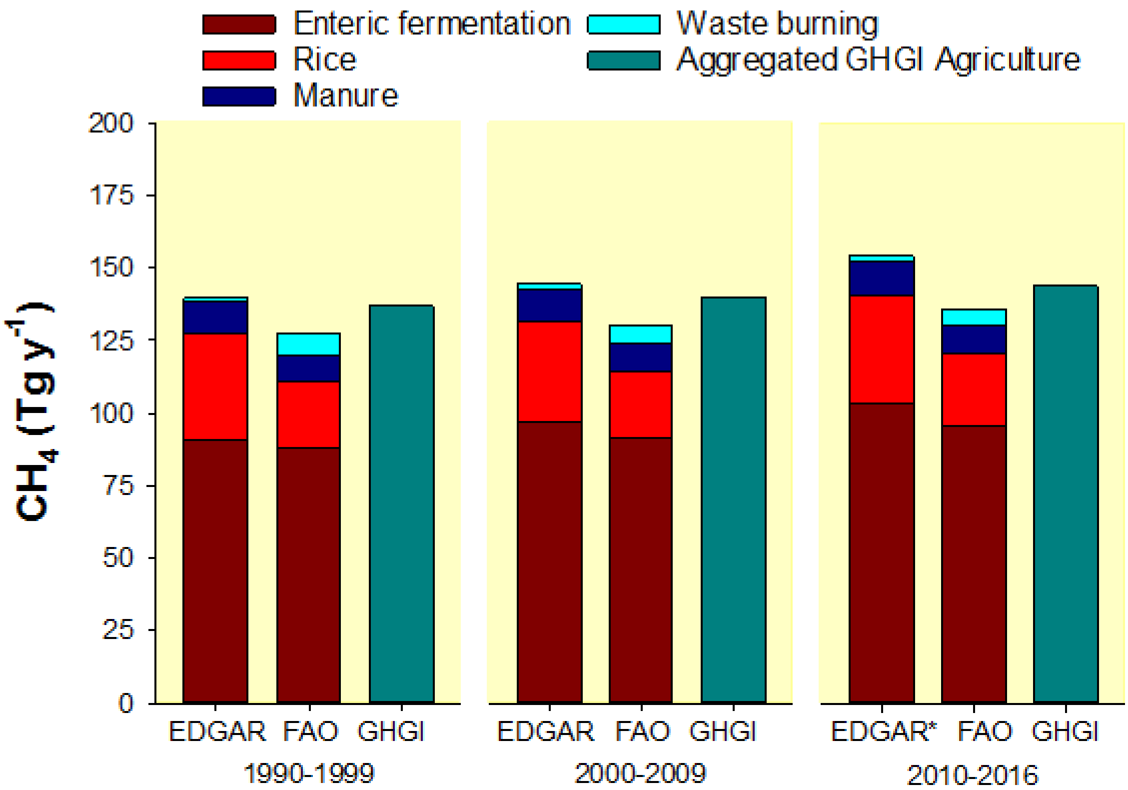

Figure 2.9 Average agricultural CH_4 emissions estimates from 1990. Sub-sectorial agricultural emissions are based on the Emissions Database for Global Atmospheric Research (EDGAR v4.3.2; Janssens-Maenhout et al. 2017a); FAOSTAT (Tubiello et al. 2013); and National GHGI data (Grassi et al. 2018). GHGI data are aggregate values for the sector.
*** Note that EDGAR data are complete only through 2012; the data in the right-hand panel represent the three years 2010-2012 and are presented for comparison.**

Upland soils are a net sink of atmospheric CH_4, but soils both produce and consume the gas. On the global scale climatic zone, soil texture, and land cover have an important effect on CH_4 uptake in upland soils (Tate 2015; Yu et al. 2017; Dutaur and Verchot 2007). Boreal soils take up less than temperate or tropical soils, coarse textured soils take up more CH_4 than medium and fine textured soils, and forests take up more than other ecosystems. Low levels of nitrogen fertilisation or atmospheric deposition can affect the soil microbial community and stimulate soil CH_4 uptake in nitrogen limited soils, while higher fertilisation rates decrease uptake (Edwards et al. 2018; Zhuang et al., 2013). Globally, N fertilisation on agricultural lands may have suppressed CH_4 oxidation by as much as 26 Tg between 1998 and 2004 (Zhuang et al., 2013)(*low confidence* , *low agreement*). The effect of N additions is cumulative and repeated fertilisation events have progressively greater suppression effects (*robust evidence, high agreement*) (Tate 2015). Other factors like higher temperatures, increased atmospheric concentrations, and changes in rainfall patterns stimulate soil CH_4 consumption in unfertilised ecosystems. Several studies (Yu et al. 2017; Xu et al. 2016; Curry 2009) have shown that globally, uptake has been increasing during the second half of the 20th century and it is expected to continue to increase by as much as 1 Tg in the 21st century, particularly in forests and grasslands (*medium evidence, high agreement*).

Northern peatlands (40°-70°N) are a significant source of atmospheric CH_4, emitting about 48 Tg CH_4, or about 10% of the total emissions to the atmosphere (Zhuang et al. 2006; Wuebbles and Hayhoe 2002). CH_4 emissions from natural northern peatlands are highly variable with the highest rate from fens (*medium evidence, high agreement*). Peatland management and restoration, alters the exchange of CH_4 with the atmosphere (*medium evidence, high agreement*). Management of peat soils typically converts them from CH_4 sources to sinks (Augustin et al. 2011; Strack and Waddington 2008; Abdalla et al. 2016) (*robust evidence, high agreement*). While restoration decreases CO_2 emissions (see Section 4.9.4), CH_4 emissions often increase relative to the drained conditions (*robust evidence, high agreement*) (Osterloh et al. 2018; Christen et al. 2016; Koskinen et al. 2016; Tuittila et al. 2000; Vanselow-Algan et al. 2015; Abdalla et al. 2016). Drained peatlands are usually considered to be negligible methane sources, but they emit CH_4 under wet weather conditions and from drainage ditches (Drösler et al. 2013; Sirin et al. 2012). While ditches cover only a small percentage of the drained area, emissions can be sufficiently high that drained peatlands emit comparable CH_4 as undrained ones (*medium evidence, medium agreement*) (Sirin et al. 2012; Wilson et al. 2016).

Because of the large uncertainty in the tropical peatland area, estimates of the global flux are highly uncertain. A meta-analysis of the effect of conversion of primary forest to rice production showed that emissions increased by a factor of 4 (*limited evidence, high agreement*) (Hergoualc'h and Verchot, 2012). For land uses that required drainage, emissions decreased by a factor of 3 (*limited evidence, high agreement*). There are no representative measurements of emissions from drainage ditches in tropical peatlands.

2.3.3 Nitrous Oxide

2.3.3.1 Atmospheric trends

The atmospheric abundance of N_2O has increased since 1750, from a pre-industrial concentration of 270 ppbv to 330 ppbv in 2017 (U.S. National Oceanographic and Atmospheric Agency, Earth Systems Research Laboratory; Figure 2.10) (*high agreement, robust evidence*). The rate of increase has also increased, from approximately 0.15 ppbv yr^{-1} 100 years ago, to 0.85 ppbv yr^{-1} over the period 2001 to 2015 (Wells et al. 2018). Atmospheric N_2O isotopic composition ($^{14/15}N$) was relatively constant during the pre-industrial period (Prokopiou et al. 2018) and shows a decrease in the $\delta^{15}N$ as the N_2O mixing ratio in the atmosphere has increased between 1940 and 2005. This recent decrease indicates as that terrestrial sources are the primary driver of increasing trends and marine sources contribute around 25% (Snider et al. 2015). Microbial denitrification and nitrification processes are responsible for more than 80% of total global N_2O emissions, which includes natural soils, agriculture, and oceans, with the remainder coming from non-biological sources such as biomass burning and fossil-fuel combustion (Fowler et al. 2015). The isotopic trend also indicates a shift from denitrification to nitrification as the primary source of N_2O as a result of the use of synthetic nitrogen (N) fertiliser (*high evidence, high agreement*) (Park et al. 2012; Toyoda et al. 2013; Snider et al. 2015; Prokopiou et al. 2018).

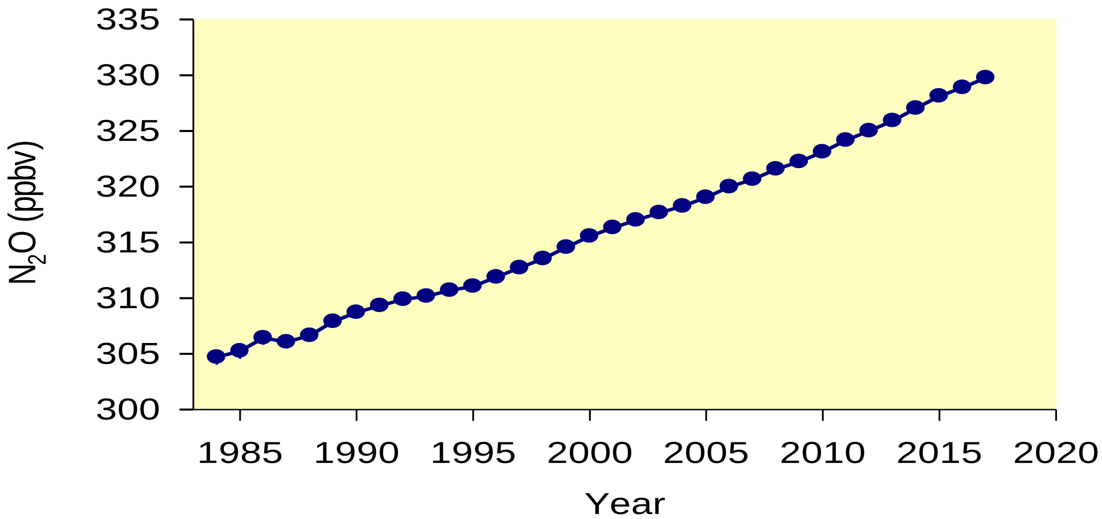

Figure 2.10 Globally averaged atmospheric N_2O mixing ratios since 1984. Data sources: NOAA/ESRL

Global Monitoring Division (https://www.esrl.noaa.gov/gmd/hats/combined/N2O.html).

The three independent sources of N_2O emissions estimates from agriculture at global, regional, and national levels are: U.S.E.P.A., EDGAR and FAOSTAT (USEPA 2013; Tubiello et al. 2015; Janssens-Maenhout et al. 2017). EDGAR and FAOSTAT have temporal resolution beyond 2005 and we these databases compare well with national inventory data (Figure 2.10). USEPA has historical estimates through 2005 and projections thereafter. The independent data use IPCC methods, with Tier 1 emission factors and national reporting of activity data. Tier 2 approaches are also available based on top-down and bottom-up approaches. Recent estimates using inversion modelling and process models estimate total annual global N_2O emissions of 16.1-18.7 (bottom-up) and 15.9-17.7 Tg N (top-down), demonstrating relatively close agreement (Thompson et al. 2014). Agriculture is the largest source and has increased with the extensification and intensification. Recent modelling estimates of terrestrial sources show a higher emissions range that is slightly more constrained than what was reported in AR5: approximately 9 (7–11) Tg N_2O-N yr^{-1} (Saikawa et al. 2014; Tian et al. 2016) compared to 6.6 (3.3–9.0) Tg N_2O-N yr^{-1} (Ciais et al. 2013a). Estimates of marine N_2O emissions are between 2.5 and 4.6 Tg N_2O-N yr^{-1}; (Buitenhuis et al., 2017; Saikawa et al., 2014).

To conclude, N_2O is continuing to accumulate in the atmosphere at an increasingly higher rate (*very high confidence*), driven primarily by increases in manure production and synthetic N fertiliser use from the mid-20th century onwards (*high confidence*). Findings since AR5 have constrained regional and global estimates of annual N_2O emissions and improved our understanding of the spatio-temporal dynamics of N_2O emissions, with soil rewetting and freeze-thaw cycles, which important determinants of total annual emission fluxes in some regions (*medium confidence*).

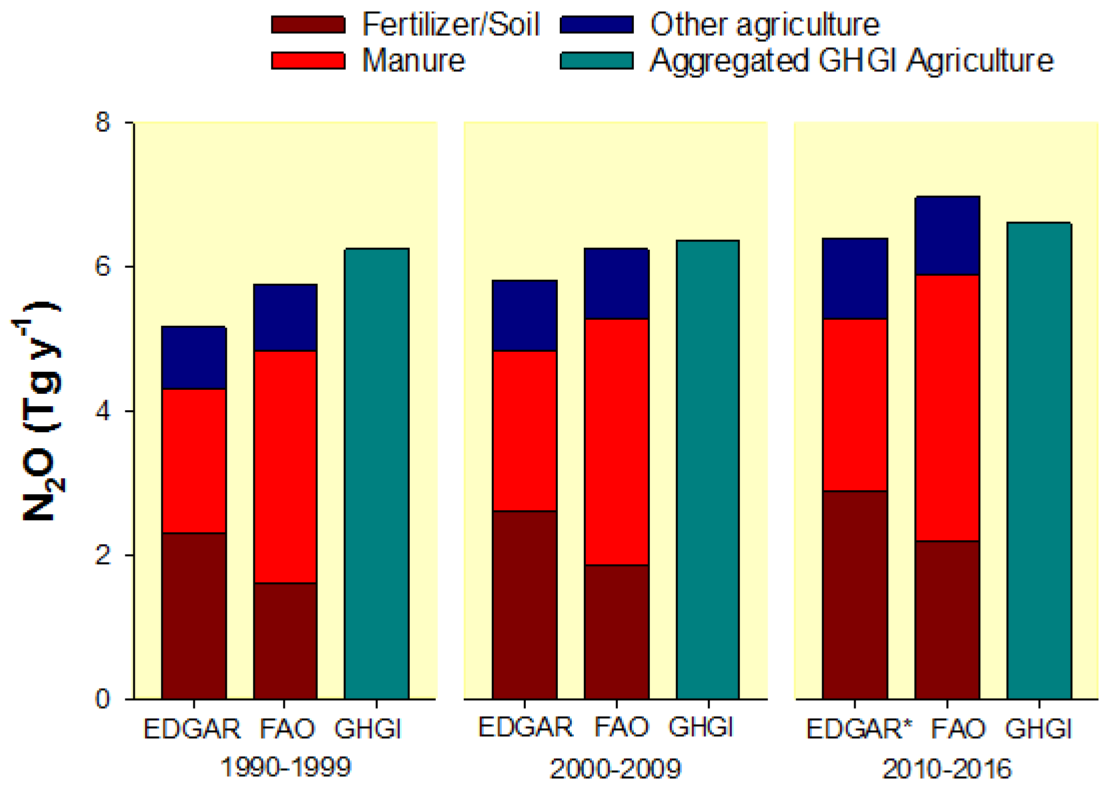

Figure 2.11 Average agricultural N_2O emissions estimates from 1990. Sub-sectorial agricultural emissions are based on the Emissions Database for Global Atmospheric Research (EDGAR v4.3.2; Janssens-Maenhout et al. 2017a); FAOSTAT (Tubiello et al. 2013); and National GHGI data (Grassi et al. 2018). GHGI data are aggregate values for the sector.
*** Note that EDGAR data are complete only through 2012; the EDGAR data in the right-hand panel represent the three years 2010-2012 and are presented for comparison.**

2.3.3.2 Land use effects

Agriculture is responsible for approximately two-thirds of N_2O emissions (*robust evidence, high agreement*)

(Janssens-Maenhout et al. 2017). Total emissions from this sector are the sum of direct and indirect emissions. Direct emissions from soils are the result of mineral fertiliser and manure application, manure management, deposition of crop residues, cultivation of organic soils and inorganic N inputs through biological nitrogen fixation. Indirect emissions come from increased warming, enrichment of downstream water bodies from runoff, and downwind N deposition on soils. The main driver of N_2O emissions in croplands is a lack of synchronisation between crop N demand and soil N supply, with approximately 50% of N applied to agricultural land not taken up by the crop (Zhang et al. 2017). Cropland soils emit over 3 Tg N_2O-N yr^{-1} (*medium evidence, high agreement*) (Janssens-Maenhout et al. 2017; Saikawa et al. 2014). Regional inverse modelling studies show larger tropical emissions than the inventory approaches and they show increases in N_2O emissions from the agricultural sector in South Asia, Central America, and South America (Saikawa et al. 2014; Wells et al. 2018).

Emissions of N_2O from pasturelands and rangelands have increased by as much as 80% since 1960 due to increased manure production and deposition (*robust evidence, high agreement*) (de Klein et al. 2014; Tian et al. 2018; Chadwick et al. 2018; Dangal et al. 2019; Cardenas et al. 2019). Studies consistently report that pasturelands and rangelands are responsible for around half of the total agricultural N_2O emissions (Davidson 2009; Oenema et al. 2014; Dangal et al. 2019). An analysis by Dangal et al. (2019) shows that while managed pastures make up around one-quarter of the global grazing lands, they contribute 86% of the net global N_2O emissions from grasslands and that more than half of these emissions are related to direct deposition of livestock excreta on soils.

Many studies calculate N_2O emissions from a linear relationship between nitrogen application rates and N_2O emissions. New studies are increasingly finding nonlinear relationships, which means that N_2O emissions per hectare are lower than the Tier 1 EFs (IPCC 2003) at low nitrogen application rates, and higher at high nitrogen application rates (*robust evidence, high agreement*) (Shcherbak et al. 2014; van Lent et al. 2015; Satria 2017). This not only has implications for how agricultural N_2O emissions are estimated in national and regional inventories, which now often use a linear relationship between nitrogen applied and N_2O emissions, it also means that in regions of the world where low nitrogen application rates dominate, increases in nitrogen fertiliser use would generate relatively small increases in agricultural N_2O emissions. Decreases in application rates in regions where application rates are high and exceed crop demand for parts of the growing season are likely to have very large effects on emissions reductions (*medium evidence, high agreement*).

Deforestation and other forms of land-use change alter soil N_2O emissions. Typically, N_2O emissions increase following conversion of native forests and grasslands to pastures or croplands (McDaniel et al. 2019; van Lent et al. 2015). This increase lasts from a few years to a decade or more, but there is a trend toward decreased N_2O emissions with time following land use change and ultimately lower N_2O emissions than had been occurring under native vegetation, in the absence of fertilisation (**Figure 2.12**) (Meurer et al. 2016; van Lent et al. 2015) (*medium evidence, high agreement*). Conversion of native vegetation to fertilised systems typically leads to increased N_2O emissions over time, with the rate of emission often being a function of nitrogen fertilisation rates, but this response can be moderated by soil characteristics and water availability (*medium evidence, high agreement*) (van Lent et al. 2015; Meurer et al. 2016). Restoration of agroecosystems to natural vegetation, over the period of one to two decades does not lead to recovery of N_2O emissions to the levels of the original vegetation (McDaniel et al. 2019). To conclude, findings since AR5 increasingly highlight the limits of linear N_2O emission factors, particularly from field to regional scales, with emissions rising nonlinearly at high nitrogen application rates (*high confidence*). Emissions from unfertilised systems often increase and then decline over time with typically lower emissions than was the case under native vegetation (*high confidence*).

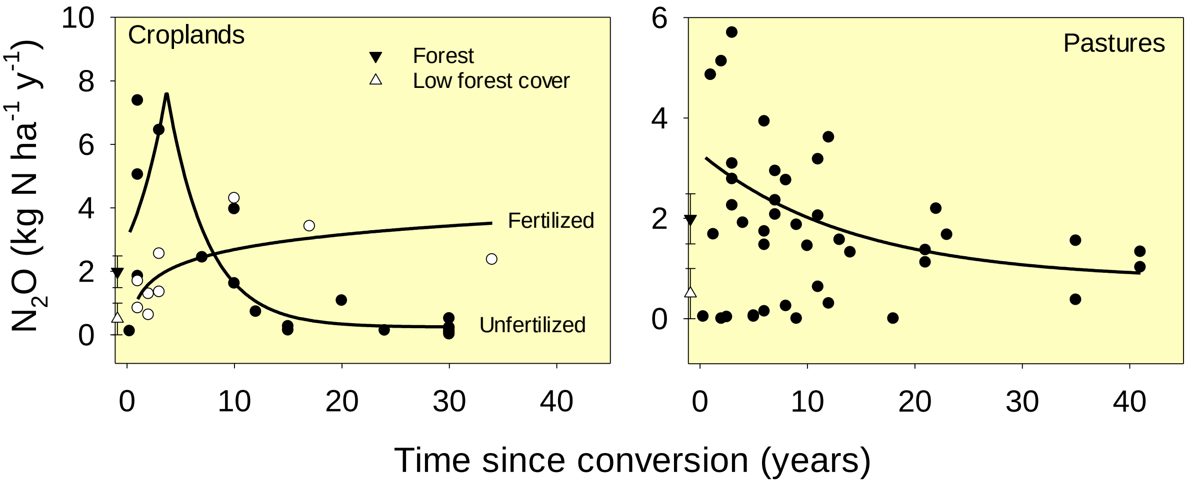

Figure 2.12 Effect of time since conversion on N_2O fluxes in unfertilised (black circles) and fertilised (white circles) tropical croplands (left frame) and in unfertilised tropical pastures (right frame). Average N_2O flux and 95% confidence intervals are given for upland forests (black inverted triangle) and low canopy forests (white inverted triangle), for comparison. The solid lines represent the trends for unfertilised and fertilised cases. Data source: (van Lent et al. 2015).

While soil emissions are the predominant source in agriculture, other sources are important or their importance is only just emerging. Biomass burning is responsible for approximately 0.7 Tg N_2O-N yr^{-1} (0.5–1.7 Tg N_2O-N yr^{-1}) or 11% of total gross anthropogenic emissions due to the release of N_2O from the oxidation of organic nitrogen in biomass (van der Werf et al. 2013). This source includes crop residue burning, forest fires, household cook stoves, and prescribed savannah, pasture and cropland burning. Aquaculture is currently not accounted for in most assessments or compilations. While it is currently responsible for less than 0.1 Tg N_2O-N yr^{-1}, is one of the fastest growing sources of anthropogenic N_2O emissions (Williams and Crutzen 2010; Bouwman et al. 2013) (*limited evidence, high agreement*). Finally, increased nitrogen deposition from terrestrial sources is leading to greater indirect N_2O emissions, particularly since 1980 (*moderate evidence*, *high agreement*). (Tian et al. 2018, 2016). In marine systems, deposition is estimated to have increased the oceanic N_2O source by 0.2 Tg N_2O-N yr^{-1} or 3% of total gross anthropogenic emissions (Suntharalingam et al. 2012).

Box 2.2: Methodologies for estimating national to global scale anthropogenic land carbon fluxes

Bookkeeping/accounting models (Houghton et al. 2012b; Hansis et al. 2015; Houghton and Nassikas 2017) calculate changes in biomass and soils that result from changes in land activity using data on biomass density and rates of growth/decomposition, typically from ground-based inventory data collection (field measurements of carbon in trees and soils). The approach includes only those changes directly caused by major categories of land-use change and management. The models do not explicitly include the indirect effects to changing environmental conditions, although some effects are implicit in biomass, growth rates and decay rates used. Thus, the models may overestimate past fluxes. The bookkeeping models include fluxes from peatland burning based on GFED estimates (Global Fire Emissions Database, (Randerson et al. 2015).)

Dynamic Global Vegetation Models (DGVMs) simulate ecological processes, such as photosynthesis, respiration, allocation, growth, decomposition etc., driven by environmental conditions (climate variability, climate change, CO_2, nitrogen concentrations). Models vary with respect to the processes included, with many since AR5 now including forest management, fire, N, and other management (Sitch et al. 2005; Le Quéré et al. 2018). Models are forced with increasing atmospheric CO_2 and changing climate, and run with and without "land use change" (land cover and forest harvest) to differentiate the anthropogenic effects from the indirect effects of climate and CO_2 - the “land sink”. Thus, indirect effects are explicitly included. This approach also includes a “lost atmospheric sink capacity”, or the carbon uptake due to environmental effects on forests that does not happen once the forests are removed (Pongratz et al. 2010).

Integrated Assessment Models (IAMs) use story-lines to construct alternative future scenarios of GHG emissions and atmospheric concentrations within a global socio-economic framework, including projections of AFOLU based on assumptions of, for example, crop yields, population growth, bioenergy use (See Cross-Chapter Box 1: Scenarios, Chapter 1). Some models include simplified DGVMs, which may include climate and CO_2 effects, while others use AFOLU emissions from other sources.

Earth system models (ESMs) couple DGVMs, surface hydrology and energy exgcnage models with a atmosphere, ocean, and sea ice models, enabling exploration of feedbacks between climate change and the carbon cycle (e.g., warming effects increase soil and plant respiration and lead to higher atsmpheric CO_2 concentrations, which in turn promote plant growth) (Friedlingstein et al. 2014). They sometimes include numerical experiments with and without land-use change to diagnose the anthropogenic AFOLU flux (Lawrence et al. 2016).

Satellite data can be used as a proxy for plant activity (e.g. greenness) and to map land cover, vegetation fires and biomass density. Algorithms, models and independent data are used to calculate fluxes of CO_2 from satellite data, although calculating the net carbon flux is difficult because of the lack of information on the respiratory flux. Some active satellite sensors (LiDAR) are able to measure three-dimensional structure in woody vegetation, which is closely related to biomass density (Zarin et al. 2016a; Baccini et al. 2012; Saatchi et al. 2011). Together with land-cover change data, these estimates of biomass density can be used to provide observational-based estimates of fluxes due to changes in forest area (e.g., (Tyukavina et al. 2015; Harris et al. 2015; Baccini et al. 2012) or degradation (Baccini et al. 2017). Satellite estimates of biomass vary considerably (Mitchard et al. 2013; Saatchi et al. 2015; Avitabile et al. 2016); data are available only for recent decades; methods generally assume that all losses of carbon are immediately released to the atmosphere; and changes in soil carbon are generally ignored. The approach implicitly includes indirect and natural disturbance effects as well as direct anthropogenic effects.

Atmospheric Inversions use observations of atmospheric concentrations with a model of atmospheric transport, based on data for wind speed and direction, to calculate implied emissions (Gatti et al. 2014; Liu et al. 2017a; van der Laan-Luijkx et al. 2017). Since AR5 there has been an increase in availability of concentration data from flux tower networks and satellites, enabling better global coverage at finer spatial scales and some national estimates (e.g. in the UK inverse techniques are used together with national GHG inventories). A combination of concentrations of different gases and isotopes enables the separation of fossil, ocean and land fluxes. However, inversions give only the net flux of CO_2 from land; they cannot separate natural and anthropogenic fluxes.

Micrometeorological flux measurements: Data on CO_2 concentrations and air movements recorded on instrumented towers enable calculation of CO_2 flux at the ecosystem scale. Global and regional Flux Networks (FluxNet (Global), AsiaFlux, Ameriflux (North America), ICOS (EU), NEON (USA), and others) contribute to a global flux data base, which is used to verify the results of modelling, inventory and remote sensing studies.

FAOSTAT: The United Nations Food and Agricultural Organization has produced country level estimates of greenhouse gas emissions (Tubiello et al. 2013) from agriculture (1961–2016) and land use (1990–2016) using a globally consistent methodological approach based largely on IPCC Tier 1 methods of the 2006 IPCC Guidelines (FAO 2015b). FAO emissions estimates were used as one of the three database inputs into the AR5 WGIII AFOLU chapter. Non-CO_2 emissions from agriculture are estimated directly from national statistics of activity data reported by countries to FAO. CO_2 emissions from land use and land-use change are computed mostly at Tier 1, albeit at fine geospatial scales to capture effects from peatland degradation and biomass fires (Rossi et al. 2016). Emissions from forest land and deforestation are based on the IPCC carbon stock change method, thus constituting a Tier 3 estimate relying of country statistics of carbon stocks and forest area collected through the FAO FRA. The carbon flux is estimated assuming instantaneous emissions in the year of forest area loss, and changes in carbon stocks within extant forests, but does not distinguish "managed" and "unmanaged" forest areas, albeit it treats separately emissions from primary, secondary and planted forest (Federici et al. 2015).

Country Reporting of GHG Inventories (GHGIs): All Parties to the UNFCCC are required to report national GHG Inventories (GHGIs) of anthropogenic emissions and removals. Reporting requirements are differentiated between developed and developing countries. Because of the difficulty of separating direct anthropogenic fluxes from indirect or natural fluxes, the (IPCC 2003) adopted the "managed land" concept as a proxy to facilitate GHGI reporting. All GHG fluxes on "managed land" are defined as anthropogenic, with each country applying their own definition of "managed land" (i.e. "where human interventions and practices have been applied to perform production, ecological or social functions" (IPCC 2006)). Fluxes may be determined on the basis of changes in carbon stocks (e.g., from forest inventories) or by activity data (e.g. area of land cover change management activity multiplied by emission factors or with modelled fluxes). Depending on the specific methods used, GHGIs include all direct anthropogenic effects and may include the indirect anthropogenic effects of environmental change (generally sinks) and natural effects (see Section 2.3.1.2). GHG fluxes from "unmanaged land" are not reported in GHGIs because they are assumed to be non-anthropogenic. The reported estimates may then be filtered through agreed "accounting rules" - i.e. what countries actually count towards their mitigation targets (Cowie AL et al. 2007; Lee, D. and Sanz 2017). The accounting aims to better quantify the additional mitigation actions by, for example, factoring out the impact of natural disturbances and forest age-related dynamics (Canadell et al. 2007; Grassi et al. 2018).

Box 2.3: CO_2 fertilisation and enhanced terrestrial uptake of carbon

All Dynamic Global Vegetation Models (DGVMs) and Earth System Models (ESMs) represent the CO_2 fertilisation effect ((Le Quéré et al. 2017; Hoffman et al. 2014). There is *high confidence* that elevated CO_2 results in increased short-term CO_2 uptake per unit leaf area (Swann et al. 2016; Field et al. 1995; Donohue et al. 2013);. However, whether this increased CO_2 uptake at the leaf level translates into increased growth for the whole plant differs among plant species and environments because growth is constrained by whole-plant resource allocation, nutrient limitation (e.g., nitrogen (N), phosphorus (P), potassium (K) and soil water and light limitations (Körner 2006; Peñuelas et al. 2017; Friend et al. 2014a). Interactions between plants and soil microbes, further modulate the degree of nutrient limitation on CO_2 fertilisation (Terrer et al. 2017).

At the ecosystems level, enhanced CO_2 uptake at decadal or longer time scales depends on changes in plant community composition and ecosystem respiration, as well disturbance and natural plant mortality (De Kauwe et al., 2016; Farrior et al., 2015; Keenan et al., 2017; Sulman et al, 2019). The results of FACE experiments (free-air carbon dioxide enrichment) over two decades are highly variable because of these factors (Norby et al. 2010; Körner 2015; Feng et al. 2015; Paschalis et al. 2017; Terrer et al. 2017; Du et al. 2019b). Under higher atmospheric CO_2 concentrations, the ratio of CO_2 uptake to water loss (water use efficiency, WUE), increases and enhances drought tolerance of plants (*high confidence*) (Berry et al., 2010; Ainsworth & Rogers, 2007).

Long-term CO_2 and water vapour flux measurements show that WUE in temperate and boreal forests of the Northern Hemisphere has increased more than predicted by photosynthetic theory and models over the past two decades (*high confidence*) (Keenan et al. 2013; Laguë and Swann 2016b). New theories have emerged on how CO_2 uptake by trees is related to water loss and to the risk of damaging xylem (water conducting tissues) in the trunk and branches (Wolf et al. 2016a; Anderegg et al. 2018a). Tree ring studies of stable carbon and oxygen isotopes also detected increased WUE in recent decades (Battipaglia et al. 2013; Silva and Anand 2013; van der Sleen et al. 2014). Yet, tree ring studies often fail to show acceleration of tree growth rates in support of CO_2 fertilisation, even when they show increased WUE (van der Sleen et al. 2014). The International Tree Ring Data Bank (ITRDB) indicated that only about 20% of the sites in the database showed increasing trends in tree growth that cannot be explained by climate variability, nitrogen deposition, elevation, or latitude. Thus there is *limited evidence (low agreement)* among observations of enhanced tree growth due to CO_2 fertilisation of forests during the 20th century (Gedalof and Berg 2010).

In grasslands, although it is possible for CO_2 fertilisation to alleviate the impacts of drought and heat stress on net carbon uptake (Roy et al. 2016b), there is *low confidence* about its projected magnitude. Because of its effect on water use efficiency, CO_2 fertilisation is expected to be pronounced in semi-arid habitats; and because of different metabolic pathways, C_3 plants are expected to be more sensitive to elevated CO_2 concentrations than C_4 grasses (Donohue et al. 2013; Morgan et al. 2011; Derner et al. 2003). Neither of these expectations was observed over a 12-year study of elevated CO_2 in a grassland system: enhanced growth was not observed during dry summers, and growth of C_4 grasses was unexpectedly stimulated, while growth of C_3 grasses was not (Reich et al. 2014, 2018).

There is *medium confidence* that CO_2 fertilisation effects have increased water use efficiency in crops and thus reduced agricultural water use per unit of crop produced (Deryng et al. 2016b; Nazemi and Wheater 2015; Elliott et al. 2014) . This effect could lead to near-term continued greening of agricultural areas. However, current assessments of these effects are based on limited observations, mostly from the temperate zone (Deryng et al. 2016a).

One line of evidence for CO_2 fertilisation is the increasing land sink ("the residual land sink" in AR5) over the last 50 years as the atmospheric CO_2 concentration has increased (Los 2013; Sitch et al. 2015b; Campbell et al. 2017; Keenan and Riley 2018). A combined analysis of atmospheric inverse analyses, ecosystem models, and forest inventory data concluded that 60% of the recent terrestrial carbon sink can be directly attributed to increasing atmospheric CO_2 (Schimel et al. 2015). A global analysis using a "reconstructed vegetation index" (RVI) for the period 1901–2006 from MODIS satellite-derived NDVI (Normalised Vegetation Difference Index) showed that CO_2 fertilisation contributed at least 40% of the observed increase in the land carbon sink (Los 2013). Without CO_2 fertilisation ESMs are unable to simulate the increasing land sink and the observed atmospheric CO_2 concentration growth rate since the middle of the 20th century (Shevliakova et al. 2013). There are other mechanisms that could explain enhanced land C uptake such as increased regional forest and shrub cover (see Cross-Chapter Box 2: Implications of large-scale conversion from non-forest to forest land, Chapter 1);(Chen et al. 2019), and, at higher latitudes, increasing temperatures and longer growing seasons (Zhu et al. 2016).

In summary, there is *low confidence* about the magnitude of the CO_2 effect and other factors that may explain at least a portion of the land sink (e.g., nitrogen deposition, increased growing season, reduced burning, erosion and re-deposition or organic sediments, and aerosol-induced cooling). Increases in atmospheric CO_2 result in increased water use efficiency and increase leaf-level photosynthesis (*high confidence*). The extent to which CO_2 fertilisation results in plant- or ecosystem-level carbon accumulation is highly variable and affected by other environmental constraints (*high confidence*). Even in ecosystems where CO_2 fertilisation has been detected in recent decades, those effects are found to weaken as a result of physiological acclimation, soil nutrient limitation, and other constraints on growth (Friend et al., 2014; Körner, 2006; Peñuelas et al., 2017).

2.4 Emissions and impacts of short-lived climate forcers (SLCF) from land

While the rising atmospheric concentration of GHGs is the largest driver of anthropogenic changes in climate, the levels of short-lived climate forcers (SLCF) can significantly modulate regional climate by altering radiation exchanges and hydrological cycle and impact ecosystems (Boucher et al. 2013; Rogelj et al. 2014; Kok et al. 2018) *(high confidence).* This section assesses the current state of knowledge with respect to past and future emissions of the three major SLCFs and their precursors: mineral dust, carbonaceous aerosols (Black Carbon and Organic Carbon), and Biogenic volatile organic compounds (BVOCs). The chapter also reports on implications of changes in their emissions for climate. Aerosols particles with diameters between about 0.010 µm to about 20 µm are recognised as SLCFs, a term that refers to their short atmospheric lifetime (a few days). BVOCs are important precursors of ozone and organic carbon (OC), both important climate forcing agents with short atmospheric lifetimes.

While the AR5 did not assess land aerosols emissions in depth, their findings stated that although progress in quantifying regional emissions of anthropogenic and natural land aerosols has been made, considerable uncertainty still remains about their historical trends, their inter-annual and decadal variability and about any changes in the future (Calvo et al. 2013; Klimont et al. 2017). Some new and improved understanding of processes controlling emissions and atmospheric processing has been developed since AR5, for example, a better understanding of the climatic role of Black Carbon (BC) as well as the understanding of the role of BVOCs in formation of secondary organic aerosols (SOA).

Depending on the chemical composition and size, aerosols can absorb or scatter sunlight and thus directly affect the amount of absorbed and scattered radiation (Fuzzi et al. 2015a; Nousiainen 2011; de Sá et al. 2019) Aerosols affect clouds formation and development, and thus can also influence precipitation patterns and amounts (Suni et al. 2015). In addition, deposition of aerosols—especially black carbon—on snow and ice surfaces can reduce albedo and increase warming as a self-reinforcing feedback. Aerosols deposition also change biogeochemical cycling in critical terrestrial ecosystems with deposition of nutrients such as nitrogen and phosphorus (Andreae et al. 2002). Primary land aerosols are emitted directly into the atmosphere due to natural or anthropogenic processes and include mineral aerosols (or dust), volcanic dust, soot from combustion, organic aerosols from industry, vehicles or biomass burning, bioaerosols from forested regions, and others. Secondary organic aerosols (SOA) are particulates that are formed in the atmosphere by gas-to-particles conversion processes from gaseous precursors, such as BVOCs, and account for a large fraction of fine mode (particles less than 2.5µm) aerosol mass (Hodzic et al. 2016; Manish et al. 2017). Land use change can affect the climate through changed emissions of SLCFs such as aerosols, ozone precursors and methane.

Aerosols from air pollution will decline in the coming years as a means for improving urban and regional air, but their removal will lead to additional warming (Boucher et al. 2013), with important regional variability, and partially offsetting projected mitigation effects for two to three decades in 1.5°C consistent pathways (IPCC 2018) (*high confidence*). It is important to emphasise that changes in emissions can either be due to external forcing or through a feedback in the climate system (Box 2.1:). For instance, enhanced dust emissions due to reduced vegetation could be a forcing if overgrazing is the cause of larger dust emission, or a feedback if dryer climate is the cause. This distinction is important in terms of mitigation measures to be implemented.

2.4.1 Mineral dust

One of the most abundant atmospheric aerosols emitted into the atmosphere is mineral dust, a "natural" aerosol that is produced by wind strong enough to initiate the emissions process of sandblasting. Mineral dust is preferentially emitted from dry and unvegetated soils in topographic depressions where deep layer of alluvium have been accumulated (Prospero et al. 2002). Dust is also emitted from disturbed soils by human activities with a 25% contribution to global emission, based on satellite-based estimate (Ginoux et al. 2012). Dust is then transported over long distances across continents and oceans. Dust cycle, which consists of mineral dust emission, transport, deposition and stabilisation, have multiple interactions with many climate processes and biogeochemical cycles.

2.4.1.1 Mineral dust as a short-lived climate forcer from land

Depending on the dust mineralogy, mixing state, and size, dust particles can absorb or scatter shortwave and long-wave radiation. Dust particles serve as cloud condensation nuclei and ice nuclei. They can influence the microphysical properties of clouds, their lifetime and precipitation rate (Kok et al. 2018). New and improved understanding of processes controlling emissions and transport of dust, its regional patterns and variability as well as its chemical composition has been developed since AR5.

While satellites remain the primary source of information to locate dust sources and atmospheric burden, in-situ data remains critical to constrain optical and mineralogical properties of the dust (Di Biagio et al. 2017; Rocha-Lima et al. 2018). Dust particles are composed of minerals, including iron oxides which strongly absorb shortwave radiation and provide nutrient for marine ecosystems. Other mineral such as feldspar is an efficient ice nuclei (Harrison et al. 2016). Dust mineralogy depending on the native soils, global databases were developed to characterise mineralogical composition of soils for use in the weather and climate models (Journet et al. 2014; Perlwitz et al. 2015). New field campaigns as well as new analysis from prior campaign have produced insights into role of dust in western Africa in climate system, for example, for dust (Veselovskii et al. 2016), long-ranged transport of dust across the Atlantic (Groß et al. 2015), and the

characterisation of aerosol particles and their ability to act as ice and cloud condensation nuclei (Price et al. 2018). Size distribution at emission is another key parameter controlling dust interactions with radiation. Most models use now the parametrisation of Kok (2011) based on the theory of brittle material. It was shown that most models underestimate the size of global dust cycle (Kok 2011) has been underestimated.
Characterisation of spatial and temporal distribution of dust emissions is essential for weather prediction and climate projections (*high confidence*). Although there is a growing confidence in characterising the seasonality and peak of dust emissions (i.e., spring-summer, (Wang et al. 2015)) and how the meteorological and soil conditions control dust sources, an understanding of long-term future dust dynamics, inter-annual dust variability and how they will affect future climate still requires substantial work. Dust is also important at high latitude, where it has impacts on snow covered surface albedo and weather (Bullard et al. 2016).

2.4.1.2 Effects of past climate change on dust emissions, and feedbacks

Limited number of model-based studies found that dust emissions increased significantly since late 19[th] century: by 25% from preindustrial to present day (e.g., from 729 Tg yr^{-1} to 912 Tg yr^{-1}) with ~50% increase driven by climate change and ~40% by land use cover change such as conversion of natural land to agriculture (Stanelle et al. 2014) (*low confidence*). These changes resulted in a clear sky radiative forcing at the top of the atmosphere of -0.14 Wm^{-2} (Stanelle et al. 2014). The authors found that, in North Africa most dust is of natural origin with a recent 15% increase in dust emissions attributed to climate change; in North America two thirds of dust emissions take place on agricultural lands and both climate change and land use change jointly drive the increase; between pre-industrial and present-day the overall effect of changes in dust is -0.14 Wm^{-2} cooling of clear sky net radiative forcing on top of the atmosphere, with -0.05 W m^{-2} from land use and -0.083 W m^{-2} from changes in climate.

The comparison of observations for vertically integrated mass of atmospheric dust mass per unit area (i.e. Dust Mass Path or DMP) obtained from the remotely sensed data and the DMP from CMIP5 models reveal that model-simulate range of DMP was much lower than the estimates from (Evan et al. 2014). ESM typically do not reproduce inter-annual and longer time scales variability seen in observations (Evan et al. 2016). Analyses of the CMIP5 models (Evan 2018; Evan et al. 2014)) reveal that all climate models systematically under-estimate dust emissions, amount of dust in the atmosphere and its inter-annual variability (*medium confidence*).

One commonly suggested reason for the lack of dust variability in climate models is the models' inability to simulate the effects of land surface changes on dust emission (Stanelle et al. 2014). Models which account for changes in land surface show more agreement with the satellite observations both in terms of Aerosol Optical Depth and DMP (Kok et al. 2014). New prognostic dust emissions models now able to account for both changes in surface winds and vegetation characteristics (e.g., leaf area index and stem area index) and soil water, ice, and snow cover (Evans et al. 2016). As a result, new modelling studies (e.g. Evans et al. 2016) indicate that in regions where soil and vegetation respond strongly to ENSO events, such as in Australia, inclusion of dynamic vegetation characteristics into dust emission parameterisations improves comparisons between the modelled and observed relationship long-term climate variability (e.g., ENSO) and dust levels (Evans et al. 2016). Thus, there has been progress in incorporating effects of vegetation, soil moisture, surface wind and vegetation on dust emission source functions but the number of studies demonstrating such improvement remains small *(limited evidence, medium agreement)*.

2.4.1.3 Future changes of dust emissions

There is no agreement about direction of future changes in dust emissions. Atmospheric dust loading is projected to increase over the southern edge of the Sahara in association with surface wind and precipitation changes (Pu and Ginoux, 2018), while Evan et al. (2016) project a decline in African dust emissions. Dust Optical Depth (DOD) is also projected to increase over the central Arabian Peninsula in all seasons and to decrease over northern China from MAM to SON (Pu and Ginoux 2018). Climate models project rising drought risks over the southwestern and central US. in the twenty-first century. The projected drier regions largely overlay the major dust sources in the US. However, whether dust activity in the US will increase in the future is not clear, due to the large uncertainty in dust modelling (Pu and Ginoux 2017). Future trends of dust emissions will depend on changes in precipitation patterns and atmospheric circulation (*limited evidence, high agreement*). However, implication of changes in human activities, including mitigation (e.g.

bioenergy production) and adaption (e.g. irrigation) are not characterised in the current literature.

2.4.2 Carbonaceous Aerosols

Carbonaceous aerosols are one of the most abundant components of aerosol particles in continental areas of the atmosphere and a key land-atmosphere component (Contini et al. 2018). They can make up to 60-80% of PM2.5 (Particulate matter with size less than 2.5 µm) in urban and remote atmosphere (Tsigaridis et al. 2014a; Kulmala et al. 2011). It comprises an organic fraction (Organic Carbon - OC) and a refractory light absorbing component, generally referred as Elemental Carbon (EC), from which Black Carbon (BC) is the optically active absorption component of EC (Gilardoni et al. 2011; Bond et al. 2013).

2.4.2.1 Carbonaceous aerosol precursors of short-lived climate forcers from land

OC is a major component of aerosol mass concentration, and it originates from different anthropogenic (combustion processes) and natural (from natural biogenic emissions) sources (Robinson et al. 2007). A large fraction of OC in the atmosphere has a secondary origin, as it can be formed in the atmosphere through condensation to the aerosol phase of low vapour pressure gaseous compounds emitted as primary pollutants or formed in the atmosphere. This component is called Secondary Organic Aerosol (SOA) (Hodzic et al. 2016). A third component of the optically active aerosols is the so-called brown carbon (BrC), an organic material that shows enhanced solar radiation absorption at short wavelengths (Wang et al. 2016b; Laskin et al. 2015; Liu et al. 2016a; Bond et al. 2013; Saturno et al. 2018).

OC and EC have distinctly different optical properties, with OC being important for the scattering properties of aerosols and EC is central for the absorption component (Rizzo et al. 2013; Tsigaridis et al. 2014a; Fuzzi et al. 2015a). While organic carbon is reflective and scatter solar radiation, it has a cooling effect on climate. On the other side, BC and BrC absorbs solar radiation and they have a warming effect in the climate system. (Bond et al. 2013).

Organic carbon is also characterised by a high solubility with a high fraction of water-soluble organic compounds (WSOC) and it is one of the main drivers of the oxidative potential of atmospheric particles. This makes particles loaded with oxidised OC an efficient CCN in most of the conditions (Pöhlker et al. 2016; Thalman et al. 2017; Schmale et al. 2018).

Biomass burning is a major global source of carbonaceous aerosols (Bowman et al. 2011b; Harrison et al. 2010; Reddington et al. 2016; Artaxo et al. 2013). As knowledge of past fire dynamics improved through new satellite observations, new fire proxies' datasets (Marlon et al. 2013; van Marle et al. 2017), and process-based models (Hantson et al. 2016), a new historic biomass burning emissions dataset starting in 1750 has been developed (Van Marle et al. 2017a) (see Cross-Chapter Box 3: Fire and Climate Change, in this chapter). Revised versions of OC biomass burning emissions (Van Marle et al. 2017a) show in general reduced trends compared to the emissions derived by (Lamarque et al. 2010) for CMIP5. CMIP6 global emissions pathways (Gidden et al. 2018; Hoesly et al. 2018) estimate global BC emissions in 2015 at 9.8 Mt BC yr^{-1}, while global OC emissions are 35 Mt OC yr^{-1}.

Land use change is critically important for carbonaceous aerosols, since biomass burning emissions consist mostly of organic aerosol, and the undisturbed forest is also a large source of organic aerosols (Artaxo et al. 2013). Additionally, urban aerosols are also mostly carbonaceous, because of the source composition (traffic, combustion, industry, etc.) (Fuzzi et al. 2015b). Burning of fossil fuel, biomass burning emissions and SOA from natural BVOC emissions are the main global sources of carbonaceous aerosols. Any change in each of these components influence directly the radiative forcing (Contini et al. 2018; Boucher et al. 2013; Bond et al. 2013).

One important component of carbonaceous aerosols is the primary biological aerosol particles (PBAP), also called bioaerosols, that correspond to a significant fraction of aerosols in forested areas (Fröhlich-Nowoisky et al. 2016; Pöschl and Shiraiwa 2015). They are emitted directly by the vegetation as part of the biological processes (Huffman et al. 2012). Airborne bacteria, fungal spores, pollen, archaea, algae, and other bioparticles are essential for the reproduction and spread of organisms across various terrestrial ecosystems.

They can serve as nuclei for cloud droplets, ice crystals, and precipitation, thus influencing the hydrological cycle and climate (Whitehead et al. 2016; Scott et al. 2015; Pöschl et al. 2010).

2.4.2.2 Effects of past climate change on carbonaceous aerosols emissions, and feedbacks

Annual global emission estimates of BC range from 7.2-7.5 Tg yr^{-1} using bottom-up inventories (Bond et al. 2013; Klimont et al. 2017) up to 17.8 ± 5.6 Tg yr^{-1} using a fully coupled climate-aerosol-urban model constrained by aerosol measurements (Cohen and Wang 2014), with considerably higher BC emissions for Eastern Europe, Southern East Asia, and Southeast Asia mostly due to higher anthropogenic BC emissions estimates. A significant source of BC, the net trend in global burned area from 2000 to 2012 was a modest decrease of 4.3 Mha yr^{-1} (-1.2% yr^{-1}).

Carbonaceous aerosols are important in urban areas as well as pristine continental regions, since they can be responsible for 50-85% of PM2.5 (Contini et al. 2018; Klimont et al. 2017). In boreal and tropical forests, carbonaceous aerosols originate from BVOC oxidation (Section 2.4.3). The largest global source of BC aerosols is open burning of forests, savannah and agricultural lands with emissions of about 2,700 Gg yr^{-1} in the year 2000 (Bond et al. 2013).

ESMs most likely underestimate globally averaged EC emissions (Bond et al. 2013; Cohen and Wang 2014), although recent emission inventories have included an upwards adjustment in these numbers (Hoesly et al. 2018). Vertical EC profiles have also been shown to be poorly constrained (Samset et al. 2014a), with a general tendency of too much EC at high altitudes. Models differ strongly in the magnitude and importance of the coating-enhancement of ambient EC absorption (Boucher et al. 2016) (Gustafsson and Ramanathan 2016), in their estimated lifetime of these particles, as well as in dry and wet removal efficiency (Mahmood et al. 2016) (*limited evidence*, *medium agreement*).

The equilibrium in emissions and concentrations between the scattering properties of organic aerosol versus the absorption component of BC is a key ingredient in the future climatic projections of aerosol effects (*limited evidence*, *high agreement*). The uncertainties in net climate forcing from BC rich sources are substantial, largely due to lack of knowledge about cloud interactions with both black carbon and co-emitted organic carbon. A strong positive forcing of about 1.1 wm^{-2} was calculated by (Bond et al. 2013), but this forcing is balanced by a negative forcing of –1.45 wm^{-2}, and shows clearly a need to work on the co-emission issue for carbonaceous aerosols. The forcing will also depend on the aerosol-cloud interactions, where carbonaceous aerosol can be coated and change their CCN capability. It is difficult to estimate the changes in any of these components in a future climate, but this will influence strongly the radiative forcing (Contini et al. 2018; Boucher et al. 2013; Bond et al. 2013) (*high confidence*).

De Coninck et al. (2018) reported studies estimating a lower global temperature effect from BC mitigation (e.g., Samset et al. 2014b; Boucher et al. 2016), although commonly used models do not capture properly observed effects of BC and co-emissions on climate (e.g., (Bond et al. 2013). Regionally, the warming effects can be substantially larger, for example, in the Arctic (Sand et al. 2015) and high mountain regions near industrialised areas or areas with heavy biomass burning impacts (Ming et al. 2013) (*high confidence*).

2.4.2.3 Future changes of carbonaceous aerosol emissions

Due to the short atmospheric lifetime of carbonaceous aerosols in the atmosphere, of the order of a few days, most studies dealing with the future concentration levels have a regional character (Cholakian et al. 2018; Fiore et al. 2012). The studies agree that the uncertainties in changes in emissions of aerosols and their precursors are generally higher than those connected to climate change itself. Confidence in future changes in carbonaceous aerosol concentration projections is limited by the reliability of natural and anthropogenic emissions (including wildfires, largely caused by human activity) of primary aerosol as well as that of the precursors. The Aerosol Chemistry Model Intercomparison Project (AerChemMIP) is endorsed by the Coupled-Model Intercomparison Project 6 (CMIP6) and is designed to quantify the climate impacts of aerosols and chemically- reactive gases (Lamarque et al. 2013). These simulations calculated future responses to SLCF emissions for the RCP scenarios in terms of concentration changes and radiative forcing. Carbonaceous aerosol emissions are expected to increase in the near future due to possible increases in open biomass burning (forest, savannah, and agricultural fires) emissions, and increase in SOA from oxidation of BVOCs (Tsigaridis et al. 2014b; Van Marle et al. 2017b; Giglio et al. 2013) (*medium confidence*).

More robust knowledge has been produced since the conclusions reported in AR5 (Boucher et al. 2013) and all lines of evidence now agree on a small effect on carbonaceous aerosol global burden due to climate change (*medium confidence*). The regional effects, however, are predicted to be much higher (Westervelt et al. 2015). With respect to possible changes in the chemical composition of PM as a result of future climate change only a few sparse data are available in the literature and the results are, as yet, inconclusive. The co-benefits of reducing aerosol emissions due to air quality issues will play an important role in future carbonaceous aerosol emissions (Gonçalves et al. 2018; Shindell et al. 2017) (*high confidence*).

2.4.3 Biogenic Volatile Organic Compounds (BVOCs)

Biogenic volatile organic compounds (BVOCs) are emitted in large amounts by forests (Guenther et al. 2012). They include isoprene, terpenes, alkanes, alkenes, alcohols, esters, carbonyls and acids (Peñuelas and Staudt 2010; Guenther et al. 1995, 2012). Their emissions represent a carbon loss to the ecosystem, which can represent up to 10% of the carbon fixed by photosynthesis under stressful conditions (Bracho-Nunez et al. 2011). The global average emission for vegetated surfaces is 0.7g C m^{-2} yr^{-1} but can exceed 100 g C m^{-2} yr^{-1} in some tropical ecosystems (Peñuelas and Llusià 2003).

2.4.3.1 BVOC precursors of short-lived climate forcers from Land

BVOCs are rapidly oxidised in the atmosphere to form less volatile compounds that can condense and form secondary organic aerosol (SOA). In boreal and tropical forests, carbonaceous aerosols originate from BVOC oxidation, of which isoprene and terpenes are the most important precursors (Claeys et al. 2004; Hu et al. 2015; De Sá et al. 2017; de Sá et al. 2018; Liu et al. 2016b, see following sub-section). BVOCs are the most important precursors of SOA. This transformation process of BVOCs affects the aerosol size distribution both by contributing to new particle formation and to the growth of larger pre-existing particles. SOA affect the scattering of radiation by the particles themselves (direct aerosol effect), but also change the amount of cloud condensation nuclei (CCN) and the lifetime and optical properties of clouds (indirect aerosol effect).

High amounts of SOA are observed over forest areas, in particular in boreal and tropical regions where they have been found to mostly originate from BVOC emissions (Manish et al. 2017). In particular, isoprene epoxydiol-derived SOA (IEPOX-SOA) is being identified in recent studies in North America and Amazonian forest as a major component in the oxidation of isoprene (Allan et al. 2014; Schulz et al. 2018; De Sá et al. 2017). In tropical regions BVOC can be convected up to the upper atmosphere, where their volatility is reduced and where they become SOA. In some cases those particles are even transported back to the lower atmosphere (Schulz et al. 2018; Wang et al. 2016a; Andreae et al. 2018). In the upper troposphere in the Amazon, SOA are important CCN and are responsible for the vigorous hydrological cycle (Pöhlker et al. 2018). This strong link between BVOC emissions by plants and hydrological cycle has been discussed in a number of studies (Fuentes et al. 2000; Schmale et al. 2018; Pöhlker et al. 2018, 2016).

Changing BVOC emissions also affect the oxidant concentrations in the atmosphere. Their impact on the concentration of ozone depends on the NOx concentrations. In polluted regions, high BVOC emissions lead to increased production of ozone, followed by the formation of more OH and a reduction in the methane lifetime. In more pristine regions (NOx-limited), increasing BVOC emissions instead lead to decreasing OH and ozone concentrations, resulting in a longer methane lifetime. The net effect of BVOCs then can change over time if NOx emissions are changing.

BVOCs' possible climate effects have received little attention because it was thought that their short lifetime would preclude them from having any significant direct influence on climate (Unger 2014a; Sporre et al. 2018). Higher temperatures and increased CO_2 concentrations are (separately) expected to increase the emissions of BVOCs (Jardine et al. 2011, 2015; Fuentes et al. 2016). This has been proposed to initiate negative climate feedback mechanisms through increased formation of SOA (Arneth et al. 2010; Kulmala 2004; Unger et al. 2017). More SOA can make the clouds more reflective, which can provide a cooling. Furthermore, the increase in SOA formation has also been proposed to lead to increased aerosol scattering, resulting in an increase in diffuse radiation. This could boost gross primary production (GPP) and further increase BVOC emissions (Kulmala et al. 2014; Cirino et al. 2014; Sena et al. 2016; Schafer et al. 2002;

Ometto et al. 2005; Oliveira et al. 2007). These important feedbacks are starting to emerge (Sporre et al. 2018; Kulmala 2004; Arneth et al. 2017b). However, there is evidence that this influence might be significant at different spatial scales, from local to global, through aerosol formation and through direct and indirect greenhouse effects *(limited evidence, medium agreement)*. Most tropical forest BVOC are primarily emitted from foliage of trees but soil microbes can also be a major source of some compounds including sesquiterpenes (Bourtsoukidis et al. 2018).

2.4.3.2 Historical changes of BVOCs and contribution to climate change

Climate warming over the past 30 years, together with the longer growing season experienced in boreal and temperate environments, have increased BVOC global emissions by since the preindustrial times (*limited evidence, medium agreement*) (Peñuelas 2009; Sanderson et al. 2003; Pacifico et al. 2012). This was opposed by lower BVOC emissions caused by the historical conversion of natural vegetation and forests to cropland (*limited evidence*, *medium agreement*) (Unger 2013, 2014a; Fu and Liao 2014). The consequences of historical anthropogenic land cover change were a decrease in the global formation of SOA (-13 %, Scott et al. 2017) and tropospheric burden (-13 %, Heald and Geddes 2016). This has resulted in a positive radiative forcing (and thus warming) from 1850 to 2000 of 0.017 W m^{-2} (Heald and Geddes 2016), 0.025 (Scott et al. 2017) and 0.09 W m^{-2} (Unger 2014b) through the direct aerosol effect. In present-day conditions, global SOA production from all sources spans between 13 and 121 Tg yr^{-1} (Tsigaridis et al. 2014a). The indirect aerosol effect (change in cloud condensation nuclei), resulting from land use induced changes in BVOC emissions, adds an additional positive radiative forcing of 0.008 W m^{-2} (Scott et al. 2017). More studies with different model setups are needed to fully assess this indirect aerosol effect associated with land use change from the preindustrial to present. CMIP6 global emissions pathways (Hoesly et al. 2018; Gidden et al. 2018) estimates global VOCs emissions in 2015 at 230 Mt VOC yr^{-1}. They also estimated that from 2000 to 2015, emissions were up from 200 to 230 Mt VOC yr^{-1}.

There is (*limited evidence, medium agreement)* that historical changes in BVOC emissions have also impacted tropospheric ozone. At most surface locations where land use has changed, the NO_x concentrations are sufficiently high for the decrease in BVOC emissions to lead to decreasing ozone concentrations (Scott et al. 2017). However, in more pristine regions (with low NO_x concentrations), the imposed conversion to agriculture has increased ozone through decreased BVOC emissions and their subsequent decrease in OH (Scott et al. 2017; Heald and Geddes 2016). In parallel, the enhanced soil NO_x emissions from agricultural land, can increase the ozone concentrations in NO_x limited regions (Heald and Geddes 2016).

Another impact of historical decrease in BVOC emissions is the reduction in the atmospheric lifetime of methane (*limited evidence, medium agreement*), which results in a negative radiative forcing that ranges from -0.007 W m^{-2} (Scott et al. 2017) to -0.07 W m^{-2} (Unger 2014b). However, the knowledge of to which degree BVOC emissions impact oxidant concentrations, in particular OH (and thus methane concentrations), is still limited and therefore these numbers are very uncertain (Heald and Spracklen 2015; Scott et al. 2017). The effect of land use change on BVOC emissions are highly heterogeneous (Rosenkranz et al. 2015) and though the global values of forcing described above are small, the local or regional values can be higher and even of opposite sign than the global values.

2.4.3.3 Future changes of BVOCs

Studies suggest that increasing temperature will change BVOC emissions through change in species composition and rate of BVOC productions. A further 2 ºC to 3ºC rise in the mean global temperature, could increase BVOC global emissions by an additional 30–45% (Peñuelas and Llusià 2003). In two modelling studies, the impact on climate from rising BVOC emissions have been found to become even larger with decreasing anthropogenic aerosol emissions (Kulmala et al. 2013; Sporre et al. 2018). A negative feedback on temperature, arising from the BVOC-induced increase in the first indirect aerosol effect have been estimated by two studies to be in the order of -0.01 W m^{-2} K (Scott et al. 2018b; Paasonen et al. 2013). Enhanced aerosol scattering from increasing BVOC emissions has been estimated to contribute with a global gain in BVOC emissions of 7% (Rap et al. 2018). In a warming planet, BVOC emissions are expected to increase but magnitude is unknown and will depend on future land use change, in addition to climate (*limited evidence, medium agreement*).

There is a very limited number of studies investigating the climate impacts of BVOCs using future land use scenarios (Ashworth et al. 2012; Pacifico et al. 2012). Scott et al. (2018a) found that a future deforestation according to the land use scenario in RCP8.5 leads to a 4% decrease in BVOC emissions at the end of the century. This resulted in a direct aerosol forcing of +0.006 W m^{-2} (decreased reflection by particles in the atmosphere) and a first indirect aerosol forcing of -0.001 W m^{-2} (change in the amount of CCN). Studies not including future land use scenarios but investigating the climate feedbacks leading to increasing future BVOC emissions, have found a direct aerosol effect of -0.06 W m^{-2} (Sporre et al. 2018) and an indirect aerosol effect of -0.45 W m^{-2} (Makkonen et al. 2012; Sporre et al. 2018). The stronger aerosol effects from the feedback compared to the land use are, at least partly, explained by a much larger change in the BVOC emissions.

A positive climate feedback could happen in a future scenario with increasing BVOC emissions, where higher ozone and methane concentrations could lead to an enhanced warming which could further increase BVOC emissions (Arneth et al. 2010). This possible feedback is mediated by NOx levels. One recent study including dynamic vegetation, land use change, CO_2 and climate change found no increase or even a slight decrease in global BVOC emissions at the end of the century (Hantson et al. 2017). There is a lack of understanding concerning the processes governing the BVOC emissions, the oxidation processes in the atmosphere, the role of the BVOC oxidation products in new particle formation and particle growth, as well as general uncertainties in aerosol-cloud interactions. There is a need for continued research into these processes but the current knowledge indicates that changing BVOC emissions need to be taken into consideration when assessing the future climate and how land use will affect it. In summary, the magnitude and sign of net effect of BVOC emissions on the radiation budget and surface temperature is highly uncertain.

2.5 Land impacts on climate and weather through biophysical and GHGs effects

The focus of this section is summarised **Figure 2.13**. We report on what we know regarding the influence land has on climate via biophysical and biogeochemical exchanges. Biogeochemical effects herein only refer to changes in net emissions of CO_2 from land. The influence of land on atmospheric composition is discussed in Section 2.3.

All sections discuss impacts of land on global and regional climate, and climate extremes, whenever the information is available. Section 2.5.1 presents effects of historical and future land use scenarios; section 2.5.2 is devoted to impacts of specific anthropogenic land uses such as forestation, deforestation, irrigation, crop and forest management; section 2.5.3 focuses on how climate driven land changes feedback on climate and section 2.5.4 puts forward that land use changes in one region can affect another region.

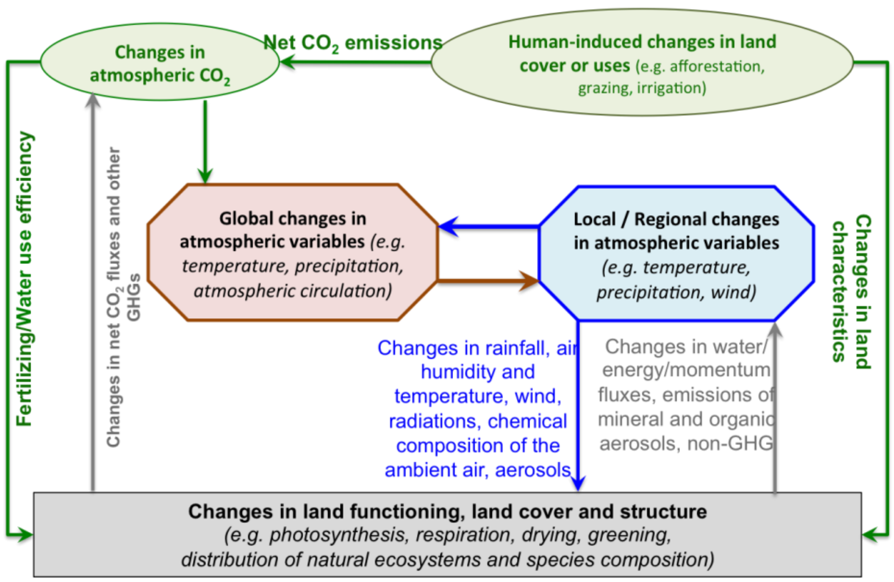

Figure 2.13 Global, local and regional climate changes are the focus of this section. They are examined

through changes in climate states (e.g., changes in air temperature and humidity, rainfall, radiation) as well as through changes in atmospheric dynamics (e.g., circulation patterns). Changes in land that influence climate are either climate- or Human- driven. Green arrows and boxes refer to what we consider herein as imposed changes (forcings). Grey box and arrows refer to responses of land to forcings (green and blue boxes) and feedbacks on those initial forcings. Red and blue boxes and arrows refer respectively to global and local/regional climate changes and their subsequent changes on land

2.5.1 Impacts of historical and future anthropogenic land cover changes

The studies reported below focus essentially on modelling experiments, as there is no direct observation of how historical land use changes have affected the atmospheric dynamics and physics at the global and regional scales. Moreover, the climate modelling experiments only assess the impacts of anthropogenic land cover changes (e.g. deforestation, urbanisation) and neglect the effects of changes in land management (e.g. irrigation, use of fertilisers, choice of species varieties among managed forests or crops). Because of this restricted accounting for land use changes we will use the term land cover changes in the following sub-sections (2.5.1.1 and 2.5.1.2).

Each section starts by describing changes at the global scale, and at the regional scale and ends with what we know about the impacts of those scenarios on extreme weather events, whenever the information is available.

2.5.1.1 Impacts of global historical land cover changes on climate

2.5.1.1.1 At the global level

The contribution of anthropogenic land cover changes to the net global warming throughout the 20th century has been derived from few model-based estimates that account simultaneously for biogeochemical and biophysical effects of land on climate (Table 2.4). The simulated net change in mean global annual surface air temperature, averaged over all the simulations, is a small warming of 0.078±0.093°C, ranging from small cooling simulated by two models (-0.05°C and -0.02°C respectively in (Brovkin et al. 2004) and (Simmons and Matthews 2016), to larger warming simulated by three models (>+0.14°C (Shevliakova et al. 2013; Pongratz et al. 2010; Matthews et al. 2004). When starting from the Holocene period (He et al. 2014) has estimated an even larger net warming effect of anthropogenic land cover changes (+0.72°C).

Table 2.4 Change in mean global annual surface air temperature resulting from anthropogenic land cover change over the historical period. This historical period varies from one simulation to another (middle column).

Reference of the study	Time period	Mean global annual change in surface air temperature (°C)
(Simmons and Matthews 2016)	1750-2000	-0.02
(Shevliakova et al. 2013)	1861-2005	+0.17
(Pongratz et al. 2010)	1900-2000	+0.14
(Matthews et al. 2004)	1700-2000	+0.15
(Brovkin et al. 2004)	1850-2000	-0.05
Mean ± standard deviation		**0.078±0.093**

This net small warming signal results from the competing effects of biophysical cooling (*medium confidence*) and biogeochemical warming (*very high confidence*; Figure 2.14 [2]). The global biophysical cooling alone has been estimated by a larger range of climate models and is -0.10 ± 0.14°C; it ranges from -0.57°C to +0.06°C (e.g. (Zhang et al. 2013a; Hua and Chen 2013; Jones et al. 2013b; Simmons and Matthews 2016), Table A2.1). This cooling is essentially dominated by increases in surface albedo: historical land cover changes have generally led to a dominant brightening of land as discussed in AR5 (Myhre et al. 2013). Reduced incoming long-wave radiation at the surface from reduced evapotranspiration and thus less water vapour in the atmosphere has also been reported as a potential contributor to this cooling (Claussen et al. 2001a). The cooling is however dampened by decreases in turbulent fluxes leading to decreased loss of

[2] FOOTNOTE: The detailed list of all values used to construct this figure is provided in Table A2.1 in the Appendix at the end of the Chapter

heat and water vapour from the land through convective processes. Those non-radiative processes are indeed well-known to often oppose the albedo-induced surface temperature changes (e.g., (Davin and de Noblet-Ducoudre 2010; Boisier et al. 2012)).

Historical land cover changes have contributed to the increase in atmospheric CO_2 content (Section 2.3) and thus to global warming (biogeochemical effect, *very high confidence*). The global mean biogeochemical warming has been calculated from observation-based estimates (+0.25±0.10°C; e.g. (Li et al. 2017a; Avitabile et al. 2016; Carvalhais et al. 2014; Le Quéré et al. 2015)), or estimated from dynamic global vegetation models (+0.24±0.12°C; e.g. (Peng et al. 2017; Arneth et al. 2017a; Pugh et al. 2015; Hansis et al. 2015)) and global climate models (+0.20±0.05°C; (Pongratz et al. 2010; Brovkin et al. 2004; Matthews et al. 2004; Simmons and Matthews 2016)).

The magnitude of these simulated biogeochemical effects may however be underestimated as they do not account for a number of processes such as land management, nitrogen/phosphorus cycles, changes in the emissions of CH_4, N_2O and non-GHG emissions from land (Ward et al. 2014; Arneth et al. 2017b; Cleveland et al. 2015; Pongratz et al. 2018). Two studies have accounted for those compounds and found a global net positive radiative forcing in response to historical anthropogenic land cover changes, indicating a net surface warming (Mahowald et al. 2017; Ward et al. 2014). However, first the estimated biophysical radiative forcing in those studies only accounts for changes in albedo and not for changes in turbulent fluxes. Secondly, the combined estimates also depend on other several key modelling estimates such as climate sensitivity, CO_2 fertilisation caused by land use emissions, possible synergistic effects, validity of radiative forcing concept for land forcing. The comparison with the other above-mentioned modelling studies is thus difficult.

In addition, most of those estimates do not account for the evolution of natural vegetation in unmanaged areas, while observations and numerical studies have reported a greening of the land in boreal regions resulting from both extended growing season and poleward migration of tree lines (Lloyd et al. 2003; Lucht et al. 1995), Section 2.2). This greening enhances global warming via a reduction of surface albedo (winter darkening of the land through the snow-albedo feedbacks, e.g. (Forzieri et al. 2017)). At the same time cooling occurs due to increased evapotranspiration during the growing season, along with enhanced photosynthesis, i.e. increased CO_2 sink (Qian et al. 2010). When feedbacks from the poleward migration of treeline is accounted for together with the biophysical effects of historical anthropogenic land cover change, the biophysical annual cooling (about -0.20°C to -0.22°C on land, -0.06°C globally) is significantly dampened by the warming (about +0.13°C) resulting from the movements of natural vegetation (Strengers et al. 2010). Accounting simultaneously for both anthropogenic and natural land cover changes reduces the cooling impacts of historical land cover change in this specific study.

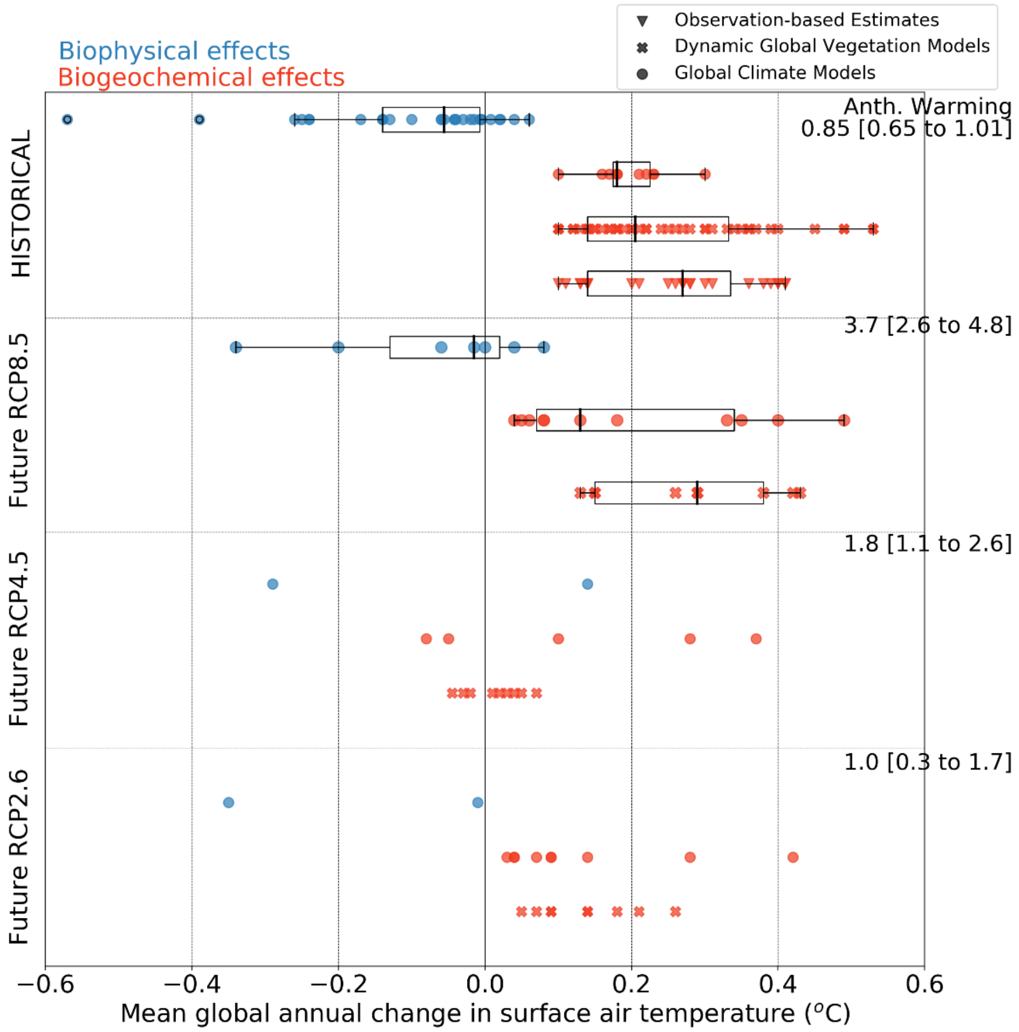

Figure 2.14 Changes in mean global annual surface air temperature (°C) in response to historical and future anthropogenic land cover changes as estimated from a range of studies (see Table A2.1 in the Appendix for detailed information). Temperature changes resulting from biophysical processes (e.g. changes in physical land surface characteristics such as albedo, evapotranspiration, and roughness length) are illustrated using blue symbols; temperature changes resulting from biogeochemical processes (e.g. changes in atmospheric CO_2 composition) use red symbols. Future changes are shown for three distinct scenarios: RCP8.5, RCP4.5 & RCP2.6. The markers 'filled circle', 'filled cross', and 'filled triangle down' represent estimates from respectively global climate models, dynamic global vegetation models (DGVMs), and observations. When the number of estimate is sufficiently large, box plots are overlaid; they show the ensemble minimum, first quartile (25th percentile), median, third quartile (75th percentile), and the ensemble maximum. Scatter points beyond the box plot are the outliers. Details about how temperature change is estimated from DGVMs and observations is provided in Appendix. Numbers on the right hand-side give the mean and the range of simulated mean global annual warming from various climate models.

2.5.1.1.2 At the regional level

The global and annual estimates reported above mask out very contrasted regional and seasonal differences. Biogeochemical effects of anthropogenic land cover change on temperature follow the spatial patterns of GHG-driven climate change with stronger warming over land than ocean, and stronger warming in northern high latitudes than in the tropics and equatorial regions (Arctic amplification). Biophysical effects on the contrary are much stronger where land cover has been modified than in their surroundings (see 2.5.4 for a discussion on non-local effects). Very contrasted regional temperature changes can thus result depending on whether biophysical processes dampen or exacerbate biogeochemical impacts.

Figure 2.15 compares, for seven climate models, the biophysical effects of historical anthropogenic land cover change in North America and Eurasia (essentially cooling) to the regional warming resulting from the increased atmospheric CO_2 content since pre-industrial times ((De Noblet-Ducoudré et al. 2012); comparing 1973–2002 to 1871–1900). It shows a dominant biophysical cooling effect of changes in land cover, at all seasons, as large as the regional footprint of anthropogenic global warming. Averaged over all agricultural areas of the world (Pongratz et al. 2010) reported a 20th century biophysical cooling of -0.10°C, and (Strengers et al. 2010) reported a land induced cooling as large as -1.5°C in western Russia and eastern China between 1871 and 2007. There is thus *medium confidence* that anthropogenic land cover change has dampened warming in many regions of the world over the historical period.

Very few studies have explored the effects of historical land cover changes on seasonal climate. There are however evidences that the seasonal magnitude and sign of those effects at the regional level are strongly related to soil-moisture/evapotranspiration and snow regimes, particularly in temperate and boreal latitudes (Teuling et al. 2010; Pitman and de Noblet-Ducoudré 2012; Alkama and Cescatti 2016). Quesada et al. (2017a) showed that atmospheric circulation changes can be significantly strengthened in winter for tropical and temperate regions. However, the lack of studies underlines the need for a more systematic assessment of seasonal, regional and other than mean temperature metrics in the future.

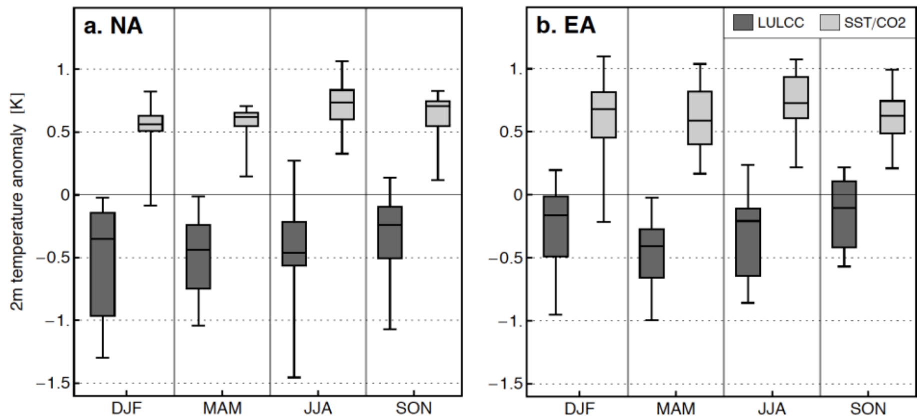

Figure 2.15 Simulated changes in mean surface air temperature (°C) between the pre-industrial period (1870–1900) and present-day (1972–2002) for all seasons and for a) North America and b) Eurasia (De Noblet-Ducoudré et al. 2012)**. Light grey boxes are the changes simulated in response to increased atmospheric GHG content between both time periods and subsequent changes in sea-surface temperature and sea-ice extent (SST/CO_2); the CO_2 changes accounted for include emissions from all sources including land use. Dark grey boxes are the changes simulated in response to the biophysical effects of historical land cover changes. The box-and-whisker plots have been drawn using results from seven climate models and ensembles of ten simulations per model and time period. The bottom and top of the each grey box are the 25th and 75th percentiles, and the horizontal line within each box is the 50th percentile (the median). The whiskers (straight lines) indicate the ensemble maximum and minimum values. Seasons are respectively December-January-February (DJF), March-April-May (MAM), June-July-August (JJA) and September-October-November (SON). North America and Eurasia are extended regions where land use changes are the largest between the two time periods considered (their contours can be found in Figure 1 of** (De Noblet-Ducoudré et al. 2012)**).**

2.5.1.1.3 Effects on extremes

The effect of historical deforestation on extreme temperature trends is intertwined with the effect of other climate forcings thus making it difficult to quantify based on observations. Based on results from four climate models, the impact of historical anthropogenic land cover change on temperature and precipitation extremes was found to be locally as important as changes arising from increases in atmospheric CO_2 and sea-surface temperatures, but with a lack of model agreement on the sign of changes (Pitman et al. 2012). In some regions the impact of land cover change masks or amplifies the effect of increased CO_2 on extremes (Avila et al. 2012; Christidis et al. 2013). Using an observational constraint for the local biophysical effect of land cover change applied to a set of CMIP5 climate models, (Lejeune et al. 2018) found that historical deforestation increased extreme hot temperatures in northern mid-latitudes. The results also indicate a stronger impact on the warmest temperatures compared to mean temperatures. Findell et al. (2017) reached similar conclusions, although using only a single climate model. Importantly, the climate models involved in these three studies did not consider the effect of management changes which have been shown to be important, as discussed Section 2.5.2.

Based on the studies discussed above there is yet *limited evidence* but *high agreement* that land cover change affects local temperature extremes more than mean values. Observational studies assessing the role of land cover on temperature extremes are still very limited (Zaitchik et al. 2006; Renaud and Rebetez 2008), but suggest that trees dampen seasonal and diurnal temperature variations at all latitudes and even more so in temperate regions compared to short vegetation (Chen et al. 2018; Duveiller et al. 2018b; Li et al. 2015a; Lee et al. 2011). Furthermore, trees also locally dampen the amplitude of heat extremes (Renaud and Rebetez 2008; Zaitchik et al. 2006) although this result depends on the forest type, coniferous trees providing less cooling effect than broadleaf trees (Renaud et al. 2011; Renaud and Rebetez 2008).

2.5.1.2 Impacts of future global land cover changes on climate

2.5.1.2.1 At the global level

The most extreme emissions scenario (RCP8.5) that has been developed for the last coordinated modelling intercomparison of climate models (CMIP5) is the one that has received the most attention in the literature with respect to how projected future anthropogenic land cover changes (Hurtt et al. 2011) will affect the very large simulated global warming.

Seven model-based studies have examined both the biophysical and biogeochemical effects of anthropogenic changes in land-cover, as projected in RCP8.5, on future climate change (Table 2.5 ; (Simmons and Matthews 2016; Davies-Barnard et al. 2014; Boysen et al. 2014)). They all agree on a biogeochemical warming, ranging from +0.04°C to +0.35°C, in response to land cover change. Two models predict an additional biophysical warming, while the others agree on a biophysical cooling that dampens (or overrule) the biogeochemical warming. Using a wider range of global climate models, the biogeochemical warming (*high confidence*) is +0.20±0.15°C whereas it is +0.28±0.11°C when estimated from dynamic global vegetation models (Pugh et al. 2015; Stocker et al. 2014). This biogeochemical warming is compensated for by a biophysical cooling (*medium confidence*) of -0.10±0.14°C (Quesada et al. 2017a; Davies-Barnard et al. 2015; Boysen et al. 2014). The estimates of temperature changes resulting from anthropogenic land cover changes alone remain very small compared to the projected mean warming of +3.7°C by the end of the 21st century (ranging from 2.6 to 4.8°C depending on the model and compared to 1986-2005; Figure 2.14).

Table 2.5 Change in mean global annual surface air temperature resulting from anthropogenic land cover changes projected for the future, according to three different scenarios: RCP8.5, RCP4.5 and RCP2.6. Temperature change resulting from biophysical and biogeochemical effects of land cover change are examined.

Reference of the study	Time period	Mean global annual change in surface air temperature (°C) Biophysical / Biogeochemical		
		RCP2.6	RCP4.5	RCP8.5
(Simmons and Matthews 2016)	2000-2100	-0.35 / +0.42	-0.29 / + 0.37	-0.34 / + 0.35
(Davies-Barnard et al. 2014)	2005-2100	-0.01 / +0.04	+0.14 / -0.08	-0.015 / +0.04
(Boysen et al. 2014)	2005-2100			+0.04 / +0.08 0 / +0.05 +0.08 / +0.06 -0.20 / +0.13 -0.06 / +0.33

Two other projected land cover change scenarios have been examined (RCP4.5 and RCP2.6; Table 2.5 ; Figure 2.14) but only one climate modelling experiment has been carried out for each, to estimate the biophysical impacts on climate of those changes (Davies-Barnard et al. 2015). For RCP2.6, earth system and dynamic global vegetation models agree on a systematic biogeochemical warming resulting from the imposed land cover changes, ranging from +0.03 to +0.28°C (Brovkin et al. 2013a), which is significant compared to the projected mean climate warming of +1°C by the end of the 21st century (ranging from 0.3 to 1.7°C depending on the models, compared to 1986-2005). A very small biophysical cooling is expected from the one estimate. For RCP4.5 biophysical warming is expected from only one estimate, and results from a projected large forestation in the temperate and high latitudes. There is no agreement on the sign of the biogeochemical effect: there are as many studies predicting cooling as warming, whichever the method to compute those effects (earth system models or dynamic global vegetation models).

Previous scenarios (Special Report on Emission Scenarios (SRES), results of climate studies using those scenarios were reported in AR4) displayed larger land use changes than the more recent ones (RCP, AR5). There is *low confidence* from some of those previous scenarios (SRES A2 and B1) of a small warming effect (+0.2 to +0.3°C) of anthropogenic land cover change on mean global climate, this being dominated by the release of CO_2 in the atmosphere from land conversions (Sitch et al. 2005). This additional warming remains quite small when compared to the one resulting from the combined anthropogenic influences [+1.7°C for SRES B1 and +2.7°C for SRES A2]. A global biophysical cooling of -0.14°C is estimated in response to the extreme land cover change projected in SRES A2, a value that far exceeds the impacts of historical land use changes (-0.05°C) calculated using the same climate model (Davin et al. 2007). The authors derived a biophysical climatic sensitivity to land use change of about -0.3°C $W.m^{-2}$ for their model, whereas a warming of about 1°C $W.m^{-2}$ is obtained in response to changes in atmospheric CO_2 concentration.

Those studies generally do not report on changes in atmospheric variables other than surface air temperature, thereby limiting our ability to assess the effects of anthropogenic land cover changes on regional climate Sitch et al. (2005). However, reported small reductions in rainfall via changes in biophysical properties of the land, following the massive tropical deforestation in SRES A2 (+0.5 and +0.25 mm day^{-1} respectively in the Amazon and Central Africa). They also report opposite changes, that is increased rainfall of about 0.25 mm day^{-1} across the entire tropics and subtropics, triggered by biogeochemical effects of this same deforestation.

2.5.1.2.2 At the regional level

In regions that will undergo land cover changes, dampening of the future anthropogenic warming can be as large as -26% while enhancement is always smaller than 9% within RCP8.5 by the end of the 21st century (Boysen et al. 2014). Voldoire (2006) show that, by 2050 and following the SRES B2 scenario, the contribution of land cover changes to the total temperature change can be as large as 15% in many boreal regions, and as large as 40% in south western tropical Africa. Feddema et al. (2005) simulate large decreases in the diurnal temperature range in the future (2050 and 2100 in SRES B1 and A2) following tropical deforestation in both scenarios. In the Amazon for example the diurnal temperature range is lowered by 2.5°C due to increases in minimum temperature while little change is obtained for the maximum value.
There is thus *medium evidence* that future anthropogenic land cover change will have a significant effect on regional temperature via biophysical effects in many regions of the world. There is however *no agreement* on whether warming will be dampened or enhanced and there is *no agreement* on the sign of the contribution across regions.

There are very few studies that go beyond analysing the changes in mean surface air temperature. Some studies attempted to look at global changes in rainfall and found no significant influence of future land cover changes (Brovkin et al. 2013a; Sitch et al. 2005; Feddema et al. 2005). Quesada et al. (2017a,b) however carried out a systematic multi-model analysis of the response of a number of atmospheric, radiative and hydrological variables (e.g. rainfall, sea level pressure, geopotential height, wind speed, soil-moisture, turbulent heat fluxes, shortwave and longwave radiation, cloudiness) to RCP8.5 land cover scenario. In particular, they found a significant reduction of rainfall in 6 out of 8 monsoon regions studied (Figure 2.16) of about 1.9% to 3% (which means more than -0.5mm day^{-1} in some areas) in response to future anthropogenic land cover changes. Including those changes in global climate models reduces the projected increase in rainfall by about 9% to 41% in those same regions, when all anthropogenic forcings are accounted for (30% in the Global Monsoon region as defined by (Wang and Ding 2008)). In addition, they found a shortening of the monsoon season of one to four days. They conclude that the projected future increase in monsoon rains may be overestimated by those models that do not yet include biophysical effects of land cover changes. Overall, the regional hydrological cycle was found to be substantially reduced and wind speed significantly strengthened in response to regional deforestation within the tropics, with magnitude comparable to projected changes with all forcings (Quesada et al. 2017b).

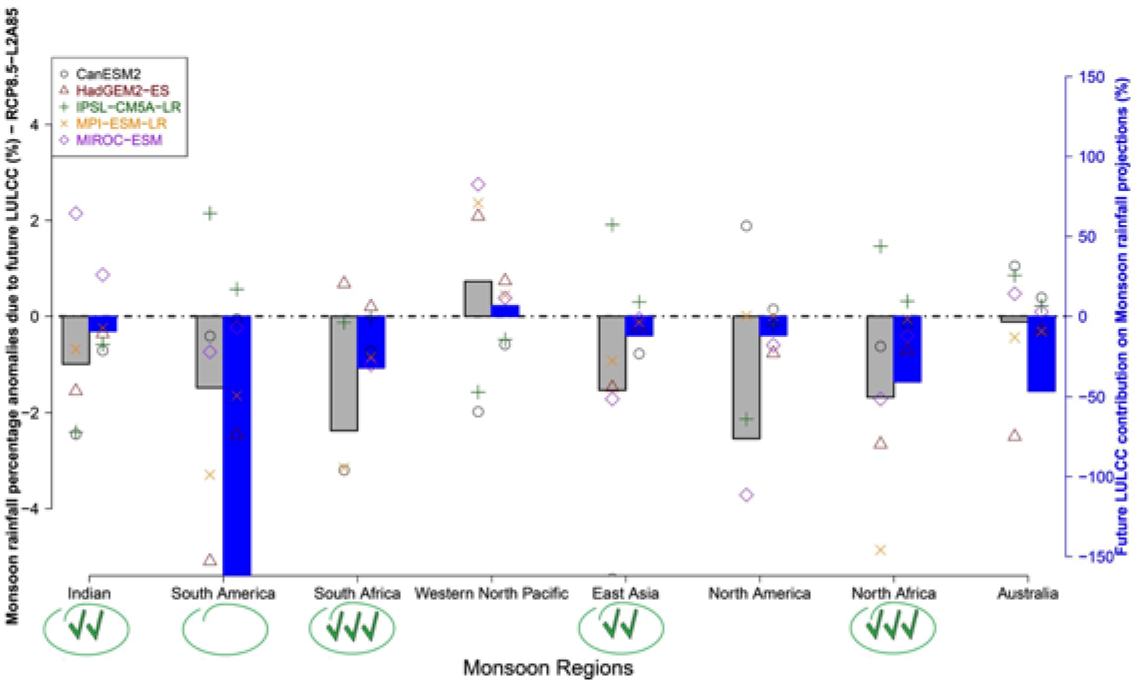

Figure 2.16 Changes in monsoon rainfall in RCP8.5 scenario resulting from projected changes in anthropogenic land cover, in eight monsoonal regions (%, grey bars). Differences are calculated between the end of the 21st century (2071–2100) and the end of the 20th century (1976–2005); percent change is calculated with reference to 1976-2005. Blue bars refer to the relative contribution of land cover changes (in %) to future rainfall projections: it is the ratio between the change in rainfall responding to land cover changes and the one responding to all anthropogenic changes (Quesada et al. 2017b). Negative values mean that changes in land cover have an opposite effect (dampening) on rainfall compared to the effects of all anthropogenic changes. Monsoon regions have been defined following (Yim et al. 2014). The changes have been simulated by five climate models (Brovkin et al. 2013, symbols). Results are shown for December-January-February for southern hemisphere regions, and for June-July-August for northern hemisphere regions. Statistical significance is given by green tick marks and circles: one, two, and three green tick marks are displayed for the regions where at least 80% of the climate models have regional changes significant at the 66th, 75th, and 80th confidence level, respectively; green circles are added when the regional values are also significant at 90th confidence level. Note that future land cover change impacts on South American monsoon are neither significant nor robust among models, along with very small future projected changes in South American monsoon rainfall.

2.5.1.2.3 Effects on extremes

Results from a set of climate models have shown that the impact of future anthropogenic land cover change on extreme temperatures can be of similar magnitude as the changes arising from half a degree global mean annual surface temperature change (Hirsch et al. 2018). However, this study also found a lack of agreement between models with respect to the magnitude and sign of changes, thus making land cover change a factor of uncertainty in future climate projections.

2.5.2 Impacts of specific land use changes

2.5.2.1 Impacts of deforestation and forestation

Deforestation or forestation[3], wherever it occurs, triggers simultaneously warming and cooling of the surface and of the atmosphere via changes in its various characteristics (Pitman 2003; Strengers et al. 2010; Bonan 2008b). Following deforestation, warming results from a) the release of CO_2 and other GHG in the atmosphere (biogeochemical impact) and subsequent increase in incoming infrared radiation at surface (greenhouse effect), b) a decreased in the total loss of energy through turbulent fluxes (latent and sensible

[3] FOOTNOTE: The term « forestation » is used herein as this chapter does not distinguish between afforestation and reforestation. In model-based studies, simulations with and without trees are compared; in observation-based estimates, sites with and without trees are compared.

heat fluxes) resulting from reduced surface roughness, c) an increased incoming solar radiation following reduced cloudiness that often (but not always) accompanies the decreased total evapotranspiration. Cooling occurs in response to d) increased surface albedo that reduces the amount of absorbed solar radiation, e) reduced incoming infrared radiation triggered by the decreased evapotranspiration and subsequent decrease in atmospheric water vapour. Points b-c-d-e are referred to as biophysical effects. Deforestation and forestation also alter rainfall and winds (horizontal as well as vertical as will be further discussed below).

The literature that discusses the effects of forestation on climate is more limited than for deforestation, but they reveal a similar climatic response with opposite sign as further discussed below. For each latitudinal band (tropical, temperate and boreal) how very large scale deforestation or forestation impacts global mean climate is examined, followed by examination of the large-scale changes in the specific latitudinal band and end up with more regionally focused analysis. Large scale idealised deforestation or forestation experiments are often carried out with global or regional climate models as they allow to understand and measure how sensitive climate is to very large changes in land cover (similar to the instant doubling of CO_2 in climate models to calculate the climatic sensitivity to GHGs). Details of the model-based studies discussed below can be found in Table A2.2 in the Appendix.

2.5.2.1.1 Global and regional impacts of deforestation/forestation in tropical regions

A pan-tropical deforestation would lead to the net release of CO_2 from land and thus to mean global annual warming, with model-based estimates of biogeochemical effects ranging from +0.19 to +1.06°C, with a mean value of +0.53±0.32°C (Ganopolski et al. 2001; Snyder et al. 2004; Devaraju et al. 2015a; Longobardi et al. 2016; Perugini et al. 2017). There is however *no agreement* between models on the magnitude and sign of the biophysical effect of such changes at the global scale (the range spans from -0.5°C to +0.7°C with a mean value of +0.1±0.27°C; Figure 2.17 ; e.g. (Devaraju et al. 2015c; Snyder 2010; Longobardi et al. 2016a)). This is the result of many compensation effects in action: increased surface albedo following deforestation, decreased atmospheric water vapour content due to less tropical evapotranspiration, decreased loss of energy from tropical land in the form of latent and sensible heat fluxes.

There is however *high confidence* that such large land cover change would lead to a mean biophysical warming when averaged over the deforested land. A mean warming of +0.61±0.48°C is found over the entire tropics. Reversely, biophysical regional cooling and global warming is expected from forestation (Wang et al. 2014b; Bathiany et al. 2010a).

Large-scale deforestation (whether pan-tropical or imposed at the sub-continent level, e.g. the Amazon) results in significant mean rainfall decrease (Lawrence and Vandecar 2015; Lejeune et al. 2015; Perugini et al. 2017). In their review (Perugini et al. 2017) reported an average simulated decrease of -288 ± 75 mm yr^{-1} (95%-confidence interval). Inversely large-scale forestation increases tropical rainfall by 41 ± 21 mm yr^{-1}. The magnitude of the change in precipitation strongly depends on the type of land cover conversion. For instance, conversion of tropical forest to bare soil causes larger reductions in regional precipitation than conversion to pasture (respectively -470 ± 60 mm yr^{-1} and -220 ± 100 mm yr^{-1}). Biogeochemical effects in response to pan-tropical deforestation, particularly CO_2 release, are generally not taken into account in those studies but could intensify the hydrological cycle and thus precipitation (Kendra Gotangco Castillo and Gurney 2013).

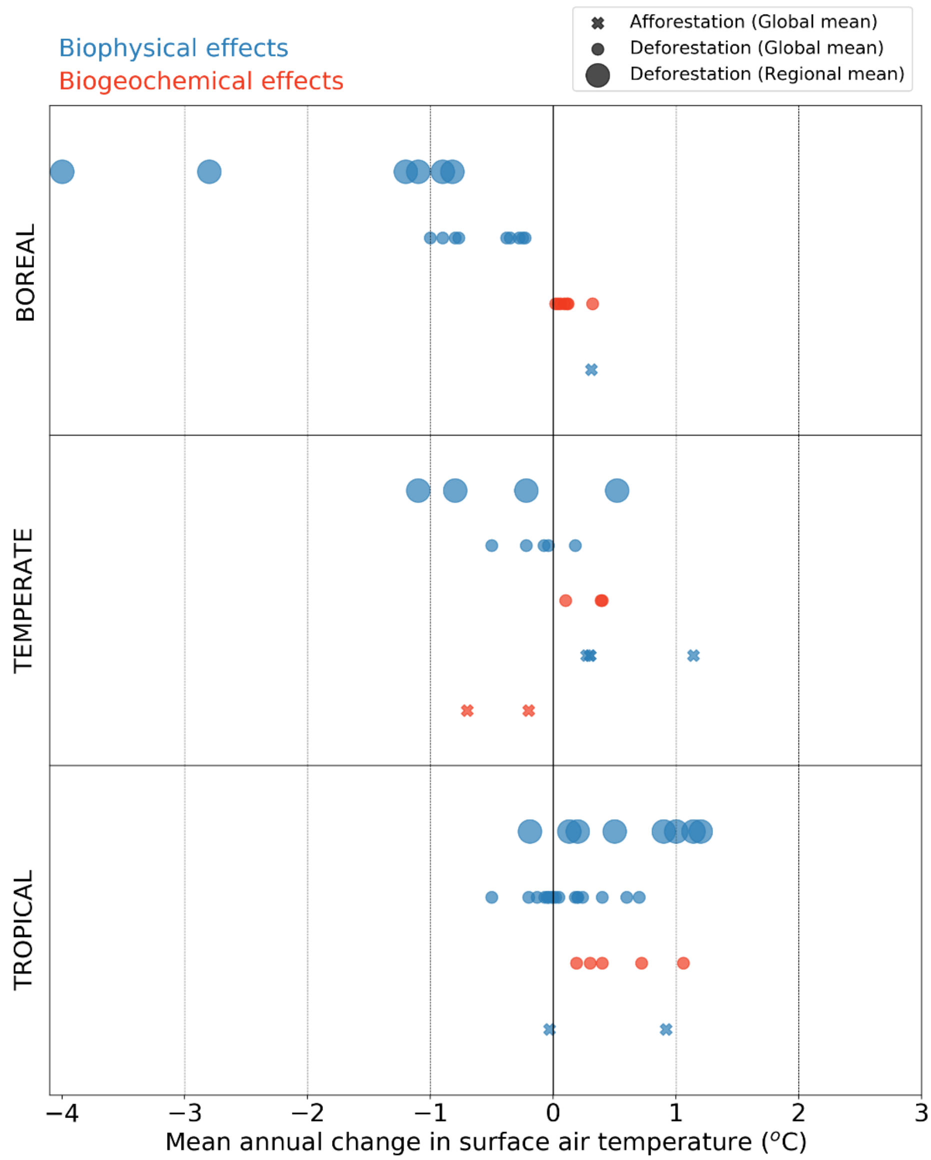

Figure 2.17 Changes in mean annual surface air temperature (°C) in response to idealised large scale deforestation (circles) or forestation (crosses), estimated from a range of studies (see Table A2.2 in the Appendix for detailed information and references to the studies). Temperature changes resulting from biophysical processes (e.g. changes in physical land surface characteristics such as albedo, evapotranspiration, and roughness length) are illustrated using blue symbols; temperature changes resulting from biogeochemical processes (e.g. changes in atmospheric CO_2 composition) use red symbols. Small blue and red circles, and crosses, are model-based estimates of changes in temperature averaged globally. Large circles are estimates averaged only over the latitudinal band where deforestation is imposed.

Specific model-based deforestation studies have been carried out for Africa (Hagos et al. 2014; Boone et al. 2016; Xue et al. 2016; Nogherotto et al. 2013; Hartley et al. 2016; Klein et al. 2017; Abiodun et al. 2012), southern America (Butt et al. 2011; Wu et al. 2017; Spracklen and Garcia-Carreras 2015; Lejeune et al. 2015), South-East Asia (Ma et al. 2013b; Werth and Avissar 2005; Mabuchi et al. 2005; Tölle et al. 2017). All found decreases in evapotranspiration following deforestation (*high agreement*), resulting in surface warming despite the competing effect from increased surface albedo (*high agreement*). Changes in thermal gradients between deforested and adjacent regions, between land and ocean, affect horizontal surface winds (*high agreement*) and thus modify the areas where rainfalls as discussed in Section 2.5.4. An increase in the land-sea thermal contrast has been found in many studies as surface friction is reduced by deforestation, thus increasing the monsoon flow in Africa and South America (Wu et al. 2017).

Observation-based estimates all agree that deforestation increases local land-surface and ambient air temperatures in the tropics, while forestation has the reverse effect (*very high confidence;* (Prevedello et al. 2019; Schultz et al. 2017; Li et al. 2015b; Alkama and Cescatti 2016)). There is *very high confidence* that forests are cooler than any shorter vegetation (crops, grasses, bare soil) during daytime due to larger transpiration rates, and there is *high confidence* that the amplitude of the diurnal cycle is smaller in the presence of forests.

Large-scale forestation scenarios of West Africa (Abiodun et al. 2012), eastern China (Ma et al. 2013a) or Saharan and Australian deserts (Ornstein et al. 2009; Kemena et al. 2017) all concluded that regional surface cooling is simulated wherever trees are grown (-2.5°C in the Sahel and -1°C in the Savanna area of West Africa, up to -8°C in western Sahara, -1.21°C over land in eastern China) while cooling of the ambient air is smaller (-0.16°C). In the case of Savanna forestation this decrease entirely compensates the GHG induced future warming (+1°C following the SRES A1B scenario). West African countries thus have the potential to reduce, or even totally cancel at some places, the GHG-induced warming in the deforested regions (Abiodun et al. 2012). However, this is compensated by enhanced warming in adjacent countries (non-local effect).

2.5.2.1.2 Global and regional impacts of deforestation/forestation in temperate regions

As for the tropics, model-based experiments show that large-scale temperate deforestation would induce a small mean global annual warming through the net release of CO_2 into the atmosphere (ranging from +0.10 to +0.40°C with a mean value of +0.20±0.13°C, Figure 2.17), whereas there is less agreement on the sign of the mean global annual temperature change resulting from biophysical processes: estimates range from -0.5°C to +0.18°C with a mean value of -0.13 ± 0.22°C. There is also *very low agreement* on the mean annual temperature change in the temperate zone (-0.4±0.62°C; (Phillips et al. 2007; Snyder et al. 2004b; Longobardi et al. 2016a; Devaraju et al. 2015a, 2018b)). There is *medium agreement* on a global and latitudinal biophysical warming in response to forestation (Figure 2.17 ; (Laguë and Swann 2016a; Swann et al. 2012a; Gibbard et al. 2005; Wang et al. 2014b)), but this is based on a smaller number of studies.

The lack of agreement at the annual scale among the climate models is however masking *rising agreement* regarding seasonal impacts of deforestation at those latitudes. There is *high agreement* that temperate deforestation leads to summer warming and winter cooling (Bright et al. 2017; Zhao and Jackson 2014; Gálos et al. 2011, 2013; Wickham et al. 2013; Ahlswede and Thomas 2017; Anderson-Teixeira et al. 2012; Anderson et al. 2011; Chen et al. 2012; Strandberg and Kjellström 2018). The winter cooling is driven by the increased surface albedo, amplified by the snow-albedo feedback. In some models and when deforestation is simulated for very large areas, the cooling is further amplified by high latitude changes in sea-ice and snow extent (polar amplification). Summer warming occurs because the latent and sensible heat fluxes, that take energy out of the surface, diminish with the smaller roughness length and lower evapotranspiration efficiency of low vegetation as compared to tree canopies (Davin and de Noblet-Ducoudre 2010; Anav et al. 2010). Conversely, there is *high agreement* that forestation in North America or in Europe cools surface climate during summer time, especially in regions where water availability can support large evapotranspiration rates. In temperate regions with water deficits, the simulated change in evapotranspiration following forestation will be insignificant while the decreased surface albedo will favour surface warming.

Observation-based estimates confirm the existence of a seasonal pattern of response to deforestation, with colder winters anytime there is snow on the ground and anywhere soils are brighter than trees, and warmer summers (Schultz et al. 2017; Wickham et al. 2014; Juang et al. 2007; Tang et al. 2018; Peng et al. 2014; Zhang et al. 2014b; Prevedello et al. 2019; Li et al. 2015b; Alkama and Cescatti 2016). In contrast forestation induces cooler summers wherever trees have access to sufficient soil moisture to transpire. The magnitude of the cooling depends on the wetness of the area of concern (Wickham et al. 2013) as well as on the original and targeted species and varieties implicated in the vegetation conversion (Peng et al. 2014; Juang et al. 2007).

There is also *high confidence* from observation-based estimates that mean annual daytime temperatures are warmer following deforestation, while night time temperatures are cooler (Schultz et al. 2017; Wickham et al. 2014; Juang et al. 2007; Tang et al. 2018; Prevedello et al. 2019; Peng et al. 2014; Zhang et al. 2014b; Li et al. 2015b; Alkama and Cescatti 2016). Deforestation then increases the amplitude of diurnal temperature variations while forestation reduces it (*high confidence*). Two main reasons have been put forward to explain why nights are warmer in forested areas: their larger capacity to store heat, and the existence of a nocturnal temperature inversion bringing warmer air from aloft.

In addition to those seasonal and diurnal fluctuations, (Lejeune et al. 2018) found systematic warming of the hottest summer days following historical deforestation in the northern mid-latitudes, and this echoes (Strandberg and Kjellström 2018) who argue that the August 2003 and July 2010 heat-waves could have been largely mitigated if Europe had been largely forested.

In a combined modelling of large-scale forestation of western Europe and climate change scenario (SRES A2) (Gálos et al. 2013) found a relatively small dampening potential of additional forest on ambient air temperature at the end of the 21st century when compared to the beginning (the cooling resulting from land cover changes is -0.5°C whereas the GHG-induced warming exceeds 2.5°C). Influence on rainfall was however much larger and significant. Projected annual rainfall decreases following warming were cancelled in Germany and significantly reduced in both France and Ukraine through forestation. In addition forestation also decreased the number of warming-induced dry days, but increased the number of extreme precipitation events.

The net impact of forestation, combining both biophysical and biogeochemical effects, has been tested in the warmer world predicted by RCP 8.5 scenario (Sonntag et al. 2016, 2018). The cooling effect from the addition of 8 Mkm^2 of forests following the land use RCP 4.5 scenario was too small (-0.27°C annually) to dampen the RCP 8.5 warming. It however reached about -1°C in some temperate regions and -2.5°C in boreal ones. This is accompanied by a reduction in the number of extremely warm days.

2.5.2.1.3 Global and regional impacts of deforestation/forestation in boreal regions

Consistent with what we have previously discussed for temperate and tropical regions, large-scale boreal deforestation induces a biogeochemical warming of +0.11±0.09°C (Figure 2.17). But contrary to those other latitudinal bands, the biophysical effect is a consistent cooling across all models (-0.55±0.29°C when averaged globally). It is also significantly larger than the biogeochemical warming (e.g. (Dass et al. 2013; Longobardi et al. 2016a; Devaraju et al. 2015a; Bathiany et al. 2010a; Devaraju et al. 2018b)). It is driven by the increased albedo, enhanced by the snow-albedo feedback as well as by an increase in sea-ice extent in the Arctic. Over the boreal lands, the cooling is as large as -1.8 ± 1.2°C. This mean annual cooling however masks out a seasonal contrast as discussed in (Strandberg and Kjellström 2018) and (Gao et al. 2014): during summer time, following the removal of forest, the decreased evapotranspiration results in a significant summer warming that outweighs the effect of an increased albedo effect.

The same observation-based estimates as discussed in the previous sub-section show similar patterns as for the temperate latitudes: seasonal and daily contrasts. (Schultz et al. 2017) however found that mean annual nighttime changes are as large as daytime ones in those regions (mean annual nocturnal cooling -1.4±0.10°C, balanced by mean annual daytime warming of 1.4±0.04°C). This contrasts with both temperate and tropical regions where daytime changes are always larger than nighttime ones.

Arora and Montenegro (2011) combined large-scale forestation and climate change scenario (SRES A2): forestation of either 50% or 100% of the total agricultural area was gradually prescribed between years 2011 and 2060 everywhere. In addition, boreal, temperate and tropical forestation have been tested separately. Both biophysical and biogeochemical effects were accounted for. The net simulated impact of forestation was a cooling varying from -0.04°C to -0.45°C, depending on the location and magnitude of the additional forest cover. It was, however, quite marginal compared to the large global warming resulting from anthropogenic GHG emissions (+3°C at the end of the 21st century). In their experiment, forestation in boreal regions led to biophysical warming and biogeochemical cooling that compensated each other, whereas forestation in the tropics led to both biophysical and biogeochemical cooling. The authors concluded that tropical forestation is three times more effective in cooling down climate than are boreal or temperate forestation.

2.5.2.1.4 Conclusion

In conclusion, planting trees will always result in capturing more atmospheric CO_2 and thus in mean annual cooling of the globe (*very high confidence*). At the regional level however the magnitude and sign of the local temperature change depends on a) where forestation occurs, b) its magnitude, c) the level of warming under which the land cover change is applied and d) the land conversion type. This is because the background climatic conditions (e.g., precipitation and snow regimes, mean annual temperature) within which the land cover changes occur vary across regions (Pitman et al. 2011; Montenegro et al. 2009; Juang et al. 2007; Wickham et al. 2014; Hagos et al. 2014; Voldoire 2006; Feddema et al. 2005; Strandberg and Kjellström 2018). In addition there is *high confidence* that estimates of the influence of any land cover or land use change on surface temperature from the sole consideration of the albedo and the CO_2 effects is incorrect as changes in turbulent fluxes (i.e., latent and sensible heat fluxes) are large contributors to local temperature change (Bright et al. 2017).

There is *high confidence* that in boreal and temperate latitudes the presence of forest cools temperature in warmer locations and seasons provided that the soil is not dry, whereas it warms temperature in colder locations and seasons provided the soil is brighter than trees or covered with snow. In the humid tropics forestation increases evapotranspiration year round and thus decreases temperature (*high confidence*). In tropical areas with a strong seasonality of rainfall, forestation will also increase evapotranspiration year round unless the soil becomes too dry. In all regions there is *medium confidence* that the diurnal temperature range decreases with increasing forest cover, with potentially reduced extreme values of temperature

Although there is not enough literature yet that rigorously compares both biophysical and biogeochemical effects of realistic scenarios of forestation, there is *high confidence* that, at the local scale (that is where the forest change occurs) biophysical effects on surface temperature are far more important than the effects resulting from the changes in emitted CO_2.

What is lacking in the literature as of today is an estimate of the impacts natural disturbances in forests will have on local climates and on the build-up of atmospheric CO_2. (O'Halloran et al. 2012) for example illustrated with many examples that changes in albedo following disturbances can result in radiative forcing changes opposite to and as large as the ones resulting from the associated changes in the net release of CO_2 by land. The resulting climate effects depend on the duration of the perturbation and of the following recovery of vegetation.

2.5.2.2 Impacts of changes in land management

There have been little changes in net cropland area over the past 50 years (at the global scale) compared to continuous changes in land management (Erb et al. 2017). Similarly, in Europe change in forest management was a very significant anthropogenic land change. Management affects water, energy and GHG fluxes exchanged between the land and the atmosphere, and thus temperature and rainfall, sometimes to the same extent as changes in land cover do as discussed in (Luyssaert et al. 2014b).

The effects of irrigation, which is a practice that has been substantially studied, and one attempt to manage solar radiation via increases in cropland albedo (geoengineering the land) is assessed, along with discussion of recent findings on the effects of forest management on local climate, although there is not enough literature yet on this topic to carry out a real assessment. The effects of urbanisation on climate are assessed in a specific cross-chapter box within this chapter (Cross-Chapter Box 4 : Climate change and urbanisation, in this chapter).

There are a number of other practices that exist, some of them being reported in Section 2.6 and chapter 6 whose importance for climate mitigation has been examined. There is however not enough literature available for assessing their biophysical effect on climate. Few papers are generally found per agricultural practice, e.g., (Jeong et al. 2014b) for double cropping, (Bagley et al. 2017) for the timing of the growing season, and (Erb et al. 2017) for a review of ten management practices.

Similarly there are very few studies that have examined how choosing species varieties and harvesting strategies in forest management impacts climate through biophysical effects, and how those effects compare to the consequences of the chosen strategies on the net CO_2 sink of the managed forest. The modelling studies highlight the existence of competing effects between e.g. the capacity of certain species to store more carbon than others (thus inducing cooling) while at the same time reducing the total evapotranspiration loss and absorbing more solar radiation via lower albedo (thus inducing warming) (Naudts et al. 2016a; Luyssaert et al. 2018).

2.5.2.2.1 Irrigation

There is substantial literature on the effects of irrigation on local, regional and global climate as this is a major land management. There is *very high confidence* that irrigation increases total evapotranspiration, increases the total amount of water vapour in the atmosphere, and decreases mean surface daytime temperature within the irrigated area and during the time of irrigation(Bonfils and Lobell 2007; Alter et al. 2015; Chen and Jeong 2018; Christy et al. 2006; Im and Eltahir 2014; Im et al. 2014; Mueller et al. 2015). Decreases in maximum daytime temperature can locally be as large as -3°C to -8°C (Cook et al. 2015; Han and Yang 2013; Huber et al. 2014; Alter et al. 2015; Im et al. 2014). Estimates of the contribution of irrigation to past historical trends in ambient air temperature vary between -0.07°C and -0.014°C/decade in Northern China (Han and Yang 2013; Chen and Jeong 2018) while being quite larger in California (-0.14°C to -0.25°C/decade; (Bonfils and Lobell 2007)). Surface cooling results from increased energy being taken up from the land via larger evapotranspiration rates. In addition, there is growing evidence from modelling studies that such cooling can locally mitigate the effect of heatwaves (Thiery et al. 2017; Mueller et al. 2015).

There is *no agreement* on changes in nighttime temperatures as discussed in (Chen and Jeong 2018) who summarised the findings from observations in many regions of the World (India, China, North America and eastern Africa; Figure 2.18). Where nighttime warming is found (Chen and Jeong 2018; Christy et al. 2006), two explanations are put forward: the first is an increase in incoming long-wave radiation in response to increased atmospheric water vapour content (greenhouse effect); the second is an increased storage of heat in the soil during daytime, because of the larger heat capacity of a moister soil, heat that is then released to the atmosphere at night.

There is *robust evidence* from modelling studies that implementing irrigation enhances rainfall although there is *very low confidence* on where this increase occurs. When irrigation occurs in Sahelian Africa, during the monsoon period, rainfall is decreased over the irrigated areas (*high agreement*) and increases south-west if the crops are located in western Africa (Alter et al. 2015) and east / north-east when crops are located further East in Sudan (Im and Eltahir 2014; Im et al. 2014) The cooler irrigated surfaces in the Sahel, because of their greater evapotranspiration, inhibits convection and creates an anomalous descending motion over crops that suppresses rainfall but influences the circulation of monsoon winds. Irrigation in India occurs prior to the start of the monsoon season and the resulting land cooling decreases the land-sea temperature contrast. This can delay the onset of the Indian monsoon and decrease its intensity (Niyogi et al. 2010; Guimberteau et al. 2012). Results from (De Vrese et al. 2016a) modelling study suggest that part of the excess rainfall triggered by Indian irrigation falls westward, in the horn of Africa. The theory behind those local and downwind changes in rainfall support the findings from the models but we do not yet have sufficient literature to robustly assess the magnitude and exact location of the expected changes driven by irrigation.

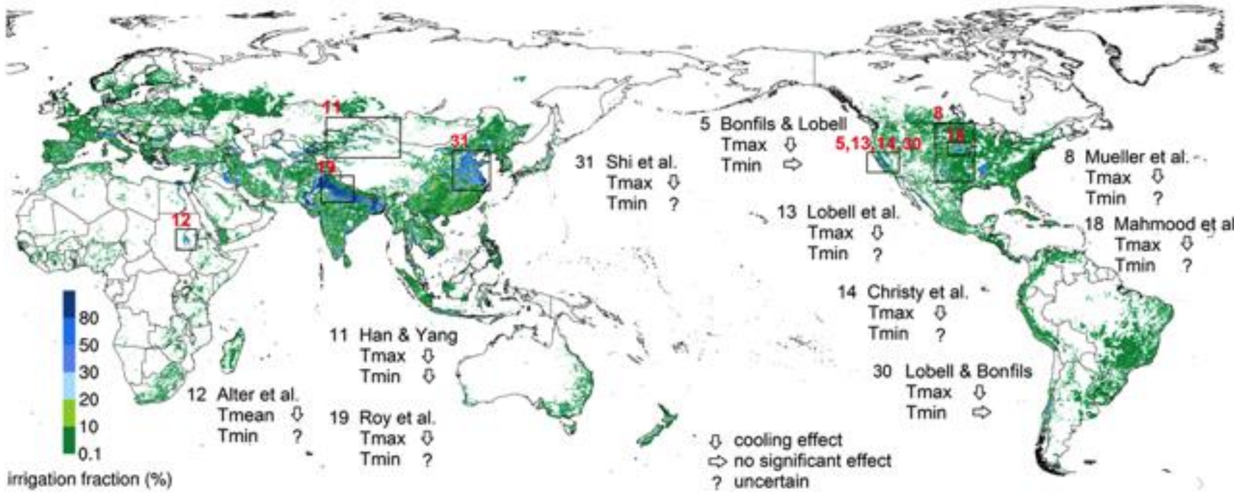

Figure 2.18 Global map of areas equipped for irrigation (colours), expressed as a percentage of total area, or irrigation fraction (Siebert et al. 2013). Numbered boxes show regions where irrigation causes cooling (down arrow) of surface mean (Tmean), maximum (Tmax) or minimum (Tmin) temperature, or else no significant effect (right arrow) or where the effect is uncertain (question mark), based on observational studies as reviewed in (Chen and Jeong 2018). Tmax refers to the warmest daily temperature while Tmin to the coldest one which generally occurs at night. References are (Alter et al. 2015; Han and Yang 2013; Roy et al. 2007; Shi et al. 2013; Bonfils and Lobell 2007; Lobell et al. 2008; Lobell and Bonfils 2008; Christy et al. 2006; Mahmood et al. 2006; Mueller et al. 2015)

2.5.2.2.2 Cropland albedo

Various methods have been proposed to increase surface albedo in cropland and thus reduce locally surface temperature (*high confidence*): choose ‘brighter’ crop varieties (Ridgwell et al. 2009; Crook et al. 2015; Hirsch et al. 2017; Singarayer et al. 2009; Singarayer and Davies-Barnard 2012), abandon tillage (Lobell et al. 2006; Davin et al. 2014), include cover crops into the rotation in areas where soils are darker than vegetation (Carrer et al. 2018; Kaye and Quemada 2017) or use greenhouses (as in (Campra et al. 2008), see (Seneviratne et al. 2018) for a review).

Whatever the solution chosen, the induced reduction in absorbed solar radiation cools the land, more specifically during the hottest summer days ((Davin et al. 2014; Wilhelm et al. 2015); *low confidence*) (Figure 2.19). Changes in temperature are essentially local and seasonal (limited to crop growth season) or sub-seasonal (when resulting from inclusion of cover crop or tillage suppression). Such management action on incoming solar radiation thus holds the potential to counteract warming in cultivated areas during crop growing season.

Introducing cover crops into a rotation can also have a warming effect in areas where vegetation has a darker albedo than soil, or in winter during snow periods if the cover crops or their residues are tall enough to overtop the snow cover (Kaye and Quemada 2017; Lombardozzi et al. 2018). In addition evapotranspiration greater than that of bare soil during this transitional period reduces soil temperature (Ceschia et al. 2017). Such management strategy can have another substantial mitigation effect as it allows to store carbon in the soil and to reduce both direct and indirect N_2O emissions (Basche et al. 2014; Kaye and Quemada 2017), in particular if fertilisation of the subsequent crop is reduced (Constantin et al. 2010, 2011). The use of cover crops thus improves substantially the GHG budget of croplands (Kaye and Quemada 2017; Tribouillois et al. 2018). More discussion on the role of the management practices for mitigation can be found in section 2.6 and chapter 6.

Only a handful of modelling studies have looked at effects other than changes in atmospheric temperature in response to increased cropland albedo. (Seneviratne et al. 2018) have found significant changes in rainfall following an idealised increase in cropland albedo, especially within the Asian monsoon regions. The benefits of cooler temperature on production, resulting from increased albedo, is cancelled by decreases in rainfall that are harmful for crop productivity. The rarity of a concomitant evaluation of albedo management impact on crop productivity prevents us from providing a robust assessment of this practice in terms of both climate mitigation and food security.

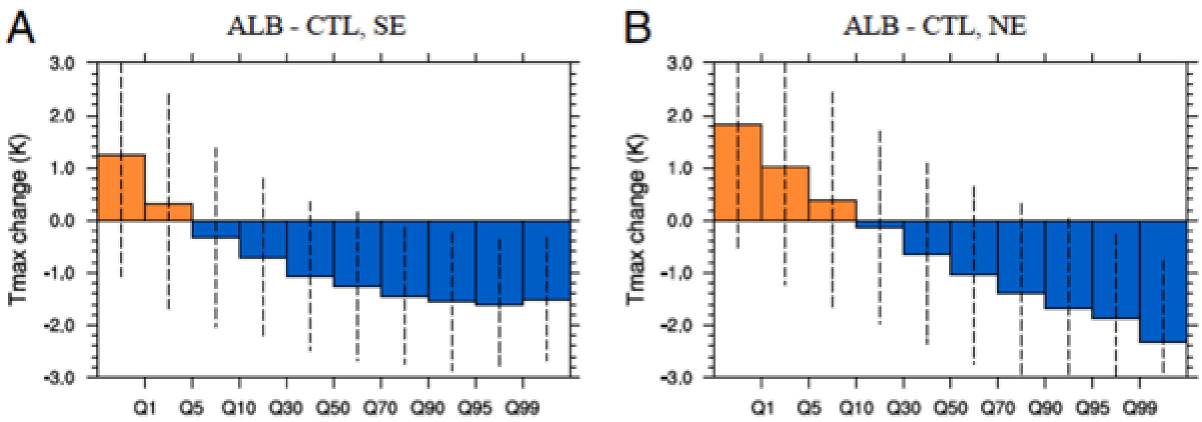

Figure 2.19 Change in summer (July-August) daily maximum temperature (°C) resulting from increased surface albedo in unploughed versus ploughed land, in (A) Southern and (B) Northern Europe, during the period 1986–2009. Changes are simulated for different quantiles of the daily maximum temperature distribution, where Q1 represents the coolest 1% and Q99 the warmest 1% of summer days. Only grid cells with more than 60% of their area in cropland are included. The dashed bars represent the standard deviation calculated across all days and grid points. SE refers to southern Europe (below 45°N) and NE to northern Europe (above 45°N)

2.5.3 Amplifying / dampening climate changes via land responses

Section 2.1 and Box 2.1: illustrates the various mechanisms through which land can affect the atmosphere and thereby climate and weather. Section 2.2 illustrates the many impacts climate changes have on the functioning of land ecosystems. Section 2.3 discusses the effects future climatic conditions on the capacity of the land to absorb anthropogenic CO_2, which then controls the sign of the feedback to the initial global warming. Sections 2.5.1 and 2.5.2 show effects of changes in anthropogenic land cover or land management on climate variables or processes. Land has thus the potential to dampen or amplify the GHG-induced global climate warming or can be used as a tool to mitigate regional climatic consequences of global warming such as extreme weather events, in addition to increasing the capacity of land to absorb CO_2 (Figure 2.20).
Land-to-climate feedbacks are difficult to assess with global or regional climate models as both types of models generally omit a large number of processes. Among these are 1) the response of vegetation to climate change in terms of growth, productivity, and geographical distribution, 2) the dynamics of major disturbances such as fires, 3) the nutrients dynamics, and 4) the dynamics and effects of short-lived chemical tracers such as biogenic volatile organic compounds (Section 2.4). Therefore, only those processes that are fully accounted for in climate models are considered here.

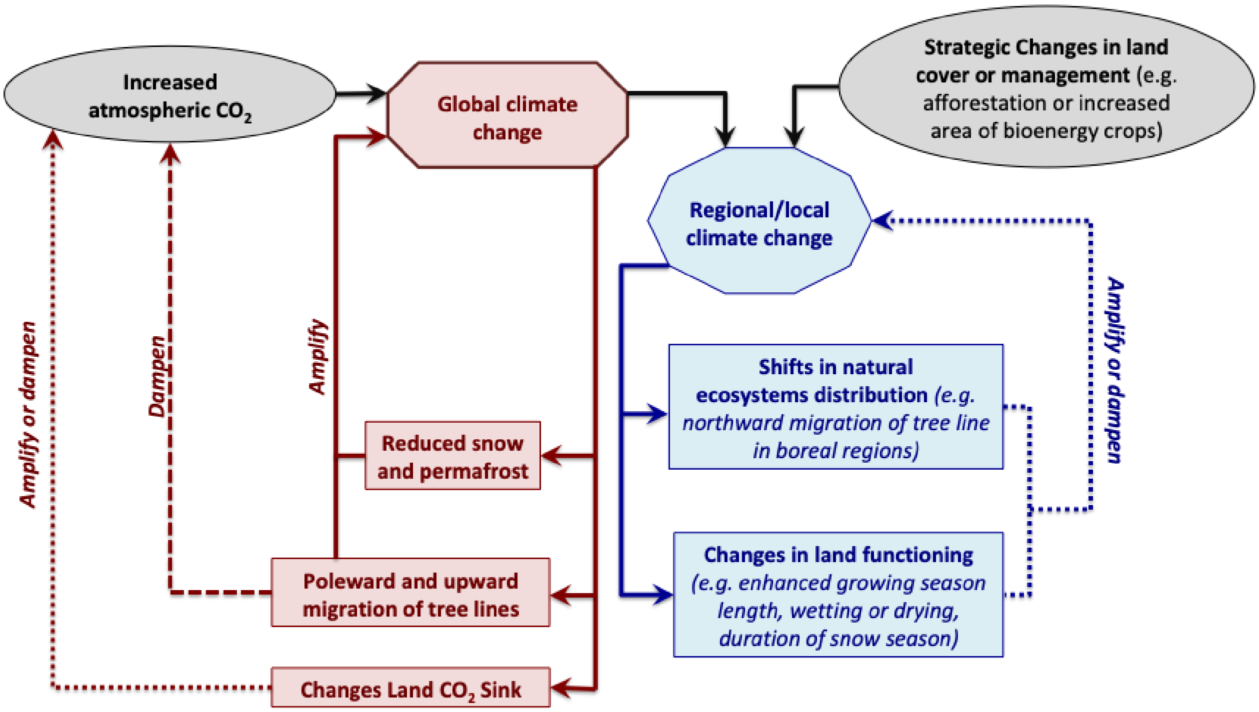

Figure 2.20 Schematics of the various ways land has been shown, in the literature, to either amplify or dampen the initial GHG-induced climatic change, at the global scale (left panel and red boxes and arrows) or at the regional/local levels (right panel and blue boxes and arrows). Grey arrows and boxes refer to what we consider herein as imposed changes, that is the initial atmospheric GHG content as well as anthropogenic land cover change and land management. Dampening feedbacks are represented with dashed lines, amplifying ones with solid lines and the feedbacks for which the direction may be variable are represented using dotted lines. The feedbacks initiated by changes in snow and permafrost areas in boreal regions is discussed in Section 2.5.3.2, the ones initiated by changes in ecosystem distribution are discussed in Sections 2.5.3.1, 2.5.1 and 2.5.2, and the feedbacks related to changes in the land functioning are discussed in Sections 2.5.3.3, 2.5.1, as well as 2.3 and 2.5 (for changes in net CO_2 fluxes). References supporting this figure can be found in each of those sections.

2.5.3.1 Effects of changes in land cover and productivity resulting from global warming

In boreal regions, the combined northward migration of the treeline and increased growing season length in response to increased temperatures in those regions (see Section 2.2) will have positive feedbacks both on global and regional annual warming (*high confidence*; Garnaud and Sushama 2015; Jeong et al. 2014a; O'ishi and Abe-Ouchi 2009; Port et al. 2012; Strengers et al. 2010). The warming resulting from the decreased surface albedo remains the dominant signal in all modelling studies at the annual time scale and during the snow season, while cooling is obtained during the growing season (see Section 2.5.2.1; Figure 2.21 right panel).

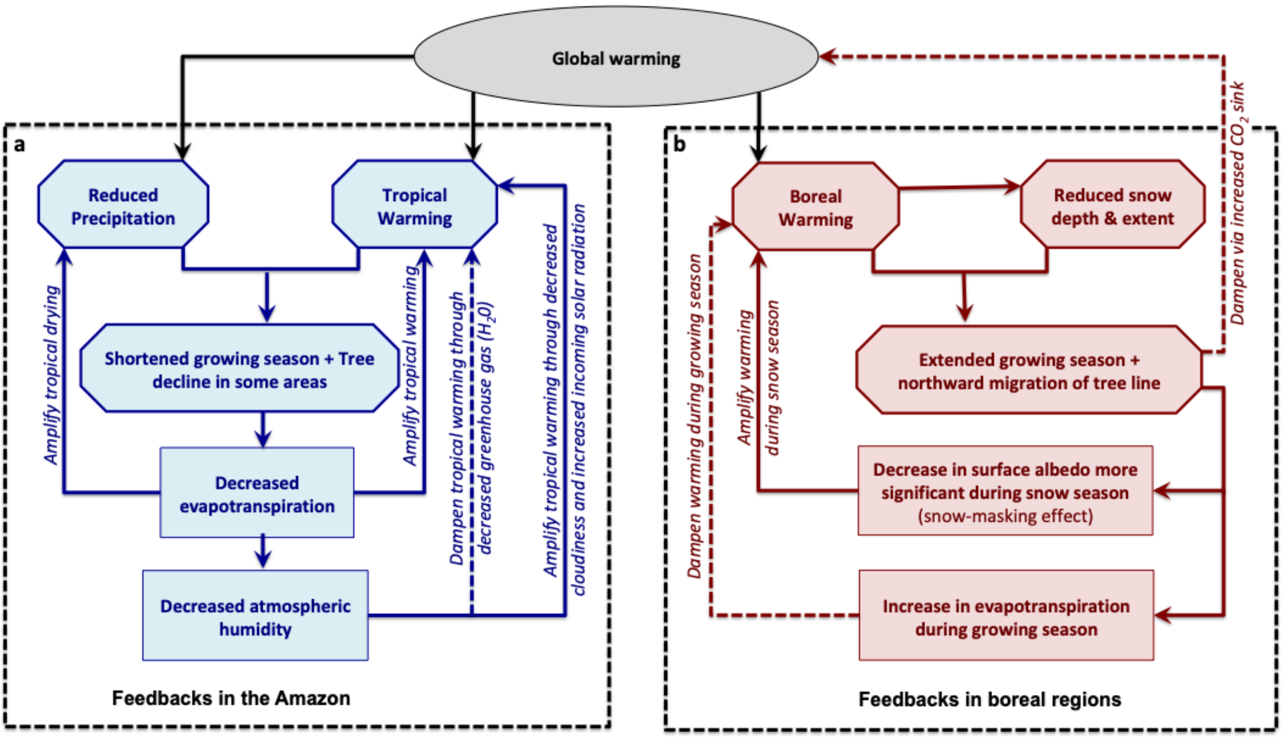

Figure 2.21 Schematic illustration of the processes through which the effects of global warming in the a) Amazon (left panel, blue boxes and arrows) and b) boreal regions (right panel, red boxes and arrows) feedback on the regional climate change. In boreal regions the sign of the feedbacks depends on the season, although annually global warming is further enhanced in those regions. Dashed lines illustrate negative feedbacks while solid lines indicate positive feedbacks. References supporting this figure can be found in the text.

In the tropics climate change will cause both greening and browning (see Section 2.2). Where global warming provokes decrease in rainfall, the induced decrease in biomass production leads to increased local warming (Port et al. 2012; Wu et al. 2016; Yu et al. 2016; *high confidence*). The reverse is true where warming generates increases in rainfall and thus greening. As an example, (Port et al. 2012) simulated decreases in tree cover and shortened growing season in the Amazon, despite the CO_2 fertilisation effects, in response to both future tropical warming and reduced precipitation (Figure 2.21 , left panel). This browning of the land decreases both evapotranspiration and atmospheric humidity. The warming driven by the drop in evapotranspiration is enhanced via decreases in cloudiness that increases incoming solar radiation, and is dampened by reduced water vapour greenhouse radiation.

There is *very low confidence* on how feedbacks affect rainfall in the tropics where vegetation changes may occur, as the sign of the change in precipitation depends on where the greening occurs and on the season (as discussed in Section 2.5.2). There is however *high confidence* that increased vegetation growth in the southern Sahel increases African monsoon rains (Yu et al. 2016; Port et al. 2012; Wu et al. 2016). Confidence on the direction of such feedbacks is also based on a significant number of paleoclimate studies that analysed how vegetation dynamics helped maintain a northward position of the African monsoon during the Holocene time period (9 to 6 kyr BP) (de Noblet-Ducoudré et al. 2000; Rachmayani et al. 2015).

2.5.3.2 Feedbacks to climate from high-latitude land-surface changes

In high latitudes, snow albedo and permafrost carbon feedbacks are the most well-known and most important surface-related climate feedbacks because of their large-scale impacts.

In response to ongoing and projected decrease in seasonal snow cover (Derksen and Brown 2012; Brutel-Vuilmet et al. 2013) warming is and will continue to be enhanced in boreal regions (*high confidence*; Brutel-Vuilmet et al. 2013; Perket et al. 2014; Thackeray and Fletcher 2015; Mudryk et al. 2017). One reason for this is the large reflectivity (albedo) the snow exerts on shortwave radiative forcing: the all-sky global land snow shortwave radiative effect is evaluated to be around -2.5±0.5 W m^{-2} (Flanner et al. 2011; Singh et al. 2015). In the Southern Hemisphere, perennial snow on the Antarctic is the dominant contribution, while in the Northern Hemisphere, this is essentially attributable to seasonal snow with a smaller contribution from snow on glaciated areas. Another reason is the sensitivity of snow cover to temperature: (Mudryk et al. 2017) recently showed that in the high latitudes, climate models tend to correctly represent this sensitivity, while in mid-latitude and alpine regions, the simulated snow cover sensitivity to temperature variations tends to be biased low. In total, the global snow albedo feedback is about 0.1 W m^{-2} K^{-1}, which amounts to about 7% of the strength of the globally dominant water vapour feedback (e.g., (Thackeray and Fletcher 2015)). While climate models do represent this feedback, a persistent spread in the modelled feedback strength has been noticed (Qu and Hall 2014) and, on average, the simulated snow albedo feedback strength tends to be somewhat weaker than in reality (Flanner et al. 2011; Thackeray and Fletcher 2015) (*medium confidence*). Various reasons for the spread and biases of the simulated snow albedo feedback have been identified, notably inadequate representations of vegetation masking of snow in forested areas (Loranty et al. 2014; Wang et al. 2016c; Thackeray and Fletcher 2015).

The second most important potential feedback from land to climate relates to permafrost decay. There is *high confidence* that, following permafrost decay from a warming climate, the resulting emissions of carbon dioxide and/or methane (caused by the decomposition of organic matter in previously frozen soil) will produce additional GHG-induced warming. There is however substantial uncertainty on the magnitude of this feedback, although recent years have seen large progress in its quantification. Lack of agreement results from several critical factors that carry large uncertainties. The most important are a) the size of the permafrost carbon pool, b) its decomposability, c) the magnitude, timing and pathway of future high-latitude climate change and d) the correct identification and model representation of the processes at play (Schuur et al. 2015b). The most recent comprehensive estimates establish a total soil organic carbon storage in permafrost of about 1500 ± 200 Pg C (Hugelius et al. 2014, 2013; Olefeldt et al. 2016), which is about 300 Pg C lower than previous estimates (*low confidence)*. Important progress has been made in recent years at incorporating permafrost-related processes in complex Earth System Models (e.g., (McGuire et al. 2018)), but representations of some critical processes such as thermokarst formation are still in their infancy (Schuur et al. 2015b). Recent model-based estimates of future permafrost carbon release (Koven et al. 2015; McGuire et al. 2018) have converged on an important insight. Their results suggest that substantial net carbon release of the coupled vegetation-permafrost system will probably not occur before about 2100 because carbon uptake by increased vegetation growth will initially compensate for GHG releases from permafrost (*limited evidence, high agreement*).

2.5.3.3 Feedbacks related to changes in soil moisture resulting from global warming

There is *medium evidence* but *high agreement* that soil moisture conditions influence the frequency and magnitude of extremes such as drought and heat waves. Observational evidence indicates that dry soil moisture conditions favour heat-waves, in particular in regions where evapotranspiration is limited by moisture availability (Mueller and Seneviratne 2012; Quesada et al. 2012; Miralles et al. 2018; Geirinhas et al. 2018; Miralles et al. 2014; Chiang et al. 2018; Dong and Crow 2019; Hirschi et al. 2014).

In future climate projections, soil moisture plays an important role in the projected amplification of extreme heat-waves and drought in many regions of the world (*medium confidence*; (Seneviratne et al. 2013; Vogel et al. 2017; Donat et al. 2018; Miralles et al. 2018)). In addition, the areas where soil moisture affects heat extremes will not be located exactly where they are today. Changes in rainfall, temperature and thus evapotranspiration will induce changes in soil moisture and therefore of where temperature and latent heat flux will be negatively coupled (Seneviratne et al. 2006; Fischer et al. 2012). Quantitative estimates of the actual role of soil moisture feedbacks are however very uncertain due to the *low confidence* in projected soil moisture changes (IPCC 2013a), to weaknesses in the representation of soil moisture-atmosphere interactions in climate models (Sippel et al. 2017; Ukkola et al. 2018; Donat et al. 2018; Miralles et al. 2018) and to methodological uncertainties associated with the soil moisture prescription framework commonly used to disentangle the effect of soil moisture on changes in temperature extremes (Hauser et al. 2017).

Where soil moisture is predicted to decrease in response to climate change in the subtropics and temperate latitudes, this drying could be enhanced by the existence of soil moisture feedbacks (*low confidence* (Berg et al. 2016)). The initial decrease in precipitation and increase in potential evapotranspiration and latent heat flux, in response to global climate change, leads to decreased soil moisture at those latitudes and can potentially amplify both. Such a feature is consistent with evidence that in a warmer climate land and atmosphere will be more strongly coupled via both the water and the energy cycles (Dirmeyer et al. 2014;

Guo et al. 2006). This increased sensitivity of atmospheric response to land perturbations implies that changes in land uses and cover are expected, in the future, to have more impact on climate in the future than they do today.

Beyond temperature, it has been suggested that soil moisture feedbacks influence precipitation occurrence and intensity. But the importance and even the sign of this feedback is still largely uncertain and debated (Tuttle and Salvucci 2016; Yang et al. 2018; Froidevaux et al. 2014; Guillod et al. 2015).

2.5.4 Non-local and downwind effects resulting from changes in land cover

Changes in land cover or land management do not just have local consequences but also affect adjacent or more remote areas. Those non-local impacts may occur in three different ways.

(1) Any action on land that affects photosynthesis and respiration has an impact on the atmospheric CO_2 content as this GHG is well mixed in the atmosphere. This change in turn affects the downwelling long-wave radiation everywhere on the planet and contributes to global climate change. This is more thoroughly discussed in Section 2.6 where various land-based mitigation solutions are examined. Local land use changes thus have the potential to affect global climate via changes in atmospheric CO_2.

(2) Any change in land cover or land management may impact local surface air temperature and moisture and thus sea-level pressure. Thermal, moisture and surface pressure gradients between the area of change and neighbouring areas are then modified and affect the amount of heat, water vapour and pollutants flowing out (downwind) of the area (e.g. Ma et al. 2013b; McLeod et al. 2017; Abiodun et al. 2012; Keys 2012).
Forests for example provide water vapour to the atmosphere which supports terrestrial precipitation downwind (Ellison et al. 2017a; Layton and Ellison 2016; Spracklen et al. 2012, 2018). Within a few days water vapour can travel several hundreds of kilometres before being condensed into rain and potentially being transpired again (Makarieva et al. 2009). This cascading moisture recycling (succession of evapotranspiration, water vapour transport and condensation-rainfall) has been observed in south America (Spracklen et al. 2018; Zemp et al. 2014; Staal et al. 2018; Spracklen et al. 2012). Deforestation can thus potentially decrease rainfall downwind, while combining 'small-scale' forestation and irrigation in the semi-arid region is susceptible to boost the precipitation-recycling mechanism with better vegetation growth downwind (Figure 2.22; (Ellison et al. 2017a; Layton and Ellison 2016)).

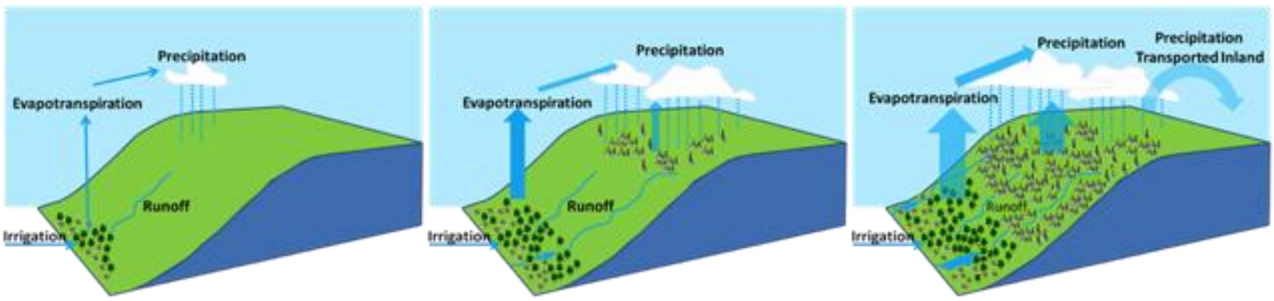

Figure 2.22 Schematic illustration of how combined forestation and irrigation can influence downwind precipitation on mountainous areas (here in Los Angeles, California area), favour vegetation growth and feeds back to the forested area via increased runoff (Layton and Ellison 2016). Areas of forests plantation and irrigation are located on the left panel, whereas consequent downwind effects and feedbacks are illustrated in the middle and right panels.

(3) Many studies using global climate models have reported that the climatic changes resulting from changes in land are not limited to the lower part of the atmosphere but can reach the upper levels via changes in large scale ascent (convection) or descent (subsidence) of air. This coupling to the upper atmosphere triggers perturbations in large-scale atmospheric transport (of heat, energy and water) and subsequent changes in temperature and rainfall in regions located quite far away from the original perturbation (**Figure 2.23**, Laguë and Swann 2016; Feddema et al. 2005, badger & dirmeyer 2016, Garcia 2016, Stark 2015, Devaraju 2018, Quesada et al. (2017a)).

De Vrese et al. (2016) for example, using a global climate model, found that irrigation in India could affect regions as remote as eastern Africa through changes in the atmospheric transport of water vapour. At the onset of boreal spring (February to March) evapotranspiration is already large over irrigated crops and the resulting excess moisture in the atmosphere is transported south-westward by the low-level winds. This results in increases in precipitation as large as 1mm d^{-1} in the horn of Africa. Such finding implies that if irrigation is to decrease in India, rainfall can decrease in eastern Africa where the consequences of drought are already disastrous.

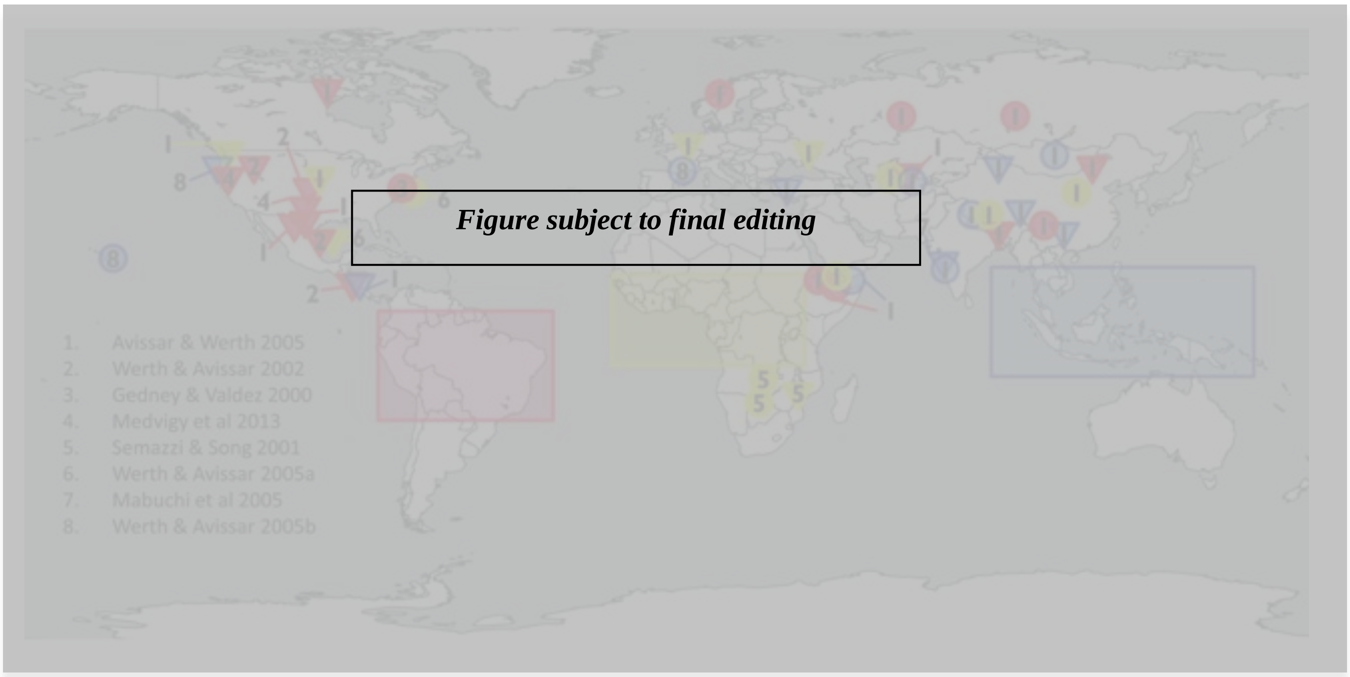

Figure 2.23 Extra-tropical effects on precipitation due to deforestation in each of the three major tropical regions. Increasing (circles) and decreasing (triangles) precipitation result from complete deforestation of either Amazonia (red), Africa (yellow), or Southeast Asia (blue) as reviewed by (Lawrence and Vandecar 2015). Boxes indicate the area in which tropical forest was removed in each region. Numbers refer to the study from which the data were derived. Cited papers are the following (Avissar and Werth 2005; Gedney and Valdes 2000; Semazzi and Song 2001; Werth 2002; Mabuchi et al. 2005; Werth 2005)

Changes in sea-surface temperature have also been simulated in response to large-scale vegetation changes (Cowling et al. 2009; Davin and de Noblet-Ducoudre 2010; Wang et al. 2014b, Notaro Liu 2007).
Most of those modelling studies have been carried out with land cover changes that are extremely large and often exaggerated with respect to reality. The existence of such teleconnections can thus be biased as discussed in Lorenz et al. (2016).

In conclusion, there is *high confidence* that any action on land (for example to dampen global warming effects), wherever they occur, will not only have effects on local climate but also generate atmospheric changes in neighbouring regions, and potentially as far as few hundreds of kilometres downwind. More remote teleconnections, thousands of kilometres away from the initial perturbation, are impossible to observe and have only been reported by modelling studies using extreme land cover changes. There is *very low confidence* that detectable changes due to such long-range processes can occur.

Cross-Chapte Box 4: Climate Change and Urbanisation

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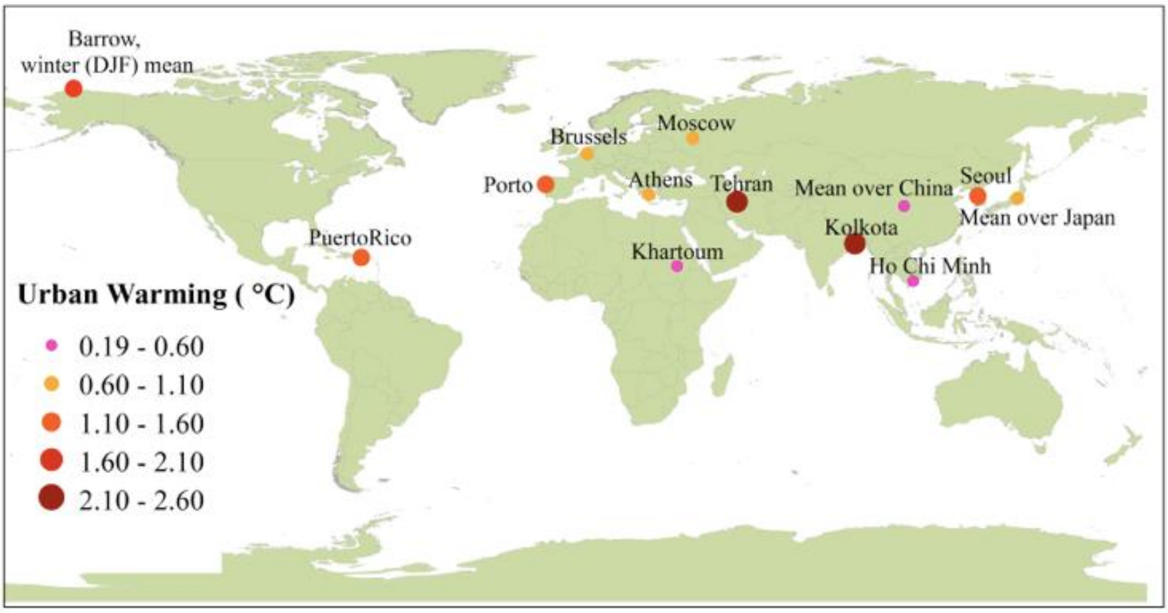
Cities extent, population, and expected growth

Despite only covering 0.4-0.9% of the global land surface (Esch et al. 2017; Zhou et al. 2015), over half the world's population live in towns and cities (United Nations 2017) generating around three-quarters of the global total carbon emissions from energy use (Creutzig et al. 2015b; Intergovernmental Panel on Climate Change 2014). Urban food consumption is a large source of these anthropogenic greenhouse gas emissions (Goldstein et al. 2017). In developed countries, per capita emissions are larger in small cities than bigger ones, while the opposite is found in developing countries (Gudipudi et al. 2019). Climate change is expected to increase the energy demand of people living in urban areas (Santamouris et al. 2015; Wenz et al. 2017).

In addition to being a driver of emissions, urbanisation contributes to forest degradation, converts neighbouring agricultural, forested, or otherwise undeveloped land to urban use, altering natural or semi-natural ecosystems both within and outside of urban areas (Du and Huang 2017). It has been identified as a major driver of land degradation as illustrated in Chapters 3, 4 and 5. Highly productive lands are experiencing the highest rate of conversion to urbanised landscapes (Nizeyimana et al. 2001; Pandey et al. 2018), affecting food security. Loss of agricultural land, and increased pollution and waste are some of key challenges arising from urbanisation and urban growth (Chen 2007). The proportion of urban population is predicted to reach ~70% by the middle of the century (United Nations 2017) with growth especially taking place in the developing world (Angel et al. 2011; Dahiya 2012). Urban sprawl is projected to consume 1.8–2.4% and 5% of the current cultivated land by 2030 and 2050 respectively (Pradhan et al. 2014; Bren d'Amour et al. 2016) driven by both general population increase and migration from rural areas (Adger et al. 2015; Seto et al. 2011; Geddes et al. 2012). New city dwellers in developing countries will require land for housing to be converted from non-urban to urban land (Barbero-Sierra et al. 2013), indicating future degradation. These growing urban areas will experience direct and indirect climate change impacts, such as sea level rise and storm surges (Boettle et al. 2016; Revi et al. 2014), increasing soil salinity, and landslides from precipitation extremes. Furthermore, poorly planned urbanisation can increase people's risk to climate hazards as informal settlements and poorly built infrastructure are often the most exposed to hazards from fire, flooding, and landslides (Adger et al. 2015; Geddes et al. 2012; Revi et al. 2014). Currently, avoiding land degradation and maintaining/enhancing ecosystem services are rarely considered in planning processes (Kuang et al. 2017).

Climate change, urban heat island and threats specific to urban populations

Cities alter the local atmospheric conditions as well as those of the surrounding areas (Wang et al. 2016b; Zhong et al. 2017). There is *high confidence* that urbanisation increases mean annual surface air temperature in cities and in their surroundings, with increases ranging from 0.19°C to 2.60°C (Cross Chapter Box 4 Figure 1) (Torres-Valcárcel et al. 2015; Li et al. 2018a; Doan et al. 2016). This phenomenon is referred to as the urban heat island (UHI) effect (Oke et al. 2017; Bader et al. 2018). The magnitude and diurnal amplitude of the UHI varies from one city to another and depends on the local background climate (Wienert and Kuttler 2005; Zhao et al. 2014; Ward et al. 2016). There is nevertheless *high confidence* that urbanisation affects night time temperatures more substantially than daytime ones (Argüeso et al. 2014; Alghamdi and Moore 2015; Alizadeh-Choobari et al. 2016; Fujibe 2009; Hausfather et al. 2013; Liao et al. 2017; Sachindra et al. 2016; Camilloni and Barrucand 2012; Wang et al. 2017a; Hamdi 2010; Arsiso et al. 2018; Elagib 2011; Lokoshchenko 2017; Robaa 2013). In addition there is *high confidence* that the UHI effect makes heatwaves more intense in cities by 1.22°C to 4°C, particularly at night (Li and Bou-Zeid 2013; Li et al. 2017b; Hamdi et al. 2016; Founda and Santamouris 2017; Wang et al. 2017a). As there is a well-established relationship between extremely high temperatures and morbidity, mortality (Watts et al. 2015) and labour productivity (Costa et al. 2016), expected increase in extreme heat events with future climate change will worsen the conditions in cities.



Cross Chapter Box 4, Figure 1: Change in annual mean surface air temperature resulting from urbanisation (°C). Colour and size of the circles refer to the magnitude of the change. This map has been compiled using the following studies: (Kim et al. 2016; Sun et al. 2016; Chen et al. 2016a; Founda et al. 2015; Rafael et al. 2017; Hinkel and Nelson 2007; Chrysanthou et al. 2014; Dou et al. 2014; Zhou et al. 2016, 2017; Polydoros et al. 2018; Li et al. 2018a; Bader et al. 2018; Alizadeh-Choobari et al. 2016; Fujibe 2009; Lokoshchenko 2017; Torres-Valcárcel et al. 2015; Doan et al. 2016; Elagib 2011; Liao et al. 2017).

Individual city case studies show that precipitation mean and extremes are increased over and downwind of urban areas, especially in the afternoon and early evening when convective rise of the atmosphere is the strongest (*medium confidence)*. The case studies covered: different inland and coastal US cities (M. et al. 2014; McLeod et al. 2017; Ganeshan and Murtugudde 2015); Dutch coastal cities (Daniels et al. 2016); Hamburg (Schlünzen et al. 2010); Shanghai (Liang and Ding 2017); Beijing (Dou et al. 2014); and Jakarta and Kuala Lumpur (Lorenz et al. 2016). Increased aerosol concentrations however can interrupt the precipitation formation process and thereby reduce heavy rainfall (Daniels et al. 2016; Zhong et al. 2017). Urban areas also experience altered water cycle in other aspects, the evaporative demand for plants in cities are increased by as much as 10% (Zipper et al. 2017) while high proportion of paving in cities mean that surface runoff of water is high (Hamdi et al. 2011; Pataki et al. 2011). In addition, water retention is lower in degraded, sealed soils beneath urban surfaces compared to intact soils. Increased surface water runoff, especially when and where the rainfall intensity is likely to intensify (IPCC 2013b), leads to a greater likelihood of flooding in urban areas without implementation of adaptation measures (Shade and Kremer 2019; Wang et al. 2013; Environmental Protection Agency 2015).

Urbanisation alters the stock size of soil organic carbon (SOC) and its stability. The conversion of vegetated land to urban land results in a loss of carbon stored in plants, while stresses associated with the urban environment (e.g., heat, limited water availability and pollution) reduce plant growth and survival in cities (Xu et al. 2016b). Overall, carbon densities or stocks decrease from natural land areas to the urban core along the rural-urban gradient (Tao et al. 2015; Zhang et al. 2015). For example the Seoul Forest Park, an urban park, shows a tenfold difference in SOC stocks across its land cover types (Bae and Ryu 2015). In Changchun in Northeast China, however, SOC density is higher in recreational forests within urban areas compared to a production forest (Zhang et al. 2015).

Urban air pollution as an environmental risk increases with climate change. Increased air temperatures can lead to reduced air quality by enhancing the formation of photochemical oxidants and increasing the concentration of air pollutants such as ozone, with corresponding threats to human health (Sharma et al. 2013). The occurrence of bronchial asthma and allergic respiratory diseases is increasing worldwide, and urban residents are experiencing poor air quality conditions more frequently than rural residents (D'Amato et al. 2010). Excess morbidity and mortality related to extremely poor air quality are found in many cities worldwide (Harlan and Ruddell 2011). Some emissions that lead to reduced air quality are also contributors to climate change (Shindell et al. 2018; de Coninck et al. 2018).

Urban response options for climate change, desertification, land degradation and food security

Urban green infrastructure (UGI; see glossary) has been proposed as a solution to mitigate climate change directly through carbon sequestration (Davies et al. 2011; Edmondson et al. 2014). However, compared to overall carbon emissions from cities, its mitigation effects are likely to be small (*medium confidence*). UGI nevertheless has an important role in adapting cities to climate change (Demuzere et al. 2014; Sussams et al. 2015; Martin et al. 2016; Gill et al. 2007; Revi et al. 2014). Adaptation through UGIs is achieved through, for example: (i) reduction in air temperature (Cavan et al. 2014; Di Leo et al. 2016; Feyisa et al. 2014; Zölch et al. 2016; Li et al. 2019) which can help improve human health and comfort (e.g. (Brown and Nicholls 2015; Klemm et al. 2015)); (ii) reduction in the energy demands of buildings through the use of green roofs and walls (e.g.(Coma et al. 2017)); (iii) reduction in surface water runoff and flood risk (Zeleňáková et al. 2017). Given that UGI necessarily involves the retention and management of non-sealed surfaces, co-benefits for land degradation will also be apparent (Murata and Kawai 2018; Scalenghe and Marsan 2009) (*limited evidence, high agreement*).

Urban agriculture is one aspect of UGI that has the potential to both meet some of the food needs of cities and reduce land degradation pressures in rural areas (*low confidence*; e.g. Wilhelm and Smith 2018). Urban agriculture has many forms, such as backyard gardening, allotments, plants on roof-tops or balconies, urban-fringe/peri-urban agriculture, hydroponics, aquaponics, livestock grazing in open spaces and vertical farming (Gerster-Bentaya 2013) (see also Section 5.6.5).

Consuming locally produced food and enhancing the efficiency of food processing and transportation can minimise food losses, contribute to food security, and in some circumstances reduce GHG emissions (Brodt et al. 2013; Michalský and Hooda 2015; Tobarra et al. 2018) (see also Section 5.5.2.3). Furthermore, urban agriculture has the potential to counteract the separation of urban populations from food production. This separation is one driver of the transition towards more homogeneous, high protein diets, which are associated with increased greenhouse gas emissions (Goldstein et al. 2017; Moragues-Faus and Marceau 2018; Magarini and Calori 2015). Barriers to the uptake of urban agriculture as a climate change mitigation option include the need for efficient distribution systems to ensure lowered carbon emissions (Newman et al. 2012) and the concern that urban agriculture may harbour pathogenic diseases, or that its products be contaminated by soil or air pollution (Hamilton et al. 2014; Ercilla-Montserrat et al. 2018).

In summary

Climate change is already affecting the health and energy demand of large numbers of people living in urban areas (*high confidence*; see also Section 2.2). Future changes to both climate and urbanisation will enhance warming in cities and their surroundings, especially during heat waves (*high confidence*). Urban and peri-urban agriculture, and more generally the implementation of urban green infrastructure, can contribute to climate change mitigation (*medium confidence*) as well as to adaptation (*high confidence*), including co-benefits for food security and reduced soil-water-air pollution.

2.6 Climate consequences of response options

Response options can affect climate mitigation and adaptation simultaneously, therefore this Special Report on Climate Change and Land (SRCCL) discusses land-based response options in an integrated way (Chapter 1). In this chapter we assess response options that that have an effect on climate. A description of the full set of response options across the SRCCL can be found in Chapter 6, including the interplay between mitigation, adaptation, desertification, land degradation, food security and other co-benefits and trade-offs. Response options specific to desertification, degradation and food security are described in more detail in Chapters 3, 4 and 5.

Some response options lead to land use change and can compete with other land uses, including other response options, while others may free-up land that can be used for further mitigation/adaptation by reducing demand for land or products e.g. agricultural intensification, diet shifts, and reduction of waste *(high confidence).*

Some response options result in a net removal of GHGs from the atmosphere and storage in living or dead organic material, or in geological stores (IPCC SR15). Such options are frequently referred to in the literature as carbon dioxide removal (CDR), Greenhouse Gas Removal (GGR) or negative emissions technologies (NETs). CDR options are assessed alongside emissions reduction options. Although they have a land footprint, solar and wind farms are not are not assessed here as they affect greenhouse gas flux in the energy industrial sectors with minimal effect in the land sector, but the impact of solar farms on agricultural land competition is dealt with in Chapter 7.

A number of different types of scenario approach exist for estimating climate contribution of land-based response options (see Cross-Chapter Box 1: Scenarios, Chapter 1). Mitigation potentials have been estimated for single and sometimes multiple response options using stylised "bottom-up" scenarios. Response options are not mutually exclusive (e.g., management of soil carbon and cropland management). Different options interact with each other; they may have additive effects or compete with each other for land or other resources, thus these potentials cannot necessarily be added up. The interplay between different land-based mitigation options, as well as with mitigation options in other sectors (such as energy or transport), in contributing to specific mitigation pathways has been assessed using Integrated Assessment Models, see Section 2.7.2. These include interactions with wider socioeconomic conditions (see Cross-Chapter Box 1: Scenarios, Chapter 1) and other sustainability goals (see chapter 6).

2.6.1 Climate impacts of individual response options

Since AR5, there have been many new estimates of the climate impacts of single or multiple response options, summarised in Figure 2.24 and discussed in sub-sections below. Recently published syntheses of mitigation potential of land-based response options (e.g. Hawken 2017a; Smith et al. 2016b; Griscom et al. 2017a; Minx et al. 2018; Fuss et al. 2018b; Nemet et al. 2018) are also included in Figure 2.24. The wide range in mitigation estimates reflects differences in methodologies that may not be directly comparable, and estimates cannot be necessarily be added if they were calculated independently as they may be competing for land and other resources.

Some studies assess a "technical mitigation potential" - the amount possible with current technologies. Some include resource constraints (e.g., limits to yields, limits to natural forest conversion) to assess a "sustainable potential". Some assess an "economic potential" mitigation at different carbon prices. Few include social and political constraints (e.g. behaviour change, enabling conditions, see Chapter 7), the biophysical climate effects (Section 2.5), or the impacts of future climate change (Section 2.3). Carbon stored in biomass and soils may be at risk of future climate change (see Section 2.2), natural disturbances such as wildfire (see Cross-Chapter Box 3: Fire and Climate Change, in this chapter) and future changes in land use or management changes that result in a net loss of carbon (Gren and Aklilu 2016).

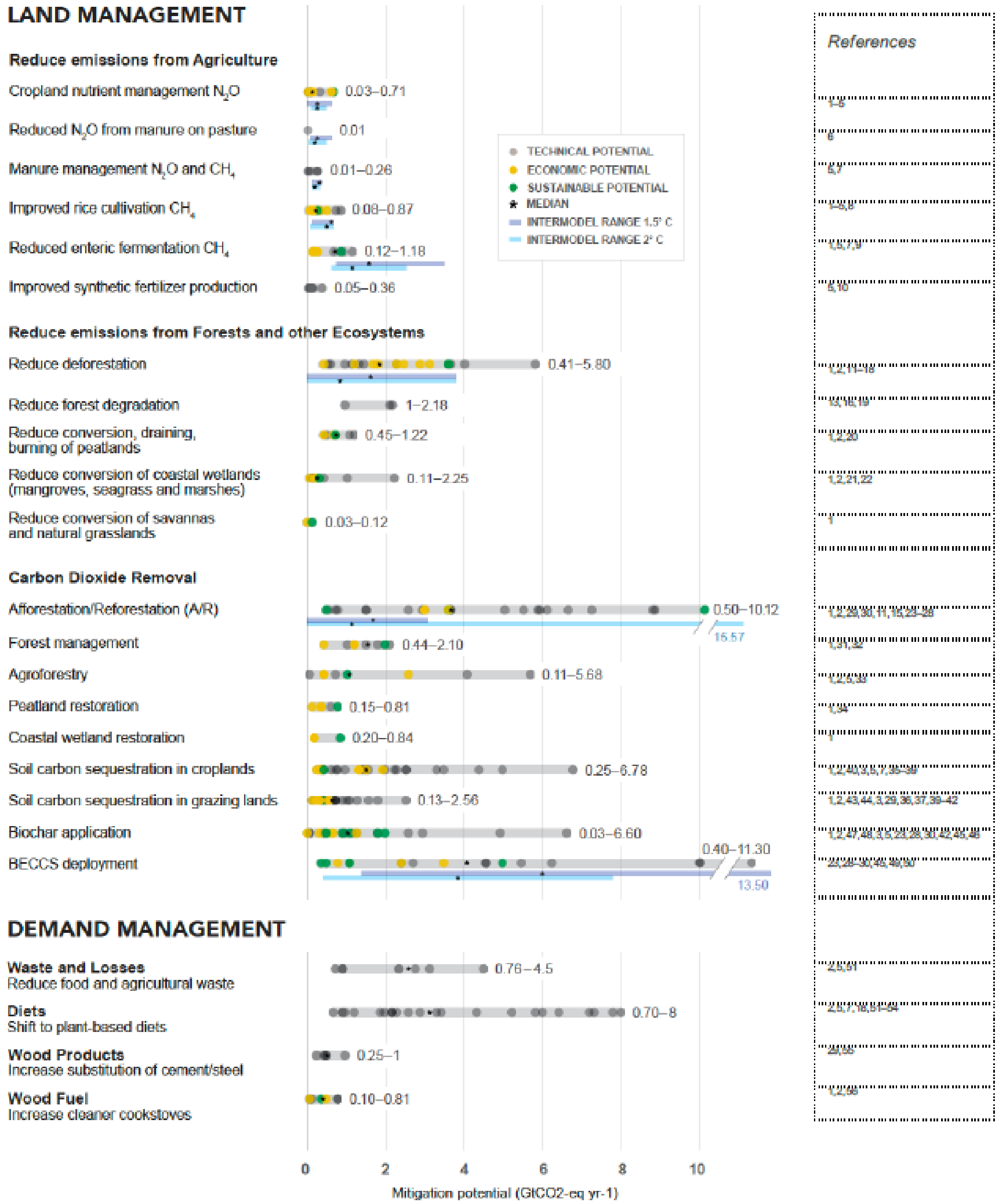


Figure 2.24 Mitigation potential of response options in 2020-2050, measured in GtCO$_2$-eq yr^{-1}, adapted from Roe et al. (2017). Mitigation potentials reflect the full range of low to high estimates from studies published after 2010, differentiated according to technical (possible with current technologies), economic (possible given economic constraints) and sustainable potential (technical or economic potential constrained by sustainability considerations). Medians are calculated across all potentials in categories with >4 data points. Only includes references that explicitly provide mitigation potential estimates in CO$_2$-eq yr^{-1} (or similar derivative) by 2050. Not all options land management potentials are additive as some may compete for land. Estimates reflect a range of methodologies (including definitions, global warming potentials and time horizons) that may not be directly comparable or additive. Results from Integrated Assessment Models are shown to compare with single option "bottom-up" estimates, in available categories from the 2°C and 1.5°C scenarios in the SSP Database (Version 2.0). The models reflect land management changes, yet in some instances, can also reflect demand-side effects from carbon prices, so may not be defined exclusively as "supply-side". References: 1. Griscom et al. (2017); 2. Hawken (2017); 3. Paustian et al. (2016); 4. Beach et al. (2016); 5. Dickie et al. (2014); 6. Herrero et al. (2013); 7. Herrero et al. (2016); 8. Hussain et al. (2015); 9. Hristov, et al. (2013); 10. Zhang et al. (2013); 11. Houghton &

Nassikas (2018); 12. Busch & Engelmann (2017); 13. Baccini et al. (2017); 14. Zarin et al. (2016); 15. Houghton, et al. (2015); 16. Federici et al. (2015); 17. Carter et al. (2015); 18. Smith et al. (2013); 19. Pearson et al. (2017); 20. Hooijer et al. (2010); 21. Howard (2017); 22. Pendleton et al. (2012); 23. Fuss et al. (2018); 24. Dooley & Kartha (2018); 25. Kreidenweis et al. (2016); 26. Yan et al. (2017); 27. Sonntag et al. (2016); 28. Lenton (2014); 29. McLaren (2012); 30. Lenton (2010); 31. Sasaki et al. (2016); 32. Sasaki et al. (2012); 33. Zomer et al. (2016); 34. Couwenberg et al. (2010); 35. Conant et al. (2017); 36. Sanderman et al. (2017); 37. Frank et al. (2017); 38. Henderson et al. (2015); 39. Sommer & Bossio (2014); 40. Lal (2010); 41. Zomer et al. (2017); 42. Smith et al. (2016); 43. Poeplau & Don (2015); 44. Powlson et al. (2014); 45. Powell & Lenton (2012); 46. Woolf et al. (2010); 47. Roberts et al. (2010); 48. Pratt & Moran (2010); 49. Turner et al. (2018); 50. Koornneef et al. (2012); 51. Bajželj et al. (2014); 52. Springmann et al. (2016); 53. Tilman & Clark (2014); 54. Hedenus et al. (2014); 55. Miner (2010); 56. Bailis et al. (2015)

2.6.1.1 Land management in agriculture

Reducing non-CO_2 emissions from agriculture through cropland nutrient management, enteric fermentation, manure management, rice cultivation and fertiliser production has a total mitigation potential of 0.30–3.38 $GtCO_2$-eq yr^{-1} *(medium confidence)* (combined sub-category measures in **Figure 2.24**, details below) with a further 0.25–6.78 $GtCO_2$-eq yr^{-1} from soil carbon management (Section 2.6.1.3). Other literature that looks at broader categories finds mitigation potential of 1.4-2.3 GtCO2-eq yr-1 from improved cropland management (Smith et al. 2008, 2014; Pradhan et al., 2013); 1.4-1.8 GtCO2-eq yr-1 from improved grazing land management (Conant et al. 2017; Herrero et al. 2016; Smith et al. 2008, 2014) and 0.2-2.4 GtCO2-eq yr-1 from improved livestock management (Smith et al. 2008, 2014; Herrero et al. 2016, FAO 2007). A detailed discussion of the mitigation potential of agricultural response options and their co-benefits are provided in Chapter 5, Section 5.5. and 5.6).

The three main measures to reduce enteric fermentation include improved animal diets (higher quality, more digestible livestock feed), supplements and additives (reduce methane by changing the microbiology of the rumen), and animal management and breeding (improve husbandry practices and genetics) – and applying these measures can mitigate 0.12–1.18 $GtCO_2$-eq yr^{-1} *(medium confidence)* (Hristov et al. 2013; Dickie et al. 2014; Herrero et al. 2016; Griscom et al. 2017). However, these measures may have limitations such as need of crop-based feed (Pradhan et al. 2013) and associated ecological costs and toxicity and animal welfare issues related to food additives (Llonch et al. 2017). Measures to manage manure include anaerobic digestion for energy use, composting as a nutrient source, reducing storage time, and changing livestock diets, and have a potential of 0.01–0.26 $GtCO_2$-eq yr^{-1} (Herrero et al. 2016a; Dickie et al. 2014).

On croplands, there is a mitigation potential of 0.03–0.71 $GtCO_2$-eq yr^{-1} for cropland nutrient management (fertiliser application) *(medium confidence)* (Griscom et al. 2017a; Hawken 2017; Paustian et al. 2016; Dickie et al. 2014; Beach et al. 2015). Reducing emissions from rice production through improved water management (periodic draining of flooded fields to reduce methane emissions from anaerobic decomposition), and straw residue management (apply in dry conditions instead of on flooded fields, avoid burning to reduce methane and nitrous oxide emissions) has the potential to mitigate up to 60% of emissions (Hussain et al. 2015), or 0.08–0.87 $GtCO_2$-eq yr^{-1} *(medium confidence)* (Griscom et al. 2017a; Hawken 2017; Paustian et al. 2016; Hussain et al. 2015; Dickie et al. 2014; Beach et al. 2015). Further, sustainable intensification through the integration of crop and livestock systems can increase productivity, decrease emission intensity and act as a climate adaptation option (see chapter 5.5.1.4).

Agroforestry is a land management system that combines woody biomass (e.g., trees or shrubs) with crops and/or livestock). The mitigation potential from agroforestry ranges between 0.08 to 5.7 $GtCO_2$ yr^1, *(medium confidence)* (Griscom et al. 2017c; Dickie et al. 2014; Zomer et al. 2016; Hawken 2017). The high estimate is from an optimum scenario combing four agroforestry solutions (silvopasture, tree intercropping, multistrata agroforestry and tropical staple trees) of Hawken (2017a). Zomer et al. (2016) reported that the trees in agroforestry landscapes had increased carbon stock by 7.33 $GtCO_2$ between 2000 and 2010, or 0.7 $GtCO_2$ yr^{-1}. For more details see Chapter 5, Section 5.5.1.3.

2.6.1.2 Land management in forests

The mitigation potential for reducing and/or halting deforestation and degradation ranges from 0.4 to 5.8

$GtCO_2$ yr^{-1} *(high confidence)* (Griscom et al. 2017a; Hawken 2017; Busch and Engelmann 2017; Baccini et al. 2017; Zarin et al. 2016b; Federici et al. 2015; Carter et al. 2015; Houghton et al. 2015; Smith et al. 2013a; Houghton and Nassikas 2018). The higher figure represents a complete halting of land use conversion in forests and peatlands (i.e. assuming recent rates of carbon loss are saved each year). Separate estimates of degradation only range from 1.0-2.18 $GtCO_2$ yr^{-1}. Reduced deforestation and forest degradation include conservation of existing carbon pools in vegetation and soil through protection in reserves, controlling disturbances such as fire and pest outbreaks, and changing management practices. Differences in estimates stem from varying land cover definitions, time periods assessed, and carbon pools included (most higher estimates include belowground, dead wood, litter, soil, and peat carbon). When deforestation and degradation are halted, it may take many decades to fully recover the biomass initially present in native ecosystems (Meli et al. 2017, See also Chapter 4.8.3).

Afforestation/Reforestation (A/R) and forest restoration can increase carbon sequestration in both vegetation and soils by 0.5–10.1 $GtCO_2$ yr^{-1} *(medium confidence)* (Fuss et al. 2018; Griscom et al. 2017a; Hawken 2017; Kreidenweis et al. 2016; Li et al. 2016; Huang et al. 2017; Sonntag et al. 2016; Lenton 2014; McLaren 2012; Lenton 2010; Erb et al. 2018a; Dooley and Kartha 2018; Yan et al. 2017; Houghton et al. 2015; Houghton and Nassikas 2018). Afforestation is the conversion to forest of land that historically has not contained forests. Reforestation is the conversion to forest of land that has previously contained forests but that has been converted to some other use. Forest restoration refers to practices aimed at regaining ecological integrity in a deforested or degraded forest landscape. The lower estimate represents the lowest range from an earth system model (Yan et al. 2017) and of sustainable global negative emissions potential (Fuss et al. 2018), and the higher estimate reforests all areas where forests are the native cover type, constrained by food security and biodiversity considerations (Griscom et al. 2017a). It takes time for full carbon removal to be achieved as the forest grows. Removal occurs at faster rates in young to medium aged forests and declines thereafter such that older forest stands have smaller carbon removals but larger stocks with net uptake of carbon slowing as forests reach maturity (Yao et al. 2018; Poorter et al. 2016; Tang et al. 2014). The land intensity of afforestation and reforestation has been estimated at 0.0029 km^2 tC^{-1} yr^{-1} (Smith et al. 2016a). Boysen et al. (2017) estimated that to sequester about 100 GtC by 2100 would require 13 Mkm^2 of abandoned cropland and pastures. See also Chapter 4.8.3.

Forest management has the potential to mitigate 0.4-2.1 $GtCO_2$-eq yr^{-1} *(medium confidence)* (Sasaki et al. 2016; Griscom et al. 2017; Sasaki et al. 2012). Forest management can alter productivity, turnover rates, harvest rates carbon in soil, and carbon in wood products (Erb et al. 2017; Campioli et al. 2015; Birdsey and Pan 2015; Erb et al. 2016; Noormets et al. 2015; Wäldchen et al. 2013; Malhi et al. 2015; Quesada et al. 2018; Nabuurs et al. 2017; Bosello et al. 2009) (see also Chapter 4, Section 4.8.4). Fertilisation may enhance productivity but would increase N_2O emissions. Preserving and enhancing carbon stocks in forests has immediate climate benefits but the sink can saturate and is vulnerable to future climate change (Seidl et al. 2017). Wood can be harvested and used for bioenergy substituting for fossil fuels (with or without carbon capture and storage) (Section 2.6.1.5), for long-lived products such as timber (see below), to be buried as biochar (Section 2.6.1.1) or to be used in the wider bioeconomy, enabling areas of land to be used continuously for mitigation. This leads to initial carbon loss and lower carbon stocks but with each harvest cycle, the carbon loss (debt) can be paid back and after a parity time, result in net savings (Laganière et al. 2017; Bernier and Paré 2013; Mitchell et al. 2012; Haberl et al. 2012; Haberl 2013; Ter-Mikaelian et al. 2015; Macintosh et al. 2015). The trade-off between maximising forest C stocks and maximising substitution is highly dependent on the counterfactual assumption (no-use vs. extrapolation of current management), initial forest conditions and site-specific contexts such as regrowth rates and, the displacement factors and efficiency of substitution, and relative differences in emissions released during extraction, transport and processing of the biomass- or fossil-based resources as well as assumptions about emission associated with the product or energy source that is substituted (Grassi et al. 2018b; Nabuurs et al. 2017; Pingoud et al. 2018; Smyth et al. 2017a; Luyssaert et al. 2018; Valade et al. 2017; York 2012; Ter-Mikaelian et al. 2014; Naudts et al. 2016b; Mitchell et al. 2012; Haberl et al. 2012; Macintosh et al. 2015; Laganière et al. 2017; Haberl 2013). This leads to uncertainty about optimum mitigation strategies in managed forests, while high carbon ecosystems such as primary forests would have large initial carbon losses and long pay-back times and thus protection of stocks would be more optimal (Lemprière et al. 2013; Kurz et al. 2016; Keith et al. 2014). See also 4.8.4.

Global mitigation potential from increasing the demand of wood products to replace construction materials range from 0.25–1 $GtCO_2$-eq yr^{-1} (McLaren 2012; Miner 2010) (*medium confidence)*, the uncertainty is determined in part by consideration of the factors described above, and is sensitive to the displacement factor, or the substitution benefit in CO_2, when wood is used instead of another material, which may vary in the future as other sectors reduce emissions, as well as market factors (Sathre and O'Connor 2010; Nabuurs et al. 2018; Iordan et al. 2018; Braun et al. 2016; Gustavsson et al. 2017; Peñaloza et al. 2018; Soimakallio et al. 2016; Grassi et al. 2018b). Using harvested carbon in long-lived products (e.g., for construction) can represent a store that can sometimes be from decades to over a century while the wood can also substitute for intensive building materials, avoiding emissions from the production of concrete and steel (Sathre and O'Connor 2010; Smyth et al. 2017b; Nabuurs et al. 2007; Lemprière et al. 2013). The harvest of carbon and storage in products affects the net carbon balance of the forest sector, with the aim of sustainable forest management strategies being to optimise carbon stocks and use of harvested products to generate sustained mitigation benefits (Nabuurs et al. 2007).

Biophysical effects of forest response options are variable depending on the location and scale of activity (Section 2.6). Reduced deforestation or afforestation in the tropics contributes to climate mitigation through both biogeochemical and biophysical effects. It also maintains rainfall recycling to some extent. In contrast, in higher latitude boreal areas observational and modelling studies show that afforestation and reforestation lead to local and global warming effects, particularly in snow covered regions in the winter as the albedo is lower for forests than bare snow (Bathiany et al. 2010a; Dass et al. 2013; Devaraju et al. 2018b; Ganopolski et al. 2001b; Snyder et al. 2004a; West et al. 2011; Arora and Montenegro 2011) (Section 2.6). Management, e.g. thinning practices in forestry, could increase the albedo in regions where albedo decreases with age. The length of rotation cycles in forestry affects tree height and thus roughness, and through the removal of leaf mass, harvest reduces evapotranspiration (Erb et al. 2017) which could lead to increased fire susceptibility in the tropics. In temperate and boreal sites, biophysical forest management effects on surface temperature were shown to be of similar magnitude than changes in land cover (Luyssaert et al. 2014b). These biophysical effects could be of a magnitude to overcompensate biogeochemical effects, e.g. the sink strength of regrowing forests after past depletions (Luyssaert et al. 2018; Naudts et al. 2016b), but many parameters and assumptions on counterfactual influence the account (Anderson et al. 2011; Li et al. 2015b; Bright et al. 2015).

Forest cover also affects climate through reactive gases and aerosols with *limited evidence* and *medium agreement* that the decrease in the emissions of BVOC resulting from the historical conversion of forests to cropland has, resulted in a positive radiative forcing through direct and indirect aerosol effects, a negative radiative forcing through the reduction in the atmospheric lifetime of methane it has increased an decreased ozone concentrations in different regions (see Section 2.4).

2.6.1.3 Land management of soils

The global mitigation potential for increasing soil organic matter stocks in mineral soils is estimated to be in the range of 0.4–8.64 $GtCO_2$ yr^{-1} *(high confidence)* though the full literature range is wider with high uncertainty related to some practices (Fuss et al. 2018; Sommer and Bossio 2014; Lal 2010; Lal et al. 2004; Conant et al. 2017; Dickie et al. 2014; Frank et al. 2017a; Griscom et al. 2017c; Herrero et al. 2015, 2016b; McLaren 2012; Paustian et al. 2016; Poeplau and Don 2015; Powlson et al. 2014b; Smith et al. 2016d; Zomer et al. 2017). Some studies have separate potentials for soil carbon sequestration in croplands (0.25-6.78 $GtCO_2$ yr^{-1}) (Griscom et al. 2017a; Hawken 2017; Frank et al. 2017a; Paustian et al. 2016; Herrero et al. 2016a; Henderson et al. 2015b; Dickie et al. 2014; Conant et al. 2017; Lal 2010) and soil carbon sequestration in grazing lands (0.13-2.56 $GtCO_2$ yr^{-1}) (Griscom et al. 2017a; Hawken 2017; Frank et al. 2017a; Paustian et al. 2016; Powlson et al. 2014a; McLaren 2012; Zomer et al. 2017; Smith et al. 2015; Sommer and Bossio 2014; Lal 2010). The potential for soil carbon sequestration and storage varies considerably, depending on prior and current land management approaches, soil type, resource availability, environmental conditions, microbial composition and nutrient availability among others (Hassink and Whitmore 1997; Smith and Dukes 2013; Palm et al. 2014; Lal 2013; Six et al. 2002; Feng et al. 2013). Soils are a finite carbon sink and sequestration rates may decline to negligible levels over as little as a couple of decades as soils reach carbon saturation (West et al. 2004; Smith and Dukes 2013). The sink is at risk of reversibility, in particular due to increased soil respiration under higher temperatures (section 2.3)

Land management practices to increase carbon interact with agricultural and fire management practices (see Cross-chapter box 3: Fire and Climate Change, and Chapter 5) and include improved rotations with deeper rooting cultivars, addition of organic materials, and agroforestry (Lal 2011; Smith et al. 2008; Lorenz and Pitman 2014; Lal 2013; Vermeulen et al. 2012a; de Rouw et al. 2010). Adoption of green manure cover crops, while increasing cropping frequency or diversity, helps sequester SOC (Poeplau and Don 2015; Mazzoncini et al. 2011; Luo et al. 2010). Studies of the long-term SOC sequestration potential of conservation agriculture, i.e. the simultaneous adoption of minimum tillage, (cover) crop residue retention and associated soil surface coverage, and crop rotations, includes results that are both positive (Powlson et al. 2016; Zhang et al. 2014) and inconclusive (Cheesman et al. 2016; Palm et al. 2014; Govaerts et al. 2009).

The efficacy of reduced and zero-till practices is highly context-specific; many studies demonstrate increased carbon storage (e.g. Paustian et al., 2000; Six et al., 2004; van Kessel et al., 2013), while others show the opposite effect (Sisti et al. 2004; Álvaro-Fuentes et al. 2008; Christopher et al. 2009). On the other hand, deep ploughing can contribute to SOC sequestration by burying soil organic matter in the subsoil where it decomposes slowly (Alcántara et al. 2016). Meta-analyses (Haddaway et al. 2017; Luo et al. 2010; Meurer et al. 2018) also show a mix of positive and negative responses, and the lack of robust comparisons of soils on an equivalent mass basis continues to be a problem for credible estimates (Wendt and Hauser 2013; Powlson et al. 2011; Powlson et al. 2014).

Soil carbon management interacts with N_2O (Paustian et al. 2016). For example, (Li et al. 2005) estimate that the management strategies required to increase C sequestration (reduced tillage, crop residue, and manure recycling) would increase N_2O emissions significantly, offsetting 75–310% of the C sequestered in terms of CO_2 equivalence, while other practices such as cover crops can reduce N_2O emissions (Kaye and Quemada 2017).

The management of soil erosion could avoid a net emissions of 1.36– 3.67 $GtCO_2$ yr^{-1} and create a sink of 0.44–3.67 $GtCO_2$ yr^{-1} *(low confidence)* (Jacinthe and Lal 2001; Lal et al. 2004; Stallard 1998; Smith et al. 2001; Van Oost et al. 2007). The overall impact of erosion control on mitigation is context-specific and uncertain at the global level, uncertain and the final fate of eroded material is still debated (Hoffmann et al., 2013).

Biochar is produced by thermal decomposition of biomass in the absence of oxygen (pyrolysis) into a stable, long-lived product like charcoal that is relatively resistant to decomposition (Lehmann et al. 2015) and which can stabilise organic matter added to soil (Han Weng et al. 2017). Although charcoal has been used traditionally by many cultures as a soil amendment, "modern biochar", produced in facilities that control emissions, is not widely used. The range of global potential of biochar is 0.03-6.6 $GtCO_2$-eq yr^{-1} by 2050 including energy substitution, with 0.03-4.9 $GtCO_2$ yr^{-1} for CDR only *(medium confidence)* (Griscom et al. 2017a; Hawken 2017; Paustian et al. 2016; Fuss et al. 2018; Lenton 2014, 2010; Powell and Lenton 2012a; Woolf et al. 2010; Pratt and Moran 2010; Smith 2016; Roberts et al. 2010). An analysis in which biomass supply constraints were applied to protect against food insecurity, loss of habitat and land degradation, estimated *technical potential* abatement of 3.7–6.6 $GtCO_2$-eq yr^{-1} (including 2.6–4.6 $GtCO_2$ yr^{-1} carbon stabilisation) (Woolf et al. 2010). Fuss et al. (2018) propose a range of 0.5–2 $GtCO_2$-eq yr^{-1} as the *sustainable potential* for negative emissions through biochar. (Griscom et al. 2017b) suggest a potential of 1.0 $GtCO_2$ yr^{-1} based on available residues. Biochar can provide additional climate change mitigation benefits by decreasing nitrous oxide (N_2O) emissions from soil and reducing nitrogen fertiliser requirements in agricultural soils (Borchard et al. 2019). Application of biochar to cultivated soils can darken the surface and reduce its mitigation potential via decreases in surface albedo, but the magnitude of this effect depends on soil moisture content, biochar application method and type of land use (*low confidence*) (Verheijen et al. 2013; Bozzi et al. 2015). Biochar is discussed in more detail in Chapter 4, Section 4.9.5.

2.6.1.4 Land management in other ecosystems

Protection and restoration of wetlands, peatlands and coastal habitats reduces net carbon loss (primarily from sediment/soils) and provides continued or enhanced natural CO_2 removal (Chapter 4, section 4.9.4). Reducing annual emissions from peatland conversion, draining and burning could mitigate 0.45-1.22 $GtCO_2$-eq yr^{-1} up to 2050 *(medium confidence)* (Hooijer et al. 2010; Griscom et al. 2017; Hawken 2017) and peatland restoration 0.15 to 0.81 *(low confidence)* (Couwenberg et al. 2010; Griscom et al. 2017b). The upper end from Griscom et al. (2017) represents a maximum sustainable potential (accounting for biodiversity and food security safeguards) for rewetting and biomass enhancement. Wetland drainage and rewetting was included as a flux category under the second commitment Period of the Kyoto protocol, with significant management knowledge gained over the last decade (IPCC 2013c). However, there are high uncertainties as to the carbon storage and flux rates, in particular the balance between CH_4 sources and CO_2 sinks (Spencer et al. 2016). Peatlands are sensitive to climate change which may increase carbon uptake by vegetation and carbon emissions due to respiration, with the balance being regionally dependent (*high confidence*). There is *low confidence* about the future peatland sink globally. Some peatlands have been found to be resilient to climate change (Minayeva and Sirin 2012), but the combination of land use change and climate change may make them vulnerable to fire (Sirin et al. 2011). While models show mixed results for the future sink (Spahni et al. 2013; Chaudhary et al. 2017; Ise et al. 2008), a study that used extensive historical data sets to project change under future warming scenarios found that the currently global peatland sink could increase slightly until 2100 and decline thereafter (Gallego-Sala et al. 2018).

Reducing the conversion of coastal wetlands (mangroves, seagrass and marshes) could reduce emissions by 0.11–2.25 $GtCO_2$-eq yr^{-1} by 2050 *(medium confidence)* (Pendleton et al. 2012; Griscom et al. 2017a; Howard et al. 2017; Hawken 2017). Mangrove restoration can mitigate the release of 0.07 $GtCO_2$ yr^{-1} through rewetting (Crooks et al. 2011) and take up 0.02–0.84 $GtCO_2$ yr^{-1} from biomass and soil enhancement (Griscom et al. 2017b) *(medium confidence)*. The ongoing benefits provided by mangroves as a natural carbon sink can be nationally-important for Small Island Developing States (SIDS) and other countries with extensive coastlines, based on estimates of high carbon sequestration rates per unit area (McLeod et al. 2011; Duarte et al. 2013; Duarte 2017; Taillardat et al. 2018). There is only *medium confidence* in the effectiveness of enhanced carbon uptake using mangroves, due to the many uncertainties regarding the response of mangroves to future climate change (Jennerjahn et al. 2017); dynamic changes in distributions (Kelleway et al. 2017) and other local-scale factors affecting long-term sequestration and climatic benefits (e.g., methane release; Dutta et al. 2017). The climate mitigation potential of coastal vegetated habitats (mangrove forests, tidal marshes and seagrasses) is considered in Chapter 5 of the IPCC Special Report on the Ocean, Cryosphere and Climate Change (SROCC), in a wider 'blue carbon' context.

2.6.1.5 Bioenergy and bioenergy with carbon capture and storage (BECCS)

An introduction and overview of bioenergy and BECCS can be found in the Cross-Chapter Boxes: Cross-Chapter Box 12: Traditional biomass use, Chapter 7; Cross-Chapter Box 7: Bioenergy and BECCS in mitigation scenarios, Chapter 6. CCS technologies are discussed in SR15. The discussion below refers to modern bioenergy only, (e.g., liquid biofuels for transport and the use of solid biofuels in combined heat and power plants).

The mitigation potential of bioenergy coupled with CCS, i.e., BECCS, is estimated to be between 0.4 and 11.3 $GtCO_2$ yr^{-1} *(medium confidence)* based on studies that directly estimate mitigation for BECCS (not bioenergy) in units of CO_2 (not EJ) (McLaren 2012; Lenton 2014; Fuss et al. 2018; Turner et al. 2018b; Lenton 2010; Koornneef et al. 2012; Powell and Lenton 2012b). SR15 reported a potential of 1-85 $GtCO_2$ yr^{-1} which they noted could be narrowed to a range of 0.5 to 5 $GtCO_2$ yr^{-1} when taking account of sustainability aims (Fuss et al. 2018). The upper end of the SR15 range is considered as a theoretical potential. Previously, the IPCC Special Report on Renewable Energy Sources concluded the technical potential of biomass supply for energy (without BECCS) could reach 100-300 EJ yr^{-1} by 2050, which would be 2-15 $GtCO_2$ yr^{-1} (using conversion factors 1 EJ = 0.02-0.05 $GtCO_2$ yr^{-1} emission reduction, SR15). A range of recent studies including sustainability or economic constraints estimate that 50-244 EJ (1 to 12 $GtCO_2$ yr^{-1} using conversion factors above) of bioenergy could be produced on 0.1-13 Mkm^2 of land (Fuss et al. 2018; Chan and Wu 2015; Schueler et al. 2016; Wu et al. 2013; Searle and Malins 2015; Wu et al. 2019; Heck et al. 2018; Fritz et al. 2013) SR15 SPM).

There is *high confidence* that the most important factors determining future biomass supply for energy are land availability and land productivity (Berndes et al. 2013; Creutzig et al. 2015a; Woods et al. 2015; Daioglou et al. 2019). Estimates of marginal/degraded lands currently considered available for bioenergy range from 3.2 to 14.0 Mkm^2, depending on the adopted sustainability criteria, land class definitions, soil conditions, land mapping method and environmental and economic considerations (Campbell et al. 2008; Cai et al. 2011; Lewis and Kelly 2014).

Bioenergy production systems can lead to net emissions in the short term that can be "paid-back" over time, with multiple harvest cycles and fossil fuel substitution, unlike fossil carbon emissions (Campbell et al. 2008; Cai et al. 2011; Lewis and Kelly 2014; De Oliveira Bordonal et al. 2015). Stablising bioenergy crops in previous high carbon forestland or peatland results in high emissions of carbon that may take from decades to more than a century to be re-paid in terms of net CO_2 emission savings from replacing fossil fuels, depending on previous forest carbon stock, bioenergy yields, and displacement efficiency (Elshout et al. 2015; Harper et al. 2018; Daioglou et al. 2017). In the case of bioenergy from managed forests, the magnitude and timing of the net mitigation benefits is controversial, as it varies with differences due to local climate conditions, forest management practice, fossil fuel displacement efficiency and methodological approaches (Hudiburg et al. 2011; Berndes et al. 2013; Guest et al. 2013; Lamers and Junginger 2013; Cherubini et al. 2016; Cintas et al. 2017; Laurance et al. 2018; Valade et al. 2018; Baker et al. 2019). Suitable bioenergy crops can be integrated in agricultural landscapes to reverse ecosystem carbon depletion (Creutzig et al. 2015a; Robertson et al. 2017; Vaughan et al. 2018; Daioglou et al. 2017). Cultivation of short rotation woody crops and perennial grasses on degraded land or cropland previously used for annual crops typically accumulate carbon in soils due to their deep root systems (Don et al. 2012; Robertson et al. 2017). The use of residues and organic waste as bioenergy feedstock can mitigate land use change pressures associated with bioenergy deployment, but residues are limited and the removal of residues that would otherwise be left on the soil could lead soil degradation (Chum et al. 2011; Liska et al. 2014; Monforti et al. 2015; Zhao et al. 2015; Daioglou et al. 2016).

The steps required to cultivate, harvest, transport, process and use biomass for energy generate emissions of GHGs and other climate pollutants (Chum et al. 2011; Creutzig et al. 2015b; Staples et al. 2017; Daioglou et al. 2019). Life-cycle GHG emissions of modern bioenergy alternatives are usually lower than those for fossil fuels (*robust evidence, medium agreement*) (Chum et al. 2011; Creutzig et al. 2015b). The magnitude of these emissions largely depends on location, (e.g. soil quality, climate), prior land use, feedstock used (e.g., residues, dedicated crops, algae), land use practice (e.g., soil management, fertiliser use), biomass transport (distances and transport modes), the bioenergy conversion pathway and product (e.g., wood pellets, ethanol). Use of conventional food and feed crops as a feedstock generally provides the highest bioenergy yields per hectare, but also causes more GHG emissions per unit energy compared to agriculture residues, biomass from managed forests, and lignocellulosic crops such as short-rotation coppice and perennial grasses (Chum et al. 2011; Gerbrandt et al. 2016). This is due to the application of fertilisers and other inputs (Oates et al. 2016; Rowe et al. 2016; Lai et al. 2017; Robertson et al. 2017).

Bioenergy from dedicated crops are in some cases held responsible for GHG emissions resulting from indirect land use change (iLUC), that is the bioenergy activity may lead to displacement of agricultural or forest activities into other locations, driven by market-mediated effects. Other mitigation options may also cause iLUC. At a global level of analysis, indirect effects are not relevant because all land-use emissions are direct. iLUC emissions are potentially more significant for crop-based feedstocks such as corn, wheat and soybean, than for advanced biofuels from lignocellulosic materials (Chum et al. 2011; Wicke et al. 2012; Valin et al. 2015; Ahlgren and Di Lucia 2014). Estimates of emissions from iLUC are inherently uncertain, are widely debated in the scientific community, and are highly dependent on modelling assumptions, such as supply/demand elasticities, productivity estimates, incorporation or exclusion of emission credits for coproducts, scale of biofuel deployment (Rajagopal and Plevin 2013; Finkbeiner 2014; Kim et al. 2014; Zilberman 2017). In some cases, iLUC effects are estimated to result in emission reductions. For example, market-mediated effects of bioenergy in North America showed potential for increased carbon stocks by inducing conversion of pasture or marginal land to forestland (Cintas et al. 2017; Duden et al. 2017; Dale et al. 2017; Baker et al. 2019). There is a wide range of variability in iLUC values for different types of biofuels, from -75 to +55 g CO_2 MJ^{-1} (Ahlgren and Di Lucia 2014; Valin et al. 2015; Plevin et al. 2015; Taheripour and Tyner 2013; Bento and Klotz 2014). There is low confidence in attribution of emissions from iLUC to bioenergy.

Bioenergy deployment can have large biophysical effects on regional climate, with the direction and magnitude of the impact depending on the type of bioenergy crop, previous land use and seasonality (*limited* *evidence, medium agreement*). A study of two alternative future bioenergy scenarios using 15 Mkm^2 of intensively used managed land or conversion of natural areas showed a nearly neutral effect on surface temperature at global levels (considering biophysical effects and CO_2 and N_2O fluxes from land but not substitution effects), although there were significant seasonal and regional differences (Kicklighter et al. 2013). Modelling studies on biofuels in the US found the switch from annual crops to perennial bioenergy plantations like Miscanthus in the US could lead to regional cooling due to increases in evapotranspiration and albedo (Georgescu et al. 2011; Harding et al. 2016), with perennial bioenergy crop expansion over suitable abandoned and degraded farmlands causing near-surface cooling up to 5°C during the growing season (Wang et al. 2017b). Similarly, growing sugarcane on existing cropland in Brazil cools down the local surface during daytime conditions up to -1°C, but warmer conditions occurs if sugar cane is deployed at the expense of natural vegetation (Brazilian Cerrado) (Loarie et al. 2011). In general, bioenergy crops (as for all crops) induce a cooling of ambient air during the growing season, but after harvest the decrease in evapotranspiration can induce warming (Harding et al. 2016; Georgescu et al. 2013; Wang et al. 2017b). Bioenergy crops were found to cause increased isoprene emissions in a scenario where 0.69 Mkm^2 of oil palm for biodiesel in the tropics and 0.92 Mkm^2 of short rotation coppice (SRC) in the mid-latitudes were planted, but effects on global climate were negligible (Ashworth et al. 2012).

2.6.1.6 Enhanced weathering

Weathering is the natural process of rock decomposition via chemical and physical processes in which CO_2 is removed from the atmosphere and converted to bicarbonates and/or carbonates (IPCC 2005). Formation of calcium carbonates in the soil provides a permanent sink for mineralised organic carbon (Manning 2008; Beerling et al. 2018). Mineral weathering can be enhanced through grinding up rock material to increase the surface area, and distributing it over land to provide carbon removals of 0.5–4.0 $GtCO_2$ yr^{-1} *(medium confidence)* (Beerling et al. 2018; Lenton 2010; Smith et al. 2016a; Taylor et al. 2016). While the geochemical potential is quite large, agreement on the technical potential is low due to a variety of unknown parameters and of limits such as rates of mineral extraction, grinding, delivery, and challenges with scaling and deployment.

2.6.1.7 Demand management in the food sector (diet change, waste reduction)

Demand-side management has the potential for climate change mitigation via reducing emissions from production, switching to consumption of less emission intensive commodities, and making land available for carbon dioxide removal (see Chapter 5 Section 5.5.2). Reducing food losses and waste increases the overall efficiency of food value chains (less land and inputs needed) along the entire supply chain, and has the potential to mitigate 0.8-4.5 $GtCO_2$-eq yr^{-1} (*high confidence*) (Chapter 5 section 5.5.2.5, Bajželj et al. 2014; Dickie et al 2014; Hawken 2017; Hiç et al. 2016).

Shifting to diets that are lower in emissions-intensive foods like beef delivers a mitigation potential of 0.7-8.0 $GtCO_2$-eq yr^{-1} (*high confidence*) (Bajželj et al. 2014; Dickie et al. 2014; Herrero et al. 2016; Hawken 2017; Springmann et al. 2016; Tilman and Clark 2014; Hedenus et al. 2014; Stehfest et al. 2009) with most of the higher end estimates (> 6 $GtCO_2$-eq yr^{-1}) based on veganism, vegetarianism or very low ruminant meat consumption) (see Chapter 5, Section 5.5.2). In addition to direct mitigation gains, decreasing meat consumption, primarily of ruminants, and reducing wastes further reduces water use, soil degradation, pressure on forests, land used for feed potentially freeing up land for mitigation (Tilman and Clark 2014) (see chapters 5 and 6). Additionally, consumption of locally produced food, shortening the supply chain, can in some cases minimise food loss, contribute to food security, and reduce GHG emissions associated with energy consumption and food loss (see Chapter 5 Section 5.5.2.6).

2.6.2 Integrated pathways for climate change mitigation

Land-based response options have the potential to interact, resulting in additive effects (e.g., climate co-benefits) or negating each other (e.g., through competition for land), they also interact with mitigation options in other sectors (such as energy or transport), thus they need to be assessed collectively under different climate mitigation targets and in combination with other sustainability goals (Popp et al. 2017; Obersteiner et al. 2016; Humpenöder et al. 2018). Integrated Assessment Models (IAMs) with distinctive land-use modules are the basis for the assessment of mitigation pathways as they combine insights from various disciplines in a single framework and cover the largest sources of anthropogenic GHG emissions from different sectors (see also SR15 Chapter 2 and technical Annex for more details). IAMs consider a limited, but expanding, portfolio of land-based mitigation options. Furthermore, the inclusion and detail of a specific mitigation measure differs across IAMs and studies (see also SR15 and Chapter 6). For example, the IAM scenarios based on the Shared Socio-economic Pathways (SSPs) (Riahi et al. 2017)(see more details on the SSPs in Cross-Chapter Box 1: Scenarios, Chapter 1) include possible trends in agriculture and land use for five different socioeconomic futures, but cover a limited set of land-based mitigation options: dietary changes, higher efficiency in food processing (especially in livestock production systems), reduction of food waste, increasing agricultural productivity, methane reductions in rice paddies, livestock and grazing management for reduced methane emissions from enteric fermentation, manure management, improvement of N-efficiency, 1st generation biofuels, reduced deforestation, afforestation, 2nd generation bioenergy crops and BECCS (Popp et al. 2017). However, many "natural climate solutions" (Griscom et al. 2017b), such as forest management, rangeland management, soil carbon management or wetland management, are not included in most of these scenarios. In addition, most IAMs neglect the biophysical effects of land-use such as changes in albedo or evapotranspiration with few exceptions (Kreidenweis et al. 2016).

Mitigation pathways, based on IAMs, are typically designed to find the least cost pathway to achieve a pre-defined climate target (Riahi et al. 2017). Such cost-optimal mitigation pathways, especially in RCP2.6 (broadly a 2°C target) and 1.9 scenarios (broadly a 1.5°C target), project GHG emissions to peak early in the 21st century, strict GHG emission reduction afterwards and, depending on the climate target, net carbon dioxide removal (CDR) from the atmosphere in the second half of the century (see Chapter 2 of SR15, (Tavoni et al. 2015; Riahi et al. 2017). In most of these pathways, land use is of great importance because of its mitigation potential as discussed in section 2.7.1: these pathways are based on the assumptions that large-scale afforestation and reforestation removes substantial amounts of CO_2 from the atmosphere; biomass grown on cropland or from forestry residues can be used for energy generation or BECCS substituting fossil fuel emissions and generating CDR; non-CO_2 emissions from agricultural production can be reduced, even under improved agricultural management (Popp et al. 2017; Rogelj et al. 2018a; Van Vuuren et al. 2018, Frank et al. 2018).

From the IAM scenarios available to this assessment, a set of feasible mitigation pathways has been identified which is illustrative of the range of possible consequences on land use and GHG emissions (presented in this chapter) and sustainable development (see Chapter 6). Thus, the IAM scenarios selected here vary due to underlying socio-economic and policy assumptions, mitigation options considered, long-term climate goal, the level of inclusion of other sustainability goals (such as land and water restrictions for biodiversity conservation or food production), and models by which they are generated.

In the baseline case without climate change mitigation, global CO_2 emissions from land-use change decrease over time in most scenarios due to agricultural intensification and decreases in demand for agricultural commodities – some turning even negative by the end of the century due to abandonment of agricultural land and associated carbon uptake through vegetation regrowth. Median global CO_2 emissions from land-use change across 5 SSPs and 5 IAMs decrease throughout the 21st century: 3, 1.9 and -0.7 $GtCO_2\ yr^{-1}$ in 2030, 2050 and 2100 respectively (Figure 2.25). In contrast, CH_4 and N_2O emissions from agricultural production remain rather constant throughout the 21st century (CH_4: 214, 231.7 and 209.1 Mt $CH_4\ yr^{-1}$ in 2030, 2050 and 2100 respectively; N_2O: 9.1, 10.1 and 10.3 Mt $N_2O\ yr^{-1}$ in 2030, 2050 and 2100 respectively).

In the mitigation cases (RCP4.5, RCP2.6 and RCP1.9), most of the scenarios indicate strong reductions in CO_2 emissions due to i) reduced deforestation and ii) carbon uptake due to afforestation. However, CO_2 emissions from land use can occur in some mitigation scenarios as a result of weak land-use change regulation (Fujimori et al. 2017; Calvin et al. 2017) or displacement effects into pasture land caused by high bioenergy production combined with forest protection only (Popp et al. 2014). The level of carbon dioxide removal globally (median value across SSPs and IAMs) increases with the stringency of the climate target (RCP4.5, RCP2.6 and RCP1.9) for both afforestation (-1.3, -1.7 and -2.4 $GtCO_2\ yr^{-1}$ in 2100) and BECCS (-6.5, -11 and -14.9 $GtCO_2\ yr^{-1}$ in 2100; see also Cross-Chapter Box 7: Bioenergy and BECCS in mitigation scenarios, Chapter 6). In the mitigation cases (RCP4.5, RCP2.6 and RCP1.9), CH_4 and N_2O emissions are remarkably lower compared to the baseline case (CH_4: 133.2, 108.4 and 73.5 Mt $CH_4\ yr^{-1}$ in 2100; N_2O: 7.4, 6.1 and 4.5 Mt $N_2O\ yr^{-1}$ in 2100; see previous paragraph for CH_4 and N_2O emissions in the baseline case). The reductions in the mitigation cases are mainly due to improved agricultural management such as improved nitrogen fertiliser management, improved water management in rice production, improved manure management by for example covering of storages or adoption of biogas plants, better herd management and better quality of livestock through breeding and improved feeding practices. In addition, dietary shifts away from emission-intensive livestock products also lead to decreased CH_4 and N_2O emissions especially in RCP2.6 and RCP1.9 scenarios. However, high levels of bioenergy production can result in increased N_2O emissions due to nitrogen fertilisation of dedicated bioenergy crops.

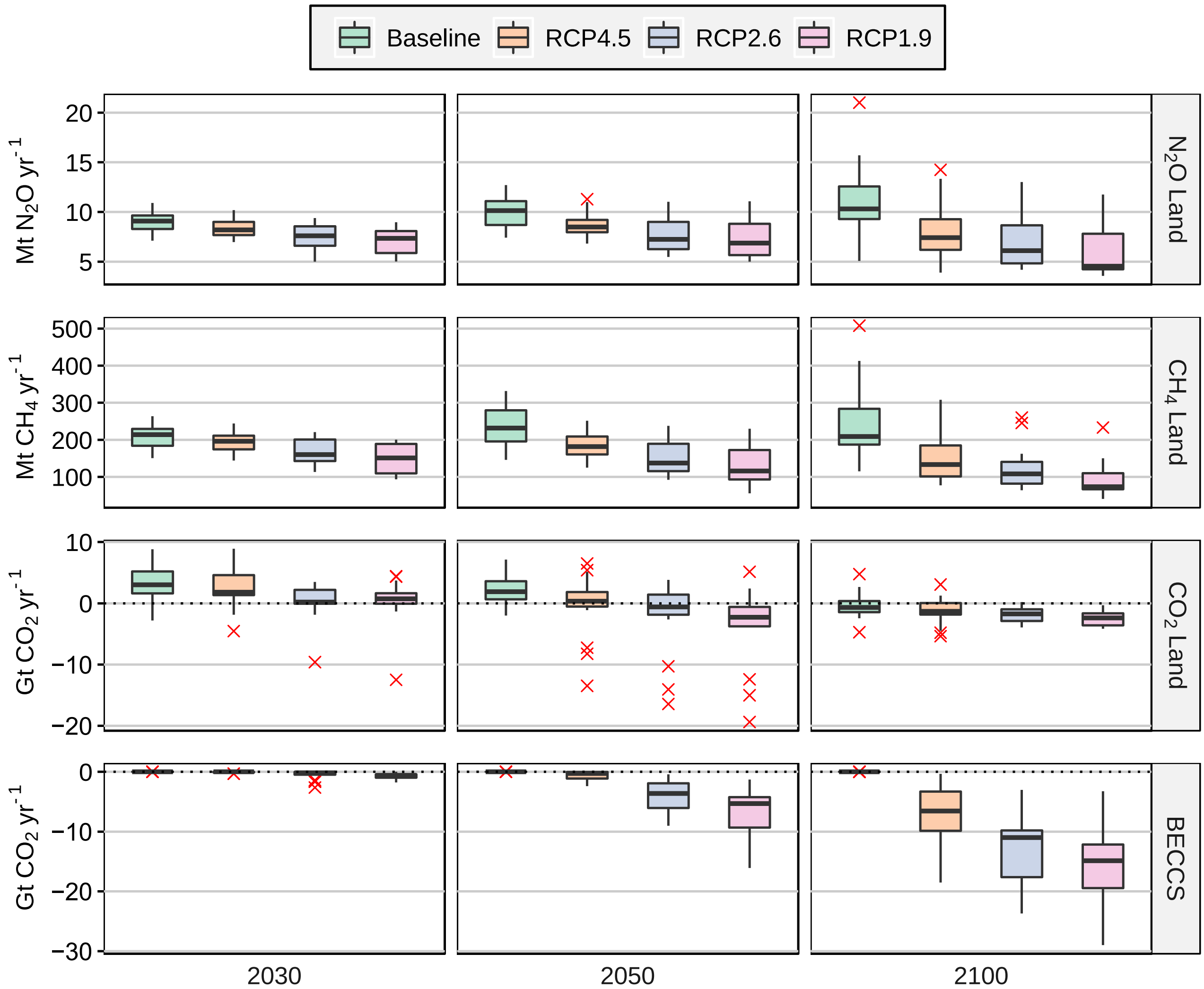


Figure 2.25 Land-based global GHG emissions and removals in 2030, 2050 and 2100 for Baseline, RCP4.5, RCP2.6 and RCP1.9 based on the Shared Socioeconomic Pathways (SSP) (Popp et al. 2017; Rogelj et al. 2018; Riahi et al. 2017). Data is from an update of the IAMC Scenario Explorer developed for the SR15 (Huppmann et al. 2018; Rogelj et al. 2018). Boxplots (Tukey style) show median (horizontal line), interquartile range (IQR box) and the range of values within 1.5 x IQR at either end of the box (vertical lines) across 5 SSPs and across 5 IAMs. Outliers (red crosses) are values greater than 1.5 x IQR at either end of the box. The categories CO_2 Land, CH4 Land and N_2O Land include GHG emissions from land-use change and agricultural land use (including emissions related to bioenergy production). In addition, the category CO_2 Land includes negative emissions due to afforestation. BECCS reflects the CO_2 emissions captured from bioenergy use and stored in geological deposits.

Such high levels of carbon dioxide removal through mitigation options that require land conversion (BECCS and afforestation) shape the land system dramatically (Figure 2.26). Across the different RCPs, SSPs and IAMs median change of global forest area throughout the 21st century ranges from about -0.2 to +7.2 Mkm^2 between 2010 and 2100, and agricultural land used for 2nd generation bioenergy crop production ranges from about 3.2–6.6 Mkm^2 in 2100 (Popp et al. 2017; Rogelj et al. 2018). Land requirements for bioenergy and afforestation for a RCP1.9 scenario are higher than for a RCP2.6 and especially a RCP4.5 mitigation scenario. As a consequence of the expansion of mainly land-demanding mitigation options, global pasture land is reduced in most mitigation scenarios much stronger compared to baseline scenarios (median reduction of 0, 2.6, 5.1 and 7.5 Mkm^2 between 2010 and 2100 in Baseline, RCP4.5, RCP2.6 and RCP1.9 respectively). In addition, cropland for food and feed production decreases with the stringency of the climate target (+1.2, +0.2, -1.8 and -4 Mkm^2 in 2100 compared to 2010 in Baseline, RCP4.5, RCP2.6 and RCP1.9 respectively). These reductions in agricultural land for food and feed production are facilitated by agricultural intensification on agricultural land and in livestock production systems (Popp et al. 2017) but also by changes in consumption patterns (Fujimori et al. 2017; Frank et al. 2017b). The pace of projected land-use change over the coming decades in ambitious mitigation scenarios goes well beyond historical changes in some instances (Turner et al. 2018c), see also SR15). This raises issues for societal acceptance, and distinct policy and governance for avoiding negative consequences for other sustainability goals (Humpenöder et al. 2018; Obersteiner et al. 2016; Calvin et al. 2014), see Chapter 6 and 7).

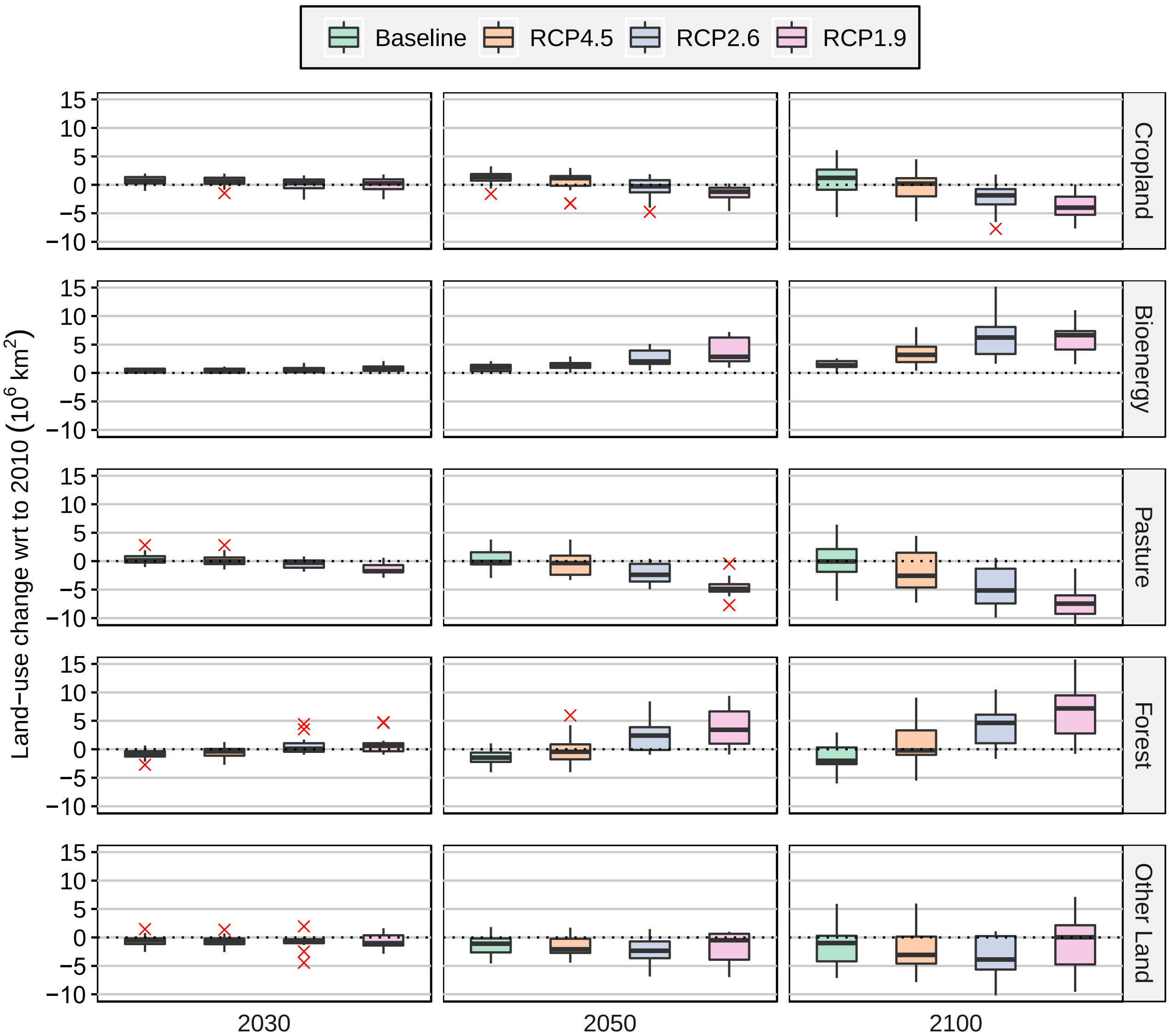


Figure 2.26 Global change of major land cover types by 2030, 2050 and 2100 relative to 2010 for Baseline, RCP4.5, RCP2.6 and RCP1.9 based on the Shared Socioeconomic Pathways (SSP) (Popp et al. 2017; Rogelj et al. 2018; Riahi et al. 2017). Data is from an update of the IAMC Scenario Explorer developed for the SR15 (Huppmann et al. 2018; Rogelj et al. 2018). Boxplots (Tukey style) show median (horizontal line), interquartile range IQR (box) and the range of values within 1.5 x IQR at either end of the box (vertical lines) across 5 SSPs and across 5 IAMs. Outliers (red crosses) are values greater than 1.5 x IQR at either end of the box. In 2010, total land cover at global scale was estimated 15-16 Mkm^2 for cropland, 0-0.14 Mkm^2 for

bioenergy, 30-35 Mkm^2 for pasture and 37-42 Mkm^2 for forest, across the IAMs that reported SSP pathways (Popp et al. 2017).

Different mitigation strategies can achieve the net emissions reductions that would be required to follow a Pathway that limits global warming to 2°C or 1.5°C, with very different consequences on the land system.

Figure 2.27 shows six alternative pathways (archetypes) for achieving ambitious climate targets (RCP2.6 and RCP1.9) highlighting land-based strategies and GHG emission. All pathways are assessed by different models but are all based on the Shared Socioeconomic Pathway 2 (SSP2) (Riahi et al. 2017), with all based on an RCP 1.9 mitigation pathway expect for Pathway 1, which is RCP2.6. All scenarios show land-based negative emissions but the amount varies across pathways, as do the relative contributions of different land-based Carbon Dioxide Removal (CDR) options, such as afforestation/reforestation and bioenergy with carbon capture and storage (BECCS).

Pathway 1 RCP2.6 "Portfolio" (Fricko et al. 2017) shows a strong near-term decrease of CO_2 emissions from land-use change, mainly due to reduced deforestation, as well as slightly decreasing N_2O and CH_4 emissions after 2050 from agricultural production due to improved agricultural management and dietary shifts away from emissions-intensive livestock products. However, in contrast to CO_2 emissions, which turn net-negative around 2050 due to afforestation/reforestation, CH_4 and N_2O emissions persist throughout the century due to difficulties of eliminating these residual emissions based on existing agricultural management methods (Stevanović et al. 2017; Frank et al. 2017b). In addition to abating land-related GHG emissions as well as increasing the terrestrial sink, this example also shows the importance of the land sector in providing biomass for BECCS and hence CDR in the energy sector. In this scenario, annual BECCS-based CDR is about 3-times higher than afforestation-based CDR in 2100 (-11.4 and -3.8 $GtCO_2$ yr^{-1} respectively). Cumulative CDR throughout the century amounts to -395 $GtCO_2$ for BECCS and -73 $GtCO_2$ for afforestation. Based on these GHG dynamics, the land sector turns GHG emission neutral in 2100. However, accounting also for BECCS-based CDR taking place in the energy sector but with biomass provided by the land sector turns the land sector GHG emission neutral already in 2060, and significantly net-negative by the end of the century.

Pathway 2 RCP1.9 "Increased Ambition" (Rogelj et al. 2018) has dynamics of land-based GHG emissions and removals that are very similar to those in Pathway 1 (RCP2.6) but all GHG emission reductions as well as afforestation/reforestation and BECCS-based CDR start earlier in time at a higher rate of deployment. Cumulative CDR throughout the century amounts to -466 $GtCO_2$ for BECCS and -117 $GtCO_2$ for afforestation.

Pathway 3 RCP 1.9 "Only BECCS", in contrast to Pathway 2, includes only BECCS-based CDR (Kriegler et al. 2017). In consequence, CO_2 emissions are persistent much longer, predominantly from indirect land-use change due to large-scale bioenergy cropland expansion into non-protected natural areas (Popp et al. 2017; Calvin et al. 2014). While annual BECCS CDR rates in 2100 are similar to Pathway 1 and 2 (-15.9 $GtCO_2$ yr^{-1}), cumulative BECCS-based CDR throughout the century is much larger (-944 $GtCO_2$).

Pathway 4 RCP1.9 "Early CDR" (Bertram et al. 2018) indicates that a significant reduction in the later century in the BECCS-related CDR as well as CDR in general can be achieved with earlier and mainly terrestrial CDR, starting already in 2030. In this scenario, terrestrial CDR is based on afforestation but could also be supported by soil organic carbon sequestration (Paustian et al. 2016) or other natural climate solutions such as rangeland or forest management (Griscom et al. 2017b). This scenario highlights the importance of the timing for CDR-based mitigation pathways (Obersteiner et al. 2016). As a result of near-term and mainly terrestrial CDR deployment, cumulative BECCS-based CDR throughout the century is limited to -300 $GtCO_2$, while cumulative afforestation-based CDR amounts to -428 $GtCO_2$.

In Pathway 5 RCP1.9 "Low residual emissions" (van Vuuren et al. 2018), land-based mitigation is driven by stringent enforcement of measures and technologies to reduce end-of-pipe non-CO_2 emissions and by introduction of in-vitro (cultured) meat, reducing residual N_2O and CH_4 emissions from agricultural production. In consequence, much lower amounts of CDR from afforestation and BECCS are needed with much later entry points to compensate for residual emissions. Cumulative CDR throughout the century amounts to -252 $GtCO_2$ for BECCS and -128 $GtCO_2$ for afforestation. Therefore, total cumulative land-based CDR in Pathway 5 is substantially lower compared to Pathways 2-4 (380 $GtCO_2$).

Finally, Pathway 6 RCP1.9 "Low Energy" (Grubler et al. 2018) – equivalent to pathway LED in SR15 – indicates the importance of other sectoral GHG emission reductions for the land sector. In this example, rapid and early reductions in energy demand and associated drops in energy-related CO_2 emissions, limit overshoot and decrease the requirements for negative emissions technologies, especially for land-demanding CDR such as biomass production for BECCS and afforestation. While BECCS is not used at all in Pathway 6, cumulative CDR throughout the century for afforestation amounts to -124 $GtCO_2$.

Besides their consequences on mitigation pathways and land consequences, those archetypes can also affect multiple other sustainable development goals that provide both challenges and opportunities for climate action (see Chapter 6).

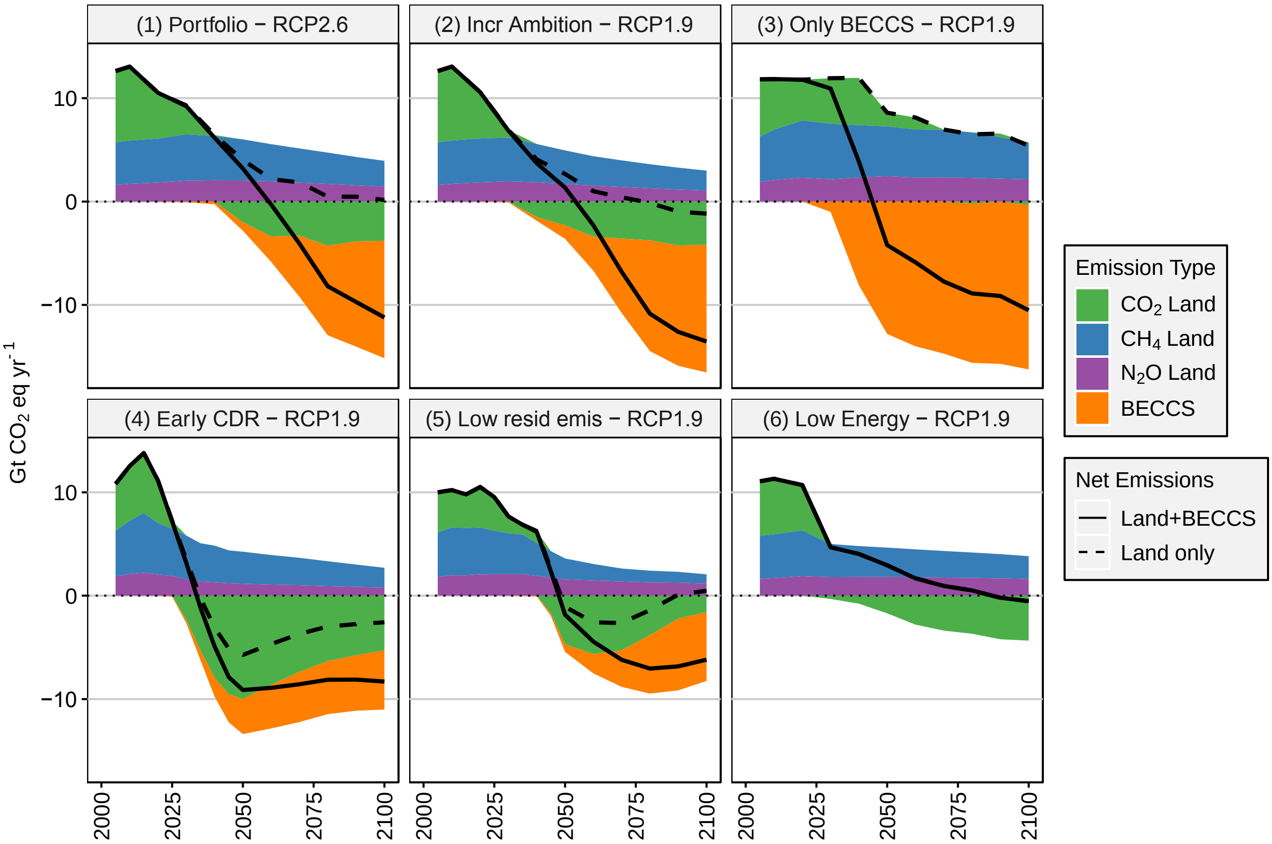


Figure 2.27 Evolution and break down of global land-based GHG emissions and removals under six alternative mitigation pathways, which illustrate the differences in timing and magnitude of land-based mitigation approaches including afforestation and BECCS. All pathways are based on different IAM realisations of SSP2. Pathway 1 is based on RCP 2.6, while all other pathways are based on RCP 1.9. Pathway 1: MESSAGE-GLOBIOM (Fricko et al. 2017); Pathway 2: MESSAGE-GLOBIOM (Rogelj et al. 2018); Pathway 3: REMIND-MAgPIE (Kriegler et al. 2017); Pathway 4: REMIND-MAgPIE (Bertram et al. 2018); Pathway 5: IMAGE (van Vuuren et al. 2018); Pathway 6: MESSAGE-GLOBIOM (Grubler et al. 2018). Data is from an update of the IAMC Scenario Explorer developed for the SR15 (Rogelj et al. 2018). The categories CO_2 Land, CH_4 Land and N_2O Land include GHG emissions from land-use change and agricultural land use (including emissions related to bioenergy production). In addition, the category CO_2 Land includes negative emissions due to afforestation. BECCS reflects the CO_2 emissions captured from bioenergy use and stored in geological deposits. Solid lines show the net effect of all land-based GHG emissions and removals (CO_2 Land, CH_4 Land, N_2O Land and BECCS), while dashed lines show the net effect excluding BECCS. CH_4 and N_2O emissions are converted to CO_2-eq using GWP factors of 28 and 265 respectively.

2.6.3 The contribution of response options to the Paris Agreement

The previous sections indicated how land based response options have the potential to contribute to the Paris Agreement, not only though reducing anthropogenic emissions but also for providing anthropogenic sinks that can contribute to *"...a balance between anthropogenic emissions by sources and removals by sinks of greenhouse gases in the second half of this century ..."* (Paris Agreement, Article 4). The balance applies globally, and relates only to greenhouse gases, not aerosols (Section 2.4) or biophysical effects (Section 2.5).

The Paris Agreement includes an Enhanced Transparency Framework, to track countries' progress towards achieving their individual targets (i.e., NDCs), and a Global Stocktake (every five years starting in 2023), to assess the countries' collective progress towards the long-term goals of the Paris Agreement. The importance of robust and transparent definitions and methods (including the approach to separating anthropogenic from natural fluxes) (Fuglestvedt et al. 2018) and the needs for reconciling country GHG inventories and models (Grassi et al. 2018a) was highlighted in 2.3 in relation to estimating emissions. Issues around estimating mitigation is also key to transparency and credibility and is part of the Paris Rulebook.

The land sector is expected to deliver up to 25% of GHG mitigation pledged by countries by 2025-2030 in their NDCs, based on early assessments of "Intended" NDCs submitted ahead of the Paris Agreement and updates immediately after *(low confidence)* (Grassi et al. 2017; Forsell et al. 2016). While most NDCs submitted to date include commitments related to the land sector, they vary with how much information is given and the type of target, with more ambitious targets for developing countries often being "conditional" on support and climate finance. Some do not specify the role of AFOLU but include it implicitly as part of economy-wide pledges (e.g. reducing total emission or emission intensity), a few mention multi-sectoral mitigation targets which include AFOLU in a fairly unspecified manner. Many NDCs include specific AFOLU response options, with most focused on the role of forests, a few included soil carbon sequestration or agricultural mitigation, few explicitly mention bioenergy (e.g., Cambodia, Indonesia and Malaysia), but this could be implicitly included with reduced emission in energy sector through fuel substitution (see Cross-Chapter Box 7: Bioenergy and BECCS in mitigation scenarios, Chapter 6, for discussion on cross sector flux reporting). The countries indicating AFOLU mitigation most prominently were Brazil and Indonesia, followed by other countries focusing either on avoiding carbon emissions (e.g., Ethiopia, Gabon, Mexico, DRC, Guyana and Madagascar) or on promoting the sink through large afforestation programs (e.g., China, India) (Grassi et al. 2017).

Figure 2.28 shows the CO_2 mitigation potential of NDCs compared to historical fluxes from LULUCF[4]. It shows future fluxes based on current policies in place and on country-stated Business As Usual (BAU) activities (these are different from current policies as many countries are already implementing polices that they do not include as part of their historical business-as-usual baseline) (Grassi et al. 2017). Under implementation of unconditional pledges, the net LULUCF flux in 2030 has been estimated to be a sink of -0.41 ± 0.68 $GtCO_2$ yr^{-1}, which increases to -1.14 ± 0.48 $GtCO_2$ yr^{-1} in 2030 with conditional activities. This compares to net LULUCF in 2010 calculated from the GHG Inventories of 0.01 ± 0.86 $GtCO_2$ yr^{-1} (Grassi et al. 2017). Forsell et al. (2016) similarly find a reduction in 2030 compared to 2010 of 0.5 $GtCO_2$ yr^{-1} (range: 0.2–0.8) by 2020 and 0.9 $GtCO_2$ yr^{-1} (range: 0.5–1.3) by 2030 for unconditional and conditional cases.

The approach of countries to calculating the LULUCF contribution towards the NDC varies, with implications for comparability and transparency. For example, by following the different approaches used to include LULUCF in country NDCs, (Grassi et al. 2017) found a 3-fold difference in estimated mitigation: 1.2–1.9 $GtCO_2$-eq yr^{-1} when 2030 expected emissions are compared to 2005 emissions; 0.7–1.4 $GtCO_2$-eq yr^{-1} when 2030 emissions are compared to reference scenarios based on current policies or 2.3–3.0 $GtCO_2$-eq yr^{-1} when compared to BAU, and 3.0–3.8 $GtCO_2$-eq yr^{-1} when based on using each countries' approach to calculation stated in the NDC (i.e., when based on a mix of country approaches, using either past years or BAU projections as reference)

[4] FOOTNOTE: CO_2 fluxes due to Land Use, Land-Use Change and Forestry, i.e. not including the part of AFOLU fluxes that are from agriculture

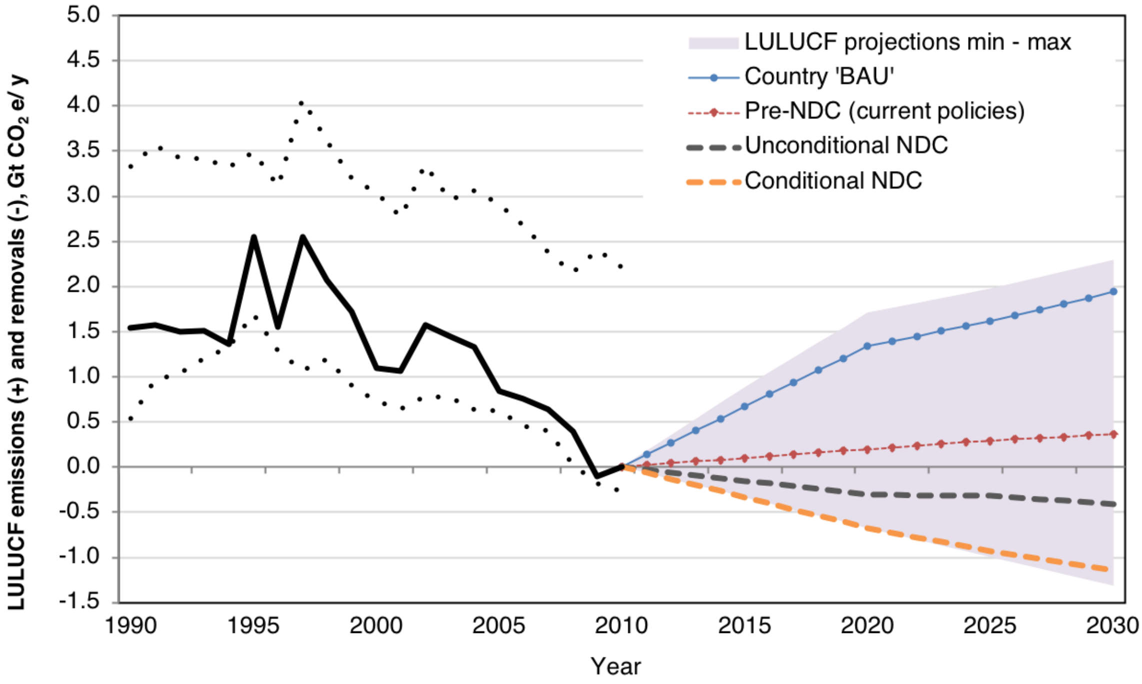


Figure 2.28 Global Land Use, Land Use Change and Forestry (LULUCF) net greenhouse gas flux for the historical period and future scenarios based on analyses of countries' Nationally Determined Contributions NDCs. The LULUCF historical data (black solid line) reflect the following countries' documents (in order of priority): data submitted to UNFCCC (NDCs[5], 2015 GHG Inventories[6], recent National Communications[7],[8]); other official countries' documents; FAO-based datasets, i.e. FAO-FRA for forest (Tian et al. 2015) as elaborated by (Federici et al. 2015) and FAOSTAT for non-forest land use emissions (FAO 2015a) . The future four scenarios reflect official countries' information, mostly INDCs or updated NDCs available at the time of the analysis (Feb 2016), complemented by Biennial Update Reports[9] and National Communications, and show: the Business as Usual (BAU) scenario as defined by the country; the trend based on pre-NDC levels of activity (current policies in place in countries); and the unconditional NDC and conditional NDC scenarios. The shaded area indicates the full range of countries' available projections (min-max), expressing the available countries' information on uncertainties beyond the specific scenarios shown. The range of historical country datasets (dotted lines) reflects differences between alternative selections of country sources, i.e. GHG inventories for developed countries complemented by FAO-based datasets (upper range) or by data in National Communications (lower range) for developing countries

In Exploring the effectiveness of the NDCs, SR15 concluded "*Estimates of global average temperature increase are 2.9–3.4°C above preindustrial levels with a greater than 66% probability by 2100* (Roberts et al. 2006; Rogelj et al. 2016), *under a full implementation of unconditional NDCs and a continuation of climate action similar to that of the NDCs.* In order to achieve 1.5°C or 2°C pathways, this shortfall would imply the need for submission (and achievement) of more ambitious NDCs, and plan for a more rapid transformation of their national energy, industry, transport, and land use sectors (Peters and Geden 2017; Millar et al. 2017; Rogelj et al. 2016).

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[5] FOOTNOTE: UNFCCC. INDCs as communicated by Parties, http://www4.unfccc.int/submissions/indc/Submission%20Pages/submissions.aspx. (UNFCCC, 2015).
[6] FOOTNOTE: UNFCCC. Greenhouse Gas Inventories, http://unfccc.int/national_reports/annex_i_ghg_inventories/national_inventories_submissions/items/8812.php. (UNFCCC, 2015).
[7] FOOTNOTE : UNFCCC. National Communications Non-Annex 1, http://unfccc.int/nationalreports/non-annexinatcom/submittednatcom/items/653.php (UNFCCC, 2015).
[8] FOOTNOTE : UNFCCC. National Communications Annex 1, http://unfccc.int/nationalreports/annexinatcom/submittednatcom/items/7742.php; (UNFCCC, 2015).
[9] FOOTNOTE : UNFCCC. Biennial Update Reports, http://unfccc.int/national_reports/non-annex_i_natcom/reporting_on_climate_change/items/8722.php (UNFCCC, 2015).

Response options relying on the use of land could provide around a third of the additional mitigation needed in the near term (2030) to close the gap between current policy trajectories based on NDCs and what is required to achieve a 2°C (>66% chance) or 1.5°C (50 to 66% chance) pathway according to the UNEP Emissions Gap Report (Roberts et al. 2006). The report estimates annual reduction potentials in 2030 from agriculture 3.0 (2.3–3.7) $GtCO_2$-eq yr^{-1}, a combination of "uncertain measures" (biochar, peat-related emission reductions, and demand-side management) 3.7 (2.6-4.8) $GtCO_2$-eq yr^{-}1; forests 5.3 (4.1–6.5) $GtCO_2$-eq yr^{-1}, bioenergy 0.9 $GtCO_2$-eq yr^{-1}, and BECCS 0.3 (0.2 to 0.4) $GtCO_2$-eq yr^{-1}((UNEP 2017) Table 4.1). These response options account for 35% of potential reduction (or 32% without bioenergy and BECCS) out of a total (all sector) potential of 38 (35–41) $GtCO_2$-eq yr^{-1}. The potentials estimated in the UNEP Emissions Gap Report are based on the technical potential of individual response options from literature including that presented in Section 2.1. CDR related to land use, while not a substitute for strong action in the energy sector, has the technical potential to balance unavoidable emissions that are difficult to eliminate with current technologies (*high confidence*), with early action avoiding deeper and more rapid action later (*very high confidence*) (Strefler et al. 2018; Elmar et al. 2018, SR15).

2.7 Plant and soil processes underlying land-climate interactions

Projecting future complex interactions between land and climate require Earth system models (ESMs). A growing number of studies suggested that many processes important for interactions between land and climate were missing in the CMIP5-class ESMs and that Dynamic vegetation models (DGVM) used tended to elevate CO_2 emission and removals (*high confidence*) (Busch and Sage 2017; Rogers et al. 2017; Anderegg et al. 2016; Tjoelker 2018; Sulman et al. 2014a; Wieder et al. 2018; Davidson et al. 2006a).

Ecosystem complexity stemming from the diversity of plants, animals and microbes, as well as their biological responses to gradual climate changes (e.g., adaptive migration) and disturbance events (e.g., extreme weather events, fire, pest outbreaks; Section 2.2), are of potential importance. Of these processes, this section focuses on plant and soil processes, as recent empirical work, including those explained in the following subsections, offer potential for improved model projections under warmer and CO_2 rich futures.

The magnitude of future uptake and release of CO_2 and other greenhouse gases by vegetation are among the greatest uncertainties (Ciais et al. 2013b). One reason for this uncertainty stems from the lack of understanding of the mechanisms responsible for plant responses to increasing temperatures. The short- and long-term projections of gross photosynthesis responses to changes in temperature, CO_2, nutrient availability vary greatly among the models (Busch and Sage 2017; Rogers et al. 2017). Net CO_2 exchange requires estimation of autotrophic respiration, which is another source of uncertainty in ESM projections (Malhi et al. 2011). The importance of plant acclimation of photosynthesis and respiration in understanding vegetation response to climate change is now widely recognised (*high confidence*) (Rogers et al., 2017; Tan et al., 2017; Tjoelker, 2018; Vanderwel et al., 2015 ; Section 2.7.1). Acclimation is broadly defined as the biochemical, physiological, morphological or developmental adjustments within the lifetime of organisms that result in improved performance at the new condition. Acclimation often operates over a time span of days to weeks, and can mitigate negative effects of climate change on organismal growth and ecosystem functions (Tjoelker 2018).

Soil carbon and microbial processes, which interact with plant responses to climate, represent another large source of uncertainty in model projections (*medium confidence*) (Sections 2.7.2, 2.7.3 and 2.7.4). Given the wide range of uncertainty associated with SOC size estimates, CMIP5 models use a wide range of starting SOC stocks from 510 to 3040 GtC (Todd-Brown et al. 2013). Soil microbial respiration is estimated to release 40–70 GtC annually from the soil to the atmosphere globally (Hawkes et al. 2017). Projections of changes in global SOC stocks during the 21st century by CMIP5 models also ranged widely, from a loss of 37 Gt to a gain of 146 Gt, with differences largely explained by initial SOC stocks, differing C input rates, and different decomposition rates and temperature sensitivities (Todd-Brown et al. 2013). With respect to land-climate interactions, the key processes affecting SOC stocks are warming (which is expected to accelerate SOC losses through microbial respiration) and acceleration of plant growth (which increases inputs of C to soils). However, complex mechanisms underlying SOC responses to moisture regimes, carbon addition, and warming drive considerable uncertainty in projections of future changes in SOC stocks (Sulman et al. 2014a; Singh et al. 2010; Wieder et al. 2018).

2.7.1 Temperature responses of plant and ecosystem production

Climate-change responses of net ecosystem production cannot be modelled by simple instantaneous response functions, because of thermal acclimation responses of plants and soil microbes, as well as delayed responses arising from interactions between plants and the soil (*high confidence*) (Slot et al. 2014; Rogers et al. 2017; Tan et al. 2017; Tjoelker 2018). Photosynthesis and respiration of component plant species exhibit different functional shapes among species (Slot et al. 2014), and carbon balance at the stand level is influenced by respiration of ecosystem biomass other than plants. Large uncertainty remains for thermal responses of bacteria and other soil organisms (Section 2.7.5). Bayesian statistical estimates of global photosynthesis and total ecosystem respirations suggest that they exhibit different responses to thermal anomaly during the last 35 years (Li et al. 2018b).

Thermal responses of plant respiration, which consumes approximately one half of GPP, have not been appropriately incorporated in most ESMs (Davidson et al., 2006; Tjoelker, 2018). Assumptions associated with respiration have been a major source of uncertainty for ESMs at the time of AR5. In most existing models, a simple assumption that respiration doubles with each 10°C increase of temperature (i.e., $Q_{10} = 2$) is adopted, ignoring acclimation. Even a small error stemming from this assumption can strongly influence estimated net carbon balance at large spatial scales of ecosystems and biomes over the time period of multiple decades (Smith and Dukes 2013; Smith et al. 2016b). In order to estimate more appropriate thermal response curves of respiration, a global database including data from 899 plant species has been compiled (Atkin et al. 2015), and respiration data from 231 plants species across seven biomes have been analysed (Heskel et al. 2016). These empirical data on thermal responses of respiration demonstrate a globally convergent pattern (Huntingford et al. 2017). According to a sensitivity analysis of a relatively small number of ESMs, a newly derived function of instantaneous responses of plant respiration to temperature (instead of a traditional exponential function of $Q_{10} = 2$) makes a significant difference in estimated autotrophic respiration especially in cold biomes (Heskel et al. 2016).

Acclimation results in reduced sensitivity of plant respiration with rising temperature, i.e., down regulation of warming-related increase in respiratory carbon emission (Atkin et al. 2015; Slot and Kitajima 2015; Tjoelker 2018) (*high confidence*). For example, experimental data from a tropical forest canopy show that temperature acclimation ameliorates the negative effects of rising temperature to leaf and plant carbon balance (Slot et al. 2014). Analysis of CO_2 flux data to quantify optimal temperature of net primary production of tropical forests also suggest acclimation potential for many tropical forests (Tan et al. 2017). Comparisons of models with and without thermal acclimation of respiration show that acclimation can halve the increase of plant respiration with projected temperature increase by the end of 21st century (Vanderwel et al. 2015).

It is typical that acclimation response to warming results in increases of the optimum temperature for photosynthesis and growth (Slot and Winter 2017; Yamori et al. 2014; Rogers et al. 2017). Although such shift is a result of a complex interactions of biochemical, respiratory, and stomatal regulation (Lloyd and Farquhar 2008), it can be approximated by a simple algorithm to address acclimation (Kattge et al. 2007). Mercado et al., (2018), using this approach, found that inclusion of biogeographical variation in photosynthetic temperature response was critically important for estimating future land surface carbon uptake. In the tropics, CO_2 fertilisation effect (c.f., Box 2.3) is suggested to be more important for observed increases in carbon sink strength than increased leaf area index or longer growing season (Zhu et al. 2016). Acclimation responses of photosynthesis and growth to simultaneous changes of temperature and CO_2, as well as stress responses above the optimal temperature for photosynthesis, remain a major knowledge gap in modelling responses of plant productivity under future climate change (Rogers et al. 2017).

2.7.2 Water transport through soil-plant-atmosphere continuum and drought mortality

How climate change, especially changes of precipitation patterns, influence water transport through the soil-plant-atmosphere continuum, is a key element in projecting the future of water vapour flux from land and cooling via latent heat flux (Sellers et al. 1996; Bonan 2008a; Brodribb 2009; Choat et al. 2012; Sperry and Love 2015; Novick et al. 2016; Sulman et al. 2016)(*high confidence*). Even without changes in leaf area per unit area of land, when plants close stomata in response to water shortage, dry atmosphere, or soil moisture deficit, the stand-level fluxes of water (and associated latent heat flux) decrease (Seneviratne et al. 2018). Closing stomata enhances drought survival at the cost of reduced photosynthetic production, while not closing stomata avoids loss of photosynthetic production at the cost of increased drought mortality (Sperry and Love 2015). Hence, species-specific responses to drought, in terms of whether they close stomata or not, have short and long-term consequences (Anderegg et al. 2018a; Buotte et al. 2019). Increased drought-induced mortality of forest trees, often exacerbated by insect outbreak and fire (e.g., (Breshears et al. 2005; Kurz et al. 2008a; Allen et al. 2010)) (Section 2.2.4), have long-term impact on hydrological interactions between land and atmosphere (Anderegg et al. 2018b).

New models linking plant water transport with canopy gas exchange and energy fluxes are expected to improved projections of climate change impacts on forests and land-atmosphere interactions (Bohrer et al., 2005; Anderegg et al., 2016; Sperry and Love, 2015; Wolf et al., 2016)(*medium confidence*). Yet, there is much uncertainty in the ability of current vegetation and land surface models to adequately capture tree mortality and the response of forests to climate extremes like drought (Rogers et al. 2017; Hartmann et al. 2018). Most vegetation models use climate stress envelopes or vegetation carbon balance estimations to project climate-driven mortality and loss of forests (McDowell et al. 2011); these may not adequately project biome shifts and impacts of disturbance in future climates. For example, a suite of vegetation models was compared to a field drought experiment in the Amazon on mature rainforest trees and all models performed poorly in projecting the timing and magnitude of biomass loss due to drought (Powell et al. 2013). More recently, the loss of water transport due to embolism (disruption of xylem water continuity) (Sperry and Love 2015), rather than carbon starvation (Rowland et al. 2015), is receiving attention as a key physiological process relevant for drought-induced tree mortality (Hartmann et al. 2018). A key challenge to modelling effort is to consider differences among plant species and vegetation types in their drought responses. One approach is to classify plant species to "functional types" that exhibit similar responses to environmental variations (Anderegg et al. 2016). Certain traits of species, such as tree height, is shown to be predictive of growth decline and mortality in response to drought (Xu et al. 2016a). Similarly, tree rooting depth is positively related to mortality, contrary to expectation, during prolonged droughts in tropical dry forest (Chitra-Tarak et al. 2017).

2.7.3 Soil microbial effects on soil nutrient dynamics and plant responses to elevated CO_2

Soil microbial processes influencing nutrient and carbon dynamics represent a large source of uncertainty in projecting land-climate interactions. For example, ESMs incorporating nitrogen and phosphorus limitations (but without considering the effects of mycorrhizae and rhizosphere priming) indicate that the simulated future C-uptake on land is reduced significantly when both nitrogen and phosphorus are limited as compared to only C-stimulation, by 63% (of 197 Pg C) under RCP2.6 and by 67% (of 425 Pg C) under RCP8.5 (Zhang et al. 2013c). Mineral nutrient limitation progressively reduces the CO_2 fertilisation effects on plant growth and productivity over time (Norby et al. 2010; Sardans et al. 2012; Reich and Hobbie 2013; Feng et al. 2015; Terrer et al. 2017) (*robust evidence, medium agreement*). The rates at which nutrient limitation develops differ among studies and sites. A recent meta-analysis shows that experimental CO_2 enrichment generally results in lower nitrogen and phosphorus concentrations in plant tissues (Du et al. 2019a), and isotopic analysis also suggest a global trend of decreases in leaf nutrient concentration (Craine et al. 2018; Jonard et al. 2015). However, reduced responses to elevated CO_2 (eCO_2) may not be a simple function of nitrogen dilution per se, as they result from complex interactions of ecosystem factors that influence nitrogen acquisition by plants (Liang et al. 2016; Rutting 2017; Du et al. 2019a).

Increasing number of case studies suggest that soil microbial processes, such as nitrogen mineralisation rates and symbiosis with plants, influence nutrient limitation on eCO_2 effects on plant growth (Drake et al. 2011; Zak et al. 2011; Hungate et al. 2013; Talhelm et al. 2014; Du et al. 2019a) (*medium confidence*). Rhizosphere priming effects (i.e., release of organic matters by roots to stimulate microbial activities) and mycorrhizal associations are proposed to explain why some sites becoming nitrogen limited after a few years and others sustaining growth through accelerated nitrogen uptake (Phillips et al. 2011; Terrer et al. 2017) (*limited evidence, medium agreement*).

Model assessments that including rhizosphere priming effects and ectomycorrhizal symbiosis suggest that soil organic matter (SOM) cycling is accelerated through microbial symbiosis (Elbert et al. 2012; Sulman et al. 2017; Orwin et al. 2011; Baskaran et al. 2017) (*medium confidence*). Uncertainty exists in differences among ectomycorrhizal fungal species in their ability to decompose SOM (Pellitier and Zak 2018) and the capacity of ecosystems to sustain long-term growth with these positive symbiotic feedbacks is still under debate (Terrer et al. 2017). ESMs include only biological nitrogen cycles, even though a recent study suggests that bedrock weathering can be a significant source of nitrogen to plants (Houlton et al. 2018). In contrast, rock weathering is widely considered to be the key for P availability, and tropical forests with highly weathered soils are considered to be limited by P availability rather than nitrogen availability (Reed et al. 2015). Yet, evidence from P-fertilisation experiments is lacking (Schulte-Uebbing and de Vries 2018) and P limitation of tropical tree growth may be strongly species-specific (Ellsworth et al. 2017; Turner et al. 2018a). Limitation by availability of soil nutrients other than nitrogen and P has not been studied in the context of land-climate interactions, except potassium (K) as a potentially limiting factor for terrestrial plant productivity in interaction with N, P and hydrology (Sardans and Peñuelas 2015; Zhao et al. 2017; Wright et al. 2018).

Anthropogenic alteration of global and regional nitrogen and P cycles, largely through use of chemical fertilisers and pollution, has major implications for future ecosystem attributes, including C storage, in natural and managed ecosystems (Peñuelas et al. 2013, 2017; Wang et al. 2017c; Schulte-Uebbing and de Vries 2018; Yuan et al. 2018) *(high confidence).* During 1997-2013, the contribution of nitrogen deposition to the global C sink has been estimated at 0.27 (± 0.13) GtC yr^{-1}, and the contribution of P deposition as 0.054 (± 0.10) GtC yr^{-1}; these constitute about 9% and 2% of the total land C sink, respectively (Wang et al. 2017c). Anthropogenic deposition of nitrogen enhances carbon sequestration by vegetation (Schulte-Uebbing and de Vries 2018), but this effect of nitrogen deposition on carbon sequestration may be offset by increased emission of GHGs such as N_2O and CH_4 (Liu and Greaver 2009). Furthermore, nitrogen deposition may lead to imbalance of nitrogen *vs.* phosphorus availability (Peñuelas et al. 2013), soil microbial activity and SOM decomposition (Janssens et al. 2010) , and reduced ecosystem stability (Chen et al. 2016b).

2.7.4 Vertical distribution of soil organic carbon

It has long been recognised that dynamics of soil organic carbon (SOC) represent a large source of uncertainties on biogeochemical interactions of land with atmosphere and climate as detailed below. Since AR5, there have been new understanding on SOC size, as well as microbial processes that influence SOM dynamics under climate change and LULCC. Three existing data bases (SoilGrids, the Harmonized World Soil Data Base, Northern Circumpolar Soil Database) substantially differ in estimated size of global soil organic carbon (SOC) stock down to 1 m depth, varying between 2500 Pg to 3400 Pg with differences among databases largely attributable to C stored in permafrost (Joosten 2015; Köchy et al. 2015; Tifafi et al. 2018). These values are four to eight times larger than the carbon stock associated with the terrestrial vegetation (Bond-Lamberty et al. 2018). New estimates since AR5 show that much larger areas in Amazon and Congo basins are peatlands (Gumbricht et al. 2017; Dargie et al. 2019).

Deep soil layers can contain much more carbon than previously assumed (e.g., González-Jaramillo et al., 2016) (*limited evidence, medium agreement*). Based on radiocarbon measurements, deep SOC can be very old, with residence times up to several thousand years (Rumpel and Kögel-Knabner 2011) or even several tens of thousands of years (Okuno and Nakamura 2003). Dynamics associated with such deeply buried carbon remain poorly studied and ignored by the models, and not addressed in most of the studies assessed in this subsection. Deep soil C is thought to be stabilised by mineral interactions, but recent experiments suggest that CO_2 release from deep soils can also be increased by warming, with a 4˚C warming enhancing annual soil respiration by 34–37% (Hicks Pries et al. 2017) or addition of fresh carbon (Sebastien Fontaine Pierre Barre, Nadia Bdioui, Bruno Mary, Cornelia Rumpel 2007). While erosion is not typically modelled as a carbon flux in ESMs, erosion and burial of carbon-containing sediments is likely a significant carbon transfer from land to ocean (Berhe et al. 2007; Asefaw et al. 2008; Wang et al. 2017e) (*medium confidence).*

2.7.5 Soil carbon responses to warming and changes in soil moisture

Annually, 119 GtC is estimated to be emitted from the terrestrial ecosystem to the atmosphere, of which ca. 50% is attributed to soil microbial respiration (Auffret et al. 2016; Shao et al. 2013). It is yet not possible yet to make mechanistic and quantitative projections about how multiple environmental factors influence soil microbial respiration (Davidson et al. 2006a; Dungait et al. 2012). Soil warming experiments show significant variability in temperature and moisture responses across biomes and climates; Crowther et al., (2016) found that warming-induced SOC loss is greater in regions with high initial carbon stocks, while an analysis of an expanded version of the same dataset did not support this conclusion (Gestel et al. 2018). Studies of SOC responses to warming over time have also shown complex responses. In a multi-decadal warming experiment, Melillo et al., (2017) found that soil respiration response to warming went through multiple phases of increasing and decreasing strength, which were related to changes in microbial communities and available substrates over time. (Conant et al., (2011) and Knorr et al., (2005) suggested that transient decomposition responses to warming could be explained by depletion of labile substrates, but that long-term SOC losses could be amplified by high temperature sensitivity of slowly decomposing SOC components. Overall, long-term SOC responses to warming remain uncertain (Davidson et al. 2006a; Dungait et al. 2012; Nishina et al. 2014; Tian et al. 2015).

It is widely known that soil moisture plays an important role in SOM decomposition by influencing microbial processes (e.g., Monard et al., 2012; Moyano et al., 2013; Yan et al., 2018), as confirmed by a recent global meta-analysis (Hawkes et al. 2017) (*high confidence*). A likely mechanism involves that increased soil moisture lowers C mineralisation rates under anaerobic conditions resulting in enhanced C stocks, but experimental analyses have shown that this effect may last for only 3–4 weeks after which iron reduction can actually accelerate loss of previously protected organic C by facilitating microbial access (Huang and Hall 2017).

Experimental studies of responses of microbial respiration to warming have found variable results (Luo et al. 2001; Bradford et al. 2008; Zhou et al. 2011; Carey et al. 2016; Teramoto et al. 2016). No acclimation was observed in C-rich calcareous temperate forest soils (Schindlbacher et al. 2015) and arctic soils (Hartley et al. 2008), and a variety of ecosystems from the Arctic to the Amazon indicated that microbes appear to enhance the temperature sensitivity of soil respiration in Arctic and boreal soils, thereby releasing even more carbon than currently projected (Karhu et al. 2014). In tropical forests, P limitation of microbial processes is a key factor influencing soil respiration (Camenzind et al. 2018). Temperature responses of symbiotic mycorrhizae differ widely among host plant species, without a clear pattern that may allow generalisation across plant species and vegetation types (Fahey et al. 2016).

Some new insights have been obtained since AR5 from investigations of improved mechanistic understanding of factors that regulate temperature responses of soil microbial respiration. Carbon use efficiency and soil nitrogen dynamics have large influence on SOC responses to warming (Allison et al. 2010; Frey et al. 2013; Wieder, William R., Bonan, Gordon B., Allison 2013; García-Palacios et al. 2015) (*high confidence*). More complex community interactions including competitive and trophic interactions could drive unexpected responses to SOC cycling to changes in temperature, moisture, and C inputs (Crowther et al. 2015; Buchkowski et al. 2017). Competition for nitrogen among bacteria and fungi could also suppress decomposition (Averill et al. 2014). Overall, the roles of soil microbial community and trophic dynamics in global SOC cycling remain very uncertain.

2.7.6 Soil carbon responses to changes in organic-matter inputs by plants

While current ESM structures mean that increasing C inputs to soils drive corresponding increases in SOC stocks, long-term carbon addition experiments have found contradictory SOC responses. Some litter addition experiments have observed increased SOC accumulation (Lajtha et al. 2014b; Liu et al. 2009), while others suggest insignificant SOC responses (Lajtha et al. 2014a; van Groenigen et al. 2014). Microbial dynamics are believed to have an important role in driving complex responses to C additions. The addition of fresh organic material can accelerate microbial growth and SOM decomposition via priming effects (Kuzyakov et al. 2014; Cheng et al. 2017). SOM cycling is dominated by “hot spots” including the rhizosphere as well as areas surrounding fresh detritus (*medium evidence*, *high agreement*) (Finzi et al. 2015; Kuzyakov and Blagodatskaya 2015). This complicates projections of SOC responses to increasing plant productivity; increasing C inputs could promote higher SOC storage, but these fresh C inputs could also deplete SOC stocks by promoting faster decomposition (Hopkins et al. 2014; Guenet et al. 2018; Sulman et al. 2014b). A meta-analysis by (van Groenigen et al. 2014) suggested that elevated CO_2 accelerated SOC turnover rates across several biomes. These effects could be especially important in high-latitude regions where soils have high organic matter content and plant productivity is increasing (Hartley et al. 2012), but have also been observed in the tropics (Sayer et al. 2011).

Along with biological decomposition, another source of uncertainty in projecting responses of SOC to climate change is stabilisation via interactions with mineral particles (Kögel-Knabner et al. 2008; Kleber et al. 2011; Marschner et al. 2008; Schmidt 2011) (*high confidence*). Historically, conceptual models of SOC cycling have centred on the role of chemical recalcitrance, the hypothesis that long-lived components of SOC are formed from organic compounds that are inherently resistant to decomposition. Under the emerging new paradigm, stable SOC is primarily formed by the bonding of microbially-processed organic material to mineral particles, which limits the accessibility of organic material to microbial decomposers (Lützow et al. 2006; Keiluweit et al. 2015; Kallenbach et al. 2016; Kleber et al. 2011; Hopkins et al. 2014). SOC in soil aggregates can be protected from microbial decomposition by being trapped in soil pores too small for microbes to access (Blanco-Canqui and Lal 2004; Six et al. 2004) or by oxygen limitation (Keiluweit et al. 2016). Some new models are integrating these mineral protection processes into SOC cycling projections (Wang et al. 2017a; Sulman et al. 2014b; Riley et al. 2014; Wieder et al. 2015), although the sensitivity of mineral-associated organic matter to changes in temperature, moisture, fire (see Box 2.1) and carbon inputs is highly uncertain. Improved quantitative understanding of soil ecosystem processes will be critically important for projection of future land-climate feedback interactions.

Frequently Asked Questions

FAQ 2.1 How does climate change affect land use and land cover?

Contemporary land cover and land use is adapted to current climate variability within particular temperature and/or rainfall ranges (referred to as climate envelopes). Anthropogenic greenhouse gas emissions impact land through changes in the weather and climate and also through modifications in atmospheric composition through increased greenhouse gasses, especially CO_2. A warming climate alters the current regional climate variability and results in a shift of regional climate envelopes poleward and to higher elevations. The shift of warmer climate envelopes into high latitude areas has potential benefits for agriculture here through extended growing seasons and warmer seasonal temperatures and increased atmospheric CO_2 concentrations enhances photosynthetic activity. However, this warming will also lead to enhanced snowmelt and reduced albedo, permafrost melting and the further release of methane and CO_2 into the atmosphere as the permafrost begins to decompose. Concurrent with these climate envelope shifts will be the emergence of new, hot climates in the tropics and increases in the frequency, intensity and duration of extreme events (e.g. heat waves, very heavy rainfall, drought). These emergent hot climates will negatively affect land use (through changes in crop productivity, irrigation needs, management practices) and land cover through loss of vegetation productivity in many parts of the world, and overwhelm any benefits to land use and land cover derived from increased atmospheric CO_2 concentrations.

FAQ 2.2 How do the land and land use contribute to climate change?

Any changes to the land and how it is used can affect exchanges of water, energy, greenhouse gases (e.g., CO_2, CH_4, N_2O), non-greenhouse gases (e.g., biogenic volatile organic compounds – BVOCs), and aerosols (mineral, e.g., dust, or, carbonaceous, e.g., black carbon) between the land and the atmosphere. Land and land use change therefore alter the state (e.g., chemical composition and air quality, temperature and humidity) and the dynamics (e.g., strength of horizontal and vertical winds) of the atmosphere, which in turn can dampen or amplify local climate change. Land-induced changes in energy, moisture and wind can affect neighbouring, and sometimes more distant, areas. For example, deforestation in Brazil warms the surface, in addition to global warming, and enhances convection which increases the relative temperature difference between the land and the ocean, boosting moisture advection from the ocean and thus rainfall further inland. Vegetation absorbs carbon dioxide (CO_2) to use for growth and maintenance. Forests contain more carbon in their biomass and soils than croplands and so a conversion of forest to cropland, for example, results in emissions of CO_2 to the atmosphere, thereby enhancing the greenhouse gas-induced global warming. Terrestrial ecosystems are both sources and sinks of chemical compounds such as nitrogen and ozone. BVOCs contribute to forming tropospheric ozone and secondary aerosols, which respectively affect surface warming and cloud formation. Semi-arid and arid regions release dust, as do cropland areas after harvest. Increasing the amount of aerosols in the atmosphere impacts temperature in both positive and negative ways depending on the particle size, altitude, and nature (carbonaceous or mineral for example). Although global warming will impact the functioning and state of the land (see FAQ 2.**1),** this is not a one-way interaction as changes in land and land use can also affect climate and thus modulate climate change. Understanding this two-way interaction can help improve adaptation and mitigation strategies as well as manage landscapes.

FAQ 2.3 How does climate change affect water resources?

Renewable freshwater resources are essential for the survival of terrestrial and aquatic ecosystems and human use in agriculture, industry and domestically. As increased water vapour concentrations are expected in a warmer atmosphere, climate change will alter the hydrological cycle and therefore regional freshwater resources. In general, wet regions are projected to get wetter and dry regions drier, although there are regional exceptions to this. The consequent impacts vary regionally; where rainfall is projected to be lower in the future (many arid subtropical regions and those with a Mediterranean climate), a reduction of water resources is expected. Here increased temperatures and decreased rainfall will reduce surface and groundwater resources, increase plant evapotranspiration and increase evaporation rates from open water (rivers, lakes, wetlands) and water supply infrastructure (canals, reservoirs). In regions where rainfall is projected to be higher in the future (many high latitude regions and the wet tropics), an increase in water resources can be expected to benefit terrestrial and freshwater ecosystems, agriculture and domestic use, however, these benefits may be limited due to increased temperatures. An increase in extreme rainfall events is also expected which will lead to increases in surface runoff, regional flooding and nutrient removal as well as a reduction in soil water and groundwater recharge in many places. Anthropogenic land use change may amplify or moderate the climate change effect on water resources therefore informed land management strategies need to be developed. A warming climate will exacerbate the existing pressures on renewable freshwater resources in water-stressed regions of the Earth and result in increased competition for water between human and natural systems.

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Appendix

This appendix provides all numbers that support **Figure 2.14** and **Figure 2.17** located in Section 2.5 It lists all model-based studies, with their references, that have been used to create the figures. Studies that examine the effects of historical and future scenarios of changes in anthropogenic land cover are presented in Table 1. The responses to idealised latitudinal deforestation and forestation can be found in Table A2.2.

The biophysical effects of changes in anthropogenic land cover reflect the impacts of changes in physical land surface characteristics such as albedo, evapotranspiration, and roughness length. The biogeochemical effects reflect changes in atmospheric CO_2 composition resulting from anthropogenic changes in land cover.

The biogeochemical effects are estimated using three different methods:

1. Directly calculated within global climate models (Tables A2.1 and A2.2);
2. Calculated from off-line dynamic global vegetation models (DGVMs) estimates of net changes in the emissions of CO_2 from land (Table A2.1);
3. Calculated from observation-based estimates of net changes in the emissions of CO_2 from land (for historical reconstruction only, Table A2.1)

The mean annual and global temperature change (ΔT) resulting from biogeochemical effects is calculated as follows, for both DGMVs and observation-based estimates:

$$\Delta T = \Delta LCO_2 * TCRE$$

Where ΔLCO_2 is the cumulative changes in net emissions of CO_2 resulting from anthropogenic land cover changes during the time period considered (in Tera tons of carbon, TtC), and TCRE is the transient climate response to cumulative carbon emissions (Gillett et al. 2013; Matthews et al. 2009). TCRE is a measure of the global temperature response to cumulative emissions of CO_2 and has been identified as a useful and practical tool for evaluating CO_2-induced climate changes (expressed in °C per Tera tons of Carbon, °C/TtC). TCRE values have been estimated for a range of Earth system models (Gillett et al. 2013), (MacDougall et al. 2016). In the following, we use the 5th percentile, mean, and 95th percentile derived from the range of available TCRE values. For each DGVM or observation-based estimate, we then calculate three potential temperature change to bracket the range of climate sensitivities.

Table A2.1: Model-based and observation-based estimates of the effects historical and future anthropogenic land cover changes have on mean annual global surface air temperature (°C). BGC and BPH correspond to the change in temperature resulting from respectively biogeochemical processes (e.g., changes in atmospheric CO2 composition) and biophysical processes (e.g., changes in physical land surface characteristics such as albedo, evapotranspiration, and roughness length).

Reference of the study	Time period	Cumulative CO_2 emissions from anthropogenic land cover change (TtC)	TCRE (°C/TtC)	Change in mean global annual (°C)	
				BGC	BPH
Historical period (global climate models)					
(Lawrence et al. 2018)	1850-2005	0.123	1.9	0.23	
(Simmons and Matthews 2016)	1750-2000[10]			0.22	-0.24
(Devaraju et al. 2016)	1850-2005	0.112	1.9	0.21	
(Zhang et al. 2013a)	1850-2005[11]	0.097	1.75	0.17	-0.06
(Hua and Chen 2013)	~1850-2000 (average of two estimates)				-0.015
(Jones et al. 2013a)	Preindustrial (no exact dates)				-0.57
(Lawrence et al. 2012)	1850-2005	0.120	1.9	0.23	-0.10
(De Noblet-Ducoudré et al. 2012)	1972-2002 relative to 1900-1970				-0.042 ;-0.056 ; -0.005 ; -0.041 ; 0.021 ;-0.007 ; -0.005
(Pongratz et al. 2010)	20th century			0.16, 0.18	-0.03
(Arora and Boer 2010)	1850-2000	0.040,0.077	2.4	0.1, 0.18	
(Strengers et al. 2010)	20th century				-0.06
(Kvalevåg et al. 2010)	Preindustrial (no exact dates)				+0.04 (CASE I)

[10] FOOTNOTE: Land Use Change + Fossil Fuel emission simulation values are considered.
[11] FOOTNOTE: Carbon-Nitrogen-Phosphorous simulation values are considered.

(Findell et al. 2009)	1901-2004				+0.02
(Findell et al. 2007)	1990 relative to potential vegetation				+0.008
(Brovkin et al. 2006)	1700-1992 (5 models)				-0.24 ;-0.13 ;-0.14 ;-0.25 ; -0.17
(Betts et al. 2007; Betts 2001)	1750-1990				-0.02
(Hansen et al. 2005)	1880-1990				-0.04
(Feddema et al. 2005)	Preindustrial land-cover changes (no exact dates, "prehuman" simulations)				-0.39
(Matthews et al. 2004)	1700-2000 (average of 7 simulations)			0.3	-0.14
(Brovkin et al. 2004)	1800-2000			0.18	-0.26
(Zhao and Pitman 2002; Chase et al. 2000, 2001)	Preindustrial				+0.06
(Hansen et al. 1998)	Preindustrial land-cover changes				-0.14
Mean (±Standard deviation) of all studies				**0.2±0.05**	**-0.1±0.14**
Historical period (Dynamic Global Vegetation Models/Bookkeeping model results)					
(Li et al. 2017a)	1901-2012 (median of models)	0.148	0.88-1.72-2.52	0.13-0.25-0.37	
(Peng et al. 2017)	1850-1990 (realistic cases range)	0.087,0.139	0.88-1.72-2.52	0.1-0.15-0.22, 0.12-0.24-0.35	
(Arneth et al. 2017a)	1901-2014[12]	0.089	0.88-1.72-2.52	0.1-0.15-0.22	
		0.210	0.88-1.72-2.52	0.18-0.36-0.53	
		0.179	0.88-	0.16-0.31-	

[12] FOOTNOTE: FLULCC,1 refers to land use change related fluxes accounting for new processes in their study.

			1.72-2.52	0.45	
		0.195	0.88-1.72-2.52	0.17-0.33-0.49	
		0.083	0.88-1.72-2.52	0.1-0.14-0.21	
		0.161	0.88-1.72-2.52	0.14-0.28-0.4	
		0.117	0.88-1.72-2.52	0.1-0.2-0.3	
		0.104	0.88-1.72-2.52	0.1-0.18-0.26	
		0.196	0.88-1.72-2.52	0.17-0.34-0.49	
(Pugh et al. 2015)	1850-2012 (gross land clearance flux)	0.157	0.88-1.72-2.52	0.14-0.27-0.39	
(Hansis et al. 2015)	1850-2012	0.269	0.88-1.72-2.52	0.19-0.36-0.53	
(Houghton et al. 2012a; Hansis et al. 2015)	1920-1999 (multi-model range)	0.072, 0.115	0.88-1.72-2.52	0.1-0.12-0.18, 0.1-0.2-0.3	
Mean (±Standard deviation) of all studies				**0.24±0.12**	
Historical period (Observation-based estimates)					
(Li et al. 2017a)	1901-2012	0.155	0.88-1.72-2.52	0.14-0.27-0.39	
(Li et al. 2017a; Avitabile et al. 2016; Carvalhais et al. 2014)	1901-2012[13]	0.160,0.165	0.88-1.72-2.52	0.14-0.27-0.40,0.14-0.28-0.41	
(Liu et al. 2015; Li et al. 2017a)	1901-2012	0.161,0.163	0.88-1.72-2.52	0.14-0.28-0.41	
(Le Quéré et al. 2015)	1870-2014	0.145	0.88-1.72-2.52	0.13-0.25-0.36	
(Carvalhais et al. 2014; Li et al. 2017a)	1901-2012	0.152,0.159	0.88-1.72-2.52	0.13-0.26-0.38, 0.14-0.27-0.4	
(Pan et al. 2011b; Li et al. 2017a)	1901-2012	0.119,0.122	0.88-1.72-2.52	0.10-0.20-0.30, 0.11-	

[13] FOOTNOTE: Different harmonization methods. Method A assumes increase in cropland area in a grid cell taken from forest. Method C assumes increase in cropland and pasture taken from forest and then natural grassland if no more forest area available.

				0.21-0.31	
Mean (±Standard deviation) of all studies				**0.25±0.10**	
Future -RCP8.5 (global climate models)					
(Tharammal et al. 2018)	2006-2100	0.093	1.9	0.18	
(Lawrence et al. 2018)	2006-2100	0.211	1.9	0.40	
(Simmons and Matthews 2016)	2000-2100	-	-	0.35	-0.34
(Hua et al. 2015)	2006-2100	0.032	2.4	0.08	-
(Davies-Barnard et al. 2014)	2005-2100	0.02	2.1	0.04	-0.015
(Boysen et al. 2014; Quesada et al. 2017a; Brovkin et al. 2013b)	2005-2100	0.034	2.4	0.08	0.04
		0.025	2.1	0.05	0.0
		0.037	1.6	0.06	0.08
		0.062	2.2	0.13	-0.20
		0.205	1.6	0.33	-0.06
(Lawrence et al. 2012)	2006-2100	0.256	1.9	0.49	
Mean (±Standard deviation) of all studies				**0.20±0.15**	**-0.1±0.14**
Future -RCP8.5 (Dynamic Global Vegetation Model results)					
(Pugh et al. 2015)	2006-2100	0.169,0.171	0.88-1.72-2.52	0.15-0.29-0.42,0.15-0.29-0.43	
(IPCC 2014)	2005-2099	0.151	0.88-1.72-2.52	0.13-0.26-0.38	
Mean (±Standard deviation) of all studies				**0.28±0.11**	
Future RCP4.5 (global climate models)					
(Tharammal et al. 2018)	2005-2100	-0.029	1.9	-0.05	
(Lawrence et al. 2018)	2006-2100	0.053	1.9	0.10	
(Simmons and Matthews 2016)	2000-2100			0.37	-0.29
(Davies-Barnard et al. 2014)	2005-2100	-0.040	2.1	-0.08	0.14

(Lawrence et al. 2012)	2006-2100	0.148	1.9	0.28	
Mean (±Standard deviation) of all studies				**0.12±0.17**	**-0.1±0.21**
Future RCP4.5 (Dynamic Global Vegetation Model results)					
(Pugh et al. 2015)	2006-2100	0.016,-0.018	0.88-1.72-2.52	0.01-0.03-0.04,-0.02-(-0.03)-(-0.045)	
(IPCC 2014)	2005-2099	0.027	0.88-1.72-2.52	0.02-0.05-0.07	
Mean (±Standard deviation) of all studies				**0.01±0.04**	
Future RCP2.6 (global climate models)					
(Tharammal et al. 2018)	2005-2100	0.039	1.9	0.07	
(Simmons and Matthews 2016)	2000-2100			0.42	-0.35
(Hua et al. 2015)	2006-2100	0.036	2.4	0.09	
(Davies-Barnard et al. 2014)	2005-2100			0.04	-0.01
(Brovkin et al. 2013b)	2005-2100	0.039	2.4	0.09	
		0.019	2.1	0.04	
		0.065	2.2	0.14	
		0.175	1.6	0.28	
(Lawrence et al. 2012)	2006-2100	0.0154	1.9	0.03	
Mean (±Standard deviation) of all studies				**0.13±0.12**	**-0.18±0.17**
Future RCP2.6 (Dynamic Global Vegetation Model results)					
(Pugh et al. 2015)	2006-2100 (no harvest, managed cases)	0.057,0.084	0.88-1.72-2.52	0.05-0.09-0.14,0.07-0.14-0.21	
(IPCC 2014)	2005-2099	0.105	0.88-1.72-2.52	0.09-0.18-0.26	
Mean (±Standard deviation) of all studies				**0.14±0.06**	

Table A2.2: Model-based estimates of the effects idealised and latitudinal deforestation or forestation have on mean annual global and latitudinal surface air temperature (°C). BGC and BPH correspond to the change in temperature resulting from respectively biogeochemical processes (e.g. changes in atmospheric CO2 composition) and biophysical processes (e.g. changes in physical land surface characteristics such as albedo, evapotranspiration, and roughness length).

Idealised Deforestation/Afforestation (global climate models)					
Reference	Change in forest area (Mkm2)	Cumulative LCC flux (TtC)	TCRE (K/TtC)	Mean annual change in surface air temperature, averaged globally (and for the latitudinal band where trees are removed or added) (°C)	
				BGC	BPH
Tropical Deforestation					
(Devaraju et al. 2018b)	36.1				0.02 (1.14)
(Longobardi et al. 2016b)	23[14]	0.127	1.72	0.30	0.044 (-0.19)
(Devaraju et al. 2015c)	23			1.06	-0.04 (0.20)
(Brovkin et al. 2015)					-0.01,-0.13,-0.05
(Bathiany et al. 2010b)	23.1			0.40	0.18 (0.9)
(Snyder 2010)	23				0.2 (1.0)
(Bala et al. 2007)	23	0.418	1.72	0.72	0.70
(Voldoire 2006)					0.2,0.4,0.6
(Snyder et al. 2004b)	22.7				0.24(1.2)
(Claussen et al. 2001b)	7.5			0.19 (0.15)	-0.04 (0.13)
(Ganopolski et al. 2001c)	7.5				-0.5 (0.5)
(Henderson-Sellers and Gornitz 1984)					0.00
(Potter et al. 1981; POTTER et al. 1975)					-0.2
(Sagan et al.					-0.07

[14] FOOTNOTE: For some studies that do not provide area deforested, IPSL-CM5 model grids used to calculate the area.

1979)					
Mean (±Standard deviation) of all studies				0.53±0.32	0.1±0.27(0.61±0.48)
Tropical Afforestation					
(Wang et al. 2014a) (Average of 4 simulations)					0.925
(Bathiany et al. 2010b)	23.1				-0.03 (-0.1)
Temperate Deforestation					
(Devaraju et al. 2018a)	18.8				0.18 (0.52)
(Longobardi et al. 2016b)	15	0.047	1.72	0.10	-0.077 (-0.22)
(Devaraju et al. 2015b)	15.3			0.39	-0.5 (-0.8)
(Bala et al. 2007)	15	0.231	1.72	0.40	-0.04
(Snyder et al. 2004b)	19.1				-0.22 (-1.1)
Mean (±Standard deviation) of all studies				0.29±0.13	-0.13±0.22 (-0.4±0.62)
Temperate Afforestation					
(Laguë and Swann 2016b)					0.3 (1.5)
(Wang et al. 2014a)					1.14
(Swann et al. 2012b)	15.3			-0.2, -0.7	0.3
(Gibbard et al. 2005)					0.27
Mean (±Standard deviation) of all studies				-0.45	0.50±0.36
Boreal Deforestation					
(Devaraju et al. 2018a)	23.5				-0.25 (-1.2)
(Longobardi et al. 2016b)	13.7	0.050	1.72	0.11	-0.38 (-0.9)

(Devaraju et al. 2015b)	13.7			0.06	-0.9 (-4)
(Dass et al. 2013)	18.5			0.12, 0.32	-0.35
(Bathiany et al. 2010b)	18.5	0.02	2.04	0.04	-0.28 (-1.1)
(Bala et al. 2007)	13.7	0.0105	1.72	0.02	-0.8
(Snyder et al. 2004b)	22.4				-0.77(-2.8)
(Caussen et al. 2001)	6			0.09 (0.12)	-0.23 (-0.82)
(Ganopolski et al. 2001c)	6				-1.0
Mean (±Standard deviation) of all studies				0.11±0.09	-0.55±0.29 (-1.8±1.2)
Boreal Afforestation					
(Bathiany et al. 2010b)					0.31 (1.2)

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Executive Summary

Desertification is land degradation in arid, semi-arid, and dry sub-humid areas, collectively known as drylands, resulting from many factors, including human activities and climatic variations. The range and intensity of desertification have increased in some dryland areas over the past several decades (*high confidence*). Drylands currently cover about 46.2% (±0.8%) of the global land area and are home to 3 billion people. The multiplicity and complexity of the processes of desertification make its quantification difficult. Desertification hotspots, as identified by a decline in vegetation productivity between 1980s and 2000s, extended to about 9.2% of drylands (±0.5%), affecting about 500 (±120) million people in 2015. The highest numbers of people affected are in South and East Asia, North Africa and Middle East (*low confidence*). Desertification has already reduced agricultural productivity and incomes (*high confidence*) and contributed to the loss of biodiversity in some dryland regions (*medium confidence*). In many dryland areas, spread of invasive plants has led to losses in ecosystem services (*high confidence*), while over-extraction is leading to groundwater depletion (*high confidence*). Unsustainable land management, particularly when coupled with droughts, has contributed to higher dust storm activity, reducing human wellbeing in drylands and beyond (*high confidence*). Dust storms were associated with global cardiopulmonary mortality of about 402,000 people in a single year. Higher intensity of sand storms and sand dune movements are causing disruption and damage to transportation and solar and wind energy harvesting infrastructures (*high confidence*). {3.1.1, 3.1.4, 3.2.1, 3.3.1, 3.4.1, 3.4.2, 3.4.2, 3.7.3, 3.7.4}

Attribution of desertification to climate variability and change and human activities varies in space and time (*high confidence*). Climate variability and anthropogenic climate change, particularly through increases in both land surface air temperature and evapotranspiration, and decreases in precipitation, are *likely* to have played a role, in interaction with human activities, in causing desertification in some dryland areas. The major human drivers of desertification interacting with climate change are expansion of croplands, unsustainable land management practices and increased pressure on land from population and income growth. Poverty is limiting both capacities to adapt to climate change and availability of financial resources to invest in sustainable land management (SLM) (*high confidence*). {3.1.4, 3.2.2, 3.4.2}

Climate change will exacerbate several desertification processes (*medium confidence*). Although CO_2-fertilisation effect is enhancing vegetation productivity in drylands (*high confidence*), decreases in water availability have a larger effect than CO_2-fertilisation in many dryland areas. There is *high confidence* that aridity will increase in some places, but no evidence for a projected global trend in dryland aridity (*medium confidence*). The area at risk of salinisation is projected to increase in the future (*limited evidence, high agreement*). Future climate change is projected to increase the potential for water driven soil erosion in many dryland areas (*medium confidence*), leading to soil organic carbon decline in some dryland areas. {3.1.1, 3.2.2, 3.5.1, 3.5.2, 3.7.1, 3.7.3}

Risks from desertification are projected to increase due climate change (*high confidence*). Under shared socioeconomic pathway SSP2 ("Middle of the Road") at 1.5°C, 2°C and 3°C of global warming, the number of dryland population exposed (vulnerable) to various impacts related to water, energy and land sectors (e.g. water stress, drought intensity, habitat degradation) are projected to reach 951 (178) million, 1,152 (220) million and 1,285 (277) million, respectively. While at global warming of 2°C, under SSP1 (sustainability), the exposed (vulnerable) dryland population is 974 (35) million, and under SSP3 (Fragmented World) it is 1,267 (522) million. Around half of the vulnerable population is in South Asia, followed by Central Asia, West Africa and East Asia. {2.2, 3.1.1, 3.2.2, 3.5.1, 3.5.2, 7.2.2}

Desertification and climate change, both individually and in combination, will reduce the provision of dryland ecosystem services and lower ecosystem health, including losses in biodiversity (*high confidence*). Desertification and changing climate are projected to cause reductions in crop and livestock productivity (*high confidence*), modify the composition of plant species and reduce biological diversity across drylands (*medium confidence*). Rising CO_2 levels will favour more rapid expansion of some invasive plant species in some regions. A reduction in the quality and quantity of resources available to herbivores can have knock-on consequences for predators, which can potentially lead to disruptive ecological cascades (*limited evidence, low agreement*). Projected increases in temperature and the severity of drought events across some dryland areas can increase chances of wildfire occurrence (*medium confidence*). {3.1.4, 3.4.1, 3.5.2, 3.7.3}

Increasing human pressures on land combined with climate change will reduce the resilience of dryland populations and constrain their adaptive capacities (*medium confidence*). The combination of pressures coming from climate variability, anthropogenic climate change and desertification will contribute to poverty, food insecurity, and increased disease burden (*high confidence*), as well as potentially to conflicts (*low confidence*). Although strong impacts of climate change on migration in dryland areas are disputed (*medium evidence*, *low agreement*), in some places, desertification under changing climate can provide an added incentive to migrate (*medium confidence*). Women will be impacted more than men by environmental degradation, particularly in those areas with higher dependence on agricultural livelihoods (*medium evidence*, *high agreement*). {3.4.2, 3.6.2}

Desertification exacerbates climate change through several mechanisms such as changes in vegetation cover, sand and dust aerosols and greenhouse gas fluxes (*high confidence*). The extent of areas in which dryness controls CO_2 exchange (rather than temperature) has increased by 6% between 1948-2012, and is projected to increase by at least another 8% by 2050 if the expansion continues at the same rate. In these areas, net carbon uptake is about 27% lower than in other areas (*low confidence*). Desertification also tends to increase albedo, decreasing energy available at the surface and associated surface temperatures, producing a negative feedback on climate change (*high confidence*). Through its effect on vegetation and soils, desertification changes the absorption and release of associated greenhouse gases (GHGs). Vegetation loss and drying of surface cover due to desertification increases the frequency of dust storms (*high confidence*). Arid ecosystems could be an important global carbon sink depending on soil water availability (*medium evidence*, *high agreement*). {3.3.3, 3.4.1, 3.5.2}

Site-specific technological solutions, based both on new scientific innovations and indigenous and local knowledge (ILK), are available to avoid, reduce and reverse desertification, simultaneously contributing to climate change mitigation and adaptation (*high confidence*). SLM practices in drylands increase agricultural productivity and contribute to climate change adaptation and mitigation (*high confidence*). Integrated crop, soil and water management measures can be employed to reduce soil degradation and increase the resilience of agricultural production systems to the impacts of climate change (*high confidence*). These measures include crop diversification and adoption of drought-tolerant crops, reduced tillage, adoption of improved irrigation techniques (e.g. drip irrigation) and moisture conservation methods (e.g. rainwater harvesting using indigenous and local practices), and maintaining vegetation and mulch cover. Conservation agriculture increases the capacity of agricultural households to adapt to climate change (*high confidence*) and can lead to increases in soil organic carbon over time, with quantitative estimates of the rates of carbon sequestration in drylands following changes in agricultural practices ranging between 0.04-0.4 t ha^{-1} (*medium confidence*). Rangeland management systems based on sustainable grazing and re-vegetation increase rangeland productivity and the flow of ecosystem services *(high confidence*). The combined use of salt-tolerant crops, improved irrigation practices, chemical remediation measures and appropriate mulch and compost is effective in reducing the impact of secondary salinisation (*medium confidence*). Application of sand dune stabilisation techniques contributes to reducing sand and dust storms (*high confidence*). Agroforestry practices and shelterbelts help reduce soil erosion and sequester carbon. Afforestation programmes aimed at creating windbreaks in the form of "green walls" and "green dams" can help stabilise and reduce dust storms, avert wind erosion, and serve as carbon sinks, particularly when done with locally adapted tree species *(high confidence*). {3.4.2, 3.6.1, 3.7.2}

Investments into SLM, land restoration and rehabilitation in dryland areas have positive economic returns (***high confidence***). Each USD invested into land restoration can have social returns of about 3–6 USD over a 30-year period. Most SLM practices can become financially profitable within three to 10 years (*medium evidence*, *high agreement*). Despite their benefits in addressing desertification, mitigating and adapting to climate change, and increasing food and economic security, many SLM practices are not widely adopted due to insecure land tenure, lack of access to credit and agricultural advisory services, and insufficient incentives for private land users *(robust evidence, high agreement*). {3.6.3}

Indigenous and local knowledge (ILK) often contribute to enhancing resilience against climate change and combating desertification (***medium confidence***). Dryland populations have developed traditional agroecological practices which are well adapted to resource-sparse dryland environments. However, there is *robust evidence* documenting losses of traditional agroecological knowledge. Traditional agroecological practices are also increasingly unable to cope with growing demand for food. Combined use of ILK and new SLM technologies can contribute to raising the resilience to the challenges of climate change and desertification (*high confidence*). {3.1.3, 3.6.1, 3.6.2}

Policy frameworks promoting the adoption of SLM solutions contribute to addressing desertification as well as mitigating and adapting to climate change, with co-benefits for poverty reduction and food security among dryland populations (***high confidence***). **Implementation of Land Degradation Neutrality policies allows to avoid, reduce and reverse desertification, thus, contributing to climate change adaptation and mitigation (*high confidence*).** Strengthening land tenure security is a major factor contributing to the adoption of soil conservation measures in croplands (*high confidence*). On-farm and off-farm livelihood diversification strategies increase the resilience of rural households against desertification and extreme weather events, such as droughts (*high confidence*). Strengthening collective action is important for addressing causes and impacts of desertification, and for adapting to climate change (*medium confidence*). A greater emphasis on understanding gender-specific differences over land use and land management practices can help make land restoration projects more successful (*medium confidence*). Improved access to markets raises agricultural profitability and motivates investment into climate change adaptation and SLM (*medium confidence*). Payments for ecosystem services give additional incentives to land users to adopt SLM practices (*medium confidence*). Expanding access to rural advisory services increases the knowledge on SLM and facilitates their wider adoption (*medium confidence*). Transition to modern renewable energy sources can contribute to reducing desertification and mitigating climate change through decreasing the use of fuelwood and crop residues for energy (*medium confidence*). Policy responses to droughts based on pro-active drought preparedness and drought risk mitigation are more efficient in limiting drought-caused damages than reactive drought relief efforts (*high confidence*). {3.4.2, 3.6.2, 3.6.3, Cross-Chapter Box 5 in this chapter}

The knowledge on limits to adaptation to combined effects of climate change and desertification is insufficient. However, the potential for residual risks and maladaptive outcomes is high (*high confidence*). Empirical evidence on the limits to adaptation in dryland areas is limited, potential limits to adaptation include losses of land productivity due to irreversible forms of desertification. Residual risks can emerge from the inability of SLM measures to fully compensate for yield losses due to climate change impacts, as well as foregone reductions in ecosystem services due to soil fertility loss even when applying SLM measures could revert land to initial productivity after some time. Some activities favouring agricultural intensification in dryland areas can become maladaptive due to their negative impacts on the environment (*medium confidence*) {3.6.4}.

Improving capacities, providing higher access to climate services, including local level early warning systems, and expanding the use of remote sensing technologies are high return investments for enabling effective adaptation and mitigation responses that help address desertification (***high confidence***). Reliable and timely climate services, relevant to desertification, can aid the development of appropriate adaptation and mitigation options reducing the impact of desertification on human and natural systems (*high confidence*), with quantitative estimates pointing that every USD invested in strengthening hydro-meteorological and early warning services in developing countries can yield between 4 to 35 USD (*low confidence)*. Knowledge and flow of knowledge on desertification is currently fragmented. Improved knowledge and data exchange and sharing will increase the effectiveness of efforts to achieve Land Degradation Neutrality (*high confidence*). Expanded use of remotely sensed information for data collection helps in measuring progress towards achieving Land Degradation Neutrality (*low evidence, high agreement*). {3.2.1, 3.6.2, 3.6.3, Cross-Chapter Box 5 in this chapter}

3.1. The Nature of Desertification

3.1.1. Introduction

In this report, desertification is defined as land degradation in arid, semi-arid, and dry sub-humid areas resulting from many factors, including climatic variations and human activities (United Nations Convention to Combat Desertification (UNCCD 1994). Land degradation is a negative trend in land condition, caused by direct or indirect human-induced processes including anthropogenic climate change, expressed as long-term reduction or loss of at least one of the following: biological productivity, ecological integrity or value to humans (4.1.3). Arid, semi-arid, and dry sub-humid areas, together with hyper-arid areas, constitute drylands (UNEP, 1992), home to about 3 billion people (van der Esch et al., 2017). The difference between desertification and land degradation is not process-based but geographic. Although land degradation can occur anywhere across the world, when it occurs in drylands, it is considered desertification (FAQ 1.3). Desertification is not limited to irreversible forms of land degradation, nor is it equated to desert expansion, but represents all forms and levels of land degradation occurring in drylands.

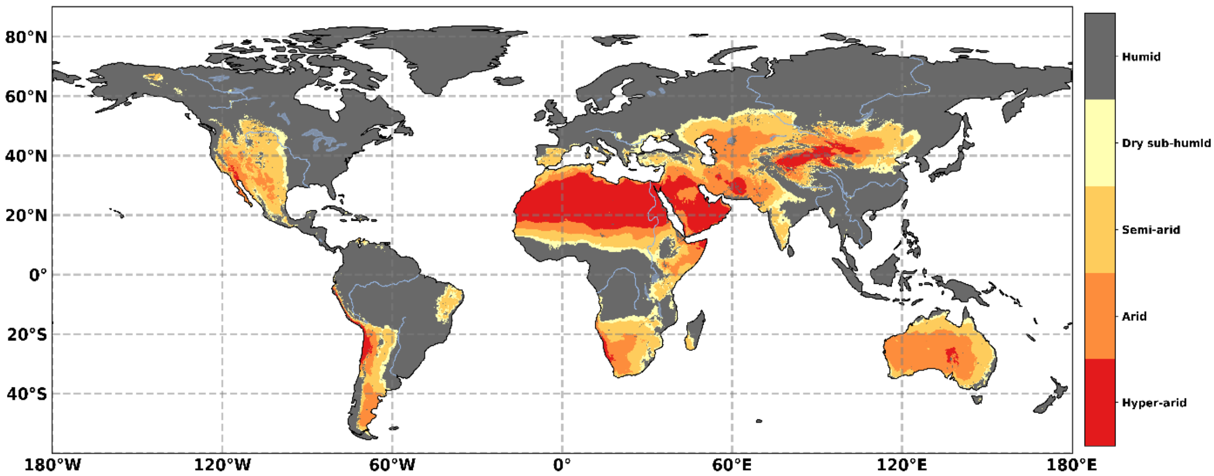


Figure 3.1 Geographical distribution of drylands, delimited based on the Aridity Index (AI). The classification of AI is: Humid AI > 0.65, Dry sub-humid 0.50 < AI ≤ 0.65, Semi-arid 0.20 < AI ≤ 0.50, Arid 0.05 < AI ≤ 0.20, Hyper-arid AI < 0.05. Data: TerraClimate precipitation and potential evapotranspiration (1980-2015) (Abatzoglou et al., 2018).

The geographic classification of drylands is often based on the aridity index (AI) - the ratio of average annual precipitation amount (P) to potential evapotranspiration amount (PET, see glossary) (Figure 3.1). Recent estimates, based on AI, suggest that drylands cover about 46.2% (±0.8%) of the global land area (Koutroulis, 2019; Prăvălie, 2016) *(low confidence*). Hyper-arid areas, where the aridity index is below 0.05, are included in drylands, but are excluded from the definition of desertification (UNCCD, 1994). Deserts are valuable ecosystems (UNEP, 2006; Safriel, 2009) geographically located in drylands and vulnerable to climate change. However, they are not considered prone to desertification. Aridity is a long-term climatic feature characterised by low average precipitation or available water (Gbeckor-Kove, 1989; Türkeş, 1999). Thus, aridity is different from drought which is a temporary climatic event (Maliva and Missimer, 2012). Moreover, droughts are not restricted to drylands, but occur both in drylands and humid areas (Wilhite et al., 2014). Following the Synthesis Report (SYR) of the IPCC Fifth Assessment Report (AR5), drought is defined here as "a period of abnormally dry weather long enough to cause a serious hydrological imbalance" (Mach et al., 2014; Cross-Chapter Box 5: Case study on policy responses to drought, in this chapter).

AI is not an accurate proxy for delineating drylands in an increasing CO_2 environment (3.2.1). The suggestion that most of the world has become more arid, since the AI has decreased, is not supported by changes observed in precipitation, evaporation or drought (Sheffield et al., 2012; Greve et al., 2014). While climate change is expected to decrease the AI due to increases in potential evaporation, the assumptions that underpin the potential evaporation calculation are not consistent with a changing CO_2 environment and the effect this has on transpiration rates (3.2.1; Roderick et al., 2015; Milly and Dunne, 2016; Greve et al., 2017). Given that future climate is characterised by significant increases in CO_2, the usefulness of currently applied AI thresholds to estimate dryland areas is limited under climate change. If instead of the AI, other variables such as precipitation, soil moisture, and primary productivity are used to identify dryland areas, there is no clear indication that the extent of drylands will change overall under climate change (Roderick et al., 2015; Greve et al., 2017; Lemordant et al., 2018). Thus, some dryland borders will expand, while some others will contract (*high confidence*).

Approximately 70% of dryland areas are located in Africa and Asia (Figure 3.2). The biggest land use/cover in terms of area in drylands, if deserts are excluded, are grasslands, followed by forests and croplands (Figure 3.3). The category of “other lands” in Figure 3.3 includes bare soil, ice, rock, and all other land areas that are not included within the other five categories (FAO, 2016). Thus, hyper-arid areas contain mostly deserts, with some small exceptions, for example, where grasslands and croplands are cultivated under oasis conditions with irrigation (3.7.4). Moreover, FAO (2016) defines grasslands as permanent pastures and meadows used continuously for more than five years. In drylands, transhumance, i.e. seasonal migratory grazing, often leads to non-permanent pasture systems, thus, some of the areas under “other land” category are also used as non-permanent pastures (Ramankutty et al., 2008; Fetzel et al., 2017; Erb et al., 2016).

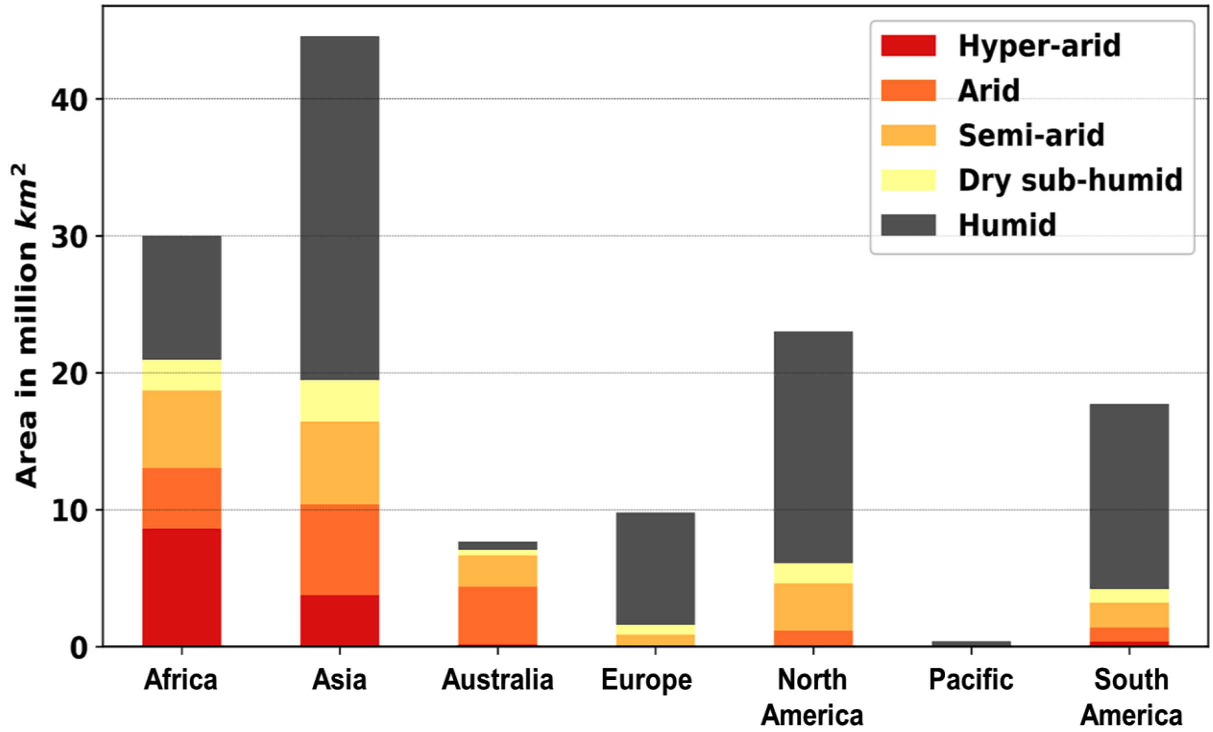


Figure 3.2 Dryland categories across geographical areas (continents and Pacific region). Data: TerraClimate precipitation and potential evapotranspiration (1980-2015) (Abatzoglou et al., 2018).

In the earlier global assessments of desertification (since the 1970s), which were based on qualitative expert evaluations, the extent of desertification was found to range between 4% and 70% of the area of drylands (Safriel, 2007). More recent estimates, based on remotely sensed data, show that about 24–29% of the global land area experienced reductions in biomass productivity between 1980s and

2000s (Bai et al., 2008; Le et al., 2016), corresponding to about 9.2% of drylands (±0.5%) experiencing declines in biomass productivity during this period (*low confidence*), mainly due to anthropogenic causes. Both of these studies consider rainfall dynamics, thus, accounting for the effect of droughts. While less than 10% of drylands is undergoing desertification, it is occurring in areas that contain around 20% of dryland population (Klein Goldewijk et a., 2017). In these areas the population has increased from ~172 million in 1950 to over 630 million today (Figure 1.1).

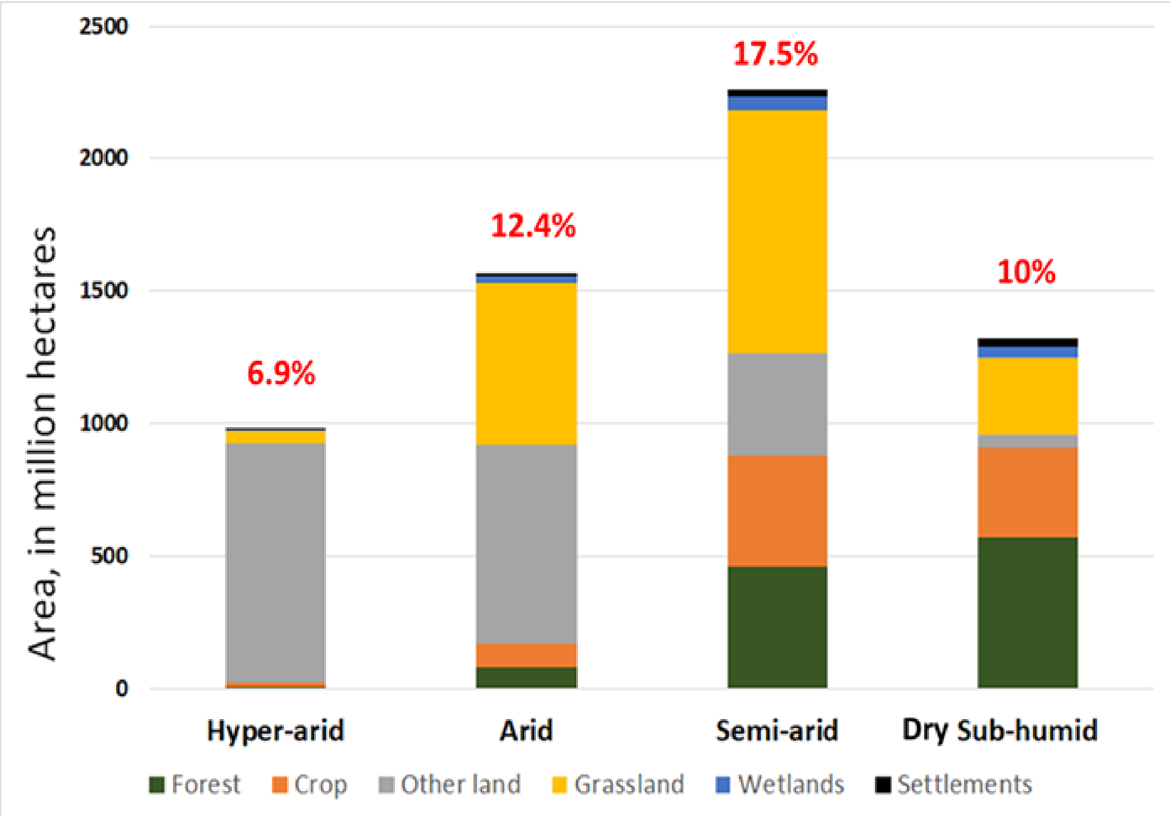


Figure 3.3 Land use and land cover in drylands and share of each dryland category in global land area. Source: FAO (2016).

Available assessments of the global extent and severity of desertification are relatively crude approximations with considerable uncertainties, for example, due to confounding effects of invasive bush encroachment in some dryland regions. Different indicator sets and approaches have been developed for monitoring and assessment of desertification from national to global scales (Imeson, 2012; Sommer et al., 2011; Zucca et al., 2012; Bestelmeyer et al., 2013). Many indicators of desertification only include a single factor or characteristic of desertification, such as the patch size distribution of vegetation (Maestre and Escudero, 2009; Kéfi et al., 2010), Normalized Difference Vegetation Index (NDVI) (Piao et al., 2005), drought-tolerant plant species (An et al., 2007), grass cover (Bestelmeyer et al., 2013), land productivity dynamics (Baskan et al., 2017), ecosystem net primary productivity (Zhou et al., 2015) or environmentally sensitive land area index (Symeonakis et al., 2016). In addition, some synthetic indicators of desertification have also been used to assess desertification extent and desertification processes, such as climate, land use, soil, and socioeconomic parameters (Dharumarajan et al., 2018), or changes in climate, land use, vegetation cover, soil properties and population as the desertification vulnerability index (Salvati et al., 2009). Current data availability and methodological challenges do not allow for accurately and comprehensively mapping desertification at a global scale (Cherlet et al., 2018). However, the emerging partial evidence points to a lower global extent of desertification than previously estimated (*medium confidence*) (3.2).

This assessment examines the socio-ecological links between drivers (3.1) and feedbacks (3.3) that influence desertification-climate change interactions, and then examines associated observed and projected impacts (3.4, 3.5) and responses (3.6). Moreover, this assessment highlights that dryland populations are highly vulnerable to desertification and climate change (3.2, 3.4). At the same time, dryland populations also have significant past experience and sources of resilience embodied in indigenous and local knowledge and practices in order to successfully adapt to climatic changes and address desertification (3.6). Numerous site-specific technological response options are also available for SLM in drylands that can help increase the resilience of agricultural livelihood systems to climate change (3.6). However, continuing environmental degradation combined with climate change are straining the resilience of dryland populations. Enabling policy responses for SLM and livelihoods diversification can help maintain and strengthen the resilience and adaptive capacities in dryland areas (3.6). The assessment finds that policies promoting SLM in drylands will contribute to climate change adaptation and mitigation, with co-benefits for broader sustainable development (*high confidence*) (3.4).

3.1.2. Desertification in previous IPCC and related reports

The IPCC Fifth Assessment report (AR5) and Special Report on Global Warming of 1.5°C include a limited discussion of desertification. In AR5 Working Group I desertification is mentioned as a forcing agent for the production of atmospheric dust (Myhre et al., 2013). The same report had *low confidence* in the available projections on the changes in dust loadings due to climate change (Boucher et al., 2013). In AR5 Working Group II, desertification is identified as a process that can lead to reductions in crop yields and the resilience of agricultural and pastoral livelihoods (Field et al., 2014; Klein et al., 2015). AR5 Working Group II notes that climate change will amplify water scarcity with negative impacts on agricultural systems, particularly in semi-arid environments of Africa (*high confidence*), while droughts could exacerbate desertification in south-western parts of Central Asia (Field et al., 2014). AR5 Working Group III identifies desertification as one of a number of often overlapping issues that must be dealt with when considering governance of mitigation and adaptation (Fleurbaey et al., 2014). The IPCC Special Report on Global Warming of 1.5°C noted that limiting global warming to 1.5°C instead of 2°C is strongly beneficial for land ecosystems and their services (*high confidence*) such as soil conservation, contributing to avoidance of desertification (Hoegh-Guldberg et al., 2018).

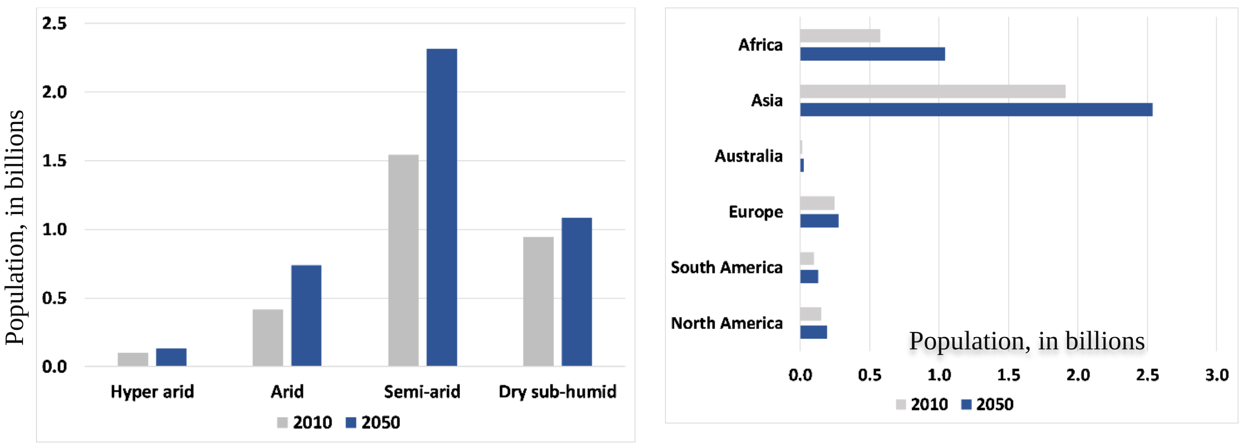
The recent Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) Assessment report on land degradation and restoration (IPBES, 2018a) is also of particular relevance. While acknowledging a wide variety of past estimates of the area undergoing degradation, IPBES (2018a) pointed at their lack of agreement about where degradation is taking place. IPBES (2018a) also recognised the challenges associated with differentiating the impacts of climate variability and change on land degradation from the impacts of human activities at a regional or global scale.

The third edition of the World Atlas of Desertification (Cherlet et al., 2018) indicated that it is not possible to deterministically map the global extent of land degradation, and its subset - desertification, pointing out that the complexity of interactions between social, economic, and environmental systems make land degradation not amenable to mapping at a global scale. Instead, Cherlet et al. (2018) presented global maps highlighting the convergence of various pressures on land resources.

3.1.3. Dryland Populations: Vulnerability and Resilience

Drylands are home to approximately 38.2% ((±0.6%) of the global population (Koutroulis, 2019; van der Esch et al., 2017), that is about 3 billion people. The highest number of people live in the drylands of South Asia (Figure 3.4), followed by Sub-Saharan Africa and Latin America (van der Esch et al., 2017). In terms of the number of people affected by desertification, Reynolds et al. (2007) indicated that desertification was directly affecting 250 million people. More recent estimates show that 500 (±120) million people lived in 2015 in those dryland areas which experienced significant loss in biomass productivity between 1980s and 2000s (Bai et al., 2008; Le et al., 2016). The highest numbers of affected people were in South and East Asia, North Africa and Middle East (*low confidence*). The population in drylands is projected to increase about twice as rapidly as non-drylands, reaching 4 billion people by 2050 (van der Esch et al., 2017). This is due to higher population growth rates in drylands. About 90% of the population in drylands live in developing countries (UN-EMG, 2011).

Dryland populations are highly vulnerable to desertification and climate change (Howe et al., 2013; Huang et al., 2016, 2017; Liu et al., 2016b; Thornton et al., 2014; Lawrence et al., 2018) because their livelihoods are predominantly dependent on agriculture; one of the sectors most susceptible to climate change (Rosenzweig et al., 2014; Schlenker and Lobell, 2010). Climate change is projected to have substantial impacts on all types of agricultural livelihood systems in drylands (CGIAR-RPDS, 2014) (3.4.1, 3.4.2).



(a) (b)

Figure 3.4 Current (a) and projected population (under SSP2) (b) in drylands, in billions. Source: van der Esch et al. (2017)

One key vulnerable group in drylands are pastoral and agropastoral households[1]. There are no precise figures about the number of people practicing pastoralism globally. Most estimates range between 100 to 200 million (Rass, 2006; Secretariat of the Convention on Biological Diversity, 2010), of whom 30–63 million are nomadic pastoralists (Dong, 2016; Carr-Hill, 2013)[2]. Pastoral production systems represent an adaptation to high seasonal climate variability and low biomass productivity in dryland ecosystems (Varghese and Singh, 2016; Krätli and Schareika, 2010), which require large areas for

[1]FOOTNOTE: Pastoralists derive more than 50% of their income from livestock and livestock products, whereas agro-pastoralists generate more than 50% of their income from crop production and at least 25% from livestock production (Swift, 1988).

[2]FOOTNOTE: The estimates of the number of pastoralists, and especially of nomadic pastoralists, are very uncertain, because often nomadic pastoralists are not fully captured in national surveys and censuses (Carr-Hill, 2013).

livestock grazing through migratory pastoralism (Snorek et al., 2014). Grazing lands across dryland environments are being degraded, and/or being converted to crop production, limiting the opportunities for migratory livestock systems, and leading to conflicts with sedentary crop producers (Abbass, 2014; Dimelu et al., 2016). These processes, coupled with ethnic differences, perceived security threats, and misunderstanding of pastoral rationality, have led to increasing marginalisation of pastoral communities and disruption of their economic and cultural structures (Elhadary, 2014; Morton, 2010). As a result, pastoral communities are not well prepared to deal with increasing weather/climate variability and weather/climate extremes due to changing climate (Dong, 2016; López-i-Gelats et al., 2016), and remain amongst the most food insecure groups in the world (FAO, 2018).

There is an increasing concentration of poverty in the dryland areas of Sub-Saharan Africa and South Asia (von Braun and Gatzweiler, 2014; Barbier and Hochard, 2016), where 41% and 12% of the total populations live in extreme poverty, respectively (World Bank, 2018). For comparison, the average share of global population living in extreme poverty is about 10% (World Bank, 2018). Multidimensional poverty, prevalent in many dryland areas, is a key source of vulnerability (Safriel et al., 2005; Thornton et al., 2014; Fraser et al., 2011; Thomas, 2008). Multidimensional poverty incorporates both income-based poverty, and also other dimensions such as poor healthcare services, lack of education, lack of access to water, sanitation and energy, disempowerment, and threat from violence (Bourguignon and Chakravarty, 2003; Alkire and Santos, 2010, 2014). Contributing elements to this multidimensional poverty in drylands are rapid population growth, fragile institutional environment, lack of infrastructure, geographic isolation and low market access, insecure land tenure systems, and low agricultural productivity (Sietz et al., 2011; Reynolds et al., 2011; Safriel and Adeel, 2008; Stafford Smith, 2016). Even in high-income countries, those dryland areas that depend on agricultural livelihoods represent relatively poorer locations nationally, with fewer livelihood opportunities, for example in Italy (Salvati, 2014). Moreover, in many drylands areas, female-headed households, women and subsistence farmers (both male and female) are more vulnerable to the impacts of desertification and climate change (Nyantakyi-Frimpong and Bezner-Kerr, 2015; Sultana, 2014; Rahman, 2013). Some local cultural traditions and patriarchal relationships were found to contribute to higher vulnerability of women and female-headed households through restrictions on their access to productive resources (Nyantakyi-Frimpong and Bezner-Kerr, 2015; Sultana, 2014; Rahman, 2013) (3.4.2, 3.6.3; Cross-Chapter Box 11: Gender, Chapter 7).

Despite these environmental, socio-economic and institutional constraints, dryland populations have historically demonstrated remarkable resilience, ingenuity and innovations, distilled into indigenous and local knowledge to cope with high climatic variability and sustain livelihoods (Safriel and Adeel, 2008; Davis, 2016; Davies, 2017; 3.6.1, 3.6.2; Cross-Chapter Box 13: Indigenous and Local Knowledge, Chapter 7). For example, across the Arabian Peninsula and North Africa, informal community bylaws were successfully used for regulating grazing, collection and cutting of herbs and wood, that limited rangeland degradation (Gari, 2006; Hussein, 2011). Pastoralists in Mongolia developed indigenous classifications of pasture resources which facilitated ecologically optimal grazing practices (Fernandez-Gimenez, 2000) (3.6.2). Currently, however, indigenous and local knowledge and practices are increasingly lost or can no longer cope with growing demands for land-based resources (Dominguez, 2014; Fernández-Giménez and Fillat Estaque, 2012; Hussein, 2011; Kodirekkala, 2017; Moreno-Calles et al., 2012; 3.4.2). Unsustainable land management is increasing the risks from droughts, floods and dust storms (3.4.2, 3.5). Policy actions promoting the adoption of SLM practices in dryland areas, based on both indigenous and local knowledge and modern science, and expanding alternative livelihood opportunities outside agriculture can contribute to climate change adaptation and mitigation, addressing desertification, with co-benefits for poverty reduction and food security (*high confidence*) (Cowie et al., 2018; Liniger et al., 2017; Safriel and Adeel, 2008; Stafford-Smith et al., 2017).

3.1.4. Processes and Drivers of Desertification under Climate Change

3.1.4.1 Processes of Desertification and Their Climatic Drivers

Processes of desertification are mechanisms by which drylands are degraded. Desertification consists of both biological and non-biological processes. These processes are classified under broad categories of degradation of physical, chemical and biological properties of terrestrial ecosystems. The number of desertification processes is large and they are extensively covered elsewhere (IPBES, 2018a; Lal, 2016; Racine, 2008; UNCCD, 2017). Section 4.2.1 and Tables 4.1-4.2 in Chapter 4 highlight those which are particularly relevant for this assessment in terms of their links to climate change and land degradation, including desertification.

Drivers of desertification are factors which trigger desertification processes. Initial studies of desertification during the early-to-mid 20th century attributed it entirely to human activities. In one of the influential publications of that time, Lavauden (1927) stated that: "Desertification is purely artificial. It is only the act of the man...” However, such a uni-causal view on desertification was shown to be invalid (Geist et al., 2004; Reynolds et al., 2007) (3.1.4.2, 3.1.4.3). Tables 4.1-4.2 in Chapter 4 summarise drivers, linking them to the specific processes of desertification and land degradation under changing climate.

Erosion refers to removal of soil by the physical forces of water, wind, or often caused by farming activities such as tillage (Ginoux et al., 2012). The global estimates of soil erosion differ significantly, depending on scale, study period and method used (García-Ruiz et al., 2015), ranging from approximately 20 Gt yr^{-1} to more than 200 Gt yr^{-1} (Boix-Fayos et al., 2006; FAO, 2015). There is a significant potential for climate change to increase soil erosion by water particularly in those regions where precipitation volumes and intensity are projected to increase (Panthou et al., 2014; Nearing et al., 2015). On the other hand, while it is a dominant form of erosion in areas such as West Asia and the Arabian Peninsula (Prakash et al., 2015; Klingmüller et al., 2016), there is *limited evidence* concerning climate change impacts on wind erosion (Tables 4.1-4.2 in Chapter 4; 3.5).

Saline and sodic soils (see glossary) occur naturally in arid, semiarid and dry sub-humid regions of the world. Climate change or hydrological change can cause soil salinisation by increasing the mineralised ground water level. However, secondary salinisation occurs when the concentration of dissolved salts in water and soil is increased by anthropogenic processes, mainly through poorly managed irrigation schemes. The threat of soil and groundwater salinisation induced by sea level rise and sea water intrusion are amplified by climate change (4.9.7).

Global warming is expected to accelerate soil organic carbon (SOC) turnover, since the decomposition of the soil organic matter by microbial activity begins with low soil water availability, but this moisture is insufficient for plant productivity (Austin et al., 2004; 3.4.1.1), as well as losses by soil erosion (Lal, 2009); therefore, in some dryland areas leading to SOC decline (3.3.3; 3.5.2) and the transfer of carbon (C) from soil to the atmosphere (Lal, 2009).

Sea surface temperature (SST) anomalies can drive rainfall changes, with implications for desertification processes. North Atlantic SST anomalies are positively correlated with Sahel rainfall anomalies (Knight et al., 2006; Gonzalez-Martin et al., 2014; Sheen et al., 2017). While the eastern tropical Pacific SST anomalies have a negative correlation with Sahel rainfall (Pomposi et al., 2016), a cooler north Atlantic is related to a drier Sahel, with this relationship enhanced if there is a simultaneous relative warming of the south Atlantic (Hoerling et al., 2006). Huber and Fensholt (2011) explored the relationship between SST anomalies and satellite observed Sahel vegetation dynamics finding similar relationships but with substantial west-east variations in both the significant SST regions and the vegetation response. Concerning the paleoclimatic evidence on aridification after the early Holocene “Green Sahara” period (11,000 to 5000 years ago), Tierney et al. (2017) indicate that a cooling of the north Atlantic played a role (Collins et al., 2017; Otto-Bliesner et al., 2014; Niedermeyer et al., 2009) similar to that found in modern observations. Besides these SST relationships, aerosols have also been suggested as a potential driver of the Sahel droughts (Rotstayn and Lohmann, 2002; Booth et al., 2012; Ackerley et al., 2011). For Eastern Africa, both recent droughts and decadal declines have been linked to human-induced warming in the western Pacific (Funk et al., 2018).

Invasive plants contributed to desertification and loss of ecosystem services in many dryland areas in the last century (*high confidence*) (3.7.3). Extensive woody plant encroachment altered runoff and soil erosion across much of the drylands, because the bare soil between shrubs is very susceptible to water erosion, mainly in high-intensity rainfall events (Manjoro et al., 2012; Pierson et al., 2013; Eldridge et al., 2015). Rising CO_2 levels due to global warming favour more rapid expansion of some invasive plant species in some regions. An example is the Great Basin region in western North America where over 20% of ecosystems have been significantly altered by invasive plants, especially exotic annual grasses and invasive conifers resulting in loss of biodiversity. This land cover conversion has resulted in reductions in forage availability, wildlife habitat, and biodiversity (Pierson et al., 2011, 2013; Miller et al., 2013).

The wildfire is a driver of desertification, because it reduces vegetation cover, increases runoff and soil erosion, reduces soil fertility and affects the soil microbial community (Vega et al., 2005; Nyman et al., 2010; Holden et al., 2013; Pourreza et al., 2014; Weber et al., 2014; Liu and Wimberly, 2016). Predicted increases in temperature and the severity of drought events across some dryland areas (2.2) can increase chances of wildfire occurrence (*medium confidence*) (Jolly et al., 2015; Williams et al., 2010; Clarke and Evans, 2018; Cross-Chapter Box 3: Fire and Climate Change, Chapter 2). In semiarid and dry sub-humid areas, fire can have a profound influence on observed vegetation and particularly the relative abundance of grasses to woody plants (Bond et al., 2003; Bond and Keeley, 2005; Balch et al., 2013).

While large uncertainty exists concerning trends in droughts globally (AR5, 2.2), examining the drought data by Ziese et al.(2014) for drylands only reveals a large inter-annual variability combined with a trend toward increasing dryland area affected by droughts since 1950s (Figure 1.1).

3.1.4.2. Anthropogenic Drivers of Desertification under Climate Change

The literature on the human drivers of desertification is substantial (D'Odorico et al., 2013; Sietz et al., 2011; Yan and Cai, 2015; Sterk et al., 2016; Varghese and Singh, 2016; to list a few) and there have been several comprehensive reviews and assessments of these drivers very recently (Cherlet et al., 2018; IPBES, 2018a; UNCCD, 2017). IPBES (2018a) identified cropland expansion, unsustainable land management practices including overgrazing by livestock, urban expansion, infrastructure development, and extractive industries as the main drivers of land degradation. IPBES (2018a) also found that the ultimate driver of land degradation is high and growing consumption of land-based resources, e.g. through deforestation and cropland expansion, escalated by population growth. What is particularly relevant in the context of the present assessment is to evaluate if, how and which human drivers of desertification will be modified by climate change effects.

Growing food demand is driving conversion of forests, rangelands, and woodlands into cropland (Bestelmeyer et al., 2015; D'Odorico et al., 2013). Climate change is projected to reduce crop yields across dryland areas (3.4.1; 5.2.2), potentially reducing local production of food and feed. Without research breakthroughs mitigating these productivity losses through higher agricultural productivity, and reducing food waste and loss, meeting increasing food demands of growing populations will require expansion of cropped areas to more marginal areas (with most prime areas in drylands already being under cultivation) (Lambin, 2012; Lambin et al., 2013; Eitelberg et al., 2015; Gutiérrez-Elorza, 2006; Kapović Solomun et al., 2018). Borrelli et al. (2017) showed that the primary driver of soil erosion in 2012 was cropland expansion. Although local food demands could also be met by importing from other areas, this would mean increasing the pressure on land in those areas (Lambin and Meyfroidt, 2011). The net effects of such global agricultural production shifts on land condition in drylands are not known.

Climate change will exacerbate poverty among some categories of dryland populations (3.4.2; 3.5.2). Depending on the context, this impact comes through declines in agricultural productivity, changes in agricultural prices and extreme weather events (Hertel and Lobell, 2014; Hallegatte and Rozenberg, 2017). There is *high confidence* that poverty limits both capacities to adapt to climate change and availability of financial resources to invest into SLM (3.5.2; 3.6.2; 3.6.3; Gerber et al., 2014; Way, 2016; Vu et al., 2014).

Labour mobility is another key human driver which will interact with climate change. Although strong impacts of climate change on migration in dryland areas are disputed, in some places, it is *likely* to provide an added incentive to migrate (3.4.2.7). Out-migration will have several contradictory effects on desertification. On one hand, it reduces an immediate pressure on land if it leads to less dependence on land for livelihoods (Chen et al., 2014; Liu et al., 2016a). Moreover, migrant remittances could be used to fund the adoption of SLM practices. Labour mobility from agriculture to non-agricultural sectors could allow land consolidation, gradually leading to mechanisation and agricultural intensification (Wang et al., 2014, 2018). On the other hand, this can increase the costs of labour-intensive SLM practices due to lower availability of rural agricultural labour and/or higher rural wages. Out-migration increases the pressure on land if higher wages that rural migrants earn in urban centres will lead to their higher food consumption. Moreover, migrant remittances could also be used to fund land use expansion to marginal areas (Taylor et al., 2016; Gray and Bilsborrow, 2014). The net effect of these opposite mechanisms varies from place to place (Qin and Liao, 2016). There is very little literature evaluating these joint effects of climate change, desertification and labour mobility (7.3.2).

There are also many other institutional, policy and socio-economic drivers of desertification, such as land tenure insecurity, lack of property rights, lack of access to markets, and to rural advisory services, lack of technical knowledge and skills, agricultural price distortions, agricultural support and subsidies contributing to desertification, and lack of economic incentives for SLM (D'Odorico et al., 2013; Geist et al., 2004; Moussa et al., 2016; Mythili and Goedecke, 2016; Sow et al., 2016; Tun et al., 2015; García-Ruiz, 2010). There is no evidence that these factors will be materially affected by climate change, however, serving as drivers of unsustainable land management practices, they do play a very important role in modulating responses for climate change adaptation and mitigation (3.6.3).

3.1.4.3 Interaction of Drivers: Desertification Syndrome versus Drylands Development Paradigm

Two broad narratives have historically emerged to describe responses of dryland populations to environmental degradation. The first is "desertification syndrome" which describes the vicious cycle of resource degradation and poverty, whereby dryland populations apply unsustainable agricultural practices leading to desertification, and exacerbating their poverty, which then subsequently further limits their capacities to invest in SLM (MEA, 2005; Safriel and Adeel, 2008). The alternative paradigm is one of "drylands development", which refers to social and technical ingenuity of dryland populations as a driver of dryland sustainability (MEA, 2005; Reynolds et al., 2007; Safriel and Adeel, 2008). The major difference between these two frameworks is that the "drylands development paradigm" recognises that human activities are not the sole and/or most important drivers of desertification, but there are interactions of human and climatic drivers within coupled social-ecological systems (Reynolds et al., 2007). This led Behnke and Mortimore (2016), and earlier Swift (1996), to conclude that the concept of desertification as irreversible degradation distorts policy and governance in the dryland areas. Mortimore (2016) suggested that instead of externally imposed technical solutions, what is needed is for populations in dryland areas to adapt to this variable environment which they cannot control. All in all, there is *high confidence* that anthropogenic and climatic drivers interact in complex ways in causing desertification. As discussed in Section 3.2.2, the relative influence of human or climatic drivers on desertification varies from place to place (*high confidence*) (Bestelmeyer et al., 2018; D'Odorico et al., 2013; Geist and Lambin, 2004; Kok et al., 2016; Polley et al., 2013; Ravi et al., 2010; Scholes, 2009; Sietz et al., 2017; Sietz et al., 2011).

3.2. Observations of Desertification

3.2.1. Status and Trends of Desertification

Current estimates of the extent and severity of desertification vary greatly due to missing and/or unreliable information (Gibbs and Salmon, 2015). The multiplicity and complexity of the processes of desertification make its quantification difficult (Prince, 2016; Cherlet et al., 2018). The most common definition for the drylands is based on defined thresholds of the AI (Figure 3.1) (UNEP, 1992). While past studies have used the AI to examine changes in desertification or extent of the drylands (Feng and Fu, 2013; Zarch et al., 2015; Ji et al., 2015; Spinoni et al., 2015; Huang et al., 2016; Ramarao et al., 2018), this approach has several key limitations: (i) the AI does not measure desertification, (ii) the impact of changes in climate on the land surface and systems is more complex than assumed by AI, and (iii) the relationship between climate change and changes in vegetation is complex due to the influence of CO_2. Expansion of the drylands does not imply desertification by itself, if there is no long-term loss of at least one of the following: biological productivity, ecological integrity, and value to humans.

The use of the AI to define changing aridity levels and dryland extent in an environment with changing atmospheric CO_2 has been strongly challenged (Roderick et al., 2015; Milly and Dunne, 2016; Greve et al., 2017; Liu et al., 2017). The suggestion that most of the world has become more arid, since the AI has decreased, is not supported by changes observed in precipitation, evaporation or drought (Sheffield et al., 2012; Greve et al., 2014) (*medium confidence*). A key issue is the assumption in the calculation of potential evapotranspiration that stomatal conductance remains constant which is invalid if atmospheric CO_2 changes. Given that atmospheric CO_2 has been increasing over the last century or more, and is projected to continue increasing, this means that AI with constant thresholds (or any other measure that relies on potential evapotranspiration) is not an appropriate way to estimate aridity or dryland extent (Donohue et al., 2013; Roderick et al., 2015; Greve et al., 2017). This issue helps explain the apparent contradiction between the drylands becoming more arid according to the AI and also becoming greener according to satellite observations (Fensholt et al., 2012; Andela et al., 2013; Figure 3.5). Other climate type classifications based on various combinations of temperature and precipitation (Köppen-Trewartha, Köppen-Geiger) have also been used to examine historical changes in climate zones finding a tendency toward drier climate types (Feng et al., 2014; Spinoni et al., 2015).

The need to establish a baseline when assessing change in the land area degraded has been extensively discussed in Prince et al. (2018). Desertification is a process not a state of the system, hence an "absolute" baseline is not required; however, every study uses a baseline defined by the start of their period of interest.

Depending on the definitions applied and methodologies used in evaluation, the status and extent of desertification globally and regionally still show substantial variations (D'Odorico et al., 2013) (*high confidence*). There is *high confidence* that the range and intensity of desertification has increased in some dryland areas over the past several decades (3.2.1.1, 3.2.1.2). The three methodological approaches applied for assessing the extent of desertification: expert judgement, satellite observation of net primary productivity, and use of biophysical models, together provide a relatively holistic assessment but none on its own captures the whole picture (Gibbs and Salmon, 2015; Vogt et al., 2011; Prince, 2016; 4.2.4).

3.2.1.1. Global Scale

Complex human-environment interactions coupled with biophysical, social, economic and political factors unique to any given location render desertification difficult to map at a global scale (Cherlet et al., 2018). Early attempts to assess desertification focused on expert knowledge in order to obtain global coverage in a cost-effective manner. **Expert judgement** continues to play an important role because degradation remains a subjective feature whose indicators are different from place to place (Sonneveld and Dent, 2007). GLASOD (Global Assessment of Human-Induced Soil Degradation) estimated nearly 2000 million hectares (M ha) (15.3% of the total land area) had been degraded by early 1990s since mid-20th century. GLASOD was criticised for perceived subjectiveness and exaggeration (Helldén and Tottrup, 2008; Sonneveld and Dent, 2007). Dregne and Chou (1992) found 3000 M ha in drylands (i.e. about 50% of drylands) were undergoing degradation. Significant improvements have been made through the efforts of WOCAT (World Overview of Conservation Approaches and Technologies), LADA (Land Degradation Assessment in Drylands) and DESIRE (Desertification Mitigation and Remediation of Land) who jointly developed a mapping tool for participatory expert assessment, with which land experts can estimate current area coverage, type and trends of land degradation (Reed et al., 2011).

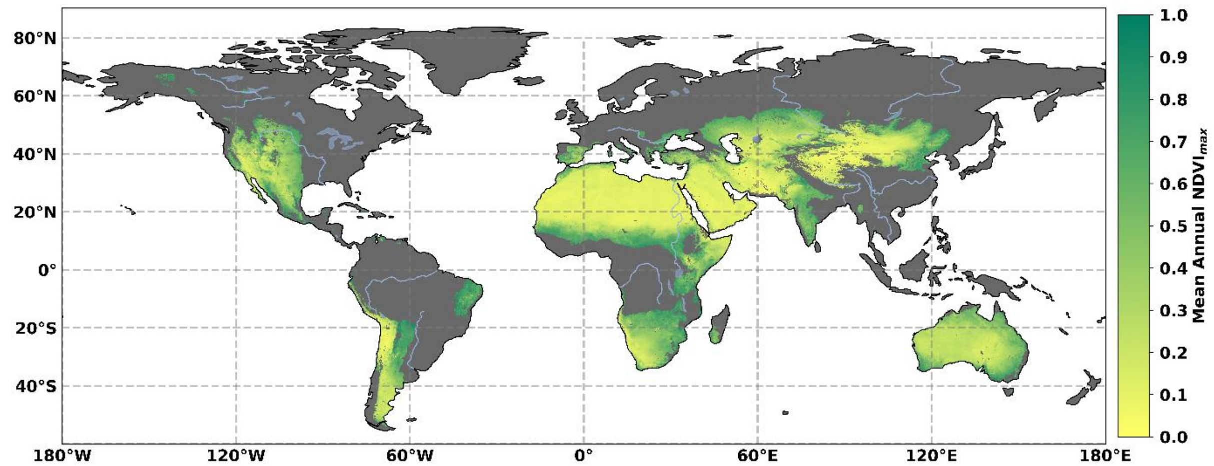


Figure 3.5 Mean Annual Maximum NDVI 1982-2015 (Global Inventory Modelling and Mapping Studies NDVI3g v1). Non-dryland regions (Aridity Index > 0.65) are masked in grey.

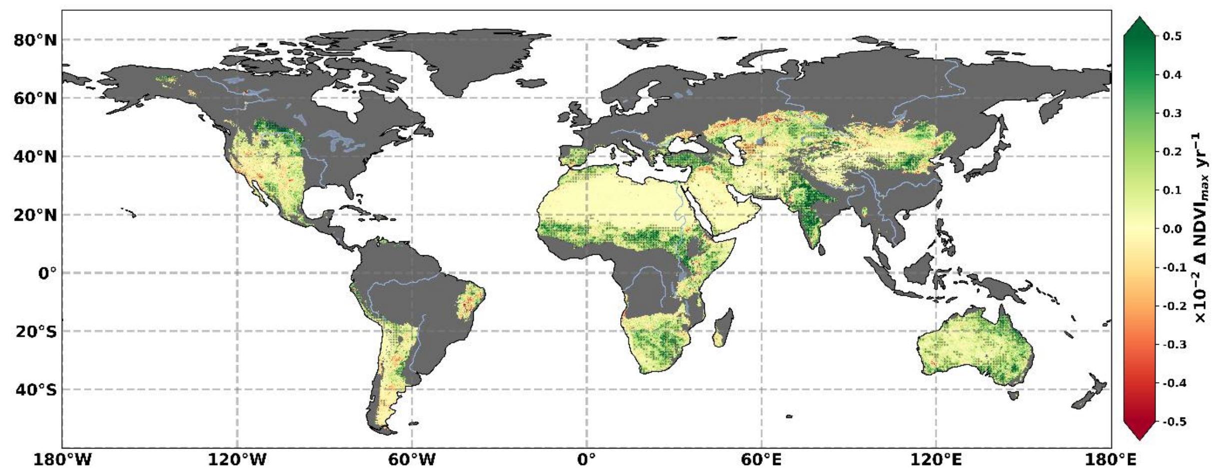


Figure 3.6 Trend in the Annual Maximum NDVI 1982-2015 (Global Inventory Modelling and Mapping Studies NDVI3g v1) calculated using the Theil-Sen estimator which is a median based estimator, and is robust to outliers. Non-dryland regions (Aridity Index > 0.65) are masked in grey.

A number of studies have used **satellite-based remote sensing** to investigate long-term changes in the vegetation and thus identify parts of the drylands undergoing desertification. Satellite data provides information at the resolution of the sensor which can be relatively coarse (up to 25 km) and interpretations of the data at sub-pixel levels are challenging. The most widely used remotely sensed vegetation index is the NDVI providing a measure of canopy greenness, which is related to the quantity of standing biomass (Bai et al., 2008; de Jong et al., 2011; Fensholt et al., 2012; Andela et al., 2013; Fensholt et al., 2015; Le et al., 2016; Figure 3.5). A main challenge associated with NDVI is that although biomass and productivity are closely related in some systems, they can differ widely when looking across land uses and ecosystem types, giving a false positive in some instances (Pattison et al., 2015; Aynekulu et al., 2017). For example, bush encroachment in rangelands and intensive monocropping with high fertiliser application gives an indication of increased productivity in satellite data though these could be considered as land degradation. According to this measure there are regions undergoing desertification, however, the drylands are greening on average (Figure 3.6).

A simple linear trend in NDVI is an unsuitable measure for dryland degradation for several reasons (Wessels et al., 2012; de Jong et al., 2013; Higginbottom and Symeonakis, 2014; Le et al., 2016). NDVI is strongly coupled to precipitation in drylands where precipitation has high inter-annual variability. This means that NDVI trend can be dominated by any precipitation trend and is sensitive to wet or dry periods, particularly if they fall near the beginning or end of the time series. Degradation may only occur during part of the time series, while NDVI is stable or even improving during the rest of the time series. This reduces the strength and representativeness of a linear trend. Other factors such as CO_2 fertilisation also influence the NDVI trend. Various techniques have been proposed to address these issues, including the residual trends (RESTREND) method to account for rainfall variability (Evans and Geerken, 2004), time-series break point identification methods to find major shifts in the vegetation trends (de Jong et al., 2013; Verbesselt et al., 2010a) and methods to explicitly account for the effect of CO_2 fertilisation (Le et al., 2016).

Using the RESTREND method, Andela et al. (2013) found that human activity contributed to a mixture of improving and degrading regions in drylands. In some locations these regions differed substantially from those identified using the NDVI trend alone, including an increase in the area being desertified in southern Africa and northern Australia, and a decrease in southeast and west Australia and Mongolia. De Jong et al. (2013) examined the NDVI time series for major shifts in vegetation activity and found that 74% of drylands experienced such a shift between 1981 and 2011. This suggests that monotonic linear trends are unsuitable for accurately capturing the changes that have occurred in the majority of the drylands. Le et al. (2016) explicitly accounted for CO_2 fertilisation effect and found that the extent of degraded areas in the world is 3% larger when compared to the linear NDVI trend.

Besides NDVI, there are many vegetation indices derived from satellite data in the optical and infrared wavelengths. Each of these datasets has been derived to overcome some limitation in existing indices. Studies have compared vegetation indices globally (Zhang et al., 2017) and specifically over drylands (Wu, 2014). In general, the data from these vegetation indices are available only since around 2000, while NDVI data is available since 1982. With less than 20 years of data, the trend analysis remains problematic with vegetation indices other than NDVI. However, given the various advantages in terms of resolution and other characteristics, these newer vegetation indices will become more useful in the future as more data accumulates.

Vegetation Optical Depth (VOD) has been available since the 1980s. VOD is based on microwave measurements and is related to total above ground biomass water content. Unlike NDVI which is only sensitive to green canopy cover, VOD is also sensitive to water in woody parts of the vegetation and hence provides a view of vegetation changes that can be complementary to NDVI. Liu et al. (2013) used VOD trends to investigate biomass changes and found that VOD was closely related to precipitation changes in drylands. To complement their work with NDVI, Andela et al. (2013) also applied the RESTREND method to VOD. By interpreting NDVI and VOD trends together they were able to differentiate changes to the herbaceous and woody components of the biomass. They reported that many dryland regions are experiencing an increase in the woody fraction often associated with shrub encroachment and suggest that this was aided by CO_2 fertilisation.

A major shortcoming of these studies based on vegetation datasets derived from satellite sensors is that they do not account for changes in vegetation composition, thus leading to inaccuracies in the estimation of the extent of degraded areas in drylands. For example, drylands of Eastern Africa currently face growing encroachment of invasive plant species, such as *Prosopis juliflora* (Ayanu et al., 2015), which constitutes land degradation since it leads to losses in economic productivity of affected areas but appears as a greening in the satellite data. Another case study in central Senegal found degradation manifested through a reduction in species richness despite satellite observed greening (Herrmann and Tappan, 2013). A number of efforts to identify changes in vegetation composition from satellites have been made (Brandt et al., 2016a,b; Evans and Geerken, 2006; Geerken, 2009; Geerken et al., 2005; Verbesselt et al., 2010a,b). These depend on well-identified reference NDVI time series for particular vegetation groupings, can only differentiate vegetation types that have distinct spectral phenology signatures and require extensive ground observations for validation. A recent alternative approach to differentiating woody from herbaceous vegetation involves the combined use of optical/infrared based vegetation indices, indicating greenness, with microwave based Vegetation Optical Depth (VOD) which is sensitive to both woody and leafy vegetation components (Andela et al., 2013; Tian et al., 2017).

Biophysical models use global data sets that describe climate patterns and soil groups, combined with observations of land use, to define classes of potential productivity and map general land degradation (Gibbs and Salmon, 2015). All biophysical models have their own set of assumptions and limitations that contribute to their overall uncertainty, including: model structure; spatial scale; data requirements (with associated errors); spatial heterogeneities of socioeconomic conditions; and agricultural technologies used. Models have been used to estimate the vegetation productivity potential of land (Cai et al., 2011) and to understand the causes of observed vegetation changes. Zhu et al. (2016) used an ensemble of ecosystem models to investigate causes of vegetation changes from 1982-2009, using a factorial simulation approach. They found CO_2 fertilisation to be the dominant effect globally though climate and land cover change were the dominant effects in various dryland locations. Borrelli et al. (2017) modelled that about 6.1% of the global land area experienced very high soil erosion rates (exceeding $10\ \mathrm{Mg\ ha^{-1}\ yr^{-1}}$) in 2012, particularly in South America, Africa, and Asia.

Overall, improved estimation and mapping of areas undergoing desertification are needed. This requires a combination of rapidly expanding sources of remotely sensed data, ground observations and new modelling approaches. This is a critical gap, especially in the context of measuring progress towards achieving the land degradation-neutrality target by 2030 in the framework of SDGs.

3.2.1.2. Regional Scale

While global scale studies provide information for any region, there are numerous studies that focus on sub-continental scales, providing more in-depth analysis and understanding. Regional and local studies are important to detect location-specific trends in desertification and heterogeneous influences of climate change on desertification. However, these regional and local studies use a wide variety of methodologies, making direct comparisons difficult. For details of the methodologies applied by each study refer to the individual papers.

3.2.1.2.1 Africa

It is estimated that 46 out of the 54 countries in Africa are vulnerable to desertification, with some already affected (Prăvălie, 2016). Moderate or higher severity degradation over recent decades have been identified in many river basins including the Nile (42% of area), Niger (50%), Senegal (51%), Volta (67%), Limpopo (66%) and Lake Chad (26%) (Thiombiano and Tourino-Soto, 2007).

The Horn of Africa is getting drier (Damberg and AghaKouchak, 2014; Marshall et al., 2012) exacerbating the desertification already occurring (Oroda, 2001). The observed decline in vegetation cover is diminishing ecosystem services (Pricope et al., 2013). Based on NDVI residuals, Kenya experienced persistent negative (positive) trends over 21.6% (8.9%) of the country, for the period 1992–2015 (Gichenje and Godinho, 2018). Fragmentation of habitats, reduction in the range of livestock grazing, higher stocking rates are considered to be the main drivers for vegetation structure loss in the rangelands of Kenya (Kihiu, 2016; Otuoma et al., 2009)

Despite desertification in the Sahel being a major concern since the 1970s, wetting and greening conditions have been observed in this region over the last three decades (Anyamba and Tucker, 2005; Huber et al., 2011; Brandt et al., 2015; Rishmawi et al., 2016; Tian et al., 2016; Leroux et al., 2017; Herrmann et al., 2005; Damberg and AghaKouchak, 2014). Cropland areas in the Sahel region of West Africa have doubled since 1975, with settlement area increasing by about 150% (Traore et al., 2014). Thomas and Nigam (2018) found that the Sahara expanded by 10% over the 20th century based on annual rainfall. In Burkina Faso, Dimobe et al. (2015) estimated that from 1984 to 2013, bare soils and agricultural lands increased by 18.8% and 89.7%, respectively, while woodland, gallery forest, tree savannas, shrub savannas and water bodies decreased by 18.8%, 19.4%, 4.8%, 45.2% and 31.2%, respectively. In Fakara region in Niger, 5% annual reduction in herbaceous yield between 1994 and 2006 was largely explained by changes in land use, grazing pressure and soil fertility (Hiernaux et al., 2009). Aladejana et al. (2018) found that between 1986 and 2015, 18.6% of the forest cover around the Owena River basin was lost. For the period 1982–2003, Le et al. (2012) found that 8% of the Volta River basin's landmass had been degraded with this increasing to 65% after accounting for the effects of CO_2 (+NO_x) fertilisation.

Greening has also been observed in parts of Southern Africa but it is relatively weak compared to other regions of the continent (Helldén and Tottrup, 2008; Fensholt et al., 2012). However, greening can be accompanied by desertification when factors such as decreasing species richness, changes in species composition and shrub encroachment are observed (Smith et al., 2013; Herrmann and Tappan, 2013; Kaptué et al., 2015; Herrmann and Sop, 2016; Saha et al., 2015) (3.1.4, 3.7.3). In the Okavango river Basin in Southern Africa, conversion of land towards higher utilisation intensities, unsustainable agricultural practises and overexploitation of the savanna ecosystems have been observed in recent decades (Weinzierl et al., 2016).

In arid Algerian High Plateaus, desertification due to both climatic and human causes led to the loss of indigenous plant biodiversity between 1975 and 2006 (Hirche et al., 2011). Ayoub (1998) identified 64 M ha in Sudan as degraded, with the Central North Kordofan state being most affected. However, reforestation measures in the last decade sustained by improved rainfall conditions have led to low-medium regrowth conditions in about 20% of the area (Dawelbait and Morari, 2012). In Morocco, areas affected by desertification were dominantly on plains with high population and livestock pressure (del Barrio et al., 2016; Kouba et al., 2018; Lahlaoi et al., 2017). The annual costs of soil degradation were estimated at about 1% of Gross Domestic Product (GDP) in Algeria and Egypt, and about 0.5% in Morocco and Tunisia (Réquier-Desjardins and Bied-Charreton, 2006).

3.2.1.2.2 Asia

Prăvălie (2016) found that desertification is currently affecting 38 of 48 countries in Asia. The changes in drylands in Asia over the period 1982–2011 were mixed, with some areas experiencing vegetation improvement while others showed reduced vegetation (Miao et al., 2015a). Major river basins undergoing salinisation include: Indo-Gangetic Basin in India (Lal and Stewart, 2012), Indus Basin in Pakistan (Aslam and Prathapar, 2006), Yellow River Basin in China (Chengrui and Dregne, 2001), Yinchuan Plain, in China (Zhou et al., 2013), Aral Sea Basin of Central Asia (Cai et al., 2003; Pankova, 2016; Qadir et al., 2009).

Helldén and Tottrup (2008) highlighted a greening trend in East Asia between 1982 and 2003. Over the past several decades, air temperature and the rainfall increased in the arid and hyper-arid region of Northwest China (Chen et al., 2015; Wang et al., 2017). Within China, rainfall erosivity has shown a positive trend in dryland areas between 1961 and 2012 (Yang and Lu, 2015). While water erosion area in Xinjiang China, has decreased by 23.2%, erosion considered as severe or intense was still increasing (Zhang et al., 2015). Xue et al. (2017) used remote sensing data covering 1975 to 2015 to show that wind-driven desertified land in north Shanxi in China had expanded until 2000, before contracting again. Li et al. (2012) used satellite data to identify desertification in Inner Mongolia China and found a link between policy changes and the locations and extent of human-caused desertification. Several oasis regions in China have seen increases in cropland area, while forests, grasslands and available water resources have decreased (Fu et al. 2017; Muyibul et al., 2018; Xie et al., 2014). Between 1990 and 2011 15.3% of Hogno Khaan nature reserve in central Mongolia was subjected to desertification (Lamchin et al., 2016). Using satellite data Liu et al. (2013) found the area of Mongolia undergoing non-climatic desertification was associated with increases in goat density and wildfire occurrence.

In Central Asia, drying up of the Aral Sea is continuing having negative impacts on regional microclimate and human health (Issanova and Abuduwaili, 2017; Lioubimtseva, 2015; Micklin, 2016; Xi and Sokolik, 2015). Half of the region's irrigated lands, especially in the Amudarya and Syrdarya river basins, were affected by secondary salinisation (Qadir et al., 2009). Le et al., (2016) showed that about 57% of croplands in Kazakhstan and about 20% of croplands in Kyrgyzstan had lost in their vegetation productivity between 1982 and 2006. Chen et al. (2019) indicated that about 58% of the grasslands in the region lost in their vegetation productivity between 1999 and 2015. Anthropogenic factors were the main driver of this loss in Turkmenistan and Uzbekistan, while the role of human drivers was smaller than that of climate-related factors in Tajikistan and Kyrgyzstan (Chen et al., 2019). The total costs of land degradation in Central Asia were estimated to equal about USD 6 billion annually (Mirzabaev et al., 2016).

Damberg and AghaKouchak (2014) found that parts of South Asia experienced drying over the last three decades. More than 75% of the area of northern, western and southern Afghanistan is affected by overgrazing and deforestation (UNEP-GEF, 2008). Desertification is a serious problem in Pakistan with a wide range of human and natural causes (Irshad et al., 2007; Lal, 2018). Similarly, desertification affects parts of India (Kundu et al., 2017; Dharumarajan et al., 2018; Christian et al., 2018). Using satellite data to map various desertification processes, Ajai et al. (2009) identified 81.4 M ha were subject to various processes of desertification in India in 2005, while salinisation affected 6.73 M ha in the country (Singh, 2009).

Saudi Arabia is highly vulnerable to desertification (Ministry of Energy Industry and Mineral Resources, 2016), with this vulnerability expected to increase in the north-western parts of the country in the coming decades. Yahiya (2012) found that Jazan, south-western Saudi Arabia, lost about 46% of its vegetation cover from 1987 to 2002. Droughts and frequent dust storms were shown to impose adverse impacts over Saudi Arabia especially under global warming and future climate change (Hasanean et al., 2015). In north-west Jordan, 18% of the area was prone to severe to very severe desertification (Al-Bakri et al., 2016). Large parts of the Syrian drylands have been identified as undergoing desertification (Evans and Geerken, 2004; Geerken and Ilaiwi, 2004). Moridnejad et al. (2015) identified newly desertified regions in the Middle East based on dust sources, finding that these regions accounted for 39% of all detected dust source points. Desertification has increased substantially in Iran since the 1930s. Despite numerous efforts to rehabilitate degraded areas, it still poses a major threat to agricultural livelihoods in the country (Amiraslani and Dragovich, 2011).

3.2.1.2.3 Australia

Damberg and AghaKouchak (2014) found that wetter conditions were experienced in northern Australia over the last three decades with widespread greening observed between 1981 and 2006 over much of Australia, except for eastern Australia where large areas were affected by droughts from 2002 to 2009 based on Advanced High Resolution Radiometer (AVHRR) satellite data (Donohue, McVicar, and Roderick, 2009). For the period 1982–2013, Burrell et al. (2017) also found widespread greening over Australia including eastern Australia over the post-drought period. This dramatic change in the trend found for eastern Australia emphasises the dominant role played by precipitation in the drylands. Degradation due to anthropogenic activities and other causes affects over 5% of Australia, particularly near the central west coast. Jackson and Prince (2016) used a local NPP scaling approach applied with MODIS derived vegetation data to quantify degradation in a dryland watershed in Northern Australia from 2000 to 2013. They estimated that 20% of the watershed was degraded. Salinisation has also been found to be degrading parts of the Murray-Darling Basin in Australia (Rengasamy, 2006). Eldridge and Soliveres (2014) examined areas undergoing woody encroachment in eastern Australia and found that rather than degrading the landscape, the shrubs often enhanced ecosystem services.

3.2.1.2.4 Europe

Drylands cover 33.8% of northern Mediterranean countries; approximately 69% of Spain, 66% of Cyprus, and between 16% and 62% in Greece, Portugal, Italy and France (Zdruli, 2011). The European Environment Agency (EEA) indicated that 14 M ha, i.e. 8% of the territory of the European Union (in Bulgaria, Cyprus, Greece, Italy, Romania, Spain and Portugal), had a "very high" and "high sensitivity" to desertification (European Court of Auditors, 2018). This figure increases to 40 M ha (23% of the EU territory) if "moderately" sensitive areas are included (Prăvălie et al., 2017; European Court of Auditors, 2018). Desertification in the region is driven by irrigation developments and encroachment of cultivation on rangelands (Safriel, 2009) caused by population growth, agricultural policies and markets. According to a recent assessment report (ECA, 2018), Europe is increasingly affected by desertification leading to significant consequences on land use, particularly in Portugal, Spain, Italy, Greece, Malta, Cyprus, Bulgaria and Romania. Using the Universal Soil Loss Equation, it was estimated that soil erosion can be as high as 300 t $ha^{-1}yr^{-1}$ (equivalent to a net loss of 18 mm yr^{-1}) in Spain (López-Bermúdez, 1990). For the badlands region in south-east Spain, however, it was shown that biological soil crusts effectively prevent soil erosion (Lázaro et al., 2008). In Mediterranean Europe, Guerra et al. (2016) found a reduction of erosion due to greater effectiveness of soil erosion prevention between 2001 and 2013. Helldén and Tottrup (2008) observed a greening trend in the Mediterranean between 1982–2003, while Fensholt et al. (2012) also show a dominance of greening in Eastern Europe.

In Russia, at the beginning of the 2000s, about 7% of the total area (i.e. ~130 M ha) was threatened by desertification (Gunin and Pankova, 2004; Kust et al., 2011). Turkey is considered highly vulnerable to drought, land degradation and desertification (Türkeş, 1999; Türkeş, 2003). About 60% of Turkey's land area is characterised with hydro-climatological conditions favourable for desertification (Türkeş, 2013). ÇEMGM (2017) estimated that about half of Turkey's land area (48.6%) is prone to moderate to high desertification.

3.2.1.2.5 North America

Drylands cover approximately 60% of Mexico. According to Pontifes et al. (2018), 3.5% of the area was converted from natural vegetation to agriculture and human settlements between 2002 to 2011.

The region is highly vulnerable to desertification due to frequent droughts and floods (Méndez and Magaña, 2010; Stahle et al., 2009; Becerril-Pina Rocio et al., 2015).

For the period 2000-2011 the overall difference between potential and actual NPP in different land capability classes in the south-western United States was 11.8% (Noojipady et al., 2015); reductions in grassland-savanna and livestock grazing area and forests were the highest. Bush encroachment is observed over a fairly wide area of western USA grasslands; including Jornada Basin within the Chihuahuan Desert, and is spreading at a fast rate despite grazing restrictions intended to curb the spread (Yanoff and Muldavin, 2008; Browning and Archer, 2011; Van Auken, 2009; Rachal et al., 2012). In comparing sand dune migration patterns and rates between 1995 and 2014, Potter and Weigand (2016) established that the area covered by stable dune surfaces, and sand removal zones, decreased while sand accumulation zones increased from 15.4 to 25.5 km^2 for Palen dunes in Southern California Desert, while movement of Kelso Dunes is less clear (Lam et al., 2011). Within the United States, average soil erosion rates on all croplands decreased by about 38% between 1982-2003 due to better soil management practices (Kertis, 2003).

3.2.1.2.6 Central and South America

Morales et al. (2011) indicated that desertification costs between 8 and 14% of gross agricultural product in many Central and South American countries. Parts of the dry Chaco and Caldenal regions in Argentina have undergone widespread degradation over the last century (Verón et al., 2017; Fernández et al., 2009). Bisigato and Laphitz (2009) identified overgrazing as a cause of desertification in the Patagonian Monte region of Argentina. Vieira et al. (2015) found that 94% of northeast Brazilian drylands were susceptible to desertification. It is estimated that up to 50% of the area was being degraded due to frequent prolonged droughts and clearing of forests for agriculture. This land-use change threatens the extinction of around 28 native species (Leal et al., 2005). In Central Chile, dryland forest and shrubland area was reduced by 1.7% and 0.7%, respectively, between 1975-2008 (Schulz et al., 2010).

3.2.2. Attribution of Desertification

Desertification is a result of complex interactions within coupled social-ecological systems. Thus, the relative contributions of climatic, anthropogenic and other drivers of desertification vary depending on specific socioeconomic and ecological contexts. The high natural climate variability in dryland regions is a major cause of vegetation changes but does not necessarily imply degradation. Drought is not degradation as the land productivity may return entirely once the drought ends (Kassas, 1995). However, if droughts increase in frequency, intensity and/or duration they may overwhelm the vegetation's ability to recover (ecosystem resilience, Prince et al., 2018), causing degradation. Assuming a stationary climate and no human influence, rainfall variability results in fluctuations in vegetation dynamics which can be considered temporary as the ecosystem tends to recover with rainfall, and desertification does not occur (Ellis, 1995; Vetter, 2005; von Wehrden et al., 2012). Climate change on the other hand, exemplified by a non-stationary climate, can gradually cause a persistent change in the ecosystem through aridification and CO_2 changes. Assuming no human influence, this 'natural' climatic version of desertification may take place rapidly, especially when thresholds are reached (Prince et al., 2018), or over longer periods of time as the ecosystems slowly adjust to a new climatic norm through progressive changes in the plant community composition. Accounting for this climatic variability is required before attributions to other causes of desertification can be made.

For attributing vegetation changes to climate versus other causes, rain use efficiency (RUE - the change in net primary productivity (NPP) per unit of precipitation) and its variations in time have been used (Prince et al., 1998). Global applications of RUE trends to attribute degradation to climate or other (largely human) causes has been performed by Bai et al. (2008) and Le et al.(2016) (3.2.1.1).

The RESTREND (residual trend) method analyses the correlation between annual maximum NDVI (or other vegetation index as a proxy for NPP) and precipitation by testing accumulation and lag periods for the precipitation (Evans and Geerken, 2004). The identified relationship with the highest correlation represents the maximum amount of vegetation variability that can be explained by the precipitation, and corresponding RUE values can be calculated. Using this relationship, the climate component of the NDVI time series can be reconstructed, and the difference between this and the original time series (the residual) is attributed to anthropogenic and other causes.

The RESTREND method, or minor variations of it, have been applied extensively. (Herrmann and Hutchinson, 2005) concluded that climate was the dominant causative factor for widespread greening in the Sahel region from 1982 to 2003, and anthropogenic and other factors were mostly producing land improvements or no change. However, pockets of desertification were identified in Nigeria and Sudan. Similar results were also found from 1982 to 2007 by Huber et al. (2011). Wessels et al. (2007) applied RESTREND to South Africa and showed that RESTREND produced a more accurate identification of degraded land than RUE alone. RESTREND identified a smaller area undergoing desertification due to non-climate causes compared to the NDVI trends. Liu et al. (2013) extended the climate component of RESTREND to include temperature and applied this to VOD observations of the cold drylands of Mongolia. They found the area undergoing desertification due to non-climatic causes is much smaller than the area with negative VOD trends. RESTREND has also been applied in several other studies to the Sahel (Leroux et al., 2017), Somalia (Omuto et al., 2010), West Africa (Ibrahim et al., 2015), China (Li et al., 2012; Yin et al., 2014), Central Asia (Jiang et al., 2017), Australia (Burrell et al., 2017) and globally (Andela et al., 2013). In each of these studies the extent to which desertification can be attributed to climate versus other causes varies across the landscape.

These studies represent the best regional, remote sensing based attribution studies to date, noting that RESTREND and RUE have some limitations (Higginbottom and Symeonakis, 2014). Vegetation growth (NPP) changes slowly compared to rainfall variations and may be sensitive to rainfall over extended periods (years) depending on vegetation type. Detection of lags and the use of weighted antecedent rainfall can partially address this problem though most studies do not do this. The method addresses changes since the start of the time series, it cannot identify whether an area is already degraded at the start time. It is assumed that climate, particularly rainfall, are principal factors in vegetation change which may not be true in more humid regions.

Another assumption in RESTREND is that any trend is linear throughout the period examined. That is, there are no discontinuities (break points) in the trend. Browning et al. (2017) have shown that break points in NDVI time series reflect vegetation changes based on long-term field sites. To overcome this limitation, Burrell et al. (2017) introduced the Time Series Segmentation-RESTREND (TSS-RESTREND) which allows a breakpoint or turning point within the period examined (Figure 3.7). Using TSS-RESTREND over Australia they identified more than double the degrading area than could be identified with a standard RESTREND analysis. The occurrence and drivers of abrupt change (turning points) in ecosystem functioning were also examined by Horion et al. (2016) over the semi-arid Northern Eurasian agricultural frontier. They combined trend shifts in RUE, field data and expert knowledge, to map environmental hotspots of change and attribute them to climate and human activities. One third of the area showed significant change in RUE, mainly occurring around the fall of the Soviet Union (1991) or as the result of major droughts. Recent human-induced turning points in ecosystem functioning were uncovered nearby Volgograd (Russia) and around Lake Balkhash (Kazakhstan), attributed to recultivation, increased salinisation, and increased grazing.

Attribution of vegetation changes to human activity has also been done within modelling frameworks. In these methods ecosystem models are used to simulate potential natural vegetation dynamics, and this is compared to the observed state. The difference is attributed to human activities. Applied to the Sahel region during the period of 1982–2002, it showed that people had a minor influence on vegetation changes (Seaquist et al., 2009). Similar model/observation comparisons performed globally found that CO_2 fertilisation was the strongest forcing at global scales, with climate having regionally varying effects (Mao et al., 2013; Zhu et al., 2016). Land use/land cover change was a dominant forcing in localised areas. The use of this method to examine vegetation changes in China (1982–2009) attributed most of the greening trend to CO_2 fertilisation and nitrogen (N) deposition (Piao et al., 2015). However in some parts of northern and western China, which includes large areas of drylands, Piao et al. (2015) found climate changes could be the dominant forcing. In the northern extratropical land surface, the observed greening was consistent with increases in greenhouse gases (notably CO_2) and the related climate change, and not consistent with a natural climate that does not include anthropogenic increase in greenhouse gases (Mao et al., 2016). While many studies found widespread influence of CO_2 fertilisation, it is not ubiquitous, for example, Lévesque et al. (2014) found little response to CO_2 fertilisation in some tree species in Switzerland/northern Italy.

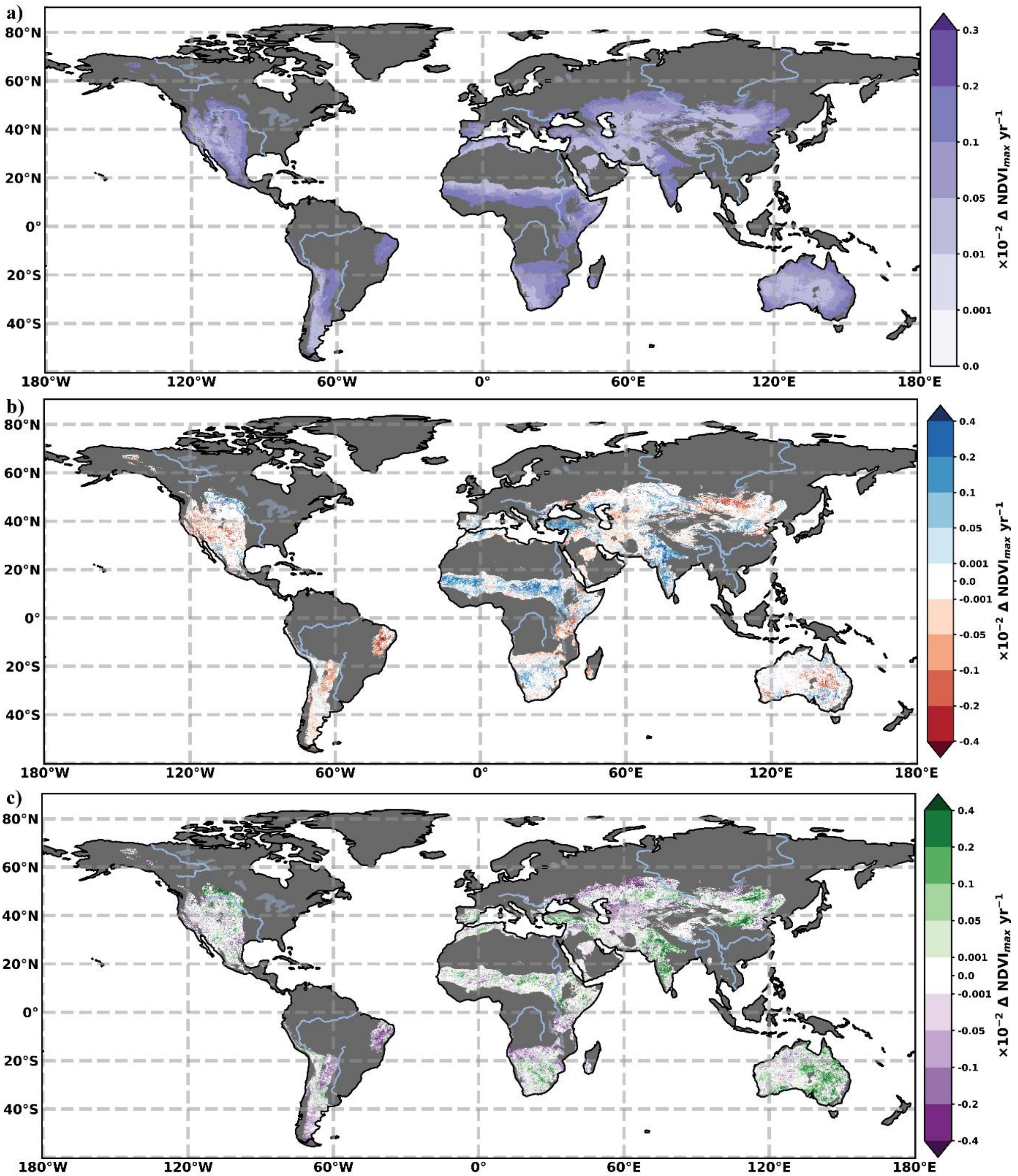


Figure 3.7 The Drivers of Dryland Vegetation Change. The mean annual change in NDVImax between 1982 and 2015 (See Figure 3.6 for total change using Global Inventory Modelling and Mapping Studies NDVI3g v1 dataset) attributable to a) CO_2 fertilisation b) climate and c) land use. The change attributable to CO_2 fertilisation was calculated using the CO_2 fertilisation relationship described in (Franks et al., 2013). The Time Series Segmented Residual Trends (TSS-RESTREND) method (Burrell et al., 2017) applied to the CO_2 adjusted NDVI was used to separate Climate and Land Use. A multi climate dataset ensemble was used to reduce the impact of dataset errors (Burrell et al., 2018). Non-dryland regions (Aridity Index > 0.65) are masked in dark grey. Areas where the change did not meet the multi-run ensemble significance criteria, or are smaller than the error in the sensors (±0.00001) are masked in white.

Using multiple extreme event attribution methodologies, Uhe et al. (2018) shows that the dominant influence for droughts in Eastern Africa during October to December 'short rains' season is the prevailing tropical SST patterns, although temperature trends mean that the current drought conditions are hotter than it would have been without climate change. Similarly, Funk et al. (2019) found that 2017 March-June East African drought was influenced by Western Pacific SST, with high SST conditions attributed to climate change.

There are numerous local case studies on attribution of desertification, which use different periods, focus on different land uses and covers, and consider different desertification processes. For example, two-thirds of the observed expansion of the Sahara Desert from 1920–2003 has been attributed to natural climate cycles (the cold phase of Atlantic Multi-Decadal Oscillation and Pacific Decadal Oscillation) (Thomas and Nigam, 2018). Some studies consider drought to be the main driver of desertification in Africa (e.g. Masih et al., 2014). However, other studies suggest that although droughts may contribute to desertification, the underlying causes are human activities (Kouba et al., 2018). Brandt et al. (2016a) found that woody vegetation trends are negatively correlated with human population density. Changes in land use, water pumping and flow diversion have enhanced drying of wetlands and salinisation of freshwater aquifers in Israel (Inbar, 2007). The dryland territory of China has been found to be very sensitive to both climatic variations and land use/land cover changes (Fu et al., 2000; Liu and Tian, 2010; Zhao et al., 2013, 2006). Feng et al. (2015) shows that socioeconomic factors were dominant in causing desertification in north Shanxi, China, between 1983 and 2012, accounting for about 80% of desertification expansion. Successful grass establishment has been impeded by overgrazing and nutrient depletion leading to the encroachment of shrubs into the northern Chihuahuan Desert (USA) since the mid-1800s (Kidron and Gutschick, 2017). Human activities led to rangeland degradation in Pakistan and Mongolia during 2000-2011 (Lei et al., 2011). More equal shares of climatic (temperature and precipitation trends) and human factors were attributed for changes in rangeland condition in China (Yang et al., 2016).

This kaleidoscope of local case studies demonstrates how attribution of desertification is still challenging for several reasons. Firstly, desertification is caused by an interaction of different drivers which vary in space and time. Secondly, in drylands, vegetation reacts closely to changes in rainfall so the effect of rainfall changes on biomass needs to be 'removed' before attributing desertification to human activities. Thirdly, human activities and climatic drivers impact vegetation/ecosystem changes at different rates. Finally, desertification manifests as a gradual change in ecosystem composition and structure (e.g., woody shrub invasion into grasslands). Although initiated at a limited location, ecosystem change may propagate throughout an extensive area via a series of feedback mechanisms. This complicates the attribution of desertification to human and climatic causes as the process can develop independently once started.

Rasmussen et al. (2016) studied the reasons behind the overall lack of scientific agreement in trends of environmental changes in the Sahel, including their causes. The study indicated that these are due to differences in conceptualisations and choice of indicators, biases in study site selection, differences in methods, varying measurement accuracy, differences in time and spatial scales. High resolution, multi-sensor airborne platforms provide a way to address some of these issues (Asner et al., 2012).

The major conclusion of this section is that, with all the shortcomings of individual case studies, relative roles of climatic and human drivers of desertification are location-specific and evolve over time (*high confidence*). Biophysical research on attribution and socio-economic research on drivers of land degradation have long studied the same topic, but in parallel, with little interdisciplinary integration. Interdisciplinary work to identify typical patterns, or typologies, of such interactions of biophysical and human drivers of desertification (not only of dryland vulnerability), and their relative shares, done globally in comparable ways, will help in the formulation of better informed policies to address desertification and achieve land degradation neutrality.

3.3. Desertification Feedbacks to Climate

While climate change can drive desertification (3.1.4.1), the process of desertification can also alter the local climate providing a feedback (Sivakumar, 2007). These feedbacks can alter the carbon cycle, and hence the level of atmospheric CO_2 and its related global climate change, or they can alter the surface energy and water budgets directly impacting the local climate. While these feedbacks occur in all climate zones (Chapter 2), here we focus on their effects in dryland regions and assess the literature concerning the major desertification feedbacks to climate. The main feedback pathways discussed throughout section 3.3 are summarised in Figure 3.8.

Drylands are characterised by limited soil moisture compared to humid regions. Thus, the sensible heat (heat that causes the atmospheric temperature to rise) accounts for more of the surface net radiation than latent heat (evaporation) in these regions (Wang and Dickinson, 2013). This tight coupling between the surface energy balance and the soil moisture in semi-arid and dry sub-humid zones makes these regions susceptible to land-atmosphere feedback loops that can amplify changes to the water cycle (Seneviratne et al., 2010). Changes to the land surface caused by desertification can change the surface energy budget, altering the soil moisture and triggering these feedbacks.

3.3.1. Sand and Dust Aerosols

Sand and mineral dust are frequently mobilised from sparsely vegetated drylands forming “sand storms” or “dust storms” (UNEP et al., 2016). The African continent is the most important source of desert dust, perhaps 50% of atmospheric dust comes from the Sahara (Middleton, 2017). Ginoux et al. (2012) estimated that 25% of global dust emissions have anthropogenic origins, often in drylands. These events can play an important role in the local energy balance. Through reducing vegetation cover and drying the surface conditions, desertification can increase the frequency of these events. Biological or structural soil crusts have been shown to effectively stabilise dryland soils and thus their loss, due to intense land use and/or climate change, can be expected to cause an increase in sand and dust storms (*high confidence*) (Rajot et al., 2003; Field et al., 2010; Rodriguez-Caballero et al., 2018). These sand and dust aerosols impact the regional climate in several ways (Choobari et al., 2014). The direct effect is the interception, reflection and absorption of solar radiation in the atmosphere, reducing the energy available at the land surface and increasing the temperature of the atmosphere in layers with sand and dust present (Kaufman et al., 2002; Middleton, 2017; Kok et al., 2018). The heating of the dust layer can alter the relative humidity and atmospheric stability, which can change cloud lifetimes and water content. This has been referred to as the semi-direct effect (Huang et al., 2017). Aerosols also have an indirect effect on climate through their role as cloud condensation nuclei, changing cloud radiative properties as well as the evolution and development of precipitation (Kaufman et al., 2002). While these indirect effects are more variable than the direct effects, depending on the types and amounts of aerosols present, the general tendency is toward an increase in the number, but a reduction in the size of cloud droplets, increasing the cloud reflectivity and decreasing the chances of precipitation. These effects are referred to as aerosol-radiation and aerosol-cloud interactions (Boucher et al., 2013).

There is *high confidence* that there is a negative relationship between vegetation green-up and the occurrence of dust storms (Engelstaedter et al., 2003; Fan et al., 2015; Yu et al., 2015; Zou and Zhai, 2004). Changes in groundwater can affect vegetation and the generation of atmospheric dust (Elmore et al., 2008). This can occur through groundwater processes such as the vertical movement of salt to the surface causing salinisation, supply of near surface soil moisture, and sustenance of groundwater dependent vegetation. Groundwater dependent ecosystems have been identified in many dryland regions around the world (Decker et al., 2013; Lamontagne et al., 2005; Patten et al., 2008). In these locations declining groundwater levels can decrease vegetation cover. Cook et al., (2009) found that dust aerosols intensified the “dust bowl” drought in North America during the 1930s.

By decreasing the amount of green cover and hence increasing the occurrence of sand and dust storms, desertification will increase the amount of shortwave cooling associated with the direct effect (*high confidence*). There is *medium confidence* that the semi-direct and indirect effects of this dust would tend to decrease precipitation and hence provide a positive feedback to desertification (Huang et al., 2009; Konare et al., 2008; Rosenfeld et al., 2001; Solmon et al., 2012; Zhao et al., 2015). However, the combined effect of dust has also been found to increase precipitation in some areas (Islam and Almazroui, 2012; Lau et al., 2009; Sun et al., 2012). The overall combined effect of dust aerosols on desertification remains uncertain with *low agreement* between studies that find positive (Huang et al., 2014), negative (Miller et al., 2004) or no feedback on desertification (Zhao et al., 2015).

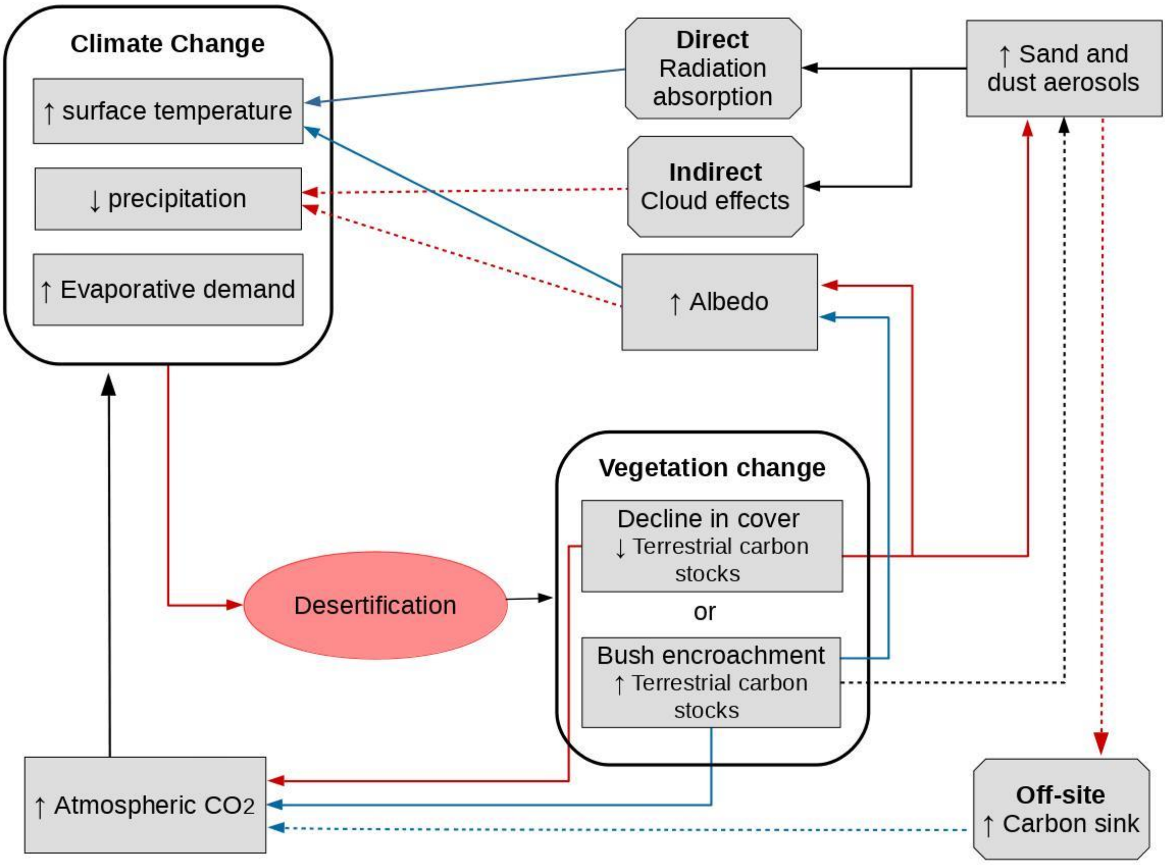


Figure 3.8 Schematic of main pathways through which desertification can feedback on climate as discussed in section 3.4. Note: red arrows indicate a positive effect. Blue arrows indicate a negative effect. Black arrows indicate an indeterminate effect (potentially both positive and negative). Solid arrows are direct while dashed arrows are indirect.

3.3.1.1. Off-site Feedbacks

Aerosols can act as a vehicle for the long-range transport of nutrients to oceans (Jickells et al., 2005; Okin et al., 2011) and terrestrial land surfaces (Das et al., 2013). In several locations, notably the Atlantic Ocean, west of northern Africa and the Pacific Ocean east of northern China, a considerable amount of mineral dust aerosols, sourced from nearby drylands, reaches the oceans. It was estimated that 60% of dust transported off Africa is deposited in the Atlantic Ocean (Kaufman et al., 2005), while 50% of the dust generated in Asia reaches the Pacific Ocean or further (Uno et al., 2009; Zhang et al., 1997). The Sahara is also a major source of dust for the Mediterranean basin (Varga et al., 2014). The direct effect of atmospheric dust over the ocean was found to be a cooling of the ocean surface (*limited evidence*, *high agreement*) (Evan and Mukhopadhyay, 2010; Evan et al., 2009) with the tropical North Atlantic mixed layer cooling by over 1°C (Evan et al., 2009).

It has been suggested that dust may act as a source of nutrients for the upper ocean biota, enhancing the biological activity and related C sink (*medium evidence*, *low agreement*) (Lenes et al., 2001; Shaw et al., 2008; Neuer et al., 2004). The overall response depends on the environmental controls on the ocean biota, the type of aerosols including their chemical constituents, and the chemical environment in which they dissolve (Boyd et al., 2010).

Dust deposited on snow can increase the amount of absorbed solar radiation leading to more rapid melting (Painter et al., 2018), impacting a region's hydrological cycle (*high confidence*). Dust deposition on snow and ice has been found in many regions of the globe (e.g. Painter et al., 2018; Kaspari et al., 2014; Qian et al., 2015; Painter et al. 2013), however quantification of the effect globally and estimation of future changes in the extent of this effect remain knowledge gaps.

3.3.2. Changes in Surface Albedo

Increasing surface albedo in dryland regions will impact the local climate, decreasing surface temperature and precipitation, and provide a positive feedback on the albedo (*high confidence*) (Charney et al., 1975). This albedo feedback can occur in desert regions worldwide (Zeng and Yoon, 2009). Similar albedo feedbacks have also been found in regional studies over the Middle East (Zaitchik et al., 2007), Australia (Evans et al., 2017; Meng et al., 2014a,b), South America (Lee and Berbery, 2012) and the USA (Zaitchik et al., 2013).

Recent work has also found albedo in dryland regions can be associated with soil surface communities of lichens, mosses and cyanobacteria (Rodriguez-Caballero et al., 2018). These communities compose the soil crust in these ecosystems and due to the sparse vegetation cover, directly influence the albedo. These communities are sensitive to climate changes with field experiments indicating albedo changes greater than 30% are possible. Thus, changes in these communities could trigger surface albedo feedback processes (*limited evidence, high agreement*) (Rutherford et al., 2017).

A further pertinent feedback relationship exists between changes in land-cover, albedo, C stocks and associated GHG emissions, particularly in drylands with low levels of cloud cover. One of the first studies to focus on the subject was Rotenberg and Yakir (2010), who used the concept of 'radiative forcing' to compare the relative climatic effect of a change in albedo with a change in atmospheric GHGs due to the presence of forest within drylands. Based on this analysis, it was estimated that the change in surface albedo due to the degradation of semi-arid areas has decreased radiative forcing in these areas by an amount equivalent to approximately 20% of global anthropogenic GHG emissions between 1970 and 2005 (Rotenberg and Yakir, 2010).

3.3.3. Changes in Vegetation and Greenhouse Gas Fluxes

Terrestrial ecosystems have the ability to alter atmospheric GHGs through a number of processes (Schlesinger et al., 1990). This may be through a change in plant and soil C stocks, either sequestering atmospheric carbon dioxide (CO_2) during growth or releasing C during combustion and respiration, or through processes such as enteric fermentation of domestic and wild ruminants that lead to the release of methane and nitrous oxide (Sivakumar, 2007). It is estimated that 241-470 Gt C is stored in dryland soils (top 1m Lal, 2004; Plaza et al., 2018). When evaluating the effect of desertification, the net balance of all the processes and associated GHG fluxes needs to be considered.

Desertification usually leads to a loss in productivity and a decline in above- and below-ground C stocks (Abril et al., 2005; Asner et al., 2003). Drivers such as overgrazing lead to a decrease in both plant and SOC pools (Abdalla et al., 2018). While dryland ecosystems are often characterised by open vegetation, not all drylands have low biomass and C stocks in an intact state (Lechmere-Oertel et al., 2005; Maestre et al., 2012). Vegetation types such as the subtropical thicket of South Africa have over 70 t C ha^{-1} in an intact state, greater than 60% of which is released into the atmosphere during degradation through overgrazing (Lechmere-Oertel et al., 2005; Powell, 2009). In comparison, semi-arid grasslands and savannas with similar rainfall, may have only 5-35 t C ha^{-1} (Scholes and Walker, 1993; Woomer et al., 2004)

At the same time, it is expected that a decline in plant productivity may lead to a decrease in fuel loads and a reduction in CO_2, nitrous oxide and methane emissions from fire. In a similar manner, decreasing productivity may lead to a reduction in ruminant animals that in turn would decrease methane emissions. Few studies have focussed on changes in these sources of emissions due to desertification and it remains a field that requires further research.

In comparison to desertification through the suppression of primary production, the process of woody plant encroachment can result in significantly different climatic feedbacks. Increasing woody plant cover in open rangeland ecosystems leads to an increase in woody C stocks both above- and below-ground (Asner et al., 2003; Hughes et al., 2006; Petrie et al., 2015; Li et al., 2016). Within the drylands of Texas, shrub encroachment led to a 32% increase in aboveground C stocks over a period of 69 years (3.8 t C ha^{1} to 5.0 t C ha^{-1}) (Asner et al., 2003). Encroachment by taller woody species, can lead to significantly higher observed biomass and C stocks, for example, encroachment by *Dichrostachys cinerea* and several Vachellia species in the sub-humid savannas of north-west South Africa led to an increase of 31–46 t C ha^{-1} over a 50–65 year period (1936–2001) (Hudak et al., 2003). In terms of potential changes in SOC stocks, the effect may be dependent on annual rainfall and soil type. Woody cover generally leads to an increase in SOC stocks in drylands that have less than 800 mm of annual rainfall, while encroachment can lead to a loss of soil C in more humid ecosystems (Barger et al., 2011; Jackson et al., 2002).

The suppression of the grass layer through the process of woody encroachment may lead to a decrease in C stocks within this relatively small C pool (Magandana, 2016). Conversely, increasing woody cover may lead to a decrease and even halt in surface fires and associated GHG emissions. In analysis of drivers of fire in southern Africa, Archibald et al. (2009) note that there is a potential threshold around 40% canopy cover, above which surface grass fires are rare. Whereas there have been a number of studies on changes in C stocks due to desertification in North America, southern Africa and Australia, a global assessment of the net change in C stocks as well as fire and ruminant GHG emissions due to woody plant encroachment has not been done yet.

3.4. Desertification Impacts on Natural and Socio-Economic Systems under Climate Change

3.4.1. Impacts on Natural and Managed Ecosystems

3.4.1.1. Impacts on Ecosystems and their Services in Drylands

The Millenium Ecosystem Assessement (2005) proposed four classes of ecosystem services: provisioning, regulating, supporting and cultural services (Cross-Chapter Box 8: Ecosystem Services, Chapter 6). These ecosystem services in drylands are vulnerable to the impacts of climate change due to high variability in temperature, precipitation and soil fertility (Enfors and Gordon, 2008; Mortimore, 2005). There is *high confidence* that desertification processes such as soil erosion, secondary salinisation, and overgrazing have negatively impacted provisioning ecosystem services in drylands, particularly food and fodder production (Majeed and Muhammad, 2019; Mirzabaev et al., 2016; Qadir et al., 2009; Van Loo et al., 2017; Tokbergenova et al., 2018) (3.4.2.2). Zika and Erb (2009) reported an estimation of NPP losses between 0.8 and 2.0 Gt C yr^{-1} due to desertification, comparing the potential NPP and the NPP calculated for the year 2000. In terms of climatic factors, although climatic changes between 1976 and 2016 were found overall favourable for crop yields in Russia (Ivanov et al., 2018), yield decreases of up to 40-60% in dryland areas were caused by severe and extensive droughts (Ivanov et al., 2018). Increase in temperature can have a direct impact on animals in the form of increased physiological stress (Rojas-Downing et al., 2017), increased water requirements for drinking and cooling, a decrease in the production of milk, meat and eggs, increased stress during conception and reproduction (Nardone et al., 2010) or an increase in seasonal diseases and epidemics (Thornton et al., 2009; Nardone et al., 2010). Furthermore, changes in temperature can indirectly impact livestock through reducing the productivity and quality of feed crops and forages (Thornton et al., 2009; Polley et al., 2013). On the other hand, fewer days with extreme cold temperatures during winter in the temperate zones are associated with lower livestock mortality. The future projection of impacts on ecosystems is presented in section 3.5.2.

Over-extraction is leading to groundwater depletion in many dryland areas (*high confidence*) (Mudd, 2000; Mays, 2013; Mahmod and Watanabe, 2014; Jolly et al., 2008). Globally, groundwater reserves have been reduced since 1900, with the highest rate of estimated reductions of 145 km^3 yr^{-1} between 2000 and 2008 (Konikow, 2011). Some arid lands are very vulnerable to groundwater reductions, because the current natural recharge rates are lower than during the previous wetter periods (e.g., Atacama Desert and Nubian aquifer system in Africa; (Squeo et al., 2006; Mahmod and Watanabe, 2014; Herrera et al., 2018).

Among regulating services, desertification can influence levels of atmospheric CO_2. In drylands, the majority of C is stored below ground in the form of biomass and SOC (FAO, 1995) (3.3.3). Land-use changes often lead to reductions in SOC and organic matter inputs into soil (Albaladejo et al., 2013; Almagro et al., 2010; Hoffmann et al., 2012; Lavee et al., 1998; Rey et al., 2011), increasing soil salinity and soil erosion (Lavee et al., 1998; Martinez-Mena et al., 2008). In addition to the loss of soil, erosion reduces soil nutrients and organic matter, thereby impacting land's productive capacity. To illustrate, soil erosion by water is estimated to result in the loss of 23–42 M t of N and 14.6–26.4 Mt of phosphorus from soils annually in the world (Pierzynski et al., 2017).

Precipitation, by affecting soil moisture content, is considered to be the principal determinant of the capacity of drylands to sequester C (Fay et al., 2008; Hao et al., 2008; Mi et al., 2015; Serrano-Ortiz et al., 2015; Vargas et al., 2012; Sharkhuu et al., 2016). Lower annual rainfall resulted in the release of C into the atmosphere for a number of sites located in Mongolia, China and North America (Biederman et al., 2017; Chen et al., 2009; Fay et al., 2008; Hao et al., 2008; Mi et al., 2015; Sharkhuu et al., 2016). Low soil water availability promotes soil microbial respiration, yet there is insufficient moisture to stimulate plant productivity (Austin et al., 2004), resulting in net C emissions at an ecosystem level. Under even drier conditions, photodegradation of vegetation biomass may often constitute an additional loss of C from ecosystem (Rutledge et al., 2010). In contrast, years of good rainfall in drylands resulted in the sequestration of C (Biederman et al., 2017; Chen et al., 2009; Hao et al., 2008) In an exceptionally rainy year (2011) in the southern hemisphere, the semiarid ecosystems of this region contributed 51% of the global net C sink (Poulter et al., 2014). These results suggest that arid ecosystems could be an important global C sink depending on soil water availability (*medium evidence, high agreement*). However, drylands are generally predicted to become warmer with an increasing frequency of extreme drought and high rainfall events (Donat et al., 2016).

When desertification reduces vegetation cover, this alters the soil surface, affecting the albedo and the water balance (Gonzalez-Martin et al., 2014) (3.3). In such situations, erosive winds have no more obstacles, which favour the occurrence of wind erosion and dust storms. Mineral aerosols have an important influence on the dispersal of soil nutrients and lead to changes in soil characteristics (Goudie and Middleton, 2001; Middleton, 2017). Thereby, the soil formation as a supporting ecosystem service is negatively affected (3.3.1.). Soil erosion by wind results in a loss of fine soil particles (silt and clay), reducing the ability of soil to sequester C (Wiesmeier et al., 2015). Moreover, dust storms reduce crop yields by loss of plant tissue caused by sandblasting (resulting loss of plant leaves and hence reduced photosynthetic activity (Field et al., 2010), exposing crop roots, crop seed burial under sand deposits, and leading to losses of nutrients and fertiliser from top soil (Stefanski and

Sivakumar, 2009). Dust storms also impact crop yields by reducing the quantity of water available for irrigation because it could decrease the storage capacity of reservoirs by siltation and block conveyance canals (Middleton, 2017; Middleton and Kang, 2017; Stefanski and Sivakumar, 2009). Livestock productivity is reduced by injuries caused by dust storms (Stefanski and Sivakumar, 2009). Additionally, dust storms favor the dispersion of microbial and plants species, which can make local endemic species vulnerable to extinction and promote the invasion of plant and microbial species (Asem and Roy, 2010; Womack et al., 2010). Dust storms increase microbial species in remote sites (*high confidence)*; (Kellogg et al., 2004; Prospero et al., 2005; Griffin et al., 2006; Schlesinger et al., 2006; Griffin, 2007; De Deckker et al., 2008; Jeon et al., 2011; Abed et al., 2012; Favet et al., 2013; Woo et al., 2013; Pointing and Belnap, 2014).

3.4.1.2. Impacts on Biodiversity: Plant and Wildlife

3.4.1.2.1. Plant Biodiversity

Over 20% of global plant biodiversity centres are located within drylands (White and Nackoney, 2003). Plant species located within these areas are characterised by high genetic diversity within populations (Martínez-Palacios et al., 1999). The plant species within these ecosystems are often highly threatened by climate change and desertification (Millennium Ecosystem Assessment, 2005b; Maestre et al., 2012). Increasing aridity exacerbates the risk of extinction of some plant species, especially those that are already threatened due to small populations or restricted habitats (Gitay et al., 2002). Desertification, including through land use change, already contributed to the loss of biodiversity across drylands (*medium confidence)* (Newbold et al., 2015; Wilting et al., 2017). For example, species richness decreased from 234 species in 1978 to 95 in 2011 following long periods of drought and human driven degradation on the steppe land of south western Algeria (Observatoire du Sahara et du Sahel, 2013). Similarly, drought and overgrazing led to loss of biodiversity in Pakistan, where only drought-adapted species have by now survived on arid rangelands (Akhter and Arshad, 2006). Similar trends were observed in desert steppes of Mongolia (Khishigbayar et al., 2015). In contrast, the increase in annual moistening of southern European Russia from the late 1980s to the beginning of the 21st century caused the restoration of steppe vegetation, even under conditions of strong anthropogenic pressure (Ivanov et al., 2018). The seed banks of annual species can often survive over the long-term, germinating in wet years, suggesting that these species could be resilient to some aspects of climate change (Vetter et al., 2005). Yet, Hiernaux and Houérou (2006) showed that overgrazing in the Sahel tended to decrease the seed bank of annuals which could make them vulnerable to climate change over time. Perennial species, considered as the structuring element of the ecosystem, are usually less affected as they have deeper roots, xeromorphic properties and physiological mechanisms that increase drought tolerance (Le Houérou, 1996). However, in North Africa, long-term monitoring (1978–2014) has shown that important plant perennial species have also disappeared due to drought (*Stipa tenacissima* and *Artemisia herba alba*) (Hirche et al., 2018; Observatoire du Sahara et du Sahel, 2013). The aridisation of the climate in the south of Eastern Siberia led to the advance of the steppes to the north and to the corresponding migration of steppe mammal species between 1976 and 2016 (Ivanov et al., 2018). The future projection of impacts on plant biodiversity is presented in the section 3.5.2.

3.4.1.2.2. Wildlife biodiversity

Dryland ecosystems have high levels of faunal diversity and endemism (MEA, 2005; Whitford, 2002). Over 30% of the endemic bird areas are located within these regions, which is also home to 25% of vertebrate species (Maestre et al., 2012; MEA, 2005). Yet, many species within drylands are threatened with extinction (Durant et al., 2014; Walther, 2016). Habitat degradation and desertification are generally associated with biodiversity loss (Ceballos et al. 2010; Tang et al. 2018; Newbold et al. 2015). The "grazing value" of land declines with both a reduction in vegetation cover and shrub encroachment, with the former being more detrimental to native vertebrates (Parsons et al., 2017). Conversely, shrub encroachment may buffer desertification by increasing resource and microclimate availability, resulting in an increase in vertebrate species abundance and richness observed in the shrub encroached arid grasslands of North America (Whitford, 1997) and Australia (Parsons et al., 2017). However, compared to historically resilient drylands, these encroached habitats and their new species assemblages may be more sensitive to droughts, which may more prevalent with climate change (Schooley et al., 2018). Mammals and birds may be particularly sensitive to droughts because they rely on evaporative cooling to maintain their body temperatures within an optimal range (Hetem et al., 2016) and risk lethal dehydration in water limited environments (Albright et al., 2017). The direct effects of reduced rainfall and water availability are *likely* to be exacerbated by the indirect effects of desertification through a reduction in primary productivity. A reduction in the quality and quantity of resources available to herbivores due to desertification under changing climate can have knock-on consequences for predators and may ultimately disrupt trophic cascades (*limited evidence, low agreement)* (Rey et al. 2017; Walther 2010). Reduced resource availability may also compromise immune response to novel pathogens, with increased pathogen dispersal associated with dust storms (Zinabu et al., 2018). Responses to desertification are species-specific and mechanistic models are not yet able to accurately predict individual species responses to the many factors associated with desertification (Fuller et al., 2016).

3.4.2. Impacts on Socio-economic Systems

Combined impacts of desertification and climate change on socio-economic development in drylands are complex. Figure 3.9 schematically represents our qualitative assessment of the magnitudes and the uncertainties associated with these impacts on attainment of the SDGs in dryland areas (UN, 2015). The impacts of desertification and climate change are difficult to isolate from the effects of other socio-economic, institutional and political factors (Pradhan et al., 2017). However, there is *high confidence* that climate change will exacerbate the vulnerability of dryland populations to desertification, and that the combination of pressures coming from climate change and desertification will diminish opportunities for reducing poverty, enhancing food and nutritional security, empowering women, reducing disease burden, improving access to water and sanitation. Desertification is embedded in SDG 15 (target 15.3) and climate change is under SDG 13, the *high confidence* and high magnitude impacts depicted for these SDGs (Figure 3.9) indicate that the interactions between desertification and climate change strongly affect the achievement of the targets of SDGs 13 and 15.3, pointing at the need for the coordination of policy actions on land degradation neutrality and mitigation and adaptation to climate change. The following subsections present the literature and the assessment which serve as the basis for Figure 3.9.

3.4.2.1 Impacts on Poverty

Climate change has a high potential to contribute to poverty particularly through the risks coming from extreme weather events (Olsson et al., 2014). However, the evidence rigourously attributing changes in observed poverty to climate change impacts is currently not available. On the other hand, most of the research on links between poverty and desertification (or more broadly, land degradation) focused on whether or not poverty is a cause of land degradation (Gerber et al., 2014; Vu et al., 2014; Way, 2016; 4.7.1). The literature measuring to what extent desertification contributed to poverty globally is lacking: the related literature remains qualitative or correlational (Barbier and Hochard, 2016). At the local level, on the other hand, there is *limited evidence* and *high agreement* that desertification increased multidimensional poverty. For example, Diao and Sarpong (2011) estimated that land degradation lowered agricultural incomes in Ghana by USD 4.2 billion between 2006 and 2015, increasing the national poverty rate by 5.4% in 2015. Land degradation increased the probability of households becoming poor by 35% in Malawi and 48% in Tanzania (Kirui, 2016). Desertification in China was found to have resulted in substantial losses in income, food production and jobs (Jiang et al., 2014). On the other hand, Ge et al. (2015) indicated that desertification was positively associated with growing incomes in Inner Mongolia in China in the short run since no costs were incurred for SLM, while in the long run higher incomes allowed allocation of more investments to reduce desertification. This relationship corresponds to the Environmental Kuznets Curve, which posits that environmental degradation initially rises and subsequently falls with rising income (e.g. Stern, 2017). There is *limited evidence* on the validity of this hypothesis regarding desertification.

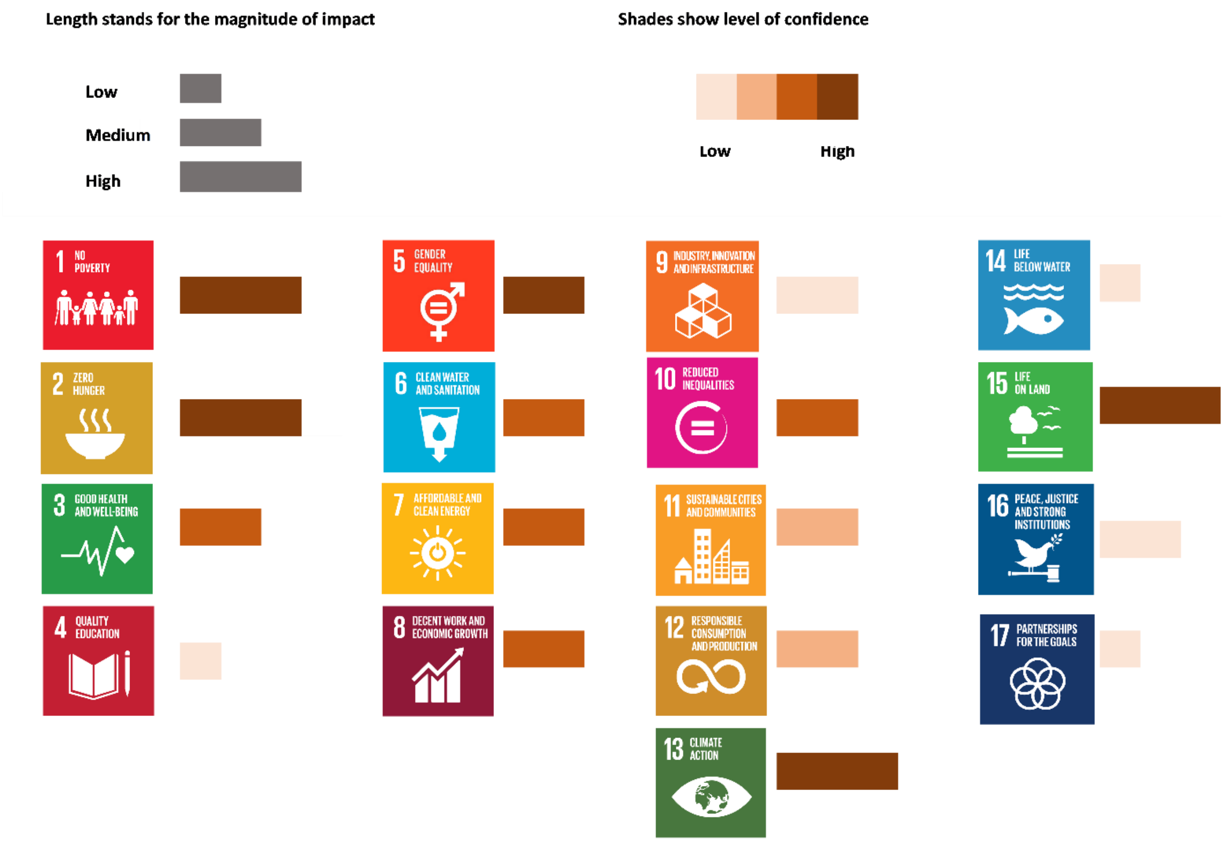


Figure 3.9 Socio-economic impacts of desertification and climate change with the SDG framework

3.4.2.2 Impacts on Food and Nutritional Insecurity

About 821 million people globally were food insecure in 2017, of whom 63% in Asia, 31% in Africa and 5% in Latin America and the Caribbean (FAO et al., 2018). The global number of food insecure people rose by 37 million since 2014. Changing climate variability, combined with a lack of climate resilience, was suggested as a key driver of this increase (FAO et al., 2018). Sub-Saharan Africa, East Africa and South Asia had the highest share of undernourished populations in the world in 2017, with 28.8%, 31.4% and 33.7%, respectively (FAO et al., 2018). The major mechanism through which climate change and desertification affect food security is through their impacts on agricultural productivity. There is *robust evidence* pointing to negative impacts of climate change on crop yields in dryland areas (*high agreement*) (Hochman et al., 2017; Nelson et al., 2010; Zhao et al., 2017; 3.4.1; 5.2.2; 4.7.2). There is also *robust evidence and high agreement* on the losses in agricultural productivity and incomes due to desertification (Kirui, 2016; Moussa et al., 2016; Mythili and Goedecke, 2016; Tun et al., 2015). Nkonya et al. (2016a) estimated that cultivating wheat, maize, and rice with unsustainable land management practices is currently resulting in global losses of USD 56.6 billion annually, with another USD 8.7 billion of annual losses due to lower livestock productivity caused by rangeland degradation. However, the extent to which these losses affected food insecurity in dryland areas is not known. Lower crop yields and higher agricultural prices worsen existing food insecurity, especially for net-food buying rural households and urban dwellers. Climate change and desertification are not the sole drivers of food insecurity, but especially in the areas with high dependence on agriculture, they are among the main contributors.

3.4.2.3 Impacts on Human Health through Dust Storms

The frequency and intensity of dust storms are increasing due to land use and land cover changes and climate-related factors (2.4) particularly in some regions of the world such as the Arabian Peninsula (Jish Prakash et al., 2015; Yu et al., 2015; Gherboudj et al., 2017; Notaro et al., 2013; Yu et al. 2013; Alobaidi et al., 2017; Maghrabi et al., 2011; Almazroui et al. 2018) and broader Middle East (Rashki et al., 2012; Türkeş, 2017; Namdari et al., 2018) as well as Central Asia (Indoitu et al., 2015; Xi and Sokolik, 2015), with growing negative impacts on human health (Díaz et al., 2017; Goudarzi et al., 2017; Goudie, 2014; Samoli et al., 2011) (*high confidence*). Dust storms transport particulate matter, pollutants, pathogens and potential allergens that are dangerous for human health over long distances (Goudie and Middleton, 2006; Sprigg, 2016). Particulate matter (PM), i.e. the suspended particles in the air having sizes of 10 micrometre (PM10) or less, have damaging effects on human health (Díaz et al., 2017; Goudarzi et al., 2017; Goudie, 2014; Samoli et al., 2011). The health effects of dust storms are largest in areas in the immediate vicinity of their origin, primarily the Sahara Desert, followed by Central and Eastern Asia, the Middle East and Australia (Zhang et al., 2016), however, there is *robust evidence* showing that the negative health effects of dust storms reach a much wider area (Bennett et al., 2006; Díaz et al., 2017; Kashima et al., 2016; Lee et al., 2014; Samoli et al., 2011; Zhang et al., 2016). The primary health effects of dust storms include damage to the respiratory and cardiovascular systems (Goudie, 2013). Dust particles with a diameter smaller than 2.5μm were associated with global cardiopulmonary mortality of about 402,000 people in 2005, with 3.47 million years of life lost in that single year (Giannadaki et al., 2014). If globally only 1.8% of cardiopulmonary deaths were caused by dust storms, in the countries of the Sahara region, Middle East, South and East Asia, dust storms were suggested to be the reason for 15–50% of all cardiopulmonary deaths (Giannadaki et al., 2014). A $10\mu gm^{-3}$ increase in PM10 dust particles was associated with mean increases in non-accidental mortality from 0.33% to 0.51% across different calendar seasons in China, Japan and South Korea (Kim et al., 2017). The percentage of all-cause deaths attributed to fine particulate matter in Iranian cities affected by Middle Eastern dust storms (MED) were 0.56–5.02%, while the same percentage for non-affected cities were 0.16–4.13% (Hopke et al., 2018). The Meningococcal Meningitis epidemics occur in the Sahelian region during the dry seasons with dusty conditions (Agier et al., 2012; Molesworth et al., 2003). Despite a strong concentration of dust storms in the Sahel, North Africa, the Middle East and Central Asia, there is relatively little research on human health impacts of dust storms in these regions. More research on health impacts and related costs of dust storms as well as on public health response measures can help in mitigating these health impacts.

3.4.2.4. Impacts on Gender Equality

Environmental issues such as desertification and impacts of climate change have been increasingly investigated through a gender lens (Bose; Broeckhoven and Cliquet, 2015; Kaijser and Kronsell, 2014; Kiptot et al., 2014; Villamor and van Noordwijk, 2016). There is *medium evidence* and *high agreement* that women will be impacted more than men by environmental degradation (Arora-Jonsson, 2011; Gurung et al., 2006; Cross-Chapter Box 11: Gender, Chapter 7). Socially structured gender-specific roles and responsibilities, daily activities, access and control over resources, decision-making and opportunities lead men and women to interact differently with natural resources and landscapes. For example, water scarcity affected women more than men in rural Ghana as they had to spend more time in fetching water, which has implications on time allocations for other activities (Ahmed et al., 2016). Despite the evidence pointing to differentiated impact of environmental degradation on women and men, gender issues have been marginally addressed in many land restoration and rehabilitation efforts, which often remain gender-blind. Although there is *robust* *evidence* on the location-specific impacts of climate change and desertification on gender equality, however, there is *limited evidence* on the gender-related impacts of land restoration and rehabilitation activities. Women are usually excluded from local decision making on actions regarding desertification and climate change. Socially constructed gender-specific roles and responsibilities are not static because they are shaped by other factors such as wealth, age, ethnicity, and formal education (Kaijser and Kronsell, 2014; Villamor et al., 2014). Hence, women's and men's environmental knowledge and priorities for restoration often differ (Sijapati Basnett et al., 2017). In some areas where sustainable land options (e.g. agroforestry) are being promoted, women were not able to participate due to culturally-embedded asymmetries in power relations between men and women (Catacutan and Villamor, 2016). Nonetheless, women particularly in the rural areas remain heavily involved in securing food for their households. Food security for them is associated with land productivity and women's contribution to address desertification is crucial.

3.4.2.5. Impacts on Water Scarcity and Use

Reduced water retention capacity of degraded soils amplifies floods (de la Paix et al., 2011), reinforces degradation processes through soil erosion, and reduces annual intake of water to aquifers, exacerbating existing water scarcities (Le Roux et al., 2017; Cano et al., 2018). Reduced vegetation cover and more intense dust storms were found to intensify droughts (Cook et al., 2009). Moreover, secondary salinisation in the irrigated drylands often requires leaching with considerable amounts of water (Greene et al., 2016; Wichelns and Qadir, 2015). Thus, different types of soil degradation increase water scarcity both through lower water quantity and quality (Liu et al., 2017; Liu et al., 2016c). All these processes reduce water availability for other needs. In this context, climate change will further intensify water scarcity in some dryland areas and increase the frequency of droughts (*medium confidence*) (2.2; IPCC, 2013; Zheng et al., 2018). Higher water scarcity may imply growing use of wastewater effluents for irrigation (Pedrero et al., 2010). The use of untreated wastewater exacerbates soil degradation processes (Tal, 2016; Singh et al., 2004; Qishlaqi et al., 2008; Hanjra et al., 2012), in addition to negative human health impacts (Faour-Klingbeil and Todd, 2018; Hanjra et al., 2012). Climate change, thus, will amplify the need for integrated land and water management for sustainable development.

3.4.2.6 Impacts on Energy Infrastructure through Dust Storms

Desertification leads to conditions that favour the production of dust storms (*high confidence*) (3.3.1). There is *robust evidence and high agreement* that dust storms negatively affect the operational potential of solar and wind power harvesting equipment through dust deposition, reduced reach of solar radiation and increasing blade surface roughness, and can also reduce effective electricity distribution in high-voltage transmission lines (Zidane et al., 2016; Costa et al., 2016; Lopez-Garcia et al., 2016; Maliszewski et al., 2012; Mani and Pillai, 2010; Mejia and Kleissl, 2013; Mejia et al., 2014; Middleton, 2017; Sarver et al., 2013; Kaufman et al., 2002; Kok et al., 2018). Direct exposure to desert dust storm can reduce energy generation efficiency of solar panels by 70–80% in one hour (Ghazi et al., 2014). Saidan et al.(2016) indicated that in the conditions of Baghdad, Iraq, one month exposure to weather reduced the efficiency of solar modules by 18.74% due to dust deposition. In Atacama desert, Chile, one month exposure reduced thin-film solar module performance by 3.7-4.8% (Fuentealba et al., 2015). This has important implications for climate change mitigation efforts using the expansion of solar and wind energy generation in dryland areas for substituting fossil fuels. Abundant access to solar energy in many dryland areas makes them high potential locations for the installation of solar energy generating infrastructure. Increasing desertification, resulting in higher frequency and intensity of dust storms imposes additional costs for climate change mitigation through deployment of solar and wind energy harvesting facilities in dryland areas. Most frequently used solutions to this problem involve physically wiping or washing the surface of solar devices with water. These result in additional costs and excessive use of already scarce water resources and labour

(Middleton, 2017). The use of special coatings on the surface of solar panels can help prevent the deposition of dusts (Costa et al., 2016; Costa et al., 2018; Gholami et al., 2017).

3.4.2.7 Impacts on Transport Infrastructure through Dust Storms and Sand Movement

Dust storms and movement of sand dunes often threaten the safety and operation of railway and road infrastructure in arid and hyper-arid areas, and can lead to road and airport closures due to reductions in visibility. For example, the dust storm on 10th March 2009 over Riyadh was assessed to be the strongest in the previous two decades in Saudi Arabia, causing limited visibility, airport shutdown and damages to infrastructure and environment across the city (Maghrabi et al., 2011). There are numerous historical examples of how moving sand dunes led to the forced decommissioning of early railway lines built in Sudan, Algeria, Namibia and Saudi Arabia in the late 19th and early 20th century (Bruno et al., 2018). Currently, the highest concentration of railways vulnerable to sand movements are located in north-western China, Middle East and North Africa (Bruno et al., 2018; Cheng and Xue, 2014). In China, sand dune movements are periodically disrupting the railway transport in Linhai-Ceke line in north-western China and Lanzhou-Xinjiang High-speed Railway in western China, with considerable clean-up and maintenance costs (Bruno et al., 2018; Zhang et al., 2010). There are large-scale plans for expansion of railway networks in arid areas of China, Central Asia, North Africa, the Middle East, and Eastern Africa. For example, “The Belt and Road Initiative” promoted by China, the Gulf Railway project by the countries of the Arab Gulf Cooperation Council (GCC), or Lamu Port-South Sudan- Ethiopia Transport Corridor in Eastern Africa. These investments have long-term return and operation periods. Their construction and associated engineering solutions will therefore benefit from careful consideration of potential desertification and climate change effects on sand storms and dune movements.

3.4.2.8 Impacts on Conflicts

There is *low confidence* in climate change and desertification leading to violent conflicts. There is *medium evidence* and *low agreement* that climate change and desertification contribute to already existing conflict potentials (Herrero, 2006; von Uexkull et al., 2016; Theisen, 2017; Olsson, 2017; Wischnath and Buhaug, 2014; 4.7.3). To illustrate, Hsiang et al. (2013) found that each one standard deviation increase in temperature or rainfall was found to increase interpersonal violence by 4% and intergroup conflict by 14% (Hsiang et al., 2013). However, this conclusion was disputed by Buhaug et al., (2014), who found no evidence linking climate variability to violent conflict after replicating Hsiang et al. (2013) by studying only violent conflicts. Almer et al. (2017) found that a one-standard deviation increase in dryness raised the likelihood of riots in Sub-Saharan African countries by 8.3% during the 1990–2011 period. On the other hand, Owain and Maslin (2018) found that droughts and heatwaves were not significantly affecting the level of regional conflict in East Africa. Similarly, it was suggested that droughts and desertification in the Sahel have played a relatively minor role in the conflicts in the Sahel in the 1980s, with the major reasons for the conflicts during this period being political, especially the marginalisation of pastoralists (Benjaminsen, 2016), corruption and rent-seeking (Benjaminsen et al., 2012). Moreover, the role of environmental factors as the key drivers of conflicts were questioned in the case of Sudan (Verhoeven, 2011) and Syria (De Châtel, 2014). Selection bias, when the literature focuses on the same few regions where conflicts occurred and relates them to climate change, is a major shortcoming, as it ignores other cases where conflicts did not occur (Adams et al., 2018) despite degradation of the natural resource base and extreme weather events.

3.4.2.9 Impacts on Migration

Environmentally-induced migration is complex and accounts for multiple drivers of mobility as well as other adaptation measures undertaken by populations exposed to environmental risk (*high confidence*). There is *medium evidence* and *low agreement* that climate change impacts migration. The World Bank (2018) predicted that 143 million people would be forced to move internally by 2050 if no climate action is taken. Focusing on asylum seekers alone, rather than the total number of migrants, Missirian and Schlenker (2017) predict the asylum applications to the European Union will increase from 28% (98,000 additional asylum applications per year) up to 188% (660,000 additional applications per year) depending on the climate scenario by 2100. While the modelling efforts have greatly improved over the years (Hunter et al., 2015; McLeman, 2011; Sherbinin and Bai, 2018) and in particular, these recent estimates provide an important insight into potential future developments, the quantitative projections are still based on the number of people exposed to risk rather than the number of people who would actually engage in migration as a response to this risk (Gemenne, 2011; McLeman, 2013) and they do not take into account individual agency in migration decision nor adaptive capacities of individuals (Hartmann, 2010; Kniveton et al., 2011; Piguet, 2010) (see Section 3.6.2 discussing migration as a response to desertification). Accordingly, the available micro-level evidence suggests that climate-related shocks are one of the many drivers of migration (Adger et al., 2014; London Government Office for Science and Foresight, 2011; Melde et al., 2017), but the individual responses to climate risk are more complex than commonly assumed (Gray and Mueller, 2012a). For example, despite strong focus on natural disasters, neither flooding (Gray and Mueller, 2012b; Mueller et al., 2014) nor earthquakes (Halliday, 2006) were found to induce long-term migration; but instead, slow-onset changes, especially those provoking crop failures and heat stress, could affect household or individual migration decisions (Gray and Mueller, 2012a; Missirian and Schlenker, 2017; Mueller et al., 2014). Out-migration from drought-prone areas has received particular attention (de Sherbinin et al., 2012; Ezra and Kiros, 2001) and indeed, a substantial body of literature suggests that households engage in local or internal migration as a response to drought (Findlay, 2011; Gray and Mueller, 2012a), while international migration decreases with drought in some contexts (Henry et al., 2004), but might increase in contexts where migration networks are well established (Feng et al., 2010; Nawrotzki and DeWaard, 2016; Nawrotzki et al., 2015, 2016). Similarly, the evidence is not conclusive with respect to the effect of environmental drivers, in particular desertification, on mobility. While it has not consistently entailed out-migration in the case of Ecuadorian Andes (Gray, 2009, 2010) environmental and land degradation increased mobility in Kenya and Nepal (Gray, 2011; Massey et al., 2010), but marginally decreased mobility in Uganda (Gray, 2011). These results suggest that in some contexts, environmental shocks actually undermine household's financial capacity to undertake migration (Nawrotzki and Bakhtsiyarava, 2017), especially in the case of the poorest households (Barbier and Hochard, 2018; Koubi et al., 2016; Kubik and Maurel, 2016; McKenzie and Yang, 2015). Adding to the complexity, migration, especially to frontier areas, by increasing pressure on land and natural resources, might itself contribute to environmental degradation at the destination (Hugo, 2008; IPBES, 2018a; McLeman, 2017). The consequences of migration can also be salient in the case of migration to urban or peri-urban areas; indeed, environmentally-induced migration can add to urbanisation (3.6.2.2), often exacerbating problems related to poor infrastructure and unemployment.

3.4.2.10 Impacts on Pastoral Communities

Pastoral production systems occupy a significant portion of the world (Rass, 2006; Dong, 2016). Food insecurity among pastoral households is often high (3.1.3; Gomes, 2006). The Sahelian droughts of the 1970-80s provided an example of how droughts could affect livestock resources and crop productivity, contributing to hunger, out-migration and suffering for millions of pastoralists (Hein and De Ridder, 2006; Molua and Lambi, 2007). During these Sahelian droughts low and erratic rainfall exacerbated desertification processes, leading to ecological changes that forced people to use marginal lands and ecosystems. Similarly, the rate of rangeland degradation is now increasing because of environmental changes and overexploitation of resources (Kassahun et al., 2008; Vetter, 2005). Desertification coupled with climate change is negatively affecting livestock feed and grazing species (Hopkins and Del Prado, 2007), changing the composition in favour of species with low forage quality, ultimately reducing livestock productivity (D'Odorico et al., 2013; Dibari et al., 2016) and increasing livestock disease prevalence (Thornton et al., 2009). There is *robust evidence and high agreement* that weak adaptive capacity, coupled with negative effects from other climate-related factors, are predisposing pastoralists to increased poverty from desertification and climate change globally (López-i-Gelats et al., 2016; Giannini et al., 2008; IPCC, 2007). On the other hand, misguided policies such as enforced sedentarisation and in certain cases protected area delineation (fencing), which restrict livestock mobility have hampered optimal use of grazing land resources (Du, 2012); and led to degradation of resources and out-migration of people in search of better livelihoods (Gebeye, 2016; Liao et al., 2015). Restrictions on the mobile lifestyle is reducing the resilient adaptive capacity of pastoralists to natural hazards including extreme and variable weather conditions, drought and climate change (Schilling et al., 2014). Furthermore, the exacerbation of the desertification phenomenon due to agricultural intensification (D'Odorico et al., 2013) and land fragmentation caused by encroachment of agriculture into rangelands (Otuoma et al., 2009; Behnke and Kerven, 2013) is threatening pastoral livelihoods. For example, commercial cotton (*Gossypium hirsutum*) production is crowding out pastoral systems in Benin (Tamou et al., 2018). Food shortages and the urgency to produce enough crop for public consumption are leading to the encroachment of agriculture into productive rangelands and those converted rangelands are frequently prime lands used by pastoralists to produce feed and graze their livestock during dry years (Dodd, 1994). The sustainability of pastoral systems is therefore coming into question because of social and political marginalisation of the system (Davies et al., 2016) and also because of the fierce competition it is facing from other livelihood sources such as crop farming (Haan et al., 2016).

3.5. Future Projections

3.5.1. Future Projections of Desertification

Assessing the impact of climate change on future desertification is difficult as several environmental and anthropogenic variables interact to determine its dynamics. The majority of modelling studies regarding the future evolution of desertification rely on the analysis of specific climate change scenarios and Global Climate Models and their effect on a few processes or drivers that trigger desertification (Cross-Chapter Box 1: Scenarios, Chapter 1).

With regards to climate impacts, the analysis of global and regional climate models concludes that under all representative concentration pathways (RCPs) potential evapotranspiration (PET) would increase worldwide as a consequence of increasing surface temperatures and surface water vapour deficit (Sherwood and Fu, 2014). Consequently, there would be associated changes in aridity indices that depend on this variable (*high agreement*, *robust evidence*) (Cook et al., 2014a; Dai, 2011; Dominguez et al., 2010; Feng and Fu, 2013; Ficklin et al., 2016; Fu et al., 2016; Greve and Seneviratne, 1999; Koutroulis, 2019; Scheff and Frierson, 2015). Due to the large increase in PET and decrease in precipitation over some subtropical land areas, aridity index will decrease in some drylands (Zhao and Dai, 2015), with one model estimating ~10% increase in hyper-arid areas globally (Zeng and Yoon, 2009). Increases in PET are projected to continue due to climate change (Cook et al., 2014a; Fu et al., 2016; Lin et al., 2015; Scheff and Frierson, 2015). However, as noted in sections 3.1.1 and 3.2.1, these PET calculations use assumptions that are not valid in an environment with changing CO_2. Evidence from precipitation, runoff or photosynthetic uptake of CO_2 suggest that a future warmer world will be less arid (Roderick et al., 2015). Observations in recent decades indicate that the Hadley cell has expanded poleward in both hemispheres (Fu et al., 2006; Hu and Fu, 2007; Johanson et al., 2009; Seidel and Randel, 2007), and under all RCPs would continue expanding (Johanson et al., 2009; Lu et al., 2007). This expansion leads to the poleward extension of sub-tropical dry zones and hence an expansion in drylands on the poleward edge (Scheff and Frierson, 2012).

Overall, this suggests that while aridity will increase in some places (*high confidence*), there is insufficient evidence to suggest a global change in dryland aridity (*medium confidence*).

Regional modelling studies confirm the outcomes of Global Climate Models (Africa: Terink et al., 2013; China: Yin et al., 2015; Brazil: Marengo and Bernasconi, 2015; Cook et al., 2012; Greece: Nastos et al., 2013; Italy: Coppola and Giorgi, 2009). According to the IPCC AR5 (IPCC, 2013), decreases in soil moisture are detected in the Mediterranean, Southwest USA and southern African regions. This is in line with alterations in the Hadley circulation and higher surface temperatures. This surface drying will continue to the end of this century under the RCP8.5 scenario (*high confidence*). Ramarao et al., (2015) showed that a future climate projection based on RCP4.5 scenario indicated the possibility for detecting the summer-time soil drying signal over the Indian region during the 21st century in response to climate change. The IPCC Special Report on Global Warming of 1.5°C (SR15, Chapter 3) (Hoegh-Guldberg et al., 2018) report concluded with "*medium confidence*" that global warming by more than 1.5°C increases considerably the risk of aridity for the Mediterranean area and Southern Africa. Miao et al., (2015b) showed an acceleration of desertification trends under the RCP8.5 scenario in the middle and northern part of Central Asia and some parts of north western China. It is also useful to consider the effects of the dynamic–thermodynamical feedback of the climate. Schewe and Levermann (2017) show increases up to 300 % in the central Sahel rainfall by the end of the century due to an expansion of the West African monsoon. Warming could trigger an intensification of monsoonal precipitation due to increases in ocean moisture availability.

The impacts of climate change on dust storm activity are not yet comprehensively studied and represent an important knowledge gap. Currently, GCMs are unable to capture recent observed dust emission and transport (Evan, 2018; Evan et al., 2014) limiting confidence in future projections. Literature suggests that climate change decreases wind erosion/dust emission overall with regional variation (*low confidence*). Mahowald et al. (2006) and Mahowald (2007) found that climate change led to a decrease in desert dust source areas globally using CMIP3 GCMs. Wang et al. (2009) found a decrease in sand dune movement by 2039 (increasing thereafter) when assessing future wind erosion driven desertification in arid and semiarid China using a range of SRES scenarios and HadCM3 simulations. Dust activity in the US Southern Great Plains was projected to increase, while in the Northern Great Plains it was projected to decrease under RCP 8.5 climate change scenario (Pu and Ginoux, 2017). Evan et al. (2016) project a decrease in African dust emission associated with a slowdown of the tropical circulation in the high CO_2 RCP8.5 scenario.

Global estimates of the impact of climate change on soil salinisation show that under the IS92a emissions scenario (a scenario prepared in 1992 that contains "business as usual" assumptions) (Leggett et al., 1992) the area at risk of salinisation would increase in the future (*limited evidence, high agreement*; (Schofield and Kirkby, 2003). Climate change has an influence on soil salinisation that induces further land degradation through several mechanisms that vary in their level of complexity. However, only a few examples can be found to illustrate this range of impacts, including the effect of groundwater table depletion (Rengasamy, 2006) and irrigation management (Sivakumar, 2007), salt migration in coastal aquifers with decreasing water tables (Sherif and Singh, 1999; 4.10.7), and surface hydrology and vegetation that affect wetlands and favour salinisation (Nielsen and Brock, 2009).

3.5.1.1. Future Vulnerability and Risk to Desertification

Following the conceptual framework developed in the Managing the Risks of Extreme Events and Disasters to Advance Climate Change Adaptation special report (SREX) (IPCC, 2012), future risks are assessed by examining changes in exposure (i.e. presence of people; livelihoods; species or ecosystems; environmental functions, service, and resources; infrastructure; or economic, social or cultural assets; see glossary), changes in vulnerability (i.e. propensity or predisposition to be adversely affected; see glossary) and changes in the nature and magnitude of hazards (i.e. potential occurrence of a natural or human-induced physical event that causes damage; see glossary). Climate change is expected to further exacerbate the vulnerability of dryland ecosystems to desertification by increasing PET globally (Sherwood and Fu, 2014). Temperature increases between 2°C and 4°C are projected in drylands by the end of the 21st century under RCP4.5 and RCP8.5 scenarios, respectively (IPCC, 2013). An assessment by (Carrão et al., 2017) showed an increase in drought hazards by late-century (2071–2099) compared to a baseline (1971–2000) under high RCPs in drylands around the Mediterranean, south-eastern Africa, and southern Australia. In Latin America, Morales et al. (2011) indicated that areas affected by drought will increase significantly by 2100 under SRES scenarios A2 and B2. The countries expected to be affected include Guatemala, El Salvador, Honduras and Nicaragua. In CMIP5 scenarios, Mediterranean types of climate are projected to become drier (Alessandri et al., 2014; Polade et al., 2017), with the equatorward margins being potentially replaced by arid climate types (Alessandri et al., 2014). Globally, climate change is predicted to intensify the occurrence and severity of droughts (*medium confidence*) (2.2; Dai, 2013; Sheffield and Wood, 2008; Swann et al., 2016; Wang, 2005; Zhao and Dai, 2015; Carrão et al., 2017; Naumann et al., 2018). Ukkola et al. (2018) showed large discrepancies between CMIP5 models for all types of droughts, limiting the confidence that can be assigned to projections of drought.

Drylands are characterised by high climatic variability. Climate impacts on desertification are not only defined by projected trends in mean temperature and precipitation values but are also strongly dependent on changes in climate variability and extremes (Reyer et al., 2013). The responses of ecosystems depend on diverse vegetation types. Drier ecosystems are more sensitive to changes in precipitation and temperature (Li et al., 2018; Seddon et al., 2016; You et al., 2018), increasing vulnerability to desertification. It has also been reported that areas with high variability in precipitation tend to have lower livestock densities and that those societies that have a strong dependence on livestock that graze natural forage are especially affected (Sloat et al., 2018). Social vulnerability in drylands increases as a consequence of climate change that threatens the viability of pastoral food systems (Dougill et al., 2010; López-i-Gelats et al., 2016). Social drivers can also play an important role with regards to future vulnerability (Máñez Costa et al., 2011). In the arid region of north-western China, Liu et al. (2016b) estimated that under RCP4.5 areas of increased vulnerability to climate change and desertification will surpass those with decreased vulnerability.

Using an ensemble of global climate, integrated assessment and impact models, Byers et al. (2018) investigated 14 impact indicators at different levels of global mean temperature change and socioeconomic development. The indicators cover water, energy and land sectors. Of particular relevance to desertification are the water (e.g. water stress, drought intensity) and the land (e.g. habitat degradation) indicators. Under shared socioeconomic pathway SSP2 (“Middle of the Road”) at 1.5°C, 2°C and 3°C of global warming, the numbers of dryland populations exposed (vulnerable) to various impacts related to water, energy and land sectors (e.g. water stress, drought intensity, habitat degradation) are projected to reach 951 (178) million, 1,152 (220) million and 1,285 (277) million, respectively. While at global warming of 2°C, under SSP1 (sustainability), the exposed (vulnerable) dryland population is 974 (35) million, and under SSP3 (Fragmented World) it is 1,267 (522) million. Steady increases in the exposed and vulnerable populations are seen for increasing global mean temperatures. However much larger differences are seen in the vulnerable population under different SSPs. Around half the vulnerable population is in South Asia, followed by Central Asia, West Africa and East Asia.

3.5.2. Future Projections of Impacts

Future climate change is expected to increase the potential for increased soil erosion by water in dryland areas (*medium confidence*). Yang et al. (2003) use a Revised Universal Soil Loss Equation (RUSLE) model to study global soil erosion under historical, present and future conditions of both cropland and climate. Soil erosion potential has increased by about 17%, and climate change will increase this further in the future. In northern Iran, under the SRES A2 emission scenario the mean erosion potential is projected to grow by 45% comparing the period 1991-2010 with 2031-2050 (Zare et al., 2016). A strong decrease in precipitation for almost all parts of Turkey was projected for the period of 2021–2050 compared to 1971-2000 using Regional Climate Model, RegCM4.4 of the International Centre for Theoretical Physics (ICTP) under RCP4.5 and RCP8.5 scenarios (Türkeş et al., 2019). The projected changes in precipitation distribution can lead to more extreme precipitation events and prolonged droughts, increasing Turkey's vulnerability to soil erosion (Türkeş et al., 2019). In Portugal, a study comparing wet and dry catchments under A1B and B1 emission scenarios showed an increase in erosion in dry catchments (Serpa et al., 2015). In Morocco an increase in sediment load is projected as a consequence of reduced precipitation (Simonneaux et al., 2015). WGII AR5 concluded the impact of increases in heavy rainfall and temperature on soil erosion will be modulated by soil management practices, rainfall seasonality and land cover (Jiménez Cisneros et al., 2014). Ravi et al. (2010) predicted an increase in hydrologic and aeolian soil erosion processes as a consequence of droughts in drylands. However, there are some studies that indicate that soil erosion will be reduced in Spain (Zabaleta et al., 2013), Greece (Nerantzaki et al., 2015) and Australia (Klik and Eitzinger, 2010), while others project changes in erosion as a consequence of the expansion of croplands (Borrelli et al., 2017).

Potential dryland expansion implies lower C sequestration and higher risk of desertification (Huang et al., 2017), with severe impacts on land usability and threatening food security. At the level of biomes (global-scale zones, generally defined by the type of plant life that they support in response to average rainfall and temperature patterns; see glossary), soil C uptake is determined mostly by weather variability. The area of the land in which dryness controls CO_2 exchange has risen by 6% since 1948 and is projected to expand by at least another 8% by 2050. In these regions net C uptake is about 27% lower than elsewhere (Yi et al., 2014). Potential losses of soil C are projected to range from 9 to 12% of the total C stock in the 0-20 cm layer of soils in the southern European Russia by end of this century (Ivanov et al., 2018).

Desertification under climate change will threaten biodiversity in drylands (*medium confidence*). Rodriguez-Rodriguez-Caballero et al. (2018) analysed the cover of biological soil crusts under current and future environmental conditions utilising an environmental niche modelling approach. Their results suggest that biological soil crusts currently cover ~1600 M ha in drylands. Under RCP scenarios 2.6 to 8.5, 25–40% of this cover will be lost by 2070 with climate and land use contributing equally. The predicted loss is expected to substantially reduce their contribution to N cycling (6.7–9.9 Tg yr^{-1} of N) and C cycling (0.16–0.24 Pg yr^{-1} of C) (Rodriguez-Caballero et al., 2018). A study in Colorado Plateau, USA showed that changes in climate in drylands may damage the biocrust communities by promoting rapid mortality of foundational species (Rutherford et al., 2017), while in southern California deserts climate change-driven extreme heat and drought may surpass the survival thresholds of some desert species (Bachelet et al., 2016). In semiarid Mediterranean shrublands in eastern Spain, plant species richness and plant cover could be reduced by climate change and soil erosion (García-Fayos and Bochet, 2009). The main drivers of species extinctions are land use change, habitat pollution, over-exploitation, and species invasion, while the climate change is indirectly linked to species extinctions (Settele et al., 2014). Malcolm et al. (2006) found that more than 2000 plant species located within dryland biodiversity hotspots could become extinct within 100 years starting 2004 (within the Cape Floristic Region, Mediterranean Basin and Southwest Australia). Furthermore, it is suggested that land use and climate change could cause the loss of 17% of species within shrublands and 8% within hot deserts by 2050 (van Vuuren et al., 2006) (*low confidence*). A study in the semi-arid Chinese Altai Mountains showed that mammal species richness will decline, and rates of species turnover will increase, and more than 50% of their current ranges will be lost (Ye et al., 2018).

Changing climate and land use have resulted in higher aridity and more droughts in some drylands, with the rising role of precipitation, wind and evaporation on desertification (Fischlin et al., 2007). In a 2°C world, annual water discharge is projected to decline, and heatwaves are projected to pose risk to food production by 2070 (Waha et al., 2017). However, Betts et al. (2018) found a mixed response of water availability (runoff) in dryland catchments to global temperature increases from 1.5ºC to 2ºC. The forecasts for Sub-Saharan Africa suggest that higher temperatures, increase in the number of heatwaves, and increasing aridity, will affect the rainfed agricultural systems (Serdeczny et al., 2017). A study by (Wang et al., 2009) in arid and semiarid China showed decreased livestock productivity and grain yields from 2040 to 2099, threatening food security. In Central Asia, projections indicate a decrease in crop yields, and negative impacts of prolonged heat waves on population health (3.7.2; Reyer et al., 2017). World Bank (2009) projected that, without the C fertilisation effect, climate change will reduce the mean yields for 11 major global crops, such as millet, field pea, sugar beet, sweet potato, wheat, rice, maize, soybean, groundnut, sunflower, and rapeseed, by 15% in Sub-Saharan Africa, 11% in Middle East and North Africa, 18% in South Asia, and 6% in Latin America and Caribbean by 2046–2055, compared to 1996–2005. A separate meta-analysis suggested a similar reduction in yields in Africa and South Asia due to climate change by 2050 (Knox et al., 2012). Schlenker and Lobell (2010) estimated that in sub-Saharan Africa, crop production may be reduced by 17–22% due to climate change by 2050. At the local level, climate change impacts on crop yields vary by location (5.2.2). Negative impacts of climate change on agricultural productivity contribute to higher food prices. The imbalance between supply and demand for agricultural products is projected to increase agricultural prices in the range of 31% for rice to 100% for maize by 2050 (Nelson et al., 2010), and cereal prices in the range between a 32% increase and a 16% decrease by 2030 (Hertel et al., 2010). In the southern European Russia, it is projected that the yields of grain crops will decline by 5 to 10% by 2050 due to the higher intensity and coverage of droughts (Ivanov et al., 2018).

Climate change can have strong impacts on poverty in drylands (*medium confidence*) (Hallegatte and Rozenberg, 2017; Hertel and Lobell, 2014). Globally, Hallegatte et al. (2015) project that without rapid and inclusive progress on eradicating multidimensional poverty, climate change can increase the number of the people living in poverty by 35 to 122 million people until 2030. Although these numbers are global and not specific to drylands, the highest impacts in terms of the share of the national populations being affected are projected to be in the drylands areas of the Sahel region, eastern Africa and South Asia (Stephane Hallegatte et al., 2015). The impacts of climate change on poverty vary depending on whether the household is a net agricultural buyer or seller. Modelling results showed that poverty rates would increase by about one-third among the urban households and non-agricultural self-employed in Malawi, Uganda, Zambia, and Bangladesh due to high agricultural prices and low agricultural productivity under climate change (Hertel et al., 2010). On the contrary, modelled poverty rates fell substantially among agricultural households in Chile, Indonesia, Philippines and Thailand, because higher prices compensated for productivity losses (Hertel et al., 2010).

3.6. Responses to Desertification under Climate Change

Achieving sustainable development of dryland livelihoods requires avoiding dryland degradation through SLM and restoring and rehabilitating the degraded drylands due to their potential wealth of ecosystem benefits and importance to human livelihoods and economies (Thomas, 2008). A broad suite of on the ground response measures exist to address desertification (Scholes, 2009), be it in the form of improved fire and grazing management, the control of erosion; integrated crop, soil and water management, among others (Liniger and Critchley, 2007; Scholes, 2009). These actions are part of the broader context of dryland development and long-term SLM within coupled socio-economic systems (Reynolds et al., 2007; Stringer et al., 2017; Webb et al., 2017). Many of these response options correspond to those grouped under land transitions in the IPCC Special Report on Global Warming of 1.5°C (Table 6.4; Coninck et al., 2018). It is therefore recognised that such actions require financial, institutional and policy support for their wide-scale adoption and sustainability over time (3.6.3; 4.8.5; 6.4.4).

3.6.1. SLM Technologies and Practices: on the Ground Actions

A broad range of activities and measures can help avoid, reduce and reverse degradation across the dryland areas of the world. Many of these actions also contribute to climate change adaptation and mitigation, with further sustainable development co-benefits for poverty reduction and food security *(high confidence*) (6.3). As preventing desertification is strongly preferable and more cost-effective than allowing land to degrade and then attempting to restore it (IPBES, 2018b; Webb et al., 2013), there is a growing emphasis on avoiding and reducing land degradation, following the Land Degradation Neutrality framework (Cowie et al., 2018; Orr et al., 2017; 4.8.5).

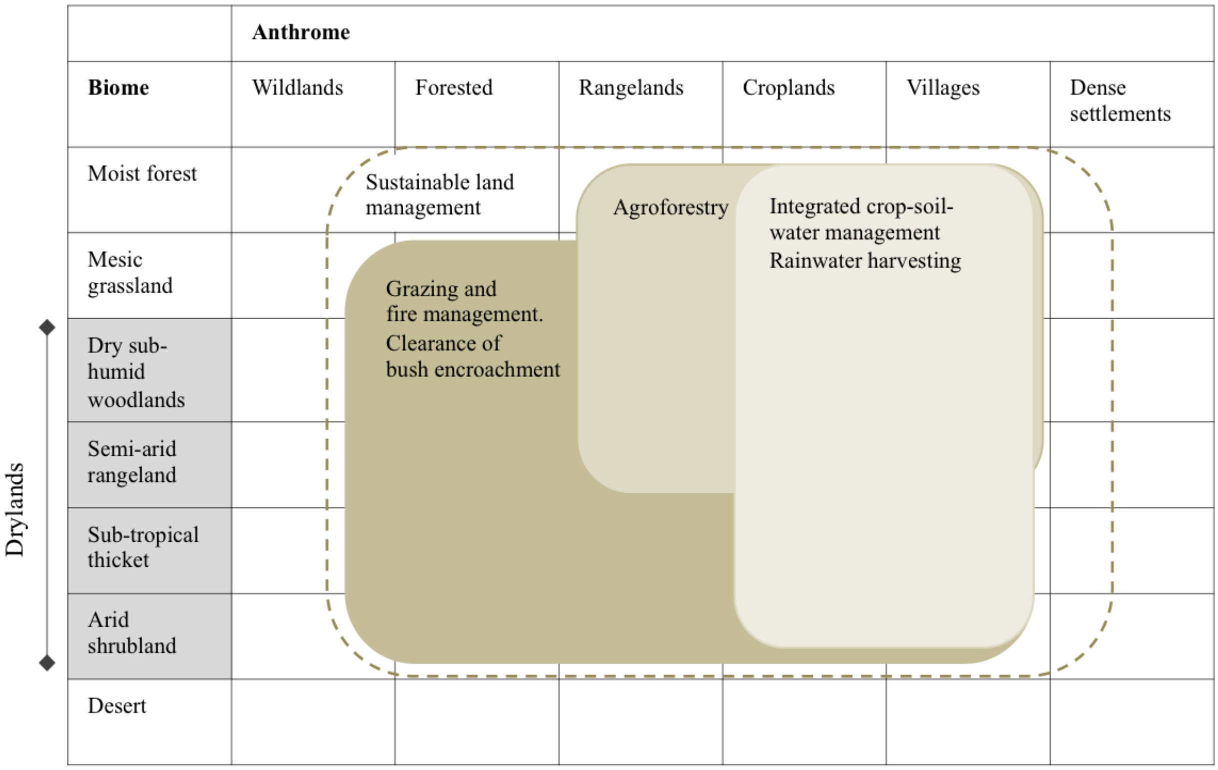


Figure 3.10 The typical distribution of on-the-ground actions across global biomes and anthromes

An assessment is made of six activities and measures practicable across the biomes and anthromes of the dryland domain (Figure 3.10). This suite of actions is not exhaustive, but rather a set of activities that are particularly pertinent to global dryland ecosystems. They are not necessarily exclusive to drylands and are often implemented across a range of biomes and anthromes (Figure 3.10). For afforestation, see 3.7.2, Cross-Chapter Box 2 in Chapter 1 and Chapter 4 (4.8.3). The use of anthromes as a structuring element for response options is based on the essential role of interactions between social and ecological systems in driving desertification within coupled socio-ecological systems (Cherlet et al., 2018). The concept of the anthromes is defined in the glossary and explored further in Chapters 1, 4 and 6.

The assessment of each action is twofold: firstly, to assess the ability of each action to address desertification and enhance climate change resilience, and secondly, to assess the potential impact of future climate change on the effectiveness of each action.

3.6.1.1. Integrated Crop-Soil-Water Management

Forms of integrated cropland management have been practiced in drylands for over thousands of years (Knörzer et al., 2009). Actions include planting a diversity of species including drought tolerant crops, reducing tillage, applying organic compost and fertiliser, adopting different forms of irrigation and maintaining vegetation and mulch cover. In the contemporary era, several of these actions have been adopted in response to climate change.

In terms of climate change *adaptation*, the resilience of agriculture to the impacts of climate change is strongly influenced by the underlying health and stability of soils as well as improvements in crop varieties, irrigation efficiency and supplemental irrigation, e.g. through rainwater harvesting (*medium evidence, high agreement*, Altieri et al., 2015; Amundson et al., 2015; Derpsch et al., 2010; Lal, 1997; de Vries et al., 2012). Desertification often leads to a reduction in ground cover that in turn results in accelerated water and wind erosion and an associated loss of fertile topsoil that can greatly reduce the resilience of agriculture to climate change (*medium evidence, high agreement*, (Touré et al., 2019; Amundson et al., 2015; Borrelli et al., 2017; Pierre et al., 2017). Amadou et al. (2011) note that even a minimal cover of crop residues (100 kg ha^{-1}) can substantially decrease wind erosion.

Compared to conventional (flood or furrow) irrigation, drip irrigation methods are more efficient in supplying water to the plant root zone, resulting in lower water requirements and enhanced water use efficiency (*robust evidence* and *high agreement*) (Ibragimov et al., 2007; Narayanamoorthy, 2010; Niaz et al., 2009). For example, in the rainfed area of Fetehjang, Pakistan, the adoption of drip methods reduced water usage by 67-68% during the production of tomato, cucumber and bell peppers, resulting in a 68-79% improvement in water use efficiency compared to previous furrow irrigation (Niaz et al., 2009). In India, drip irrigation reduced the amount of water consumed in the production of sugarcane by 44%, grapes by 37%, bananas by 29% and cotton by 45%, while enhancing the yields by up to 29% (Narayanamoorthy, 2010). Similarly, in Uzbekistan, drip irrigation increased the yield of cotton by 10-19% while reducing water requirements by 18-42% (Ibragimov et al., 2007).

A prominent response that addresses soil loss, health and cover is altering cropping methods. The adoption of intercropping (inter- and intra- row planting of companion crops) and relay cropping (temporally differentiated planting of companion crops) maintains soil cover over a larger fraction of the year, leading to an increase in production, soil N, species diversity and a decrease in pest abundance (*robust evidence and medium agreement*, (Altieri and Koohafkan, 2008; Tanveer et al., 2017; Wilhelm and Wortmann, 2004). For example, intercropping maize and sorghum with *Desmodium* (an insect repellent forage legume) and *Brachiaria* (an insect trapping grass), which is being promoted in drylands of East Africa, led to a two-three fold increase in maize production and an 80% decrease in stem boring insects (Khan et al., 2014). In addition to changes in cropping methods, forms of agroforestry and shelter belts are often used to reduce erosion and improve soil conditions (3.7.2). For example, the use of tree belts of mixed species in northern China led to a reduction of surface wind speed and an associated reduction in soil temperature by up to 40% and an increase in soil moisture by up to 30% (Wang et al., 2008).

A further measure that can be of increasing importance under climate change is rainwater harvesting (RWH), including traditional zai (small basins used to capture surface runoff), earthen bunds and ridges (Nyamadzawo et al., 2013), *fanya juus* infiltration pits (Nyagumbo et al., 2019), contour stone bunds (Garrity et al., 2010) and semi-permeable stone bunds (often referred to by the French term "digue filtrante") (Taye et al., 2015). RWH increases the amount of water available for agriculture and livelihoods through the capture and storage of runoff, while at the same time, reducing the intensity of peak flows following high intensity rainfall events. It is therefore often highlighted as a practical response to dryness (i.e. long-term aridity and low seasonal precipitation) and rainfall variability projected to become more acute over time in some dryland areas (Dile et al., 2013; Vohland and Barry, 2009). For example, for Wadi Al-Lith drainage in Saudi Arabia, the use of rainwater harvesting was suggested as a key climate change adaptation action (Almazroui et al., 2017). There is *robust evidence and high agreement* that the implementation of RWH systems leads to an increase in agricultural production in drylands (see reviews by Biazin et al., 2012; Bouma and Wösten, 2016; Dile et al., 2013). A global meta-analysis of changes in crop production due to the adoption of RWH techniques noted an average increase in yields of 78%, ranging from –28% to 468% (Bouma and Wösten, 2016). Of particular relevance to climate change in drylands is that the relative impact of RWH on agricultural production generally increases with increasing dryness. Relative yield improvements due to the adoption of RWH were significantly higher in years with less than 330 mm rainfall, compared to years with more than 330 mm (Bouma and Wösten, 2016). Despite delivering a clear set of benefits, there are some issues that need to be considered. The impact RWH may vary at different temporal and spatial scales (Vohland and Barry, 2009). At a plot scale, RWH structures may increase available water and enhance agricultural production, SOC and nutrient availability, yet at a catchment scale, they may reduce runoff to downstream uses (Meijer et al., 2013; Singh et al., 2012; Vohland and Barry, 2009; Yosef and Asmamaw, 2015). Inappropriate storage of water in warm climes can lead to an increase in water related diseases unless managed correctly, for example, schistosomiasis and malaria (see review by Boelee et al., 2013).

Integrated crop-soil-water management may also deliver climate change *mitigation* benefits through avoiding, reducing and reversing the loss of SOC (Table 6.5). Approximately 20-30 Pg of SOC have been released into the atmosphere through desertification processes, for example, deforestation, overgrazing and conventional tillage (Lal, 2004). Activities, such as those associated with conservation agriculture (minimising tillage, crop rotation, maintaining organic cover and planting a diversity of species), reduce erosion, improve water use efficiency and primary production, increase inflow of organic material and enhance SOC over time, contributing to climate change mitigation aand adaptation (*high confidence*) (Plaza-Bonilla et al., 2015; Lal, 2015; Srinivasa Rao et al., 2015; Sombrero and de Benito, 2010). Conservation agriculture practices also lead to increases in SOC (*medium confidence*). However, sustained C sequestration is dependent on net primary productivity and on the availability of crop-residues that may be relatively limited and often consumed by livestock or used elsewhere in dryland contexts (Cheesman et al., 2016; Plaza-Bonilla et al., 2015). For this reason, expected rates of C sequestration following changes in agricultural practices in drylands are relatively low (0.04-0.4 t C ha^{-1}) and it may take a protracted period of time, even several decades, for C stocks to recover if lost (*medium confidence*) (Farage et al., 2007; Hoyle, D'Antuono, Overheu, and Murphy, 2013; Lal, 2004). This long recovery period enforces the rationale for prioritising avoiding and reducing land degradation and loss of C, in addition to restoration activities.

3.6.1.2. Grazing and Fire Management in Drylands

Rangeland management systems such as sustainable grazing approaches and re-vegetation increase rangeland productivity (*high confidence*) (Table 6.5). Open grassland, savanna and woodland are home to the majority of world's livestock production (Safriel et al., 2005). Within these drylands areas, prevailing grazing and fire regimes play an important role in shaping the relative abundance of trees versus grasses (Scholes and Archer, 1997; Staver et al., 2011; Stevens et al., 2017), as well as the health of the grass layer in terms of primary production, species richness and basal cover (the propotion of the plant that is in the soil) (Plaza-Bonilla et al., 2015; Short et al., 2003). This in turn influences levels of soil erosion, soil nutrients, secondary production and additional ecosystem services (Divinsky et al., 2017; Pellegrini et al., 2017). A further set of drivers, including soil type, annual rainfall and changes in atmospheric CO_2 may also define observed rangeland structure and composition (Devine et al., 2017; Donohue et al., 2013), but the two principal factors that pastoralists can manage are grazing and fire by altering their frequency, type and intensity.

The impact of grazing and fire regimes on biodiversity, soil nutrients, primary production and further ecosystem services is not constant and varies between locations (Divinsky et al., 2017; Fleischner, 1994; van Oijen et al., 2018). Trade-offs may therefore need to be considered to ensure that rangeland diversity and production are resilient to climate change (Plaza-Bonilla et al., 2015; van Oijen et al., 2018). In certain locations, even light to moderate grazing have led to a significant decrease in the occurrence of particular species, especially forbs (O'Connor et al., 2011; Scott-shaw and Morris, 2015). In other locations, species richness is only significantly impacted by heavy grazing and is able to withstand light to moderate grazing (Divinsky et al., 2017). A context specific evaluation of how grazing and fire impact particular species may therefore be required to ensure the persistence of target species over time (Marty, 2005). A similar trade-off may need to be considered between soil C sequestration and livestock production. As noted by Plaza-Bonilla et al. (2015) increasing grazing pressure has been found to both increase and decrease SOC stocks in different locations. Where it has led to a decrease in soil C stocks, for example in Mongolia (Han et al., 2008) and Ethiopia (Bikila et al., 2016), trade-offs between C sequestration and the value of livestock to local livelihoods need be considered.

Although certain herbaceous species may be unable to tolerate grazing pressure, a complete lack of grazing or fire may not be desired in terms of ecosystems health. It can lead to a decrease in basal cover and the accumulation of moribund, unpalatable biomass that inhibits primary production (Manson et al., 2007; Scholes, 2009). The utilisation of the grass sward through light to moderate grazing stimulates the growth of biomass, basal cover and allows water services to be sustained over time (Papanastasis et al., 2017; Scholes, 2009). Even, moderate to heavy grazing in periods of higher rainfall may be sustainable, but constant heavy grazing during dry periods and especially droughts can lead to a reduction in basal cover, SOC, biological soil crusts, ecosystem services and an accelerated erosion (*high agreement, robust evidence*, (Archer et al., 2017; Conant and Paustian, 2003; D'Odorico et al., 2013; Geist and Lambin, 2004; Havstad et al., 2006; Huang et al., 2007; Manzano and Návar, 2000; Pointing and Belnap, 2012; Weber et al., 2016). For this reason, the inclusion of drought forecasts and contingency planning in grazing and fire management programs is crucial to avoid desertification (Smith and Foran, 1992; Torell et al., 2010). It is an important component of avoiding and reducing early degradation. Although grasslands systems may be relatively resilient and can often recover from a moderately degraded state (Khishigbayar et al., 2015; Porensky et al., 2016), if a tipping point has been exceeded, restoration to a historic state may not be economical or ecologically feasible (D'Odorico et al., 2013).

Together with livestock management (Table 6.5), the use of fire is an integral part of rangeland management and can be applied to remove moribund and unpalatable forage, exotic weeds and woody species (Archer et al., 2017). Fire has less of an effect on SOC and soil nutrients in comparison to grazing (Abril et al., 2005), yet elevated fire frequency has been observed to lead to a decrease in soil C and N (Abril et al., 2005; Bikila et al., 2016; Bird et al., 2000; Pellegrini et al., 2017). Although the impact of climate change on fire frequency and intensity may not be clear due to its differing impact on fuel accumulation, suitable weather conditions and sources of ignition (Abatzoglou et al., 2018; Littell et al., 2018; Moritz et al., 2012), there is an increasing use of prescribed fire to address several global change phenomena, for example, the spread of invasive species and bush encroachment as well as the threat of intense runaway fires (Fernandes et al., 2013; McCaw, 2013; van Wilgen et al., 2010). Cross-Chapter Box 3 located in Chapter 2 provides a further review of the interaction between fire and climate change.

There is often much emphasis on reducing and reversing the degradation of rangelands due to the wealth of benefits they provide, especially in the context of assisting dryland communities to adapt to climate change (Webb et al., 2017; Woollen et al., 2016). The emerging concept of ecosystem-based adaptation has highlighted the broad range of important ecosystem services that healthy rangelands can provide in a resilient manner to local residents and downstream economies (Kloos and Renaud, 2016; Reid et al., 2018). In terms of climate change mitigation, the contribution of rangelands, woodland and sub-humid dry forest (e.g. Miombo woodland in south-central Africa) is often undervalued due to relatively low C stocks per hectare. Yet due to their sheer extent, the amount of C sequestered in these ecosystems is substantial and can make a valuable contribution to climate change mitigation (Lal, 2004; Pelletier et al., 2018).

3.6.1.3. Clearance of Bush Encroachment

The encroachment of open grassland and savanna ecosystems by woody species has occurred for at least the past 100 years (Archer et al., 2017; O'Connor et al., 2014; Schooley et al., 2018). Dependent on the type and intensity of encroachment, it may lead to a net loss of ecosystem services and be viewed as a form of desertification (Dougill et al., 2016; O'Connor et al., 2014). However, there are circumstances where bush encroachment may lead to a net increase in ecosystem services, especially at intermediate levels of encroachment, where the ability of the landscape to produce fodder for livestock is retained, while the production of wood and associated products increases (Eldridge et al., 2011; Eldridge and Soliveres, 2014). This may be particularly important in regions such as southern Africa and India where over 65% of rural households depend on fuelwood from surrounding landscapes as well as livestock production (Komala and Prasad, 2016; Makonese et al., 2017; Shackleton and Shackleton, 2004).

This variable relationship between the level of encroachment, C stocks, biodiversity, provision of water and pastoral value (Eldridge and Soliveres, 2014) can present a conundrum to policy makers, especially when considering the goals of three Rio Conventions - UNFCCC, UNCCD and UNCBD. Clearing intense bush encroachment may improve species diversity, rangeland productivity, the provision of water and decrease desertification, thereby contributing to the goals of the UNCBD, UNCCD as well as adaptation aims of the UNFCCC. However, it would lead to the release of biomass C stocks into the atmosphere and potentially conflict with the mitigation aims of the UNFCCC.

For example, Smit et al. (2015) observed an average increase in above-ground woody C stocks of 44 t C ha^{-1} in savannas in northern Namibia. However, since bush encroachment significantly inhibited livestock production, there are often substantial efforts to clear woody species (Stafford-Smith et al., 2017). Namibia has an early national programme aimed at clearing woody species through mechanical measures (harvesting of trees) as well as the application of arboricides (Smit et al., 2015). However, the long-term success of clearance and subsequent improved fire and grazing management remains to be evaluated, especially restoration back towards an 'original open grassland state'. For example, in northern Namibia, the rapid reestablishment of woody seedlings has raised questions about whether full clearance and restoration is possible (Smit et al., 2015). In arid landscapes, the potential impact of elevated atmospheric CO_2 (Donohue et al., 2013; Kgope et al., 2010) and opportunity to implement high intensity fires that remove woody species and maintain rangelands in an open state has been questioned (Bond and Midgley, 2000). If these drivers of woody plant encroachment cannot be addressed, a new form of "emerging ecosystem' (Milton, 2003) may need to be explored that includes both improved livestock and fire management as well as the utilisation of biomass as a long-term commodity and source of revenue (Smit et al., 2015). Initial studies in Namibia and South Africa (Stafford-Smith et al., 2017) indicate that there may be good opportunity to produce sawn timber, fencing poles, fuel wood and commercial energy, but factors such as the cost of transport can substantially influence the financial feasibility of implementation.

The benefit of proactive management that prevents land from being degraded (altering grazing systems or treating bush encroachment at early stages before degradation has been initiated) is more cost-effective in the long-term and adds resistance to climate change then treating lands after degradation has occurred (Webb et al., 2013; Weltz and Spaeth, 2012). The challenge is getting producers to alter their management paradigm from short-term objectives to long-term objectives.

3.6.1.4. Combating sand and dust storms through sand dune stabilisation

Dust and sand storms have a considerable impact on natural and human systems (3.4.1, 3.4.2). Application of sand dune stabilisation techniques contributes to reducing sand and dust storms (*high confidence*). Using a number of methods, sand dune stabilisation aims to avoid and reduce the occurrence of dust and sand storms (Mainguet and Dumay, 2011). Mechanical techniques include building palisades to prevent the movement of sand and reduce sand deposits on infrastructure. Chemical methods include the use of calcium bentonite or using silica gel to fix mobile sand (Aboushook et al., 2012; Rammal and Jubair, 2015). Biological methods include the use of mulch to stabilise surfaces (Sebaa et al., 2015; Yu et al., 2004) and establishing permanent plant cover using pasture species that improve grazing at the same time (Abdelkebir and Ferchichi, 2015; Zhang et al., 2015; 3.7.1.3). When the dune is stabilised, woody perennials are introduced that are selected according to climatic and ecological conditions (FAO, 2011). For example, such revegetation processes have been implemented on the shifting dunes of the Tengger Desert in northern China leading to the stabilisation of sand and the sequestration of up to 10 t C ha^{-1} over a period of 55 years (Yang et al., 2014).

3.6.1.5 Use of Halophytes for the Revegetation of Saline Lands

Soil salinity and sodicity can severely limit the growth and productivity of crops (Jan et al., 2017) and lead to a decrease in available arable land. Leaching and drainage provides a possible solution, but can be prohibitively expensive. An alternative, more economical option, is the growth of halophytes (plants that are adapted to grow under highly saline conditions) that allow saline land to be used in a productive manner (Qadir et al., 2000). The biomass produced can be used as forage, food, feed, essential oils, biofuel, timber, fuelwood (Chughtai et al., 2015; Mahmood et al.,2016; Sharma et al., 2016). A further co-benefit is the opportunity to mitigate climate change through the enhancement of terrestrial C stocks as land is revegetated (Dagar et al., 2014; Wicke et al., 2013). The combined use of salt-tolerant crops, improved irrigation practices, chemical remediation measures and appropriate mulch and compost is effective in reducing the impact of secondary salinisation (*medium confidence*).

In Pakistan, where about 6.2 M ha of agricultural land is affected by salinity, pioneering work on utilising salt tolerant plants for the revegetation of saline lands (Biosaline Agriculture) was done in the early 1970s (NIAB, 1997). A number of local and exotic varieties were initially screened for salt tolerance in lab- and greenhouse based studies, and then distributed to similar saline areas (Ashraf et al., 2010). These included tree species (*Acacia ampliceps*, *A. nilotica, Eucalyptus camaldulensis, Prosopis juliflora, Azadirachta indica*) (Awan and Mahmood, 2017), forage plants (*Leptochloa fusca*, *Sporobolus arabicus*, *Brachiaria mutica*, *Echinochloa* sp., *Sesbania* and *Atriplex* spp.) and crop species including varieties of barley (*Hordeum vulgare*), cotton, wheat (*Triticum aestivum*) and *Brassica* spp (Mahmood et al.,2016) as well as fruit crops in the form of Date Palm (*Phoenix dactylifera*) that has high salt tolerance with no visible adverse effects on seedlings (Yaish and Kumar, 2015; Al-Mulla et al., 2013; Alrasbi et al., 2010). Pomegranate (*Punica granatum L.*) is another fruit crop of moderate to high salt tolerance. Through regulating growth form and nutrient balancing, it can maintain water content, chlorophyll fluorescence and enzyme activity at normal levels (Ibrahim, 2016; Okhovatian-Ardakani et al., 2010).

In India and elsewhere, tree species including *Prosopis juliflora*, *Dalbergia sissoo*, *Eucalyptus tereticornis* have been used to revegetate saline land. Certain biofuel crops in the form of *Ricinus communis* (Abideen et al., 2014), *Euphorbia antisyphilitica* (Dagar et al., 2014), *Karelinia caspia* (Akinshina et al., 2016) and *Salicornia* spp. (Sanandiya and Siddhanta, 2014) are grown in saline areas, and *Panicum turgidum* (Koyro et al., 2013) and *Leptochloa fusca* (Akhter et al., 2003) have been grown as fodder crop on degraded soils with brackish water. In China, intense efforts are being made on the use of halophytes (Sakai et al., 2012; Wang et al., 2018). These examples reveal that there is great scope still use saline areas in a productive manner through the utilisation of halophytes. The most productive species often have yields equivalent to conventional crops, at salinity levels matching even that of sea water.

3.6.2. Socio-economic Responses

Socio-economic and policy responses are often crucial in enhancing the adoption of SLM practices (Cordingley et al., 2015; Fleskens and Stringer, 2014; Nyanga et al., 2016) and for assisting agricultural households to diversify their sources of income (Barrett et al., 2017; Shiferaw and Djido, 2016). Technology and socio-economic responses are not independent, but continuously interact.

3.6.2.1. Socio-economic Responses for Combating Desertification Under Climate Change

Desertification limits the choice of potential climate change mitigation and adaptation response options by reducing climate change adaptive capacities. Furthermore, many additional factors, for example, a lack of access to markets or insecurity of land tenure, hinder the adoption of SLM. These factors are largely beyond the control of individuals or local communities and require broader policy interventions (3.6.3). Nevertheless, local collective action and indigenous and local knowledge are still crucial to the ability of households to respond to the combined challenge of climate change and desertification. Raising awareness, capacity building and development to promote collective action and indigenous and local knowledge contribute to avoiding, reducing and reversing desertification under changing climate.

The use of indigenous and local knowledge enhances the success of SLM and its ability to address desertification (Altieri and Nicholls, 2017; Engdawork and Bork, 2016). Using indigenous and local knowledge for combating desertification could contribute to climate change adaptation strategies (Belfer et al., 2017; Codjoe et al., 2014; Etchart, 2017; Speranza et al., 2010; Makondo and Thomas, 2018; Maldonado et al., 2016; Nyong et al., 2007). There are abundant examples of how indigenous and local knowledge, which are an important part of broader agroecological knowledge (Altieri, 2018), have allowed livelihood systems in drylands to be maintained despite environmental constraints. An example is the numerous traditional water harvesting techniques that are used across the drylands to adapt to dry spells and climate change. These include creating planting pits ("zai", "ngoro") and micro-basins, contouring hill slopes and terracing (Biazin et al., 2012) (3.6.1). Traditional "ndiva" water harvesting system in Tanzania enables the capture of runoff water from highland areas to downstream community-managed micro-dams for subsequent farm delivery through small scale canal networks (Enfors and Gordon, 2008). A further example are pastoralist communities located in drylands who have developed numerous methods to sustainably manage rangelands. Pastoralist communities in Morocco developed the "agdal" system of seasonally alternating use of rangelands to limit overgrazing (Dominguez, 2014) as well as to manage forests in the Moroccan High Atlas Mountains (Auclair et al., 2011). Across the Arabian Peninsula and North Africa, a rotational grazing system "hema" was historically practiced by the Bedouin communities (Hussein, 2011; Louhaichi and Tastad, 2010). The Beni-Amer herders in the Horn of Africa have developed complex livestock breeding and selection systems (Fre, 2018). Although well adapted to resource-sparse dryland environments, traditional practices are currently not able to cope with increased demand for food and environmental changes (Enfors and Gordon, 2008; Engdawork and Bork, 2016). Moreover, there is *robust evidence* documenting the marginalisation or loss of indigenous and local knowledge (Dominguez, 2014; Fernández-Giménez and Fillat Estaque, 2012; Hussein, 2011; Kodirekkala, 2017; Moreno-Calles et al., 2012). Combined use of indigenous and local knowledge and new SLM technologies can contribute to raising resilience to the challenges of climate change and desertification (*high confidence*) (Engdawork and Bork, 2016; Guzman et al., 2018).

Collective action has the potential to contribute to SLM and climate change adaptation (*medium confidence*) (Adger, 2003; Engdawork and Bork, 2016; Eriksen and Lind, 2009; Ostrom, 2009; Rodima-Taylor et al., 2012). Collective action is a result of social capital. Social capital is divided into structural and cognitive forms, structural corresponding to strong networks (including outside one's immediate community) and cognitive encompassing mutual trust and cooperation within communities (van Rijn et al., 2012; Woolcock and Narayan, 2000). Social capital is more important for economic growth in settings with weak formal institutions, and less so in those with strong enforcement of formal institutions (Ahlerup et al., 2009). There are cases throughout the drylands showing that community bylaws and collective action successfully limited land degradation and facilitated SLM (Ajayi et al., 2016; Infante, 2017; Kassie et al., 2013; Nyangena, 2008; Willy and Holm-Müller, 2013; Wossen et al., 2015). However, there are also cases when they did not improve SLM where they were not strictly enforced (Teshome et al., 2016). Collective action for implementing responses to dryland degradation is often hindered by local asymmetric power relations and "elite capture" (Kihiu, 2016; Stringer et al., 2007). This illustrates that different levels and types of social capital result in different levels of collective action. In a sample of East, West and southern African countries, structural social capital in the form of access to networks outside one's own community was suggested to stimulate the adoption of agricultural innovations, whereas cognitive social capital, associated with inward-looking community norms of trust and cooperation, was found to have a negative relationship with the adoption of agricultural innovations (van Rijn et al., 2012). The latter is indirectly corroborated by observations of the impact of community-based rangeland management organisations in Mongolia. Although levels of cognitive social capital did not differ between them, communities with strong links to outside networks were able to apply more innovative rangeland management practices in comparison to communities without such links (Ulambayar et al., 2017).

Farmer-led innovations. Agricultural households are not just passive adopters of externally developed technologies, but are active experimenters and innovators (Reij and Waters-Bayer, 2001; Tambo and Wünscher, 2015; Waters-Bayer et al., 2009). SLM technologies co-generated through direct participation of agricultural households have higher chances of being accepted by them (*medium confidence*) (Bonney et al., 2016; Vente et al., 2016). Usually farmer-driven innovations are more frugal and better adapted to their resource scarcities than externally introduced technologies (Gupta et al., 2016). Farmer-to-farmer sharing of their own innovations and mutual learning positively contribute to higher technology adoption rates (Dey et al., 2017). This innovative ability can be given a new dynamism by combining it with emerging external technologies. For example, emerging low-cost phone applications that are linked to soil and water monitoring sensors can provide farmers with previously inaccessible information and guidance (Cornell et al., 2013; Herrick et al., 2017; McKinley et al., 2017; Steger et al., 2017).

Currently, the adoption of SLM practices remains insufficient to address desertification and contribute to climate change adaptation and mitigation more extensively. This is due to the constraints on the use of indigenous and local knowledge and collective action, as well as economic and institutional barriers for SLM adoption (3.1.4.2; 3.6.3; Banadda, 2010; Cordingley et al., 2015; Lokonon and Mbaye, 2018; Mulinge et al., 2016; Wildemeersch et al., 2015). Sustainable development of drylands under these socio-economic and environmental (climate change, desertification) conditions will also depend on the ability of dryland agricultural households to diversify their livelihoods sources (Boserup, 1965; Safriel and Adeel, 2008).

3.6.2.2. Socio-Economic Responses for Economic Diversification

Livelihood diversification through non-farm employment increases the resilience of rural households against desertification and extreme weather events by diversifying their income and consumption (*high confidence*). Moreover, it can provide the funds to invest into SLM (Belay et al., 2017; Bryan et al., 2009; Dumenu and Obeng, 2016; Salik et al., 2017; Shiferaw et al., 2009). Access to non-agricultural employment is especially important for poorer pastoral households as their small herd sizes make them less resilient to drought (Fratkin, 2013; Lybbert et al., 2004). However, access to alternative opportunities is limited in the rural areas of many developing countries, especially for women and marginalised groups who lack education and social networks (Reardon et al., 2008).

Migration is frequently used as an adaptation strategy to environmental change (*medium confidence*). Migration is a form of livelihood diversification and a potential response option to desertification and increasing risk to agricultural livelihoods under climate change (Walther et al., 2002). Migration can be short-term (e.g., seasonal) or long-term, internal within a country or international. There is *medium evidence* showing rural households responding to desertification and droughts through all forms of migration, for example: during the Dust Bowl in the United States in the 1930s (Hornbeck, 2012); during droughts in Burkina Faso in the 2000s (Barbier et al., 2009); in Mexico in the 1990s (Nawrotzki et al., 2016); and by the Aymara people of the semiarid Tarapacá region in Chile between 1820-1970 responding to declines in rainfall and growing demands for labor outside the region (Lima et al., 2016). There is *robust evidence and high agreement* showing that migration decisions are influenced by a complex set of different factors, with desertification and climate change playing relatively lesser roles (Liehr et al., 2016) (3.4.2). Barrios et al. (2006) found that urbanisation in Sub-Saharan Africa was partially influenced by climatic factors during the 1950 to 2000 period, in parallel to liberalisation of internal restrictions on labour movements: with 1% reduction in rainfall associated with 0.45% increase in urbanisation. This migration favoured more industrially-diverse urban areas in Sub-Saharan Africa (Henderson et al., 2017), because they offer more diverse employment opportunities and higher wages. Similar trends were also observed in Iran in response to water scarcity (Madani et al., 2016). However, migration involves some initial investments. For this reason, reductions in agricultural incomes due to climate change or desertification have the potential to decrease out-migration among the poorest agricultural households who become less able to afford migration (Cattaneo and Peri, 2016), thus increasing social inequalities. There is *medium evidence* and *high agreement* that households with migrant worker members are more resilient against extreme weather events and environmental degradation compared to non-migrant households who are more dependent on agricultural income (Liehr et al., 2016; Salik et al., 2017; Sikder and Higgins, 2017). Remittances from migrant household members potentially contribute to SLM adoptions, however, substantial out-migration was also found to constrain the implementation of labour-intensive land management practices (Chen et al., 2014; Liu et al., 2016a).

3.6.3. Policy Responses

The adoption of SLM practices depends on the compatibility of the technology with prevailing socio-economic and biophysical conditions (Sanz et al., 2017). Globally, it was shown that every USD invested into restoring degraded lands yields social returns, including both provisioning and non-provisioning ecosystem services, in the range of USD 3–6 over a 30-year period (Nkonya et al., 2016a). A similar range of returns from land restoration activities were found in Central Asia (Mirzabaev et al., 2016), Ethiopia (Gebreselassie et al., 2016), India (Mythili and Goedecke, 2016), Kenya (Mulinge et al., 2016), Niger (Moussa et al., 2016) and Senegal (Sow et al., 2016). Despite these relatively high returns, there is *robust evidence* that the adoption of SLM practices remains low (Cordingley et al., 2015; Giger et al., 2015; Lokonon and Mbaye, 2018). Part of the reason for these low adoption rates is that the major share of the returns from SLM are social benefits, namely in the form of non-provisioning ecosystem services (Nkonya et al., 2016a). The adoption of SLM technologies does not always provide implementers with immediate private benefits (Schmidt et al., 2017). High initial investment costs, institutional and governance constraints and a lack of access to technologies and equipment may inhibit their adoption further (Giger et al., 2015; Sanz et al., 2017; Schmidt et al., 2017). However, not all SLM practices have high upfront costs. Analysing the World Overview of Conservation Approaches and Technologies (WOCAT) database, a globally acknowledged reference database for SLM, Giger et al. (2015) found that the upfront costs of SLM technologies ranged from about USD 20 to USD 5000, with the median cost being around USD 500 . Many SLM technologies are profitable within three to 10 years (*medium evidence*, *high agreement*) (Djanibekov and Khamzina, 2016; Giger et al., 2015; Moussa et al., 2016; Sow et al., 2016). About 73% of 363 SLM technologies evaluated were reported to become profitable within three years, while 97% were profitable within 10 years (Giger et al., 2015). Similarly, it was shown that social returns from investments in restoring degraded lands will exceed their costs within six years in many settings across drylands (Nkonya et al., 2016a). However, even with affordable upfront costs, market failures in the form of lack of access to credit, input and output markets, and insecure land tenure (3.1.3) result in the lack of adoption of SLM technologies (Moussa et al., 2016). Payments for ecosystem services, subsidies for SLM, encouragement of community collective action can lead to a higher level of adoption of SLM and land restoration activities (*medium confidence*) (Bouma and Wösten, 2016; Lambin et al., 2014; Reed et al., 2015; Schiappacasse et al., 2012; van Zanten et al., 2014; 3.6.3). Enabling policy responses discussed in this section contribute to overcoming these market failures.

Many socio-economic factors shaping individual responses to desertification typically operate at larger scales. Individual households and communities do not exercise control over these factors, such as land tenure insecurity, lack of property rights, lack of access to markets, availability of rural advisory services, and agricultural price distortions. These factors are shaped by national government policies and international markets. As in the case with socio-economic responses, policy responses are classified below in two ways: those which seek to combat desertification under changing climate; and those which seek to provide alternative livelihood sources through economic diversification. These options are mutually complementary and contribute to all the three hierarchical elements of the Land Degradation Neutrality (LDN) framework, namely, avoiding, reducing and reversing land degradation (Cowie et al., 2018; Orr et al., 2017; 4.8.5; Table 7.2; 7.4.5). Enabling policy environment is a critical element for the achievement of LDN (Chasek et al., 2019). Implementation of LDN policies can contribute to climate change adaptation and mitigation (*high confidence*) (3.6.1, 3.7.2).

3.6.3.1. Policy Responses towards Combating Desertification under Climate Change

Policy responses to combat desertification take numerous forms (Marques et al., 2016). Below we discuss major policy responses consistently highlighted in the literature in connection with SLM and climate change, because these response options were found to strengthen adaptation capacities and to contribute to climate change mitigation. They include improving market access, empowering women, expanding access to agricultural advisory services, strengthening land tenure security, payments for ecosystem services, decentralised natural resource management, investing into research and monitoring of desertification and dust storms, and investing into modern renewable energy sources.

Policies aiming at improving market access, that is the ability to access output and input markets at lower costs, help farmers and livestock producers earn more profit from their produce. Increased profits both motivate and enable them to invest more in SLM. Higher access to input, output and credit markets was consistently found as a major factor in the adoption of SLM practices in a wide number of settings across the drylands (*medium confidence*) (Aw-Hassan et al., 2016; Gebreselassie et al., 2016; Mythili and Goedecke, 2016; Nkonya and Anderson, 2015; Sow et al., 2016). Lack of access to credit limits adjustments and agricultural responses to the impacts of desertification under changing climate, with long-term consequences on the livelihoods and incomes, as was shown for the case of the American Dust Bowl during 1930s (Hornbeck, 2012). Government policies aimed at improving market access usually involve constructing and upgrading rural-urban transportation infrastructure and agricultural value chains, such as investments into construction of local markets, abattoirs and cold storage warehouses, as well as post-harvest processing facilities (Mcpeak et al., 2006). However, besides infrastructural constraints, providing improved access often involves relieving institutional constraints to market access (Little, 2010), such as improved coordination of cross-border food safety and veterinary regulations (Ait Hou et al., 2015; Keiichiro et al., 2015; Mcpeak et al., 2006; Unnevehr, 2015), and availability and access to market information systems (Bobojonov et al., 2016; Christy et al., 2014; Nakasone et al., 2014).

Women's empowerment. A greater emphasis on understanding gender-specific differences over land-use and land management practices as an entry point can make land restoration projects more successful (*medium confidence*) (Broeckhoven and Cliquet, 2015; Carr and Thompson, 2014; Catacutan and Villamor, 2016; Dah-gbeto and Villamor, 2016). In relation to representation and authority to make decisions in land management and governance, women's participation remains lacking particularly in the dryland regions. Thus, ensuring women's rights means accepting women as equal members of the community and citizens of the state (Nelson et al., 2015). This includes equitable access of women to resources (including extension services), networks, and markets. In areas where socio-cultural norms and practices devalue women and undermine their participation, actions for empowering women will require changes in customary norms, recognition of women's (land) rights in government policies and programmes to assure that their interests are better represented (1.4.2; Cross-Chapter Box 11: Gender, Chapter 7). In addition, several novel concepts are recently applied for an in-depth understanding of gender in relation to science-policy interface. Among these are the concepts of intersectionality, i.e. how social dimensions of identity and gender are bound up in systems of power and social institution (Thompson-Hall et al., 2016), bounded rationality for gendered decision making, related to incomplete information interacting with limits to human cognition leading to judgement errors or objectively poor decision making (Villamor and van Noordwijk, 2016), anticipatory learning for preparing for possible contingencies and consideration of long-term alternatives (Dah-gbeto and Villamor, 2016) and systematic leverage points for interventions that produce, mark, and entrench gender inequality within communities (Manlosa et al., 2018), which all aim to improve gender equality within agro-ecological landscapes through a systems approach.

Education and expanding access to agricultural services. Providing access to information about SLM practices facilitates their adoption (*medium confidence*) (Kassie et al., 2015; Nkonya et al., 2015; Nyanga et al., 2016). Moreover, improving the knowledge of climate change, capacity building and development in rural areas can help strengthen climate change adaptive capacities (Berman et al., 2012; Chen et al., 2018; Descheemaeker et al., 2018; Popp et al., 2009; Tambo, 2016; Yaro et al., 2015). Agricultural initiatives to improve the adaptive capacities of vulnerable populations were more successful when they were conducted through reorganised social institutions and improved communication, e.g. in Mozambique (Osbahr et al., 2008). Improved communication and education could be facilitated by wider use of new information and communication technologies (Peters et al., 2015). Investments into education were associated with higher adoption of soil conservation measures, e.g. in Tanzania (Tenge et al., 2004). Bryan et al. (2009) found that access to information was the prominent facilitator of climate change adaptation in Ethiopia. However, resource constraints of agricultural services, and disconnects between agricultural policy and climate policy can hinder the dissemination of climate smart agricultural technologies (Morton, 2017). Lack of knowledge was also found to be a significant barrier to implementation of soil rehabilitation programmes in the Mediterranean region (Reichardt, 2010). Agricultural services will be able to facilitate SLM best when they also serve as platforms for sharing indigenous and local knowledge and farmer innovations (Mapfumo et al., 2016). Participatory research initiatives conducted jointly with farmers have higher chances of resulting in technology adoption (Bonney et al., 2016; Rusike et al., 2006; Vente et al., 2016). Moreover, rural advisory services are often more successful in disseminating technological innovations when they adopt commodity/value chain approaches, remain open to engagement in input supply, make use of new opportunities presented by information and communication technologies (ICTs), facilitate mutual learning between multiple stakeholders (Morton, 2017), and organise science and SLM information in a location-specific manner for use in education and extension (Bestelmeyer et al., 2017).

Strengthening land tenure security. Strengthening land tenure security is a major factor contributing to the adoption of soil conservation measures in croplands (*high confidence*) (Bambio and Bouayad Agha, 2018; Higgins et al., 2018; Holden and Ghebru, 2016; Paltasingh, 2018; Rao et al., 2016; Robinson et al., 2018) , thus, contributing to climate change adaptation and mitigation. Moreover, land tenure security can lead to more investment in trees (Deininger and Jin, 2006; Etongo et al., 2015). Land tenure recognition policies were found to lead to higher agricultural productivity and incomes, although with inter-regional variations, requiring an improved understanding of overlapping formal and informal land tenure rights (Lawry et al., 2017). For example, secure land tenure increased investments into SLM practices in Ghana, however, without affecting farm productivity (Abdulai et al., 2011). Secure land tenure, especially for communally managed lands, helps reduce arbitrary appropriations of land for large scale commercial farms (Aha and Ayitey, 2017; Baumgartner, 2017; Dell'Angelo et al., 2017). In contrast, privatisation of rangeland tenures in Botswana and Kenya led to the loss of communal grazing lands and actually increased rangeland degradation (Basupi et al., 2017; Kihiu, 2016) as pastoralists needed to graze livestock on now smaller communal pastures. Since food insecurity in drylands is strongly affected by climate risks, there is *robust evidence and high agreement* that resilience to climate risks is higher with flexible tenure for allowing mobility for pastoralist communities, and not fragmenting their areas of movement (Behnke, 1994; Holden and Ghebru, 2016; Liao et al., 2017; Turner et al., 2016; Wario et al., 2016). More research is needed on the optimal tenure mix, including low-cost land certification, redistribution reforms, market-assisted reforms and gender-responsive reforms, as well as collective forms of land tenure such as communal land tenure and cooperative land tenure (see 7.6.5 for a broader discussion of land tenure security under climate change).

Payment for ecosystem services (PES) provide incentives for land restoration and SLM (*medium confidence*) (Lambin et al., 2014; Li et al., 2018; Reed et al., 2015; Schiappacasse et al., 2012). Several studies illustrate that social cost of desertification are larger than its private cost (Costanza et al., 2014; Nkonya et al., 2016a). Therefore, although SLM can generate public goods in the form of provisioning ecosystem services, individual land custodians underinvest in SLM as they are unable to reap these benefits fully. Payment for ecosystem services provides a mechanism through which some of these benefits can be transferred to land users, thereby stimulating further investment in SLM. The effectiveness of PES schemes depends on land tenure security and appropriate design taking into account specific local conditions (Börner et al., 2017). However, PES has not worked well in countries with fragile institutions (Karsenty and Ongolo, 2012). Equity and justice in distributing the payments for ecosystem services were found to be key for the success of the PES programmes in Yunnan, China (He and Sikor, 2015). Yet, when reviewing the performance of PES programmes in the tropics, Calvet-Mir et al. (2015), found that they are generally effective in terms of environmental outcomes, despite being sometimes unfair in terms of payment distribution. It is suggested that the implementation of PES will be improved through decentralised approaches giving local communities a larger role in the decision making process (He and Lang, 2015).

Empowering local communities for decentralised natural resource management. Local institutions often play a vital role in implementing SLM initiatives and climate change adaptation (*high confidence*) (Gibson et al., 2005; Smucker et al., 2015). Pastoralists involved in community-based natural resource management in Mongolia had greater capacity to adapt to extreme winter frosts resulting in less damage to their livestock (Fernandez-Gimenez et al., 2015). Decreasing the power and role of traditional community institutions, due to top-down public policies, resulted in lower success rates in community-based programmes focused on rangeland management in Dirre, Ethiopia (Abdu and Robinson, 2017). Decentralised governance was found to lead to improved management in forested landscapes (Dressler et al., 2010; Ostrom and Nagendra, 2006). However, there are also cases when local elites were placed in control, decentralised natural resource management negatively impacted the livelihoods of the poorer and marginalised community members due to reduced access to natural resources (Andersson and Ostrom, 2008; Cullman, 2015; Dressler et al., 2010). The success of decentralised natural resource management initiatives depends on increased participation and empowerment of diverse set of community members, not only local leaders and elites, in the design and management of local resource management institutions (Kadirbeyoglu and Özertan, 2015; Umutoni et al., 2016), while considering the interactions between actors and institutions at different levels of governance (Andersson and Ostrom, 2008; Carlisle and Gruby, 2017; McCord et al., 2017). An example of such programmes where local communities played a major role in land restoration and rehabilitation activities is the cooperative project on "The National Afforestation and Erosion Control Mobilization Action Plan" in Turkey, initiated by the Turkish Ministry of Agriculture and Forestry (Çalişkan and Boydak, 2017), with the investment of USD 1.8 billion between 2008 and 2012. The project mobilised local communities in cooperation with public institutions, municipalities, and non-governmental organisations, to implement afforestation, rehabilitation and erosion control measures, resulting in the afforestation and reforestation of 1.5 M ha (Yurtoglu, 2015). Moreover, some 1.75 M ha of degraded forest and 37880 ha of degraded rangelands were rehabilitated. Finally, the project provided employment opportunities for 300,000 rural residents for six months every year, combining land restoration and rehabilitation activities with measures to promote socio-economic development in rural areas (Çalişkan and Boydak, 2017).

Investing in research and development. Desertification has received substantial research attention over recent decades (Turner et al., 2007). There is also a growing research interest on climate change adaptation and mitigation interventions that help address desertification (Grainger, 2009). Agricultural research on SLM practices has generated a significant number of new innovations and technologies that increase crop yields without degrading the land, while contributing to climate change adaptation and mitigation (3.6.1). There is *robust evidence* that such technologies help improve the food security of smallholder dryland farming households (Harris and Orr, 2014, 6.3.5). Strengthening research on desertification is of high importance not only to meet SDGs but also effectively manage ecosystems based on solid scientific knowledge. More investment in research institutes and training the younger generation of researchers is needed for addressing the combined challenges of desertification and climate change (Akhtar-Schuster et al., 2011; Verstraete et al., 2011). This includes improved knowledge management systems that allow stakeholders to work in a coordinated manner by enhancing timely, targeted and contextualised information sharing (Chasek et al., 2011). Knowledge and flow of knowledge on desertification is currently highly fragmented, constraining effectiveness of those engaged in assessing and monitoring the phenomenon at various levels (Reed et al., 2011). Improved knowledge and data exchange and sharing increase the effectiveness of efforts to address desertification (*high confidence*).

Developing modern renewable energy sources. Transitioning to renewable energy resources contributes to reducing desertification by lowering reliance on traditional biomass in dryland regions (*medium confidence*). Populations in most developing countries continue to rely on traditional biomass, including fuelwood, crop straws and livestock manure, for a major share of their energy needs, with the highest dependence in Sub-Saharan Africa (Amugune et al., 2017; IEA, 2013). Use of biomass for energy, mostly fuelwood (especially as charcoal), was associated with deforestation in some dryland areas (Iiyama et al., 2014; Mekuria et al., 2018; Neufeldt et al., 2015; Zulu, 2010), while in some other areas there was no link between fuelwood collection and deforestation (Simon and Peterson, 2018; Swemmer et al., 2018; Twine and Holdo, 2016). Moreover, the use of traditional biomass as a source of energy was found to have negative health effects through indoor air pollution (de la Sota et al., 2018; Lim and Seow, 2012), while also being associated with lower female labor force participation (Burke and Dundas, 2015). Jiang et al., (2014) indicated that providing improved access to alternative energy sources such as solar energy and biogas could help reduce the use of fuelwood in south-western China, thus alleviating the spread of rocky desertification. The conversion of degraded lands into cultivation of biofuel crops will affect soil C dynamics (Albanito et al., 2016; Nair et al., 2011; Cross-Chapter Box 7: Bioenergy and BECCS, Chapter 6). The use of biogas slurry as soil amendment or fertiliser can increase soil C (Galvez et al., 2012; Negash et al., 2017). Large-scale installation of wind and solar farms in the Sahara desert was projected to create a positive climate feedback through increased surface friction and reduced albedo, doubling precipitation over the neighbouring Sahel region with resulting increases in vegetation (Li et al., 2018). Transition to renewable energy sources in high-income countries in dryland areas primarily contributes to reducing greenhouse gas emissions and mitigating climate change, with some other co-benefits such as diversification of energy sources (Bang, 2010), while the impacts on desertification are less evident. The use of renewable energy has been proposed as an important mitigation option in dryland areas as well (El-Fadel et al., 2003). Transitions to renewable energy are being promoted by governments across drylands (Cancino-Solórzano et al., 2016; Hong et al., 2013; Sen and Ganguly, 2017) including in fossil-fuel rich countries (Farnoosh et al., 2014; Dehkordi et al., 2017; Stambouli et al., 2012; Vidadili et al., 2017), despite important social, political and technical barriers to expanding renewable energy production (Afsharzade et al., 2016; Baker et al., 2014; Elum and Momodu, 2017; Karatayev et al., 2016). Improving the social awareness about the benefits of transitioning to renewable energy resources and access to hydro-energy, solar and wind energy contributes to their improved adoption (Aliyu et al., 2017; Katikiro, 2016).

Developing and strengthening climate services relevant for desertification. Climate services provide climate, drought and desertification-related information in a way that assists decision making by individuals and organisations. For monitoring desertification, integration of biogeophysical (climate, soil, ecological factors, biodiversity) and socio-economic aspects (use of natural resources by local population) provides a basis for better vulnerability prediction and assessment (OSS, 2012; Vogt et al., 2011). Examples of relevant services include: drought monitoring and early warning systems often implemented by national climate and meteorological services but also encompassing regional and global systems (Pozzi et al., 2013); and the Sand and Dust Storm Warning Advisory and Assessment System (SDS-WAS), created by WMO in 2007, in partnership with the World Health Organization (WHO) and the United Nations Environment Program (UNEP). Currently, there is also a lack of ecological monitoring in arid and semi-arid regions to study surface winds, dust and sandstorms, and their impacts on ecosystems and human health (Bergametti et al., 2018; Marticorena et al., 2010). Reliable and timely climate services, relevant to desertification, can aid the development of appropriate adaptation and mitigation options reducing the impact of desertification under changing climate on human and natural systems (*high confidence*) (Beegum et al., 2016; Beegum et al., 2018; Cornet, 2012; Haase et al., 2018; Sergeant, Moynahan, & Johnson, 2012).

3.6.3.2. Policy Responses Supporting Economic Diversification

Despite policy responses for combating desertification, climate change, growing food demands, as well as the need to reduce poverty and strengthen food security, will put strong pressures on the land (Cherlet et al., 2018; 6.1.4; 7.2.2). Sustainable development of drylands and their resilience to combined challenges of desertification and climate change will thus also depend on the ability of governments to promote policies for economic diversification within agriculture and in non-agricultural sectors in order make dryland areas less vulnerable to desertification and climate change.

Investing into irrigation. Investments into expanding irrigation in dryland areas can help increase the resilience of agricultural production to climate change, improve labour productivity and boost production and income revenue from agriculture and livestock sectors (Geerts and Raes, 2009; Olayide et al., 2016; Oweis and Hachum, 2006). This is particularly true for Sub-Saharan Africa, where currently only 6% of the cultivated areas are irrigated (Nkonya et al., 2016b). While renewable groundwater resources could help increase the share of irrigated land to 20.5%-48.6% of croplands in the region (Altchenko and Villholth, 2015). On the other hand, over-extraction of groundwaters, mainly for irrigating crops, is becoming an important environmental problem in many dryland areas (Cherlet et al., 2018), requiring careful design and planning of irrigation expansion schemes and use of water efficient irrigation methods (Bjornlund, van Rooyen, and Stirzaker, 2017; Woodhouse et al., 2017). For example, in Saudi Arabia, improving the efficiency of water management, e.g. through the development of aquifers, water recycling and rainwater harvesting is part of policy actions to combat desertification (Bazza, et al., 2018; Kingdom of Saudi Arabia, 2016). The expansion of irrigation to riverine areas, crucial for dry season grazing of livestock, needs to consider the loss of income from pastoral activities, which is not always lower than income from irrigated crop production (Behnke and Kerven, 2013). Irrigation development could be combined with the deployment of clean energy technologies in economically viable ways (Chandel et al., 2015). For example, solar-powered drip irrigation was found to increase household agricultural incomes in Benin (Burney et al., 2010). The sustainability of irrigation schemes based on solar-powered extraction of groundwaters depends on measures to avoid over-abstraction of groundwater resources and associated negative environmental impacts (Closas and Rap, 2017).

Expanding agricultural commercialisation. Faster poverty rate reduction and economic growth enhancement is realised when countries transition into the production of non-staple, high value commodities and manage to build a robust agro-industry sector (Barrett et al., 2017). Ogutu and Qaim (2019) found that agricultural commercialisation increased incomes and decreased multidimensional poverty in Kenya. Similar findings were earlier reported by Muriithi and Matz (2015) for commercialisation of vegetables in Kenya. Commercialisation of rice production was found to have increased smallholder welfare in Nigeria (Awotide et al., 2016). Agricultural commercialisation contributed to improved household food security in Malawi, Tanzania and Uganda (Carletto et al., 2017). However, such a transition did not improve farmers' livelihoods in all cases (Reardon et al., 2009). High value cash crop/animal production can be bolstered by wide-scale use of technologies, for example, mechanisation, application of inorganic fertilisers, crop protection and animal health products. Market oriented crop/animal production facilitates social and economic progress with labour increasingly shifting out of agriculture into non-agricultural sectors (Cour, 2001). Modernised farming, improved access to inputs, credit and technologies enhances competitiveness in local and international markets (Reardon et al., 2009).

Facilitating structural transformations in rural economies implies that the development of non-agricultural sectors encourages the movement of labour from land-based livelihoods, vulnerable to desertification and climate change, to non-agricultural activities (Haggblade et al., 2010). The movement of labour from agriculture to non-agricultural sectors is determined by relative labour productivities in these sectors (Shiferaw and Djido, 2016). Given already high underemployment in the farm sector, increasing labour productivity in the non-farm sector was found as the main driver of labour movements from farm sector to non-farm sector (Shiferaw and Djido, 2016). More investments into education can facilitate this process (Headey et al., 2014). However, in some contexts, such as pastoralist communities in Xinjiang, China, income diversification was not found to improve the welfare of pastoral households (Liao et al., 2015). Economic transformations also occur through urbanisation, involving the shift of labour from rural areas into gainful employment in urban areas (Jedwab and Vollrath, 2015). The larger share of world population will be living in urban centres in the 21st century and this will require innovative means of agricultural production with minimum ecological footprint and less dependence on fossil fuels (Revi and Rosenzweig, 2013), while addressing the demand of cities (see 4.9.1 for discussion on urban green infrastructure). Although there is some evidence of urbanisation leading to the loss of indigenous and local ecological knowledge, however, indigenous and local knowledge systems are constantly evolving, and are also getting integrated into urban environments (Júnior et al., 2016; Reyes-García et al., 2013; van Andel and Carvalheiro, 2013). Urban areas are attracting an increasing number of rural residents across the developing world (Angel et al., 2011; Cour, 2001; Dahiya, 2012). Urban development contributes to expedited agricultural commercialisation by providing market outlet for cash and high value crop and livestock products. At the same time, urbanisation also poses numerous challenges in the form of rapid urban sprawl and pressures on infrastructure and public services, unemployment and associated social risks, which have considerable implications on climate change adaptive capacities (Bulkeley, 2013; Garschagen and Romero-Lankao, 2015).

Cross-Chapter Box 5: Policy Responses to Drought

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Drought is a highly complex natural hazard (for floods, see Box 7.2). It is difficult to precisely identify its start and end. It is usually slow and gradual (Wilhite and Pulwarty, 2017), but sometimes can evolve rapidly (Ford and Labosier, 2017; Mo and Lettenmaier, 2015). It is context-dependent, but its impacts are diffuse, both direct and indirect, short-term and long-term (Few and Tebboth, 2018; Wilhite and Pulwarty, 2017). Following the Synthesis Report (SYR) of the IPCC Fifth Assessment Report (AR5), drought is defined here as "a period of abnormally dry weather long enough to cause a serious hydrological imbalance" (Mach et al., 2014). Although drought is considered abnormal relative to the water availability under the mean climatic characteristics, it is also a recurrent element of any climate, not only in drylands, but also in humid areas (Cook et al., 2014b; Seneviratne and Ciais, 2017; Spinoni et al., 2019; Türkeş, 1999; Wilhite et al., 2014). Climate change is projected to increase the intensity or frequency of droughts in some regions across the world (for detailed assessment see 2.2, and IPCC Special Report on Global Warming of 1.5°C (Hoegh-Guldberg et al., 2018, Chapter 3)). Droughts often amplify the effects of unsustainable land management practices, especially in drylands, leading to land degradation (Cook et al., 2009; Hornbeck, 2012). Especially in the context of climate change, the recurrent nature of droughts requires pro-actively planned policy instruments both to be well-prepared to respond to droughts when they occur and also undertake ex ante actions to mitigate their impacts by strengthening the societal resilience against droughts (Gerber and Mirzabaev, 2017).

Droughts are among the costliest of natural hazards (*robust evidence*, *high agreement*). According to the International Disaster Database (EM-DAT), droughts affected more than 1.1 billion people between 1994-2013, with the recorded global economic damage of USD 787 billion (CRED, 2015), corresponding to an average of USD 41.4 billion per year. Drought losses in the agricultural sector alone in the developing countries were estimated to equal USD 29 billion between 2005-2015 (FAO, 2018). Usually, these estimates capture only direct and on-site costs of droughts. However, droughts have also wide-ranging indirect and off-site impacts, which are seldom quantified. These indirect impacts are both biophysical and socio-economic, with the poor households and communities being particularly exposed to them (Winsemius et al., 2018). Droughts affect not only water quantity, but also water quality (Mosley, 2014). The costs of these water quality impacts are yet to be adequately quantified. Socio-economic indirect impacts of droughts are related to food insecurity, poverty, lowered health and displacement (Gray and Mueller, 2012; Johnstone and Mazo, 2011; Linke et al., 2015; Lohmann and Lechtenfeld, 2015; Maystadt and Ecker, 2014; Yusa et al., 2015 see also 3.4.2.9, Box 5.5), which are difficult to quantify comprehensively. Research is required for developing methodologies that could allow for more comprehensive assessment of these indirect drought costs. Such methodologies require the collection of highly granular data, which is currently lacking in many countries due to high costs of data collection. However, the opportunities provided by remotely sensed data and novel analytical methods based on big data and artificial intelligence, including use of citizen science for data collection, could help in reducing these gaps.

There are three broad (and sometimes overlapping) policy approaches for responding to droughts (also see 7.4.8). These approaches are often pursued simultaneously by many governments. Firstly, responding to drought when it occurs by providing direct drought relief, known as crisis management. Crisis management is also the costliest among policy approaches to droughts because it often incentivises the continuation of activities vulnerable to droughts (Botterill and Hayes, 2012; Gerber and Mirzabaev, 2017).

The second approach involves development of drought preparedness plans, which coordinate the policies for providing relief measures when droughts occur. For example, combining resources to respond to droughts at regional level in Sub-Saharan Africa was found more cost-effective than separate individual country drought relief funding (Clarke and Hill, 2013). Effective drought preparedness plans require well-coordinated and integrated government actions - a key lesson learnt from 2015-2017 drought response in Cape Town, South Africa (Visser, 2018). Reliable, relevant and timely climate and weather information helps respond to droughts appropriately (Sivakumar and Ndiang'ui, 2007). Improved knowledge and integration of weather and climate information can be achieved by strengthening drought early warning systems at different scales (Verbist et al., 2016). Every USD invested into strengthening hydro-meteorological and early warning services in developing countries was found to yield between USD 4 to 35 (Hallegatte, 2012). Improved access and coverage by drought insurance, including index insurance, can help alleviate the impacts of droughts on livelihoods (Guerrero-Baena et al., 2019; Kath et al., 2019; Osgood et al., 2018; Ruiz et al., 2015; Tadesse et al., 2015).

The third category of responses to droughts involves drought risk mitigation. Drought risk mitigation is a set of proactive measures, policies and management activities aimed at reducing the future impacts of droughts (Vicente-Serrano et al., 2012). For example, policies aimed at improving water use efficiency in different sectors of the economy, especially in agriculture and industry, or public advocacy campaigns raising societal awareness and bringing about behavioural change to reduce wasteful water consumption in the residential sector are among such drought risk mitigation policies (Tsakiris, 2017). Public outreach and monitoring of communicable diseases, air and water quality were found useful for reducing health impacts of droughts (Yusa et al., 2015). The evidence from household responses to drought in Cape Town, South Africa, between 2015-2017, suggests that media coverage and social media could play a decisive role in changing water consumption behaviour, even more so than official water consumption restrictions (Booysen et al., 2019). Drought risk mitigation approaches are less costly than providing drought relief after the occurrence of droughts. To illustrate, Harou et al. (2010) found that establishment of water markets in California considerably reduced drought costs. Application of water saving technologies reduced drought costs in Iran by USD 282 million (Salami et al., 2009). Booker et al. (2005) calculated that interregional trade in water could reduce drought costs by 20–30% in the Rio Grande basin, USA. Increasing rainfall variability under climate change can make the forms of index insurance based on rainfall less efficient (Kath et al., 2019). A number of diverse water property instruments, including instruments allowing water transfer, together with the technological and institutional ability to adjust water allocation, can improve timely adjustment to droughts (Hurlbert, 2018). Supply-side water management providing for proportionate reductions in water delivery prevents the important climate change adaptation option of managing water according to need or demand (Hurlbert and Mussetta, 2016). Exclusive use of a water market to govern water allocation similarly prevents the recognition of the human right to water at times of drought (Hurlbert, 2018). Policies aiming to secure land tenure, to expand access to markets, agricultural advisory services and effective climate services, as well as to create off-farm employment opportunities can facilitate the adoption of drought risk mitigation practices (Alam, 2015; Kusunose and Lybbert, 2014), increasing the resilience to climate change (3.6.3), while also contributing to SLM (3.6.3, 4.8.1, Table 5.7).

The excessive burden of drought relief funding on public budgets is already leading to a paradigm shift towards proactive drought risk mitigation instead of reactive drought relief measures (Verner et al., 2018; Wilhite, 2016). Climate change will reinforce the need for such proactive drought risk mitigation approaches. Policies for drought risk mitigation that are already needed now will be even more relevant under higher warming levels (Jerneck and Olsson, 2008; McLeman, 2013; Wilhite et al., 2014). Overall, there is *high confidence* that responding to droughts through ex post drought relief measures is less efficient compared to ex ante investments into drought risk mitigation, particularly under climate change.

3.6.4. Limits to Adaptation, Maladaptation, and Barriers for Mitigation

Chapter 16 in the Fifth Assessment Report of IPCC (Klein et al., 2015) discusses the existence of soft and hard limits to adaptation, highlighting that values and perspectives of involved agents are relevant to identify limits (4.8.5.1, 7.4.9). In that sense, adaptation limits vary from place to place and are difficult to generalise (Barnett et al., 2015; Dow et al., 2013; Klein et al., 2015). Currently, there is a lack of knowledge on adaptation limits and potential maladaptation to combined effects of climate change and desertification (see 4.8.6 in Chapter 4 for discussion on resilience, thresholds, and irreversible land degradation also relevant for desertification). However, the potential for residual risks and maladaptive outcomes is high (*high confidence*). Some examples of residual risks are illustrated below (those risks which remain after adaptation efforts were taken, irrespective whether they are tolerable or not, tolerability being a subjective concept). Although SLM measures can help lessen the effects of droughts, they cannot fully prevent water stress in crops and resulting lower yields (Eekhout and de Vente, 2019). Moreover, although in many cases SLM measures can help reduce and reverse desertification, there would be still short-term losses in land productivity. Irreversible forms of land degradation (e.g. loss of topsoil, severe gully erosion) can lead to the complete loss of land productivity. Even when solutions are available, their costs could be prohibitive presenting the limits to adaptation (Dixon et al., 2013). If warming in dryland areas surpasses human thermal physiological thresholds (Klein et al., 2015; Waha et al., 2013), adaptation could eventually fail (Kamali et al., 2018). Catastrophic shifts in ecosystem functions and services, e.g. coastal erosion (4.9.8; Chen et al., 2015; Schneider and Kéfi, 2016), and economic factors can also result in adaptation failure (Evans et al., 2015). Despite the availability of numerous options that contribute to combating desertification, climate change adaptation and mitigation, there are also chances of maladaptive actions (*medium confidence*) (Glossary). Some activities favouring agricultural intensification in dryland areas can become maladaptive due to their negative impacts on the environment (*medium confidence*). Agricultural expansion to meet food demands can come through deforestation and consequent diminution of C sinks (Godfray and Garnett, 2014; Stringer et al., 2012). Agricultural insurance programs encouraging higher agricultural productivity and measures for agricultural intensification can result in detrimental environmental outcomes in some settings (Guodaar et al., 2019; Müller et al., 2017; Table 6.12). Development of more drought-tolerant crop varieties is considered as a strategy for adaptation to shortening rainy season, but this can also lead to a loss of local varieties (Al Hamndou and Requier-Desjardins, 2008). Livelihood diversification to collecting and selling firewood and charcoal production can exacerbate deforestation (Antwi-Agyei et al., 2018). Avoiding maladaptive outcomes can often contribute both to reducing the risks from climate change and combating desertification (Antwi-Agyei et al., 2018). Avoiding, reducing and reversing desertification would enhance soil fertility, increase C storage in soils and biomass, thus reducing C emissions from soils to the atmosphere (3.7.2; Cross-Chapter Box 2: Implications of large-scale conversion from non-forest to forest land, Chapter 1). In specific locations, there may be barriers for some of these activities. For example, afforestation and reforestation programs can contribute to reducing sand storms and increasing C sinks in dryland regions (3.6.1, 3.7.2) (Chu et al., 2019). However, implementing agroforestry measures in arid locations can be constrained by lack of water (Apuri et al., 2018), leading to a trade-off between soil C sequestration and other water uses (Cao et al., 2018).

3.7. Hotspots and Case Studies

The challenges of desertification and climate change in dryland areas across the world often have very location-specific characteristics. The five case studies in this section present rich experiences and lessons learnt on: 1) soil erosion, 2) afforestation and reforestation through “green walls”, 3) invasive plant species, 4) oases in hyper-arid areas, and 5) integrated watershed management. Although it is impossible to cover all hotspots of desertification and on the ground actions from all dryland areas, these case studies present a more focused assessment of these five issues that emerged as salient in the group discussions and several rounds of review of this chapter. The choice of these case studies was also motivated by the desire to capture a wide diversity of dryland settings.

3.7.1. Climate Change and Soil Erosion

3.7.1.1. Soil Erosion under Changing Climate in Drylands

Soil erosion is a major form of desertification occurring in varying degrees in all dryland areas across the world (3.2), with negative effects on dryland ecosystems (3.4). Climate change is projected to increase soil erosion potential in some dryland areas through more frequent heavy rainfall events and rainfall variability than currently (see Section 3.5.2 for more detailed assessment, (Achite and Ouillon, 2007; Megnounif and Ghenim, 2016; Vachtman et al., 2013; Zhang and Nearing, 2005). There are numerous soil conservation measures that can help reduce soil erosion (3.6.1). Such soil management measures include afforestation and reforestation activities, rehabilitation of degraded forests, erosion control measures, prevention of overgrazing, diversification of crop rotations, and improvement in irrigation techniques, especially in sloping areas (Anache et al., 2018; ÇEMGM, 2017; Li and Fang, 2016; Poesen, 2018; Ziadat and Taimeh, 2013). Effective measures for soil conservation can also use spatial patterns of plant cover to reduce sediment connectivity, and the relationships between hillslopes and sediment transfer in eroded channels (García-Ruiz et al., 2017). The following three examples present lessons learnt from the soil erosion problems and measures to address them in different settings of Chile, Turkey and the Central Asian countries.

3.7.1.2. No-Till Practices for Reducing Soil Erosion in Central Chile

Soil erosion by water is an important problem in Chile. National assessments conducted in 1979, which examined 46% of the continental surface of the country, concluded that very high levels of soil erosion affected 36% of the territory. The degree of soil erosion increases from south to north. The leading locations in Chile are the region of Coquimbo with 84% of eroded soils (Lat 29°S, Semiarid climate), the region of Valparaíso with 57% of eroded souls (Lat 33° S, Mediterranean climate) and the region of O’Higging with 37% of eroded soils (Lat 34°S Mediterranean climate). The most important drivers of soil erosion are soil, slope, climate erosivity (i.e., precipitation, intensity, duration and frequency) due to a highly concentrated rainy season, and vegetation structure and cover. In the region of Coquimbo, goat and sheep overgrazing have aggravated the situation (CIREN, 2010).

Erosion rates reach up to 100 t ha^{-1} annually, having increased substantially over the last 50 years (Ellies, 2000). About 10.4% of central Chile exhibits high erosion rates (greater than 1.1 t ha^{-1} annually) (Bonilla et al., 2010).

Over the last few decades there has been an increasing interest in the development of no-till (also called zero tillage) technologies to minimise soil disturbance, reduce the combustion of fossil fuels and increase soil organic matter. No-till in conjunction with the adoption of strategic cover crops have positively impacted soil biology with increases in soil organic matter. Early evaluations by Crovetto, (1998) showed that no-till application (after seven years) had doubled the biological activity indicators compared to traditional farming and even surpassed those found in pasture (grown for the previous 15 years). Besides erosion control, additional benefits are an increase of water holding capacity and reduction in bulk density. Currently, the above no-till farm experiment has lasted for 40 years and continues to report benefits to soil health and sustainable production (Reicosky and Crovetto, 2014). The influence of this iconic farm has resulted in the adoption of soil conservation practices and specially no-till in dryland areas of the Mediterranean climate region of central Chile (Martínez et al., 2011). Currently, it has been estimated that the area under no-till farming in Chile varies between 0.13 and 0.2 M ha (Acevedo and Silva, 2003).

3.7.1.3. Combating Wind Erosion and Deflation in Turkey: The Greening Desert of Karapınar

In Turkey, the amount of sediment recently released through erosion into seas was estimated to be 168 Mt yr^{-1}, which is considerably lower than the 500 Mt yr^{-1} that was estimated to be lost in the 1970s (ÇEMGM, 2017). The decrease in erosion rates is attributed to an increase in spatial extent of forests, rehabilitation of degraded forests, erosion control, prevention of overgrazing, and improvement in irrigation technologies. Soil conservation measures conducted in the Karapınar district, Turkey, exemplify these activities. The district is characterised by a semi-arid climate and annual average precipitation of 250–300 mm (Türkeş, 2003; Türkeş and Tatlı, 2011). In areas where vegetation was overgrazed or inappropriately tilled, the surface soil horizon was removed through erosion processes resulting in the creation of large drifting dunes that threatened settlements around Karapınar (Groneman, 1968). Such dune movement had begun to affect the Karapınar settlement in 1956 (Kantarcı et al., 2011). Consequently, by early 1960s, Karapınar town and nearby villages were confronted with the danger of abandonment due to out-migration in early 1960s (Figure 3.11-1). The reasons for increasing wind erosion in the Karapınar district can be summarised as follows: sandy material was mobilised following drying of the lake; hot and semi-arid climate conditions; overgrazing and use of pasture plants for fuel; excessive tillage; and strong prevailing winds.

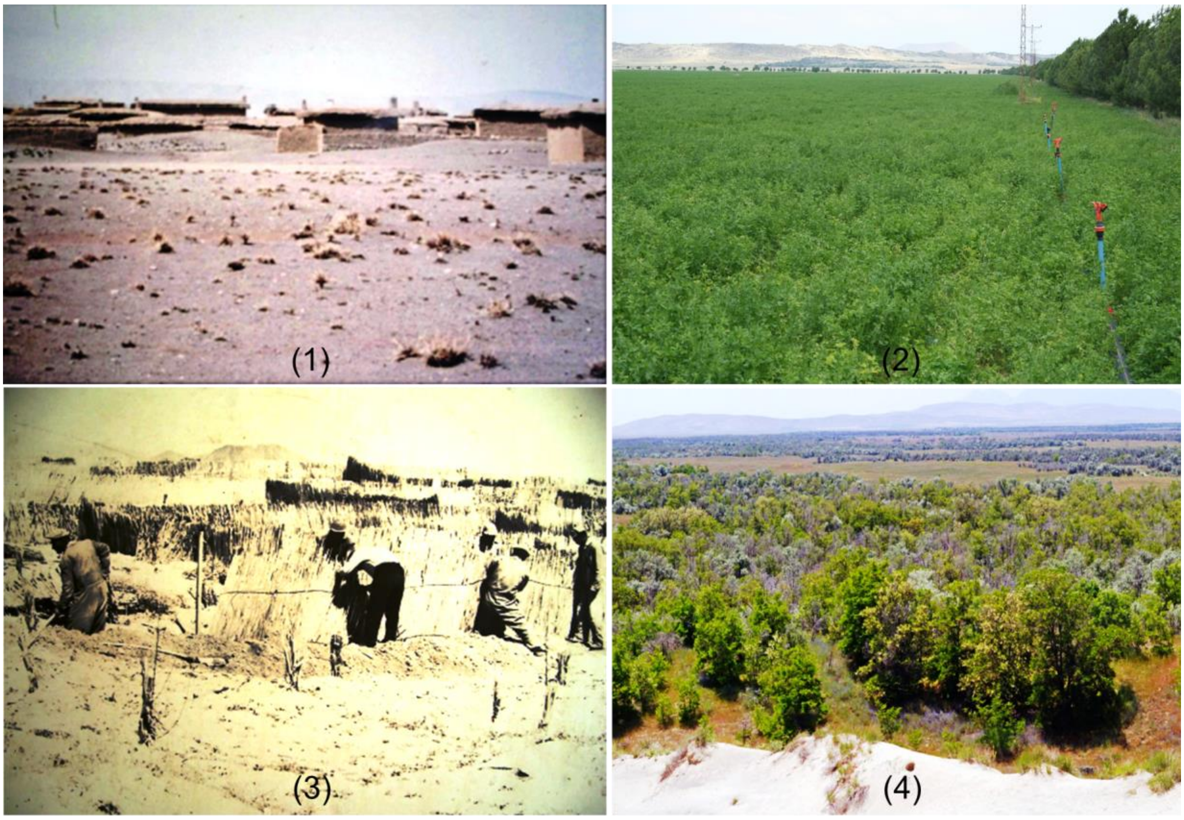

Figure 3.11 (1) A general view of a nearby village of Karapınar town in early 1960s (Çarkaci, 1999). (2) A view of the Karapınar wind erosion area in 2013 (Photograph: Murat Türkeş, 17.06.2013). (3) Construction of Cane Screens in early 1960s in order to decrease speed of the wind and prevent movement of the sand accumulations and dunes, which was one of the physical measures during the prevention and mitigation period (Çarkaci, 1999). (4) A view of mix vegetation in most of the Karapınar wind erosion area in 2013, the main tree species of which were selected for afforestation with respect to their resistance to the arid continental climate conditions along with a warm/hot temperature regime over the district (Photograph: Murat Türkeş, 17.06.2013)

Restoration and mitigation strategies were initiated in 1959 and today, 4300 ha of land have been restored (Akay and Yildirim, 2010) (Figure 3.11-2), using specific measures: (1) Physical measures: construction of cane screens to decrease wind speed and prevent sand movement (Figure 3.11); (2) Restoration of cover: increasing grass cover between screens using seeds collected from local pastures or the cultivation of rye (*Secale* sp.) and wheat grass (*Agropyron elongatum*) that are known to grow in arid and hot conditions; (3) Afforestation: saplings obtained from nursery gardens were planted and grown between these screens. Main tree species selected were oleaster (*Eleagnus* sp.), acacia (*Robinia pseudeaccacia*), ash (*Fraxinus* sp.), elm (*Ulmus* sp.) and maple (*Acer* sp.) (Figure 3.11-4). Economic growth occurred after controlling erosion and new tree nurseries have been established with modern irrigation. Potential negative consequences through the excessive use of water can be mitigated through engagement with local stakeholders and transdisciplinary learning processes, as well as by restoring the traditional land uses in the semi-arid Konya closed basin (Akça et al., 2016).

3.7.1.4. Soil Erosion in Central Asia under Changing Climate

Soil erosion is widely acknowledged to be a major form of degradation of Central Asian drylands, affecting considerable share of croplands and rangelands. However, up-to-date information on the actual extent of eroded soils at the regional or country level is not available. The estimates compiled by Pender et al. (2009), based on the Central Asian Countries Initiative for Land Management

(CACILM), indicate that about 0.8 M ha of the irrigated croplands were subject to high degree of soil erosion in Uzbekistan. In Turkmenistan, soil erosion was indicated to be occurring in about 0.7 M ha of irrigated land. In Kyrgyzstan, out of 1 M ha irrigated land in the foothill zones, 0.76 M ha were subject to soil erosion by water, leading to losses in crop yields of 20-60% in these eroded soils. About 0.65 M ha of arable land were prone to soil erosion by wind (Mavlyanova et al., 2017). Soil erosion is widespread in rainfed and irrigated areas in Kazakhstan (Saparov, 2014). About 5 M ha of rainfed croplands were subject to high levels of soil erosion (Pender et al., 2009). Soil erosion by water was indicated to be a major concern in sloping areas in Tajikistan (Pender et al., 2009).

The major causes of soil erosion in Central Asia are related to human factors, primarily excessive water use in irrigated areas (Gupta et al., 2009), deep ploughing and lack of maintenance of vegetative cover in rainfed areas (Suleimenov et al., 2014), and overgrazing in rangelands (Mirzabaev et al., 2016). Lack of good maintenance of watering infrastructure for migratory livestock grazing and fragmentation of livestock herds led to overgrazing near villages, increasing the soil erosion by wind (Alimaev et al., 2008). Overgrazing in the rangeland areas of the region (e.g. particularly in Kyzylkum) contributes to dust storms, coming primarily from Ustyurt Plateau, desertified areas of Amudarya and Syrdarya rivers' deltas, dried seabed of the Aral Sea (now called Aralkum), and the Caspian Sea (Issanova and Abuduwaili, 2017; Xi and Sokolik, 2015). Xi and Sokolik (2015) estimated that total dust emissions in Central Asia were 255.6 Mt in 2001, representing 10-17% of the global total.

Central Asia is one of the regions highly exposed to climate change, with warming levels projected to be higher than the global mean (Hoegh-Guldberg et al., 2018), leading to more heat extremes (Reyer et al., 2017). There is no clear trend in precipitation extremes, with some potential for moderate rise in occurrence of droughts. The diminution of glaciers is projected to continue in the Pamir and Tian Shan mountain ranges, a major source of surface waters along with seasonal snowmelt. Glacier melting will increase the hazards from moraine-dammed glacial lakes and spring floods (Reyer et al., 2017). Increased intensity of spring floods creates favourable conditions for higher soil erosion by water especially in the sloping areas in Kyrgyzstan and Tajikistan. The continuation of some of the current unsustainable cropland and rangeland management practices may lead to elevated rates of soil erosion particularly in those parts of the region where climate change projections point to increases in floods (Kyrgyzstan, Tajikistan) or increases in droughts (Turkmenistan, Uzbekistan) (Hijioka et al., 2014). Increasing water use to compensate for higher evapotranspiration due to growing temperatures and heat waves could increase soil erosion by water in the irrigated zones, especially sloping areas and crop fields with uneven land levelling (Bekchanov et al., 2010). The desiccation of the Aral Sea resulted in hotter and drier regional microclimate, adding to the growing wind erosion in adjacent deltaic areas and deserts (Kust, 1999).

There are numerous sustainable land and water management practices available in the region for reducing soil erosion (Abdullaev et al., 2007; Gupta et al., 2009; Kust et al., 2014; Nurbekov et al., 2016). These include: improved land levelling and more efficient irrigation methods such as drip, sprinkler and alternate furrow irrigation (Gupta et al., 2009); conservation agriculture practices, including no-till methods and maintenance of crop residues as mulch in the rainfed and irrigated areas (Kienzler et al., 2012; Pulatov et al., 2012); rotational grazing; institutional arrangements for pooling livestock for long-distance mobile grazing; reconstruction of watering infrastructure along the livestock migratory routes (Han et al., 2016; Mirzabaev et al., 2016); afforesting degraded marginal lands (Djanibekov and Khamzina, 2016; Khamzina et al., 2009; Khamzina et al., 2016); integrated water resource management (Dukhovny et al., 2013; Kazbekov et al., 2009), planting salt and drought tolerant halophytic plants as windbreaks in sandy rangelands (Akinshina et al., 2016; Qadir et al., 2009; Toderich et al., 2009; Toderich et al., 2008), and potentially the dried seabed of the former Aral Sea (Breckle, 2013). The adoption of enabling policies, such as those discussed in Section 3.6.3, can facilitate the adoption of these sustainable land and water management practices in Central Asia (*high confidence*) (Aw-Hassan et al., 2016; Bekchanov et al., 2016; Bobojonov et al., 2013; Djanibekov et al., 2016; Hamidov et al., 2016; Mirzabaev et al., 2016).

3.7.2. Green Walls and Green Dams

This case study evaluates the experiences of measures and actions implemented to combat soil erosion, decrease dust storms, and to adapt to and mitigate climate change under the Green Wall and Green Dam programmes in East Asia (e.g., China) and Africa (e.g., Algeria, Sahara and the Sahel region). These measures have also been implemented in other countries, such as Mongolia (Do & Kang, 2014; Lin et al., 2009), Turkey (Yurtoglu, 2015; Çalişkan and Boydak, 2017) and Iran (Amiraslani and Dragovich, 2011), and are increasingly considered as part of many national and international initiatives to combat desertification (Goffner et al., 2019; Cross-Chapter Box 2, chapter 1). Afforestation and reforestation programs can contribute to reducing sand storms and increasing C sinks in dryland regions (*high confidence*). On the other hand, Green Wall and Green Dam programmes also decrease the albedo and hence increase the surface absorption of radiation, increasing the surface temperature. The net effect will largely depend on the balance between these and will vary from place to place depending on many factors.

3.7.2.1. The Experiences of Combating Desertification in China

Arid and semiarid areas of China, including north-eastern, northern and north-western regions, cover an area of more than 509 M ha, with annual rainfall of below 450 mm. Over the past several centuries, more than 60% of the areas in arid and semiarid regions were used as pastoral and agricultural lands. The coupled impacts of past climate change and human activity have caused desertification and dust storms to become a serious problem in the region (Xu et al., 2010). In 1958, the Chinese government recognised that desertification and dust storms jeopardised livelihoods of nearly 200 million people, and afforestation programmes for combating desertification have been initiated since 1978. China is committed to go beyond the Land Degradation Neutrality objective as indicated by the following programmes that have been implemented. The Chinese Government began the Three North's Forest Shelterbelt programme in Northeast China, North China, and Northwest China, with the goal to combat desertification and to control dust storms by improving forest cover in arid and semiarid regions. The project is implemented in three stages (1978–2000, 2001–2020, and 2021–2050). In addition, the Chinese government launched Beijing and Tianjin Sandstorm Source Treatment Project (2001–2010), Returning Farmlands to Forest Project (2003–present), Returning Grazing Land to Grassland Project (2003–present) to combat desertification, and for adaptation and mitigation of climate change (State Forestry Administration of China, 2015; Tao, 2014; Wang et al., 2013).

The results of the fifth period monitoring (2010–2014) showed: (1) Compared with 2009, the area of degraded land decreased by 12,120 km^2 over a five-year period; (2) In 2014, the average coverage of vegetation in the sand area was 18.33%, an increase of 0.7% compared with 17.63% in 2009, and the C sequestration increased by 8.5%; (3) Compared with 2009, the amount of wind erosion decreased by 33%, the average annual occurrence of sandstorms decreased by 20.3% in 2014; (4) As of 2014, 203,700 km^2 of degraded land were effectively managed, accounting for 38.4% of the 530,000 km^2 of manageable desertified land; (5) The restoration of degraded land has created an annual output of 53.63 M tonnes of fresh and dried fruits, accounting for 33.9% of the total national annual output of fresh and dried fruits (State Forestry Administration of China, 2015). This has become an important pillar for economic development and a high priority for peasants as a method to eradicate poverty (State Forestry Administration of China, 2015).

Stable investment mechanisms for combating desertification have been established along with tax relief policies and financial support policies for guiding the country in its fight against desertification. The investments in scientific and technological innovation for combating desertification have been improved, the technologies for vegetation restoration under drought conditions have been developed, the popularisation and application of new technologies has been accelerated, and the training of technicians for farmers and herdsmen has been strengthened. To improve the monitoring capability and technical level of desertification, the monitoring network system has been strengthened, and the popularisation and application of modern technologies are intensified (e.g., information and remote sensing) (Wu et al., 2015). Special laws on combating desertification have been decreed by the government. The provincial government responsibilities for desertification prevention and controlling objectives and laws have been strictly implemented.

Many studies showed that the projects generally played an active role in combating desertification and fighting against dust storms in China over the past several decades (*high confidence*) (Cao et al., 2018; State Forestry Administration of China, 2015; Wang et al., 2013; Wang et al., 2014; Yang et al. 2013). At the beginning of the project, some problems appeared in some places due to lack of enough knowledge and experience (*low confidence*) (Jiang, 2016; Wang et al., 2010). For example, some tree species selected were not well suited to local soil and climatic conditions (Zhu et al., 2007), and there was an inadequate consideration of the limitation of the amount of effective water on the carrying capacity of trees in some arid regions (Dai, 2011; Feng et al., 2016; 3.6.4). In addition, at the beginning of the project, there was an inadequate consideration of the effects of climate change on combating desertification (Feng et al., 2015; Tan and Li, 2015). Indeed, climate change and human activities over past years have influenced the desertification and dust storm control effects in China (Feng et al., 2015; Wang et al., 2009; Tan and Li, 2015), and future climate change will bring new challenges for combating desertification in China (Wang et al., 2017; Yin et al., 2015; Xu et al., 2019). In particular, the desertification risk in China will be enhanced at 2°C compared to 1.5°C global temperature rise (Ma et al., 2018). Adapting desertification control to climate change involves: improving the adaptation capacity to climate change for afforestation and grassland management by executing SLM practices; optimising the agricultural and animal husbandry structure; and using big data to fulfil the water resources regulation (Zhang and Huisingh, 2018). In particular, improving scientific and technological supports in desertification control is crucial for adaptation to climate change and combating desertification, including protecting vegetation in desertification-prone lands by planting indigenous plant species, facilitating natural restoration of vegetation to conserve biodiversity, employing artificial rain or snow, water saving irrigation and water storage technologies (Jin et al., 2014; Yang et al., 2013).

3.7.2.2. The Green Dam in Algeria

After independence in 1962, the Algerian government initiated measures to replant forests destroyed by the war and the steppes affected by desertification among its top priorities (Belaaz, 2003). In 1972, the government invested in the “Green Dam" (“Barrage Vert”) project. This was the first significant experiment to combat desertification, influence the local climate and decrease the aridity by restoring a barrier of trees. The Green Dam extends across arid and semi-arid zones between the isohyets 300 and 200 mm. It is a 3 M ha band of plantation running from east to west (Figure 3.12). It is over 1,200 km long (from the Algerian-Moroccan border to the Algerian-Tunisian border) and has an average width of about 20 km. The soils in the area are shallow, low in organic matter and susceptible to erosion. The main objectives of the project were to conserve natural resources, improve the living conditions of local residents and avoid their exodus to urban areas. During the first four decades (1970–2000) the success rate was low (42%) due to lack of participation by the local population and the choice of species (Bensaid, 1995).

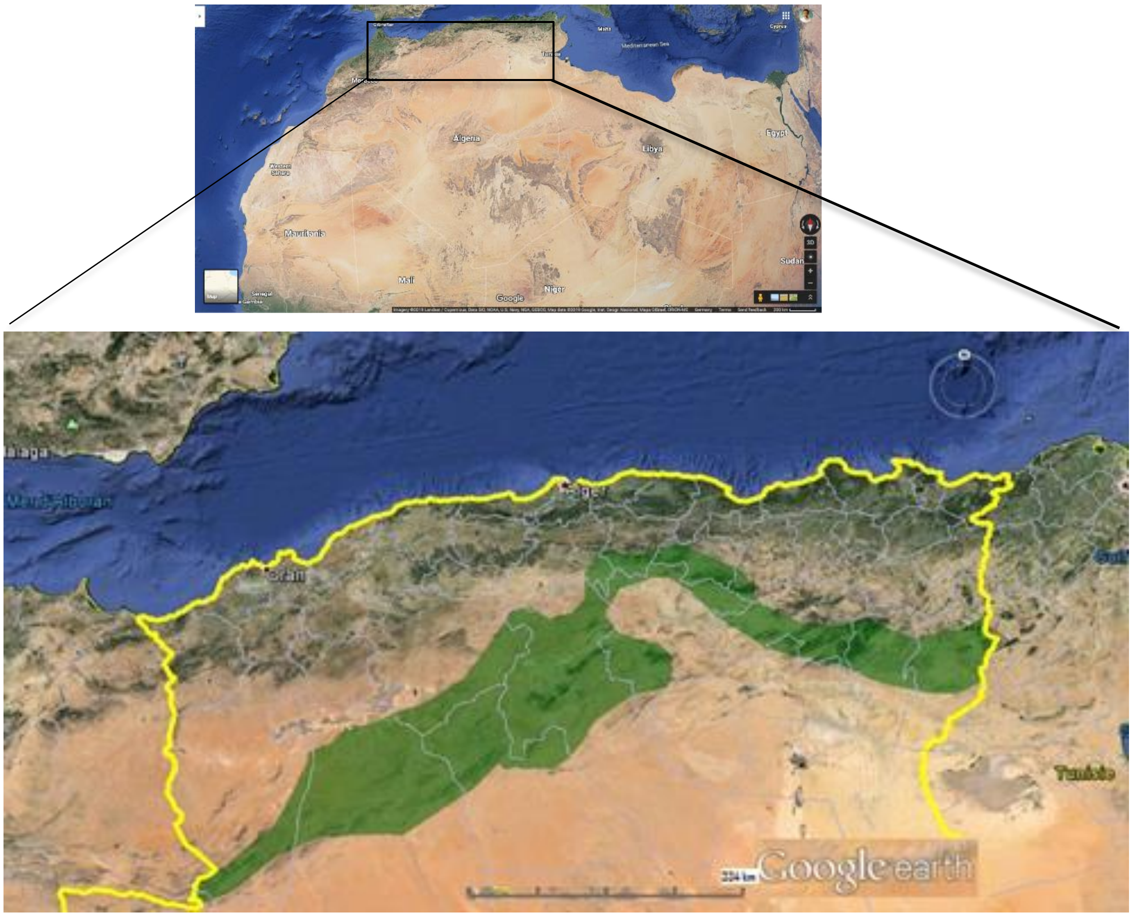

Figure 3.12 Localisation of the Green Dam in Algeria (Saifi et al., 2015). Note: The green coloured band represents the location of the Green Dam; the yellow band delineates the national border of Algeria. Source: GoogleEarth

The Green Dam did not have the desired effects. Despite tree planting efforts, desertification intensified on the steppes, especially in south-western Algeria due to the prolonged drought during the 1980s. Rainfall declined from 18% to 27%, and the dry season has increased by two months in the last century (Belala et al., 2018). Livestock numbers in the Green Dam regions, mainly sheep, have grown exponentially, leading to severe overgrazing, causing trampling and soil compaction, which greatly increased the risk of erosion. Wind erosion, very prevalent in the region, is due to climatic conditions and the strong anthropogenic action that reduced the vegetation cover. The action of the wind carries fine particles such as sands and clays and leaves on the soil surface a lag gravel pavement, which is unproductive. Water erosion is largely due to torrential rains in the form of severe thunderstorms that disintegrate the bare soil surface from raindrop impact (Achite et al., 2016). The detached soil and nutrients are transported offsite via runoff resulting in loss of fertility and water holding capacity. The risk of and severity of water erosion is a function of human land use activities that increase soil loss through removal of vegetative cover. The National Soil Sensitivity to Erosion Map (Salamani et al., 2012) shows that more than 3 M ha of land in the steppe provinces are currently experiencing intense wind activity (Houyou et al., 2016) and are areas at particular risk of soil erosion. Mostephaoui et al. (2013), estimates that each year there is a loss of 7 t h^{-1} of soils due to erosion. Nearly 0.6 M ha of land in the steppe zone are fully degraded without the possibility of biological recovery.

To combat the effects of erosion and desertification, the government has planned to relaunch the rehabilitation of the Green Dam by incorporating new concepts related to sustainable development, and adaptation to climate change. The experience of previous years has led to integrated rangeland management, improved tree and fodder shrub plantations and the development of water conservation techniques. Reforestation is carried out using several species, including fruit trees, to increase and diversify the sources of income of the population.

The evaluation of the Green Dam from 1972 to 2015 (Merdas et al., 2015) shows that 0.3 M ha of forest plantation have been planted, which represents 10% of the project area. Estimates of the success rate of reforestation vary considerably between 30% and 75%, depending on the region. Through demonstration, the Green Dam has inspired several African nations to build a Great Green Wall to combat land degradation, mitigate climate change effects, loss of biodiversity and poverty in a region that stretches from Senegal to Djibouti (Sahara and Sahel Observatory (OSS), 2016).

3.7.2.3. The Great Green Wall of the Sahara and the Sahel Initiative

The Great Green Wall is an initiative of the Heads of State and Government of the Sahelo-Saharan countries to mitigate and adapt to climate change, and to improve the food security of the Sahel and Saharan peoples (Sacande, 2018; M'Bow, 2017). Launched in 2007, this regional project aims to restore Africa's degraded arid landscapes, reduce the loss of biodiversity and support local communities to sustainable use of forests and rangelands. The Great Green Wall focuses on establishing plantations and neighbouring projects covering a distance of 7,775 km from Senegal on the Atlantic coast to Eritrea on the Red Sea coast, with a width of 15 km (Figure 3.13). The wall passes through Djibouti, Eritrea, Ethiopia, Sudan, Chad, Niger, Nigeria, Mali, Burkina Faso and Mauritania and Senegal.

The choice of woody and herbaceous species that will be used to restore degraded ecosystems is based on biophysical and socio-economic criteria, including socio-economic value (food, pastoral, commercial, energetic, medicinal, cultural); ecological importance (C sequestration, soil cover, water infiltration) and species that are resilient to climate change and variability. The Pan-African Agency of the Great Green Wall (PAGGW) was created in 2010 under the auspices of the African Union and CEN-SAD to manage the project. The initiative is implemented at the level of each country by a national structure. A monitoring and evaluation system has been defined, allowing nations to measure outcomes and to propose the necessary adjustments.

In the past, reforestation programs in the arid regions of the Sahel and North Africa that have been undertaken to stop desertification were poorly studied and cost a lot of money without significant success (Benjaminsen and Hiernaux, 2019). Today, countries have changed their strategies and opted for rural development projects that can be more easily funded. Examples of scalable practices for land restoration: Managing water bodies for livestock and crop production, promoting fodder trees reducing runoff (Mbow, 2017).

The implementation of the initiative has already started in several countries. For example, the FAO's Action Against Desertification project was restoring 18000 hectares of land in 2018 through planting native tree species in Burkina Faso, Ethiopia, the Gambia, Niger, Nigeria and Senegal (Sacande, 2018). Berrahmouni et al. (2016) estimated that 166 M ha can be restored in the Sahel, requiring the restoration of 10 M ha per year to achieve Land Degradation Neutrality targets by 2030. Despite this early implementation actions on the ground, the achievement of the planned targets is questionable and challenging without significant additional funding.

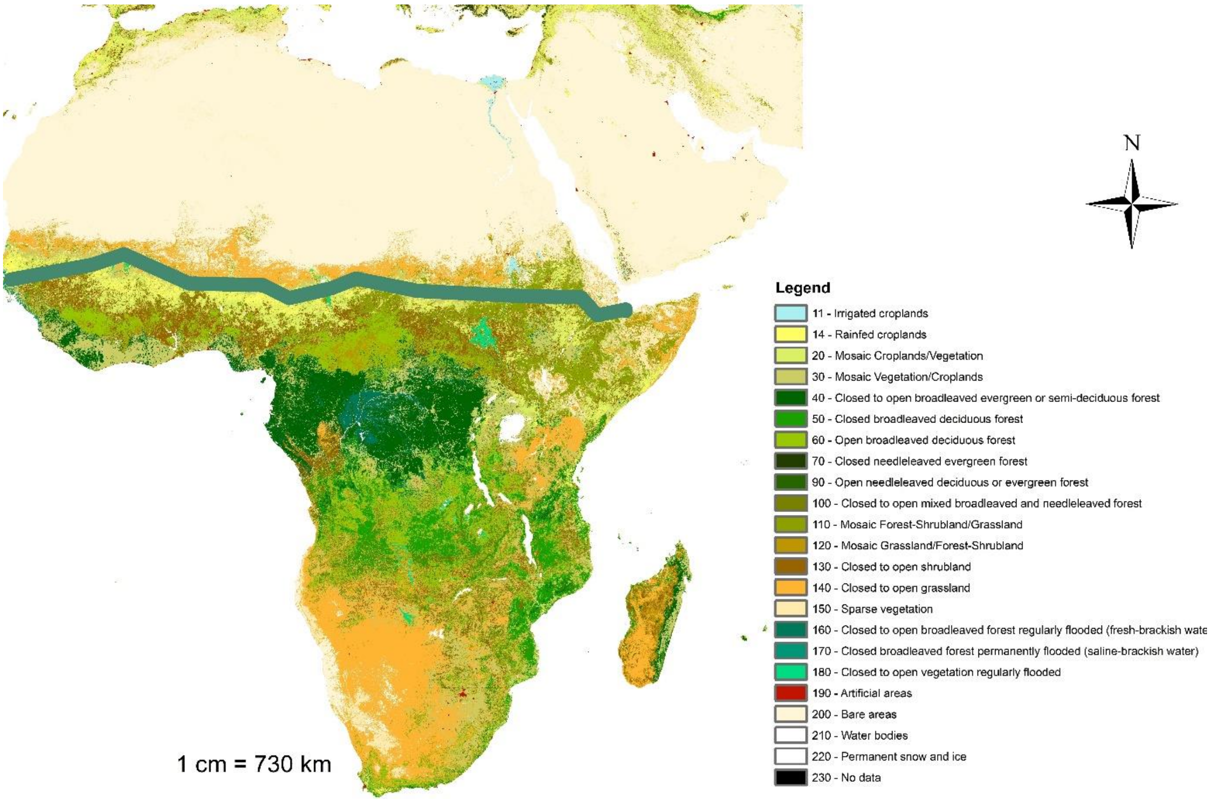

Figure 3.13 The Great Green Wall of the Sahara and the Sahel.
Source for the data layer: This dataset is an extract from the GlobCover 2009 land cover map, covering Africa and the Arabian Peninsula. The GlobCover 2009 land cover map is derived by an automatic and regionally-tuned classification of a time series of global MERIS (MEdium Resolution Imaging Spectrometer) FR mosaics for the year 2009. The global land cover map counts 22 land cover classes defined with the United Nations (UN) Land Cover Classification System (LCCS).

3.7.3. Invasive Plant Species

3.7.3.1. Introduction

The spread of invasive plants can be exacerbated by climate change (Bradley et al., 2010; Davis et al., 2000). In general, it is expected that the distribution of invasive plant species with high tolerance to drought or high temperatures may increase under most climate change scenarios (*medium* to *high confidence*; Bradley et al., 2010; Settele et al., 2014; Scasta et al., 2015). Invasive plants are considered a major risk to native biodiversity and can disturb the nutrient dynamics and water balance in affected ecosystems (Ehrenfeld, 2003). Compared to more humid regions, the number of species that succeed in invading dryland areas is low (Bradley et al., 2012), yet they have a considerable impact on biodiversity and ecosystem services (Le Maitre et al., 2015; 2011; Newton et al., 2011). Moreover, human activities in dryland areas are responsible for creating new invasion opportunities (Safriel et al., 2005).

Current drivers of species introductions include expanding global trade and travel, land degradation and changes in climate (Chytrý et al., 2012; Richardson et al., 2011; Seebens et al., 2018). For example, Davis et al. (2000) suggests that high rainfall variability promotes the success of alien plant species - as reported for semiarid grasslands and Mediterranean-type ecosystems (Cassidy et al., 2004; Reynolds et al., 2004; Sala et al., 2006). Furthermore, Panda et al. (2018) demonstrated that many invasive species could withstand elevated temperature and moisture scarcity caused by climate change. Dukes et al. (2011) observed that the invasive plant yellow-star thistle (*Centaurea solstitialis)* grew six time larger under elevated atmospheric CO_2 expected in future climate change scenarios.

Climate change is *likely* going to aggravate the problem as existing species continue to spread unabated and other species develop invasive characteristics (Hellmann et al., 2008). Although the effects of climate change on invasive species distributions have been relatively well explored, the greater impact on ecosystems is less well understood (Bradley et al., 2010; Eldridge et al., 2011).

Due to the time lag between the initial release of invasive species and their impact, the consequence of invasions is not immediately detected and may only be noticed centuries after introduction (Rouget et al., 2016). Climate change and invading species may act in concert (Bellard et al., 2013; Hellmann et al., 2008; Seebens et al., 2015). For example, invasion often changes the size and structure of fuel loads, which can lead to an increase in the frequency and intensity of fire (Evans et al., 2015). In areas where the climate is becoming warmer, an increase in the likelihood of suitable weather conditions for fire may promote invasive species, which in turn may lead to further desertification. Conversely, fire may promote plant invasions via several of mechanisms (by reducing cover of competing vegetation, destroying native vegetation and clearing a path for invasive plants or creating favourable soil conditions) (Brooks et al., 2004; Grace et al., 2001; Keeley and Brennan, 2012).

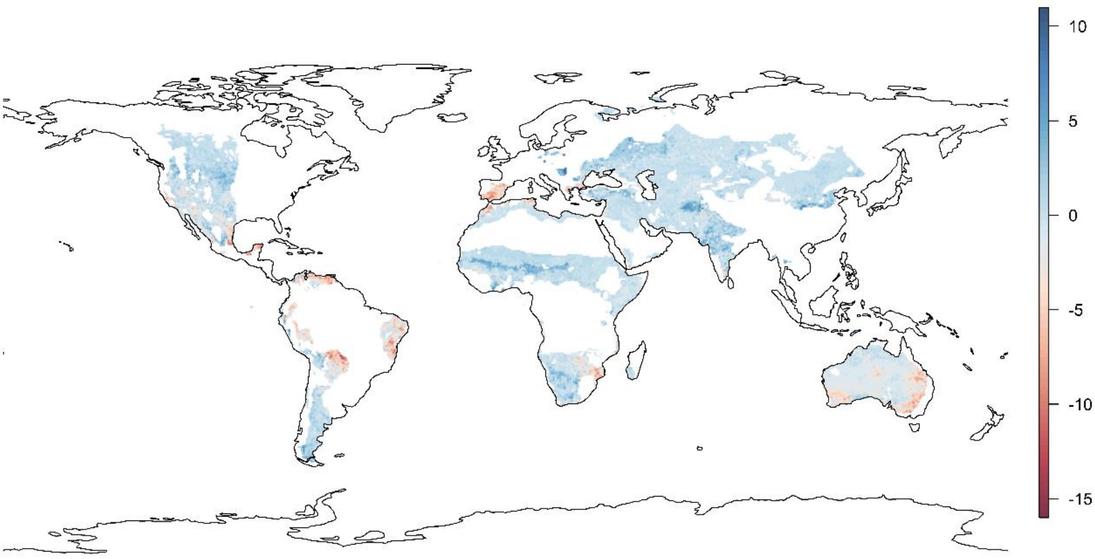

Figure 3.14 Difference between the number of invasive alien species (n=99, from(Bellard et al., 2013)) predicted to occur by 2050 (under A1B scenario) and current period "2000" within the dryland areas.

At a regional scale, Bellard et al. (2013) predicted increasing risk in Africa and Asia, with declining risk in Australia (Figure 3.14). This projection does not represent an exhaustive list of invasive alien species occurring in drylands.

A set of four case studies in Ethiopia, Mexico, the USA and Pakistan is presented below to describe the nuanced nature of invading plant species, their impact on drylands and their relationship with climate change.

3.7.3.2. Ethiopia

The two invasive plants that inflict the heaviest damage to ecosystems, especially biodiversity, are the annual herbaceous weed, *Parthenium hysterophorus (Asteraceae)* also known as Congress weed; and the tree species, *Prosopis juliflora (Fabaceae)* also called Mesquite both originating from southwestern United States to central - south America (Adkins and Shabbir, 2014). *Prosopis* was introduced in the 1970s and has since spread rapidly. *Prosopis,* classified as the highest priority invader in the country, is threatening livestock production and challenging the sustainability of the pastoral systems. Parthenium is believed to have been introduced along with relief aid during the debilitating droughts of the early 1980s, and a recent study reported that *the weed* has spread into 32 out of 34 districts in Tigray, the northernmost region of Ethiopia (Teka, 2016). A study by Etana et al. (2011) indicated that *Parthenium* caused a 69% decline in the density of herbaceous species in Awash National Park within a few years of introduction. In the presence of *Parthenium*, the growth and development of crops is suppressed due to its allelopathic properties. McConnachie et al. (2011) estimated a 28% crop loss across the country, including a 40-90% reduction in sorghum yield in eastern Ethiopia alone (Tamado et al., 2002). The weed is a substantial agricultural and natural resource problem and constitutes a significant health hazard (Fasil, 2011). Parthenium causes acute allergic respiratory problems, skin dermatitis, and reportedly mutagenicity both in human and livestock (Mekonnen, 2017; Patel, 2011). The eastern belt of Africa including Ethiopia presents a very suitable habitat, and the weed is expected to spread further in the region in the future (Mainali et al., 2015).

There is neither a comprehensive intervention plan nor a clear institutional mandate to deal with invasive weeds, however, there are fragmented efforts involving local communities even though they are clearly inadequate. The lessons learned are related to actions that have contributed to the current scenario are several. First, lack of coordination and awareness - mesquite was introduced by development agencies as a drought tolerant shade tree with little consideration of its invasive nature. If research and development institutions had been aware, a containment strategy could have been implemented early on. The second major lesson is the cost of inaction. When research and development organisations did sound the alarm, but the warnings went largely unheeded, resulting in the spread and buildup of two of the worst invasive plant species in the world (Fasil, 2011).

3.7.3.3. Mexico

Buffelgrass (*Cenchrus ciliaris* L.), a native species from southern Asia and East Africa, was introduced into Texas and northern Mexico in the 1930s and 1940s, as it is highly productive in drought conditions (Cox et al., 1988; Rao et al., 1996). In the Sonoran desert of Mexico, the distribution of buffelgrass has increased exponentially, covering 1 M ha in Sonora State (Castellanos-Villegas et al., 2002). Furthermore, its potential distribution extended to 53% of Sonora State and 12% of semiarid and arid ecosystems in Mexico (Arriaga et al., 2004). Buffelgrass has also been reported as an aggressive invader in Australia and the United States resulting in altered fire cycles that enhance further spread of this plant and disrupts ecosystem processes (Marshall et al., 2012; Miller et al., 2010; Schlesinger et al., 2013).

Castellanos et al. (2016) reported that soil moisture was lower in the buffelgrass savanna cleared 35 years ago than in the native semi-arid shrubland, mainly during the summer. The ecohydrological changes induced by buffelgrass can therefore displace native plant species over the long term. Invasion by buffelgrass can also affect landscape productivity, as it is not as productive as native vegetation (Franklin and Molina-Freaner, 2010). Incorporation of buffelgrass is considered a good management practice by producers and the government. For this reason, no remedial actions are undertaken.

3.7.3.4. United States

Sagebrush ecosystems have declined from 25 to 13 M ha since the late 1800s (Miller et al., 2011). A major cause is the introduction of non-native cheatgrass (*Bromus tectorum*), which is the most prolific invasive plant in the United States. Cheatgrass infests more than 10 M ha in the Great Basin and is expanding every year (Balch et al., 2013). It provides a fine-textured fuel that increases the intensity, frequency and spatial extent of fire (Balch et al., 2013). Historically, wildfire frequency was 60 to 110 years in Wyoming big sagebrush communities and has increased to five years following the introduction of cheatgrass (Balch et al., 2013; Pilliod et al., 2017).

The conversion of the sagebrush steppe biome into to annual grassland with higher fire frequencies has severely impacted livestock producers as grazing is not possible for a minimum of two years after fire. Furthermore, cheatgrass and wildfires reduce critical habitat for wildlife and negatively impact species richness and abundance – for example, the greater sage-grouse (*Centocercus urophasianus*) and pygmy rabbit (*Brachylagus idahoensis*) which are on the verge of being listed for federal protection (Crawford et al., 2004; Larrucea and Brussard, 2008; Lockyer et al., 2015).

Attempts to reduce cheatgrass impacts through reseeding of both native and adapted introduced species have occurred for more than 60 years (Hull and Stewart, 1949) with little success. Following fire, cheatgrass becomes dominant and recovery of native shrubs and grasses is improbable, particularly in relatively low elevation sites with minimal annual precipitation (less than 200 mm yr^{-1}) (Davies et al., 2012; Taylor et al., 2014). Current rehabilitation efforts emphasise the use of native and non-native perennial grasses, forbs, and shrubs (Bureau of Land Management, 2005). Recent literature suggests that these treatments are not consistently effective at displacing cheatgrass populations or re-establishing sage-grouse habitat with success varying with elevation and precipitation (Arkle et al., 2014; Knutson et al., 2014). Proper post-fire grazing rest, season-of-use, stocking rates, and subsequent management are essential to restore resilient sagebrush ecosystems before they cross a threshold and become an annual grassland (Chambers et al., 2014; Miller et al., 2011; Pellant et al., 2004). Biological soil crust protection may be an effective measure to reduce cheatgrass germination, as biocrust disturbance has been shown to be a key factor promoting germination of non-native grasses (Hernandez and Sandquist, 2011). Projections of increasing temperature (Abatzoglou and Kolden, 2011), and observed reductions in and earlier melting of snowpack in the Great Basin region (Harpold and Brooks, 2018; Mote et al., 2005) suggest that there is a need to understand current and past climatic variability as this will drive wildfire and invasions of annual grasses.

3.7.3.5. Pakistan

The alien plants invading local vegetation in Pakistan include *Brossentia papyrifera* (found in Islamabad Capital territory), *Parthenium hysterophorus* (found in Punjab and Khyber Pakhtunkhwa provinces), *Prosopis juliflora* (found all over Pakistan), *Eucalyptus camaldulensis* (found in Punjab and Sindh provinces), *Salvinia* (aquatic plant widely distributed in water bodies in Sindh), *Cannabis sativa* (found in Islamabad Capital Territory), *Lantana camara* and *Xanthium strumarium* (found in upper Punjab and Khyber Pakhtunkhwa provinces) (Khan et al., 2010; Qureshi et al., 2014). Most of these plants were introduced by the Forest Department decades ago for filling the gap between demand and supply of timber, fuelwood and fodder. These non-native plants have some uses but their disadvantages outweigh their benefits (Marwat et al., 2010; Rashid et al., 2014).

Besides being a source of biological pollution and a threat to biodiversity and habitat loss, the alien plants reduce the land value and cause huge losses to agricultural communities (Rashid et al., 2014). *Brossentia papyrifera,* commonly known as Paper Mulberry, is the root cause of inhalant pollen allergy for the residents of lush green Islamabad during spring. From February to April, the pollen allergy is at its peak with symptoms of severe persistent coughing with difficulty in breathing and wheezing. The pollen count, although variable at different times and days, can be as high as 55,000 m^{-3}. Early symptoms of the allergy include sneezing, itching in the eyes and skin, and blocked nose. With changing climate, the onset of disease is getting earlier, and pollen count is estimated to cross 55,000 m^{-3} (Rashid et al., 2014). About 45% of allergic patients in the twin cities of Islamabad and Rawalpindi showed positive sensitivity to the pollens (Marwat et al., 2010). Millions of rupees have been spent by the Capital Development Authority on pruning and cutting of Paper Mulberry trees but because of its regeneration capacity growth is regained rapidly (Rashid et al., 2014). Among other invading plants, *Prosopis juliflora* has allelopathic properties, and *Eucalyptus* is known to transpire huge amounts of water and deplete the soil of its nutrient elements (Qureshi et al., 2014).

Although Biodiversity Action Plan exists in Pakistan, it is not implemented in letter and spirit. The Quarantine Department focuses only on pests and pathogens but takes no notice of plant and animal species being imported. Also, there is no provision of checking the possible impacts of imported species on the environment (Rashid et al., 2014) and of carrying out bio assays of active allelopathic compounds of alien plants.

3.7.4. Oases in Hyper-arid Areas in the Arabian Peninsula and Northern Africa

Oases are isolated areas with reliable water supply from lakes and springs located in hyper-arid and arid zones (Figure 3.15). Oasis agriculture has long been the only viable crop production system throughout the hot and arid regions of the Arabian Peninsula and North Africa. Oases in hyper-arid climates are usually subject to water shortage as evapotranspiration exceeds rainfall. This often causes salinisation of soils. While many oases have persisted for several thousand years, many others have been abandoned, often in response to changes in climate or hydrologic conditions (Jones et al., 2019), providing testimony to societies' vulnerability to climatic shifts and raising concerns about similarly severe effects of anthropogenic climate change (Jones et al., 2019).

Figure 3.15. Oases across the Arabian Peninsula and North Africa (alphabetically by country): (a) Masayrat ar Ruwajah oasis, Ad Dakhiliyah Governorate, Oman. Photo: Eike Lüdeling; (b) Tasselmanet oasis, Ouarzazate Province, Morocco. Photo: Abdellatif Khattabi. (c) Al-Ahsa oasis, Al-Ahsa Governarate, Saudi Arabia. Photo: Shijan Kaakkara; (d) Zarat oasis, Governorate of Gabes, Tunisia. Photo: Hamda Aloui; The use rights for (a), (b) and (d) were granted by copyright holders; (c) is licensed under the Creative Commons Attribution 2.0 Generic license.

On the Arabian Peninsula and in North Africa, climate change is projected to have substantial and complex effects on oasis areas (Abatzoglou and Kolden, 2011; Ashkenazy et al., 2012; Bachelet et al., 2016; Guan et al., 2018; Iknayan and Beissinger, 2018; Ling et al., 2013). To illustrate, by the 2050s, the oases in southern Tunisia are expected to be affected by hydrological and thermal changes, with an average temperature increase of 2.7°C, a 29% decrease in precipitation and a 14% increase in evapotranspiration rate (Ministry of Agriculture and Water Resources of Tunisia and GIZ, 2007). In Morocco, declining aquifer recharge is expected to impact the water supply of the Figuig oasis (Jilali, 2014), as well as for the Draa Valley (Karmaoui et al., 2016). Saudi Arabia is expected to experience a 1.8–4.1°C increase in temperatures by 2050, which is forecast to raise agricultural water demand by 5-15% in order to maintain the level of production equal to that in 2011 (Chowdhury and Al-Zahrani, 2013). The increase of temperatures and variable pattern of rainfall over the central, north and south-western regions of Saudi Arabia may pose challenges for sustainable water resource management (Tarawneh and Chowdhury, 2018). Moreover, future climate scenarios are expected to increase the frequency of floods and flash floods, such as in the coastal areas along the central parts of the Red Sea and the south-southwestern areas of Saudi Arabia (Almazroui et al., 2017).

While many oases are cultivated with very heat-tolerant crops such as date palms, even such crops eventually lose in their productivity when temperatures exceed certain thresholds or hot conditions prevail for extended periods. Projections so far do not indicate severe losses in land suitability for date palm for the Arabian Peninsula (Aldababseh et al., 2018; Shabani et al., 2015). It is unclear, however, how reliable the climate response parameters in the underlying models are, and actual responses may differ substantially. Date palms are routinely assumed to be able to endure very high temperatures, but recent transcriptomic and metabolomic evidence suggests that heat stress reactions already occur at 35°C (Safronov et al., 2017), which is not exceptionally warm for many oases in the region. Given current assumptions about the heat-tolerance of date palm, however, adverse effects are expected to be small (Aldababseh et al., 2018; Shabani et al., 2015). For some other perennial oasis crops, impacts of temperature increases are already apparent. Between 2004-2005 and 2012-2013, high-mountain oases of Al Jabal Al Akhdar in Oman lost almost all fruit and nut trees of temperate-zone origin, with the abundance of peaches, apricots, grapes, figs, pears, apples, and plums dropping by between 86% and 100% (Al-Kalbani et al., 2016). This implies that that the local climate may not remain suitable for species that depend on cool winters to break their dormancy period (Luedeling et al., 2009). A similar impact is very probable in Tunisia and Morocco, as well as in other oasis locations in the Arabian Peninsula and North Africa (Benmoussa et al., 2007). All these studies expect strong decreases in winter chill, raising concerns that many currently well-established species will no longer be viable in locations where they are grown today. The risk of detrimental chill shortfalls is expected to increase gradually, slowly diminishing the economic prospects to produce such species. Without adequate adaptation actions, the consequences of this development for many traditional oasis settlements and other plantations of similar species could be highly negative.

At the same time, population growth and agricultural expansion in many oasis settlements are leading to substantial increases in water demand for human consumption (Al-Kalbani et al., 2014). For example, a large unmet water demand has been projected for future scenarios for the valley of Seybouse in East Algeria (Aoun-Sebaiti et al., 2014), and similar conclusions were drawn for Wadi El Natrun in Egypt (Switzman et al., 2018). Modelling studies have indicated long-term decline in available water and increasing risk of water shortages, e.g. for oases in Morocco (Johannsen et al., 2016; Karmaoui et al., 2016), the Dakhla oasis in Egypt's Western Desert (Sefelnasr et al., 2014) and for the large Upper Mega Aquifer of the Arabian Peninsula (Siebert et al., 2016). Mainly due to the risk of water shortages, Souissi et al. (2018) classified almost half of all farmers in Tunisia as non-resilient to climate change, especially those relying on tree crops, which limit opportunities for short-term adaptation actions.

The maintenance of the oasis systems and the safeguarding of their population's livelihoods are currently threatened by continuous water degradation, increasing soil salinisation, and soil contamination (Besser et al., 2017). Waterlogging and salinisation of soils due to rising saline groundwater tables coupled with inefficient drainage systems have become common to all continental oases in Tunisia, most of which are concentrated around saline depressions, known locally as chotts (Ben Hassine et al., 2013). Similar processes of salinisation are also occurring in the oasis areas of Egypt due to agricultural expansion, excessive use of water for irrigation and deficiency of the drainage systems (Abo-Ragab, 2010; Masoud and Koike, 2006). A prime example for this is Siwa oasis (Figure 3.16), a depression extending over 1050 km^2 in the north-western desert of Egypt in the north of the sand dune belt of the Great Sand Sea (Abo-Ragab, and Zaghloul, 2017). Siwa oasis has been recognised as a Globally Important Agricultural Heritage Site (GIAHS) by the FAO for being an *in situ* repository of plant genetic resources, especially of uniquely adapted varieties of date palm, olive and secondary crops that are highly esteemed for their quality and continue to play a significant role in rural livelihoods and diets (FAO, 2016).

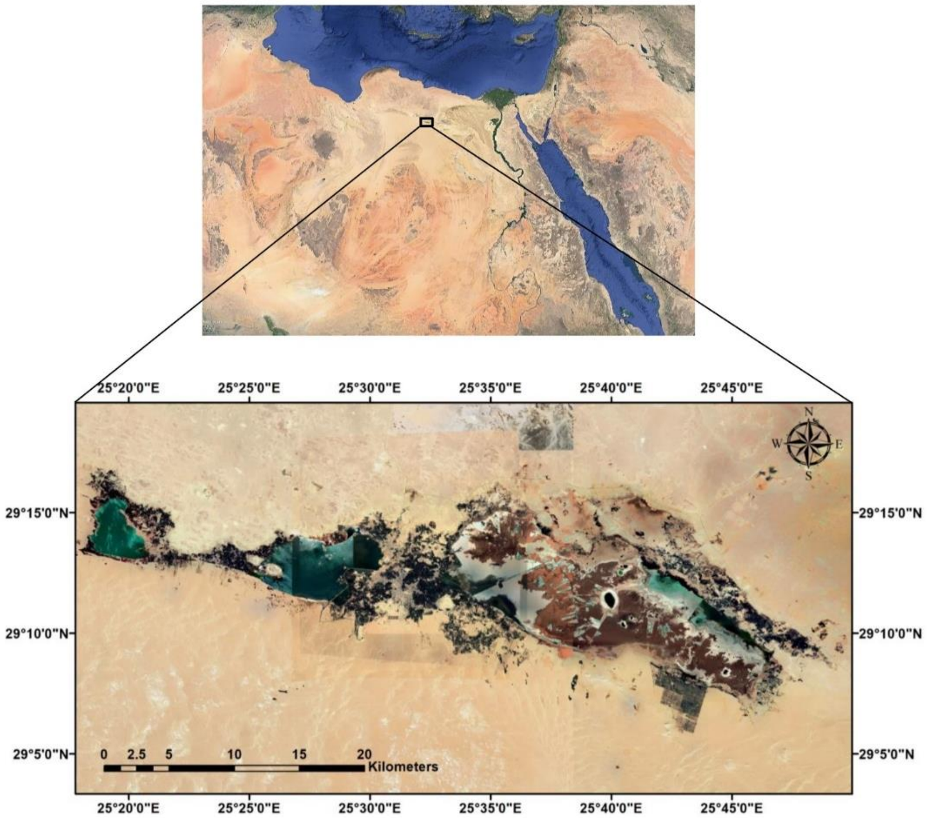

Figure 3.16. The Satellite Image of the Siwa Oasis, Egypt. Source: Google Maps.

The population growth in Siwa is leading rapid agricultural expansion and land reclamation. The Siwan farmers are converting the surrounding desert into reclaimed land by applying their old inherited traditional practices. Yet, agricultural expansion in the oasis mainly depends on non-renewable groundwaters. Soil salinisation and vegetation loss have been accelerating since 2000 due to water mismanagement and improper drainage systems (Masoud and Koike, 2006). Between 1990-2008, the cultivated area increased from 53 to 88 km^2, lakes from 60 to 76 km^2, sabkhas (salt flats) from 335 to 470 km^2, and the urban area from 6 to 10 km^2 (Abo-Ragab, 2010). The problem of rising groundwater tables was exacerbated by climatic changes (Askri et al., 2010; Gad and Abdel-Baki, 2002; Marlet et al., 2009).

Water supply is *likely* to become even scarcer for oasis agriculture under changing climate in the future than it is today, and viable solutions are difficult to find. While some authors stress the possibility to use desalinated water for irrigation (Aldababseh et al., 2018), the economics of such options, especially given the high evapotranspiration rates in the Arabian Peninsula and North Africa, are debatable. Many oases are located far from water sources that are suitable for desalination, adding further to feasibility constraints. Most authors therefore stress the need to limit water use (Sefelnasr et al., 2014), e.g. by raising irrigation efficiency (Switzman et al., 2018), reducing agricultural areas (Johannsen et al., 2016) or imposing water use restrictions (Odhiambo, 2017), and to carefully monitor desertification (King and Thomas, 2014). Whether adoption of crops with low water demand, such as sorghum (*Sorghum bicolor* (L.) Moench) or jojoba (*Simmondsia chinensis* (Link) C. K. Schneid.) (Aldababseh et al., 2018), can be a viable option for some oases remains to be seen, but given their relatively low profit margins compared to currently grown oasis crops, there are reasons to doubt the economic feasibility of such proposals. While it is currently unclear, to what extent oasis agriculture can be maintained in hot locations of the region, cooler sites offer potential for shifting towards new species and cultivars. Especially for tree crops, which have particular climatic needs across seasons. Resilient options can be identified, but procedures to match tree species and cultivars with site climate need to be improved to facilitate effective adaptation.

There is *high confidence* that many oases of North Africa and the Arabian Peninsula are vulnerable to climate change. While the impacts of recent climate change are difficult to separate from the consequences of other change processes, it is *likely* that water resources have already declined in many places and the suitability of the local climate for many crops, especially perennial crops, has already decreased. This decline of water resources and thermal suitability of oasis locations for traditional crops is *very likely* to continue throughout the 21st century. In the coming years, the people living in oasis regions across the world will face challenges due to increasing impacts of global environmental change (Chen et al., 2018). Hence, efforts to increase their adaptive capacity to climate change can facilitate the sustainable development of oasis regions globally. This will concern particularly addressing the trade-offs between environmental restoration and agricultural livelihoods (Chen et al., 2018). Ultimately, sustainability in oasis regions will depend on policies integrating the provision of ecosystem services and social and human welfare needs (Wang et al., 2017).

3.7.5. Integrated Watershed Management

Desertification has resulted in significant loss of ecosystem processes and services as described in detail in this chapter. The techniques and processes to restore degraded watersheds are not linear and integrated watershed management (IWM) must address physical, biological and social approaches to achieve SLM objectives (German et al., 2007).

3.7.5.1. Jordan

Population growth, migration into Jordan and changes in climate have resulted in desertification of the Jordan Badia region. The Badia region covers more than 80% of the country’s area and receives less than 200 mm of rainfall per year, with some areas receiving less than 100 mm (Al-Tabini et al., 2012). Climate analysis has indicated a generally increasing dryness over the West Asia and Middle Eastern region (AlSarmi and Washington, 2011; Tanarhte et al., 2015) with reduction in average annual rainfall in Jordan’s Badia area (De Pauw et al., 2015). The incidence of extreme rainfall events has not declined over the region. Locally increased incidence of extreme events over the Mediterranean region have been proposed (Giannakopoulos et al., 2009).

The practice of intensive and localised livestock herding, in combination with deep ploughing and unproductive barley agriculture, are the main drivers of severe land degradation and depletion of the rangeland natural resources. This affected both the quantity and the diversity of vegetation as native plants with a high nutrition value were replaced with invasive species with low palatability and nutritional content (Abu-Zanat et al., 2004). The sparsely covered and crusted soils in Jordan's Badia area have a low rainfall interception and infiltration rate, which leads to increased surface runoff and subsequent erosion and gullying, speeding up the drainage of rainwater from the watersheds that can result in downstream flooding in Amman, Jordan (Oweis, 2017).

Figure 3.17. Fresh Vallerani micro water harvesting catchment (a) and aerial imaging showing micro water harvesting catchment treatment after planting (b) and 1 year after treatment (c). Source: Stefan Strohmeier

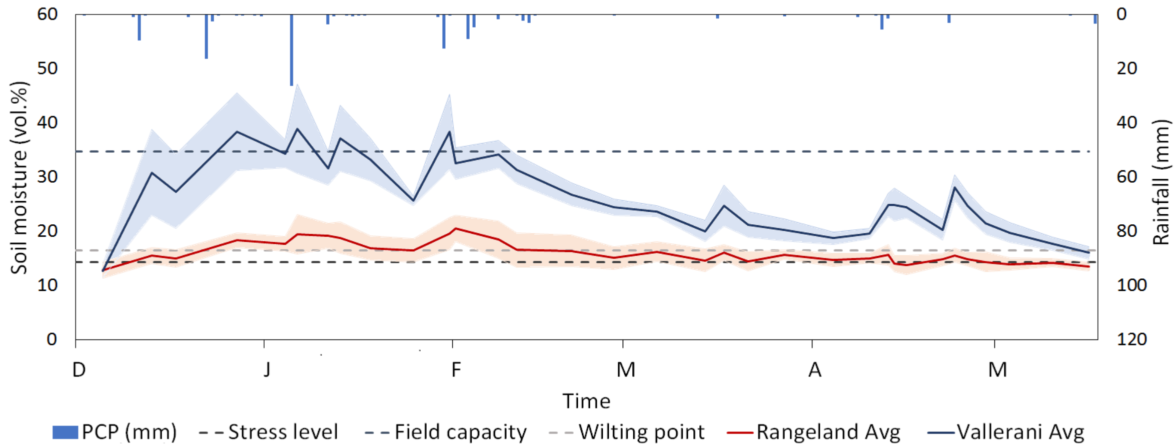

Figure 3.18 Illustration of enhanced soil water retention in the Mechanized Micro Rainwater Harvesting compared to untreated Badia rangelands in Jordan, showing precipitation (PCP), sustained stress level resulting in decreased production, Field Capacity and Wilting Point for available soil moisture, and then measured soil moisture content between the two treatments (degraded rangeland and the restored rangeland with the Vallerani plow).

To restore the desertified Badia an IWM plan was developed using hillslope implemented water harvesting micro catchments as a targeted restoration approach (Tabieh et al., 2015). Mechanized Micro Rainwater Harvesting (MIRWH) technology using the 'Vallerani plough' (Antinori and Vallerani, 1994; Gammoh and Oweis, 2011; Ngigi, 2003) is being widely applied for rehabilitation of highly degraded rangeland areas in Jordan. Tractor digs out small water harvesting pits on the contour of the slope (Figure 3.17) allowing the retention, infiltration and the local storage of surface runoff in the soil (Oweis, 2017). The micro catchments are planted with native shrub seedlings, such as saltbush (*Atriplex halimus*), with enhanced survival as a function of increased soil moisture (Figure 3.18) and increased dry matter yields (>300 kg ha^{-1}) that can serve as forage for livestock (Oweis, 2017; Tabieh et al., 2015).

Simultaneously to MIRWH upland measures, the gully erosion is being treated through intermittent stone plug intervention (Figure 3.19), stabilising the gully beds, increasing soil moisture in proximity of the plugs and dissipating the surface runoff's energy, and mitigating further back-cutting erosion and quick drainage of water. Eventually, the treated gully areas silt up and dense vegetation cover can re-establish. In addition, grazing management practices are implemented to increase the longevity of the treatment. Ultimately, the recruitment processes and revegetation shall control the watershed's hydrological regime through rainfall interception, surface runoff deceleration and filtration, combined with the less erodible and enhanced infiltration characteristics of the rehabilitated soils. In-depth understanding of the Badia's rangeland status transition, coupled with sustainable rangeland management, are still subject to further investigation, development and adoption; required to mitigate the ongoing degradation of the Middle Eastern rangeland ecosystems.

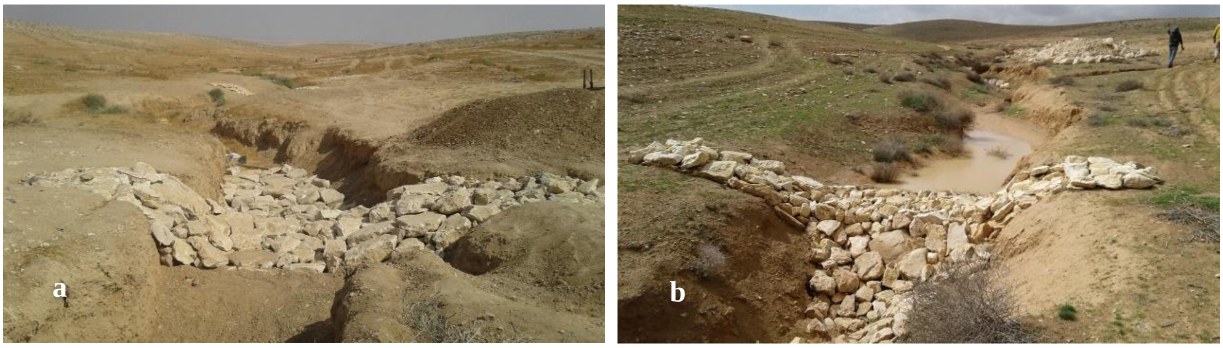

Figure 3.19 Gully plug development in September 2017 (a) and post rainfall event in March 2018 (b) near Amman, Jordan. Source: Stefan Strohmeire.

Oweis (2017) indicated that costs of the fully automated Vallerani technique was approximately USD 32 ha^{-1}. The total cost of the restoration package included the production, planting, and maintenance of the shrub seedlings (USD 11 ha^{-1}). Tabieh et al. (2015) calculated a benefit cost ratio (BCR) of > 1.5 for revegetation of degraded Badia areas through MIRWH and saltbush. However, costs vary based on the seedling's costs and availability of trained labour.

Water harvesting is not a recent scientific advancement. Water harvesting has been documented having evolved during the Bronze Age and was widely practiced in the Negev Desert during the Byzantine time period (1300-1600 years ago) (Fried et al., 2018; Stavi et al., 2017). Through construction of various structures made for packed clay and stone, water was either held on site in half-circular dam structures (Hafir) that faced up slope to capture runoff or on terraces that slowed water allowing it to infiltrate and to be stored in the soil profile. Numerous other systems were designed to capture water in below ground cisterns to be used later to provide water to livestock or for domestic use. Other water harvesting techniques divert runoff from hillslopes or wadis and spread the water in a systematic manner across playas and the toe slope of a hillslope. These systems allow production of crops in areas with 100 mm of average annual precipitation by harvesting an additional 300+ mm of water (Beckers et al., 2013). Water harvesting provides a proven technology to mitigate or adapt to climate change where precipitation maybe reduced and allow for small scale crop and livestock production to continue supporting local needs.

3.7.5.2. India

The Green Revolution that transformed irrigated agriculture in India had little effect on agricultural productivity in the rainfed and semi-arid regions, where land degradation and drought were serious concerns. In response to this challenge, integrated watershed management (IWM) projects were implemented over large areas in semi-arid biomes over the past few decades. IWM was meant to become a key factor in meeting a range of social development goals in many semi-arid rainfed agrarian landscapes in India (Bouma et al., 2007; Kerr et al., 2002). Over the years, watershed development has become the fulcrum of rural development that has the potential to achieve the twin objectives of ecosystem restoration and livelihood assurance in the drylands of India (Joy et al., 2004).

Some reports indicate significant improvements in mitigation of drought impacts, raising crop, fodder and livestock productivity, expanding the availability of drinking water and increasing incomes as a result of IWM (Rao, 2000), but overall the positive impact of the programme has been questioned and except in a few cases the performance has not lived up to expectations (Joy et al., 2004; JM Kerr et al., 2002). Rigorous comparisons of catchments with and without IWM projects have shown no significant enhancement of biomass (Bhalla et al., 2013). The factors contributing to the successful cases were found to include effective participation of stakeholders in management (Rao, 2000; Ratna Reddy et al., 2004) .

Attribution of success to soil and water conservation measures was confounded by inadequate monitoring of rainfall variability and lack of catchment hydrologic indicators (Bhalla et al., 2013). Social and economic trade-offs included bias of benefits to downstream crop producers at the expense of pastoralists, women and upstream communities. This biased distribution of IWM benefits could potentially be addressed by compensation for environmental services between communities (Kerr et al., 2002). The successes in some areas also led to increased demand for water, especially groundwater, since there has been no corresponding social regulation of water use after improvement in water regime (Samuel et al., 2007). Policies and management did not ensure water allocation to sectors with the highest social and economic benefits (Batchelor et al., 2003). Limited field evidence of the positive impacts of rainwater harvesting at the local scale is available, but there are several potential negative impacts at the watershed scale (Glendenning et al., 2012). Furthermore, watershed projects are known to have led to more water scarcity, higher expectations for irrigation water supply, further exacerbating water scarcity (Bharucha et al., 2014).

In summary, the overall poor performance of IWM projects have been linked to several factors. These include inequity in the distribution of benefits (Kerr et al., 2002), focus on institutional aspects rather than application of appropriate watershed techniques and functional aspects of watershed restoration (Joy et al., 2006; Vaidyanathan, 2006), mismatch between scales of focus and those that are optimal for catchment processes (Kerr, 2007), inconsistencies in criteria used to select watersheds for IWM projects (Bhalla et al., 2011), and in a few cases additional costs and inefficiencies of local non-governmental organisations (Chandrasekhar et al., 2006; Deshpande, 2008). Enabling policy responses for improvement of IWM performance include a greater emphasis on ecological restoration rather than civil engineering, sharper focus on sustainability of livelihoods than just conservation, adoption of a water justice as a normative goal and minimising externalities on non-stakeholder communities, rigorous independent biophysical monitoring with feedback mechanisms and integration with larger schemes for food and ecological security and maintenance of environmental flows for downstream areas (Bharucha et al., 2014; Calder et al., 2008; Joy et al., 2006). Successful adaptation of IWM would largely depend on how IWM creatively engages with dynamics of large scale land use and hydrology under a changing climate, involvement of livelihoods and rural incomes in ecological restoration, regulation of groundwater use and changing aspirations of rural population (*robust evidence*, *high agreement*) (O'Brien et al., 2004; Samuel et al., 2007; Samuel and Joy, 2018).

3.7.5.3. Limpopo River Basin

Covering an area of 412938 km^2, the Limpopo River basin spans parts of Botswana, South Africa, Zimbabwe and Mozambique, eventually entering into the Mozambique Channel. It has been selected as a case study as it provides a clear illustration of the combined effect of desertification and climate change, and why IWM may be crucial component of reducing exposure to climate change. It is predominantly a semi-arid area with an average annual rainfall of 400 mm (Mosase and Ahiablame, 2018). Rainfall is both highly seasonal and variable with the prominent impact of the El Nino / La Nina phenomena and the Southern Oscillation leading to severe droughts (Jury, 2016). It is also exposed to tropical cyclones that sweep in from the Mozambique Channel often leading to extensive casualties and the destruction of infrastructure (Christie and Hanlon, 2001). Furthermore, there is good agreement across climate models that the region is going to become warmer and drier, with a change in the frequency of floods and droughts (Engelbrecht et al., 2011; Zhu and Ringler, 2012).

Seasonality is predicted to increase, which in turn may increase the frequency of flood events in an area that is already susceptible to flooding (Spaliviero et al., 2014) .

A clear need exists to both address exposure to flood events as well as predicted decreases in water availability, which are already acute. Without the additional impact of climate change, the basin is rapidly reaching a point where all available water has been allocated to users (Kahinda et al., 2016; Zhu and Ringler, 2012). The urgency of the situation was identified several decades ago (FAO, 2004), with the countries of the Basin recognising that responses are required at several levels, both in terms of system governance as well as addressing land degradation.

Recent reviews of the governance and implementation of IWM within the basin recognise that an integrated approach is needed and that a robust institutional, legal, political, operational, technical and support is crucial (Alba et al., 2016; Gbetibouo et al., 2010; Machethe et al., 2004; Spaliviero et al., 2011; van der Zaag and Savenije, 1999). Within the scope of emerging lessons, two principal ones emerge. The first is capacity and resource constraints at most levels. Limited capacity within Limpopo Watercourse Commission (LIMCOM) and national water management authorities constrains the implementation of IWM planning processes (Kahinda et al., 2016; Spaliviero et al., 2011). Whereas strategy development is often relatively well-funded and resourced through donor funding, long-term implementation is often limited due to competing priorities. The second is adequate representation of all parties in the process in order to address existing inequalities and ensure full integration of water management. For example, within Mozambique, significant strides have been made towards the decentralisation of river basin governance and IWM. Despite a good progress, Alba et al. (2016) found that the newly implemented system may enforce existing inequalities as not all stakeholders, particularly smallholder farmers, are adequately represented in emerging water management structures and are often inhibited by financial and institutional constraints. Recognising economic and socio-political inequalities and explicitly considering them to ensure the representation of all participants can increase the chances of successful IWM implementation.

3.8. Knowledge Gaps and Key Uncertainties

- Desertification has been studied for decades and different drivers of desertification have been described, classified, and are generally understood (e.g., overgrazing by livestock or salinisation from inappropriate irrigation) (D'Odorico et al., 2013). However, there are knowledge gaps on the extent and severity of desertification at global, regional, and local scales (Zhang and Huisingh, 2018; Zucca et al., 2012). Overall, improved estimation and mapping of areas undergoing desertification is needed. This requires a combination of rapidly expanding sources of remotely sensed data, ground observations and new modelling approaches. This is a critical gap, especially in the context of measuring progress towards achieving the Land Degradation Neutrality target by 2030 in the framework of SDGs.

- Despite numerous relevant studies, consistent indicators for attributing desertification to climatic and/or human causes are still lacking due to methodological shortcomings.

- Climate change impacts on dust and sand storm activity remain a critical gap. In addition, the impacts of dust and sand storms on human welfare, ecosystems, crop productivity and animal health are not measured, particularly in the highly affected regions such as the Sahel, North Africa, the Middle East and Central Asia. Dust deposition on snow and ice has been found in many regions of the globe (e.g. Painter et al., 2018; Kaspari et al., 2014; Qian et al., 2015; Painter et al. 2013), however, the quantification of the effect globally and estimation of future changes in the extent of this effect remain knowledge gaps.

- Future projections of combined impacts of desertification and climate change on ecosystem services, fauna and flora, are lacking, even though this topic is of considerable social importance. Available information is mostly on separate, individual impacts of either (mostly) climate change or desertification. Responses to desertification are species-specific and mechanistic models are not yet able to accurately predict individual species responses to the many factors associated with desertification under changing climate.

- Previous studies have focused on the general characteristics of past and current desertification feedbacks to the climate system, however, the information on the future interactions between climate and desertification (beyond changes in the aridity index) are lacking. The knowledge of future climate change impacts on such desertification processes as soil erosion, salinisation, and nutrient depletion remains limited both at the global and at the local levels.

- Further research to develop technologies and innovations needed to combat desertification is required but also better understanding of reasons for the observed poor adoption of available innovations is important to improve adoption rates.

- Desertification under changing climate has a high potential to increase poverty particularly through the risks coming from extreme weather events (Olsson et al., 2014). However, the evidence rigorously attributing changes in observed poverty to climate change impacts is currently not available.

- The knowledge on limits to adaptation to combined effects of climate change and desertification is insufficient. This is an important gap since the potential for residual risks and maladaptive outcomes is high.

- Filling these gaps involves considerable investments in research and data collection. Using Earth observation systems in a standardised approach could help fill some of these gaps. This would increase data comparability and reduce uncertainty in approaches and costs. Systematically collected data would provide far greater insights than incomparable fragmented data.

Frequently Asked Questions

FAQ 3.1 How does climate change affect desertification?

Desertification is land degradation in drylands. Climate change and desertification have strong interactions. Desertification affects climate change through loss of fertile soil and vegetation. Soils contain large amounts of carbon some of which could be released to the atmosphere due to desertification, with important repercussions for the global climate system. The impacts of climate change on desertification are complex and knowledge on the subject is still insufficient. On the one hand, some dryland regions will receive less rainfall and increases in temperatures can reduce soil moisture harming plant growth. On the other hand, the increase of CO_2 in the atmosphere can enhance plant growth if there are enough water and soil nutrients available.

FAQ 3.2 How can climate change induced desertification be avoided, reduced or reversed?

Managing land sustainably can help avoid, reduce or reverse desertification, and contribute to climate change mitigation and adaptation. Such sustainable land management practices include reducing soil tillage and maintaining plant residues to keep soils covered, planting trees on degraded lands, growing a wider variety of crops, applying efficient irrigation methods, improving rangeland grazing by livestock and many others.

FAQ 3.2 How do sustainable land management practices affect ecosystem services and biodiversity?

Sustainable land management practices help improve ecosystems services and protect biodiversity. For example, conservation agriculture and better rangeland management can increase the production of food and fibres. Planting trees on degraded lands can improve soil fertility and fix carbon in soils. Sustainable land management practices also support biodiversity through habitat protection. Biodiversity protection allows to safeguard precious genetic resources, thus, contributing to human wellbeing.

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Chapter 4: Land Degradation

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Executive Summary

Land degradation affects people and ecosystems throughout the planet and is both affected by climate change and contributes to it. In this report, land degradation is defined as a *negative trend in land condition, caused by direct or indirect human-induced processes including anthropogenic climate change, expressed as long-term reduction or loss of at least one of the following: biological productivity, ecological integrity, or value to humans.* Forest degradation is land degradation which occurs in forest land. Deforestation is the conversion of forest to non-forest land and can result in land degradation. {4.1.3}

Land degradation adversely affects people's livelihoods *(very high confidence)* and occurs over a quarter of the Earth's ice-free land area *(medium confidence)*. The majority of the 1.3 to 3.2 billion affected people *(low confidence)* are living in poverty in developing countries *(medium confidence)*. Land use changes and unsustainable land management are direct human causes of land degradation (*very high confidence*), with agriculture being a dominant sector driving degradation (*very high confidence*). Soil loss from conventionally tilled land exceeds the rate of soil formation by >2 orders of magnitude (*medium confidence*). Land degradation affects humans in multiple ways, interacting with social, political, cultural and economic aspects, including markets, technology, inequality and demographic change (*very high confidence*). Land degradation impacts extend beyond the land surface itself, affecting marine and freshwater systems, as well as people and ecosystems far away from the local sites of degradation (*very high confidence*). {4.1.6, 4.2.1, 4.2.3, 4.3, 4.6.1, 4.7, Table 4.1}

Climate change exacerbates the rate and magnitude of several ongoing land degradation processes and introduces new degradation patterns (*high confidence*). Human-induced global warming has already caused observed changes in two drivers of land degradation: increased frequency, intensity and/or amount of heavy precipitation (*medium confidence*), and increased heat stress *(high confidence*). Global warming beyond that of present-day will further exacerbate ongoing land degradation processes through increasing floods (*medium confidence*), drought frequency and severity (*medium confidence*), intensified cyclones (*medium confidence*), and sea-level rise (*very high confidence*), with outcomes being modulated by land management (*very high confidence*). Permafrost thawing due to warming (*high confidence*), and coastal erosion due to sea level rise and impacts of changing storm paths (*low confidence*), are examples of land degradation affecting places in which it has not typically been a problem. Erosion of coastal areas because of sea level rise will increase worldwide (*high confidence*). In cyclone prone areas the combination of sea level rise and more intense cyclones will cause land degradation with serious consequences for people and livelihoods (*very high confidence*). {4.2.1, 4.2.2, 4.2.3, 4.4.1, 4.4.2, 4.9.6, Table 4.1}

Land degradation and climate change, both individually and in combination, have profound implications for natural resource-based livelihood systems and societal groups (*high confidence*). The number of people whose livelihood depends on degraded lands has been estimated to ~1.5 billion worldwide (*very low confidence*). People in degraded areas who directly depend on natural resources for subsistence, food security and income, including women and youth with limited adaptation options, are especially vulnerable to land degradation and climate change (*high confidence*). Land degradation reduces land productivity and increases the workload of managing the land, affecting women disproportionally in some regions. Land degradation and climate change act as threat multipliers for already precarious livelihoods (*very high confidence*), leaving them highly sensitive to extreme climatic events, with consequences such as poverty and food insecurity (*high confidence*), and in some cases migration, conflict and loss of cultural heritage (*low confidence*). Changes in vegetation cover and distribution due to climate change increase risks of land degradation in some areas (*medium confidence*). Climate change will have detrimental effects on livelihoods, habitats, and infrastructure through increased rates of land degradation (*high confidence*) and from new degradation patterns (*low evidence, high agreement*). {4.1.6, 4.2.1, 4.7}

Land degradation is a driver of climate change through emission of greenhouse gases and reduced rates of carbon uptake *(very high confidence*). Since 1990, globally the forest area has decreased by 3% (*low confidence*) with net decreases in the tropics and net increases outside the tropics (*high confidence*). Lower carbon density in re-growing forests compared to carbon stocks before deforestation results in net emissions from land use change (*very high confidence*). Forest management that reduces carbon stocks of forest land also leads to emissions, but global estimates of these emissions are uncertain. Cropland soils have lost 20-60% of their organic carbon content prior to cultivation, and soils under conventional agriculture continue to be a source of greenhouse gases (*medium confidence*). Of the land degradation processes, deforestation, increasing wildfires, degradation of peat soils, and permafrost thawing contribute most to climate change through the release of greenhouse gases and the reduction in land carbon sinks following deforestation (*high confidence*). Agricultural practices also emit non-CO_2 greenhouse gases from soils and these emissions are exacerbated by climate change (*medium confidence*). Conversion of primary to managed forests, illegal logging and unsustainable forest management result in greenhouse gas emissions (*very high confidence*) and can have additional physical effects on the regional climate including those arising from albedo shifts (*medium confidence*). These interactions call for more integrative climate impact assessments. {4.2.2, 4.3, 4.5.4, 4.6}

Large-scale implementation of dedicated biomass production for bioenergy increases competition for land with potentially serious consequences for food security and land degradation *(high confidence)*. Increasing the extent and intensity of biomass production through e.g. fertiliser additions, irrigation or monoculture energy plantations can result in local land degradation. Poorly implemented intensification of land management contributes to land degradation (e.g., salinisation from irrigation) and disrupted livelihoods (*high confidence*). In areas where afforestation and reforestation occur on previously degraded lands, opportunities exist to restore and rehabilitate lands with potentially significant co-benefits (*high confidence*) that depend on whether restoration involves natural or plantation forests. The total area of degraded lands has been estimated at 1-6 Mkm^2 (*very low confidence*). The extent of degraded and marginal lands suitable for dedicated biomass production is highly uncertain and cannot be established without due consideration of current land use and land tenure. Increasing the area of dedicated energy crops can lead to land degradation elsewhere through indirect land use change (*medium confidence*). Impacts of energy crops can be reduced through strategic integration with agricultural and forestry systems (*high confidence*) but the total quantity of biomass that can be produced through synergistic production systems is unknown. {4.1.6, 4.4.2, 4.5, 4.7.1, 4.8.1, 4.8.3, 4.8.4, 4.9.3}

Reducing unsustainable use of traditional biomass reduces land degradation and emissions of CO_2, while providing social and economic co-benefits (*very high confidence*). Traditional biomass in the form of fuelwood, charcoal and agricultural residues remains a primary source of energy for more than one-third of the global population leading to unsustainable use of biomass resources and forest degradation and contributing around 2% of global greenhouse gas (GHG) emissions (*low confidence*). Enhanced forest protection, improved forest and agricultural management, fuel-switching and adoption of efficient cooking and heating appliances can promote more sustainable biomass use and reduce land degradation, with co-benefits of reduced GHG emissions, improved human health, and reduced workload especially for women and youth (*very high confidence*). {4.1.6, 4.5.4}

Land degradation can be avoided, reduced or reversed by implementing sustainable land management, restoration and rehabilitation practices that simultaneously provide many co-benefits, including adaptation to and mitigation of climate change *(high confidence)*. Sustainable land management is a comprehensive array of technologies and enabling conditions, which have proven to address land degradation at multiple landscape scales, from local farms *(very high confidence)* to entire watersheds *(medium confidence)*. Sustainable forest management can prevent deforestation, maintain and enhance carbon sinks and can contribute towards greenhouse gas emissions reduction goals. Sustainable forest management generates socio-economic benefits, provides fiber, timber and biomass to meet society's growing needs. While sustainable forest management sustains high carbon sinks, the conversion from primary forests to sustainably managed forests can result in carbon emission during the transition and can result in loss of biodiversity *(high confidence)*. Conversely, in areas of degraded forests, sustainable forest management can increase carbon stocks and biodiversity *(medium confidence)*. Carbon storage in long-lived wood products and reductions of emissions from use of wood products to substitute for emissions-intensive materials also contribute to mitigation objectives. {4.8, 4.9, Table 4.2}

Lack of action to address land degradation will increase emissions and reduce carbon sinks and is inconsistent with the emission reductions required to limit global warming to 1.5°C or 2°C. (*high confidence*). Better management of soils can offset 5–20% of current global anthropogenic GHG emissions (*medium confidence*). Measures to avoid, reduce and reverse land degradation are available but economic, political, institutional, legal and socio-cultural barriers, including lack of access to resources and knowledge, restrict their uptake (*very high confidence*). Proven measures that facilitate implementation of practices that avoid, reduce, or reverse land degradation include tenure reform, tax incentives, payments for ecosystem services, participatory integrated land use planning, farmer networks and rural advisory services. Delayed action increases the costs of addressing land degradation, and can lead to irreversible biophysical and human outcomes (*high confidence*). Early actions can generate both site specific and immediate benefits to communities affected by land degradation, and contribute to long-term global benefits through climate change mitigation (*high confidence*). {4.1.5, 4.1.6, 4.7.1, 4.8, Table 4.2}

Even with adequate implementation of measures to avoid, reduce and reverse land degradation there will be residual degradation in some situations *(high confidence)*. Limits to adaptation are dynamic, site specific and are determined through the interaction of biophysical changes with social and institutional conditions. Exceeding the limits of adaptation will trigger escalating losses or result in undesirable changes, such as forced migration, conflicts, or poverty. Examples of potential limits to adaptation due to climate change induced land degradation are coastal erosion where land disappears, collapsing infrastructure and livelihoods due to thawing of permafrost, and extreme forms of soil erosion. {4.7, 4.8.5, 4.8.6, 4.9.6, 4.9.7, 4.9.8}

Land degradation is a serious and widespread problem, yet key uncertainties remain concerning its extent, severity, and linkages to climate change *(very high confidence)*. Despite the difficulties of objectively measuring the extent and severity of land degradation given its complex and value-based characteristics, land degradation represents, like climate change, one of the biggest and most urgent challenges for humanity (*very high confidence*). The current global extent, severity and rates of land degradation are not well quantified. There is no single method by which land degradation can be measured objectively and consistently over large areas because it is such a complex and value laden concept (*very high confidence*). However, many scientific and locally-based approaches, including the use of indigenous and local knowledge, exist that can assess different aspects of land degradation or provide proxies. Remote sensing, corroborated by other data, can generate geographically explicit and globally consistent data that can be used as proxies over relevant time scales (several decades). Few studies have specifically addressed the impacts of proposed land-based negative emission technologies on land degradation. Much research has tried to understand how livelihoods and ecosystems are affected by a particular stressor, for example drought, heat stress, or water logging. Important knowledge gaps remain in understanding how plants, habitats and ecosystems are affected by the cumulative and interacting impacts of several stressors, including potential new stressors resulting from large-scale implementation of negative emission technologies. {4.10}

4.1 Introduction

4.1.1 Scope of the chapter

This chapter examines the scientific understanding of how climate change impacts land degradation, and vice versa, with a focus on non-drylands. Land degradation of drylands is covered in Chapter 3. After providing definitions and the context (Section 4.1) we proceed with a theoretical explanation of the different processes of land degradation and how they are related to climate and to climate change, where possible (Section 4.2). Two sections are devoted to a systematic assessment of the scientific literature on status and trend of land degradation (Section 4.3) and projections of land degradation (Section 4.4). Then follows a section where we assess the impacts of climate change mitigation options, bioenergy and land-based technologies for carbon dioxide removal (CDR), on land degradation (Section 4.5). The ways in which land degradation can impact climate and climate change are assessed in Section 4.6. The impacts of climate related land degradation on human and natural systems are assessed in Section 4.7. The remainder of the chapter assesses land degradation mitigation options based on the concept of sustainable land management: avoid, reduce and reverse land degradation (Section 4.8), followed by a presentation of eight illustrative case studies of land degradation and remedies (Section 4.9). The chapter ends with a discussion of the most critical knowledge gaps and areas for further research (Section 4.10).

4.1.2 Perspectives of land degradation

Land degradation has accompanied humanity at least since the widespread adoption of agriculture during Neolithic time, some 10,000 to 7,500 years ago (Dotterweich 2013; Butzer 2005; Dotterweich 2008) and the associated population increase (Bocquet-Appel 2011). There are indications that the levels of greenhouse gases (particularly carbon dioxide and methane) of the atmosphere started to increase already more than 3,000 years ago as a result of expanding agriculture, clearing of forests, and domestication of wild animals (Fuller et al. 2011; Kaplan et al. 2011; Vavrus et al. 2018; Ellis et al. 2013). While the development of agriculture (cropping and animal husbandry) underpinned the development of civilisations, political institutions, and prosperity, farming practices led to conversion of forests and grasslands to farmland, and the heavy reliance on domesticated annual grasses for our food production meant that soils started to deteriorate through seasonal mechanical disturbances (Turner et al. 1990; Steffen et al. 2005; Ojima et al. 1994; Ellis et al. 2013). More recently, urbanisation has significantly altered ecosystems, see further Cross-chapter Box 4 on Climate Change and Urbanisation, Chapter 2. Since about 1850, about 35% of the human caused emissions of CO_2 to the atmosphere comes from land as a combined effect of land degradation and land-use change (Foley et al. 2005) and about 38% of Earth's land area has been converted to agriculture (Foley et al. 2011), see Chapter 2 for more details.

Not all human impacts on land result in degradation according to the definition of land degradation used in this report (see 4.2.1). There are many examples of long-term sustainably managed land around the world (such as terraced agricultural systems and sustainably managed forests) although degradation and its management are the focus of this chapter. We also acknowledge that human use of land and ecosystems provides essential goods and services for society (Foley et al. 2005; MEA (Millennium Ecosystem Assessment) 2005).

Land degradation was long subject to a polarised scientific debate between disciplines and perspectives in which social scientists often proposed that natural scientists exaggerated land degradation as a global problem (Blaikie and Brookfield 1987; Forsyth 1996; Lukas 2014; Zimmerer 1993). The elusiveness of the concept in combination with the difficulties of measuring and monitoring land degradation at global and regional scales by extrapolation and aggregation of empirical studies at local scales, such as the Global Assessment of Soil Degradation database

(GLASOD) (Sonneveld and Dent 2009) contributed to conflicting views. The conflicting views were not confined to science only, but also caused tension between the scientific understanding of land degradation and policy (Andersson et al. 2011; Behnke and Mortimore 2016; Grainger 2009; Toulmin and Brock 2016). Another weakness of many land degradation studies is the exclusion of the views and experiences of the land users, whether farmers or forest dependent communities (Blaikie and Brookfield 1987; Fairhead and Scoones 2005; Warren 2002; Andersson et al. 2011). More recently, the polarised views described above have been reconciled under the umbrella of Land Change Science, which has emerged as an interdisciplinary field aimed at examining the dynamics of land cover and land-use as a coupled human–environment system (Turner et al. 2007). A comprehensive discussion about concepts and different perspectives of land degradation was presented in Chapter 2 of the recent report from the Intergovernmental Platform on Biodiversity and Ecosystem Services (IPBES) on land degradation (Montanarella et al. 2018).

In summary, agriculture and clearing of land for food and wood products have been the main drivers of land degradation for millennia (*high confidence*). This does not mean, however, that agriculture and forestry always cause land degradation (*high confidence*); sustainable management is possible but not always practiced (*high confidence*). Reasons for this are primarily economic, political and social.

4.1.3 Definition of land degradation

To clarify the scope of this chapter it is important to start by defining land itself. The Special Report on Climate Change and Land (SRCCL) defines land as "the terrestrial portion of the biosphere that comprises the natural resources (soil, near surface air, vegetation and other biota, and water), the ecological processes, topography, and human settlements and infrastructure that operate within that system" (Henry et al. 2018), adapted from (FAO 2007; UNCCD 1994).

Land degradation is defined in many different ways within the literature, with differing emphases on biodiversity, ecosystem functions and ecosystem services (e.g., Montanarella et al. 2018). In this report, land degradation is defined as a *negative trend in land condition, caused by direct or indirect human-induced processes including anthropogenic climate change, expressed as long-term reduction or loss of at least one of the following: biological productivity, ecological integrity or value to humans*. This definition applies to forest and non-forest land: forest degradation is land degradation that occurs in forest land. Soil degradation refers to a subset of land degradation processes that directly affect soil.

The SRCCL definition is derived from the IPCC AR5 definition of desertification, which is in turn taken from the UNCCD: "Land degradation in arid, semi-arid, and dry sub-humid areas resulting from various factors, including climatic variations and human activities. Land degradation in arid, semi-arid, and dry sub-humid areas is a reduction or loss of the biological or economic productivity and integrity of rainfed cropland, irrigated cropland, or range, pasture, forest, and woodlands resulting from land uses or from a process or combination of processes, including processes arising from human activities and habitation patterns, such as (1) soil erosion caused by wind and/or water; (2) deterioration of the physical, chemical, biological, or economic properties of soil; and (3) long-term loss of natural vegetation" (UNCCD 1994, Article 1).

The SRCCL definition is intended to complement the more detailed UNCCD definition, expanding the scope to all regions, not just drylands, providing an operational definition that emphasises the relationship between land degradation and climate for use in this report. Through its attention to the three aspects biological productivity, ecological integrity and value to humans, the SRCCL definition is consistent with the Land Degradation Neutrality (LDN) concept, which aims to maintain or enhance the land-based natural capital, and the ecosystem services that flow from it (Cowie et al. 2018).

In the SRCCL definition of land degradation, changes in land condition resulting solely from natural processes (such as volcanic eruptions and tsunamis) are not considered land degradation, as these are not direct or indirect human-induced processes. Climate variability exacerbated by human-induced climate change can contribute to land degradation. Value to humans can be expressed in terms of ecosystem services or Nature's Contribution to People.

The definition recognises the reality presented in the literature that land-use and land management decisions often result in trade-offs between time, space, ecosystem services, and stakeholder groups (e.g. Dallimer and Stringer 2018). The interpretation of a negative trend in land condition is somewhat subjective, especially where there is a trade-off between ecological integrity and value to humans. The definition also does not consider the magnitude of the negative trend or the possibility that a negative trend in one criterion may be an acceptable trade-off for a positive trend in another criterion. For example, reducing timber yields to safeguard biodiversity by leaving on site more wood that can provide habitat, or vice versa, is a trade-off that needs to be evaluated based on context (i.e. the broader landscape) and society's priorities. Reduction of biological productivity *or* ecological integrity *or* value to humans *can* constitute degradation, but any one of these changes need not necessarily be considered degradation. Thus, a land-use change that reduces ecological integrity and enhances **sustainable** food production at a specific location is not necessarily degradation. Different stakeholder groups with different world views value ecosystem services differently. As Warren (2002) explained: land degradation is contextual. Further, a decline in biomass carbon stock does not always signify degradation, such as when caused by periodic forest harvest. Even a decline in productivity may not equate to land degradation, such as when a high intensity agricultural system is converted to a lower input more sustainable production system.

In the SRCCL definition, degradation is indicated by a negative trend in land condition during the period of interest, thus the baseline is the land condition at the start of this period. The concept of baseline is theoretically important but often practically difficult to implement for conceptual and methodological reasons (Herrick et al. 2019; Prince et al. 2018; see also Sections 4.3.1 and 4.4.1). Especially in biomes characterised by seasonal and interannual variability, the baseline values of the indicators to be assessed should be determined by averaging data over a number of years prior to the commencement of the assessment period (Orr et al. 2017; see also 4.2.4).

Forest degradation is land degradation in forest remaining forest. In contrast, deforestation refers to the conversion of forest to non-forest that involves a loss of tree cover and a change in land-use. Internationally accepted definitions of forest (FAO 2015; UNFCCC 2013) include lands where tree cover has been lost temporarily, due to disturbance or harvest, with an expectation of forest regrowth. Such temporary loss of forest cover therefore is not deforestation.

4.1.4 Land degradation in previous IPCC reports

Several previous IPCC assessment reports include brief discussions of land degradation. In AR5 WGIII land degradation is one factor contributing to uncertainties of the mitigation potential of land-based ecosystems, particularly in terms of fluxes of soil carbon (Smith et al., 2014, p. 817). In AR5 WGI, soil carbon was discussed comprehensively but not in the context of land degradation, except forest degradation (Ciais et al. 2013) and permafrost degradation (Vaughan et al. 2013). Climate change impacts were discussed comprehensively in AR5 WGII, but land degradation was not prominent. Land use and land cover changes were treated comprehensively in terms of effects on the terrestrial carbon stocks and flows (Settele et al. 2015) but links to land degradation were to a large extent missing. Land degradation was discussed in relation to human security as one factor which in combination with extreme weather events has been proposed to contribute to human migration (Adger et al. 2014), an issue discussed more comprehensively in this chapter (see section 4.7.3). Drivers and processes of degradation by which land-based carbon is released to the atmosphere and/or the long-term reduction in the capacity of the land to remove atmospheric carbon and to store this in biomass and soil carbon, have been discussed in the methodological reports of IPCC (IPCC 2006, 2014a) but less so in the assessment reports.

The Special Report on Land Use, Land-Use Change and Forestry (SR-LULUCF) (Watson et al. 2000) focused on the role of the biosphere in the global cycles of greenhouse gases (GHG). Land degradation was not addressed in a comprehensive way. Soil erosion was discussed as a process by which soil carbon is lost and the productivity of the land is reduced. Deposition of eroded soil carbon in marine sediments was also mentioned as a possible mechanism for permanent sequestration of terrestrial carbon (Watson et al. 2000) (p. 194). The possible impacts of climate change on land productivity and degradation were not discussed comprehensively. Much of the report was about how to account for sources and sinks of terrestrial carbon under the Kyoto Protocol.

The IPCC Special Report on Managing the Risks of Extreme Events and Disasters to Advance Climate Change Adaptation (SREX) (IPCC 2012) did not provide a definition of land degradation. Nevertheless, it addressed different aspects related to some types of land degradation in the context of weather and climate extreme events. From this perspective, it provided key information on both observed and projected changes in weather and climate (extremes) events that are relevant to extreme impacts on socio-economic systems and on the physical components of the environment, notably on permafrost in mountainous areas and coastal zones for different geographic regions, but little explicit links to land degradation. The report also presented the concept of sustainable land management as an effective risk reduction tool.

Land degradation has been treated in several previous IPCC reports but mainly as an aggregated concept associated with emissions of GHG or as an issue that can be addressed through adaptation and mitigation.

4.1.5 Sustainable land management and sustainable forest management

Sustainable land management (SLM) is defined as "the stewardship and use of land resources, including soils, water, animals and plants, to meet changing human needs, while simultaneously ensuring the long-term productive potential of these resources and the maintenance of their environmental functions" (Adapted from World Overview of Conservation Approaches and Technologies, WOCAT). Achieving the objective of ensuring that productive potential is maintained in the long term will require implementation of adaptive management and "triple loop learning", that seeks to monitor outcomes, learn from experience and emerging new knowledge, modifying management accordingly (Rist et al. 2013).

Sustainable Forest Management (SFM) is defined as "the stewardship and use of forests and forest lands in a way, and at a rate, that maintains their biodiversity, productivity, regeneration capacity, vitality and their potential to fulfill, now and in the future, relevant ecological, economic and social functions, at local, national, and global levels, and that does not cause damage to other ecosystems" (Forest Europe 2016; Mackey et al. 2015). This SFM definition was developed by the Ministerial Conference on the Protection of Forests in Europe and has since been adopted by the Food and Agriculture Organization. Forest management that fails to meet these sustainability criteria can contribute to land degradation. Land degradation can be reversed through restoration and rehabilitation, which are defined in the Glossary, where other terms that are used but not explicitly defined in this section can also be found. While the definitions of SLM and SFM are very similar and could be merged, both are included to maintain the subtle differences in the existing definitions.

Climate change impacts interact with land management to determine sustainable or degraded outcome (Figure 4.1). Climate change can exacerbate many degradation processes (Table 4.1) and introduce novel ones (e.g., permafrost thawing or biome shifts). To avoid, reduce or reverse degradation, land management activities can be selected to mitigate the impact of, and adapt to, climate change. In some cases, climate change impacts may result in increased productivity and carbon stocks, at least in the short term. For example, longer growing seasons due to climate warming can lead to higher forest productivity (Henttonen et al. 2017; Kauppi et al. 2014; Dragoni et al. 2011), but warming alone many not increase productivity where other factors such a water supply are limiting (Hember et al. 2017).

The types and intensity of human land-use and climate change impacts on lands affect their carbon stocks and their ability to operate as carbon sinks. In managed agricultural lands, degradation can result in reductions of soil organic carbon stocks, which also adversely affects land productivity and carbon sinks (See Figure 4.1).

The transition from natural to managed forest landscapes usually results in an initial reduction of landscape-level carbon stocks. The magnitude of this reduction is a function of the differential in frequency of stand replacing natural disturbances (e.g. wildfires) and harvest disturbances, as well as the age-dependence of these disturbances (Harmon et al. 1990; Kurz et al. 1998a; Trofymow et al. 2008).

Sustainable forest management applied at the landscape scale to existing unmanaged forests can first reduce average forest carbon stocks and subsequently increase the rate at which carbon dioxide is removed from the atmosphere, because net ecosystem production of forest stands is highest in intermediate stand ages (Kurz et al. 2013; Volkova et al. 2018; Tang et al. 2014). The net impact on the atmosphere depends on the magnitude of the reduction in carbon stocks, the fate of the harvested biomass (i.e. use in short or long-lived products and for bioenergy, and therefore displacement of emissions associated with GHG-intensive building materials and fossil fuels), and the rate of regrowth. Thus, the impacts of sustainable forest management on one indicator (e.g., past reduction in C stocks in the forested landscape) can be negative, while those on another indicator (e.g., current forest productivity and rate of CO_2 removal from the atmosphere, avoided fossil fuel emissions) can be positive. Sustainably managed forest landscapes can have a lower biomass carbon density than unmanaged forest, but the younger forests can have a higher growth rate, and therefore contribute stronger carbon sinks, than older forests (Trofymow et al. 2008; Volkova et al. 2018; Poorter et al. 2016).

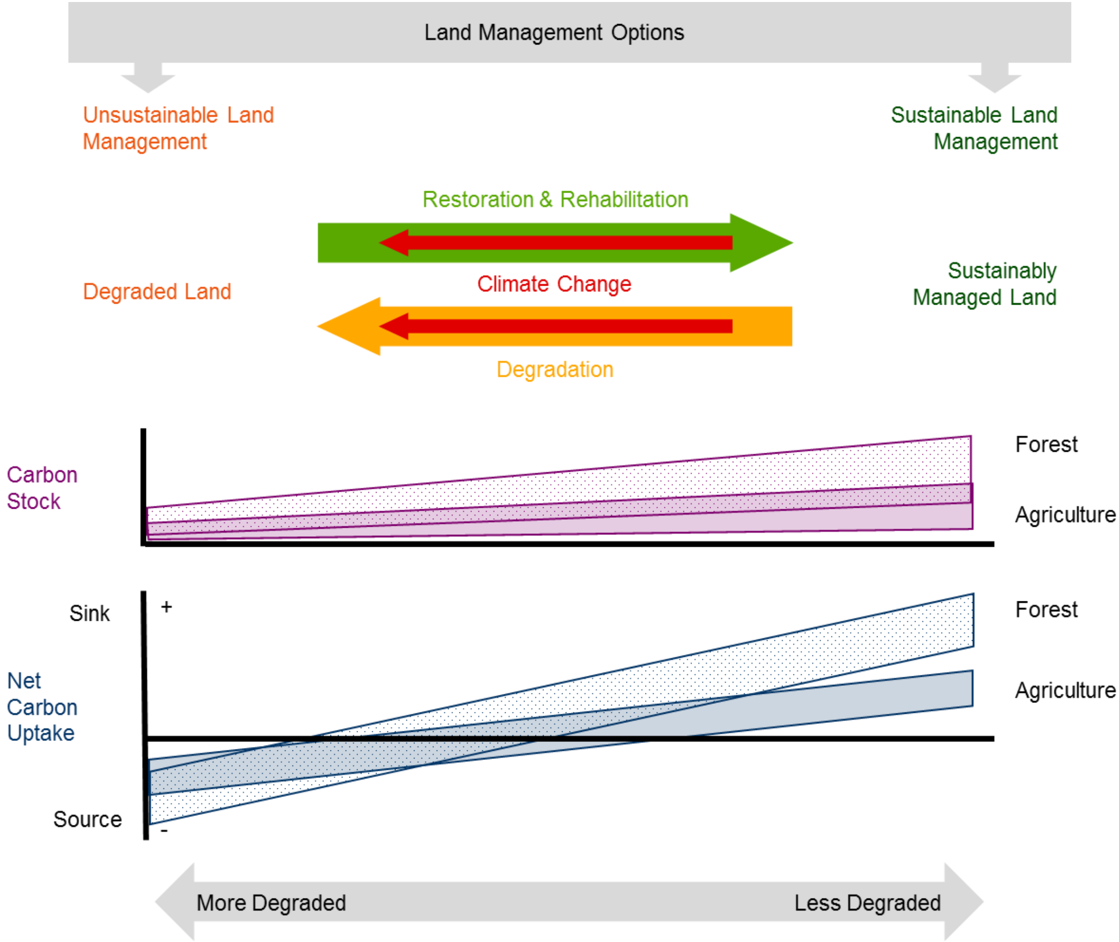


Figure 4.1 Conceptual figure illustrating that climate change impacts interact with land management to determine sustainable or degraded outcome. Climate change can exacerbate many degradation processes (Table 4.1) and introduce novel ones (e.g., permafrost thawing or biome shifts), hence management needs to respond to climate impacts in order to avoid, reduce or reverse degradation. The types and intensity of human land-use and climate change impacts on lands affect their carbon stocks and their ability to operate as carbon sinks. In managed agricultural lands, degradation typically results in reductions of soil organic carbon stocks, which also adversely affects land productivity and carbon sinks. In forest land, reduction in biomass carbon stocks alone is not necessarily an indication of a reduction in carbon sinks. Sustainably managed forest landscapes can have a lower biomass carbon density but the younger forests can have a higher growth rate, and therefore contribute stronger carbon sinks, than older forests. Ranges of carbon sinks in forest and agricultural lands are overlapping. In some cases, climate change impacts may result in increased productivity and carbon stocks, at least in the short term.

Selective logging and thinning can maintain and enhance forest productivity and achieve co-benefits when conducted with due care for the residual stand and at intensity and frequency that does not exceed the rate of regrowth (Romero and Putz 2018). In contrast, unsustainable logging practices can lead to stand-level degradation. For example, degradation occurs when selective logging (high-grading) removes valuable large-diameter trees, leaving behind damaged, diseased, non-commercial or otherwise less productive trees, reducing carbon stocks and also adversely affecting subsequent forest recovery (Belair and Ducey 2018; Nyland 1992).

Sustainable forest management is defined using several criteria (see above) and its implementation will typically involve trade-offs among these criteria. The conversion of primary forests to sustainably managed forest ecosystems increases relevant economic, social and other functions but often with adverse impacts on biodiversity (Barlow et al. 2007). In regions with infrequent or no stand replacing natural disturbances, the timber yield per hectare harvested in managed secondary forests is typically lower than the yield per hectare from the first harvest in the primary forest (Romero and Putz 2018).

The sustainability of timber yield has been achieved in temperate and boreal forests where intensification of management has resulted in increased growing stocks and increased harvest rates in countries where forests had previously been overexploited (Henttonen et al. 2017; Kauppi et al. 2018). However, intensification of management to increase forest productivity can be associated with reductions in biodiversity. For example, when increased productivity is achieved by periodic thinning and removal of trees that would otherwise die due to competition, thinning reduces the amount of dead organic matter of snags and coarse woody debris that can provide habitat and this loss reduces biodiversity (Spence 2001; Ehnström 2001) and forest carbon stocks (Russell et al. 2015; Kurz et al. 2013). Recognition of adverse biodiversity impacts of high yield forestry is leading to modified management aimed at increasing habitat availability through, for example, variable retention logging and continuous cover management (Roberts et al. 2016) and through the re-introduction of fire disturbances in landscapes where fires have been suppressed (Allen et al. 2002). Biodiversity losses are also observed during the transition from primary to managed forests in tropical regions (Barlow et al. 2007) where tree species diversity can be very high, e.g. in the Amazon region about 16,000 tree species are estimated to exist (ter Steege et al. 2013).

Forest certification schemes have been used to document SFM outcomes (Rametsteiner and Simula 2003) by assessing a set of criteria and indicators (e.g., Lindenmayer et al. 2000). While many of the certified forests are found in temperate and boreal countries (Rametsteiner and Simula 2003; MacDicken et al. 2015), examples from the tropics also show that SFM can improve outcomes. For example, selective logging emits 6% of the tropical GHG annually and improved logging practices can reduce emissions by 44 % while maintaining timber production (Ellis et al. 2019). In the Congo Basin, implementing reduced impact logging (RIL-C) practices can cut emissions in half without reducing the timber yield (Umunay et al. 2019). SFM adoption depends on the socio-economic and political context and its improvement depends mainly on better reporting and verification (Siry et al. 2005).

The successful implementation of SFM requires well established and functional governance, monitoring, and enforcement mechanisms to eliminate deforestation, illegal logging, arson, and other activities that are inconsistent with SFM principles (Nasi et al. 2011). Moreover, following human and natural disturbances forest regrowth must be ensured through reforestation, site rehabilitation activities or natural regeneration. Failure of forests to regrow following disturbances will lead to unsustainable outcomes and long-term reductions in forest area, forest cover, carbon density, forest productivity and land-based carbon sinks (Nasi et al. 2011).

Achieving all of the criteria of the definitions of SLM and SFM is an aspirational goal that will be made more challenging where climate change impacts, such as biome shifts and increased disturbances, are predicted to adversely affect future biodiversity and contribute to forest degradation (Warren et al. 2018). Land management to enhance land sinks will involve trade-offs that need to be assessed within their spatial, temporal and societal context.

4.1.6 The human dimension of land degradation and forest degradation

Studies of land and forest degradation are often biased towards biophysical aspects both in terms of its processes, such as erosion or nutrient depletion, and its observed physical manifestations, such as gullying or low primary productivity. Land users' own perceptions and knowledge about land conditions and degradation have often been neglected or ignored by both policy makers and scientists (Reed et al. 2007; Forsyth 1996; Andersson et al. 2011). A growing body of work is nevertheless beginning to focus on land degradation through the lens of local land users (Kessler and Stroosnijder 2006; Fairhead and Scoones 2005; Zimmerer 1993; Stocking et al. 2001) and the importance of local and indigenous knowledge within land management is starting to be appreciated (Montanarella et al. 2018). Climate change impacts directly and indirectly the social reality, the land users, and the ecosystem and vice versa. Land degradation can also have an impact on climate change (see Section 4.6).

The use and management of land is highly gendered and is expected to remain so for the foreseeable future (Kristjanson et al. 2017). Women have often less formal access to land than men and less influence over decisions about land, even if they carry out many of the land management tasks (Jerneck 2018a; Elmhirst 2011; Toulmin 2009; Peters 2004; Agarwal 1997; Jerneck 2018b). Many oft-cited sweeping statements about women's subordination in agriculture are difficult to substantiate, yet it is clear that gender inequality persists (Doss et al. 2015). Even if women's access to land is changing formally (Kumar and Quisumbing 2015), the practical outcome is often limited due to several other factors related to both formal and informal institutional arrangements and values (Lavers 2017; Kristjanson et al. 2017; Djurfeldt et al. 2018). Women are also affected differently than men when it comes to climate change, having lower adaptive capacities due to factors such as prevailing land tenure frameworks, lower access to other capital assets and dominant cultural practices (Vincent et al. 2014; Antwi-Agyei et al. 2015; Gabrielsson et al. 2013). This affects the options available to women to respond to both land degradation and climate change. Indeed, access to land and other assets (e.g., education and training) is key in shaping land-use and land management strategies (Liu et al. 2018b; Lambin et al. 2001). Young people is another category that is often disadvantaged in terms of access to resources and decision making power, even though they carry out much of the day-to-day work (Wilson et al. 2017; Kosec et al. 2018; Naamwintome and Bagson 2013).

Land rights differ between places and are dependent on the political-economic and legal context (Montanarella et al. 2018). This means there is no universally applicable best arrangement. Agriculture in highly erosion prone regions requires site specific and long lasting soil and water conservation measures, such as terraces (see 4.8.1), which may benefit from secure private land rights (Tarfasa et al. 2018; Soule et al. 2000). Pastoral modes of production and community based forest management systems are often dominated by communal land tenure arrangements, which may conflict with agricultural/forestry modernization policies implying private property rights (Antwi-Agyei et al. 2015; Benjaminsen and Lund 2003; Itkonen 2016; Owour et al. 2011; Gebara 2018)

Cultural ecosystem services, defined as the non-material benefits people obtain from ecosystems through spiritual enrichment, cognitive development, reflection, recreation and aesthetic experiences (Millennium Assessment 2005) are closely linked to land and ecosystems, although often underrepresented in the literature on ecosystem services (Tengberg et al. 2012; Hernández-Morcillo et al. 2013). Climate change interacting with land conditions can impact cultural aspects, such as sense of place and sense of belonging (Olsson et al. 2014).

4.2 Land degradation in the context of climate change

Land degradation results from a complex chain of causes making the clear distinction between direct and indirect drivers difficult. In the context of climate change, an additional complex aspect is brought by the reciprocal effects that both processes have on each other (i.e. climate change influencing land degradation and vice versa). In this chapter, we use the terms processes and drivers with the following meanings:

Processes of land degradation are those direct mechanisms by which land is degraded and are similar to the notion of "direct drivers" in the Millennium Ecosystem Assessment (MA, Millennium Ecosystem Assessment, 2005) framework. In this report, a comprehensive list of land degradation processes is presented in Table 4.1.

Drivers of land degradation are those indirect conditions which may drive processes of land degradation and are similar to the notion of "indirect drivers" in the MA framework. Examples of indirect drivers of land degradation are changes in land tenure or cash crop prices, which can trigger land-use or management shifts that affect land degradation.

An exact demarcation between processes and drivers is not possible. Drought and fires are described as drivers of land degradation in the next section but they can also be a process: for example, if repeated fires deplete seed sources they can affect regeneration and succession of forest ecosystems. The responses to land degradation follow the logic of the Land Degradation Neutrality concept: avoiding, reducing and reversing land degradation (Orr et al. 2017b; Cowie et al. 2018).

In research on land degradation, climate and climate variability are often intrinsic factors. The role of climate change, however, is less articulated. Depending on what conceptual framework is used, climate change is understood either as a process or a driver of land degradation, and sometimes both.

4.2.1 Processes of land degradation

A large array of interactive physical, chemical, biological and human processes led to what we define in this report as land degradation (Johnson and Lewis 2007). The biological productivity, ecological integrity (which encompasses both functional and structural attributes of ecosystems) or the human value (which includes any benefit that people get from the land) of a given territory can deteriorate as the result of processes triggered at scales that range from a single furrow (e.g., water erosion under cultivation) to the landscape level (e.g., salinisation through raising groundwater levels under irrigation). While pressures leading to land degradation are often exerted on specific components of the land systems (i.e., soils, water, biota), once degradation processes start, other components become affected through cascading and interactive effects. For example, different pressures and degradation processes can have convergent effects, as can be the case of overgrazing leading to wind erosion, landscape drainage resulting in wetland drying, and warming causing more frequent burning; all of which can independently lead to reductions of the soil organic matter pools as second order process. Still, the reduction of organic matter pools is also a first order process triggered directly by the effects of rising temperatures (Crowther et al., 2016) as well as other climate changes such as precipitation shifts (Viscarra Rossel et al. 2014). Beyond this complexity, a practical assessment of the major land degradation processes helps to reveal and categorise the multiple pathways in which climate change exerts a degradation pressure (Table 4.1).

Conversion of freshwater wetlands to agricultural land has historically been a common way of increasing the area of arable land. Despite the small areal extent (~1% of the earth's surface (Hu et al. 2017; Dixon et al. 2016)), freshwater wetlands provide a very large number of ecosystem services, such as groundwater replenishment, flood protection, and nutrient retention, and are biodiversity hotspots (Reis et al. 2017; Darrah et al. 2019; Montanarella et al. 2018). The loss of wetlands since 1900 has been estimated at ~55% globally (Davidson 2014) (*low confidence*) and 35% since 1970 (Darrah et al. 2019) (*medium confidence*) which in many situations pose a problem for adaptation to climate change. Drainage causes loss of wetlands, which can be further exacerbated by climate change, and reduces the capacity to adapt to climate change (Barnett et al. 2015; Colloff et al. 2016; Finlayson et al. 2017) (*high confidence*).

4.2.1.1 Types of land degradation processes

Land degradation processes can affect the soil, water or biotic components of the land or in their respective interfaces (Table 4.1). Across land degradation processes, those affecting the soil have received more attention. The most widespread and studied land degradation processes affecting soils are water and wind erosion, which have accompanied agriculture since its onset and are still dominant (Table 4.1). Degradation through erosion processes is not restricted to soil loss in detachment areas but includes impacts on transport and deposition areas as well (less commonly, deposition areas can have their soils improved by these inputs). Larger scale degradation processes related to the whole continuum of soil erosion, transport and deposition include dune field expansion/displacement, development of gully networks and the accumulation of sediments (siltation) of natural and artificial water bodies (Poesen and Hooke 1997; Ravi et al. 2010). Long-distance sediment transport during erosion events can have remote effects on land systems as documented for the fertilisation effect of African dust on the Amazon (Yu et al. 2015).

Coastal erosion represents a special case among erosional, with reports linking it to climate change. While human interventions in coastal areas (e.g., expansion of shrimp farms) and rivers (e.g., upstream dams cutting coastal sediment supply), and economic activities causing land subsidence (Keogh and Törnqvist 2019; Allison et al. 2016) are dominant human drivers, storms and sea level rise have already left a significant global imprint on coastal erosion (Mentaschi et al. 2018). Recent projections that take into account geomorphological and socioecological feedbacks suggest that coastal wetlands may not get reduced by sea level rise if their inland growth is accommodated with proper management actions (Schuerch et al. 2018a).

Other physical degradation process in which no material detachment and transport are involved include soil compaction, hardening, sealing and any other mechanism leading to the loss of porous space crucial for holding and exchanging air and water (Hamza and Anderson 2005). A very extreme case of degradation through pore volume loss, manifested at landscape or larger scales, is ground subsidence. Typically caused by the lowering of groundwater or oil levels, subsidence involves a sustained collapse of the ground surface, which can lead to other degradation processes such as salinisation and permanent flooding. Chemical soil degradation processes include relatively simple changes, like nutrient depletion resulting from the imbalance of nutrient extraction on harvested products and fertilisation, and more complex ones, such as acidification and increasing metal toxicity. Acidification in croplands is increasingly driven by excessive nitrogen fertilisation and to a lower extent by the depletion of cation like calcium, potassium or magnesium through exports in harvested biomass (Guo et al. 2010). One of the most relevant chemical degradation processes of soils in the context of climate change is the depletion of its organic matter pool. Reduced in agricultural soils through the increase of respiration rates by tillage and the decline of belowground plant biomass inputs, soil organic matter pools have been diminished also by the direct effects of warming, not only in cultivated land but also under natural vegetation (Bond-Lamberty et al. 2018). Debate persists, however, on whether in more humid and carbon rich ecosystems the simultaneous stimulation of decomposition and productivity may result in the lack of effects on soil carbon (Crowther et al. 2016; van Gestel et al. 2018). In the case of forests, harvesting, particularly if it is exhaustive as in the case of the use of residues for energy generation, can also lead to organic matter declines (Achat et al. 2015). Affected by many other degradation processes (e.g. wildfire increase, salinisation) and having negative effects on other pathways of soil degradation (e.g. reduced nutrient availability, metal toxicity). Soil organic matter can be considered a “hub” of degradation processes and a critical link with the climate system (Minasny et al. 2017).

Land degradation processes can also start from alterations in the hydrological system that are particularly important in the context of climate change. Salinisation, although perceived and reported in soils, is typically triggered by water table-level rises driving salts to the surface under dry to sub-humid climates (Schofield and Kirkby 2003). While salty soils occur naturally under these climates (primary salinity), human interventions have expanded their distribution (secondary salinity with irrigation without proper drainage being the predominant cause of salinisation (Rengasamy 2006). Yet, it has also taken place under non-irrigated conditions where vegetation changes (particularly dry forest clearing and cultivation) had reduced the magnitude and depth of soil water uptake, triggering water table rises towards the surface. Changes in evapotranspiration and rainfall regimes can exacerbate this process (Schofield and Kirkby 2003). Salinisation can also result from the intrusion of sea water into coastal areas both as a result of sea level rise and ground subsidence (Colombani et al. 2016).

Recurring flood and waterlogging episodes (Bradshaw et al. 2007; Poff 2002), and the more chronic expansion of wetlands over dryland ecosystems are mediated by the hydrological system, on occasions aided by geomorphological shifts as well (Kirwan et al. 2011). This is also the case for the drying of continental water bodies and wetlands, including the salinisation and drying of lakes and inland seas (Anderson et al. 2003; Micklin 2010; Herbert et al. 2015). In the context of climate change, the degradation of peatland ecosystems is particularly relevant given their very high carbon storage and their sensitivity to changes in soils, hydrology and/or vegetation (Leifeld and Menichetti 2018). Drainage for land-use conversion together with peat mining are major drivers of peatland degradation, yet other factors such as the extractive use of their natural vegetation and the interactive effects of water table levels and fires (both sensitive to climate change) are important (Hergoualc'h et al. 2017a; Lilleskov et al. 2019).

The biotic components of the land can also be the focus of degradation processes. Vegetation clearing processes associated with land-use changes are not limited to deforestation but include other natural and seminatural ecosystems such as grasslands (the most cultivated biome on Earth), as well as dry steppes and shrublands, which give place to croplands, pastures, urbanisation or just barren land. This clearing process is associated with net C losses from the vegetation and soil pool. Not all biotic degradation processes involve biomass losses. Woody encroachment of open savannahs involve the expansion of woody plant cover and/or density over herbaceous areas and often limits the secondary productivity of rangelands (Asner et al. 2004, Anadon et al. 2014). These processes have been accelerated since the mid-1800s over most continents (Van Auken 2009). Change in plant composition of natural or semi-natural ecosystems without any significant vegetation structural changes is another pathway of degradation affecting rangelands and forests. In rangelands, selective grazing and its interaction with climate variability and/or fire can push ecosystems to new compositions with lower forage value and higher proportion of invasive species (Illius and O´Connor 1999, Sasaki et al. 2007), in some cases with higher carbon sequestration potential, yet with very complex interactions between vegetation and soil carbon shifts (Piñeiro et al. 2010). In forests, extractive logging can be a pervasive cause of degradation leading to long-term impoverishment and in extreme cases, a full loss of the forest cover through its interaction with other agents such as fires (Foley et al. 2007) or progressive intensification of land use. Invasive alien species are another source of biological degradation. Their arrival into cultivated systems is constantly reshaping crop production strategies making agriculture unviable on occasions. In natural and seminatural systems such as rangelands, invasive plant species not only threaten livestock production through diminished forage quality, poisoning and other deleterious effects, but have cascading effects on other processes such as altered fire regimes and water cycling (Brooks et al. 2004). In forests, invasions affect primary productivity and nutrient availability, change fire regimes, and alter species composition, resulting in long term impacts on carbon pools and fluxes (Peltzer et al. 2010).

Other biotic components of ecosystems have been shown as a focus of degradation processes. Invertebrate invasions in continental waters can exacerbate other degradation processes such as eutrophication, which is the over enrichment of nutrients leading to excessive algal growth (Walsh et al. 2016a). Shifts in soil microbial and mesofaunal composition, which can be caused by pollution with pesticides or nitrogen deposition but also by vegetation or disturbance regime shifts, alter many soil functions including respiration rates and C release to the atmosphere (Hussain et al. 2009; Crowther et al. 2015). The role of the soil biota modulating the effects of climate change on soil carbon have been recently demonstrated (Ratcliffe et al. 2017), highlighting the importance of this less known component of the biota as a focal point of land degradation. Of special relevance as both indicators and agents of land degradation recovery are mycorrhiza, which are root associated fungal organisms (Asmelash et al. 2016; Vasconcellos et al. 2016). In natural dry ecosystems, biological soil crusts composed by a broad range of organisms including mosses are a particularly sensitive focus for degradation (Field et al. 2010) with evidenced sensitivity to climate change (Reed et al. 2012).

4.2.1.2 Land degradation processes and climate change

While the subdivision of individual processes is challenged by their strong interconnectedness, it provides a useful setting to identify the most important "focal points" of climate change pressures on land degradation. Among land degradation processes those responding more directly to climate change pressures include all types of erosion and soil organic matter declines (soil focus), salinisation, sodification and permafrost thawing (soil/water focus), waterlogging of dry ecosystems and drying of wet ecosystems (water focus), and a broad group of biological mediated processes like woody encroachment, biological invasions, pest outbreaks (biotic focus), together with biological soil crust destruction and increased burning (soil/biota focus) (Table 4.1). Processes like ground subsidence can be affected by climate change indirectly through sea level rise (Keogh and Törnqvist 2019).

Even when climate change exerts a direct pressure on degradation processes, it can be a secondary driver subordinated to other overwhelming human pressures. Important exceptions are three processes in which climate change is a dominant global or regional pressure and the main driver of their current acceleration. These are coastal erosion as affected by sea level rise and increased storm frequency/intensity (*high agreement, medium evidence*) (Johnson et al. 2015; Alongi 2015; Harley et al. 2017a; Nicholls et al. 2016), permafrost thawing responding to warming (*high agreement, robust evidence*) (Liljedahl et al. 2016; Peng et al. 2016; Batir et al. 2017) and increased burning responding to warming and altered precipitation regimes (*high agreement, robust evidence*) (Jolly et al. 2015; Abatzoglou and Williams 2016; Taufik et al. 2017; Knorr et al. 2016). The previous assessment highlights the fact that climate change not only exacerbates many of the well acknowledged ongoing land degradation processes of managed ecosystems (i.e., croplands and pastures), but becomes a dominant pressure that introduces novel degradation pathways in natural and seminatural ecosystems. Climate change has influenced species invasions and the degradation that they cause by enhancing the transport, colonisation, establishment, and ecological impact of the invasive species, but also by impairing their control practices (*medium agreement, medium evidence*) (Hellmann et al. 2008).

Table 4.1 Major land degradation processes and their connections with climate change. For each process a "focal point" (soil, water, biota) on which degradation occurs first place is indicated, acknowledging that most processes propagate to other land components and cascade into or interact with some of the other processes listed below. The impact of climate change on each process is categorised based on the proximity (very direct = high, very indirect=low) and dominance (dominant=high, subordinate to other pressures = low) of effects. The major effects of climate change on each process are highlighted together with the predominant pressures from other drivers. Feedbacks of land degradation processes on climate change are categorized according to the intensity (very intense=high, subtle=low) of the chemical (greenhouse gases emissions or capture) or physical (energy and momentum exchange, aerosol emissions) effects. Warming effects are indicated in red and cooling effects in blue. Specific feedbacks on climate change are highlighted.

Processes	**Focal point**	**Impacts of Climate Change**				**Feedbacks on Climate Change**			
		proximity	**dominance**	**Climate Change pressures**	**Other pressures**	**intensity of chemical effects**	**intensity of physical effects**	**global extent**	**Specific Impacts**
Wind erosion	Soil	**high**	**medium**	Altered wind/drought patterns (*high confidence* on effect, *medium-low confidence* on trend) (*1*). Indirect effect through vegetation type and biomass production shifts	Tillage, leaving low cover, overgrazing, deforestation/vegetation clearing, large plot sizes, vegetation and fire regime shifts	low	**medium**	**high**	Radiative cooling by dust release (*medium confidence*). Ocean and land fertilisation and C burial (*medium confidence*). Albedo increase. Dust effect as condensation nuclei (19).
Water erosion	Soil	**high**	**medium**	Increasing rainfall intensity (*high confidence* on effect and trend) (2). Indirect effects on fire frequency/intensity, permafrost thawing, biomass production.	Tillage, cultivation leaving low cover, overgrazing, deforestation/vegetation clearing, vegetation burning, poorly designed roads and paths.	**medium**	**medium**	**high**	Net C release. Net release is probably less than site-specific loss due to deposition and burial (*high confidence*). Albedo increase (20).
Coastal erosion	Soil/Water	**high**	**high**	Sea level rise, increasing intensity/frequency of storm surges (*high confidence* on effects and trends)(3)	Retention of sediments by upstream dams, Coastal aquiculture, Elimination of mangrove forests, Subsidence	**high**	low	low	Release of old buried C pools (*medium confidence*)(21).

Subsidence	Soil/Water	low	low	Indirect through increasing drought leading to higher ground water use. Indirect through enhanced decomposition (e.g. through drainage) in organic soils.	Groundwater depletion / overpumping. Peatland drainage.	**low/high**	low	low	Unimportant in the case of groundwater depletion. Very high net C release in the case of drained peatlands
Compaction/Hardening	Soil	low	low	Indirect through reduced organic matter content.	Land use conversion, machinery overuse, intensive grazing, poor tillage/grazing management (e.g. under wet or waterlogged conditions)	low	low	**medium**	Contradictory effects of reduced aeration on N_2O emissions
Nutrient depletion	Soil	low	low	Indirect (e.g. shifts in cropland distribution, BECCS)	Insufficient replenishment of harvested nutrients	low	low	**medium**	Net C release due shrinking SOC pools. Larger reliance on soil liming with associated CO_2 releases.
Acidification/Overfertilisation	Soil	low	low	Indirect (e.g. shifts in cropland distribution, BECCS). Sulfidic wetland drying due to increased drought as special direct effect.	High N fertilisation. High cation depletion. Acid rain/deposition	**medium**	low	**medium**	N_2O release from overfertilised soils, increased by acidification. Inorganic C release from acidifying soils (*medium to high confidence*) (22).
Pollution	Soil/Biota	low	low	Indirect (e.g. increased pest and weed incidence)	Intensifying chemical control of weed and pests	low	low	**medium**	Unknown, probably unimportant.
Organic matter decline	Soil	**high**	**medium**	Warming accelerates soil respiration rates (*medium confidence* on effects and trends) (4). Indirect effects through changing quality of plant litter or fire/waterlogging regimes.	Tillage. reduced plant input to soil. Drainage of waterlogged soils. Influenced by most of the other soil degradation processes.	**high**	low	**high**	Net C release (*high confidence*)(23).
Metal toxicity	Soil	low	low	Indirect	High cation depletion, fertilisation, mining activities	low	low	low	unknown, probably unimportant.

Salinisation	Soil / Water	High	low	Sea level rise (*high confidence* on effects and trends) (5). Water balance shifts (*medium confidence* on effects and trends) (6). Indirect effects through irrigation expansion.	Irrigation without good drainage infrastructure. Deforestation and water table level raises under dryland agriculture	low	medium	medium	Reduced methane emissions with high sulfate load. Albedo increase.
Sodification (increased sodium and associated physical degradation in soils)	Soil / Water	High	low	Water balance shifts (*medium confidence* on effects and trends) (7). Indirect effects through irrigation expansion.	Poor water management	low	medium	low	Net C release due to soil structure and organic matter dispersion. Albedo increase.
Permafrost thawing	Soil / Water	High	high	Warming (*very high confidence* on effects and trends) (8), seasonality shifts and accelerated snow melt leading to higher erosivity.		high	low	high	Net C release. CH_4 release (*high confidence*)(24).
Waterlogging of dry systems	Water	High	medium	Water balance shifts (*medium confidence* on effects and trends) (9). Indirect effects through vegetation shifts.	Deforestation. Irrigation without good drainage infrastructure	medium	medium	low	CH_4 release. Albedo decrease
Drying of continental waters/wetland/lowlands	Water	High	medium	Increasing extent and duration of drought (*high confidence* on effects, *medium confidence* on trends) (10). Indirect effects through vegetation shifts.	Upstream surface and groundwater water consumption. Intentional drainage. Trampling/overgrazing.	medium	medium	medium	Net C release. N_2O release. Albedo increase
Flooding	Water	High	medium	Sea level raise, increasing intensity/frequency of storm surges, increasing rainfall intensity causing flash floods (*high confidence* on effects and trends)(11).	Land clearing. Increasing impervious surface. Transport infrastructure.	medium	medium	low	CH_4 and N_2O release. Albedo decrease
Eutrophication of continental waters	Water/Biota	Low	low	Indirect through warming effects on N losses from the land or climate change effects on erosion rates. Interactive effects of warming and nutrient loads on algal blooms.	Excess fertilisation. Erosion. Poor management of livestock/human sewage.	medium	low	low	CH_4 and N_20 release.

Woody encroachment	Biota	**High**	**medium**	Rainfall shifts (*medium confidence* on effects and trends), CO_2 rise (*medium confidence* on effects, *very high confidence* on trends)(12).	Overgrazing. Altered fire regimes, fire suppression. Invasive alien species.	**high**	**high**	**high**	Net C storage. Albedo decrease
Species loss, compositional shifts	Biota	**High**	**medium**	Habitat loss as a result of climate shifts (*medium confidence* on effects and trends) (13).	Selective grazing and logging causing plant species loss, Pesticides causing soil microbial and soil faunal losses, Large animal extinctions, Interruption of disturbance regimes	low	low	medium	Unknown.
Soil microbial and mesofaunal shifts	Biota	**High**	low	Habitat loss as a result of climate shifts (*medium confidence* on effects and trends) (14).	Altered fire regimes, nitrogen deposition, pesticide pollution, vegetation shifts, disturbance regime shifts	low	low	medium	Unknown.
Biological soil crust destruction	Biota/Soil	**High**	**medium**	Warming. Changing rainfall regimes. (*medium confidence* on effects, high confidence and trends). Indirect through fire regime shifts and/or invasions (15).	Overgrazing and trampling. Land use conversion.	low	**high**	**high**	Radiative cooling through albedo rise and dust release (*high confidence*)(25).
Invasions	Biota	**High**	**medium**	Habitat gain as a result of climate shifts (*medium confidence* on effects and trends) (16).	Intentional and unintentional species introductions.	low	low	medium	Unknown.
Pest outbreaks	Biota	**High**	**medium**	Habitat gain and accelerated reproduction as a result of climate shifts (*medium confidence* on effects and trends) (17).	Large scale monocultures. Poor pest management practices.	**medium**	low	medium	Net C release.

Increased burning	Soil/Biota	**High**	**high**	Warming, drought, shifting precipitation regimes, also wet spells rising fuel load. (*high confidence* on effects and trends) (18).	Fire suppression policies increasing wildfire intensity. Increasing use of fire for rangeland management. Agriculture introducing fires in humid climates without previous fire history. Invasions.	**high**	**medium**	**medium**	Net C release. CO, CH_4, N_2O release. Albedo increase. (*high confidence*). Long term decline of NPP in non-adapted ecosystems (26).

References in table 4.1:

(1) (Bärring et al. 2003; Munson et al. 2011; Sheffield et al. 2012), (2) (Nearing et al. 2004b; Shakesby 2011; Panthou et al. 2014), (3) (Johnson et al. 2015; Alongi 2015; Harley et al. 2017b), (4) (Bond-Lamberty et al. 2018; Crowther et al. 2016; van Gestel et al. 2018), (5) (Colombani et al. 2016), (6) (Schofield and Kirkby 2003; Aragüés et al. 2015; Benini et al. 2016), (7) (Jobbágy et al. 2017), (8) (Liljedahl et al. 2016; Peng et al. 2016; Batir et al. 2017), (9) (Piovano et al. 2004; Osland et al. 2016), (10) (Burkett and Kusler 2000; Nielsen and Brock 2009; Johnson et al. 2015; Green et al. 2017), (11) (Panthou et al. 2014; Arnell and Gosling 2016; Vitousek et al. 2017), (12) (Van Auken 2009; Wigley et al. 2010), (13) (Vincent et al. 2014; Gonzalez et al. 2010; Scheffers et al. 2016), (14) (Pritchard 2011; Ratcliffe et al. 2017), (15) (Reed et al. 2012; Maestre et al. 2013), (16) (Hellmann et al. 2008; Hulme 2017), (17) (Pureswaran et al. 2015; Cilas et al. 2016; Macfadyen et al. 2018), (18) (Jolly et al. 2015; Abatzoglou and Williams 2016; Taufik et al. 2017; Knorr et al. 2016), (19) (Davin et al. 2010; Pinty et al. 2011), (20) (Wang et al. 2017b; Chappell et al. 2016), (21) (Pendleton et al. 2012), (22) (Oertel et al. 2016), (23) (Houghton et al. 2012; Eglin et al. 2010), (24) (Schuur et al. 2015; Christensen et al. 2004; Walter Anthony et al. 2016; Abbott et al. 2016), (25) (Belnap, Walker, Munson, & Gill, 2014; Rutherford et al., 2017), (26) (Page et al. 2002; Pellegrini et al. 2018)

4.2.2 Drivers of land degradation

Drivers of land degradation and land improvement are many and they interact in multiple ways. Figure 4.2, illustrates how some of the most important drivers interact with the land users. It is important to keep in mind that both natural and human factors can drive both degradation and improvement (Kiage 2013; Bisaro et al. 2014).

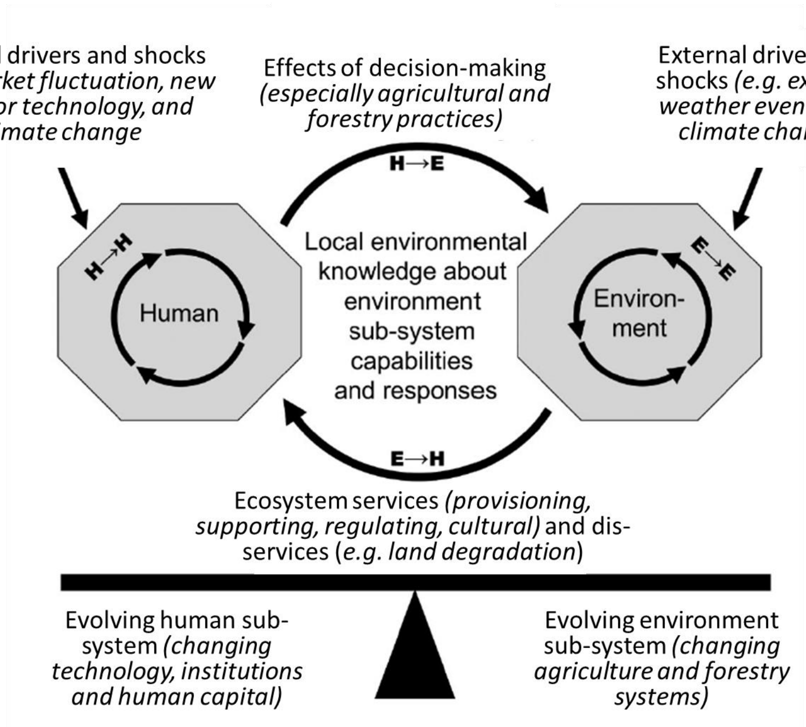


Figure 4.2 Schematic representation of the interactions between the human and environmental components of the land system showing decision making and ecosystem services as the key linkages between the components (moderated by an effective system of local and scientific knowledge), and indicating how the rates of change and the way these linkages operate must be kept broadly in balance for functional coevolution of the components. Modified with permission from (Stafford Smith et al. 2007)**.**

Land degradation is driven by the entire spectrum of factors, from very short and intensive events such as individual rain storms of 10 minutes removing topsoil or initiating a gully or a landslide (Coppus and Imeson 2002; Morgan 2005b) to century scale slow depletion of nutrients or loss of soil particles (Johnson and Lewis 2007, p. 5-6). But instead of focusing on absolute temporal variations, the drivers of land degradation can be assessed in relation to the rates of possible recovery. Unfortunately, this is impractical to do in a spatially explicit way because rates of soil formation is difficult to measure due the slow rate, usually < 5mm/century (Delgado and Gómez 2016). Studies suggest that erosion rates of conventionally tilled agricultural fields exceed the rate at which soil is generated by one to two orders of magnitude (Montgomery 2007a).

The landscape effects of gully erosion from one short intensive rainstorm can persist for decades and centuries (Showers 2005). Intensive agriculture under the Roman Empire in occupied territories in France is still leaving its marks and can be considered an example of irreversible land degradation (Dupouey et al. 2002).

The climate change related drivers of land degradation are both gradual changes of temperature, precipitation, and wind as well as changes of the distribution and intensity of extreme events (Lin et al. 2017). Importantly, these drivers can act in two directions: land improvement and land degradation. Increasing CO_2 levels in the atmosphere is a driver of land improvement even if the net effect is modulated by other factors, such as the availability of nitrogen (Terrer et al. 2016) and water (Gerten et al. 2014; Settele et al. 2015; Girardin et al. 2016).

The gradual and planetary changes that can cause land degradation/improvement have been studied by global integrated models and Earth observation technologies. Studies of global land suitability for agriculture suggest that climate change will increase the area suitable for agriculture by 2100 in the Northern high latitudes by 16% (Ramankutty et al. 2002) or 5.6 million km^2 (Zabel et al. 2014), while tropical regions will experience a loss (Ramankutty et al. 2002; Zabel et al. 2014).

Temporal and spatial patterns of tree mortality can be used as an indicator of climate change impacts on terrestrial ecosystems. Episodic mortality of trees occur naturally even without climate change, but more widespread spatio-temporal anomalies can be a sign of climate induced degradation (Allen et al. 2010). In the absence of systematic data on tree mortality, a comprehensive meta-analysis of 150 published articles suggests that increasing tree mortality around the world can be attributed to increasing drought and heat stress in forests worldwide (Allen et al. 2010).

Other and more indirect drivers can be a wide range of factors such as demographic changes, technological change, changes of consumption patterns and dietary preferences, political and economic changes, and social changes (Mirzabaev et al. 2016). It is important to stress that there are no simple or direct relationships between underlying drivers and land degradation, such as poverty or high population density, that are necessarily causing land degradation (Lambin et al. 2001). However, drivers of land degradation need to be studied in the context of spatial, temporal, economic, environmental and cultural aspects (Warren 2002). Some analyzes suggest an overall negative correlation between population density and land degradation (Bai et al. 2008) but we find many local examples of both positive and negative relationships (Brandt et al. 2018a, 2017). Even if there are correlations in one or the other direction, causality is not always the same.

Land degradation is inextricably linked to several climate variables, such as temperature, precipitation, wind, and seasonality. This means that there are many ways in which climate change and land degradation are linked. The linkages are better described as a web of causality than a set of cause – effect relationships.

4.2.3 Attribution in the case of land degradation

The question here is whether or not climate change can be attributed to land degradation and vice versa. Land degradation is a complex phenomenon often affected by multiple factors such as climatic (rainfall, temperature, and wind), abiotic ecological factors (e.g. soil characteristics and topography), type of land use (e.g. farming of various kinds, forestry, or protected area), and land management practices (e.g. tilling, crop rotation, and logging/thinning). Therefore, attribution of land degradation to climate change is extremely challenging. Because land degradation is highly dependent on land management, it is even possible that climate impacts would trigger land management changes reducing or reversing land degradation, sometimes called transformational adaptation (Kates et al. 2012). There is not much research on attributing land degradation explicitly to climate change, but there is more on climate change as a threat multiplier for land degradation. However, it is in some cases possible to infer climate change impacts on land degradation both theoretically and empirically. Section 4.2.3.1 will outline the potential direct linkages of climate change on land degradation based on current theoretical understanding of land degradation processes and drivers. Section 4.2.3.2 will investigate possible indirect impacts on land degradation.

4.2.3.1 Direct linkages with climate change

The most important direct impacts of climate change on land degradation are the results of increasing temperatures, changing rainfall patterns, and intensification of rainfall. These changes will in various combinations cause changes in erosion rates and the processes driving both increases and decreases of soil erosion. From an attribution point of view, it is important to note that projections of precipitation are in general more uncertain than projections of temperature changes (Murphy et al. 2004; Fischer and Knutti 2015; IPCC 2013a). Precipitation involves local processes of larger complexity than temperature and projections are usually less robust than those for temperature (Giorgi and Lionello 2008; Pendergrass 2018).

Theoretically the intensification of the hydrological cycle as a result of human induced climate change is well established (Guerreiro et al. 2018; Trenberth 1999; Pendergrass et al. 2017; Pendergrass and Knutti 2018) and also empirically observed (Blenkinsop et al. 2018; Burt et al. 2016a; Liu et al. 2009; Bindoff et al. 2013). AR5 WGI concluded that heavy precipitation events have increased in frequency, intensity, and/or amount since 1950 (*likely*) and that further changes in this direction are *likely* to *very likely* during the 21st century (IPCC 2013). The IPCC Special Report on 1.5°C concluded that human-induced global warming has already caused an increase in the frequency, intensity and/or amount of heavy precipitation events at the global scale (Hoegh-Guldberg et al. 2018). As an example, in central India, there has been a threefold increase in widespread extreme rain events during 1950–2015 which has influenced several land degradation processes, not least soil erosion (Burt et al. 2016b). In Europe and North America, where observation networks are dense and having long time series, it is *likely* that the frequency or intensity of heavy rainfall have increased (IPCC 2013b). It is also expected that seasonal shifts and cycles such as monsoons and ENSO (see Glossary) will further increase the intensity of rainfall events (IPCC 2013).

When rainfall regimes change, it is expected to drive changes in vegetation cover and composition, which may be a cause of land degradation in and of itself, as well as impacting other aspects of land degradation. Vegetation cover, for example is a key factor in determining soil loss through both water (Nearing et al. 2005) and wind erosion (Shao 2008). Changing rainfall regimes also affect below-ground biological processes, such as fungi and bacteria (Meisner et al. 2018; Shuab et al. 2017; Asmelash et al. 2016).

Changing snow accumulation and snow melt alter both volume and timing of hydrological flows in and from mountain areas (Brahney et al. 2017; Lutz et al. 2014), with potentially large impacts on downstream areas. Soil processes are also affected by changing snow conditions by affecting the partitioning between evaporation and streamflow and between subsurface flow and surface runoff (Barnhart et al. 2016). Rainfall intensity is a key climatic driver of soil erosion. Early modelling studies and theory suggest that light rainfall events will decrease while heavy rainfall events increase at about 7% per degree of warming (Liu et al. 2009; Trenberth 2011). Such changes result in increases in the intensity of rainfall which increase the erosive power of rainfall (erosivity) and hence increase the likelihood of water erosion. Increases in rainfall intensity can even exceed the rate of increase of atmospheric moisture content (Liu et al. 2009; Trenberth 2011). Erosivity is highly correlated to the product of total rainstorm energy and the maximum 30 minute rainfall intensity of the storm (Nearing et al. 2004a) and increases of erosivity will exacerbate water erosion substantially (Nearing et al. 2004a). However, the effects will not be uniform but highly variable across regions (Almagro et al. 2017; Mondal et al. 2016). Several empirical studies around the world have shown the increasing intensity of rainfall (IPCC 2013b; Ma et al. 2015, 2017) and also suggest that this will be accentuated with future increasing warming (Cheng and AghaKouchak 2015; Burt et al. 2016b; O'Gorman 2015).

The very comprehensive database of direct measurements of water erosion presented by García-Ruiz et al. (2015) contains 4377 entries (North America: 2776, Europe: 847, Asia: 259, Latin America: 237, Africa: 189, Australia & Pacific: 67), even though not all entries are complete (Figure 4.3).

a

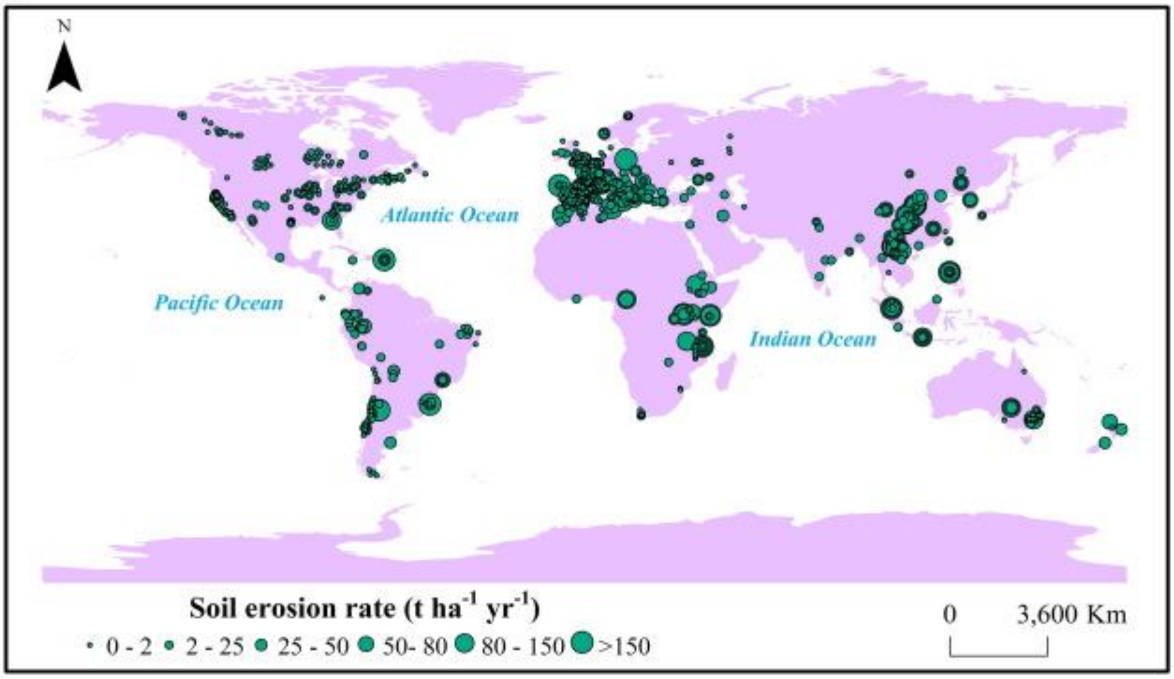


Figure 4.3. Map of observed soil erosion rates in database of 4377 entries by García-Ruiz et al., 2015). The map was published by (Li and Fang 2016).

An important finding from that database is that almost any erosion rate is possible under almost any climatic condition (García-Ruiz et al. 2015). Even if the results show few clear relationships between erosion and land conditions, the authors highlighted four observations: 1) the highest erosion rates were found in relation to agricultural activities – even though moderate erosion rates were also found in agricultural settings, 2) high erosion rates after forest fires were not observed (although the cases were few), 3) land covered by shrubs showed generally low erosion rates, 4) pasture land showed generally medium rates of erosion. Some important findings for the link between soil erosion and climate change can be noted from erosion measurements: erosion rates tend to increase with increasing mean annual rainfall, with a peak in the interval of 1000 to 1400 mm annual rainfall (García-Ruiz et al. 2015) (*low confidence*). However, such relationships are overshadowed by the fact that most rainfall events do not cause any erosion, instead erosion is caused by a few high intensity rainfall events (Fischer et al. 2016; Zhu et al. 2019). Hence mean annual rainfall is not a good predictor of erosion (Gonzalez-Hidalgo et al. 2012, 2009). In the context of climate change, it means the tendency of rainfall patterns to change towards more intensive precipitation events is serious. Such patterns have already been observed widely, even in cases where the total rainfall is decreasing (Trenberth 2011). The findings generally confirm the strong consensus about the importance of vegetation cover as a protection against soil erosion, emphasising how extremely important land management is for controlling erosion.

In the Mediterranean region, the observed and expected decrease in annual rainfall due to climate change is accompanied by an increase of rainfall intensity and hence erosivity (Capolongo et al. 2008). In tropical and sub-tropical regions, the on-site impacts of soil erosion dominate, and are manifested in very high rates of soil loss, in some cases exceeding 100 t ha^{-1} yr^{-1} (Tadesse 2001; García-Ruiz et al. 2015). In temperate regions, the off-site costs of soil erosion are often a greater concern, for example siltation of dams and ponds, downslope damage to property, roads and other infrastructure (Boardman 2010). In cases where water erosion occurs the down-stream effects, such as siltation of dams, are often significant and severe in terms of environmental and economic damages

(Kidane and Alemu 2015; Reinwarth et al. 2019; Quiñonero-Rubio et al. 2016; Adeogun et al. 2018; Ben Slimane et al. 2016).

The distribution of wet and dry spells also affects land degradation although uncertainties remain depending on resolution of climate models used for prediction (Kendon et al. 2014). Changes in timing of rainfall events may have significant impacts on processes of soil erosion through changes in wetting and drying of soils (Lado et al. 2004).

Soil moisture content is affected by changes in evapotranspiration and evaporation which may influence the partitioning of water into surface and subsurface runoff (Li and Fang 2016; Nearing et al. 2004b). This portioning of rainfall can have a decisive effect on erosion (Stocking et al. 2001).

Wind erosion is a serious problem in agricultural regions, not only in drylands (Wagner 2013). Near-surface wind speeds over land areas have decreased in recent decades (McVicar and Roderick 2010), partly as a result of changing surface roughness (Vautard et al. 2010), Theoretically (Bakun 1990; Bakun et al. 2015) and empirically (Sydeman et al. 2014; England et al. 2014) mean winds along coastal regions worldwide have increased with climate change (*medium evidence*, *high agreement*). Other studies of wind and wind erosion have not detected any long-term trend suggesting that climate change has altered wind patterns outside drylands in a way that can significantly affect the risk of wind erosion (Pryor and Barthelmie 2010; Bärring et al. 2003). Therefore, the findings regarding wind erosion and climate change are inconclusive, partly due to inadequate measurements.

Global mean temperatures are rising worldwide, but particularly in the Arctic region (*high confidence*) (IPCC 2018a). Heat stress from extreme temperatures and heatwaves (multiple days of hot weather in a row) have increased markedly in some locations in the last three decades (*high confidence*), and are *virtually certain* to continue during the 21st century (Olsson et al. 2014a). The IPCC Special Report on Global Warming of 1.5°C concluded that human-induced global warming has already caused more frequent heatwaves in most of land regions, and that climate models project robust differences between present-day and global warming up to 1.5°C and between 1.5°C and 2°C (Hoegh-Guldberg et al. 2018). Direct temperature effects on soils are of two kinds. Firstly, permafrost thawing leads to soil degradation in boreal and high altitude regions (Yang et al. 2010; Jorgenson and Osterkamp 2005). Secondly, warming alters the cycling of nitrogen (N) and carbon (C) in soils, partly due to impacts on soil microbiota (Solly et al. 2017). There are many studies with particularly strong experimental evidence, but a full understanding of cause and effect is contextual and elusive (Conant et al. 2011a,b; Wu et al. 2011). This is discussed comprehensively in Chapter 2.

Climate change, including increasing atmospheric CO_2 levels, affects vegetation structure and function and hence conditions for land degradation. Exactly how vegetation responds to changes remains a research task. In a comparison of seven global vegetation models under four representative concentration pathways Friend et al., (2014) found that all models predicted increasing vegetation carbon storage, however with substantial variation between models. An important insight compared with previous understanding is that structural dynamics of vegetation seems to play a more important role for carbon storage than vegetation production (Friend et al. 2014). The magnitude of CO_2 fertilisation of vegetation growth, and hence conditions for land degradation is still uncertain (Holtum and Winter 2010), particularly in tropical rainforests (Yang et al. 2016). For more discussion on this topic, see Chapter 2 in this report.

In summary, rainfall changes attributed to human-induced climate change have already intensified drivers of land degradation (*robust evidence, high agreement*) but attributing land degradation to climate change is challenging because of the importance of land management (*medium evidence, high agreement*). Changes in climate variability modes, such as in monsoons and ENSO events, can also affect land degradation (*low evidence, low agreement*).

4.2.3.2 Indirect and complex linkages with climate change

Many important indirect linkages between land degradation and climate change occur via agriculture, particularly through changing outbreaks of pests (Rosenzweig et al. 2001; Porter et al. 1991; Thomson et al. 2010; Dhanush et al. 2015; Lamichhane et al. 2015), which is covered comprehensively in Chapter 5. More negative impacts have been observed than positive ones (IPCC 2014b). After 2050 the risk of yield losses increase as a result of climate change in combination with other drivers (*medium confidence*) and such risks will increase dramatically if global mean temperatures increase by ~4°C (*high confidence*) (Porter et al. 2014). The reduction (or plateauing) in yields in major production areas (Brisson et al. 2010; Lin and Huybers 2012; Grassini et al. 2013) may trigger cropland expansion elsewhere, either into natural ecosystems, marginal arable lands or intensification on already cultivated lands, with possible consequences for increasing land degradation.

Precipitation and temperature changes will trigger changes in land- and crop management, such as changes in planting and harvest dates, type of crops, and type of cultivars, which may alter the conditions for soil erosion (Li and Fang 2016).

Much research has tried to understand how plants are affected by a particular stressor, for example drought, heat, or waterlogging, including effects on belowground processes. But less research has tried to understand how plants are affected by several simultaneous stressors – which of course is more realistic in the context of climate change (Mittler 2006; Kerns et al. 2016) and from a hazards point of view (see 7.2.1). From an attribution point of view, such a complex web of causality is problematic if attribution is only done through statistical significant correlation. It requires a combination of statistical links and theoretically informed causation, preferably integrated into a model. Some modelling studies have combined several stressors with geomorphologically explicit mechanisms (using the WEPP model) and realistic land use scenarios, and found severe risks of increasing erosion from climate change (Mullan et al. 2012; Mullan 2013). Other studies have included various management options, such as changing planting and harvest dates (Zhang and Nearing 2005; Parajuli et al. 2016; Routschek et al. 2014; Nunes and Nearing 2011), type of cultivars (Garbrecht and Zhang 2015), and price of crops (Garbrecht et al. 2007; O'Neal et al. 2005) to investigate the complexity of how new climate regimes may alter soil erosion rates.

In summary, climate change increases the risk of land degradation both in terms of likelihood and consequence but the exact attribution to climate change is challenging due to several confounding factors. But since climate change exacerbates most degradation processes it is clear that unless land management is improved, climate change will result in increasing land degradation (*very high confidence*).

4.2.4 Approaches to assessing land degradation

In a review of different approaches and attempts to map global land degradation, Gibbs and Salmon (2015) identified four main approaches to map the global extent of degraded lands: expert opinions (Oldeman and van Lynden 1998; Dregne 1998; Reed 2005; Bot et al. 2000), satellite observation of vegetation greenness (e.g., remote sensing of NDVI (Normalized Difference Vegetation Index), EVI (Enhanced Vegetation Index), PPI (Plant Phenology Index) (Yengoh et al. 2015; Bai et al. 2008c; Shi et al. 2017a; Abdi et al. 2019; JRC 2018), biophysical models (biogeographical/ topological) (Cai et al. 2011b; Hickler et al. 2005; Steinkamp and Hickler 2015; Stoorvogel et al. 2017) and inventories of land use/condition. Together they provide a relatively complete evaluation, but none on its own assesses the complexity of the process (Vogt et al. 2011; Gibbs and Salmon 2015). There is, however, a robust consensus that remote sensing and field-based methods are critical to assess and monitor land degradation, particularly over large areas (such as global, continental and sub-continental) although there are still knowledge gaps to be filled (Wessels et al. 2007, 2004; Prince 2016; Ghazoul and Chazdon 2017) as well as the problem of baseline (see 4.1.3).

Remote sensing can provide meaningful proxies of land degradation in terms of severity, temporal development, and areal extent. These proxies of land degradation include several indexes that have been used to assess land conditions and monitoring the changes of land condition, for example extent of gullies, severe forms of rill and sheet erosion, and deflation. The presence of open-access, quality controlled and continuously updated global databases of remote sensing data is invaluable, and is the only method for consistent monitoring of large areas over several decades (Sedano et al. 2016; Brandt et al. 2018b; Turner 2014).The NDVI, as a proxy for Net Primary Production (NPP, see glossary), is one of the most commonly used methods to assess land degradation, since it indicates land cover, an important factor for soil protection. Although NDVI is not a direct measure of vegetation biomass, there is a close coupling between NDVI integrated over a season and in situ NPP (*high agreement*, *robust evidence*) (see Higginbottom et al. 2014; Andela et al. 2013; Wessels et al. 2012).

Distinction between land degradation/improvement and the effects of climate variation is an important and contentious issue (Murthy and Bagchi 2018; Ferner et al. 2018). There is no simple and straightforward way to disentangle these two effects. The interaction of different determinants of primary production is not well understood. A key barrier to this is a lack of understanding of the inherent inter-annual variability of vegetation (Huxman et al. 2004; Knapp and Smith 2001; Ruppert et al. 2012; Bai et al. 2008a; Jobbágy and Sala 2000). One possibility is to compare potential land productivity modelled by vegetation models and actual productivity measured by remote sensing (Seaquist et al. 2009; Hickler et al. 2005; van der Esch et al. 2017), but the difference in spatial resolution, typically 0.5 degrees for vegetation models compared to 0.25–0.5 km for remote sensing data, is hampering the approach. Moderate-resolution Imaging Spectroradiometer, or MODIS, provides higher spatial resolution (up to 0.25 km), delivers data for the Enhanced Vegetation Index (EVI) which is calculated similarly to NDVI and have showed robust approach to estimate spatial patterns of global annual primary productivity (Shi et al. 2017b; Testa et al. 2018).

Another approach to disentangle the effects of climate and land use/management is to use the Rain Use Efficiency (RUE), defined as the biomass production per unit of rainfall, as an indicator (Le Houerou 1984; Prince et al. 1998; Fensholt et al. 2015). A variant of the RUE approach is the residual trend (RESTREND) of a NDVI time-series, defined as the fraction of the difference between the observed NDVI and the NDVI predicted from climate data (Yengoh et al. 2015; John et al. 2016). These two metrics aim to estimate the NPP, rainfall and the time dimensions. They are simple transforms of the same three variables: RUE shows the NPP relationship with rainfall for individual years, while RESTREND is the interannual change of RUE; also, both consider that rainfall is the only variable that affects biomass production. They are legitimate metrics when used appropriately, but in many cases they involve oversimplifications and yield misleading results (Fensholt et al. 2015; Prince et al. 1998).

Furthermore, increases in NPP do not always indicate improvement in land condition/reversal of land degradation, since this does not account for changes in vegetation composition. It could, for example, result from conversion of native forest to plantation, or due to bush encroachment, which many consider to be a form of land degradation (Ward 2005). Also, NPP may be increased by irrigation, which can enhance productivity in the short-medium term while increasing risk of soil salinisation in the long term (Niedertscheider et al. 2016).

Recent progress and expanding time series of canopy characterisations based on passive microwave satellite sensors have offered rapid progress in regional and global descriptions of forest degradation and recovery trends (Tian et al. 2017). The most common proxy is VOD (vertical optical depth) and has already been used to describe global forest/savannah carbon stock shifts over two decades, highlighting strong continental contrasts (Liu et al. 2015a) demonstrating the value of this approach to monitor forest degradation at large scales. Contrasting NDVI which is only sensitive to vegetation "greenness", from which primary production can be modelled, VOD is also sensitive to water in woody parts of the vegetation and hence provides a view of vegetation dynamics that can be complementary to NDVI. As well as the NDVI, VOD also needs to be corrected to take into account the rainfall variation (Andela et al. 2013).

Even though remote sensing offers much potential, its application to land degradation and recovery remains challenging as structural changes often occur at scales below the detection capabilities of most remote sensing technologies. Additionally, if the remote sensing is based on vegetation indexes data, other forms of land degradation, such as nutrient depletion, changes of soil physical or biological properties, loss of values for humans, among others, cannot be inferred indirectly by remote sensing. The combination of remotely sensed images and field based approach can give improved estimates of carbon stocks and tree biodiversity (Imai et al. 2012; Fujiki et al. 2016).

Additionally, the majority of trend techniques employed would be capable of detecting only the most severe of degradation processes, and would therefore not be useful as a degradation early-warning system (Higginbottom et al. 2014; Wessels et al. 2012). However, additional analyses using higher resolution imagery, such as the Landsat and SPOT satellites, would be well suited to provide further localized information on trends observed (Higginbottom et al. 2014). New approaches to assess land degradation using high spatial resolution are developing but the need for time series makes progress slow. The use of synthetic aperture radar (SAR) data has been shown to be advantageous for the estimation of soil surface characteristics, in particular surface roughness and soil moisture (Gao et al. 2017; Bousbih et al. 2017), and detecting and quantifying selective logging (Lei et al. 2018). It is still necessary to maintain the efforts to fully assess land degradation using remote sensing.

Computer simulation models can be used alone or combined with the remote sensing observations to assess land degradation. The RUSLE (Revised Universal Soil Loss Equation) can be used, to some extent, to predict the long-term average annual soil loss by water erosion. RUSLE has been constantly revisited to estimate soil loss based on the product of rainfall–runoff erosivity, soil erodibility, slope length and steepness factor, conservation factor, and support practice parameter (Nampak et al. 2018). Inherent limitations of RUSLE include data-sparse regions, inability to account for soil loss from gully erosion or mass wasting events, and that it does not predict sediment pathways from hillslopes to water bodies (Benavidez et al. 2018). Since RUSLE models only provide gross erosion, the integration of a further module in the RUSLE scheme to estimate the sediment yield from the modelled hillslopes is needed. The spatially distributed sediment delivery model WaTEM/SEDEM has been widely tested in Europe (Borrelli et al. 2018). Wind erosion is another factor that needs to be taken into account in the modelling of soil erosion (Webb et al. 2017a, 2016). Additional models need to be developed to include the limitations of the RUSLE models.

Regarding the field based approach to assess land degradation, there are multiple indicators that reflect functional ecosystem processes linked to ecosystem services and, thus, to the value for humans. These indicators are a composite set of measurable attributes from different factors, such as climate, soil, vegetation, biomass, management, among others, that can be used together or to develop indexes to better assess land degradation (Allen et al. 2011; Kosmas et al. 2014).

Declines in vegetation cover, changes in vegetation structure, decline in mean species abundances, decline in habitat diversity, changes in abundance of specific indicator species, reduced vegetation health and productivity, and vegetation management intensity and use, are the most common indicators in the vegetation condition of forest and woodlands (Stocking et al. 2001; Wiesmair et al. 2017; Ghazoul and Chazdon 2017; Alkemade et al. 2009).

Several indicators of the soil quality (soil organic matter, depth, structure, compaction, texture, pH, C:N ratio, aggregate size distribution and stability, microbial respiration, soil organic carbon, salinisation, among others) have been proposed (see also 2.2) (Schoenholtz et al. 2000). Among these, soil organic matter (SOM) directly and indirectly drives the majority of soil functions. Decreases in

SOM can lead to a decrease in fertility and biodiversity, as well as a loss of soil structure, causing reductions in water holding capacity, increased risk of erosion (both wind and water) and increased bulk density and hence soil compaction (Allen et al. 2011; Certini 2005; Conant et al. 2011a). Thus, indicators related with the quantity and quality of the SOM are necessary to identify land degradation (Pulido et al. 2017; Dumanski and Pieri 2000). The composition of the microbial community is *very likely* to be positive impacted by both climate change and land degradation processes (Evans and Wallenstein 2014; Wu et al. 2015; Classen et al. 2015), thus changes in microbial community composition can be very usefull to rapidly reflect land degradation (e.g. forest degradation increased the bacterial alpha-diversity indexes) (Flores-Rentería et al. 2016; Zhou et al. 2018). These indicators might be used as a set of indicators site-dependent, and in a plant-soil system (Ehrenfeld et al. 2005).

Useful indicators of degradation and improvement include changes in ecological processes and disturbance regimes that regulate the flow of energy and materials and that control ecosystem dynamics under a climate change scenario. Proxies of dynamics include spatial and temporal turnover of species and habitats within ecosystems (Ghazoul et al. 2015; Bahamondez and Thompson 2016). Indicators in agricultural lands include crop yield decreases and difficulty in maintaining yields (Stocking et al. 2001). Indicators of landscape degradation/improvement in fragmented forest landscapes include the extent, size, and distribution of remaining forest fragments, an increase in edge habitat, and loss of connectivity and ecological memory (Zahawi et al. 2015; Pardini et al. 2010).

In summary, as land degradation is such a complex and global process there is no single method by which land degradation can be estimated objectively and consistently over large areas (*very high confidence*). However, many approaches exist that can be used to assess different aspects of land degradation or provide proxies of land degradation. Remote sensing, complemented by other kinds of data (i.e., field observations, inventories, expert opinions), is the only method that can generate geographically explicit and globally consistent data over time scales relevant for land degradation (several decades).

4.3 Status and current trends of land degradation

The scientific literature on land degradation often excludes forest degradation, yet here we attempt to assess both issues. Because of the different bodies of scientific literature, we assess land degradation and forest degradation under different sub-headings, and where possible draw integrated conclusions.

4.3.1 Land degradation

There are no reliable global maps of the extent and severity of land degradation (Gibbs and Salmon 2015; Prince et al. 2018; van der Esch et al. 2017), despite the fact that land degradation is a severe problem (Turner et al. 2016). The reasons are both conceptual, i.e., how is land degradation defined, using what baseline (Herrick et al. 2019) or over what time period, and methodological, i.e. how can it be measured (Prince et al. 2018). Although there is a strong consensus that land degradation is a reduction in productivity of the land or soil, there are diverging views regarding the spatial and temporal scales at which land degradation occurs (Warren 2002), and how this can be quantified and mapped. Proceeding from the definition in this report, there are also diverging views concerning ecological integrity and the value to humans. A comprehensive treatment of the conceptual discussion about land degradation is provided by the recent report on land degradation from the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services, IPBES (Montanarella et al. 2018).

A review of different attempts to map global land degradation, based on expert opinion, satellite observations, biophysical models and a data base of abandoned agricultural lands, suggested that between <10 M km^2 to 60 M km^2 (corresponding to 8–45% of the ice-free land area) have been degraded globally (Gibbs and Salmon, 2015) (*very low confidence*).

One often used global assessment of land degradation used trends in NDVI as a proxy for land degradation and improvement during the period 1983 to 2006 (Bai et al. 2008b,c) with an update to 2011 (Bai et al. 2015). These studies, based on very coarse resolution satellite data (8 km NOAA AVHRR), indicated that between 22% and 24% of the global ice-free land area was subject to a downward trend, while about 16% showed an increasing trend. The study also suggested, contrary to earlier assessments (Middleton and Thomas 1997), that drylands were not among the most affected regions. Another study using a similar approach for the period 1981-2006 suggested that about 29% of the global land area is subject to 'land degradation hotspots', i.e. areas with acute land degradation in need of particular attention. These hotspot areas were distributed over all agro-ecological regions and land cover types. Two different studies have tried to link land degradation, identified by NDVI as a proxy, and number of people affected: Le et al. (2016) estimated that at least 3.2 billion people were affected, while Barbier and Hochard (2016, 2018) estimated that 1.33 billion people were affected, of which 95% were living in developing countries.

Yet another study, using a similar approach and type of remote sensing data, compared NDVI trends with biomass trends calculated by a global vegetation model over the period 1982-2010 and found that 17-36% of the land areas showed a negative NDVI trend while a positive or neutral trend was predicted in modelled vegetation (Schut et al. 2015). The World Atlas of Desertification (3rd edition) includes a global map of land productivity change over the period 1999 to 2013, which is one useful proxy for land degradation (Cherlet et al. 2018). Over that period about 20% of the global ice-free land area shows signs of declining or unstable productivity, whereas about 20% shows increasing productivity. The same report also summarized the productivity trends by land categories and found that most forest land showed increasing trends in productivity while rangelands had more declining trends than increasing trends (Fig 4.4). These productivity assessments, however, do not distinguish between trends due to climate change and trends due to other factors. A recent analysis of "greening" of the world using MODIS time series of NDVI 2000 – 2017, shows a striking increase in the greening over China and India. In China the greening is seen over both forested areas, 42%, and cropland areas, in which 32% is increasing (see Section 4.9.3). In India, the greening is almost entirely associated with cropland (82%) (Chen et al. 2019).

All the studies of vegetation trends referred to above show that there are regionally-differentiated trends of either decreasing or increasing vegetation. When comparing vegetation trends with trends in climatic variables, Schut et al. (2015) found very few areas (1-2%) where an increase in vegetation trend was independent of the climate drivers, and that study suggested that positive vegetation trends are primarily caused by climatic factors.

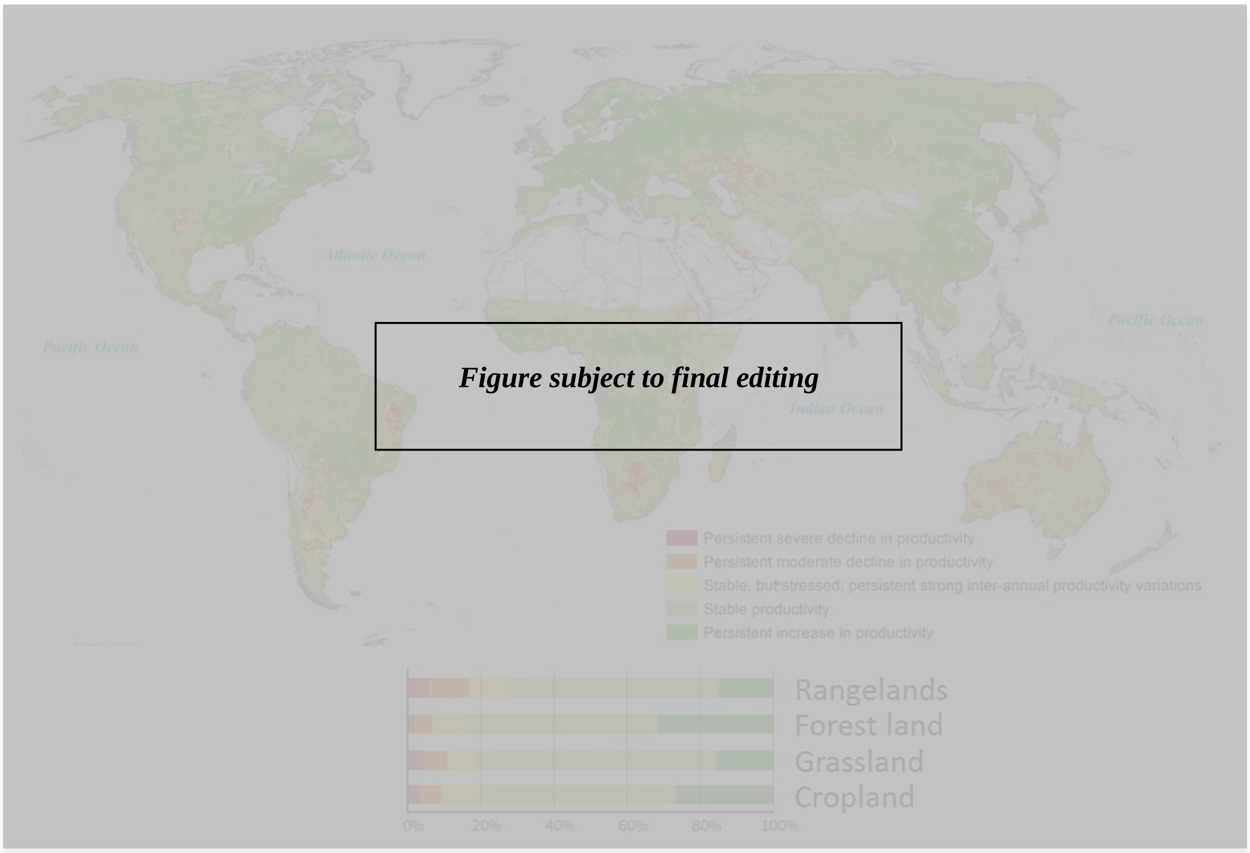


Figure 4.4. Proportional global land productivity trends by land cover/land use class. (Cropland includes arable land, permanent crops and mixed classes with over 50% crops; Grassland includes natural grassland and managed pasture land; Rangelands include shrub land, herbaceous and sparsely vegetated areas; Forest land includes all forest categories and mixed classes with tree cover greater than 40%). Data source: Copernicus Global Land SPOT VGT, 1999-2013.

In an attempt to go beyond the mapping of global vegetation trends for assessing land degradation, Borelli et al. (2017) used a soil erosion model (RUSLE) and suggested that soil erosion is mainly caused in areas of crop land expansion, particularly in sub-Saharan Africa, South America and Southeast Asia. The method is controversial for both conceptual reasons (i.e., the ability of the model to capture the most important erosion processes) and data limitations (i.e., the availability of relevant data at regional to global scales), and its validity for assessing erosion over large areas has been questioned by several studies (Baveye 2017; Evans and Boardman 2016a,b; Labrière et al. 2015).

An alternative to using remote sensing for assessing the state of land degradation is to compile field based data from around the globe (Turner et al. 2016). In addition to the problems of definitions and baselines, this approach is also hampered by the lack of standardized methods used in the field. An assessment of the global severity of soil erosion in agriculture, based on 1,673 measurements around the world (compiled from 201 peer reviewed articles), indicated that the global net median rate of soil formation (i.e., formation minus erosion) is about 0.004 mm yr^{-1} (~ 0.05 t $ha^{-1}yr^{-1}$) compared with the median net rate of soil loss in agricultural fields, 1.52 mm yr^{-1} (~ 18 t $ha^{-1}yr^{-1}$) in tilled fields and 0.065 mm yr^{-1} (~ 0.8 t $ha^{-1}yr^{-1}$) in no-till fields (Montgomery 2007a). This means that the rate of soil erosion from agricultural fields is in between 380 (conventional tilling) and 16 times (no-till) the natural rate of soil formation (*medium agreement, limited evidence*). These approximate figures are supported by another large meta-study including over 4000 sites around the world (see Figure 4.4) where the average soil loss from agricultural plots was ~ 21 t $ha^{-1}yr^{-1}$ (García-Ruiz et al. 2015). Climate change, mainly through the intensification of rainfall, will further increase these rates unless land management is improved (*high agreement, medium evidence*).

Soils contain about 1500 Gt of organic carbon (median across 28 different estimates presented by (Scharlemann et al. 2014)), which is about 1.8 times more carbon than in the atmosphere (Ciais et al. 2013) and 2.3 – 3.3 times more than what is held in the terrestrial vegetation of the world (Ciais et al. 2013). Hence, land degradation including land conversion leading to soil carbon losses has the potential to impact the atmospheric concentration of CO_2 substantially. When natural ecosystems are cultivated they lose soil carbon that accumulated over long time periods. The loss rate depends on the type of natural vegetation and how the soil is managed. Estimates of the magnitude of loss vary but figures between 20% and 59% have been reported in several meta studies (Poeplau and Don 2015; Wei et al. 2015; Li et al. 2012; Murty et al. 2002; Guo and Gifford 2002). The amount of soil carbon lost explicitly due to land degradation after conversion is hard to assess due to large variation in local conditions and management, see also Chapter 2.

From a climate change perspective, land degradation plays an important role in the dynamics of nitrous oxide (N_2O) and methane (CH_4). N_2O is produced by microbial activity in the soil and the dynamics are related to both management practices and weather conditions while CH_4 dynamics are primarily determined by the amount of soil carbon and to what extent the soil is subject to water logging (Palm et al. 2014), see also Chapter 2.

Several attempts have been made to map the human footprint on the planet (Čuček et al. 2012; Venter et al. 2016) but they in some cases confuse human impact on the planet with degradation. From our definition it is clear that human impact (or pressure) is not synonymous with degradation but information on the human footprint provides a useful mapping of potential non-climatic drivers of degradation.

In summary, there are no uncontested maps of the location, extent and severity of land degradation. Proxy estimates based on remote sensing of vegetation dynamics provide one important information source, but attribution of the observed changes in productivity to climate change, human activities, or other drivers is hard. Nevertheless, the different attempts to map the extent of global land degradation using remotely sensed proxies show some convergence and suggest that about a quarter of the ice free land area is subject to some form of land degradation (*limited evidence, medium agreement*) affecting about 3.2 billion people (*low confidence*). Attempts to estimate the severity of land degradation through soil erosion estimates suggest that soil erosion is a serious form of land degradation in croplands closely associated with unsustainable land management in combination with climatic parameters, some of which are subject to climate change (*limited evidence, high agreement*). Climate change is one among several causal factors in the status and current trends of land degradation (*limited evidence, high agreement*).

4.3.2 Forest degradation

Quantifying degradation in forests has also proven difficult. Indicators that remote sensing or inventory methods can measure more easily than reductions in biological productivity, losses of ecological integrity or value to humans include reductions in canopy cover or carbon stocks. However, the causes of reductions in canopy cover or carbon stocks can be many (Curtis et al. 2018), including natural disturbances (e.g., fires, insects and other forest pests), direct human activities (e.g., harvest, forest management) and indirect human impacts (such as climate change) and these may not reduce long-term biological productivity. In many boreal, some temperate and other forest types natural disturbances are common, and consequently these disturbance-adapted forest types are comprised of a mosaic of stands of different ages and stages of stand recovery following natural disturbances. In those managed forests where natural disturbances are uncommon or suppressed, harvesting is the primary determinant of forest age-class distributions.

Quantifying forest degradation as a reduction in productivity, carbon stocks or canopy cover also requires that an initial condition (or baseline) is established against which this reduction is assessed (see Section 4.1.4). In forest types with rare stand-replacing disturbances, the concept of “intact” or “primary” forest has been used to define the initial condition (Potapov et al. 2008) but applying a single metric can be problematic (Bernier et al. 2017). Moreover, forest types with frequent stand-replacing disturbances, such as wildfires, or with natural disturbances that reduce carbon stocks, such as some insect outbreaks, experience over time a natural variability of carbon stocks or canopy density making it more difficult to define the appropriate baseline carbon density or canopy cover against which to assess degradation. In these systems, forest degradation cannot be assessed at the stand level, but requires a landscape-level assessment that takes into consideration the stand age-class distribution of the landscape, which reflects natural and human disturbance regimes over past decades to centuries and also considers post-disturbance regrowth (van Wagner 1978; Volkova et al. 2018; Lorimer and White 2003).

The lack of a consistent definition of forest degradation also affects the ability to establish estimates of the rates or impacts of forest degradation because the drivers of degradation are not clearly defined (Sasaki and Putz 2009). Moreover, the literature at times confounds estimates of forest degradation and deforestation (i.e. the conversion of forest to non-forest land uses). Deforestation is a change in land use, while forest degradation is not, although severe forest degradation can ultimately lead to deforestation.

Based on empirical data provided by 46 countries, the drivers for deforestation (due to commercial agriculture) and forest degradation (due to timber extraction and logging) are similar in Africa, Asia and Latin America (Hosonuma et al. 2012). More recently, global forest disturbance over the period 2001 – 2015 was attributed to commodity driven deforestation (27 ± 5%), forestry (26 ± 4%), shifting agriculture (24 ± 3%) and wildfire (23 ± 4%). The remaining 0.6 ± 0.3% was attributed to the expansion of urban centers (Curtis et al. 2018).

The trends of productivity shown by several remote sensing studies (see previous section) are largely consistent with mapping of forest cover and change using a 34 year time series of coarse resolution satellite data (NOAA AVHRR) (Song et al. 2018). This study, based on a thematic classification of satellite data, suggests that (i) global tree canopy cover increased by 2.24 million km² between 1982 and 2016 (corresponding to +7.1%) but with regional differences that contribute a net loss in the tropics and a net gain at higher latitudes, and (ii) the fraction of bare ground decreased by 1.16 million km² (corresponding to -3.1%), mainly in agricultural regions of Asia (Song et al. 2018), see Figure 4.5. Other tree or land cover datasets show opposite global net trends (Li et al. 2018b), but high agreement in terms of net losses in the tropics and large net gains in the temperate and boreal zones (Li et al. 2018b; Song et al. 2018; Hansen et al. 2013). Differences across global estimates are further discussed in Chapter 1 (1.1.2.3) and Chapter 2.

The changes detected from 1982 to 2016 were primarily linked to direct human action, such as land-use changes (about 60% of the observed changes), but also to indirect effects, such as human induced climate change (about 40% of the observed changes) (Song et al. 2018), a finding also supported by a more recent study (Chen et al. 2019). The climate induced effects were clearly discernible in some regions, such as forest decline in the US Northwest due to increasing pest infestation and increasing fire frequency (Lesk et al. 2017; Abatzoglou and Williams 2016; Seidl et al. 2017), warming induced vegetation increase in the Arctic region, general greening in the Sahel probably as a result of increasing rainfall and atmospheric CO_2, and advancing treelines in mountain regions (Song et al. 2018).

Keenan et al.(Keenan et al. 2015) and Sloan and Sayer (2015) studied the 2015 Forest Resources Assessment (FRA) of the FAO (FAO 2016) and found that the total forest area from 1990 to 2015 declined by 3%, an estimate that is supported by a global remote sensing assessment of forest area change that found a 2.8% decline between 1990-2010 (D’Annunzio et al. 2017; Lindquist and D’Annunzio 2016). The trend in deforestation is, however, contradicting between these two global assessments with FAO (2016) suggesting deforestation is slowing down while the remote sensing assessments finds it to be accelerating (D'Annunzio et al. 2017). Recent estimates (Song et al. 2018) owing to semantic and methodological differences (see Chapter 1, section 1.1.2.3) suggest global tree cover to have increased over the period 1982-2016, which contradicts the forest area dynamics assessed by FAO (2016, Lindquist and D'Annunzio 2016). The loss rate in tropical forest areas from 2010 to 2015 is 55 000 km^2 yr^{-1}. According to the FRA the global natural forest area also declined from 39.61 M km^2 to 37.21 M km^2 during the period 1990 to 2015 (Keenan et al. 2015).

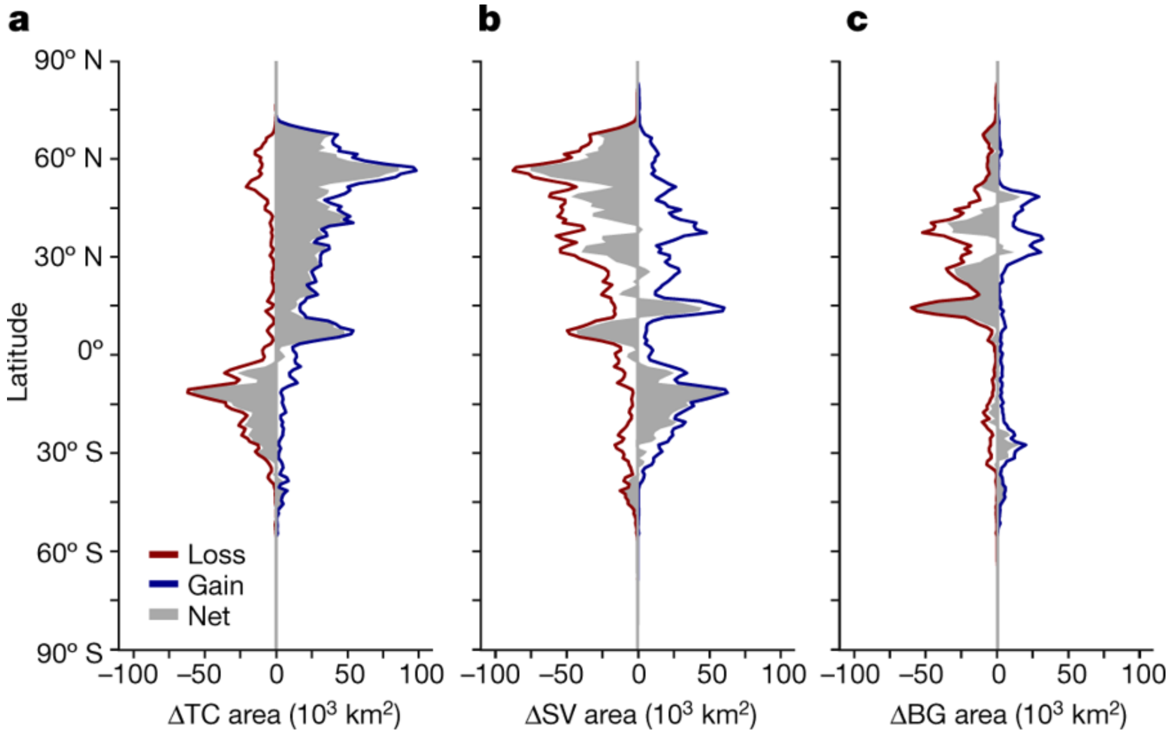


Figure 4.5. Diagrams showing latitudinal profiles of land cover change over the period 1982 to 2016 based on analysis of time-series of NOAA AVHRR imagery: a, Tree canopy cover change (ΔTC). b, Short vegetation cover change (ΔSV). c, Bare ground cover change (ΔBG). Area statistics were calculated for every 1° of latitude (Song et al. 2018). Source of data: NOAA AVHRR.

Since 1850, deforestation globally contributed 77% of the emissions from land-use and land-cover change (LULCC) while degradation contributed 10% (with the remainder originating from non-forest land uses) (Houghton and Nassikas 2018). That study also showed large temporal and regional differences with northern mid-latitude forests currently contributing carbon sinks due to increasing forest area and forest management. However, the contribution to carbon emissions of degradation as percentage of total forest emissions (degradation and deforestation) are uncertain, with estimates varying from 25% (Pearson et al. 2017) to nearly 70% of carbon losses (Baccini et al. 2017). The 25% estimate refers to an analysis of 74 developing countries within tropical and subtropical regions covering 22 million km^2 for the period 2005-2010 while the 70% estimate refers to an analysis of the tropics for the period 2003-2014, but by and large the scope of these studies is the same. Pearson et al. (2017) estimated annual gross emissions of 2.1 Gt CO_2, of which 53% were derived from timber harvest, 30% from wood fuel harvest and 17% from forest fire. Estimating gross emissions only, creates a distorted representation of human impacts on the land sector carbon cycle. While forest harvest for timber and fuel wood and land–use change (deforestation) contribute to gross emissions, to quantify impacts on the atmosphere it is necessary to estimate net emissions, i.e. the balance of gross emissions and gross removals of carbon from the atmosphere through forest regrowth (Chazdon et al. 2016a; Poorter et al. 2016; Sanquetta et al. 2018).

Current efforts to reduce atmospheric CO_2 concentrations can be supported by reductions in forest-related carbon emissions and increases in sinks, which requires that the net impact of forest management on the atmosphere be evaluated (Griscom et al. 2017). Forest management and the use of wood products in GHG mitigation strategies result in changes in forest ecosystem C stocks, changes in harvested wood product C stocks, and potential changes in emissions resulting from the use of wood products and forest biomass that substitute for other emissions-intensive materials such as concrete, steel and fossil fuels (Kurz et al. 2016; Lemprière et al. 2013; Nabuurs et al. 2007). The net impact of these changes on GHG emissions and removals, relative to a scenario without forest mitigation actions needs to be quantified, (e.g. Werner et al. 2010; Smyth et al. 2014; Xu et al. 2018). Therefore, reductions in forest ecosystem C stocks alone are an incomplete estimator of the impacts of forest management on the atmosphere (Nabuurs et al. 2007; Lemprière et al. 2013; Kurz et al. 2016; Chen et al. 2018b). The impacts of forest management and the carbon storage in long-lived products and landfills vary greatly by region, however, because of the typically much shorter life-span of wood products produced from tropical regions compared to temperate and boreal regions (Earles et al. 2012; Lewis et al. 2019; Iordan et al. 2018) (see also section 4.8.4).

Assessments of forest degradation based on remote sensing of changes in canopy density or land cover, (e.g., (Hansen et al. 2013; Pearson et al. 2017) quantify changes in aboveground biomass C stocks and require additional assumptions or model-based analyses to also quantify the impacts on other ecosystem carbon pools including belowground biomass, litter, woody debris and soil carbon. Depending on the type of disturbance, changes in aboveground biomass may lead to decreases or increases in other carbon pools, for example, windthrow and insect induced tree mortality may result in losses in aboveground biomass that are (initially) off-set by corresponding increases in dead organic matter carbon pools (Yamanoi et al. 2015; Kurz et al. 2008), while deforestation will reduce the total ecosystem carbon pool (Houghton et al. 2012).

A global study of current vegetation carbon stocks (450 Gt C), relative to a hypothetical condition without land-use (916 Gt C), attributed 42-47% of carbon stock reductions to land management effects without land-use change, while the remaining 53-58% of carbon stock reductions were attributed to deforestation and other land-use changes (Erb et al. 2018). While carbon stocks in European forests are lower than hypothetical values in the complete absence of human land use, forest area and carbon stocks have been increasing over recent decades (McGrath et al. 2015; Kauppi et al. 2018). Studies of Gingrich et al. (2015) on the long-term trends in land-use over nine European countries (Albania, Austria, Denmark, Germany, Italy, the Netherlands, Romania, Sweden and the United Kingdom) also show an increase in forest land and reduction in cropland and grazing land from the 19th century to the early 20th century. However, the extent to which human activities have affected the productive capacity of forest lands is poorly understood. Biomass Production Efficiency (BPE), i.e. the fraction of photosynthetic production used for biomass production, was significantly higher in managed forests (0.53) compared to natural forests (0.41) (and it was also higher in managed (0.44) compared to natural (0.63) grasslands) (Campioli et al. 2015). Managing lands for production may involve trade-offs. For example, a larger proportion of Net Primary Production in managed forests is allocated to biomass carbon storage, but lower allocation to fine roots is hypothesised to reduce soil C stocks in the long-term (Noormets et al. 2015). Annual volume increment in Finnish forests has more than doubled over the last century, due to increased growing stock, improved forest management and environmental changes (Henttonen et al. 2017).

As economies evolve, the patterns of land-use and C stock changes associated with human expansion into forested areas often include a period of rapid decline of forest area and carbon stocks, recognition of the need for forest conservation and rehabilitation, and a transition to more sustainable land management that is often associated with increasing carbon stocks, (e.g. Birdsey et al. 2006). Developed and developing countries around the world are in various stages of forest transition

(Kauppi et al. 2018; Meyfroidt and Lambin 2011). Thus, opportunities exist for sustainable forest management to contribute to atmospheric carbon targets through reduction of deforestation and degradation, forest conservation, forest restoration, intensification of management, and enhancements of carbon stocks in forests and harvested wood products (Griscom et al. 2017) (*medium evidence, medium agreement*).

4.4 Projections of land degradation in a changing climate

Land degradation will be affected by climate change in both direct and indirect ways, and land degradation will to some extent also feed-back into the climate system. The direct impacts are those in which climate and land interact directly in time and space. Examples of direct impacts are when increasing rainfall intensity exacerbates soil erosion, or when prolonged droughts reduce the vegetation cover of the soil making it more prone to erosion and nutrient depletion. The indirect impacts are those where climate change impacts and land degradation are separated in time and/or space. Examples of such impacts are when declining agricultural productivity due to climate change drives an intensification of agriculture elsewhere, which may cause land degradation. Land degradation, if sufficiently widespread, may also feed back into the climate system by reinforcing ongoing climate change.

Although climate change is exacerbating many land degradation processes (*high to very high confidence*), prediction of future land degradation is challenging because land management practices determine to a very large extent the state of the land. Scenarios of climate change in combination with land degradation models can provide useful knowledge on what kind and extent of land management will be necessary to avoid, reduce and reverse land degradation.

4.4.1 Direct impacts on land degradation

There are two main levels of uncertainty in assessing the risks of future climate change induced land degradation. The first level, where uncertainties are comparatively low, is the changes of the degrading agent, such as erosive power of precipitation, heat stress from increasing temperature extremes (HÜVE et al. 2011), water stress from droughts, and high surface wind speed. The second level of uncertainties, and where the uncertainties are much larger, relates to the above and belowground ecological changes as a result of changes in climate, such as rainfall, temperature, and increasing level of CO_2. Vegetation cover is crucial to protect against erosion (Mullan et al. 2012; García-Ruiz et al. 2015).

Changes in rainfall patterns, such as distribution in time and space, and intensification of rainfall events will increase the risk of land degradation, both in terms of likelihood and consequences (*high agreement, medium evidence*). Climate induced vegetation changes will increase the risk of land degradation in some areas (where vegetation cover will decline) (*medium confidence*). Landslides are a form of land degradation that is induced by extreme rainfall events. There is a strong theoretical reason for increasing landslide activity due to intensification of rainfall, but the empirical evidence is so far lacking that climate change has contributed to landslides (Crozier 2010; Huggel et al. 2012; Gariano and Guzzetti 2016), human disturbance may be a more important future trigger than climate change (Froude and Petley 2018).

Erosion of coastal areas as a result of sea level rise will increase worldwide (*very high confidence*). In cyclone prone areas (such as the Caribbean, Southeast Asia, and the Bay of Bengal) the combination of sea level rise and more intense cyclones (Walsh et al. 2016b), and in some areas also land subsidence (Yang et al. 2019; Shirzaei and Bürgmann 2018; Wang et al. 2018; Fuangswasdi et al. 2019; Keogh and Törnqvist 2019), will pose a serious risk to people and livelihoods (*very high confidence*), in some cases even exceeding limits to adaption, see further section 4.8.4.1, 4.9.6, 4.9.8.

4.4.1.1 Changes in water erosion risk due to precipitation changes

The hydrological cycle is intensifying with increasing warming of the atmosphere. The intensification means that the number of heavy rainfall events is increasing while the total number of rainfall events tends to decrease (Trenberth 2011; Li and Fang 2016; Kendon et al. 2014; Guerreiro et al. 2018; Burt et al. 2016a; Westra et al. 2014; Pendergrass and Knutti 2018) *(robust evidence, high agreement*). Modelling of changes in land degradation as a result of climate change alone is hard because of the importance of local contextual factors. As shown above, actual erosion rate is extremely dependent on local conditions, primarily vegetation cover and topography (García-Ruiz et al. 2015). Nevertheless, modelling of soil erosion risks has advanced substantially in recent decades and such studies are indicative of future changes in the risk of soil erosion while actual erosion rates will still primarily be determined by land management. In a review article, Li & Fang (Li and Fang 2016) summarised 205 representative modelling studies around the world where erosion models had been used in combination with down-scaled climate models to assess future (between 2030 to 2100) erosion rates. The meta-study by Li & Fang considered, where possible, climate change in terms of temperature increase and changing rainfall regimes and their impacts on vegetation and soils. Almost all of the sites had current soil loss rates above 1t ha^{-1} (assumed to be the upper limit for acceptable soil erosion in Europe) and 136 out of 205 studies predicted increased soil erosion rates. The percentage increase in erosion rates varied between 1.2% to as much as over 1600%, whereas 49 out of 205 studies projected more than 50% increase. Projected soil erosion rates varied substantially between studies because the important of local factors, hence climate change impacts on soil erosion should preferably be assessed at the local to regional scale, rather than the global (Li and Fang 2016).

Mesoscale convective systems (MCS), typically thunder storms, have increased markedly in recent 3-4 decades in the USA and Australia and they are projected to increase substantially (Prein et al. 2017). Using a climate model with the ability to represent MCS, Prein and colleagues were able to predict future increases in frequency, intensity, and size of such weather systems. Findings include the 30% decrease in number of MCS of <40 mm h^{-1}, but a sharp increase of 380% in the number of extreme precipitation events of >90 mm h^{-1} over the North American continent. The combined effect of increasing precipitation intensity and increasing size of the weather systems implies that the total amount of precipitation from these weather systems is expected to increase by up to 80% (Prein et al. 2017), which will substantially increase the risk of land degradation in terms of landslides, extreme erosion events, flashfloods etc.

The potential impacts of climate change on soil erosion can be assessed by modelling the projected changes in particular variables of climate change known to cause erosion, such as erosivity of rainfall. A study of the conterminous United States based on three climate models and three scenarios (A2, A1B, and B1) found that rainfall erosivity will increase in all scenarios, even if there are large spatial differences – strong increase in NE and NW, and either weak or inconsistent trends in the SW and mid-West (Segura et al. 2014).

In a study of how climate change will impact future soil erosion processes in the Himalayas, Gupta and Kumar (2017) estimated that soil erosion will increase by about 27% in the near term (2020s) and 22% in the medium term (2080s), with little difference between scenarios. A study from Northern Thailand estimated that erosity will increase by 5% in the near term (2020s) and 14% in the medium term (2080s), which would result in a similar increase of soil erosion, all other factors being constant (Plangoen and Babel 2014). Observed rainfall erosivity has increased significantly in the lower Niger Basin (Nigeria) and are predicted to increase further based on statistical downscaling of four General Circulation Models (GCM) scenarios, with an estimated increase of 14%, 19% and 24% for the 2030s, 2050s, and 2070s respectively (Amanambu et al. 2019).

Many studies from around the world where statistical downscaling of GCM results have been used in combination with process based erosion models show a consistent trend of increasing soil erosion

Using a comparative approach Serpa et al. (2015) studied two Mediterranean catchments (one dry and one humid) using a spatially explicit hydrological model (SWAT) in combination with land use and climate scenarios for 2071-2100. Climate change projections showed, on the one hand, decreased rainfall and streamflow for both catchments whereas sediment export decreased only for the humid catchment; projected land use change, from traditional to more profitable, on the other hand resulted in increase in streamflow. The combined effect of climate and land use change resulted in reduced sediment export for the humid catchment (-29% for A1B; -22% for B1) and increased sediment export for the dry catchment (+222% for A1B; +5% for B1). Similar methods have been used elsewhere, also showing the dominant effect of land use/land cover for runoff and soil erosion (Neupane and Kumar 2015).

A study of future erosion rates in Northern Ireland, using a spatially explicit erosion model in combination with downscaled climate projections (with and without sub-daily rainfall intensity changes), showed that erosion rates without land management changes would decrease by 2020s, 2050s and 2100s irrespective of changes in intensity, mainly as a result of a general decline in rainfall (Mullan et al. 2012). When land management scenarios were added to the modelling, the erosion rates started to vary dramatically for all three time periods, ranging from a decrease of 100% for no-till land use, to an increase of 3621% for row crops under annual tillage and sub-days intensity changes (Mullan et al. 2012). Again, it shows how crucial land management is for addressing soil erosion, and the important role of rainfall intensity changes.

There is a large body of literature based on modelling future land degradation due to soil erosion concluding that in spite of the increasing trend of erosive power of rainfall (*medium evidence*, *high agreement*) land degradation is primarily determined by land management (*very high confidence*).

4.4.1.2 Climate induced vegetation changes, implications for land degradation

The spatial mosaic of vegetation is determined by three factors: the ability of species to reach a particular location, how species tolerate the environmental conditions at that location (e.g. temperature, precipitation, wind, the topographic and soil conditions), and the interaction between species (including above/below ground species (Settele et al. 2015). Climate change is projected to alter the conditions and hence impact the spatial mosaic of vegetation, which can be considered a form of land degradation. Warren et al. (2018) estimated that only about 33% of globally important biodiversity conservation areas will remain intact if global mean temperature increases to 4.5°C, while twice that area (67%) will remain intact if warming is restricted to 2°C. According to AR5, the clearest link between climate change and ecosystem change is when temperature is the primary driver, with changes of Arctic tundra as a response to significant warming as the best example (Settele et al. 2015). Even though distinguishing climate induced changes from land use changes is challenging, Boit et al. (2016) suggest that 5-6% of biomes in South America will undergo biome shifts until 2100, regardless of scenario, attributed to climate change. The projected biome shifts are primarily forests shifting to shrubland and dry forests becoming fragmented and isolated from more humid forests (Boit et al. 2016). Boreal forests are subject to unprecedented warming in terms of speed and amplitude (IPCC 2013b), with significant impacts on their regional distribution (Juday et al. 2015). Globally, tree lines are generally expanding northward and to higher elevations, or remaining stable, while a reduction in tree line was rarely observed and only where disturbances occurred (Harsch et al. 2009) There is *limited evidence* of a slow northward migration of the boreal forest in eastern North America (Gamache and Payette 2005). The thawing of permafrost may increase drought induced tree mortality throughout the circumboreal zone (Gauthier et al. 2015).

Forests are a prime regulator of hydrological cycling, both fluxes of atmospheric moisture and precipitation, hence climate and forests are inextricably linked (Ellison et al. 2017; Keys et al. 2017). Forest management influences the storage and flow of water in forested watersheds, particularly harvesting, thinning and construction of roads increase the likelihood of floods as an outcome of extreme climate events (Eisenbies et al. 2007). Water balance of at least partly forested landscapes is to a large extent controlled by forest ecosystems (Sheil and Murdiyarso 2009; Pokam et al. 2014). This includes surface runoff, as determined by evaporation and transpiration and soil conditions, and water flow routing (Eisenbies et al. 2007). Water use efficiency (i.e., the ratio of water loss to biomass gain) is increasing with increased CO_2 levels (Keenan et al. 2013), hence transpiration is predicted to decrease which in turn will increase surface runoff (Schlesinger and Jasechko 2014). However, the interaction of several processes makes predictions challenging (Frank et al. 2015; Trahan and Schubert 2016). Surface runoff is an important agent in soil erosion.

Generally, removal of trees through harvesting or forest death (Anderegg et al. 2012) will reduce transpiration and hence increase the runoff during the growing season. Management induced soil disturbance (such as skid trails and roads) will affect water flow routing to rivers and streams (Zhang et al. 2017; Luo et al. 2018; Eisenbies et al. 2007).

Climate change affects forests in both positive and negative ways (Trumbore et al. 2015; Price et al. 2013) and there will be regional and temporal differences in vegetation responses (Hember et al. 2017; Midgley and Bond 2015). Several climate change related drivers interact in complex ways, such as warming, changes in precipitation and water balance, CO_2 fertilisation, and nutrient cycling, which makes projections of future net impacts challenging (see 2.3.1.2) (Kurz et al. 2013; Price et al. 2013). In high latitudes, a warmer climate will extend the growing seasons which however, could be constrained by summer drought (Holmberg et al. 2019) while increasing levels of atmospheric CO_2 will increase water use efficiency but not necessarily tree growth (Giguère-Croteau et al. 2019). Improving one growth limiting factor will only enhance tree growth if other factors are not limiting (Norby et al. 2010; Trahan and Schubert 2016; Xie et al. 2016; Frank et al. 2015). Increasing forest productivity has been observed in most of Fennoscandia (Kauppi et al. 2014; Henttonen et al. 2017), Siberia and the northern reaches of North America as a response to a warming trend (Gauthier et al. 2015) but increased warming may also decrease forest productivity and increase risk of tree mortality and natural disturbances (Price et al. 2013; Girardin et al. 2016; Beck et al. 2011; Hember et al. 2016; Allen et al. 2011). The climatic conditions in high latitudes are changing at a magnitude faster than the ability of forests to adapt with detrimental, yet unpredictable, consequences (Gauthier et al. 2015).

Negative impacts dominate, however, and have already been documented (Lewis et al. 2004; Bonan et al. 2008; Beck et al. 2011) and are predicted to increase (Miles et al. 2004 ; Allen et al. 2010; Gauthier et al. 2015; Girardin et al. 2016; Trumbore et al. 2015). Several authors have emphasized a concern that tree mortality (forest dieback) will increase due to climate induced physiological stress as well as interactions between physiological stress and other stressors, such as insect pests, diseases, and wildfires (Anderegg et al. 2012; Sturrock et al. 2011; Bentz et al. 2010; McDowell et al. 2011). Extreme events such as extreme heat and drought, storms, and floods also pose increased threats to forests in both high and low latitude forests (Lindner et al. 2010; Mokria et al. 2015). However, comparing observed forest dieback with modelled climate induced damages did not show a general link between climate change and forest dieback (Steinkamp and Hickler 2015). Forests are subject to increasing frequency and intensity of wildfires which is projected to increase substantially with continued climate change (see also Cross-Chapter Box 3: Fire and climate change, Chapter 2) (Price et al. 2013). In the tropics, interaction between climate change, CO_2 and fire could lead to abrupt shifts between woodland and grassland dominated states in the future (Shanahan et al. 2016).

Within the tropics, much research has been devoted to understanding how climate change may alter regional suitability of various crops. For example coffee is expected to be highly sensitive to both temperature and precipitation changes, both in terms of growth and yield and in terms of increasing problems of pests (Ovalle-Rivera et al. 2015). Some studies conclude that the global area of coffee production will decrease by 50% (Bunn et al. 2015). Due to increased heat stress, the suitability of Arabica coffee is expected to deteriorate in Mesoamerica, while it can improve in high altitude areas in South America. The general pattern is that the climatic suitability for Arabica coffee will deteriorate at low altitudes of the tropics as well as at the higher latitudes (Ovalle-Rivera et al. 2015). This means that climate change in and of itself can render unsustainable previously sustainable land use and land management practices and vice versa (Laderach et al. 2011).

Rangelands are projected to change in complex ways due to climate change. Increasing levels of atmospheric CO_2 stimulate directly plant growth and can potentially compensate negative effects from drying by increasing rain use efficiency. But the positive effect of increasing CO_2 will be mediated by other environmental conditions, primarily water availability but also nutrient cycling, fire regimes and invasive species. Studies over the North American rangelands suggest, for example, that warmer and dryer climatic conditions will reduce NPP in the southern Great Plains, the Southwest, and northern Mexico, but warmer and wetter conditions will increase NPP in the northern Plains and southern Canada (Polley et al. 2013).

4.4.1.3 Coastal erosion

Coastal erosion is expected to increase dramatically by sea level rise and in some areas in combination with increasing intensity of cyclones (highlighted in Section 4.9.6). Cyclone induced coastal erosion). Coastal regions are also characterised by high population density, particularly in Asia (Bangladesh, China, India, Indonesia, Vietnam), whereas the highest population increase in coastal regions is projected in Africa (East Africa, Egypt, and West Africa) (Neumann et al. 2015). For coastal regions worldwide, and particularly in developing countries with high population density in low-lying coastal areas, limiting the warming to 1.5˚C to 2.0 ˚C will have major socio-economic benefits compared with higher temperature scenarios (IPCC 2018a; Nicholls et al. 2018). For more in-depth discussions on coastal process, please refer to Chapter 4 of the upcoming IPCC Special Report on The Ocean and Cryosphere in a Changing Climate (IPCC SROCC).

Despite the uncertainty related to the responses of the large ice sheets of Greenland and west Antarctica, climate change-induced sea level rise is largely accepted and represents one of the biggest threats faced by coastal communities and ecosystems (Nicholls et al. 2011; Cazenave and Cozannet 2014; DeConto and Pollard 2016; Mengel et al. 2016). With significant socio-economic effects, the physical impacts of projected sea level rise, notably coastal erosion, have received considerable scientific attention (Nicholls et al. 2011; Rahmstorf 2010; Hauer et al. 2016).

Rates of coastal erosion or recession will increase due to rising sea levels and in some regions also in combination with increasing oceans waves (Day and Hodges 2018; Thomson and Rogers 2014; McInnes et al. 2011; Mori et al. 2010), lack or absence of sea-ice (Savard et al. 2009; Thomson and Rogers 2014) and thawing of permafrost (Hoegh-Guldberg et al. 2018), and changing cyclone paths (Tamarin-Brodsky and Kaspi 2017; Lin and Emanuel 2016a). The respective role of the different climate factors in the coastal erosion process will vary spatially. Some studies have shown that the role of sea level rise on the coastal erosion process can be less important than other climate factors, like wave heights, changes in the frequency of the storms, and the cryogenic processes (Ruggiero 2013; Savard et al. 2009). Therefore, in order to have a complete picture of the potential effects of sea level rise on rates of coastal erosion, it is crucial to consider the combined effects of the aforementioned climate controls and the geomorphology of the coast under study.

Coastal wetlands around the world are sensitive to sea-level rise. Projections of the impacts on global coastlines are inconclusive, with some projections suggesting that 20% to 90% (depending on sea-level rise scenario) of present day wetlands will disappear during the 21st century (Spencer et al. 2016). Another study, which included natural feed-back processes and management responses suggested that coastal wetlands may actually increase (Schuerch et al. 2018b).

Low-lying coastal areas in the tropics are particularly subject to the combined effect of sea-level rise and increasing intensity of tropical cyclones, conditions which in many cases pose limits to adaptation, see section 4.8.5.1.

Many large coastal deltas are subject to the additional stress of shrinking deltas as a consequence of the combined effect of reduced sediment loads from rivers due to damming and water use, and land subsidence resulting from extraction of ground water or natural gas, and aquaculture (Higgins et al. 2013; Tessler et al. 2016; Minderhoud et al. 2017; Tessler et al. 2015; Brown and Nicholls 2015; Szabo et al. 2016; Yang et al. 2019; Shirzaei and Bürgmann 2018; Wang et al. 2018; Fuangswasdi et al. 2019). In some cases the rate of subsidence can outpace the rate of sea level rise by one order of magnitude (Minderhoud et al. 2017) or even two (Higgins et al. 2013). Recent findings from the Mississippi Delta raises the risk of a systematic underestimation of the rate of land subsidence in coastal deltas (Keogh and Törnqvist 2019)

In sum, from a land degradation point of view, low lying coastal areas are particularly exposed to the nexus of climate change and increasing concentration of people (Elliott et al. 2014) (*robust evidence, high agreement*) and the situation will become particularly acute in delta areas shrinking from both reduced sediment loads and land subsidence (*robust evidence, high agreement*).

4.4.2 Indirect impacts on land degradation

Indirect impacts of climate change on land degradation are difficult to quantify because of the many conflating factors. The causes of land-use change are complex, combining physical, biological and socioeconomic drivers (Lambin et al. 2001; Lambin and Meyfroidt 2011). One such driver of land-use change is the degradation of agricultural land, which can result in a negative cycle of natural land being converted to agricultural land to sustain production levels. The intensive management of agricultural land can lead to a loss of soil function, negatively impacting the many ecosystem services provided by soils including maintenance of water quality and soil carbon sequestration (Smith et al. 2016a). The degradation of soil quality due to cropping is of particular concern in tropical regions, where it results in a loss of productive potential of the land, affecting regional food security and driving conversion of non-agricultural land, such as forestry, to agriculture (Lambin et al. 2003; Drescher et al. 2016; Van der Laan et al. 2017). Climate change will exacerbate these negative cycles unless sustainable land managed practices are implemented.

Climate change impacts on agricultural productivity (see Chapter 5) will have implications for the intensity of land use and hence exacerbate the risk of increasing land degradation. There will be both localised effects (i.e., climate change impacts on productivity affecting land use in the same region) and teleconnections (i.e., climate change impacts and land-use change are spatially and temporally separate) (Wicke et al. 2012; Pielke et al. 2007). If global temperature increases beyond 3°C it will have negative yield impacts on all crops (Porter et al. 2014) which, in combination with a doubling of demands by 2050 (Tilman et al. 2011), and increasing competition for land from the expansion of negative emissions technologies (IPCC 2018a; Schleussner et al. 2016), will exert strong pressure on agricultural lands and food security.

In sum, reduced productivity of most agricultural crops will drive land-use changes worldwide (*robust evidence, medium agreement*), but predictions of how this will impact land degradation is challenging because of several conflating factors. Social change, such as widespread changes in dietary preferences will have a huge impact on agriculture and hence land degradation (*medium evidence, high agreement*).

4.5 Impacts of bioenergy and technologies for CO_2 removal (CDR) on land degradation

4.5.1 Potential scale of bioenergy and land-based CDR

In addition to the traditional land use drivers (e.g. population growth, agricultural expansion, forest management), a new driver will interact to increase competition for land throughout this century: the potential large-scale implementation of land-based technologies for CO_2 removal (CDR). Land-based CDR include afforestation and reforestation, bioenergy with carbon capture and storage (BECCS), soil carbon management, biochar and enhanced weathering (Smith et al., 2015; Smith 2016)

Most scenarios, including two of the four pathways in the IPCC Special Report on 1.5°C (IPCC 2018a), compatible with stabilisation at 2°C involve substantial areas devoted to land-based CDR, specifically afforestation/ reforestation and BECCS (Schleussner et al. 2016; Smith et al. 2016b; Mander et al. 2017). Even larger land areas are required in most scenarios aimed at keeping average global temperature increases to below 1.5 °C , and scenarios that avoid BECCS also require large areas of energy crops in many cases (IPCC 2018b), although some options with strict demand-side management avoid this need (Grubler et al. 2018). Consequently, the addition of carbon capture and storage (CCS) systems to bioenergy facilities enhances mitigation benefits because it increases the carbon retention time and reduces emissions relative to bioenergy facilities without CCS. The IPCC SR15 states that "When considering pathways limiting warming to 1.5°C with no or limited overshoot, the full set of scenarios shows a conversion of 0.5 – 11 M km^2 of pasture into 0 – 6 M km^2 for energy crops, a 2 M km^2 reduction to 9.5 M km^2 increase forest, and a 4 M km^2 decrease to a 2.5 M km^2 increase in non-pasture agricultural land for food and feed crops by 2050 relative to 2010." (Rogelj et al., 2018, p. 145) . For comparison, the global cropland area in 2010 was 15.9 M km^2 (Table 1.1), and (Woods et al. 2015) estimate the area of abandoned and degraded land potentially available for energy crops (or afforestation/reforestation) exceeds 5 M km^2. However, the area of available land has long been debated, as much marginal land is subject customary land tenure and used informally often by impoverished communities (Baka 2013, 2014; Haberl et al. 2013; Young 1999). Thus, as noted in the SR15, "The implementation of land-based mitigation options would require overcoming socio-economic, institutional, technological, financing and environmental barriers that differ across regions" (IPCC, 2018a, p. 18).

The wide range of estimates reflects the large differences among the pathways, availability of land in various productivity classes, types of NET implemented, uncertainties in computer models, and social and economic barriers to implementation (Fuss et al. 2018; Nemet et al. 2018; Minx et al. 2018).

4.5.2 Risks of land degradation from expansion of bioenergy and land-based CDR

The large-scale implementation of high intensity dedicated energy crops, and harvest of crop and forest residues for bioenergy, could contribute to increases in the area of degraded lands: intensive land management can result in nutrient depletion, over fertilisation and soil acidification, salinisation (from irrigation without adequate drainage), wet ecosystems drying (from increased evapotranspiration), as well as novel erosion and compaction processes (from high impact biomass harvesting disturbances) and other land degradation processes described in Section 4.2.1.

Global integrated assessment models used in the analyses of mitigation pathways vary in their approaches to modelling CDR (Bauer et al. 2018) and the outputs have large uncertainties due to their limited capability to consider site-specific details (Krause et al. 2018). Spatial resolutions vary from 11 world regions to 0.25 degrees gridcells (Bauer et al. 2018). While model projections identify potential areas for CDR implementation (Heck et al. 2018), the interaction with climate change induced biome shifts, available land and its vulnerability to degradation are unknown. The crop/forest types and management practices that will be implemented are also unknown, and will be influenced by local incentives and regulations. While it is therefore currently not possible to project the area at risk of degradation from the implementation of land-based CDR, there is a clear risk that expansion of energy crops at the scale anticipated could put significant strain on land systems, biosphere integrity, freshwater supply and biogeochemical flows (Heck et al. 2018). Similarly, extraction of biomass for energy from existing forests, particularly where stumps are utilized, can impact soil health (de Jong et al. 2017). Reforestation and afforestation present a lower risk of land degradation and may in fact reverse degradation (see Section 4.5.3) although potential adverse hydrological and biodiversity impacts will need to be managed (Caldwell et al. 2018; Brinkman et al. 2017). Soil carbon management can deliver negative emissions while reducing or reversing land degradation. Chapter 6 discusses the significance of context and management in determining environmental impacts of implementation of land-based options.

4.5.3 Potential contributions of land-based CDR to reducing and reversing land degradation

Although large-scale implementation of land-based CDR has significant potential risks, the need for negative emissions and the anticipated investments to implement such technologies can also create significant opportunities. Investments into land-based CDR can contribute to halting and reversing land degradation, to the restoration or rehabilitation of degraded and marginal lands (Chazdon and Uriarte 2016; Fritsche et al. 2017) and can contribute to the goals of land degradation neutrality (Orr et al. 2017a).

Estimates of the global area of degraded land range from less than 10 to 60 M km^2 (Gibbs and Salmon 2015), see also section 4.3.1. Additionally, large areas are classified as marginal lands and may be suitable for the implementation of bioenergy and land-based CDR (Woods et al. 2015). The yield per hectare of marginal and degraded lands is lower than on fertile lands, and if CDR will be implemented on marginal and degraded lands this will increase the area demand and costs per unit area of achieving negative emissions (Fritsche et al. 2017). Selection of lands suitable for CDR must be considered carefully to reduce conflicts with existing users, to assess the possible trade-offs in biodiversity contributions of the original and the CDR land uses, to quantify the impacts on water budgets, and to ensure sustainability of the CDR land use.

Land use and land condition prior to the implementation of CDR affect the climate change benefits (Harper et al. 2018). Afforestation/ reforestation on degraded lands can increase C stocks in vegetation and soil, increase carbon sinks (Amichev et al. 2012), and deliver co-benefits for biodiversity and ecosystem services particularly if a diversity of local species are used. Afforestation and reforestation on native grasslands can reduce soil carbon stocks, although the loss is typically more than compensated by increases in biomass and dead organic matter C stocks (Bárcena et al. 2014; Li et al. 2012; Ovalle-Rivera et al. 2015; Shi et al. 2013), and may impact biodiversity (Li et al. 2012) (see also 4.4.1: Large scale forest cover expansion, what can be learned in context of the SRCCL).

Strategic incorporation of energy crops into agricultural production systems, applying an integrated landscape management approach, can provide co-benefits for management of land degradation and other environmental objectives. For example, buffers of Miscanthus and other grasses can enhance soil carbon and reduce water pollution (Cacho et al. 2018; Odgaard et al. 2019), and strip-planting of short rotation tree crops can reduce the water table where crops are affected by dryland salinity (Robinson et al. 2006). Shifting to perennial grain crops has the potential to combine food production with carbon sequestration at a higher rate than with annual grain crops and avoid the trade-off between food production and climate change mitigation (Crews, Carton, & Olsson, 2018; de Oliveira, Brunsell, Sutherlin, Crews, & DeHaan, 2018; Ryan et al., 2018, see also 4.9.2).

Changes in land cover can affect surface reflectance, water balances and emissions of volatile organic compounds and thus the non-GHG impacts on the climate system from afforestation/reforestation or planting energy crops (Anderson et al. 2011; Bala et al. 2007; Betts 2000; Betts et al. 2007), (see Section 4.6 for further details). Some of these impacts reinforce the GHG mitigation benefits, while others off-set the benefits, with strong local (slope, aspect) and regional (boreal vs. tropical biomes) differences in the outcomes (Li et al. 2015). Adverse effects on albedo from afforestation with evergreen conifers in boreal zones can be reduced through planting of broadleaf deciduous species (Astrup et al. 2018; Cai et al. 2011a; Anderson et al. 2011).

Combining CDR technologies may prove synergistic. Two soil management techniques with an explicit focus on increasing the soil carbon content rather than promoting soil conservation more broadly have been suggested: Addition of biochar to agricultural soils (see 4.9.5) and addition of ground silicate minerals to soils in order to take up atmospheric CO_2 through chemical weathering (Taylor et al. 2017; Haque et al. 2019; Beerling 2017; Strefler et al. 2018). The addition of biochar is comparatively well understood and also field tested at large scale, see section 4.9.5 for a comprehensive discussion. The addition of silicate minerals to soils is still highly uncertain in terms of its potential (from 95 $GtCO_2\ yr^{-1}$ (Strefler et al. 2018) to only 2-4 $GtCO_2\ yr^{-1}$ (Fuss et al. 2018)) and costs (Schlesinger and Amundson 2018).

Effectively addressing land degradation through implementation of bioenergy and land-based CDR will require site-specific local knowledge, matching of species with the local land, water balance, nutrient and climatic conditions, and ongoing monitoring and, where necessary, adaptation of land management to ensure sustainability under global change (Fritsche et al. 2017). Effective land governance mechanisms including integrated land-use planning, along with strong sustainability standards could support deployment of energy crops and afforestation/reforestation at appropriate scales and geographical contexts (Fritsche et al. 2017). Capacity-building and technology transfer through the international cooperation mechanisms of the Paris Agreement could support such efforts. Modelling to inform policy development is most useful when undertaken with close interaction between model developers and other stakeholders including policymakers to ensure that models account for real world constraints (Dooley and Kartha 2018).

International initiatives to restore lands, such as the Bonn Challenge (Verdone and Seidl 2017) and the New York Declaration on Forests (Chazdon et al. 2017), and interventions undertaken for Land Degradation Neutrality and implementation of NDCs (see Glossary) can contribute to NET objectives. Such synergies may increase the financial resources available to meet multiple objectives (see section 4.8.4).

4.5.4 Traditional biomass provision and land degradation

Traditional biomass (fuelwood, charcoal, agricultural residues, animal dung) used for cooking and heating by some 2.8 billion people (38% of global population) in non-OECD countries accounts for more than half of all bioenergy used worldwide (IEA 2017; REN21 2018; see Cross-Chapter Box 7 on Bioenergy, Chapter 6). Cooking with traditional biomass has multiple negative impacts on human health, particularly for women, children and youth (Machisa et al. 2013; Sinha and Ray 2015; Price 2017; Mendum and Njenga 2018; Adefuye et al. 2007) and on household productivity including high workloads for women and youth (Mendum and Njenga 2018; Brunner et al. 2018; Hou et al. 2018; Njenga et al. 2019). Traditional biomass is land-intensive due to reliance on open fires, inefficient stoves and overharvesting of woodfuel, contributing to land degradation, losses in biodiversity and reduced ecosystem services (IEA 2017; Bailis et al. 2015; Masera et al. 2015; Specht et al. 2015; Fritsche et al. 2017; Fuso Nerini et al. 2017). Traditional woodfuels account for 1.9-2.3% of global GHG emissions, particularly in “hotspots” of land degradation and fuelwood depletion in eastern Africa and South Asia, such that one-third of traditional woodfuels globally are harvested unsustainably (Bailis et al. 2015). Scenarios to significantly reduce reliance on traditional biomass in developing countries present multiple co-benefits (*high evidence*, *high agreement*), including reduced emissions of black carbon, a short-lived climate forcer that also causes respiratory disease (Shindell et al. 2012).

A shift from traditional to modern bioenergy, especially in the African context, contributes to improved livelihoods and can reduce land degradation and impacts on ecosystem services (Smeets et al. 2012; Gasparatos et al. 2018; Mudombi et al. 2018). In Sub-Saharan Africa, most countries mention woodfuel in their Nationally Determined Contribution (NDC) but fail to identify transformational processes to make fuelwood a sustainable energy source compatible with improved forest management (Amugune et al. 2017). In some regions, especially in South and Southeast Asia, a scarcity of woody biomass may lead to excessive removal and use of agricultural wastes and residues, which contributes to poor soil quality and land degradation (Blanco-Canqui and Lal 2009; Mateos et al. 2017).

In sub-Saharan Africa, forest degradation is widely associated with charcoal production although in some tropical areas rapid re-growth can offset forest losses (Hoffmann et al. 2017; McNicol et al. 2018). Overharvesting of wood for charcoal contributes to the high rate of deforestation in sub-Saharan Africa, which is five times the world average, due in part to corruption and weak governance systems (Sulaiman et al. 2017). Charcoal may also be a by-product of forest clearing for agriculture, with charcoal sale providing immediate income when the land is cleared for food crops (Kiruki et al. 2017; Ndegwa et al. 2016). Besides loss of forest carbon stock, a further concern for climate change is methane and black carbon emissions from fuelwood burning and traditional charcoal-making processes (Bond et al. 2013; Patange et al. 2015; Sparrevik et al. 2015).

A fundamental difficulty in reducing environmental impacts associated with charcoal lies in the small-scale nature of much charcoal production in sub-Saharan Africa leading to challenges in regulating its production and trade, which is often informal, and in some cases illegal, but nevertheless widespread since charcoal is the most important urban cooking fuel (Zulu 2010; Zulu and Richardson 2013; Smith et al. 2015; World Bank 2009) (World Bank, 2009). Urbanisation combined with population growth has led to continuously increasing charcoal production. Low efficiency of traditional charcoal production results in a four-fold increase in raw woody biomass required and thus much greater biomass harvest (Hojas-Gascon et al. 2016; Smeets et al. 2012). With continuing urbanisation anticipated, increased charcoal production and use will probably contribute to increasing land pressures and increased land degradation, especially in sub-Saharan Africa *(medium evidence, high agreement)*.

Although it could be possible to source this biomass more sustainably, the ecosystem and health impacts of this increased demand for cooking fuel would be reduced through use of other renewable fuels or in some cases, non-renewable fuels (LPG), as well as through improved efficiency in end-use and through better resource and supply chain management (Santos et al. 2017; Smeets et al. 2012; Hoffmann et al. 2017). Integrated response options such as agro-forestry (see Chapter 6) and good governance mechanisms for forest and agricultural management (see Chapter 7) can support the transition to sustainable energy for households and reduce the environmental impacts of traditional biomass.

4.6 Impacts of land degradation on climate

While Chapter 2 has its focus on land cover changes and their impacts on the climate system, this chapter focuses on the influences of individual land degradation processes on climate (see cross chapter Table 4.1) which may or may not take place in association to land cover changes. The effects of land degradation on CO_2 and other greenhouse gases as well as those on surface albedo and other physical controls of the global radiative balance are discussed.

4.6.1 Impacts on greenhouse gases

Land degradation processes with direct impact on soil and terrestrial biota have great relevance in terms of CO_2 exchange with the atmosphere given the magnitude and activity of these reservoirs in the global C cycle. As the most widespread form of soil degradation, erosion detaches the surface soil material which typically hosts the highest organic C stocks, favoring the mineralisation and release as CO_2, yet complementary processes such as C burial may compensate this effect, making soil erosion a long-term C sink (*low agreement, limited evidence)*, (Wang et al. 2017b), but see also (Chappell et al. 2016). Precise estimation of the CO_2 released from eroded lands is challenged by the fact that only a fraction of the detached C is eventually lost to the atmosphere. It is important to acknowledge that a substantial fraction of the eroded material may preserve its organic C load in field conditions. Moreover, C sequestration may be favored through the burial of both the deposited material and the surface of its hosting soil at the deposition location (Quinton et al. 2010). The cascading effects of erosion on other environmental processes at the affected sites can often cause net CO_2 emissions through their indirect influence on soil fertility and the balance of organic C inputs and outputs, interacting with other non-erosive soil degradation processes such as nutrient depletion, compaction and salinisation, which can lead to the same net C effects (see Table 4.1) (van de Koppel et al. 1997).

As natural and human-induced erosion can result in net C storage in very stable buried pools at the deposition locations, degradation in those locations has a high C-release potential. Coastal ecosystems such as mangrove forests, marshes and seagrasses are a typical deposition locations and their degradation or replacement with other vegetation is resulting in a substantial C release (0.15 to 1.02 Gt C yr^{-1}) (Pendleton et al. 2012), which highlights the need for a spatially-integrated assessment of land degradation impacts on climate that considers in-situ but also ex-situ emissions.

Cultivation and agricultural management of cultivated land are relevant in terms of global CO_2 land-atmosphere exchange (see also 4.8.1). Besides the initial pulse of CO_2 emissions associated with the onset of cultivation and associated vegetation clearing (see Chapter 2), agricultural management practices can increase or reduce C losses to the atmosphere. Although global croplands are considered to be at relatively neutral stage in the current decade (Houghton et al. 2012), this results from a highly uncertain balance between coexisting net losses and gains. Degradation losses of soil and biomass carbon appear to be compensated by gains from soil protection and restoration practices such as cover crops, conservation tillage and nutrient replenishment favoring organic matter build-up. Cover crops, increasingly used to improve soils, have the potential to sequester 0.12 Gt C yr^{-1} on global croplands with a saturation time of more than 150 years (Poeplau and Don 2015). No-till practices (i.e. tillage elimination favoring crop residue retention in the soil surface) which were implemented to protect soils from erosion and reduce land preparation times, were also seen with optimism as a C sequestration option, which today is considered more modest globally and, in some systems, even less certain (VandenBygaart 2016; Cheesman et al. 2016; Powlson et al. 2014). Among soil fertility restoration practices, lime application for acidity correction, increasingly important in tropical regions, can generate a significant net CO_2 source in some soils (Bernoux et al. 2003, Alemu et al 2017).

Land degradation processes in seminatural ecosystems driven by unsustainable uses of their vegetation through logging or grazing lead to reduced plant cover and biomass stocks, causing net C releases from soils and plant stocks. Degradation by logging activities is particularly important in developing tropical and subtropical regions, involving C releases that exceed by far the biomass of harvested products, including additional vegetation and soil sources that are estimated to reach 0.6 Gt C yr^{-1} (Pearson et al. 2014, 2017). Excessive grazing pressures pose a more complex picture with variable magnitudes and even signs of C exchanges. A general trend of higher C losses in humid overgrazed rangelands suggests a high potential for C sequestration following the rehabilitation of those systems (Conant and Paustian 2002) with a global potential sequestration of 0.045 Gt C yr^{-1}. A special case of degradation in rangelands are those processes leading to the woody encroachment of grass-dominated systems, which can be responsible of declining animal production but high C sequestration rates (Asner et al. 2003, Maestre et al. 2009).

Fire regime shifts in wild and seminatural ecosystems can become a degradation process in itself, with high impact on net C emission and with underlying interactive human and natural drivers such as burning policies (Van Wilgen et al. 2004), biological invasions (Brooks et al. 2009), and plant pest/disease spread (Kulakowski et al. 2003). Some of these interactive processes affecting unmanaged forests have resulted in massive C release, highlighting how degradation feedbacks on climate are not restricted to intensively used land but can affect wild ecosystems as well (Kurz et al. 2008).

Agricultural land and wetlands represent the dominant source of non-CO_2 greenhouse gases (Chen et al. 2018d). In agricultural land, the expansion of rice cultivation (increasing CH_4 sources), ruminant stocks and manure disposal (increasing CH_4, N_2O and NH_3 fluxes) and nitrogen over-fertilisation combined with soil acidification (increasing N_2O fluxes) are introducing the major impacts (*medium agreement, medium evidence*) and their associated emissions appear to be exacerbated by global warming (*medium agreement and medium evidence)* (Oertel et al. 2016).

As the major sources of global N_2O emissions, over-fertilisation and manure disposal are not only increasing in-situ sources but also stimulating those along the pathway of dissolved inorganic nitrogen transport all the way from draining waters to the ocean (*high agreement, medium evidence*). Current budgets of anthropogenically fixed nitrogen on the Earth System (Tian et al. 2015; Schaefer et al. 2016; Wang et al. 2017a) suggest that N_2O release from terrestrial soils and wetlands accounts for 10-15% of the inputs, yet many further release fluxes along the hydrological pathway remain uncertain, with emissions from oceanic "dead-zones" being a major aspect of concern (Schlesinger 2009; Rabalais et al. 2014).

Environmental degradation processes focused on the hydrological system, which are typically manifested at the landscape scale, include both drying (as in drained wetlands or lowlands) and wetting trends (as in waterlogged and flooded plains). Drying of wetlands reduces CH_4 emissions (Turetsky et al. 2014) but favors pulses of organic matter mineralization linked to high N_2O release (Morse and Bernhardt 2013; Norton et al. 2011). The net warming balance of these two effects is not resolved and may be strongly variable across different types of wetlands. In the case of flooding of non-wetland soils, a suppression of CO_2 release is typically over compensated in terms of net greenhouse impact by enhanced CH_4 fluxes, that stem from the lack of aeration but are aided by the direct effect of extreme wetting on the solubilisation and transport of organic substrates (McNicol and Silver 2014). Both wetlands rewetting/restoration and artificial creation can increase CH_4 release (Altor and Mitsch 2006; Fenner et al. 2011). Permafrost thawing is another major source of CH_4 release with substantial long-term contributions to the atmosphere that are starting to get globally quantified (Christensen et al. 2004; Schuur et al. 2015; Walter Anthony et al. 2016).

4.6.2 Physical impacts

Among the physical effects of land degradation, surface albedo changes are those with the most evident impact on the net global radiative balance and net climate warming/cooling. Degradation processes affecting wild and semi-natural ecosystems such as fire regime changes, woody encroachment, logging and overgrazing can trigger strong albedo changes before significant biogeochemical shifts take place, in most cases these two types of effects have opposite signs in terms of net radiative forcing, making their joint assessment critical for understanding climate feedbacks (Bright et al. 2015).

In the case of forest degradation or deforestation, the albedo impacts are highly dependent on the latitudinal/climatic belt to which they belong. In boreal forests the removal or degradation of the tree cover increases albedo (net cooling effect)(*medium evidence, high agreement*) as the reflective snow cover becomes exposed, which can exceed the net radiative effect of the associated C release to the atmosphere (Davin et al. 2010; Pinty et al. 2011). On the other hand, progressive greening of boreal and temperate forests has contributed to net albedo declines (*medium agreement, medium evidence*) (Planque et al. 2017; Li et al. 2018a). In the northern treeless vegetation belt (tundra), shrub encroachment leads to the opposite effect as the emergence of plant structures above the snow cover level reduce winter-time albedo (Sturm 2005).

The extent to which albedo shifts can compensate carbon storage shifts at the global level has not been estimated. A significant but partial compensation takes place in temperate and subtropical dry ecosystems in which radiation levels are higher and C stocks smaller compared to their more humid counterparts (*medium agreement, medium evidence*). In cleared dry woodlands half of the net global warming effect of net C release has been compensated by albedo increase (Houspanossian et al. 2013), whereas in afforested dry rangelands albedo declines cancelled one fifth of the net C sequestration (Rotenberg and Yakir 2010). Other important cases in which albedo effects impose a partial compensation of C exchanges are the vegetation shifts associated to wild fires, as shown for the savannahs, shrublands and grasslands of sub-Saharan Africa (Dintwe et al. 2017). Besides the net global effects discussed above, albedo shifts can play a significant role on local climate (*high agreement, medium evidence*), as exemplified by the effect of no-till agriculture reducing local heat extremes in European landscapes (Davin et al. 2014) and the effects of woody encroachment causing precipitation rises in the North American Great Plains (Ge and Zou 2013). Modeling efforts that integrate ground data from deforested areas worldwide accounting for both physical and biogeochemical effects, indicate that massive global deforestation would have a net warming impact (Lawrence and Vandecar 2015) at both local and global levels with highlight non-linear effects of forest loss on climate variables.

Beyond the albedo effects presented above, other physical impacts of land degradation on the atmosphere can contribute to global and regional climate change. Of particular continental to global relevance are the net cooling effects of dust emissions (*low agreement, medium evidence*) (Lau and Kim 2007), but see also (Huang et al. 2014). Anthropogenic emission of mineral particles from degrading land appear to have a similar radiative impact than all other anthropogenic aerosols (Sokolik and Toon 1996). Dust emissions may explain regional climate anomalies through reinforcing feedbacks, as suggested for the amplification of the intensity, extent and duration of the low precipitation anomaly of the North American “Dust Bowl” in the 1930s (Cook et al. 2009). Another source of physical effects on climate are surface roughness changes which, by affecting atmospheric drag, can alter cloud formation and precipitation (*low agreement, low evidence*), as suggested by modeling studies showing how the massive deployment of solar panels in the Sahara could increase rainfall in the Sahel (Li et al. 2018c) or how woody encroachment in the Arctic tundra could reduce cloudiness and raise temperature (Cho et al. 2018). The complex physical effects of deforestation, as explored through modeling, converge into general net regional precipitation declines, tropical temperature increases and boreal temperature declines, while net global effects are less certain (Perugini et al. 2017). Integrating all the physical effects of land degradation and its recovery or reversal is still challenge, yet modeling attempts suggest that over the last three decades the slow but persistent net global greening caused by the average increase of leaf area in the land has caused a net cooling of the Earth, mainly through the raise of evapotranspiration (Zeng et al. 2017) (*low confidence*).

4.7 Impacts of climate-related land degradation on poverty and livelihoods

Unravelling the impacts of climate-related land degradation on poverty and livelihoods is highly challenging. This complexity is due to the interplay of multiple social, political, cultural, and economic factors, such as markets, technology, inequality, population growth, (Barbier and Hochard 2018) each of which interact and shape the ways in which social-ecological systems respond (Morton 2007). We find *limited evidence* attributing the impacts of climate-related land degradation to poverty and livelihoods, with climate often not distinguished from any other driver of land degradation. Climate is nevertheless frequently noted as a risk multiplier for both land degradation and poverty (*high agreement, robust evidence*) and is one of many stressors people live with, respond to and adapt to in their daily lives (Reid and Vogel 2006). Climate change is considered to exacerbate land degradation and potentially accelerate it due to heat stress, drought, changes to evapotranspiration rates and biodiversity, as well as a result of changes to environmental conditions that allow new pests and diseases to thrive (Reed and Stringer 2016b). In general terms, the climate (and climate change) can increase human and ecological communities' sensitivity to land degradation. Land degradation then leaves livelihoods more sensitive to the impacts of climate change and extreme climatic events (*high agreement, robust evidence*). If human and ecological communities exposed to climate change and land degradation are sensitive and cannot adapt, they can be considered vulnerable to it; if they are sensitive and can adapt, they can be considered resilient (Reed and Stringer 2016b). The impacts of land degradation will vary under a changing climate both spatially and temporally, leading some communities and ecosystems to be more vulnerable or more resilient than others under different scenarios. Even within communities, groups such as women and the youth are often more vulnerable than others.

4.7.1 Relationships between land degradation, climate change and poverty

This section sets out the relationships between land degradation and poverty, and climate change and poverty, leading to inferences about the 3-way links between them. Poverty is multidimensional and includes a lack of access to the whole range of capital assets that can be used to pursue a livelihood. Livelihoods constitute the capabilities, assets, and activities that are necessary to make a living (Chambers and Conway 1992; Olsson et al. 2014b).

The literature shows *high agreement* in terms of speculation that there are <u>potential</u> links between land degradation and poverty. However, studies have not provided robust quantitative assessments of the extent and incidence of poverty within land degradation affected populations (Barbier and Hochard 2016). Some researchers, e.g. Nachtergaele et al. (2011) estimate that 1.5 billion people were dependent upon degraded land to support their livelihoods in 2007, while >42 % of the world's poor population inhabit degraded areas. However, there is overall *low confidence* in the evidence base, a lack of studies that look beyond the past and present, and the literature calls for more in-depth research to be undertaken on these issues (Gerber et al. 2014). Recent work by Barbier and Hochard (Barbier and Hochard 2018) points to biophysical constraints such as poor soils and limited rainfall which interact to limit land productivity, suggesting that those farming in climatically less favourable agricultural areas are challenged by poverty. Studies such as those by (Coomes et al. 2011), focusing on an area in the Amazon, highlight the importance of the initial conditions of land holding in the dominant (shifting) cultivation system in terms of long-term effects on household poverty and future forest cover, showing initial land tenure and socio-economic aspects can make some areas less favourable too.

Much of the qualitative literature is focused on understanding the livelihood and poverty impacts of degradation through a focus on subsistence agriculture, where farms are small, under traditional or informal tenure and where exposure to environmental (including climate) risks is high (Morton 2007). In these situations, the poor lack access to assets (financial, social, human, natural and physical) and in the absence of appropriate institutional supports and social protection, this leaves them sensitive and unable to adapt, so a vicious cycle of poverty and degradation can ensue. To further illustrate the complexity, livelihood assessments often focus on a single snapshot in time, livelihoods are dynamic and people alter their livelihood activities and strategies depending the on internal and external stressors to which they are responding (O'Brien et al. 2004). When certain livelihood activities and strategies become no longer tenable as a result of land degradation (and may push people into poverty), it can have further effects on issues such as migration (Lee 2009), as people adapt by moving (see Section 4.7.3); and may result in conflict (see Section 4.7.3), as different groups within society compete for scarce resources, sometimes through non-peaceful actions. Both migration and conflict can lead to land use changes elsewhere that further fuel climate change through increased emissions.

Similar challenges as for understanding land degradation-poverty linkages are experienced in unravelling the relationship between climate change and poverty. A particular issue in examining climate change-poverty links relates to the common use of aggregate economic statistics like GDP, as the assets and income of the poor constitute such as minor proportion of national wealth (Hallegatte et al. 2018). Aggregate quantitative measures also fail to capture the distributions of costs and benefits from climate change. Furthermore, people fall into and out of poverty, with climate change being one of many factors affecting these dynamics, through its impacts on livelihoods. Much of the literature on climate change and poverty tends to look backward rather than forward (Skoufias et al. 2011), providing a snap-shot of current or past relationships, (for example, (Dell et al. 2009) who examine the relationship between temperature and income (GDP) using cross-sectional data from countries in the Americas). Yet, simulations of future climate change impacts on income or poverty are largely lacking.

Noting the *limited evidence* that exists that explicitly focuses on the relationship between land degradation, climate change and poverty, Barbier and Hochard (2018b) suggest that those people living in less favoured agricultural areas face a poverty-environment trap that can result in increased land degradation under climate change conditions. The emergent relationships between land degradation, climate change and poverty are shown in Figure 4.6 (see also Figure 6.1).

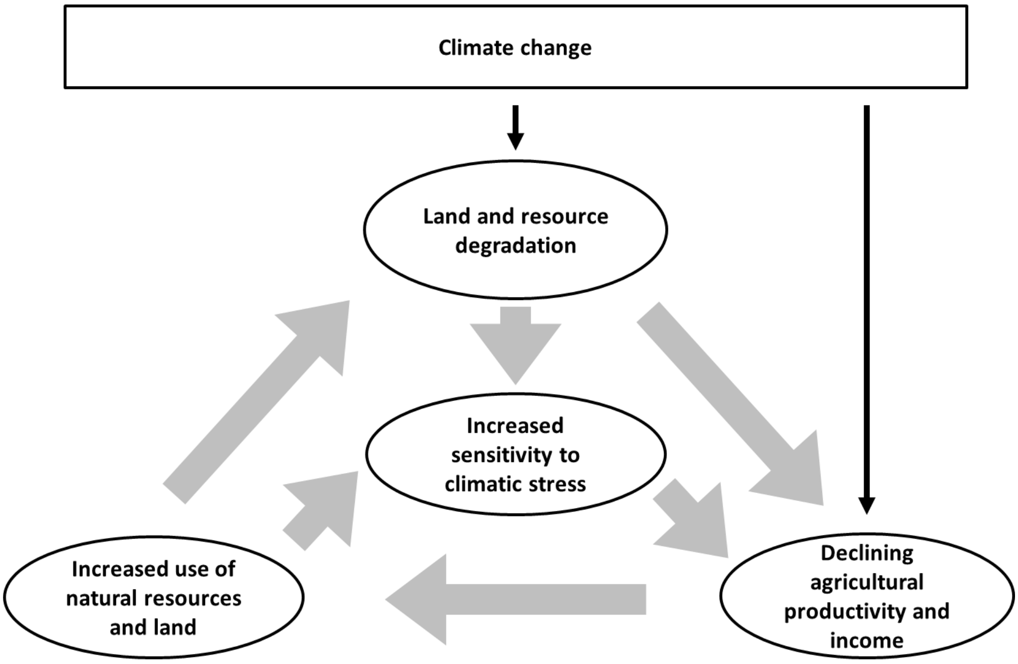


Figure 4.6 Schematic representation of links between climate change, land management and socio-economic conditions.

The poor have access to few productive assets, so land, and the natural resource base more widely, plays a key role in supporting the livelihoods of the poor. It is however, hard to make generalisations about how important income derived from the natural resource base is for rural livelihoods in the developing world (Angelsen et al. 2014) with studies focusing on forest resources having shown that approximately one quarter of the total rural household income in developing countries stems from forests, with forest-based income shares being tentatively higher for low-income households (Vedeld et al. 2007; Angelsen et al. 2014). Different groups use land in different ways within their overall livelihood portfolios and are therefore at different levels of exposure and sensitivity to climate shocks and stresses. The literature nevertheless displays *high evidence* and *high agreement* that those populations whose livelihoods are more sensitive to climate change and land degradation are often more dependent on environmental assets, and these people are often the poorest members of society. There is further *high evidence and high agreement* that both climate change and land degradation can affect livelihoods and poverty through their threat multiplier effect. Research in Bellona, in the Solomon Islands in the south Pacific (Reenberg et al. 2008) examined event-driven impacts on livelihoods, taking into account weather events as one of many drivers of land degradation and links to broader land-use and land cover changes that have taken place. Geographical locations experiencing land degradation are often the same locations that are directly affected by poverty, and are also affected by extreme events linked to climate change and variability.

Much of the assessment presented above has considered placed-based analyses examining the relationships between poverty, land degradation and climate change in the locations in which these outcomes have occurred. Altieri and Nicholls (2017) note that due to the globalised nature of markets and consumption systems, the impacts of changes in crop yields linked to climate-related land degradation (manifest as lower yields) will be far reaching, beyond the sites and livelihoods experiencing degradation. Despite these teleconnections, farmers living in poverty in developing countries will be especially vulnerable due to their exposure, dependence on the environment for income and limited options to engage in other ways to make a living (Rosenzweig and Hillel 1998). In identifying ways in which these interlinkages can be addressed, (Scherr 2000) observes that key actions that can jointly address poverty and environmental improvement often seek to increase access to natural resources, enhance the productivity of the natural resource assets of the poor, and to engage stakeholders in addressing public natural resource management issues. In this regard, it is increasingly recognised that those suffering from and being vulnerable to land degradation and poverty need to have a voice and play a role in the development of solutions, especially where the natural resources and livelihood activities they depend on are further threatened by climate change.

4.7.2 Impacts of climate related land degradation on food security

How and where we grow food compared to where and when we need to consume it is at the crux of issues surrounding land degradation, climate change and food security, especially because more than 75% of the global land surface (excluding Antarctica) faces rain-fed crop production constraints (Fischer et al. 2009), see also Chapter 5. Taken separately, knowledge on land degradation processes and human-induced climate change has attained a great level of maturity. However, their combined effects on food security, notably food supply, remain underappreciated (Webb et al. 2017b), and quantitative information is lacking. Just a few studies have shown how the interactive effects of the aforementioned challenging, interrelated phenomena can impact crop productivity and hence food security and quality (Karami et al. 2009; Allen et al. 2001; Högy and Fangmeier 2008) (*low evidence*). Along with socio-economic drivers climate change accelerates land degradation due to its influence on land-use systems (Millennium Assessment 2005; UNCCD 2017), potentially leading to a decline in agri-food system productivity, particularly on the supply side. Increases in temperature and changes in precipitation patterns are expected to have impacts on soil quality, including nutrient availability and assimilation (St.Clair and Lynch 2010). Those climate-related changes are expected to have net negative impacts on agricultural productivity, particularly in tropical regions, though the magnitude of impacts depends on the models used. Anticipated supply side issues linked to land and climate relate to biocapacity factors (including e.g. whether there is enough water to support agriculture); production factors (e.g. chemical pollution of soil and water resources or lack of soil nutrients) and distribution issues (e.g. decreased availability of and/or accessibility to the necessary diversity of quality food where and when it is needed) (Stringer et al. 2011). Climate sensitive transport infrastructure is also problematic for food security (Islam et al. 2017), and can lead to increased food waste, while poor siting of roads and transport links can lead to soil erosion and forest loss (Xiao et al. 2017), further feeding back into climate change.

Over the past decades, crop models have been useful tools for assessing and understanding climate change impacts on crop productivity and food security (White et al. 2011; Rosenzweig et al. 2014). Yet, the interactive effects of soil parameters and climate change on crop yields and food security remain limited, with *low evidence* of how they play out in different economic and climate settings (e.g. Sundström et al. 2014). Similarly, there have been few meta-analyses focusing on the adaptive capacity of land-use practices such as conservation agriculture in light of climate stress (see e.g. Steward et al. 2018), as well as *low evidence* quantifying the role of wild foods and forests (and by extension forest degradation) in both the global food basket and in supporting household scale food security (Bharucha and Pretty 2010; Hickey et al. 2016)

To be sustainable, any initiative aiming at addressing food security – encompassing supply, diversity and quality - must take into consideration the interactive effects between climate and land degradation in a context of other socio-economic stressors. Such socio-economic factors are especially important if we look at demand side issues too, which include lack of purchasing power, large scale speculation on global food markets leading to exponential price rises (Tadesse et al. 2014), competition in appropriation of supplies and changes to per capita food consumption (Stringer et al. 2011; see also Chapter 5). Lack of food security, combined with lack of livelihood options, is often an important manifestation of vulnerability, and can act as a key trigger for people to migrate. In this way, migration becomes an adaptation strategy.

4.7.3 Impacts of climate-related land degradation on migration and conflict

Land degradation may trigger competition for scarce natural resources potentially leading to migration and/or conflict, though even with *medium evidence* there is *low agreement* in the literature. Linkages between land degradation and migration occur within a larger context of multi-scale interaction of environmental and non-environmental drivers and processes, including resettlement projects, searches for education and/or income, land shortages, political turmoil, and family-related reasons (McLeman 2017; Hermans and Ide 2019). The complex contribution of climate to migration and conflict hampers retrieving any level of confidence on climate-migration and climate-conflict linkages, therefore constituting a major knowledge gap (Cramer et al. 2014; Hoegh-Guldberg et al. 2018).

There is *low evidence* on the causal linkages between climate change, land degradation processes (other than desertification) and migration. Existing studies on land degradation and migration – particularly in drylands – largely focus on the effect of rainfall variability and drought and shows how migration serves as adaptation strategy (Piguet et al. 2018; McLeman 2017; chapter 3). For example, in the Ethiopian highlands severe topsoil erosion and forest degradation is a major environmental stressor which is amplified by re-occurring droughts, with migration being an important household adaptation strategy (Morrissey 2013). In the humid tropics, land degradation, mainly as a consequence of deforestation, has been a reported reason for people leaving their homes during the Amazonian colonisation (Hecht 1983) but was also observed more recently, for example in Guatemala, where soil degradation was one of the most frequently cited migration pushes (López-Carr 2012) and Kenya, where households respond to low soil quality by sending temporary migrants for additional income generation (Gray 2011). In contrast, in the Andean highlands and the Pacific coastal plain, migration increased with land quality, probably because revenues from additional agricultural production was invested in costly forms of migration (Gray and Bilsborrow 2013). These mixed results illustrate the complex, non-linear relationship of land degradation-migration linkages and suggest explaining land degradation-migration linkages requires considering a broad socio-ecological embedding (McLeman 2017).

In addition to people moving away from an area due to "lost" livelihood activities, climate related land degradation can also reduce the availability of livelihood safety nets – environmental assets that people use during times of shocks or stress. For example, Barbier (2000) notes that wetlands in north-east Nigeria around Hadejia–Jama'are floodplain provide dry season pastures for seminomadic herders, agricultural surpluses for Kano and Borno states, groundwater recharge of the Chad formation aquifer and 'insurance' resources in times of drought. The floodplain also supports many migratory bird species. As climate change and land degradation combine, delivery of these multiple services can be undermined, particularly as droughts become more widespread, reducing the utility of this wetland environment as a safety net for people and wildlife alike.

Early studies conducted in Africa hint at a significant causal link between land degradation and violent conflict (Homer-Dixon et al. 1993). For example, Percival and Homer-Dixon (1995) identified land degradation as one of the drivers of the crisis in Rwanda in the early 1990s which allowed radical forces to stoke ethnic rivalries. With respect to the Darfur conflict, some scholars and UNEP concluded that land degradation, together with other environmental stressors, constitute a major security threat for the Sudanese people (Byers and Dragojlovic 2004; Sachs 2007; UNEP 2007). Recent studies show *low agreement*, suggesting that climate change can increase the likelihood of civil violence if certain economic, political and social factors, including low development and weak governance mechanisms, are present (Scheffran et al. 2012; Benjaminsen et al. 2012). In contrast, Raleigh (Raleigh and Urdal 2007) found in a global study that land degradation is a weak predictor for armed conflict. As such, studies addressing possible linkages between climate change – a key driver of land degradation – and the risks of conflict have yielded contradictory results and it remains largely unclear whether land degradation resulting from climate change leads to conflict or cooperation (Salehyan 2008; Solomon et al. 2018).

Land degradation-conflict linkages can be bi-directional. Research suggests that households experiencing natural resource degradation often engage in migration for securing livelihoods (Kreamer 2012), which potentially triggers land degradation at the destination leading to conflict there (Kassa et al. 2017). While this indeed holds true for some cases it may not for others given the complexity of processes, contexts and drivers. Where conflict and violence do ensue, it is often as a result of a lack of appreciation for the cultural practices of others.

4.8 Addressing land degradation in the context of climate change

Land degradation in the form of soil carbon loss is estimated to have been ongoing for at least 12,000 years, but increased exponentially in the last 200 years (Sanderman et al. 2017). Before the advent of modern sources of nutrients, it was imperative for farmers to maintain and improve soil fertility through the prevention of runoff and erosion, and management of nutrients through vegetation residues and manure. Many ancient farming systems were sustainable for hundreds and even thousands of years, such as raised field agriculture in Mexico (Crews and Gliessman 1991), tropical forest gardens in SE Asia and Central America (Ross 2011; Torquebiau 1992; Turner and Sabloff 2012), terraced agriculture in East Africa, Central America, Southeast Asia and the Mediterranean basin (Turner and Sabloff 2012; Preti and Romano 2014; Widgren and Sutton 2004; Håkansson and Widgren 2007; Davies and Moore 2016; Davies 2015), and integrated rice-fish cultivation in East Asia (Frei and Becker 2005).

Such long-term sustainable farming systems evolved in very different times and geographical contexts, but they share many common features, such as: the combination of species and structural diversity in time, and space (horizontally and vertically) in order to optimise the use of available land; recycling of nutrients through biodiversity of plants, animals, and microbes; harnessing the full range of site-specific micro-environments (e.g. wet and dry soils); biological interdependencies which helps suppression of pests; reliance on mainly local resources; reliance on local varieties of crops and sometimes incorporation of wild plants and animals; the systems are often labour and knowledge intensive (Rudel et al. 2016; Beets 1990; Netting 1993; Altieri and Koohafkan 2008). Such farming systems have stood the test of time and can provide important knowledge for adapting farming systems to climate change (Koohafkann and Altieri 2011).

In modern agriculture the importance of maintaining the biological productivity and ecological integrity of farm land has not been a necessity in the same way as in pre-modern agriculture because nutrients and water have been supplied externally. The extreme land degradation in the US Midwest during the Dust Bowl period in the 1930s became an important wake-up call for agriculture and agricultural research and development, from which we can still learn much in order to adapt to ongoing and future climate change (McLeman et al. 2014; Baveye et al. 2011; McLeman and Smit 2006).

Sustainable Land Management (SLM) is a unifying framework for addressing land degradation and can be defined as the stewardship and use of land resources, including soils, water, animals and plants, to meet changing human needs, while simultaneously ensuring the long-term productive potential of these resources and the maintenance of their environmental functions. ‘It is a comprehensive approach comprising technologies combined with social, economic and political enabling conditions (Nkonya et al. 2011). It is important to stress that farming systems are informed by both scientific and local/traditional knowledge. The power of SLM in small-scale diverse farming was demonstrated effectively in Nicaragua after the severe cyclone Mitch in 1998 (Holt-Giménez 2002). Pairwise analysis of 880 fields with and without implementation of SLM practices showed that the SLM fields systematically fared better than the fields without SLM in terms of more topsoil remaining, higher field moisture, more vegetation, less erosion and lower economic losses after the cyclone. Furthermore the difference between fields with and without SLM increased with increasing levels of storm intensity, increasing slope gradient, and increasing age of SLM (Holt-Giménez 2002).

When addressing land degradation through SLM and other approaches it is important to consider feedbacks that impact climate change. Table 4.2 shows some of the most important land degradation issues, their potential solutions, and their impacts on climate change. This table provides a link between the comprehensive lists of land degradation processes (Table 4.1) and land management solutions (Table 4.2).

Table 4.2 (Cross-chapter Ch 3 and Ch 4) Interaction of human and climate drivers can exacerbate desertification and land degradation

Climate change exacerbates the rate and magnitude of several ongoing land degradation and desertification processes. Human drivers of land degradation and desertification include expanding agriculture, agricultural practices and forest management. In turn land degradation and desertification are also drivers of climate change through the emission of greenhouse gases, reduced rates of carbon uptake and reduced capacity of ecosystems to act as carbon sinks into the future.

Human driver	Climate driver
Grazing pressure	Warming trend
Agriculture practices	Extreme temperatures
Expansion of agriculture	Drying trend
Forest clearing	Extreme rainfall
Wood fuel	Shifting rains
	Intensifying cyclones
	Sea level rise

Issue/syndrome	Impact on climate change	Human driver	Climate driver	Land management options	References	
Erosion of agricultural soils	Emission: CO_2, N_2O			Increase soil organic matter, no till, perennial crops, erosion control, agroforestry, dietary change	{3.2.4, 3.5.1, 3.6.2, 3.8.1, 4.9.1, 4.9.5, 4.10.2, 4.10.5}	{3.1.4,3.4.1, 3.5.2,3.7.1,4.8.1,4.8.5, 4.9.2,4.9.5}
Deforestation	Emission of CO_2			Forest protection, sustainable forest management and dietary change	{4.2.5, 4.6, 4.9.3, 4.9.4, 4.10.3}	{4.1.5,4.5,4.8.3,4.8.4,4.9.3}
Forest degradation	Emission of CO_2 Reduced carbon sink			Forest protection, sustainable forest management	{4.2.5, 4.6, 4.9.3, 4.9.4, 4.10.3}	{4.1.5,4.5,4.8.3,4.8.4,4.9.3}
Overgrazing	Emission: CO_2, CH_4 Increasing albedo			Controlled grazing, rangeland management	{3.2.4.2, 3.5.1, 3.7.1, 3.8.1, 4.9.1.4}	{3.1.4.2,3.4.1,3.6.1,3.7.1,4.8.1.4}
Firewood and charcoal production	Emission: CO_2, CH_4 Increasing albedo			Clean cooking (health co-benefits, particularly for women and children)	{3.7.3, 4.6.4, 4.9.3, 4.9.4}	{3.6.3,4.5.4,4.8.3,4.8.4}
Increasing fire frequency and intensity	Emission: CO_2, CH_4, N_2O Emission: aerosols, increasing albedo			Fuel management, fire management	{3.2.4, 3.7.1, 4.2.5, 4.9.3, Cross chapter box 3}	{3.1.4,3.6.1,4.1.5,4.8.3, Cross chapter box 3}
Degradation of tropical peat soils	Emission: CO_2, CH_4			Peatland restoration, erosion control, regulating the use of peat soils	{4.10.4}	{4.9.4}
Thawing of perma-frost	Emission: CO_2, CH_4			relocation of settlement and infrastructure	{4.9.5.1}	{4.8.5.1}
Coastal erosion	Emission: CO_2, CH_4			Wetland and coastal restoration, mangrove conservation, long term land use planning	{4.10.6, 4.10.7, 4.10.8}	{4.9.6,4.9.7,4.9.8}
Sand and dust storms, wind erosion	Emission: aerosols			Vegetation management, afforestation, windbreaks	{3.4.1, 3.5.1, 3.7.1, 3.8.1, 3.8.2}	{3.3.1,3.4.1,3.6.1,3.7.1,3.7.2}
Bush encroachment	Capturing: CO_2, Decreasing albedo			Grazing land management, fire management	{3.7.1.3, 3.8.3.2}	{3.6.1.3,3.7.3.2}

4.8.1 Actions on the ground to address land degradation

Concrete actions on the ground to address land degradation are primarily focused on soil and water conservation. In the context of adaptation to climate change, actions relevant for addressing land degradation are sometimes framed as ecosystem based adaptation (EBA) (Scarano 2017) or Nature Based Solutions (NBS) (Nesshöver et al. 2017), and in an agricultural context, agroecology (see glossary) provides an important frame. The site-specific biophysical and social conditions, including local and indigenous knowledge, are important for successful implementation of concrete actions.

Responses to land degradation generally take the form of agronomic measures (methods related to managing the vegetation cover), soil management (methods related to tillage, nutrient supply), and mechanical methods (methods resulting in durable changes to the landscape) (Morgan 2005a). Measures may be combined to reinforce benefits to land quality, as well as improving carbon sequestration that supports climate change mitigation. Some measures offer adaptation options and other co-benefits, such as agroforestry involving planting fruit trees that can support food security in the face of climate change impacts (Reed and Stringer 2016a) or application of compost or biochar that enhances soil water holding capacity, so increases resilience to drought.

There are important differences in terms of labour and capital requirements for different technologies, and also implications for land tenure arrangements. Agronomic measures and soil management require generally little extra capital input and comprise activities repeated annually, so have no particular implication for land tenure arrangements. Mechanical methods require substantial upfront investments in terms of capital and labour, resulting in long lasting structural change requiring more secure land tenure arrangements (Mekuriaw et al. 2018). Agroforestry is a particularly important strategy for SLM in the context of climate change because the large potential to sequester carbon in plants and soil and enhance resilience of agricultural systems (Zomer et al. 2016).

Implementation of sustainable land management practices has been shown to increase the productivity of land (Branca et al. 2013) and to provide good economic returns on investment in many different settings around the world (Mirzabaev et al. 2015). Giger et al (2018) showed in a meta study of 363 projects over the period 1990 to 2012 that 73% of the projects were perceived to have a positive or at least neutral cost/benefit ratio in the short term, and 97% were perceived to have a positive or very positive cost/benefit ratio in the long term (*robust evidence, high agreement*). Despite the positive effects, uptake is far from universal. Local factors, both biophysical conditions (e.g. soils, drainage, and topography) and socio-economic conditions (e.g. land tenure, economic status, and land fragmentation) play decisive roles in the interest in, capacity to undertake, and successful implementation of sustainable land management practices (Teshome et al. 2016; Vogl et al. 2017; Tesfaye et al. 2016; Cerdà et al. 2018; Adimassu et al. 2016). From a landscape perspective, sustainable land management can generate benefits, including adaptation to and mitigation of climate change, for entire watersheds, but challenges remain regarding coordinated and consistent implementation (Kerr et al. 2016; Wang et al. 2016a). (*medium evidence, medium agreement*)

4.8.1.1 Agronomic and soil management measures

Rebuilding soil carbon is an important goal of SLM, particularly in the context of climate change (Rumpel et al. 2018). The two most important reasons why agricultural soils have lost 20-60% of the soil carbon they contained under natural ecosystem conditions are the frequent disturbance through tillage and harvesting and the change from deep rooted perennial plants to shallow rooted annual plants (Crews and Rumsey 2017). Practices that build soil carbon are those that increase organic matter input to soil, or reduce decomposition of soil organic matter.

Agronomic practices can alter the carbon balance significantly, by increasing organic inputs from litter and roots into the soil. Practices include retention of residues, use of locally-adapted varieties, inter-cropping, crop rotations, and green manure crops that replace the bare field fallow during winter and are eventually ploughed before sowing next main crop (Henry et al., 2018). Cover crops (green manure crops and catch crops that are grown between the main cropping seasons) can increase soil carbon stock by between 0.22 and 0.4 t C $ha^{-1}yr^{-1}$ (Poeplau and Don 2015; Kaye and Quemada 2017).

Reduced tillage (or no-tillage) is an important strategy for reducing soil erosion and nutrient loss by wind and water (Van Pelt et al. 2017; Panagos et al. 2015; Borrelli et al. 2016). But the evidence that no-till agriculture also sequesters carbon is not compelling (VandenBygaart 2016). Soil sampling of only the upper 30 cm can give biased results suggesting that soils under no-till practices have higher carbon content than soils under conventional tillage (Baker et al. 2007; Ogle et al. 2012; Fargione et al. 2018; VandenBygaart 2016).

Changing from annual to perennial crops can increase soil carbon content (Culman et al. 2013; Sainju et al. 2017). A perennial grain crop (intermediate wheatgrass) was on average over four years a net carbon sink of about 13.5 t CO_2 $ha^{-1}yr^{-1}$ (de Oliveira et al. 2018). Sprunger et al. (2018) compared an annual winter wheat crop with a perennial grain crop (intermediate wheatgrass) and found that the perennial grain root biomass was 15 times larger than winter wheat, however, there was no significant difference in soil carbon pools after the four-year experiment. Exactly how much, and over what time period, carbon can be sequestered through changing from annual to perennial crops depends on the degree of soil carbon depletion and other local biophysical factors (see also section 4.9.2).

Integrated soil fertility management is a sustainable approach to nutrient management that uses a combination of chemical and organic amendments (manure, compost, biosolids, biochar), rhizobial nitrogen fixation, and liming materials to address soil chemical constraints (Henry et al., 2018). In pasture systems, management of grazing pressure, fertilisation, diverse species including legumes and perennial grasses can reduce erosion and enhance soil carbon (Conant et al. 2017).

4.8.1.2 Mechanical soil and water conservation

In hilly and mountainous terrain terracing is an ancient but still practiced soil conservation method worldwide (Preti and Romano 2014) in climatic zones from arid to humid tropics (Balbo 2017). By reducing the slope gradient of hillsides, terraces provide flat surfaces and deep, loose soils that increase infiltration, reduce erosion and thus sediment transport. They also decrease the hydrological connectivity and thus reduce hillside runoff (Preti et al. 2018; Wei et al. 2016; Arnáez et al. 2015; Chen et al. 2017). In terms of climate change, terraces are a form of adaptation which helps both in cases where rainfall is increasing or intensifying (by reducing slope gradient and the hydrological connectivity), and where rainfall is decreasing (by increasing infiltration and reducing runoff) (*robust evidence*, *high agreement*). There are several challenges, however, to continued maintenance and construction of new terraces, such as the high costs in terms of labour and/or capital (Arnáez et al. 2015) and disappearing local knowledge for maintaining and constructing new terraces (Chen et al. 2017). The propensity of farmers to invest in mechanical soil conservation methods varies with land tenure, farmers with secure tenure arrangements are more willing to invest in durable practices such as terraces (Lovo 2016; Sklenicka et al. 2015; Haregeweyn et al. 2015). Where the slope is less severe, erosion can be controlled by contour banks, and the keyline approach (Duncan 2016; Stevens et al. 2015) to soil and water conservation.

4.8.1.3 Agroforestry

Agroforestry is defined as a collective name for land-use systems in which woody perennials (trees, shrubs, etc.) are grown in association with herbaceous plants (crops, pastures) and/or livestock in a spatial arrangement, a rotation, or both, and in which there are both ecological and economic interactions between the tree and non-tree components of the system (Young, 1995, p. 11). At least since the 1980s agroforestry has been widely touted as an ideal land management practice in areas vulnerable to climate variations and subject to soil erosion. Agroforestry holds the promise of improving of soil and climatic conditions while generating income from wood energy, timber, and non-timber products – sometimes presented as a synergy of adaptation and mitigation of climate change (Mbow et al. 2014).

There is strong scientific consensus that a combination of forestry with agricultural crops and/or livestock, agroforestry systems can provide additional ecosystem services when compared with monoculture crop systems (Waldron et al. 2017; Sonwa et al. 2011a, 2014, 2017; Charles et al. 2013). Agroforestry can enable sustainable intensification by allowing continuous production on the same unit of land with higher productivity without the need to use shifting agriculture systems to maintain crop yields (Nath et al. 2016). This is especially relevant where there is a regional requirement to find a balance between the demand for increased agricultural production and the protection of adjacent natural ecosystems such as primary and secondary forests (Mbow et al. 2014). For example, the use of agroforestry for perennial crops such as coffee and cocoa are increasingly promoted as offering a route to sustainable farming with important climate change adaptation and mitigation co-benefits (Sonwa et al. 2001; Kroeger et al. 2017). Reported co-benefits of agroforestry in cocoa production include increased carbon sequestration in soils and biomass, improved water and nutrient use efficiency and the creation of a favourable micro-climate for crop production (Sonwa et al. 2017; Chia et al. 2016). Importantly, the maintenance of soil fertility using agroforestry has the potential to reduce the practice of shifting-agriculture (of cocoa) which results in deforestation (Gockowski and Sonwa 2011). However, positive interactions within these systems can be ecosystem and/or species specific, but co-benefits such as increased resilience to extreme climate events, or improved soil fertility are not always observed (Blaser et al. 2017; Abdulai et al. 2018). These contrasting outcomes indicate the importance of field scale research programs to inform agroforestry system design, species selection and management practices (Sonwa et al. 2014) .

Despite the many proven benefits, adoption of agroforestry has been low and slow (Toth et al. 2017; National Research Centre for Agroforestry et al. 1999; Pattanayak et al. 2003; Jerneck and Olsson 2014). There are several reasons for the slow uptake, but the perception of risks and the time lag between adoption and realisation of benefits are often important (Pattanayak et al. 2003; Mercer 2004; Jerneck and Olsson 2013).

An important question for agroforestry is whether it supports poverty alleviation, or if it favours comparatively affluent households. Experiences from India suggest that the overall adoption is (s)low and differential between rich and poor households. Brockington el al. (2016), studied agroforestry adoption over many years in South India, they found that overall only 18% of the households adopted agroforestry but among the relatively rich households who adopted agroforestry, 97% of them were still practicing it after 6-8 years and some had expanded their operations. Similar results were obtained in Western Kenya, that food secure households were much more willing to adopt agroforestry than food insecure households (Jerneck and Olsson 2013, 2014). Other experiences from sub-Saharan Africa illustrate the difficulties (such as local institutional support) of having a continued engagement of communities in agroforestry (Noordin et al. 2001; Matata et al. 2013; Meijer et al. 2015).

4.8.1.4 Crop-livestock interaction as an approach to manage land degradation

The integration of crop and livestock production into “mixed farming” for smallholders in developing countries became an influential model, particularly for Africa, in the early 1990s (Pritchard et al. 1992; McIntire et al. 1992). Crop-livestock integration under this model was seen as founded on three pillars; improved use of manure for crop fertility management; expanded use of animal traction (draught animals); and promotion of cultivated fodder crops. For Asia, emphasis was placed on draught power for land preparation, manure for soil fertility enhancement, and fodder production as an entry point for cultivation of legumes (Devendra and Thomas 2002). Mixed farming was seen as an evolutionary process to expand food production in the face of population increase, promote improvements in income and welfare, and protect the environment. The process could be further facilitated and steered by research, extension and policy (Pritchard et al. 1992; McIntire et al. 1992; Devendra 2002) (Pritchard et al., 1992; McIntire et al. 1992; Devendra 1992).

Scoones and Wolmer (2002) place this model in historical context, including concern about population pressure on resources and the view that mobile pastoralism was environmentally damaging. The latter view had already been critiqued by developing understandings of pastoralism, mobility and communal tenure of grazing lands (for example (Behnke 1994; Ellis 1994)). They set out a much more differentiated picture of crop livestock interactions, which can take place either within a single farm household, or between crop and livestock producers, in which case they will be mediated by formal and informal institutions governing the allocation of land, labour and capital, with the interactions evolving through multiple place-specific pathways (Ramisch et al. 2002; Scoones and Wolmer 2002). Promoting a diversity of approaches to crop-livestock interactions does not imply that the integrated model necessarily leads to land degradation, but increases the space for institutional support to local innovation (Scoones and Wolmer 2002).

However, specific managerial and technological practices that link crop and livestock production will remain an important part of the repertoire of on-farm adaptation and mitigation. Howden and coauthors (Howden et al. 2007) note the importance of innovation within existing integrated systems including use of adapted forage crops. Rivera-Ferre et al. (2016) list as adaptation strategies with high potential for grazing systems, mixed crop-livestock systems or both: crop-livestock integration in general; soil management including composting; enclosure and corralling of animal; improved storage of feed. Most of these are seen as having significant co-benefits for mitigation, and improved management of manure is seen as a mitigation measure with adaptation co-benefits.

4.8.2 Local and indigenous knowledge for addressing land degradation

In practice, responses are anchored both in scientific research, as well as local, indigenous and traditional knowledge and know-how. For example, studies in the Philippines Camacho et al. (2016) examine how traditional integrated watershed management by indigenous people sustain regulating services vital to agricultural productivity, while delivering co-benefits in the form of biodiversity and ecosystem resilience at a landscape scale. Although responses can be site specific and sustainable at a local scale, the multi-scale interplay of drivers and pressures can nevertheless cause practices that have been sustainable for centuries to become less so. Siahaya et al (2016) explore the traditional knowledge that has informed rice cultivation in the uplands of East Borneo, grounded in sophisticated shifting cultivation methods (gilir balik) which have been passed on for generations (more than 200 years) in order to maintain local food production. Gilir balik involves temporary cultivation of plots, after which, abandonment takes place as the land user moves to another plot, leaving the natural (forest) vegetation to return. This approach is considered sustainable if it has the support of other subsistence strategies, adapts to and integrates with the local context, and if the carrying capacity of the system is not surpassed (Siahaya et al. 2016). Often gilir balik cultivation involves intercropping of rice with bananas, cassava and other food crops. Once the abandoned plot has been left to recover such that soil fertility is restored, clearance takes place again and the plot is reused for cultivation. Rice cultivation in this way plays an important role in forest management, with several different types of succession forest being found in the study are of Siahaya et al (2016). Nevertheless, interplay of these practices with other pressures (large-scale land acquisitions for oil palm plantation, logging and mining), risk their future sustainability. Use of fire is critical in processes of land clearance, so there are also trade-offs for climate change mitigation which have been sparsely assessed.

Interest appears to be growing in understanding how indigenous and local knowledge inform land users' responses to degradation, as scientists engage farmers as experts in processes of knowledge co-production and co-innovation (Oliver et al. 2012; Bitzer and Bijman 2015). This can help to introduce, implement, adapt and promote the use of locally appropriate responses (Schwilch et al. 2011). Indeed, studies strongly agree on the importance of engaging local populations in both sustainable land and forest management. Meta-analyses in tropical regions that examined both forests in protected areas and community managed forests suggest that deforestation rates are lower, with less variation in deforestation rates presenting in community managed forests compared to protected forests (Porter-Bolland et al. 2012). This suggests that consideration of the social and economic needs of local human populations is vital in preventing forest degradation (Ward et al. 2018). However, while disciplines such as ethnopedology seek to record and understand how local people perceive, classify and use soil, and draw on that information to inform its management (Barrera-Bassols and Zinck 2003), links with climate change and its impacts (perceived and actual) are not generally considered.

4.8.3 Reducing deforestation and forest degradation and increasing afforestation

Improved stewardship of forests through reduction or avoidance of deforestation and forest degradation, and enhancement of forest carbon stocks can all contribute to land-based natural climate solutions (Angelsen et al. 2018; Sonwa et al. 2011b; Griscom et al. 2017). While estimates of annual emissions from tropical deforestation and forest degradation range widely from 0.5 to 3.5 Gt C yr^{-1} (Baccini et al. 2017; Houghton et al. 2012; Mitchard 2018, see also Chapter 2), they all indicate the large potential to reduce annual emissions from deforestation and forest degradation. Recent estimates of forest extent for Africa in 1900 may result in downward adjustments of historic deforestation and degradation emission estimates (Aleman et al. 2018). Emissions from forest degradation in non-Annex I countries have declined marginally from 1.1 $GtCO_2$ yr^{-1} in 2001-2010 to 1 $GtCO_2$ yr^{-1} in 2011-2015, but the relative emissions from degradation compared to deforestation have increased from a quarter to a third (Federici et al. 2015). Forest sector activities in developing countries were estimated to represent a technical mitigation potential in 2030 of 9 Gt CO_2 (Miles et al. 2015). This was partitioned into reduction of deforestation (3.5 Gt CO_2), reduction in degradation and forest management (1.7 Gt CO_2) and afforestation and reforestation (3.8 $GtCO_2$). The economic mitigation potential will be lower than the technical potential (Miles et al. 2015).

Natural regeneration of second-growth forests enhances carbon sinks in the global carbon budget (Chazdon and Uriarte 2016). In Latin America, Chazdon et al. (2016) estimated that in 2008, second-growth forests (1 to 60 years old) covered 2.4 M km^2 of land (28.1% of the total study area). Over 40 years, these lands can potentially accumulate 8.5 Gt C in aboveground biomass via low-cost natural regeneration or assisted regeneration, corresponding to a total CO_2 sequestration of 31.1 Gt CO_2 (Chazdon et al. 2016b). While aboveground biomass carbon stocks are estimated to be declining in the tropics, they are increasing globally due to increasing stocks in temperate and boreal forests (Liu et al. 2015b), consistent with the observations of a global land sector carbon sink (Le Quéré et al. 2013; Keenan et al. 2017; Pan et al. 2011).

Moving from technical mitigation potentials (Miles et al. 2015) to real reduction of emissions from deforestation and forest degradation required transformational changes (Korhonen-Kurki et al. 2018). This transformation can be facilitated by two enabling conditions: the presence of already initiated policy change; or the scarcity of forest resources combined with an absence of any effective forestry framework and policies. These authors and others (Angelsen et al. 2018) found that the presence of powerful transformational coalitions of domestic pro-REDD+ political actors combined with strong ownership and leadership, regulations and law enforcement, and performance-based funding, can provide a strong incentive for achieving REDD+ goals.

Implementing schemes such as REDD+ and various projects related to the voluntary carbon market is often regarded as a no-regrets investment (Seymour and Angelsen 2012) but the social and ecological implications (including those identified in the Cancun Safeguards) must be carefully considered for REDD+ projects to be socially and ecologically sustainable (Jagger et al. 2015). In 2018, 34 countries have submitted a REDD+ forest reference level and/or forest reference emission level to the UNFCCC. Of these REDD+ reference levels, 95% included the activity "reducing deforestation" while 34% included the activity "reducing forest degradation" (FAO 2018). Five countries submitted REDD+ results in the technical annex to their Biannual Update Report (BUR) totalling an emission reduction of 6.3 Gt CO_2 between 2006 and 2015 (FAO 2018).

Afforestation is another mitigation activity that increases carbon sequestration (see also Cross-Chapter Box 2: Implications of large-scale reforestation and afforestation, Chapter 1). Yet, it requires careful consideration about where to plant trees to achieve potential climatic benefits given an altering of local albedo and turbulent energy fluxes and increasing night-time land surface temperatures (Peng et al., 2014). A recent hydro-climatic modelling effort has shown that forest cover can account for about 40% of the observed decrease in annual runoff (Buendia et al. 2016). A meta-analysis of afforestation in Northern Europe (Bárcena and co-authors 2014) concluded that significant soil organic carbon sequestration in Northern Europe occurs after afforestation of croplands but not grasslands. Additional sequestration occurs in forest floors and biomass carbon stocks. Successful programmes of large scale afforestation activities in South Korea and China are discussed in-depth a special case study (Section 4.9.3).

The potential outcome of efforts to reduce emissions from deforestation and degradation in Indonesia through a 2011 moratorium on concessions to convert primary forests to either timber or palm oil uses was evaluated against rates of emissions over the period 2000 to 2010. The study concluded that less than 7% of emissions would have been avoided had the moratorium been implemented in 2000 because it only curtailed emissions due to a subset of drivers of deforestation and degradation (Busch et al. 2015).

In terms of ecological integrity of tropical forests, the policy focus on carbon storage and tree cover can be problematic if it leaves out other aspects of forests ecosystems, such as biodiversity – and particularly fauna (Panfil and Harvey 2016; Peres et al. 2016; Hinsley et al. 2015). Other concerns of forest based projects under the voluntary carbon market are potential negative socio-economic side effects (Edstedt and Carton 2018a; Carton and Andersson 2017; Osborne 2011; Scheidel and Work 2018; Richards and Lyons 2016; Borras and Franco 2018; Paladino and Fiske 2017) and leakage (particularly at the subnational scale), i.e. when interventions to reduce deforestation or degradation at one site displace pressures and increase emissions elsewhere (Atmadja and Verchot 2012; Phelps et al. 2010; Lund et al. 2017; Balooni and Lund 2014).

Maintaining and increasing forest area, in particular of native forests rather than monoculture and short-rotation plantations, contributes to the maintenance of global forest carbon stocks (Lewis et al. 2019) (*robust evidence, high agreement*).

4.8.4 Sustainable forest management and CO_2 removal technologies

While reducing deforestation and forest degradation may help directly meet mitigation goals, sustainable forest management aimed at providing timber, fiber, biomass and non-timber resources can provide long-term livelihood for communities, can reduce the risk of forest conversion to non-forest uses (settlement, crops, etc.), and can maintain land productivity, thus reducing the risks of land degradation (Putz et al. 2012; Gideon Neba et al. 2014; Sufo Kankeu et al. 2016; Nitcheu Tchiadje et al. 2016; Rossi et al. 2017).

Developing sustainable forest management strategies aimed at contributing towards negative emissions throughout this century requires an understanding of forest management impacts on ecosystem carbon stocks (including soils), carbon sinks, carbon fluxes in harvested wood, carbon storage in harvested wood products including landfills and the emission reductions achieved through the use of wood products and bioenergy (Nabuurs et al. 2007; Lemprière et al. 2013; Kurz et al. 2016; Law et al. 2018; Nabuurs et al. 2017). Transitions from natural to managed forest landscapes can involve a reduction in forest carbon stocks, the magnitude of which depends on the initial landscape conditions, the harvest rotation length relative to the frequency and intensity of natural disturbances and on the age-dependence of managed and natural disturbances (Harmon et al. 1990; Kurz et al. 1998a). Initial landscape conditions, in particular the age-class distribution and therefore C stocks of the landscape strongly affect the mitigation potential of forest management options (Ter-Mikaelian et al. 2013; Kilpeläinen et al. 2017). Landscapes with predominantly mature forests may experience larger reductions in carbon stocks during the transition to managed landscapes (Harmon et al. 1990;

Kurz et al. 1998b; Lewis et al. 2019) while in landscapes with predominantly young or recently disturbed forests sustainable forest management can enhance carbon stocks (Henttonen et al. 2017).

Forest growth rates, net primary productivity, and net ecosystem productivity are age-dependent with maximum rates of carbon removal from the atmosphere occurring in young to medium aged forests and declining thereafter (Tang et al. 2014). In boreal forest ecosystem, estimation of carbon stocks and carbon fluxes indicate that old growth stands are typically small carbon sinks or carbon sources (Gao et al. 2018; Taylor et al. 2014; Hadden and Grelle 2016). In tropical forests, carbon uptake rates in the first 20 years of forest recovery were 11 times higher than uptake rates in old-growth forests (Poorter et al. 2016). Age-dependent increases in forest carbon stocks and declines in forest carbon sinks mean that landscapes with older forests have accumulated more carbon but their sink strength is diminishing, while landscapes with younger forests contain less carbon but they are removing CO_2 from the atmosphere at a much higher rate (Volkova et al. 2017; Poorter et al. 2016). The rates of carbon removal are not just age-related but also controlled by many biophysical factors and human activities (Bernal et al. 2018) and in ecosystems with uneven-aged, multispecies forests the relationships between carbon stocks and sinks are more difficult and expensive to quantify.

Whether or not forest harvest and use of biomass is contributing to net reductions of atmospheric carbon depends on carbon losses during and following harvest, rates of forest regrowth, and the use of the harvested wood and the carbon retention in long-lived or short-lived products as well as the emission reductions achieved through the substitution of emissions-intensive products with wood products (Lemprière et al. 2013; Lundmark et al. 2014; Xu et al. 2018b; Olguin et al. 2018; Dugan et al. 2018; Chen et al. 2018b; Pingoud et al. 2018; Seidl et al. 2007). Studies that ignore changes in forest carbon stocks (such as some life cycle analyses that assume no impacts of harvest on forest carbon stocks), ignore changes in wood product pools (Mackey et al. 2013) or assume long-term steady state (Pingoud et al. 2018), or ignore changes in emissions from substitution benefits (Mackey et al. 2013; Lewis et al. 2019) will arrive at diverging conclusions about the benefits of sustainable forest management. Moreover, assessments of climate benefits of any mitigation action must also consider the time dynamics of atmospheric impacts as some actions will have immediate benefits (e.g. avoided deforestation) while others may not achieve net atmospheric benefits for decades or centuries. For example, the climate benefits of woody biomass use for bioenergy depend on several factors such as the source and alternate fate of the biomass, the energy type it substitutes and the rates of regrowth of the harvested forest (Laganière et al. 2017; Ter-Mikaelian et al. 2014; Smyth et al. 2017). Conversion of primary forests in regions of very low stand replacing disturbances to short-rotation plantations where the harvested wood is used for short-lived products with low displacement factors will increase emissions. In general, greater mitigation benefits are achieved if harvested wood products are used for products with long carbon retention time and high displacement factors.

With increasing forest age, carbon sinks in forests will diminish until harvest or natural disturbances such as wildfire remove biomass carbon or release it to the atmosphere (Seidl et al. 2017). While individual trees can accumulate carbon for centuries (Köhl et al. 2017), stand level carbon accumulation rates depend on both tree growth and tree mortality rates (Hember et al. 2016; Lewis et al. 2004). Sustainable forest management, including harvest and forest regeneration, can help maintain active carbon sinks by maintaining a forest age-class distribution that includes a share of young, actively growing stands (Volkova et al. 2018; Nabuurs et al. 2017). The use of the harvested carbon in either long-lived wood products (e.g. for construction), short-lived wood products (e.g., pulp and paper), or biofuels affects the net carbon balance of the forest sector (Lemprière et al. 2013; Matthews et al. 2018). The use of these wood products can further contribute to GHG emission reduction goals by avoiding the emissions from the products with higher embodied emissions that have been displaced (Nabuurs et al. 2007; Lemprière et al. 2013). In 2007 the IPCC concluded that "[i]n the long term, a sustainable forest management strategy aimed at maintaining or increasing forest carbon stocks, while producing an annual sustained yield of timber, fibre or energy from the forest, will generate the largest sustained mitigation benefit" (Nabuurs et al. 2007).–The apparent trade-offs between maximising forest C stocks and maximising ecosystem C sinks are at the origin of ongoing debates about optimum management strategies to achieve negative emissions (Keith et al. 2014; Kurz et al. 2016; Lundmark et al. 2014). Sustainable forest management, including the intensification of carbon-focussed management strategies, can make long-term contributions towards negative emissions if the sustainability of management is assured through appropriate governance, monitoring and enforcement. As specified in the definition of sustainable forest management, other criteria such as biodiversity must also be considered when assessing mitigation outcomes (Lecina-Diaz et al. 2018). Moreover, the impacts of changes in management on albedo and other non-GHG factors also need to be considered (Luyssaert et al. 2018) (See also Chapter 2). The contribution of sustainable forest management for negative emissions is strongly affected by the use of the wood products derived from forest harvest and the time horizon over which the carbon balance is assessed. Sustainable forest management needs to anticipate the impacts of climate change on future tree growth, mortality and disturbances when designing climate change mitigation and adaptation strategies (Valade et al. 2017; Seidl et al. 2017).

4.8.5 Policy responses to land degradation

The 1992 United Nations Conference on Environment and Development (UNCED), also known as the Rio de Janeiro Earth Summit, recognised land degradation as a major challenge to sustainable development, and led to the establishment of the United Nations Convention to Combat Desertification (UNCCD), which addressed specifically land degradation in the drylands. The UNCCD emphasizes sustainable land use to link poverty reduction on one hand and environmental protection on the other. The two other "Rio Conventions" emerging from the UNCED, the United Nations Framework Convention on Climate Change (UNFCCC) and the Convention on Biological Diversity (CBD), focus on climate change and biodiversity, respectively. The land has been recognized as an aspect of common interest to the three conventions, and sustainable land management (SLM) is proposed as a unifying theme for current global efforts on combating land degradation, climate change and loss of biodiversity, as well as facilitating land-based adaptation to climate change and sustainable development.

The Global Environmental Facility (GEF) funds developing countries to undertake activities that meet the goals of the conventions and deliver global environmental benefits. Since 2002, the GEF has invested in projects that support sustainable land management through its Land Degradation Focal Area Strategy, to address land degradation within and beyond the drylands.

Under the UNFCCC, parties have devised National Adaptation Plans (NAPs) that identify medium- and long-term adaptation needs. Parties have also developed their climate change mitigation plans, presented as Nationally Determined Contributions (NDCs). These programs have the potential of assisting the promotion of SLM. It is realised that the root causes of land degradation and successful adaptation will not be realised until holistic solutions to land management are explored. SLM can help address root causes of low productivity, land degradation, loss of income generating capacity as well as contribute to the amelioration of the adverse effects of climate change.

The "4 per 1000" (4p1000) initiative (Soussana et al. 2019) launched by France during the UNFCCC COP21 in 2015 aims at capturing CO_2 from the atmosphere through changes to agricultural and forestry practices at a rate that would increase the carbon content of soils by 0.4% per year (Rumpel et al. 2018). If global soil carbon content increases at this rate in the top 30-40 cm, the annual increase in atmospheric CO_2 would be stopped (Dignac et al. 2017). This is an illustration of how extremely important soils are for addressing climate change. The initiative is based on eight steps: stop carbon loss (priority #1 is peat soils); promote carbon uptake; monitor, report, and verify impacts; deploy technology for tracking soil carbon; test strategies for implementation and upscaling; involve communities; coordinate policies; provide support (Rumpel et al. 2018). Questions remain however, to what extent the 4p1000 is achievable as a universal goal (van Groenigen et al. 2017; Poulton et al. 2018; Schlesinger and Amundson 2018).

Land degradation neutrality (LDN) was introduced by the UNCCD at Rio +20, and adopted at UNCCD COP12 (UNCCD 2016a). LDN is defined as "a state whereby the amount and quality of land resources necessary to support ecosystem functions and services and enhance food security remain stable or increase within specified temporal and spatial scales and ecosystems". Pursuit of LDN requires effort to avoid further net loss of the land-based natural capital relative to a reference state, or baseline. LDN encourages a dual-pronged effort involving sustainable land management to reduce the risk of land degradation, combined with efforts in land restoration and rehabilitation, to maintain or enhance land-based natural capital, and its associated ecosystem services (Orr et al., 2017; Cowie et al. 2018;). Planning for LDN involves projecting the expected cumulative impacts of land use and land management decisions, then counterbalancing anticipated losses with measures to achieve equivalent gains, within individual land types (where land type is defined by land potential). Under LDN framework developed by UNCCD, three primary indicators are used to assess whether LDN is achieved by 2030: land cover change, net primary productivity and soil organic carbon (Cowie et al. 2018; Sims et al., 2019. Achieving LDN therefore requires integrated landscape management that seeks to optimize land use to meet multiple objectives (ecosystem health, food security, human well-being) (Cohen-Shacham, E., Walters, G., Janzen, C. and Maginnis 2016). The response hierarchy of Avoid > Reduce > Reverse land degradation articulates the priorities in planning LDN interventions. LDN provides the impetus for widespread adoption of SLM and efforts to restore or rehabilitate land. Through its focus LDN ultimately provides tremendous potential for mitigation of and adaptation to climate change by halting and reversing land degradation and transforming land from a carbon source to a sink. There are strong synergies between the concept of LDN and the Nationally Determined Contributions (NDCs) of many countries with linkages to national climate plans. LDN is also closely related to many Sustainable Development Goals (SDG) in the areas of poverty, food security, environmental protection and sustainable use of natural resources (UNCCD 2016b). The GEF is supporting countries to set LDN targets and implement their LDN plans through its land degradation focal area, which encourages application of integrated landscape approach to managing land degradation (GEF 2018).

The 2030 agenda for sustainable development, adopted by the United Nations in 2015, comprises 17 Sustainable Development Goals (SDGs). Goal 15 is of direct relevance to land degradation with the objective to protect, restore and promote sustainable use of terrestrial ecosystems, sustainably manage forests, combat desertification and halt and reverse land degradation and halt biodiversity loss. Target 15.3 specifically addresses land degradation neutrality. Other goals that are relevant for land degradation include goal 2 (Zero hunger), goal 3 (Good health and well-being), goal 7 (Affordable and clean energy), goal 11 (Sustainable cities and communities), and goal 12 (Responsible production and consumption). Sustainable management of land resources underpins the SDGs related to hunger, climate change and environment. Further goals of a cross-cutting nature include 1 (No poverty), 6 (Clean water and sanitation) and 13 (Climate action). It remains to be seen how these interconnections are dealt with in practice.

With a focus on biodiversity, IPBES published a comprehensive assessment of land degradation in 2018 (Montanarella et al. 2018). The IPBES report, together with this report focusing on climate change, may contribute to create synergy between the two main global challenges for addressing land degradation in order to help achieving the goals of SDG 15 (Protect, restore and promote sustainable use of terrestrial ecosystems, sustainably manage forests, combat desertification, and halt and reverse land degradation and halt biodiversity loss).

Market based mechanisms like the Clean Development Mechanism (CDM) under the UNFCCC and the voluntary carbon market provide incentives to enhance carbon sinks on the land through afforestation and reforestation. Implications for local land use and food security have been raised as a concern and need to be assessed (Edstedt and Carton 2018b; Olsson et al. 2014b). Many projects aimed at reducing emissions from deforestation and forest degradations (not to be confused with the national REDD+ programs in accordance with the UNFCCC Warsaw Framework) are being planned and implemented primarily targeting countries with high forest cover and high deforestation rates. Some parameters of incentivising emissions reduction, quality of forest governance, conservation priorities, local rights and tenure frameworks, and sub-national project potential are being looked into with often very mixed results (Newton et al. 2016; Gebara and Agrawal 2017).

Besides international public initiatives, some actors in the private sector are increasingly aware of the negative environmental impacts of some global value chains producing food, fibre, and energy products (Lambin et al. 2018; van der Ven and Cashore 2018; van der Ven et al. 2018; Lyons-White and Knight 2018). While improvement is under way in many supply chains, measures implemented so far are often insufficient to be effective in reducing or stopping deforestation and forest degradation (Lambin et al. 2018). The GEF is investing in actions to reduce deforestation in commodity supply chains through its Food Systems, Land Use, and Restoration Impact Program (GEF 2018).

4.8.5.1 Limits to adaptation

SLM can be deployed as a powerful adaptation strategy in most instances of climate change impacts on natural and social systems, yet there are limits to adaptation (Klein, R.J.T., G.F. Midgley, B.L. Preston, M. Alam, F.G.H. Berkhout, K. Dow 2014; Dow et al. 2013a). Such limits are dynamic and interact with social and institutional conditions (Barnett et al. 2015; Filho and Nalau 2018). Exceeding adaptation limits will trigger escalating losses or require undesirable transformational change, such as forced migration. The rate of change in relation to the rate of possible adaptation is crucial (Dow et al. 2013b). How limits to adaptation are defined and how they can be measured is contextual and contested. Limits must be assessed in relation to the ultimate goals of adaptation, which is subject to diverse and differential values (Dow et al. 2013b; Adger et al. 2009). A particularly sensitive issue is whether migration is accepted as adaptation or not (Black et al. 2011; Tacoli 2009; Bardsley and Hugo 2010). If migration were understood and accepted as a form of successful adaptation, it would change the limits to adaptation by reducing or even avoiding future humanitarian crises caused by climate extremes (Adger et al. 2009; Upadhyay et al. 2017; Nalau et al. 2018).

In the context of land degradation potential limits to adaptation exist if land degradation becomes so severe and irreversible that livelihoods cannot be maintained, and if migration is either not acceptable or possible. Examples are coastal erosion where land disappears (Gharbaoui and Blocher 2016; Luetz 2018), collapsing livelihoods due to thawing of permafrost (Landauer and Juhola 2019), and extreme forms of soil erosion (e.g., landslides (Van der Geest and Schindler 2016) and gully erosion leading to badlands (Poesen et al. 2003)).

4.8.6 Resilience and thresholds

Resilience refers to the capacity of interconnected social, economic and ecological systems, such as farming systems, to absorb disturbance (e.g., drought, conflict, market collapse), and respond or reorganise, to maintain their essential function, identity and structure. Resilience can be described as "coping capacity". The disturbance may be a shock - sudden events such as a flood or disease epidemic – or it may be a trend that develops slowly, like a drought or market shift. The shocks and trends anticipated to occur due to climate change are expected to exacerbate risk of land degradation. Therefore, assessing and enhancing resilience to climate change is a critical component of designing sustainable land management strategies.

Resilience as an analytical lens is particularly strong in ecology and related research on natural resource management (Folke et al. 2010; Quinlan et al. 2016) while in the social sciences the relevance of resilience for studying social and ecological interactions is contested (Cote and Nightingale 2012; Olsson et al. 2015; Cretney 2014; Béné et al. 2012; Joseph 2013). In the case of adaptation to climate change (and particularly regarding limits to adaptation), a crucial ambiguity of resilience is the question whether resilience is a normative concept (i.e. resilience is good or bad) or is a descriptive characteristic of a system (i.e. neither good nor bad). Previous IPCC reports have defined resilience as a normative (positive) attribute (see AR5 Glossary), while the wider scientific literature is divided on this (Weichselgartner and Kelman 2015; Strunz 2012; Brown 2014; Grimm and Calabrese 2011; Thorén and Olsson 2018). For example, is outmigration from a disaster prone area considered a successful adaptation (high resilience) or a collapse of the livelihood system (lack of resilience) (Thorén and Olsson 2018)? In this report resilience is considered a positive attribute when it maintains capacity for adaptation, learning and/or transformation.

Furthermore, resilience and the related terms adaptation and transformation are defined and used differently by different communities (Quinlan et al. 2016). The relationship and hierarchy of resilience with respect to vulnerability and adaptive capacity are also debated, with different perspectives between the disaster management, and global change communities, (e.g., Cutter et al. 2008). Nevertheless, these differences in usage need not inhibit the application of “resilience thinking” in managing land degradation; researchers using these terms, despite variation in definitions, apply the same fundamental concepts to inform management of human-environment systems, to maintain or improve the resource base, and sustain livelihoods.

Applying resilience concepts involves viewing the land as a component of an interlinked social-ecological system; identifying key relationships that determine system function and vulnerabilities of the system; identifying thresholds or tipping points beyond which the system transitions to an undesirable state; and devising management strategies to steer away from thresholds of potential concern, thus facilitating healthy systems and sustainable production (Walker et al., 2009).

A threshold is a non-linearity between a controlling variable and system function, such that a small change in the variable causes the system to shift to an alternative state. Bestelmeyer et al. (2015) and Prince et al. (2018) illustrate this concept in the context of land degradation. Studies have identified various biophysical and socio-economic thresholds in different land-use systems. For example, 50% ground cover (living and dead plant material and biological crusts) is a recognised threshold for dryland grazing systems (e.g., (Tighe et al. 2012); below this threshold infiltration rate declines, risk of erosion causing loss of topsoil increases, a switch from perennial to annual grass species occurs and there is a consequential sharp decline in productivity. This shift to a lower-productivity state cannot be reversed without significant human intervention. Similarly, the combined pressure of water limitations and frequent fire can lead to transition from closed forest to savannah or grassland: if fire is too frequent trees do not reach reproductive maturity and post-fire regeneration will fail; likewise, reduced rainfall / increased drought prevents successful forest regeneration (Reyer et al. 2015; Thompson et al. 2009) see also Cross-chapter box 3 on Fire and climate change, Chapter 2.

In managing land degradation, it is important to assess the resilience of the existing system, and the proposed management interventions. If the existing system is in an undesirable state or considered unviable under expected climate trends, it may be desirable to promote adaptation or even transformation to a different system that is more resilient to future changes. For example, in an irrigation district where water shortages are predicted, measures could be implemented to improve water use efficiency, for example by establishing drip irrigation systems for water delivery, although transformation to pastoralism or mixed dryland cropping/livestock production may be more sustainable in the longer term, at least for part of the area. Application of sustainable land management practices, especially those focussed on ecological functions (e.g., agroecology, ecosystem-based approaches, regenerative agriculture, organic farming), can be effective in building resilience of agro-ecosystems (Henry et al. 2018). Similarly, the resilience of managed forests can be enhanced by sustainable forest management that protects or enhances biodiversity, including assisted migration of tree species within their current range limit (Winder et al. 2011; Pedlar et al. 2012) or increasing species diversity in plantation forests (Felton et al. 2010; Liu et al. 2018a). The essential features of a resilience approach to management of land degradation under climate change are described by (O'Connell et al. 2016; Simonsen et al. 2014).

Consideration of resilience can enhance effectiveness of interventions to reduce or reverse land degradation (*medium agreement, limited evidence*). This approach will increase the likelihood that SLM/SFM and land restoration/rehabilitation interventions achieve long-term environmental and social benefits. Thus, consideration of resilience concepts can enhance the capacity of land systems to cope with climate change and resist land degradation, and assist land use systems to adapt to climate change.

4.8.7 Barriers to implementation of sustainable land management

There is a growing recognition that addressing barriers and designing solutions to complex environmental problems, such as land degradation, requires awareness of the larger system into which the problems and solutions are embedded (Laniak et al. 2013). An ecosystem approach to SLM based on understanding of the processes of land degradation has been recommended that can separate multiple drivers, pressures and impacts (Kassam et al. 2013), but large uncertainty in model projections of future climate, and associated ecosystem processes (IPCC 2013a) pose additional challenges to the implementation of SLM. As discussed earlier in this chapter, many SLM practices, including both technologies and approaches, are available that can increase yields and contribute to closing the yield gap between actual and potential crop or pasture yield, while also enhancing resilience to climate change (Yengoh and Ardö 2014; WOCAT). However, there are often systemic barriers to adoption and scaling up of SLM practices, especially in developing countries.

Uitto (2016) identified areas that the GEF, the financial mechanism of the UNCCD, UNFCCC and other multilateral environmental agreements, can address to solve global environmental problems. This includes removal of barriers related to knowledge and information; strategies for implementation of technologies and approaches; and institutional capacity. Strengthening these areas would drive transformational change leading to behavioral change and broader adoption of sustainable environmental practices. Detailed analysis of barriers as well as strategies, methods and approaches to scale up SLM have been undertaken for GEF programs in Africa, China and globally (Tengberg and Valencia 2018; Liniger et al. 2011; Tengberg et al. 2016). A number of interconnected barriers and bottlenecks to the scaling up of SLM have been identified in this context and are related to:

- Limited access to knowledge and information, including new SLM technologies and problem-solving capacities;
- Weak enabling environment, including the policy, institutional and legal framework for SLM, and land tenure and property rights;
- Inadequate learning and adaptive knowledge management in the project cycle, including monitoring and evaluation of impacts; and
- Limited access to finance for scaling up, including public and private funding, innovative business models for SLM technologies and financial mechanisms and incentives, such as payments for ecosystem services (PES), insurance and micro-credit schemes (see also Shames et al 2014).

Adoption of innovations and new technologies are increasingly analysed using the transition theory framework (Geels 2002), the starting point being the recognition that many global environmental problems cannot be solved by technological change alone but require more far-reaching change of social-ecological systems. Using transition theory makes it possible to analyse how adoption and implementation follow the four stages of sociotechnical transitions, from predevelopment of technologies and approaches at the niche level, take-off and acceleration, to regime shift and stabilisation at the landscape level. According to a recent review of sustainability transitions in developing countries (Wieczorek 2018), three internal niche processes are important, including the formation of networks that support and nurture innovation, the learning process and the articulation of expectations to guide the learning process. While technologies are important, institutional and political aspects form the major barriers to transition and upscaling. In developing and transition economies, informal institutions play a pivotal role and transnational linkages are also important, such as global value chains. In these countries, it is therefore more difficult to establish fully coherent regimes or groups of individuals who share expectations, beliefs or behavior, as there is a high level of uncertainty about rules and social networks or dominance of informal institutions, which creates barriers to change. This uncertainty is further exacerbated by climate change. Landscape forces comprise a set of slow changing factors, such as broad cultural and normative values, long-term economic effects such as urbanisation, and shocks such as war and crises that can lead to change.

A study on SLM in the Kenyan highlands using transition theory concluded that barriers to adoption of SLM included high poverty levels, a low input-low output farming system with limited potential to generate income, diminishing land sizes and low involvement of the youth in farming activities. Coupled with a poor coordination of government policies for agriculture and forestry, these barriers created negative feedbacks in the SLM transition process. Other factors to consider include gender issues and lack of secure land tenure. Scaling up of SLM technologies would require collaboration of diverse stakeholders across multiple scales, a more supportive policy environment and substantial resource mobilisation (Mutoko et al. 2014). Tengberg and Valencia (2018) analysed the findings from a review of the GEF integrated natural resources management portfolio of projects using the transition theory framework (Figure 4.7). They concluded that to remove barriers to SLM, an agricultural innovations systems approach that supports co-production of knowledge with multiple stakeholders, institutional innovations, a focus on value chains and strengthening of social capital to facilitate shared learning and collaboration could accelerate the scaling up of sustainable technologies and practices from the niche to the landscape level. Policy integration and establishment of financial mechanisms and incentives could contribute to overcoming barriers to a regime shift. The new SLM regime could in turn be stabilised and sustained at the landscape level by multi-stakeholder knowledge platforms and strategic partnerships. However, transitions to more sustainable regimes and practices are often challenged by lock-in mechanisms in the current system (Lawhon and Murphy 2012), such as economies of scale, investments already made in equipment, infrastructure and competencies, lobbying, shared beliefs, and practices, which could hamper wider adoption of SLM.

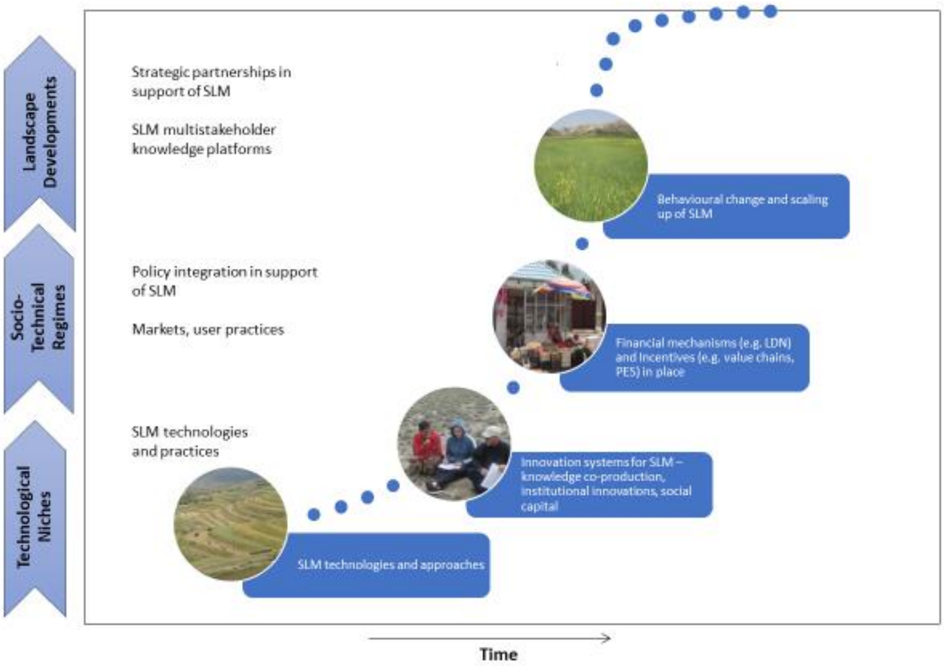


Figure 4.7 The transition from SLM niche adoption to regime shift and landscape development (figure draws inspiration from (Geels 2002)). Adapted from (Tengberg and Valencia 2018)

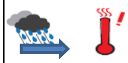
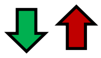
Adaptive, multi-level and participatory governance of social-ecological systems is considered important for regime shifts and transitions to take place (Wieczorek 2018) and essential to secure the capacity of environmental assets to support societal development over longer time periods (Folke et al. 2005). There is also recognition that effective environmental policies and programs need to be informed by a comprehensive understanding of the biophysical, social, and economic components and processes of a system, their complex interactions, and how they respond to different changes (Kelly (Letcher) et al. 2013). But blueprint policies will not work due to the wide diversity of rules and informal institutions used across sectors and regions of the world, especially in traditional societies (Ostrom 2009).

The most effective way of removing barriers to funding of SLM has been mainstreaming of SLM objectives and priorities into relevant policy and development frameworks and combining SLM best practices with economic incentives for land users. As the short-term costs for establishing and maintaining SLM measures are generally high and constitute a barrier to adoption, land users may need to be compensated for generation of longer-term public goods, such as ecosystem services. Cost-benefit analyses can be conducted on SLM interventions to facilitate such compensations (Liniger et al. 2011; Nkonya et al. 2016; Tengberg et al. 2016). The landscape approach is a means to reconcile competing demands on the land and remove barriers to implementation of SLM (e.g. Sayer et al. 2013; Bürgi et al. 2017). It involves an increased focus on participatory governance, development of new SLM business models, and innovative funding schemes including insurance (Shames et al. 2014). The Land Degradation Neutrality (LDN) Fund takes a landscape approach and raises private finance for SLM and promotes market-based instruments, such as Payment for Ecosystem Services (PES), certification and carbon trading, that can support scaling up of SLM to improve local livelihoods, sequester carbon and enhance the resilience to climate change.

4.9 Case-studies

Climate change impacts on land degradation can be avoided, reduced or even reversed, but need to be addressed in a context sensitive manner. Many of the responses described in this section can also provide synergies of adaptation and mitigation. In this section we provide more in-depth analysis of a number of salient aspects of how land degradation and climate change interact. Table 4.3 is a synthesis of how of these case studies relate to climate change and other broader issues in terms of co-benefits.

Table 4.3 Synthesis of how the case studies interact with climate change and a broader set of co-benefits

Case studies (4.10) (4.9)	Mitigation benefits and potential	Adaptation benefits	Co-benefits		Legend
Urban green infrastructure (4.10.1) (4.9.1) An increasing majority of the world population live in cities and land degradation is an urgent matter for urban areas			human health, recreation		carbon sink
Perennial grains (4.10.2) (4.9.2) After 40 years of breeding, perennial grains now seem to have the potential of reducing climate impacts of agriculture while increasing its overall sustainability			reduced use of herbicides, reduced soil erosion and nutrient leakage		reduced emission
Reforestation (4.10.3) (4.9.3) Two cases of successful reforestation serve as illustrations of the potential of sustained efforts into reforestation			economic return from sustainable forestry, reduced flood risk downstream		
Managment of peat soils (4.10.4) (4.9.4) Degradation of peat soils in tropical and arctic regions is a major source of greenhouse gases, hence an urgent mitigation option			improved air quality in tropical regions		reduced flood risk
Biochar (4.10.5) (4.9.5) Biochar is a land managent technique of high potential, but controversial			improved soil fertility		reduced heat stress
Protection against hurricane damages (4.10.6) (4.9.6) More severe tropical cyclones increase the risk of land degradation in some areas, hence the need for increased adaptation			reduced losses (human lives, livelihoods, and assets)		drought resistance
Responses to salt water intrusion (4.10.7) (4.9.7) The combined effect of climate induced sealevel rise and land use change in coastal regions increases the risk of saltwater intrusion in many coastal regions			improved food and water security,		storm protection
Avoiding coastal maladaptation (4.10.8) (4.9.8) Low lying coastal areas are in urgent need of adaptation, but examples have resulted in maladaptation			reduced losses (human lives, livelihoods, and assets)		protection against sea level rise

4.9.1 Urban green infrastructure

Over half the world's population now lives in towns and cities, a proportion that is predicted to increase to ~70% by the middle of the century (United Nations 2015). Rapid urbanisation is a severe threat to land and the provision of ecosystem services (Seto et al. 2012). However, as cities expand, the avoidance of land degradation, or the maintenance/enhancement of ecosystem services is rarely considered in planning processes. Instead economic development and the need for space for construction is prioritised, which can result in substantial pollution of air and water sources, the degradation of existing agricultural areas and indigenous, natural or semi-natural ecosystems both within and outside of urban areas. For instance, urban areas are characterised by extensive impervious surfaces. Degraded, sealed soils beneath these surfaces do not provide the same quality of water retention as intact soils. Urban landscapes comprising 50-90% impervious surfaces can therefore result in 40-83% of rainfall becoming surface water runoff (Pataki et al. 2011). With rainfall intensity predicted to increase in many parts of the world under climate change (Royal Society 2016), increased water runoff is going to get worse. Urbanisation, land degradation and climate change are therefore strongly interlinked, suggesting the need for common solutions (Reed and Stringer 2016b).

There is now a large body of research and application demonstrating the importance of retaining urban green infrastructure (UGI) for the delivery of multiple ecosystem services (DG Environment News Alert Service, 2012; Wentworth, 2017) as an important tool to mitigate and adapt to climate change. UGI can be defined as all green elements within a city, including but not limited to retained indigenous ecosystems, parks, public greenspaces, green corridors, street trees, urban forests, urban agriculture, green roofs/walls and private domestic gardens (Tzoulas et al. 2007). The definition is usually extended to include 'blue' infrastructure, such as rivers, lakes, bioswales and other water drainage features. The related concept of Nature Based Solutions (defined as: living solutions inspired by, continuously supported by and using nature, which are designed to address various societal challenges in a resource-efficient and adaptable manner and to provide simultaneously economic, social, and environmental benefits) has gained considerable traction within the European Commission as one approach to mainstreaming the importance of UGI (Maes and Jacobs 2017; European Union 2015).

Through retaining existing vegetation and ecosystems, revegetating previous developed land or integrating vegetation into buildings in the form of green walls and roofs, UGI can play a direct role in mitigating climate change through carbon sequestration. However, compared to overall carbon emissions from cities, effects will be small. Given that UGI necessarily involves the retention and management of non-sealed surfaces, co-benefits for land degradation (e.g. soil compaction avoidance, reduced water run-off, carbon storage and vegetation productivity; (Davies et al. 2011; Edmondson et al. 2011, 2014; Yao et al. 2015) will also be apparent. Although not currently a priority, its role in mitigating land degradation could be substantial. For instance, appropriately managed innovative urban agricultural production systems, such as vertical farms, could have the potential to both meet some of the food needs of cities and reduce the production (and therefore degradation) pressure on agricultural land in rural areas, although thus far this is unproven (for a recent review (Wilhelm and Smith 2018)).

The importance of UGI as part of a climate change adaptation approach has received greater attention and application (Gill et al. 2007; Fryd et al. 2011; Demuzere et al. 2014; Sussams et al. 2015). The EU's Adapting to Climate Change White Paper emphasises the "crucial role in adaptation in providing essential resources for social and economic purposes under extreme climate conditions" (CEC, 2009, p. 9). Increasing vegetation cover, planting street trees and maintaining/expanding public parks reduces temperatures (Cavan et al. 2014; Di Leo et al. 2016; Feyisa et al. 2014; Tonosaki K, Kawai S 2014; Zölch et al. 2016). Further, the appropriate design and spatial distribution of greenspaces within cities can help to alter urban climates to improve human health and comfort (e.g. (Brown and Nicholls 2015; Klemm et al. 2015)). The use of green walls and roofs can also reduce energy use in buildings (e.g. (Coma et al. 2017)). Similarly, natural flood management and ecosystem based approaches of providing space for water, renaturalising rivers and reducing surface run-off through the presence of permeable surfaces and vegetated features (including walls and roofs) can manage flood risks, impacts and vulnerability (e.g. (Gill et al. 2007; Munang et al. 2013)). Access to UGI in times of environmental stresses and shock can provide safety nets for people and can, therefore, be an important adaptation mechanism, both to climate change (Potschin et al. 2016) and land degradation.

Most examples of UGI implementation as a climate change adaptation strategy have centered on its role in water management for flood risk reduction. The importance for land degradation is either not stated, or not prioritized. In Beira, Mozambique, the government is using UGI to mitigate against increased flood risks predicted to occur under climate change and urbanisation, which will be done by improving the natural water capacity of the Chiveve River. As part of the UGI approach, mangrove habitats have been restored and future phases include developing new multi-functional urban green spaces along the river (World Bank 2016). The retention of green spaces within the city will have the added benefit of halting further degradation in those areas. Elsewhere, planning mechanisms promote the retention and expansion of green areas within cities to ensure ecosystem service delivery, which directly halts land degradation, but are largely viewed and justified in the context of climate change adaptation and mitigation. For instance, the Landscape Programme in Berlin includes five plans, one of which covers adapting to climate change through the recognition of the role of UGI (Green Surge 2016). Major climate related challenges facing Durban, South Africa, include sea level rise, urban heat island, water runoff and conservation (Roberts and O'Donoghue 2013). Now considered a global leader in climate adaptation planning (Roberts 2010), Durban's Climate Change Adaptation plan includes the retention and maintenance of natural ecosystems in particular those which are important for mitigating flooding, coastal erosion, water pollution, wetland siltation and climate change (eThekwini Municipal Council 2014).

4.9.2 Perennial Grains and Soil Organic Carbon

The severe ecological perturbation that is inherent in the conversion of native perennial vegetation to annual crops, and the subsequent high frequency of perturbation required to maintain annual crops, results in at least four forms of soil degradation that will be exacerbated by the effects of climate change (Crews et al. 2016). First, soil erosion is a very serious consequence of annual cropping with median losses exceeding rates of formation by 1-2 orders of magnitude in conventionally plowed agroecosystems, and while erosion is reduced with conservation tillage, median losses still exceed formation by several fold (Montgomery 2007). More severe storm intensity associated with climate change is expected to cause even greater losses to wind and water erosion (Nearing et al. 2004b). Secondly, the periods of time in which live roots are reduced or altogether absent from soils in annual cropping systems allow for substantial losses of nitrogen from fertilised croplands, averaging 50% globally (Ladha et al. 2005). This low retention of nitrogen is also expected to worsen with more intense weather events (Bowles et al. 2018). A third impact of annual cropping is the degradation of soil structure caused by tillage, which can reduce infiltration of precipitation, and increase surface runoff. It is predicted that the percentage of precipitation that infiltrates into agricultural soils will decrease further under climate change scenarios (Basche and DeLonge 2017; Wuest et al. 2006). The fourth form of soil degradation that results from annual cropping is the reduction of soil organic matter (SOM), a topic of particular relevance to climate change mitigation and adaptation.

Undegraded cropland soils can theoretically hold far more SOM (which is ~58% carbon) than they currently do (Soussana et al. 2006). We know this deficiency because, with few exceptions, comparisons between cropland soils and those of proximate mature native ecosystems commonly show a 40-75% decline in soil carbon attributable to agricultural practices. What happens when native ecosystems are converted to agriculture that induces such significant losses of SOM? Wind and water erosion commonly results in preferential removal of light organic matter fractions that can accumulate on or near the soil surface (Lal 2003). In addition to the effects of erosion, the fundamental practices of growing annual food and fiber crops alters both inputs and outputs of organic matter from most agroecosystems resulting in net reductions in soil carbon equilibria (Soussana et al. 2006; McLauchlan 2006; Crews et al. 2016). Native vegetation of almost all terrestrial ecosystems is dominated by perennial plants, and the belowground carbon allocation of these perennials is a key variable in determining formation rates of stable soil organic carbon (SOC) (Jastrow et al. 2007; Schmidt et al. 2011). When perennial vegetation is replaced by annual crops, inputs of root-associated carbon (roots, exudates, mycorrhizae) decline substantially. For example, perennial grassland species allocate around 67% of productivity to roots, whereas annual crops allocate between 13-30% (Saugier 2001; Johnson et al. 2006).

At the same time inputs of SOC are reduced in annual cropping systems, losses are increased because of tillage, compared to native perennial vegetation. Tillage breaks apart soil aggregates, which, among other functions, are thought to inhibit soil bacteria, fungi and other microbes from consuming and decomposing soil organic matter (Grandy and Neff 2008). Aggregates reduce microbial access to organic matter by restricting physical access to mineral-stabilized organic compounds as well as reducing oxygen availability (Cotrufo et al. 2015; Lehmann and Kleber 2015). When soil aggregates are broken open with tillage in the conversion of native ecosystems to agriculture, microbial consumption of SOC and subsequent respiration of CO_2 increase dramatically, reducing soil carbon stocks (Grandy and Robertson 2006; Grandy and Neff 2008).

Many management approaches are being evaluated to reduce soil degradation in general, especially by increasing mineral-protected forms of SOC in the world's croplands (Paustian et al. 2016). The menu of approaches being investigated focus either on increasing belowground carbon inputs, usually through increases in total crop productivity, or by decreasing microbial activity, usually through reduced soil disturbance (Crews and Rumsey 2017). However, the basic biogeochemistry of terrestrial ecosystems managed for production of annual crops presents serious challenges to achieving the standing stocks of SOC accumulated by native ecosystems that preceded agriculture. A novel new approach that is just starting to receive significant attention is the development of perennial cereal, legume and oilseed crops (Glover et al. 2010; Baker 2017).

There are two basic strategies that plant breeders and geneticists are using to develop new perennial grain crop species. The first involves making wide hybrid crosses between existing elite lines of annual crops, such as wheat, sorghum and rice, with related wild perennial species in order to introgress perennialism into the genome of the annual (Cox et al. 2018; Huang et al. 2018; Hayes et al. 2018). The other approach is *de novo* domestication of wild perennial species that have crop-like traits of interest (DeHaan et al. 2016; DeHaan and Van Tassel 2014). New perennial crop species undergoing *de novo* domestication include intermediate wheatgrass, a relative of wheat that produces grain also known as Kernza (DeHaan et al. 2018; Cattani and Asselin 2018) and *Silphium integrifolium*, an oilseed crop in the sunflower family (Van Tassel et al. 2017). Other perennial grain crops receiving attention include pigeon pea, barley, buckwheat and maize (Batello et al. 2014; Chen et al. 2018c) and a number of legume species (Schlautman et al. 2018). In most cases, the seed yields of perennial grain crops under development are well below those of elite modern grain varieties. In the time that it takes intensive breeding efforts to close the yield and other trait gaps between annual and perennial grains, perennial proto-crops may be used for purposes other than grain, including forage production (Ryan et al. 2018). Perennial rice stands out as a high-yielding exception, as its yields matched those of elite local varieties in the Yunnan Province for six growing seasons over three years (Huang et al. 2018).

In a perennial agroecosystem, the biogeochemical controls on SOC accumulation shift dramatically, and begin to resemble the controls that govern native ecosystems (Crews et al. 2016). When erosion is reduced or halted, and crop allocation to roots increases by 100-200%, and when soil aggregates are not disturbed thus reducing microbial respiration, SOC levels are expected to increase (Crews and Rumsey 2017). Deep roots growing year-round are also effective at increasing nitrogen retention (Culman et al. 2013; Jungers et al. 2019). Substantial increases in SOC have been measured where croplands that had historically been planted to annual grains were converted to perennial grasses, such as in the Conservation Reserve Program (CRP) of the US, or in plantings of second generation perennial biofuel crops. Two studies have assessed carbon accumulation in soils when croplands were converted to the perennial grain Kernza. In one, researchers found no differences in soil labile (permanganate-oxidizable) C after 4 years of cropping to perennial Kernza versus annual wheat in a sandy textured soil. Given that coarse textured soils do not offer the same physicochemical protection against microbial attack as many finer textured soils, these results are not surprising, but these results do underscore how variable rates of carbon accumulation can be (Jastrow et al. 2007). In the second study, researchers assessed the carbon balance of a Kernza field in Kansas USA over 4.5 years using eddy covariance observations (de Oliveira et al. 2018). They found the net C accumulation rate of about 1500 g C m^{-2} yr^{-1} in the first year of the study corresponding to the biomass of Kernza increasing, to about 300 g C m^{-2} yr^{-1}in the final year where CO_2 respiration losses from the decomposition of roots and soil organic matter approached new carbon inputs from photosynthesis. Based on measurements of soil carbon accumulation in restored grasslands in this part of US, the net carbon accumulation in stable organic matter under a perennial grain crop might be expected to sequester 30-50 g C m^{-2} yr^{-1}(Post and Kwon 2000) until a new equilibrium is reached. Sugar cane, a highly productive perennial, has been shown to accumulate a mean of 187 g C m^{-2} yr^{-1}in Brazil (La Scala Júnior et al. 2012).

Reduced soil erosion, increased nitrogen retention, greater water uptake efficiency and enhanced carbon sequestration represent improved ecosystem functions made possible in part by deep and extensive root systems of perennial crops (Figure 4.8).

Figure 4.8 Comparison of root systems between the newly domesticated intermediate wheatgrass (left) and annual wheat (right). Photo and copyright: Jim Richardson

When compared to annual grains like wheat, single species stands of deep rooted perennial grains such as Kernza are expected to reduce soil erosion, increase nitrogen retention, achieve greater water uptake efficiency and enhance carbon sequestration (Crews et al. 2018) (Figure 4.8). An even higher degree of ecosystem services can at least theoretically be achieved by strategically combining different functional groups of crops such as a cereal and a nitrogen-fixing legume (Soussana and Lemaire 2014). Not only is there evidence from plant diversity experiments that communities with higher species richness sustain higher concentrations of soil organic carbon (Hungate et al. 2017; Sprunger and Robertson 2018; Chen et al. 2018b; Yang et al. 2019), but other valuable ecosystem services such as pest suppression, lower greenhouse gas emissions, and greater nutrient retention may be enhanced (Schnitzer et al. 2011; Culman et al. 2013).

Similar to perennial forage crops such as alfalfa, perennial grain crops are expected to have a definite productive life span, probably in the range of 3-10 years. A key area of research on perennial grains cropping systems is to minimise losses of soil organic carbon during conversion of one stand of perennial grains to another. Recent work demonstrates that no-till conversion of a mature perennial grassland to another perennial crop will experience several years of high net CO_2 emissions as decomposition of copious crop residues exceeds ecosystem uptake of carbon by the new crop (Abraha et al. 2018). Most if not all of this lost carbon will be recaptured in the replacement crop. It is not known whether mineral-stabilised carbon that is protected in soil aggregates is vulnerable to loss in perennial crop succession.

Perennial grains hold promises of agricultural practices which can significantly reduce soil erosion and nutrient leakage while sequestering carbon. When cultivated in mixes with N-fixing species (legumes) such polycultures also reduce the need for external inputs of nitrogen - a large source of GHG from conventional agriculture.

4.9.3 Reversing land degradation through reforestation

4.9.3.1 South Korea Case Study on Reforestation Success

In the first half of the 20th century, forests in the Republic of South Korea were severely degraded and deforested during foreign occupations and the Korean War. Unsustainable harvest for timber and fuel wood resulted in severely degraded landscapes, heavy soil erosion and large areas denuded of vegetation cover. Recognising that South Korea's economic health would depend on a healthy environment, South Korea established a national forest service (1967) and embarked on the first phase of a 10-year reforestation program in 1973 (Forest Development Program), which was followed by subsequent reforestation programs that ended in 1987, after 2.4 Mha of forests were restored, see Figure 4.9.

As a consequence of reforestation, forest volume increased from 11.3 m^3 ha^{-1} in 1973 to 125.6 m^3 ha^{-1} in 2010 and 150.2 m^3 ha^{-1} in 2016 (Korea Forest Service 2017). Increases in forest volume had significant co-benefits such as increasing water yield by 43% and reducing soil losses by 87% from 1971 to 2010 (Kim et al. 2017).

The forest carbon density in South Korea has increased from 5–7 Mg C ha^{-1} in the period 1955–1973 to more than 30 Mg C ha^{-1} in the late 1990s (Choi et al. 2002). Estimates of C uptake rates in the late 1990s were 12 Tg C yr^{-1} (Choi et al. 2002). For the period 1954 to 2012 C uptake was 8.3 Tg C yr^{-1} (Lee et al. 2014), lower than other estimates because reforestation programs did not start until 1973. NEP in South Korea was 10.55 ± 1.09 Tg C yr^{-1} in the 1980s, 10.47 ± 7.28 Tg C yr^{-1} in the 1990s, and 6.32 ± 5.02 Tg C yr^{-1} in the 2000s, showing a gradual decline as average forest age increased (Cui et al. 2014). The estimated past and projected future increase in the carbon content of South Korea's forest area during 1992-2034 was 11.8 Tg C yr^{-1} (Kim et al. 2016).

Figure 4.9 Example of severely degraded hills in South Korea and stages of forest restoration. The top two photos are taken in the early 1970s, before and after restoration, the third photo about 5 years after restoration and the bottom photo was taken about 20 years after restoration. Many examples of such restoration success exist throughout South Korea (Source: Korea Forest Service).

During the period of forest restoration, South Korea also promoted inter-agency cooperation and coordination, especially between the energy and forest sectors, to replace firewood with fossil fuels, and by reducing demand for firewood helped forest recovery (Bae et al. 2012). As experience with forest restoration programs has increased, emphasis has shifted from fuelwood plantations, often with exotic species and hybrid varieties to planting more native species and encouraging natural regeneration (Kim and Zsuffa 1994; Lee et al. 2015). Avoiding monocultures in reforestation programs can reduce susceptibility to pests (Kim and Zsuffa 1994). Other important factors in the success of the reforestation program were that private landowners were heavily involved in initial efforts (both corporate entities and smallholders) and that the reforestation program was made part of the national economic development program (Lamb 2014).

The net present value and the benefit-cost ratio of the reforestation program were USD 54.3 billion and 5.84 billion in 2010, respectively. The breakeven point of the reforestation investment appeared within two decades. Substantial benefits of the reforestation program included disaster risk reduction and carbon sequestration (Lee et al. 2018a).

In summary, the reforestation program was a comprehensive technical and social initiative that restored forest ecosystems, enhanced the economic performance of rural regions, contributed to disaster risk reduction, and enhanced carbon sequestration (Kim et al. 2017; Lee et al. 2018a; UNDP 2017).

The success of the reforestation program in South Korea and the associated significant carbon sink indicate a high mitigation potential that might be contributed by a potential future reforestation program in the Democratic People's Republic of Korea (North Korea) (Lee et al. 2018b).

4.9.3.2 China Case Study on Reforestation Success

The dramatic decline in the quantity and quality of natural forests in China resulted in land degradation, such as soil erosion, floods, droughts, carbon emission, and damage to wildlife habitat (Liu and Diamond 2008). In response to failures of previous forestry and land policies, the severe droughts in 1997, and the massive floods in 1998, the central government decided to implement a series of land degradation control policies, including the National Forest Protection Program (NFPP), Grain for Green or the Conversion of Cropland to Forests and Grasslands Program (GFGP) (Liu et al. 2008; Yin 2009; Tengberg et al. 2016; Zhang et al. 2000). The NFPP aimed to completely ban logging of natural forests in the upper reaches of the Yangtze and Yellow rivers as well as in Hainan Province by 2000 and to substantially reduce logging in other places (Xu et al. 2006). In 2011, NFPP was renewed for the 10-year second phase, which also added another 11 counties around Danjiangkou Reservoir in Hubei and Henan Provinces, the water source for the middle route of the South-to-North Water Diversion Project (Liu et al. 2013). Furthermore, the NFPP afforested 31 Mha by 2010 through aerial seeding, artificial planting, and mountain closure (i.e., prohibition of human activities such as fuelwood collection and lifestock grazing) (Xu et al. 2006). China banned commercial logging in all natural forests by the end of 2016, which imposed logging bans and harvesting reductions in 68.2 Mha of forest land – including 56.4 Mha of natural forest (approximately 53% of China's total natural forests).

GFGP became the most ambitious of China's ecological restoration efforts with over USD 45 billion devoted to its implementation since 1990 (Kolinjivadi and Sunderland 2012) The program involves the conversion of farmland on slopes of 15-25° or greater to forest or grassland (Bennett 2008). The pilot program started in three provinces –Sichuan, Shaanxi, and Gansu – in 1999 (Liu and Diamond 2008). After initial success, it was extended to 17 provinces by 2000 and finally to all provinces by 2002, including the headwaters of the Yangtze and Yellow rivers (Liu et al. 2008).

NFPP and GFGP have dramatically improved China's land conditions and ecosystem services, and thus have mitigated the unprecedented land degradation in China (Liu et al. 2013; Liu et al 2002; Long et al. 2006; Xu et al. 2006). NFPP protected 107 Mha forest area and increased forest area by 10 Mha between 2000 and 2010. For the second phase (2011–2020), the NFPP plans to increase forest cover by a further 5.2 Mha, capture 416 million tons of carbon, provide 648,500 forestry jobs, further reduce land degradation, and enhance biodiversity (Liu et al. 2013). During 2000–2007, sediment concentration in the Yellow River had declined by 38%. In the Yellow River basin, it was estimated that surface runoff would be reduced by 450 million m^3 from 2000 to 2020, which is equivalent to 0.76% of the total surface water resources (Jia et al. 2006). GFGP had cumulatively increased vegetative cover by 25 Mha, with 8.8 Mha of cropland being converted to forest and grassland, 14.3 Mha barren land being afforested, and 2.0 million ha of forest regeneration from mountain closure. Forest cover within the GFGP region has increased 2% during the first 8 years (Liu et al. 2008). In Guizhou Province, GFGP plots had 35–53% less loss of phosphorus than non-GFGP plots (Liu et al. 2002). In Wuqi County of Shaanxi Province, the Chaigou Watershed had 48% and 55% higher soil moisture and moisture-holding capacity in GFGP plots than in non-GFGP plots, respectively (Liu et al. 2002). According to reports on China's first national ecosystem assessment (2000–2010), for carbon sequestration and soil retention, coefficients for the GTGP targeting forest restoration and NFPP are positive and statistically significant. For sand fixation, GTGP targeting grassland restoration is positive and statistically significant. Remote sensing observations confirm vegetation cover increases and bare soil decline in China over the period 2001 to 2015 (Qiu et al. 2017) (Qiu et al. 2017). But where afforestation is sustained by drip irrigation from groundwater, questions about plantation sustainability arise (Chen et al. 2018a). Moreover, greater gains in biodiversity could be achieved by promoting mixed forests over monocultures (Hua et al. 2016).

NFPP-related activities received a total commitment of 93.7 billion yuan (about USD 14 billion with today's exchange rate) between 1998 and 2009. Most of the money was used to offset economic losses of forest enterprises caused by the transformation from logging to tree plantations and forest management (Liu et al. 2008). By 2009, the cumulative total investment through the NFPP and GFGP exceeded USD 50 billion and directly involved more than 120 million farmers in 32 million households in the GFGP alone (Liu et al. 2013). All programs reduce or reverse land degradation and improve human well-being. Thus, a coupled human and natural systems perspective (Liu et al. 2008) would be helpful to understand the complexity of policies and their impacts, and to establish long-term management mechanisms to improve the livelihood of participants in these programs and other land management policies in both China and many other parts of the world.

4.9.4 Degradation and management of peat soils

Globally, peatlands cover 3-4 % of the Earth's land area (~430 Mha) (Xu et al. 2018a; Wu et al. 2017b) and store 26-44% of estimated global soil organic carbon (Moore 2002). They are most abundant in high northern latitudes, covering large areas in North America, Russia and Europe. At lower latitudes, the largest areas of tropical peatlands are located in Indonesia, the Congo Basin and the Amazon Basin in the form of peat swamp forests (Gumbricht et al. 2017; Xu et al. 2018a). It is estimated that while 80-85% of the global peatland areas is still largely in a natural state, they are such carbon-dense ecosystems that degraded peatlands (0.3% of the terrestrial land) are responsible for a disproportional 5% of global anthropogenic carbon dioxide (CO_2) emissions, that is an annual addition of 0.9-3 Gt of CO_2 to the atmosphere (Dommain et al. 2012; IPCC 2014c).

Peatland degradation is not well quantified globally, but regionally peatland degradation can involve a large percentage of the areas. Land-use change and degradation in tropical peatlands have primarily been quantified in Southeast Asia, where drainage and conversion to plantation crops is the dominant transition (Miettinen et al. 2016). Degradation of peat swamps in Peru is also a growing concern and one pilot survey showed that over 70% of the peat swamps were degraded in one region that was surveyed (Hergoualc'h et al. 2017a). Around 65,000km^2 or 10% of the European peatland area has been lost and 44% of the remaining European peatlands are degraded (Joosten, H., Tanneberger 2017). Large areas of fens have been entirely 'lost' or greatly reduced in thickness due to peat wastage (Lamers et al. 2015).

The main drivers of the acceleration of peatland degradation in the twentieth century were associated with drainage for agriculture, peat extraction and afforestation related activities (burning, over-grazing, fertilisation) with a variable scale and severity of impact depending on existing resources in the various countries (O'Driscoll et al. 2018; Abu et al. 2017; Dommain et al. 2018; Lamers et al. 2015). New drivers include urban development, wind farm construction (Smith et al. 2012), hydro-electric development, tar sands mining and recreational (Joosten, H., Tanneberger 2017). Anthropogenic pressures are now affecting peatlands in previously geographically isolated areas with consequences for global environmental concerns and impacts on local livelihoods (Dargie et al. 2017; Lawson et al. 2015; Butler et al. 2009).

Drained and managed peatlands are GHG emissions hotspots (Swails et al. 2018; Hergoualc'h et al. 2017b; Roman-Cuesta et al. 2016; Hergoualc'h et al. 2017a). In most cases, lowering of the water table leads to direct and indirect CO_2 and N_2O emissions to the atmosphere with rates dependent on a range of factors, including the groundwater level and the water content of surface peat layers, nutrient content, temperature, and vegetation communities. The exception is nutrient limited boreal peatlands (Minkkinen et al. 2018; Ojanen et al. 2014). Drainage also increases erosion and dissolved organic C loss, removing stored carbon into streams as dissolved and particulate organic carbon, which ultimately returns to the atmosphere (Moore et al. 2013; Evans et al. 2016).

In tropical peatlands, oil palm is the most widespread plantation crop and on average it emits around 40 t CO_2 ha^{-1} yr^{-1}; Acacia plantations for pulpwood are the second most widespread plantation crop and emit around 73 t CO_2 ha^{-1} yr^{-1} (Drösler et al. 2013). Other land uses typically emit less than 37 t CO_2 ha^{-1} yr^{-1}. Total emissions from peatland drainage in the region are estimated to be between 0.07 and 1.1 Gt CO_2 yr^{-1} (Houghton and Nassikas 2017; Frolking et al. 2011). Land-use change also affects the fluxes of N_2O and CH_4. Undisturbed tropical peatlands emit about 0.8 Mt CH_4 yr^{-1} and 0.002 Mt N_2O yr^{-1}, while disturbed peatlands emit 0.1 Mt CH_4 $yr^{-1}$and 0.2 Mt N_2O–N yr^{-1} (Frolking et al. 2011). These N_2O emissions are probably low as new findings show that emissions from fertilised oil palm can exceed 20 kg N_2O–N ha^{-1} yr^{-1} (Oktarita et al. 2017).

In the temperate and boreal zones, peatland drainage often leads to emissions on the order of 0.9 to 9.5 t CO_2 ha^{-1} y^{-1} in forestry plantations and 21 to 29 t CO_2 ha^{-1} y^{-1} in grasslands and croplands. Nutrient poor sites often continue to be CO_2 sinks for long periods (e.g. 50 y) following drainage and in some cases sinks for atmospheric CH_4, even when drainage ditch emissions are considered (Minkkinen et al. 2018; Ojanen et al. 2014). Undisturbed boreal and temperate peatlands emit about 030 Mt CH_4 yr^{-1} and 0.02 Mt N_2O-N yr^{-1}, while disturbed peatlands emit 0.1 Mt CH_4 $yr^{-1}$and 0.2 Mt yr^{-1} N_2O (Frolking et al. 2011).

Fire emissions from tropical peatlands are only a serious issue in Southeast Asia, where they are responsible for 634 (66–4070) Mt CO_2 yr^{-1} (van der Werf et al. 2017). Much of the variability is linked with the El Niño Southern Oscillation, which produces drought conditions in this region. Anomalously active fire seasons have also been observed in non-drought years and this has been attributed to the increasing effect of high temperatures that dry vegetation out during short dry spells in otherwise normal rainfall years (Fernandes et al. 2017; Gaveau et al. 2014). Fires have significant societal impacts; for example, the 2015 fires caused over 100,000 additional deaths across Indonesia, Malaysia and Singapore and this event was more than twice as deadly as the 2006 El Niño event (Koplitz et al. 2016).

Peatland degradation in other parts of the world differs from Asia. In Africa large peat deposits like those found in the Cuvette Centrale in the Congo Basin or in the Okavango inland delta, the principle threat is changing rainfall regimes due to climate variability and change (Weinzierl et al. 2016; Dargie et al. 2017). Expansion of agriculture is not yet a major factor in these regions. In the Western Amazon, extraction of non-timber forest products like the fruits of *Mauritia flexuosa* (moriche palm) and Suri worms are major sources of degradation that lead to losses of carbon stocks (Hergoualc'h et al. 2017a).

The effects of peatland degradation on livelihoods have not been systematically characterised. In places where plantation crops are driving the conversion of peat swamps, the financial benefits can be considerable. One study in Indonesia found that the net present value of an oil palm plantation is between USD 3,835 and 9,630 per ha to land owners (Butler et al. 2009). High financial returns are creating the incentives for the expansion of smallholder production in peatlands. Smallholder plantations extend over 22% of the peatlands in insular Southeast Asia compared to 27% for industrial plantations (Miettinen et al. 2016). In places where income is generated from extraction of marketable products, ecosystem degradation probably has a negative effect on livelihoods. For example, the sale of fruits of *M. flexuosa* in some parts of the western Amazon constitutes as much as 80% of the winter income of many rural households, but information on trade values and value chains of *M. flexuosa* is still sparse (Sousa et al. 2018; Virapongse et al. 2017).

There is little experience with peatland restoration in the tropics. Experience from northern latitudes suggests that extensive damage and changes in hydrological conditions mean that restoration in many cases is unachievable (Andersen et al. 2017). In the case of Southeast Asia, where peatlands form as raised bogs, drainage leads to collapse of the dome and this collapse cannot be reversed by rewetting. Nevertheless, efforts are underway to develop solutions or at least partial solutions in Southeast Asia, for example, by the Indonesian Peatland Restoration Agency. The first step is to restore the hydrological regime in drained peatlands and experiences with canal blocking and re-flooding of the peat. These efforts have been only partially successful (Ritzema et al. 2014). Market incentives with certification through the Roundtable on Sustainable Palm Oil have also not been particularly successful as many concessions seek certification only after significant environmental degradation has been accomplished (Carlson et al. 2017). Certification had no discernible effect on forest loss or fire detection in peatlands in Indonesia. To date there is no documentation of restoration methods or successes in many other parts of the tropics, but in situations where degradation does not involve drainage, ecological restoration may be possible. In South America, for example, there is growing interest in restoration of palm swamps, and as experiences are gained it will be important to document success factors to inform successive efforts (Virapongse et al. 2017).

In higher latitudes where degraded peatlands have been drained, the most effective option to reduce losses from these large organic carbon stocks is change hydrological conditions and increase soil moisture and surface wetness (Regina et al. 2015). Long-term GHG monitoring in boreal sites has demonstrated that rewetting and restoration noticeably reduce emissions compared to degraded drained sites and can restored the carbon sink function when vegetation is re-established (Wilson et al. 2016; IPCC 2014a; Nugent et al. 2018) although restored ecosystems may not yet be as resilient as their undisturbed counterparts (Wilson et al. 2016). Several studies have demonstrated the co-benefits of rewetting specific degraded peatlands for biodiversity, carbon sequestration, (Parry et al. 2014; Ramchunder et al. 2012; Renou-Wilson et al. 2018) and other ecosystem services such as improvement of water storage and quality (Martin-Ortega et al. 2014) with beneficial consequences for human well-being (Bonn et al. 2016; Parry et al. 2014).

4.9.5 Biochar

Biochar is organic matter that is carbonised by heating in an oxygen-limited environment, and used as a soil amendment. The properties of biochar vary widely, dependent on the feedstock and the conditions of production. Biochar could make a significant contribution to mitigating both land degradation and climate change, simultaneously.

4.9.5.1 Role of biochar in climate change mitigation

Biochar is relatively resistant to decomposition compared with fresh organic matter or compost, so represents a long-term C store (*very high confidence*). Biochars produced at higher temperature (> 450°C) and from woody material have greater stability than those produced at lower temperature (300-450°C), and from manures (*very high confidence*) (Singh et al. 2012; Wang et al. 2016b). Biochar stability is influenced by soil properties: biochar carbon can be further stabilised by interaction with clay minerals and native soil organic matter (*medium evidence*) (Fang et al. 2015). Biochar stability is estimated to range from decades to thousands of years, for different biochars in different applications (Singh et al., 2015; Wang et al., 2016). Biochar stability decreases as ambient temperature increases (*limited evidence*) (Fang et al. 2017).

Biochar can enhance soil carbon stocks through "negative priming", in which rhizodeposits are stabilised through sorption of labile C on biochar, and formation of biochar-organo-mineral complexes (Weng et al. 2015, 2017, 2018; Wang et al. 2016b). Conversely, some studies show increased turnover of native soil carbon ("positive priming") due to enhanced soil microbial activity induced by biochar. In clayey soils, positive priming is minor and short-lived compared to negative priming effects, which dominate in the medium to long-term (Singh and Cowie 2014; Wang et al. 2016b). Negative priming has been observed particularly in loamy grassland soil (Ventura et al.

2015) and clay-dominated soils, whereas positive priming is reported in sandy soils (Wang et al. 2016b) and those with low C content (Ding et al. 2018).

Biochar can provide additional climate change mitigation by decreasing nitrous oxide (N_2O) emissions from soil, due in part to decreased substrate availability for denitrifying organisms, related to the molar H/C ratio of the biochar (Cayuela et al. 2015). However, this impact varies widely: meta-analyses found an average decrease in N_2O emissions from soil of 30-54%, (Cayuela et al. 2015) (Moore 2002; Borchard et al. 2019), although another study found no significant reduction in field conditions when weighted by the inverse of the number of observations per site (Verhoeven et al. 2017). Biochar has been observed to reduce methane emissions from flooded soils, such as rice paddies, though, as for N_2O, results vary between studies and increases have also been observed (He et al. 2017; KAMMANN et al. 2017). Biochar has also been found to reduces methane uptake by dryland soils, though the effect is small in absolute terms (Jeffery et al. 2016).

Additional climate benefits of biochar can arise through reduced N fertiliser requirements, due to reduced losses of N through leaching and/or volatilization (Singh, Hatton, Balwant, & Cowie, 2010) and enhanced biological nitrogen fixation (Van Zwieten et al. 2015); increased yields of crop, forage, vegetable and tree species (Biederman and Stanley Harpole 2013), particularly in sandy soils and acidic tropical soils (Simon et al. 2017) ; avoided GHG emissions from manure that would otherwise be stockpiled, crop residues that would be burned or processing residues that would be landfilled; and reduced GHG emissions from compost when biochar is added (Agyarko-Mintah et al. 2017; Wu et al. 2017a).

Climate benefits of biochar could be substantially reduced through reduction in albedo if biochar is surface-applied at high rates to light-colored soils (Genesio et al. 2012; Bozzi et al. 2015; Woolf et al. 2010), or if black carbon dust is released (Genesio et al. 2016). Pelletizing or granulating biochar, and applying below the soil surface or incorporating into the soil, minimises the release of black carbon dust and reduces the effect on albedo (Woolf et al. 2010).

Biochar is a potential "negative emissions" technology: the thermochemical conversion of biomass to biochar slows mineralisation of the biomass, delivering long term C storage; gases released during pyrolysis can be combusted for heat or power, displacing fossil energy sources, and could be captured and sequestered if linked with infrastructure for carbon capture and storage (Smith 2016). Studies of the life cycle climate change impacts of biochar systems generally show emissions reduction in the range 0.4 -1.2 t CO_2e t^{-1} (dry) feedstock (Cowie et al. 2015). Use of biomass for biochar can deliver greater benefits than use for bioenergy, if applied in a context where it delivers agronomic benefits and/or reduces non-CO_2 GHG emissions (Ji et al. 2018; Woolf et al. 2010, 2018; Xu et al. 2019). A global analysis of technical potential, in which biomass supply constraints were applied to protect against food insecurity, loss of habitat and land degradation, estimated technical potential abatement of 3.7 - 6.6 Gt CO_2e yr^{-1} (including 2.6-4.6 GtCO_2e yr^{-1} carbon stabilization), with theoretical potential to reduce total emissions over the course of a century by 240 – 475 Gt CO_2e (Woolf et al. 2010). Fuss et al. 2018 propose a range of 0.5-2 GtCO_2e as the sustainable potential for negative emissions through biochar. Mitigation potential of biochar is reviewed in Chapter 2.

4.9.5.2 Role of biochar in management of land degradation

Biochars generally have high porosity, high surface area and surface-active properties that lead to high absorptive and adsorptive capacity, especially after interaction in soil (Joseph et al. 2010). As a result of these properties, biochar could contribute to avoiding, reducing and reversing land degradation through the following documented benefits:

- Improved nutrient use efficiency due to reduced leaching of nitrate and ammonium (e.g. (Haider et al. 2017) and increased availability of phosphorus (P) in soils with high P fixation capacity (Liu et al. 2018c), potentially reducing N and P fertiliser requirements.

- Management of heavy metals and organic pollutants: through reduced bioavailability of toxic elements (O'Connor et al., 2018; Peng ; Deng, ; Peng, & Yue, 2018), by reducing availability, through immobilization due to increased pH and redox effects (Rizwan et al. 2016) and adsorption on biochar surfaces (Zhang et al. 2013) thus providing a means of remediating contaminated soils, and enabling their utilisation for food production.
- Stimulation of beneficial soil organisms, including earthworms and mycorrhizal fungi (Thies et al. 2015).
- Improved porosity and water holding capacity (Quin et al. 2014), particularly in sandy soils (Omondi et al. 2016), enhancing microbial function during drought (Paetsch et al. 2018).
- Amelioration of soil acidification, through application of biochars with high pH and acid neutralising capacity (Chan et al. 2008)(Van Zwieten et al. 2010).

Biochar systems can deliver a range of other co-benefits including destruction of pathogens and weed propagules, avoidance of landfill, improved handling and transport of wastes such as sewage sludge, management of biomass residues such as environmental weeds and urban greenwaste, reduction of odors and management of nutrients from intensive livestock facilities, reduction in environmental N pollution and protection of waterways. As a compost additive, biochar has been found to reduce leaching and volatilisation of nutrients, increasing nutrient retention, through absorption and adsorption processes (Joseph et al. 2018).

While many studies report positive responses, some studies have found negative or zero impacts on soil properties or plant response (e.g. Kuppusamy, Thavamani, Megharaj, Venkateswarlu, & Naidu, 2016). The risk that biochar may enhance PAH in soil or sediments has been raised (Quilliam et al. 2013; Ojeda et al. 2016), but bioavailability of PAH in biochar has been shown to be very low (Hilber et al. 2017) Pyrolysis of biomass leads to losses of volatile nutrients, especially N. While availability of N and P in biochar is lower in biochar than in fresh biomass (Xu et al. 2016) the impact of biochar on plant uptake is determined by the interactions between biochar, soil minerals and activity of microorganisms (e.g. (Vanek and Lehmann 2015); (Nguyen et al. 2017). To avoid negative responses, it is important to select biochar formulations to address known soil constraints, and to apply biochar prior to planting (Nguyen et al., 2017). Nutrient enrichment improves the performance of biochar from low nutrient feedstocks (Joseph et al. 2013). While there are many reports of biochar reducing disease or pest incidence, there are also reports of nil or negative effects (Bonanomi et al. 2015). Biochar may induce systemic disease resistance (e.g., Elad et al. 2011)), though (Viger et al. 2015) reported down-regulation of plant defence genes, suggesting increased susceptibility to insect and pathogen attack. Disease suppression where biochar is applied is associated with increased microbial diversity and metabolic potential of the rhizosphere microbiome (Kolton et al. 2017). Differences in properties related to feedstock (Bonanomi et al. 2018) and differential response to biochar dose, with lower rates more effective (Frenkel et al. 2017) in contributing to variable disease responses.

Constraints to biochar adoption are high cost and limited availability due to limited large-scale production; limited amount of unutilised biomass; and competition for land for growing biomass. While early biochar research tended to use high rates of application (10 t ha^{-1} or more) subsequent studies have shown that biochar can be effective at lower rates especially when combined with chemical or organic fertilisers (Joseph et al. 2013). Biochar can be produced at many scales and levels of engineering sophistication, from simple cone kilns and cookstoves to large industrial scale units processing several tonnes of biomass per hour (Lehmann and Stephen 2015). Substantial technological development has occurred recently, though large-scale deployment is limited to date.

Governance of biochar is required to manage climate, human health and contamination risks associated with biochar production in poorly-designed or operated facilities that release methane or particulates (Downie et al. 2012)(Buss et al. 2015), to ensure quality control of biochar products, and to ensure biomass is sourced sustainably and is uncontaminated. Measures could include labelling standards, sustainability certification schemes and regulation of biochar production and use. Governance mechanisms should be tailored to context, commensurate with risks of adverse outcomes.

In summary, application of biochar to soil can improve soil chemical, physical and biological attributes, enhancing productivity and resilience to climate change, while also delivering climate change mitigation through carbon sequestration and reduction in GHG emissions (*medium agreement, robust evidence*). However, responses to biochar depend on biochar properties, in turn dependent on feedstock and biochar production conditions, and the soil and crop to which it is applied. Negative or nil results have been recorded. Agronomic and methane reduction benefits appear greatest in tropical regions, where acidic soils predominate and suboptimal rates of lime and fertiliser are common, while carbon stabilisation is greater in temperate regions. Biochar is most effective when applied in low volumes to the most responsive soils and when properties are matched to the specific soil constraints and plant needs. Biochar is thus a practice that has potential to address land degradation and climate change simultaneously, while also supporting sustainable development. The potential of biochar is limited by the availability of biomass for its production. Biochar production and use requires regulation and standardisation to manage risks (*strong agreement*).

4.9.6 Management of land degradation induced by tropical cyclones

Tropical cyclones are normal disturbances that natural ecosystems have been affected by and recovered from for millennia. Climate models mostly predict decreasing frequency of tropical cyclones, but dramatically increasing intensity of the strongest storms as well as increasing rainfall rates (Bacmeister et al. 2018; Walsh et al. 2016b). Large amplitude fluctuations in the frequency and intensity complicate both the detection and attribution of tropical cyclones to climate change (Lin and Emanuel 2016b). Yet, the intensity of high-intensity cyclones have increased and are expected to increase further due to global climate change (Knutson et al. 2010; Bender et al. 2010; Vecchi et al. 2008; Bhatia et al. 2018; Tu et al. 2018; Sobel et al. 2016) (*medium agreement, robust evidence*). Tropical cyclone paths are also shifting towards the poles increasing the area subject to tropical cyclones (Sharmila and Walsh 2018; Lin and Emanuel 2016b). Climate change alone will affect the hydrology of individual wetland ecosystems mostly through changes in precipitation and temperature regimes with great global variability (Erwin 2009). Over the last seven decades, the speed at which tropical cyclones move has decreased significantly as expected from theory, exacerbating the damage on local communities from increasing rainfall amounts and high wind speed (Kossin 2018). Tropical cyclones will accelerate changes in coastal forest structure and composition. The heterogeneity of land degradation at coasts that are affected by tropical cyclones can be further enhanced by the interaction of its components (for example, rainfall, wind speed, and direction) with topographic and biological factors (for example, species susceptibility) (Luke et al. 2016).

Small Island Developing States (SIDS) are particularly affected by land degradation induced by tropical cyclones, recent examples are Matthew (2016) in the Caribbean, and Pam (2015) and Winston (2016) in the Pacific (Klöck and Nunn 2019; Handmer and Nalau 2019). Even if the Pacific Ocean has experienced cyclones of unprecedented intensity in the recent years, their geomorphological effects may not be unprecedented (Terry and Lau 2018).

Cyclone impacts on coastal areas is not restricted to SIDS, but a problem for all low-lying coastal areas (Petzold and Magnan 2019). The Sundarban, one of the world's largest coastal wetlands, covers about one million hectares between Bangladesh and India. Large areas of the Sundarban mangroves have been converted into paddy fields over the past two centuries and more recently into shrimp farms (Ghosh et al. 2015). In 2009 the cyclone Aila caused incremental stresses on the socioeconomic conditions of the Sundarban coastal communities through rendering huge areas of land unproductive for a long time (Abdullah et al. 2016). The impact of Aila was wide spread throughout the Sundarbans mangroves showing changes between pre- and post-cyclonic period of 20-50% in the enhanced vegetation index (Dutta et al. 2015). Although the magnitude of the effects of the Sundarban mangroves derived from climate change is not yet defined (Payo et al. 2016; Loucks et al. 2010; Gopal and Chauhan 2006; Ghosh et al. 2015; Chaudhuri et al. 2015). There is *high agreement* that the joint effect of climate change and land degradation will be very negative for the area, strongly affecting the environmental services provided by these forests, including the extinction of large mammal species (Loucks et al. 2010). This changes in vegetation are mainly due to inundation and erosion (Payo et al. 2016).

The tropical cyclone Nargis hit unexpectedly the Ayeyarwady River delta (Myanmar) in 2008 with unprecedented and catastrophic damages to livelihoods, destruction of forests and erosion of fields (Fritz et al. 2009) as well as eroding the shoreline 148 m compared with the long-term average (1974-2015) of 0.62 m yr^{-1}. This is an example of the disastrous effects that changing cyclone paths can have on areas previously not affected by cyclones (Fritz et al. 2010).

4.9.6.1 Management of coastal wetlands

Tropical cyclones mainly, but not exclusively, affect coastal regions, threatening maintenance of the associated ecosystems, mangroves, wetlands, seagrasses, etc. This areas not only provide food, water and shelter for fish, birds and other wildlife, but also provide important ecosystem services such as water quality improvement, flood abatement and carbon sequestration (Meng et al. 2017).

Despite its importance coastal wetlands are listed amongst the most heavily damaged of natural ecosystems worldwide. Starting in the 1990s, wetland restoration and re-creation became a “hotspot” in the ecological research fields (Zedler 2000). The coastal wetland restoration and preservation is an extremely cost-effective strategy for society, for example the preservation of coastal wetlands in the USA provide storm protection services with the cost of 23.2 billion yr^{-1} USD (Costanza et al. 2008).

There is a *high agreement* with *medium evidence* that the success of wetland restoration depends mainly on the flow of the water through the system and the degree to which re-flooding occurs, the disturbance regimes, and the control of invasive species (Burlakova et al. 2009; López-Rosas et al. 2013). The implementation of the Ecological Mangrove Rehabilitation (EMR) protocol (López-Portillo et al. 2017) that includes monitoring and reporting tasks, has been proven to deliver successful rehabilitation of wetland ecosystem services.

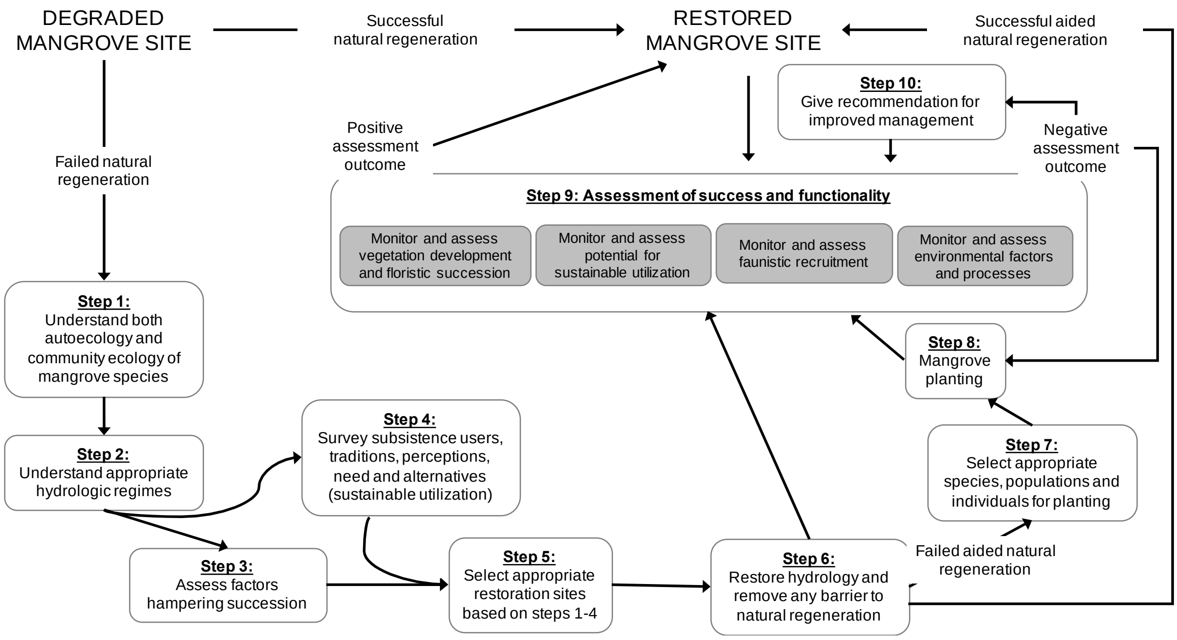


Figure 4.10 Decision tree showing recommended steps and tasks to restore a mangrove wetland based on original site conditions (Modified from Bosire et al. (2008)

4.9.7 Saltwater intrusion

Current environmental changes, including climate change, have caused sea levels to rise worldwide, particularly in tropical and subtropical regions (Fasullo and Nerem 2018). Combined with scarcity of water in river channels, such rises have been instrumental in the intrusion of highly saline seawater inland, posing a threat to coastal areas and an emerging challenge to land managers and policy makers. Assessing the extent of salinisation due to sea water intrusion at a global scale nevertheless remains challenging. Wicke et al. (2011) suggest that across the world, approximately 1.1 Gha of land is affected by salt, with 14% of this categorised as forest, wetland or some other form of protected area. Seawater intrusion is generally caused by: i) increased tidal activity, storm surges, cyclones and sea storms due to changing climate, ii) heavy groundwater extraction or land use changes as a result of changes in precipitation, and droughts/floods, iii) coastal erosion as a result of destruction of mangrove forests and wetlands iv) construction of vast irrigation canals and drainage networks leading to low river discharge in the deltaic region; and v) sea level rise contaminating nearby freshwater aquifers as a result of subsurface intrusion (Uddameri et al. 2014).

The Indus delta, located in the south-eastern coast of Pakistan near Karachi in the North Arabian sea, is one of the six largest estuaries in the world spanning an area of 600,000 ha. The Indus delta is a clear example of seawater intrusion and land degradation due to local as well as up-country climatic and environmental conditions (Rasul et al. 2012). Salinisation and waterlogging in the up-country areas including provinces of Punjab and Sindh is, however, caused by the irrigation network and over-irrigation (Qureshi 2011).

Such degradation takes the form of high soil salinity, inundation and waterlogging, erosion and freshwater contamination. The inter-annual variability of precipitation with flooding conditions in some years and drought conditions in others has caused variable river flows and sediment runoff below Kotri barrage (about 200 km upstream of the Indus delta). This has affected hydrological processes in the lower reaches of the river and the delta, contributing to the degradation (Rasul et al. 2012).

Over 480,000 ha of fertile land is now affected by sea water intrusion, wherein eight coastal subdivisions of the districts of Badin and Thatta are mostly affected (Chandio et al. 2011). A very high intrusion rate of 0.179±0.0315 km yr^{-1}, based on the analysis of satellite data, was observed in the Indus delta during the past 10 years (2004–2015) (Kalhoro et al. 2016). The area of agricultural crops under cultivation has been declining with economic losses of millions of USD (IUCN 2003). Crop yields have reduced due to soil salinity, in some places failing entirely. Soil salinity varies seasonally, depending largely on the river discharge: during the wet season (August 2014), salinity (0.18 mg L^{-1}) reached 24 km upstream while during the dry season (May 2013), it reached 84 km upstream (Kalhoro et al. 2016). The freshwater aquifers have also been contaminated with sea water rendering them unfit for drinking or irrigation purposes. Lack of clean drinking water and sanitation causes widespread diseases, of which diarrhoea is most common (IUCN 2003).

Lake Urmia in northwest Iran, the second largest saltwater lake in the world and the habitat for endemic Iranian brine shrimp, *Artemia urmiana*, has also been affected by salty water intrusion. During a 17-year period between 1998 and 2014, human disruption including agriculture and years of dam building affected the natural flow of freshwater as well as salty sea water in the surrounding area of Lake Urmia. Water quality has also been adversely affected, with salinity fluctuating over time, but in recent years reaching a maximum of 340 g L^{-1} (similar to levels in the Dead Sea). This has rendered the underground water unfit for drinking and agricultural purposes and risky to human health and livelihoods. Adverse impacts of global climate change as well as direct human impacts have caused changes in land use, overuse of underground water resources and construction of dams over rivers which resulted in the drying-up of the lake in large part. This condition created sand, dust and salt storms in the region which affected many sectors including agriculture, water resources, rangelands, forests and health, and generally presented desertification conditions around the lake (Karbassi et al. 2010; Marjani and Jamali 2014; Shadkam et al. 2016).

Rapid irrigation expansion in the basin has, however, indirectly contributed to inflow reduction. Annual inflow to Lake Urmia has dropped by 48% in recent years. About three fifths of this change was caused by climate change and two fifths by water resource development and agriculture (Karbassi et al. 2010; Marjani and Jamali 2014; Shadkam et al. 2016).

In the drylands of Mexico, intensive production of irrigated wheat and cotton using groundwater (Halvorson et al. 2003) resulted in sea water intrusion into the aquifers of La Costa de Hermosillo, a coastal agricultural valley at the center of Sonora Desert in Northwestern Mexico. Production of these crops in 1954 was on 64,000 ha of cultivated area, increasing to 132,516 ha in 1970, but decreasing to 66,044 ha in 2009 as a result of saline intrusion from the Gulf of California (Romo-Leon et al. 2014). In 2003, only 15% of the cultivated area was under production, with around 80,000 ha abandoned due to soil salinisation whereas in 2009, around 40,000 ha was abandoned (Halvorson et al. 2003; Romo-Leon et al. 2014). Salinisation of agricultural soils could be exacerbated by climate change, as Northwestern Mexico is projected to be warmer and drier under climate change scenarios (IPCC 2013a).

In other countries, intrusion of seawater is exacerbated by destruction of mangrove forests. Mangroves are important coastal ecosystems that provide spawning bed for fish, timber for building, livelihoods to dependent communities, act as barriers against coastal erosion, storm surges, tropical cyclones and tsunamis (Kalhoro et al. 2017) and are among the most carbon-rich stocks on Earth (Atwood et al. 2017). They nevertheless face a variety of threats: climatic (storm surges, tidal activities, high temperatures) and human (coastal developments, pollution, deforestation, conversion to aquaculture, rice culture, oil palm plantation), leading to declines in their areas. In Pakistan, using remote sensing (RS), the mangrove forest cover in the Indus delta decreased from 260,000 ha in 1980s to 160,000 ha in 1990 (Chandio et al. 2011). Based on remotely sensed data, a sharp decline in the mangrove area was also found in the arid coastal region of Hormozgan province in southern Iran during 1972, 1987 and 1997 (Etemadi et al. 2016). Myanmar has the highest rate (about 1% yr^{-1}) of mangrove deforestation in the world (Atwood et al. 2017). Regarding global loss of carbon stored in the mangrove due to deforestation, four countries exhibited high levels of loss: Indonesia (3,410 Gg

CO_2 yr^{-1}), Malaysia (1,288 GgCO_2 yr^{-1}), US (206 Gg CO_2 yr^{-1}) and Brazil (186 GgCO_2 yr^{-1}). Only in Bangladesh and Guinea Bissau there was no decline in the mangrove area from 2000 to 2012 (Atwood et al. 2017).

Frequency and intensity of average tropical cyclones will continue to increase (Knutson et al. 2015) and global sea level will continue to rise. The IPCC (2013) projected with *medium confidence* that sea level in the Asia Pacific region will rise from 0.4 to 0.6 m, depending on the emission pathway, by the end of this century. Adaptation measures are urgently required to protect the world's coastal areas from further degradation due to saline intrusion. A viable policy framework is needed to ensure the environmental flows to deltas in order to repulse the intruding seawater.

4.9.8 Avoiding coastal maladaptation

Coastal degradation—for example, beach erosion, coastal squeeze, and coastal biodiversity loss—as a result of rising sea levels is a major concern for low lying coasts and small islands (*high confidence*). The contribution of climate change to increased coastal degradation has been well documented in AR5 (Nurse et al. 2014; Wong et al. 2014) and is further discussed in Section 4.4.1.3. as well as in the IPCC Special Report on the Ocean and Cryosphere in a Changing Climate (SROCC). However, coastal degradation can also be indirectly induced by climate change as the result of adaptation measures that involve changes to the coastal environment, for example, coastal protection measures against increased flooding and erosion due to sea level rise and storm surges transforming the natural coast to a 'stabilised' coastline (Cooper and Pile 2014; French 2001). Every kind of adaptation response option is context-dependent, and, in fact, sea walls play an important role for adaptation in many places. Nonetheless, there are observed cases where the construction of sea walls can be considered 'maladaptation' (Barnett and O'Neill 2010; Magnan et al. 2016) by leading to increased coastal degradation, such as in the case of small islands, where due to limitations of space coastal retreat is less of an option than in continental coastal zones. There is emerging literature on the implementation of alternative coastal protection measures and mechanisms on small islands to avoid coastal degradation induced by sea walls (e.g., Mycoo and Chadwick 2012; Sovacool 2012).

In many cases, increased rates of coastal erosion due to the construction of sea walls are the result of the negligence of local coastal morphological dynamics and natural variability as well as the interplay of environmental and anthropogenic drivers of coastal change (*medium evidence*, *high agreement*). Sea walls in response to coastal erosion may be ill-suited for extreme wave heights under cyclone impacts and can lead to coastal degradation by keeping overflowing sea water from flowing back into the sea, and therefore affect the coastal vegetation through saltwater intrusion, as observed in Tuvalu (Government of Tuvalu 2006; Wairiu 2017). Similarly, in Kiribati, poor construction of sea walls has resulted in increased erosion and inundation of reclaimed land (Donner 2012; Donner and Webber 2014). In the Comoros and Tuvalu, sea walls have been constructed from climate change adaptation funds and 'often by international development organizations seeking to leave tangible evidence of their investments' (Marino and Lazrus 2015, p. 344). In these cases, they have even increased coastal erosion, due to poor planning and the negligence of other causes of coastal degradation, such as sand mining (Marino and Lazrus 2015; Betzold and Mohamed 2017; Ratter et al. 2016). On the Bahamas, the installation of sea walls as a response to coastal erosion in areas with high wave action has led to the contrary effect and even increased sand loss in those areas (Sealey 2006). The reduction of natural buffer zones—i.e., beaches and dunes—due to vertical structures, such as sea walls, increased the impacts of tropical cyclones on Reunion Island (Duvat et al. 2016). Such a process of 'coastal squeeze' (Pontee 2013) also results in the reduction of intertidal habitat zones, such as wetlands and marshes (Linham and Nicholls 2010). Coastal degradation resulting from the construction of sea walls, however, is not only observed in Small Island Developing States (SIDS), as described above, but also on islands in the Global North, for example, the North Atlantic (Muir et al. 2014; Young et al. 2014; Cooper and Pile 2014; Bush 2004).

The adverse effects of coastal protection measures may be avoided by the consideration of local social-ecological dynamics, including the critical studying of diverse drivers of ongoing shoreline changes, and the according implementation of locally adequate coastal protection options (French 2001; Duvat 2013). Critical elements for avoiding maladaptation include profound knowledge of local tidal regimes, availability of relative sea level rise scenarios and projections for extreme water levels. Moreover, the downdrift effects of sea walls need to be considered, since undefended coasts may be exposed to increased erosion (Linham and Nicholls 2010). In some cases, it may be possible to keep intact and restore natural buffer zones as an alternative to the construction of hard engineering solutions. Otherwise, changes in land-use, building codes, or even coastal realignment can be an option in order to protect and avoid the loss of the buffer function of beaches (Duvat et al. 2016; Cooper and Pile 2014). Examples of Barbados show that combinations of hard and soft coastal protection approaches can be sustainable and reduce the risk of coastal ecosystem degradation while keeping the desired level of protection for coastal users (Mycoo and Chadwick 2012). Nature-based solutions and approaches such as ‘building with nature’ (Slobbe et al. 2013) may allow for more sustainable coastal protection mechanisms and avoid coastal degradation. Examples from the Maldives, several Pacific islands and the North Atlantic show the importance of the involvement of local communities in coastal adaptation projects, considering local skills, capacities, as well as demographic and socio-political dynamics, in order to ensure the proper monitoring and maintenance of coastal adaptation measures (Sovacool 2012; Muir et al. 2014; Young et al. 2014; Buggy and McNamara 2016; Petzold 2016).

4.10 Knowledge gaps and key uncertainties

The co-benefits of improved land management, such as mitigation of climate change, increased climate resilience of agriculture, and impacts on rural areas/societies are well-known in theory but there is a lack of a coherent and systematic global inventory of such integrated efforts. Both successes and failures are important to document systematically.

Efforts to reduce climate change through land-demanding mitigation actions aimed at removing atmospheric carbon, such as afforestation, reforestation, bioenergy crops, intensification of land management and plantation forestry can adversely affect land conditions and lead to degradation. However, they may also lead to avoidance, reduction and reversal of degradation. Regionally differentiated, socially and ecologically appropriate sustainable land management strategies need to be identified, implemented, monitored and the results communicated widely to ensure climate effective outcomes.

Impacts of new technologies on land degradation and their social and economic ramifications need more research.

Improved quantification of the global extent, severity and rates of land degradation by combining remote sensing with a systematic use of ancillary data is a priority. The current attempts need a better scientific underpinning and appropriate funding.

Land degradation is defined using multiple criteria but the definition does not provide thresholds or the magnitude of acceptable change. In practice, human interactions with land will result in a variety of changes, some may contribute positively to one criterion while adversely affecting another. Research is required on the magnitude of impacts and the resulting trade-offs. Given the urgent need to remove carbon from the atmosphere and to reduce climate change impacts, it is important to reach agreement on what level of reduction in one criterion (biological productivity, ecological integrity) may be acceptable for a given increase in another criterion (ecological integrity, biological productivity)?

Attribution of land degradation to the underlying drivers is a challenge because a complex web of causality rather than simple cause-effect relationships. Also, diverging views on land degradation in relation to other challenges is hampering such efforts.

A more systematic treatment of the views and experiences of land users would be useful in land degradation studies.

Much research has tried to understand how social and ecological systems are affected by a particular stressor, for example drought, heat, or waterlogging. But less research has tried to understand how such systems are affected by several simultaneous stressors – which of course is more realistic in the context of climate change (Mittler 2006).

More realistic modelling of carbon dynamics, including better appreciation of belowground biota, would help us to better quantify the role of soils and soil management for soil carbon sequestration.

Frequently Asked Questions

FAQ 4.1 How do climate change and land degradation interact with land use?

Climate change, land degradation, and land use are linked in a complex web of causality. One important impact of climate change (e.g. flood and drought) on land degradation is that increasing global temperatures intensify the hydrological cycle resulting in more intense rainfall, which is an important driver of soil erosion. This means that sustainable land management (SLM) becomes even more important with climate change. Land-use change in the form of clearing of forest for rangeland and cropland (e.g., for provision of bio-fuels), and cultivation of peat soils, is a major source of greenhouse gas emission from both biomass and soils. Many SLM practices (e.g., agroforestry, shifting perennial crops, restoration, etc.) increase carbon content of soil and vegetation cover and hence provide both local and immediate adaptation benefits combined with global mitigation benefits in the long term, while providing many social and economic co-benefits. Avoiding, reducing and reversing land degradation has a large potential to mitigate climate change and help communities to adapt to climate change.

FAQ 4.2 How does climate change affect land-related ecosystem services and biodiversity?

Climate change will affect land-related ecosystem services (e.g. pollination, resilience to extreme climate events, water yield, soil conservation, carbon storage, etc.) and biodiversity, both directly and indirectly. The direct impacts range from subtle reductions or enhancements of specific services, such as biological productivity, resulting from changes in temperature, temperature variability or rainfall, to complete disruption and elimination of services. Disruptions of ecosystem services can occur where climate change causes transitions from one biome to another, e.g., forest to grassland as a result of changes in water balance or natural disturbance regimes. Climate change will result in range shifts and, in some cases, extinction of species. Climate change can also alter the mix of land-related ecosystem services, such as groundwater recharge, purification of water, and flood protection. While the net impacts are specific to ecosystem types, ecosystem services and time, there is an asymmetry of risk such that overall impacts of climate change are expected to reduce ecosystem services. Indirect impacts of climate change on land-related ecosystem services include those that result from changes in human behavior, including potential large-scale human migrations or the implementation of afforestation, reforestation or other changes in land management, which can have positive or negative outcomes on ecosystem services.

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Chapter 5: Food Security

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Executive summary

The current food system (production, transport, processing, packaging, storage, retail, consumption, loss and waste) feeds the great majority of world population and supports the livelihoods of ca. 200 million people. Since 1961, food supply per capita has increased more than 30%, accompanied by greater use of nitrogen fertilisers (increase of about 800%) and water resources for irrigation (increase of more than 100%). However, an estimated 821 million people are currently undernourished, 151 million children under 5 are stunted, 613 million women and girls aged 15 to 49 suffer from iron deficiency, and 2 billion adults are overweight or obese. The food system is under pressure from non-climate stressors (e.g., population and income growth, demand for animal-sourced products), and from climate change. These climate and non-climate stresses are impacting the four pillars of food security (availability, access, utilisation, and stability). {5.1.1, 5.1.2}

Observed climate change is already affecting food security through increasing temperatures, changing precipitation patterns, and greater frequency of some extreme events *(high confidence)*. Increasing temperatures are affecting agricultural productivity in higher latitudes, raising yields of some crops (maize, cotton, wheat, sugar beets), while yields of others (maize, wheat, barley) are declining in lower-latitude regions. Warming compounded by drying has caused yield declines in parts of Southern Europe. Based on indigenous and local knowledge, climate change is affecting food security in drylands, particularly those in Africa, and high mountain regions of Asia and South America. {5.2.2}

Food security will be increasingly affected by projected future climate change *(high confidence)*. Across SSPs 1, 2, and 3, global crop and economic models projected a 1-29% cereal price increase in 2050 due to climate change (RCP 6.0), which would impact consumers globally through higher food prices; regional effects will vary *(high confidence)*. Low-income consumers are particularly at risk, with models projecting increases of 1-183 million additional people at risk of hunger across the SSPs compared to a no climate change scenario *(high confidence)*. While increased CO_2 is projected to be beneficial for crop productivity at lower temperature increases, it is projected to lower nutritional quality *(high confidence)* (e.g., wheat grown at 546-586 ppm CO_2 has 5.9–12.7% less protein, 3.7–6.5% less zinc, and 5.2–7.5% less iron). Distributions of pests and diseases will change, affecting production negatively in many regions *(high confidence)*. Given increasing extreme events and interconnectedness, risks of food system disruptions are growing *(high confidence)*. {5.2.3, 5.2.4}

Vulnerability of pastoral systems to climate change is very high (*high confidence*). Pastoralism is practiced in more than 75% of countries by between 200 and 500 million people, including nomadic communities, transhumant herders, and agro-pastoralists. Impacts in pastoral systems include lower pasture and animal productivity, damaged reproductive function, and biodiversity loss. Pastoral system vulnerability is exacerbated by non-climate factors (land tenure, sedentarisation, changes in traditional institutions, invasive species, lack of markets, and conflicts). {5.2.2}

Fruit and vegetable production, a key component of healthy diets, is also vulnerable to climate change *(medium evidence, high agreement)*. Declines in yields and crop suitability are projected under higher temperatures, especially in tropical and semi-tropical regions. Heat stress reduces fruit set and speeds up development of annual vegetables, resulting in yield losses, impaired product quality, and increasing food loss and waste. Longer growing seasons enable a greater number of plantings to be cultivated and can contribute to greater annual yields. However, some fruits and vegetables need a period of cold accumulation to produce a viable harvest, and warmer winters may constitute a risk. {5.2.2}

Food security and climate change have strong gender and equity dimensions (*high confidence*). Worldwide, women play a key role in food security, although regional differences exist. Climate change impacts vary among diverse social groups depending on age, ethnicity, gender, wealth, and class. Climate extremes have immediate and long-term impacts on livelihoods of poor and vulnerable communities, contributing to greater risks of food insecurity that can be a stress multiplier for internal and external migration *(medium confidence).* {5.2.6} Empowering women and rights-based approaches to decision-making can create synergies among household food security, adaptation, and mitigation. {5.6.4}

Many practices can be optimised and scaled up to advance adaptation throughout the food system *(high confidence)*. Supply-side options include increased soil organic matter and erosion control, improved cropland, livestock, and grazing land management, and genetic improvements for tolerance to heat and drought. Diversification in the food system (e.g., implementation of integrated production systems, broad-based genetic resources, and heterogeneous diets) is a key strategy to reduce risks (*medium confidence*). Demand-side adaptation, such as adoption of healthy and sustainable diets, in conjunction with reduction in food loss and waste, can contribute to adaptation through reduction in additional land area needed for food production and associated food system vulnerabilities. Indigenous and local knowledge can contribute to enhancing food system resilience *(high confidence)*. {5.3, 5.6.3 Cross-Chapter Box 6}.

Ca. 25-30% of total GHG emissions are attributable to the food system. These are from agriculture and land use, storage, transport, packaging, processing, retail, and consumption *(medium confidence)*. This estimate includes emissions of 10–12% from crop and livestock activities within the farm gate and 8-10% from land use and land use change including deforestation and peatland degradation *(high confidence)*; 5–10% is from supply chain activities *(medium confidence)*. This estimate includes GHG emissions from food loss and waste. Within the food system, during the period 2007-2016, the major sources of emissions from the supply side were agricultural production, with crop and livestock activities within the farm gate generating respectively 142 ± 43 Tg CH_4 yr^{-1} *(high confidence)* and 8.3 ± 2.3 Tg N_2O yr^{-1} *(high confidence)*, and CO_2 emissions linked to relevant land use change dynamics such as deforestation and peatland degradation, generating 4.8 ± 2.4 Gt CO_2 yr^{-1}. Using 100-year GWP values (no climate feedback) from the IPCC AR5, this implies that total GHG emissions from agriculture were 6.2 ± 1.9 Gt CO_2eq yr^{-1}, increasing to 11.0 ± 3.1 Gt CO_2eq yr^{-1} including relevant land use. Without intervention, these are likely to increase by about 30%–40% by 2050, due to increasing demand based on population and income growth and dietary change *(high confidence)*. {5.4}

Supply-side practices can contribute to climate change mitigation by reducing crop and livestock emissions, sequestering carbon in soils and biomass, and by decreasing emissions intensity within sustainable production systems *(high confidence)*. Total mitigation potential of crop and livestock activities is estimated as 1.5–4.0 GtCO_2-eq yr^{-1} by 2030 at prices ranging from 20-100 USD/tCO_2eq *(high confidence)*. Options with large potential for GHG mitigation in cropping systems include soil carbon sequestration (at decreasing rates over time), reductions in N_2O emissions from fertilisers, reductions in CH_4 emissions from paddy rice, and bridging of yield gaps. Options with large potential for mitigation in livestock systems include better grazing land management, with increased net primary production and soil carbon stocks, improved manure management, and higher-quality feed. Reductions in GHG emissions intensity (emissions per unit product) from livestock can support reductions in absolute emissions, provided appropriate governance to limit total production is implemented at the same time (*medium confidence*). {5.5.1}

Consumption of healthy and sustainable diets presents major opportunities for reducing GHG emissions from food systems and improving health outcomes *(high confidence)*. Examples of healthy and sustainable diets are high in coarse grains, pulses, fruits and vegetables, and nuts and seeds; low in energy-intensive animal-sourced and discretionary foods (such as sugary beverages); and with a carbohydrate threshold. Total mitigation potential of dietary changes is estimated as 1.8-3.4 GtCO_2eq yr^{-1} by 2050 at prices ranging from 20-100 USD/tCO_2 *(medium confidence)*. This estimate includes reductions in emissions from livestock and soil carbon sequestration on spared land, but co-benefits with health are not taken into account. Mitigation potential of dietary change may be higher, but achievement of this potential at broad scales depends on consumer choices and dietary preferences that are guided by social, cultural, environmental, and traditional factors, as well as income growth. Meat analogues such as imitation meat (from plant products), cultured meat, and insects may help in the transition to more healthy and sustainable diets, although their carbon footprints and acceptability are uncertain. {5.5.2, 5.6.5}

Reduction of food loss and waste could lower GHG emissions and improve food security *(medium confidence)*. Combined food loss and waste amount to a third of global food production *(high confidence)*. During 2010-2016, global food loss and waste equalled 8–10% of total GHG emissions from food systems *(medium confidence)*; and cost about USD 1 trillion per year (2012 prices) *(low confidence)*. Technical options for reduction of food loss and waste include improved harvesting techniques, on-farm storage, infrastructure, and packaging. Causes of food loss (e.g., lack of refrigeration) and waste (e.g., behaviour) differ substantially in developed and developing countries, as well as across regions *(robust evidence, medium agreement)*. {5.5.2}

Agriculture and the food system are key to global climate change responses. Combining supply-side actions such as efficient production, transport, and processing with demand-side interventions such as modification of food choices, and reduction of food loss and waste, reduces GHG emissions and enhances food system resilience (*high confidence*). Such combined measures can enable the implementation of large-scale land-based adaptation and mitigation strategies without threatening food security from increased competition for land for food production and higher food prices. Without combined food system measures in farm management, supply chains, and demand, adverse effects would include increased number of malnourished people and impacts on smallholder farmers (*medium evidence, high agreement*). Just transitions are needed to address these effects. {5.5, 5.6, 5.7}

For adaptation and mitigation throughout the food system, enabling conditions need to be created through policies, markets, institutions, and governance *(high confidence)*. For adaptation, resilience to increasing extreme events can be accomplished through risk sharing and transfer mechanisms such as insurance markets and index-based weather insurance *(high confidence)*. Public health policies to improve nutrition – such as school procurement, health insurance incentives, and awareness-raising campaigns – can potentially change demand, reduce health-care costs, and contribute to lower GHG emissions *(limited evidence, high agreement)*. Without inclusion of comprehensive food system responses in broader climate change policies, the mitigation and adaptation potentials assessed in this chapter will not be realised and food security will be jeopardised (*high confidence*). {5.7}

5.1 Framing and context

The current food system (production, transport, processing, packaging, storage, retail, consumption, loss and waste) feeds the great majority of world population and supports the livelihoods of ca. 200 million people. Agriculture as an economic activity generates between 1% and 60% of national GDP in many countries, with a world average of about 4% in 2017 (World Bank 2019). Since 1961, food supply per capita has increased more than 30%, accompanied by greater use of nitrogen fertiliser (increase of about 800%) and water resources for irrigation (increase of more than 100%).

The rapid growth in agricultural productivity since the 1960s has underpinned the development of the current global food system that is both a major driver of climate change, and increasingly vulnerable to it (from production, transport, and market activities). Given the current food system, the FAO estimates that there is a need to produce about 50% more food by 2050 in order to feed the increasing world population (FAO 2018a). This would engender significant increases in GHG emissions and other environmental impacts, including loss of biodiversity. FAO (2018a) projects that by 2050 cropland area will increase 90-325 Mha, between 6-21% more than the 1,567 Mha cropland area of 2010, depending on climate change scenario and development pathway (the lowest increase arises from reduced food loss and waste and adoption of more sustainable diets).

Climate change has direct impacts on food systems, food security, and, through the need to mitigate, potentially increases the competition for resources needed for agriculture. Responding to climate change through deployment of land-based technologies for negative emissions based on biomass production would increasingly put pressure on food production and food security through potential competition for land.

Using a food system approach, this chapter addresses how climate change affects food security, including nutrition, the options for the food system to adapt and mitigate, synergies and trade-offs among these options, and enabling conditions for their adoption. The chapter assesses the role of incremental and transformational adaptation, and the potential for combinations of supply-side measures such as sustainable intensification (increasing productivity per hectare) and demand-side measures (e.g., dietary change and waste reduction) to contribute to climate change mitigation.

5.1.1 Food security and insecurity, the food system, and climate change

The *food system* encompasses all the activities and actors in the production, transport, manufacturing, retailing, consumption, and waste of food, and their impacts on nutrition, health and well-being, and the environment (Figure 5.1).

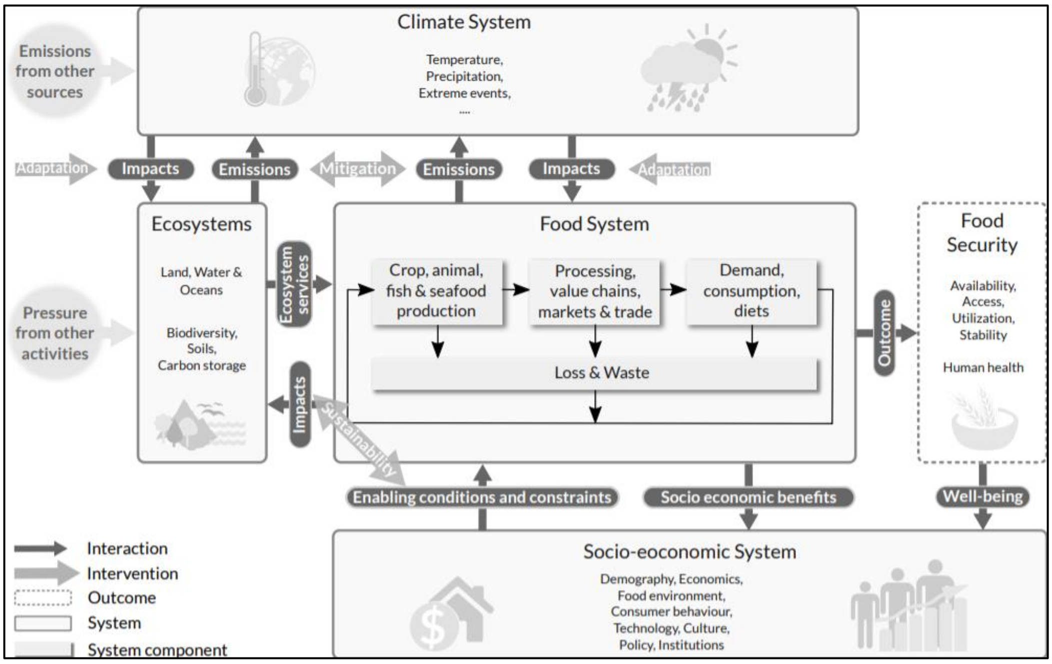


Figure 5.1 Interlinkages between the climate system, food system, ecosystem (land, water and oceans), and socio-economic system. These systems operate at multiple scales, both global and regional. Food security is an outcome of the food system leading to human well-being, which is also indirectly linked with climate and ecosystems through the socio-economic system. Response options for sustainable (S) practices, mainly in terms of climate change mitigation (M) and adaptation (A) are represented by grey arrows. Adapation measures can help to reduce negative impacts of climate change on the food system and ecosystems. Mitigation measures can reduce greenhouse gas emissions coming from the food system and ecosystems.

5.1.1.1 Food security as an outcome of the food system

The activities and the actors in the food system leads to outcomes such as food security and generate impacts on the environment. As part of the environmental impacts, food systems are a considerable contributor to greenhouse gas emissions, and thus climate change (Section 5.4). In turn climate change has complex interactions with food systems, leading to food insecurity through impacts on food availability, access, utilisation and stability (Table 5.1; Section 5.2).

We take a *food systems lens* in the Special Report on Climate Change and Land (SRCCL) to recognise that demand for and supply of food are interlinked and need to be jointly assessed in order to identify the challenges of mitigation and adaptation to climate change. Outcomes cannot be disaggregated solely to, for example, agricultural production, because the demand for food shapes what is grown, where it is grown, and how much is grown. Thus, greenhouse gas emissions from agriculture result, in large part, from ‘pull’ from the demand side. Mitigation and adaptation involve modifying production, supply chain, and demand practices (through for example dietary choices, market incentives, and trade relationships), so as to evolve a more sustainable and healthy food system.

According to FAO (2001a), *food security* is a situation that exists when all people, at all times, have physical, social, and economic access to sufficient, safe, and nutritious food that meets their dietary needs and food preferences for an active and healthy life. “All people at all times” implies the need for equitable and stable food distribution, but it is increasingly recognised that it also covers the need for inter-generational equity, and therefore “sustainability” in food production. “Safe and nutritious food …for a healthy life” implies that food insecurity can occur if the diet is not nutritious, including when there is consumption of an excess of calories, or if food is not safe, meaning free from harmful substances.

A prime impact of food insecurity is *malnourishment* (literally "bad nourishment") leading to *malnutrition*, which refers to deficiencies, excesses, or imbalances in a person's intake of energy and/or nutrients. As defined by FAO et al. (2018), undernourishment occurs when an individual's habitual food consumption is insufficient to provide the amount of dietary energy required to maintain a normal, active, healthy life. In addition to undernourishment in the sense of insufficient calories ("hunger"), undernourishment occurs in terms of nutritional deficiencies in vitamins (e.g., Vitamin A) and minerals (e.g., iron, zinc, iodine), so-called "hidden hunger". Hidden hunger tends to be present in countries with high levels of undernourishment (Muthayya et al. 2013), but micronutrient deficiency can occur in societies with low prevalence of undernourishment. For example, in many parts of the world teenage girls suffer from iron deficiency (Whitfield et al. 2015) and calcium deficiency is common in Western-style diets (Aslam and Varani 2016). Food security is related to nutrition, and conversely food insecurity is related to malnutrition. Not all malnourishment arises from food insecurity, as households may have access to healthy diets but choose to eat unhealthily, or it may arise from illness. However, in many parts of the world, poverty is linked to poor diets (FAO et al. 2018). This may be through lack of resources to produce or access food in general, or healthy food, in particular, as healthier diets are more expensive than diets rich in calories but poor in nutrition (*high confidence)* (see meta-analysis by Darmon and Drewnowski 2015). The relationship between poverty and poor diets may also be linked to unhealthy "food environments," with retail outlets in a locality only providing access to foods of low-nutritional quality (Gamba et al. 2015) – such areas are sometimes termed "food deserts" (Battersby 2012).

Whilst conceptually the definition of food security is clear, it is not straightforward to measure in a simple way that encompasses all its aspects. Although there are a range of methods to assess food insecurity, they all have some shortcomings. For example, the UN FAO has developed the Food Insecurity Experience Scale (FIES), a survey-based tool to measure the severity of overall households' inability to access food. While it provides reliable estimates of the prevalence of food insecurity in a population, it does not reveal whether actual diets are adequate or not with respect to all aspects of nutrition (see Section 5.1.2.1).

5.1.1.2 Effects of climate change on food security

Climate change is projected to negatively impact the four pillars of food security – availability, access, utilisation and stability – and their interactions (FAO et al. 2018) *(high confidence)*. This chapter assesses recent work since AR5 that has strengthened understanding of how climate change affects each of these pillars across the full range of food system activities (Table 5.1, Section 5.2).

While most studies continue to focus on availability via impacts on food production, more studies are addressing related issues of access (e.g., impacts on food prices), utilisation (e.g., impacts on nutritional quality), and stability (e.g., impacts of increasing extreme events) as they are affected by a changing climate (Bailey et al. 2015). Low-income producers and consumers are likely to be most affected because of a lack of resources to invest in adaptation and diversification measures (UNCCD 2017; Bailey et al. 2015).

Table 5.1 Relationships between food security, the food system, and climate change and guide to chapter.

Food security pillar	**Examples of observed and projected climate change impacts**	**Sections**	**Examples of adaptation and mitigation**	**Section**
Availability *Production*	Reduced yields in crop and livestock systems	5.2.2.1, 5.2.2.2	Development of adaptation practices	5.3

of food and its readiness for use through storage, processing, distribution, sale and/or exchange	Reduced yields from lack of pollinators; pests and diseases	5.2.2.3, 5.2.2.4	Adoption of new technologies, new and neglected varieties	5.3.2.3, 5.3.3.1,
	Reduced food quality affecting availability (e.g., food spoilage and loss from mycotoxins)	5.2.4.1, 5.5.2.5	Enhanced resilience by integrated practices, better food storage	5.3.2.3, 5.3.3.4, 5.6.4
	Disruptions to food storage and transport networks from change in climate, including extremes	5.2.5.1, 5.3.3.4, 5.8.1, Box 5.5	Reduction of demand on by reducing waste, modifying diets	5.3.4, 5.5.2, 5.7
			Closing of crop yield and livestock productivity gaps	5.6.4.4, 5.7
			Risk management, including marketing mechanisms, financial insurance	5.3.2, 5.7
Access: *Ability to obtain food, including effects of price*	Yield reductions, changes in farmer livelihoods, limitations on ability to purchase food	5.2.2.1, 5.2.2.2	Integrated agricultural practices to build resilient livelihoods	5.6.4
	Price rise and spike effects on low-income consumers, in particular women and children, due to lack of resources to purchase food	5.1.3, 5.2.3.1, 5.2.5.1, Box 5.1	Increased supply chain efficiency (e.g., reducing loss and waste)	5.3.3, 5.3.4
	Effects of increased extreme events on food supplies, disruption of agricultural trade and transportation infrastructure	5.8.1	More climate-resilient food systems, shortened supply chains, dietary change, market change	5.7
Utilisation *Achievement of food potential through nutrition, cooking, health*	Impacts on food safety due to increased prevalence of microorganisms and toxins	5.2.4.1	Improved storage and cold chains	5.3.3, 5.3.4
	Decline in nutritional quality resulting from increasing atmospheric CO_2	5.2.4.2	Adaptive crop and livestock varieties, healthy diets, better sanitation	5.3.4, 5.5.2, 5.7
	Increased exposure to diarrheal and other infectious diseases due to increased risk of flooding	5.2.4.1		
Stability *Continuous availability and access to food without disruption*	Greater instability of supply due to increased frequency and severity of extreme events; food price rises and spikes; instability of agricultural incomes	5.2.5, 5.8.1	Resilience via integrated systems and practices, diversified local agriculture, infrastructure investments, modifying markets and trade, reducing food loss and waste	5.6.4, 5.7, 5.8.1
	Widespread crop failure contributing to migration and conflict	5.8.2	Crop insurance for farmers to cope with extreme events	5.3.2.2, 5.7
			Capacity building to develop resilient systems	5.3.6, 5.7.4

Combined *Systemic impacts from interactions of all four pillars*	Increasing undernourishment as food system is impacted by climate change	5.1	Increased food system productivity and efficiency (e.g., supply side mitigation, reducing waste, dietary change)	5.5.1, 5.7
	Increasing obesity and ill health through narrow focus on adapting limited number commodity crops	5.1	Increased production of healthy food and reduced consumption of energy-intensive products	5.5.2, 5.7
	Increasing environmental degradation and GHG emissions	Cross-Chapter Box 6	Development of climate smart food systems by reducing GHG emissions, building resilience, adapting to climate change	5.3.3, 5.7
	Increasing food insecurity due to competition for land and natural resources (e.g., for land-based mitigation)	5.6.1	Governance and institutional responses (including food aid) that take into consideration gender and equity	5.2.5, 5.7

5.1.2 Status of the food system, food insecurity, and malnourishment

5.1.2.1 Trends in the global food system

Food is predominantly produced on land, with, on average, 83% of the 697 kg of food consumed per person per year, 93% of the 2884 kcal per day, and 80% of the 81 g of protein eaten per day coming from terrestrial production in 2013 (FAOSTAT 2018)[1]. With increases in crop yields and production (Figure 5.2), the absolute supply of food has been increasing over the last five decades. Growth in production of animal-sourced food is driving crop utilisation for livestock feed (FAOSTAT 2018; Pradhan et al. 2013a). Global trade of crop and animal-sourced food has increased by around 5 times between 1961 and 2013 (FAOSTAT 2018). During this period, global food availability has increased from 2200 kcal/cap/day to 2884 kcal/cap/day, making a transition from a food deficit to a food surplus situation (FAOSTAT 2018; Hiç et al. 2016).

The availability of cereals, animal products, oil crops, and fruits and vegetables has mainly grown (FAOSTAT 2018), reflecting shifts towards more affluent diets. This, in general, has resulted in a decrease in prevalence of underweight and an increase in prevalence of overweight and obesity among adults (Abarca-Gómez et al. 2017). During the period 1961-2016, anthropogenic greenhouse gas emissions associated with agricultural production has grown from 3.1 Gt CO_2-eq yr^{-1} to 5.8 Gt CO_2-eq yr^{-1} (Section 5.4.2, Chapter 2). The increase in emissions is mainly from the livestock sector (from enteric fermentation and manure left on pasture), use of synthetic fertiliser, and rice cultivation (FAOSTAT 2018).

[1] FOOTNOTE: Does not take into account terrestrial production of feed.

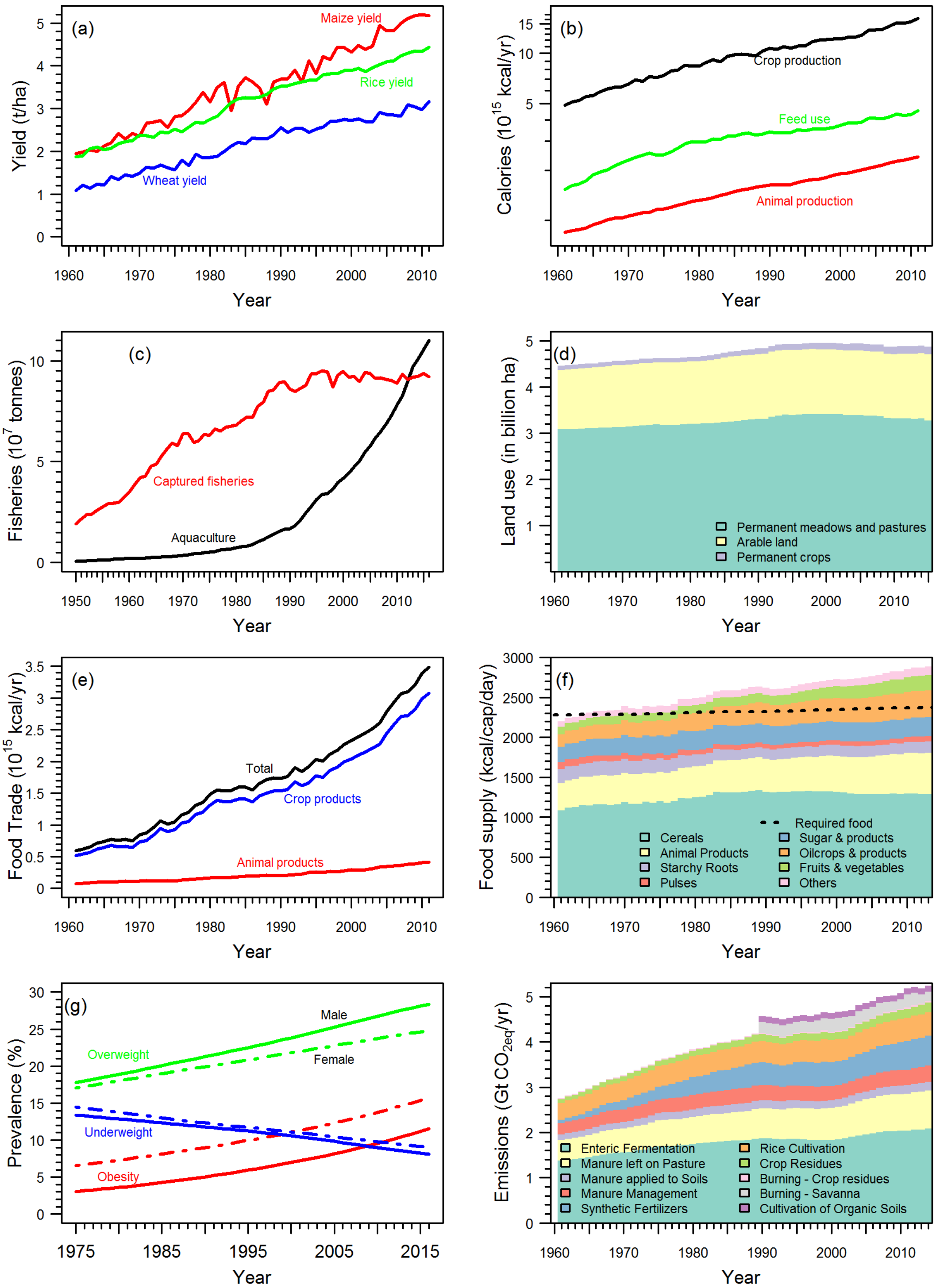


Figure 5.2 Global trends in (a) yields of maize, rice, and wheat (FAOSTAT 2018) – the top three crops grown in the world; (b) production of crop and animal calories and use of crop calories as livestock feed (FAOSTAT 2018); (c) production from marine and aquaculture fisheries (FishStat 2019); (d) land used for agriculture (FAOSTAT 2018); (e) food trade in calories (FAOSTAT 2018); (f) food supply and required food (i.e., based on human energy requirements for medium physical activities) from 1961–2012 (FAOSTAT 2018; Hiç et al. 2016); (g) prevalence of overweight, obesity and underweight from 1975–2015 (Abarca-Gómez et al. 2017); and (h) GHG emissions for the agriculture sector, excluding land use change (FAOSTAT 2018). For figures (b) and (e), data provided in mass units were converted into calories using nutritive factors (FAO 2001b). Data on emissions due to burning of savanna and cultivation of organic soils is provided only after 1990 (FAOSTAT 2018).

5.1.2.2 Food insecurity status and trends

In addressing food security the dual aspects of malnutrition – under-nutrition and micro-nutrient deficiency, as well as over-consumption, overweight, and obesity – need to be considered (Figure 5.2g and Table 5.2). The UN agencies' *State of Food Security and Nutrition* 2018 report (FAO et al. 2018) and the Global Nutrition Report 2017 (Development Initiatives 2017) summarise the global data. The *State of Food Security* report's estimate for undernourished people on a global basis is 821 million, up from 815 million the previous year and 784 million the year before that. Previous to 2014/2015 the prevalence of hunger had been declining over the last three decades. The proportion of young children (under 5) who are stunted (low height-for-age), has been gradually declining, and was 22% in 2017 compared to 31% in 2012 (150.8 million, down from 165.2 million in 2012). In 2017, 50.5 million children (7.5%) under 5 were wasted (low weight for height). Since 2014, undernutrition has worsened, particularly in parts of sub-Saharan Africa, South-Eastern Asia and Western Asia, and recently Latin America. Deteriorations have been observed most notably in situations of conflict and conflict combined with droughts or floods (FAO et al. 2018).

Regarding micronutrient deficiencies known as 'hidden hunger', reporting suggests a prevalence of one in three people globally (FAO 2013a; von Grebmer et al. 2014; Tulchinsky 2010) (Table 5.2). In the last decades, hidden hunger (measured through proxies targeting iron, vitamin A, and zinc deficiencies) worsened in Africa, while it mainly improved in Asia and Pacific (Ruel-Bergeron et al. 2015). In 2016, 613 million women and girls aged 15 to 49 suffered from iron deficiency (Development Initiatives 2018); in 2013, 28.5% of the global population suffered from iodine deficiency; and in 2005, 33.3% of children under five and 15.3% of pregnant women suffered from vitamin A deficiency, and 17.3% of the global population suffered from zinc deficiency (HLPE 2017).

Table 5.2 Global prevalence of various forms of malnutrition

	HLPE 2017 (UN)	SOFI 2017 (FAO)	GNR 2017	SOFI 2018 (FAO)	GNR2018
Overweight but not obese[a]	1.3 billion		1.93 billion		1.34 billion (38,9%)[c]
Overweight under five	41 million	41 million	41 million	38 million	38 million
Obesity[b]	600 million	600 million (13%)	641 million	672 million	678 million (13,1%)[c]
Undernourishment	800 million	815 million	815 million	821 million	
Stunting under five	155 million	155 million	155 million[d]	151 million	151 million[d] (22%)
Wasting under five	52 million	52 million (8%)	52 million[d]	50 million	51 million[d] (7%)
MND (iron)	19.2% of pregnant women[e]	33% women of reproductive age	613 million women and girls aged 15 to 49[f]	613 million (32.8%) women and girls aged 15 to 49[f]	613 million (32.8%) women and girls aged 15 to 49[f]

HLPE: High Level Panel of Experts of the committee of world food security; *SOFI*: The State of Food Security and Nutrition in the World; *GNR*: Global Nutrition Report; *MND*: Micro nutrient deficiency (Iron deficiency for year 2016, uses anemia as a proxy (percentage of pregnant women whose haemoglobin level is less than 110 grams per litre at sea level and percentage of non-pregnant women whose haemoglobin level is less than 120 grams per litre at sea level).

[a]Body mass index between 25-29.9 kg/m^2

[b]Body mass index greater than 30 kg/m^2

[c]Prevalence of overweight/obesity among adults (age ≥18) in year 2016. Data from NCD Risc data source.

[d]UNICEF WHO Joint Malnutrition;

[e]In 2011

[f]Anaemia prevalence in girls and women aged 15 to 49

Globally, as the availability of inexpensive calories from commodity crops increases, so does per capita consumption of calorie-dense foods (Ng et al. 2014; NCD-RisC 2016a; Abarca-Gómez et al. 2017; Doak and Popkin 2017). As a result, in every region of the world, the prevalence of obesity (body mass index >30 kg/m^2) and overweight (body mass index range between normality [18.5-24.9] and obesity) is increasing. There are now more obese adults in the world than underweight adults (Ng et al. 2014; NCD-RisC 2016a; Abarca-Gómez et al. 2017; Doak and Popkin 2017). In 2016, around two billion adults were overweight, including 678 million suffering from obesity (NCD-RisC 2016a; Abarca-Gómez et al. 2017). The prevalence of overweight and obesity has been observed in all age groups.

Around 41 million children under five years and 340 million children and adolescents aged 5–19 years were suffering from overweight or obesity in 2016 (NCD-RisC 2016a; FAO et al. 2017; WHO 2015). In many high-income countries, the rising trends in children and adolescents suffering from overweight and obesity have stagnated at high levels; however, these have accelerated in parts of Asia and have very slightly reduced in European and Central Asian lower and middle-income countries (Abarca-Gómez et al. 2017; Doak and Popkin 2017; Christmann et al. 2009).

There are associations between obesity and non-communicable diseases such as diabetes, dementia, inflammatory diseases (Saltiel and Olefsky 2017), cardio-vascular disease (Ortega et al. 2016) and some cancers, e.g., of the colon, kidney, and liver (Moley and Colditz 2016). There is a growing recognition of the rapid rise in overweight and obesity on a global basis and its associated health burden created through the non-communicable diseases (NCD-RisC 2016a; HLPE 2017).

Analyses reported in FAO et al. (2018) highlight the link between food insecurity, as measured by the FIES scale, and malnourishment (*medium agreement, robust evidence*). This varies by malnourishment measure as well as country (FAO et al. 2018). For example, there is *limited evidence* (*low agreement* but multiple studies) that food insecurity and childhood wasting (i.e., or low weight for height) are closely related, but it is very likely (*high agreement, robust evidence*) that childhood stunting and food insecurity are related (FAO et al. 2018). With respect to adult obesity there is *robust evidence, with medium agreement*, that food insecurity, arising from poverty reducing access to nutritious diets, is related to the prevalence of obesity, especially in high-income countries and adult females. An additional meta-analysis (for studies in Europe and North America) also finds a negative relationship between income and obesity, with some support for an effect of obesity causing low income (as well as vice versa) (Kim and von dem Knesebeck 2018).

As discussed in Section 5.1.1.1, different methods of assessing food insecurity can provide differential pictures. Of particular note is the spatial distribution of food insecurity, especially in higher-income countries. FAO et al. (2018) reports FIES estimates of severe food insecurity in Africa, Asia and Latin America of 29.8%, 6.9% and 9.8% of the population, respectivity, but of 1.4% of the population (i.e., about 20 million in total; pro rata <5 million for US, <1 million for UK) in Europe and North America. However, in the United States, USDA estimates 40 million people were exposed to varying degrees of food insecurity, from mild to severe (overall prevalence about 12%) (Coleman-Jensen et al. 2018). In the UK, estimates from 2017 and 2018 indicate about 4 million adults are moderately to severely food insecure (prevalence 8%) (End Hunger UK 2018; Bates et al. 2017). The UK food bank charity, the Trussell Trust, over a year in 2017/18, distributed 1,332,952 three-day emergency food parcels to people referred to the charity as being in food crisis. Furthermore, a 2003 study in the UK (Schenker 2003) estimated that 40% of adults, and 15% of children, admitted to hospitals were malnourished, and that 70% of undernourishment in the UK was unreported.

In total, more than half the world's population are underweight or overweight (NCD-RisC 2017a), so their diets do not provide the conditions for 'an active and healthy life'. This will be more compromised under the impacts of climate change by changing the availability, access, utilisation, and stability of diets of sufficient nutritional quality as shown in Table 5.2 and discussed in detail below (see Section 5.2).

5.1.3 Climate change, gender, and equity

Throughout, the chapter considers many dimensions of gender and equity in regard to climate change and the food system (Box 5.1). Climate change impacts differ among diverse social groups depending on factors such as age, ethnicity, ability/disability, sexual orientation, gender, wealth, and class (*high confidence*) (Vincent and Cull 2014; Kaijser and Kronsell 2014). Poverty, along with socio-economic and political marginalisation, cumulatively put women, children and the elderly in a disadvantaged position in coping with the adverse impacts of the changing climate (UNDP 2013; Skoufias et al. 2011). The contextual vulnerability of women is higher due to their differentiated relative power, roles, and responsibilities at the household and community levels (Bryan and Behrman 2013; Nelson et al. 2002). They often have a higher reliance on subsistence agriculture, which will be severely impacted by climate change (Aipira et al. 2017).

Through impacts on food prices (section 5.2.3.1) poor people's food security is particularly threatened. Decreased yields can impact nutrient intake of the poor by decreasing supplies of highly nutritious crops and by promoting adaptive behaviours that may substitute crops that are resilient but less nutritious (Thompson et al. 2012; Lobell and Burke 2010). In Guatemala, food prices and poverty have been correlated with lower micronutrient intakes (Iannotti et al. 2012). In the developed world, poverty is more typically associated with calorically-dense but nutrient-poor diets, obesity, overweight, and other related diseases (Darmon and Drewnowski 2015).

Rural areas are especially affected by climate change (Dasgupta et al. 2014), through impacts on agriculture-related livelihoods and rural income (Mendelsohn et al. 2007) and through impacts on employment. Jessoe et al. (2018) using a 28-year panel on individual employment in rural Mexico, found that years with a high occurrence of heat lead to a reduction in local employment by up to 1.4% with a medium emissions scenario, particularly for wage work and non-farm labour, with impacts on food access. Without employment opportunities in areas where extreme poverty is prevalent, people may be forced to migrate, exacerbating potential for ensuing conflicts (FAO 2018a).

Finally, climate change can affect human health in other ways that interact with food utilisation. In many parts of the world where agriculture relies still on manual labour, projections are that heat stress will reduce the hours people can work, and increase their risk (Dunne et al. 2013). For example, Takakura et al (2017) estimates that under RCP8.5, the global economic loss from people working shorter hours to mitigate heat loss may be 2.4–4% of GDP. Furthermore, as discussed by (Watts et al. 2018); people's nutritional status interacts with other stressors and affects their susceptibility to ill health (the "utilisation pillar" of food security): so food-insecure people are more likely to be adversely affected by extreme heat, for example.

In the case of food price hikes, those more vulnerable are more affected (Uraguchi 2010), especially in urban areas (Ruel et al. 2010), where livelihood impacts are particularly severe for the individuals and groups that have scarce resources or are socially isolated (Revi et al. 2014; Gasper et al. 2011) (*high confidence*). These people often lack power and access to resources, adequate urban services and functioning infrastructure. As climate events become more frequent and intense, this can increase the scale and depth of urban poverty (Rosenzweig et al. 2018b). Urban floods and droughts may result in water contamination increasing the incidence of diarrhoeal illness in poor children (Bartlett 2008). In the near destruction of New Orleans by Hurricane Katrina, about 40,000 jobs were lost (Rosemberg 2010).

Box 5.1 Gender, food security, and climate change

Differentiated impacts, vulnerability, risk perception, behaviours and coping strategies for climate change related to food security derive from cultural (gendered) norms, that is, the behaviours, tasks, and responsibilities a society defines as "male" or "female", and the differential gendered access to resources (Paris and Rola-Rubzen 2018; Aberman and Tirado 2014; Lebel et al. 2014; Bee 2016). In many rural areas women often grow most of the crops for domestic consumption and are primarily responsible for storing, processing, and preparing food; handling livestock; gathering food, fodder and fuelwood; managing domestic water supply; and providing most of the labour for post-harvest activities (FAO 2011a). They are mostly impacted through increased hardship, implications for household roles, and subsequent organisational responsibilities (Boetto and McKinnon 2013; Jost et al. 2016). Water scarcity can particularly affect women because they need to spend more time and energy to collect water, where they may be more exposed to physical and sexual violence (Sommer et al. 2015; Aipira et al. 2017). They may be forced to use unsafe water in the household increasing risk of water-borne diseases (Parikh 2009). Climate change also has differentiated gendered impacts on livestock-holders food security (McKune et al. 2015; Ongoro and Ogara 2012; Fratkin et al. 2004) (See Supplementary Material Table SM5.1).

Gender dimensions of the four pillars

Worldwide, women play a key role in food security (World Bank 2015) and the four pillars of food security have strong gender dimensions (Thompson 2018). In terms of food availability, women tend to have less access to productive resources, including land, and thus less capacity to produce food (Cross-chapter box 11: Gender in Chapter 7).

In terms of food access, gendered norms in how food is divided at mealtimes may lead to smaller food portions for women and girls. Women's intra-household inequity limits their ability to purchase food; limitations also include lack of women's mobility impacting trips to the market and lack of decision-making within the household (Ongoro and Ogara 2012; Mason et al. 2017; Riley and Dodson 2014).

In terms of food utilisation, men, women, children and the elderly have different nutritional needs (e.g., during pregnancy or breast-feeding).

In terms of stability, women are more likely to be disproportionately affected by price spikes (Vellakkal et al. 2015; Arndt et al. 2016; Hossain and Green 2011; Darnton-Hill and Cogill 2010; Cohen and Garrett 2010; Kumar and Quisumbing 2013) because when food is scarce women reduce food consumption relative to other family members, although these norms vary according to age, ethnicity, culture, region, and social position, as well as by location in rural or urban areas (Arora-Jonsson 2011; Goh 2012; Niehof 2016; Ongoro and Ogara 2012).

Integrating gender into adaptation

Women have their own capabilities to adapt to climate change. In the Pacific Islands, women hold critical knowledge on where or how to find clean water; which crops to grow in a flood or a drought season; how to preserve and store food and seeds ahead of approaching storms, floods or droughts;

and how to carry their families through the recovery months. They also play a pivotal role in managing household finances and investing their savings in education, health, livelihoods, and other activities that assist their families to adapt and respond to climate effects (Aipira et al. 2017). Decreasing women's capacity to adapt to the impacts of climate change also decreases that of the household (Bryan and Behrman 2013).

However, gender norms and power inequalities also shape the ability of men, women, boys, girls and the elderly to adapt to climate risks (Rossi and Lambrou 2008). For example, women pastoralists in the Samburu district of Kenya cannot make decisions affecting their lives, limiting their adaptive capacity (Ongoro and Ogara 2012).

Participation in decision-making and politics, division of labour, resource access and control, and knowledge and skills (Nelson and Stathers 2009) are some of the barriers to adaptation. Women's adaptive capacity is also diminished because their work often goes unrecognised (Rao 2005; Nelson and Stathers 2009). Many of women's activities are not defined as "economically active employment" in national accounts (FAO 2011a). This non-economic status of women's activities implies that they are not included in wider discussions of priorities or interventions for climate change. Their perspectives and needs are not met; and thus, interventions, information, technologies, and tools promoted are potentially not relevant, and even can increase discrimination (Alston 2009; Edvardsson Björnberg and Hansson 2013; Huynh and Resurreccion 2014).

Where gender-sensitive policies to climate change may exist, effective implementation in practice of gender equality and empowerment may not be achieved on the ground due to lack of technical capacity, financial resources and evaluation criteria, as shown in the Pacific Islands (Aipira et al. 2017). Thus, corresponding institutional frameworks that are well-resourced, coordinated, and informed are required, along with adequate technical capacity within government agencies, NGOs and project teams, to strength collaboration and promote knowledge sharing (Aipira et al. 2017).

Women's empowerment: Synergies among adaptation, mitigation, and food security

Empowered and valued women in their societies increases their capacity to improve food security under climate change, make substantial contributions to their own well-being, to that of their families and of their communities (Langer et al. 2015; Ajani et al. 2013; Alston 2014) (*high confidence*). Women's empowerment includes economic, social and institutional arrangements and may include targeting men in integrated agriculture programs to change gender norms and improve nutrition (Kerr et al. 2016). Empowerment through collective action and groups-based approaches in the near-term has the potential to equalise relationships on the local, national and global scale (Ringler et al. 2014). Empowered women are crucial to creating effective synergies among adaptation, mitigation, and food security.

In Western Kenya, widows in their new role as main livelihood providers invested in sustainable innovations like rainwater harvesting systems and agroforestry (this can serve as both adaptation and mitigation), and worked together in formalised groups of collective action (Gabrielsson and Ramasar 2013) to ensure food and water security. In Nepal, women's empowerment had beneficial outcomes in maternal and children nutrition, reducing the negative effect of low production diversity (Malapit et al. 2015). Integrated nutrition and agricultural programs have increased women's decision-making power and control over home gardens in Burkina Faso (van den Bold et al. 2015) with positive impacts on food security.

5.1.4 Food systems in AR5, SR15, and the Paris Agreement

Food, and its relationship to the environment and climate change, has grown in prominence since the Rio Declaration in 1992, where food production is Chapter 14 of Agenda 21, to the Paris Agreement of 2015, which includes the need to ensure food security under the threat of climate change on its first page. This growing prominence of food is reflected in recent IPCC reports, including its Fifth Assessment Report (IPCC 2014a) and the Special Report on Global Warming of 1.5°C (SR15) (IPCC 2018a).

5.1.4.1 Food systems in AR5 and SR15

The IPCC Working Group (WG) II AR5 chapter on Food Security and Food Production Systems broke new ground by expanding its focus beyond the effects of climate change primarily on agricultural production (crops, livestock and aquaculture) to include a food systems approach as well as directing attention to undernourished people (Porter et al. 2014). However, it focused primarily on food production systems due to the prevalence of studies on that topic (Porter et al. 2017). It highlighted that a range of potential adaptation options exist across all food system activities, not just in food production, and that benefits from potential innovations in food processing, packaging, transport, storage, and trade were insufficiently researched at that time.

The IPCC WG III AR5 chapter on Agriculture, Forestry and Other Land Use (AFOLU) assessed mitigation potential considering not only the supply, but also the demand side of land uses, by consideration of changes in diets; it also included food loss and waste (Smith et al. 2014). AR5 focused on crop and livestock activities within the farm gate and land use and land use change dynamics associated with agriculture. It did not take a full food system approach to emissions estimates that includes processing, transport, storage, and retail.

The IPCC WG II AR5 Rural Areas chapter (Revi et al. 2014) found that farm households in developing countries are vulnerable to climate change due to socio-economic characteristics and non-climate stressors, as well as climate risks (Dasgupta et al. 2014). They also found that a wide range of on-farm and off-farm climate change adaptation measures are already being implemented and that the local social and cultural context played a prominent role in the success or failure of different adaptation strategies for food security, such as trade, irrigation or diversification. The IPCC WG II AR5 Urban Areas chapter found that food security of people living in cities was severely affected by climate change through reduced supplies, including urban-produced food, and impacts on infrastructure, as well as a lack of access to food. Poor urban dwellers are more vulnerable to rapid changes of food prices due to climate change.

Many climate change response options in IPCC WG II and WG III AR5 (IPCC 2014b) address incremental adaptation or mitigation responses separately rather than being inclusive of more systemic or transformational changes in multiple food systems that are large-scale, in depth, and rapid, requiring social, technological, organisational and system responses (Rosenzweig and Solecki 2018; Mapfumo et al. 2017; Termeer et al. 2017). In many cases, transformational change will require integration of resilience and mitigation across all parts of the food system including production, supply chains, social aspects, and dietary choices. Further, these transformational changes in the food system need to encompass linkages to ameliorative responses to land degradation (see Chapter 4), desertification (see Chapter 3), and declines in quality and quantity of water resources throughout the food-energy-water nexus (Chapter 2; Section 5.7).

The IPCC Special Report on Global Warming of 1.5°C found that climate-related risks to food security are projected to increase with global warming of 1.5°C and increase further with 2°C (IPCC 2018a).

5.1.4.2 Food systems and the Paris Agreement

To reach the temperature goal put forward in the Paris Agreement of limiting warming to well below 2°C, and pursuing efforts to limit warming to 1.5°C, representatives from 196 countries signed the United Nations Framework Convention on Climate Change (UNFCCC) Paris Agreement (UNFCCC 2015) in December 2015. The Agreement put forward a temperature target of limiting warming to well below 2°C, and pursuing efforts to limit warming to 1.5°C. Under the Paris Agreement, Parties are expected to put forward their best efforts through nationally determined contributions (NDCs) and to strengthen these efforts in the years ahead. Article 2 of the Agreement makes clear the agreement is within “the context of sustainable development” and states actions should be "in a manner that does not threaten food production” to ensure food security.

Many countries have included food systems in their mitigation and adaptation plans as found in their NDCs for the Paris Agreement (Rosenzweig et al. 2018a). Richards et al. (2015) analysed 160 Party submissions and found that 103 include agricultural mitigation; of the 113 Parties that include adaptation in their NDCs, almost all (102) include agriculture among their adaptation priorities. There is much attention to conventional agricultural practices that can be climate-smart and sustainable (e.g., crop and livestock management), but less to the enabling services that can facilitate uptake (e.g., climate information services, insurance, credit). Considerable finance is needed for agricultural adaptation and mitigation by least developed countries – in the order of USD 3 billion annually for adaptation and USD 2 billion annually for mitigation, which may be an underestimate due to a small sample size (Richards et al. 2015). On the mitigation side, none of the largest agricultural emitters included sector-specific contributions from the agriculture sector in their NDCs, but most included agriculture in their economy-wide targets (Richards et al. 2018).

Carbon dioxide removal (CDR). A key aspect regarding the implementation of measures to achieve the Paris Agreement goals involves measures related to carbon dioxide removal (CDR) through bioenergy (Sections 5.5 and 5.6). To reach the temperature target put forward of limiting warming to well below 2°C, and pursuing efforts to limit warming to 1.5°C, large investments and abrupt changes in land use will be required to advance bioenergy with carbon capture and sequestration (BECCS), afforestation and reforestation (AR), and biochar technologies. Existing scenarios estimate the global area required for BECCS alone to help limit warming to 1.5°C in the range of 109-990 Mha, most commonly around 380-700 Mha.

Most scenarios assume very rapid deployment between 2030 and 2050, reaching rates of expansion in land use in 1.5°C scenarios exceeding 20 M ha yr^{-1}, which are unprecedented for crops and forestry reported in the FAO database from 1961. Achieving the 1.5 °C target would thus result in major competing demands for land between climate change mitigation and food production, with cascading impacts on food security.

This chapter assesses how the potential conflict for land could be alleviated by sustainable intensification to produce food with a lower land footprint (Section 5.6, Cross-Chapter Box 6: Agricultural intensification). To accomplish this, farmers would need to produce the same amount of food with lower land requirement, which depends on technology, skills, finance, and markets. Achieving this would also rely on demand-side changes including dietary choices that enable reduction of the land footprint for food production while still meeting dietary needs. Transitions required for such transformative changes in food systems are addressed in Section 5.7.

5.1.4.3 Charting the future of food security

This chapter utilises the common framework of the Representative Concentration Pathways (RCPs) and the Shared Socio-economic Pathways (SSPs) (Popp et al. 2017; Riahi et al. 2017; Doelman et al. 2018) to assess the impacts of future GHG emissions, mitigation measures, and adaptation on food security (See Cross-Chapter Box 1: Scenarios in Chapter 1, Section 5.2 and 5.6).

New work utilising these scenario approaches has shown that the food system externalises costs onto human health and the environment (Springmann et al. 2018a; Swinburn et al. 2019; Willett et al. 2019), leading to calls for transforming the food system to deliver better human and sustainability outcomes (Willett et al. 2019; IAP 2018; Development Initiatives 2018; Lozano et al. 2018). Such a transformation could be an important lever to address the complex interactions between climate change and food security. Through acting on mitigation and adaptation in regard to both food demand and food supply we assess the potential for improvements to both human health and the Sustainable Development Goals (Section 5.6).

This chapter builds on the food systems and scenario approaches followed by AR5 and its focus on climate change and food security, but new work since AR5 has extended beyond production to how climate change interacts with the whole food system. The analysis of climate change and food insecurity has expanded beyond undernutrition to include the overconsumption of unhealthy mass-produced food high in sugar and fat, which also threatens health in different but highly damaging ways and the role of dietary choices and consumption in greenhouse gas emissions. It focused on land-based food systems, though highlighting in places the contributions of freshwater and marine production.

The chapter assesses new work on the observed and projected effects of CO_2 concentrations on the nutritional quality of crops (Section 5.2.4.2) and emphasises the role of extreme climate events (Section 5.2.5.1), social aspects including gender and equity (Box 5.1. and Cross-chapter Box 11: Gender in Chapter 7), and dietary choices (Section 5.4.6, 5.5.2). Other topics with considerable new literature include impacts on smallholder farming systems (Section 5.2.2.6), food loss and waste (Section **Error! Reference source not found.**), and urban and peri-urban agriculture (Section 5.6.5). he chapter explores the potential competing demands for land that mitigation measures to achieve temperature targets may engender, with cascading impacts on food production, food security, and farming systems (Section 5.6), and the enabling conditions for achieving the mitigation and adaptation in equitable and sustainable ways (Section 5.7). Section 5.8 presents challenges to future food security, including food price spikes, migration, and conflict.

5.2 Impacts of climate change on food systems

There are many routes by which climate change can impact food security and thus human health (Watts et al. 2018; Fanzo et al. 2017). One major route is via climate change affecting the amount of food, both from direct impacts on yields (Section 5.2.2.1) and indirect effects through climate change's impacts on water availability and quality, pests and diseases (Section 5.2.2.3), and pollination services (Section 5.2.2.4). Another route is via changing CO_2 in the atmosphere, affecting biomass and nutritional quality (Section 5.2.4.2). Food safety risks during transport and storage can also be exacerbated by changing climate (Section 5.2.4.1).

Further, the direct impacts of changing weather can affect human health through the agricultural workforce's exposure to extreme temperatures (Section 5.2.5.1). Through changing metabolic demands and physiological stress for people exposed to extreme temperatures, there is also the potential for interactions with food availability: people may require more food to cope, whilst at the same time being impaired from producing it (Watts et al. 2018). All these factors have the potential to alter both physical health as well as cultural health, through changing the amount, safety and quality of food available for individuals within their cultural context.

This section assesses recent literature on climate change impacts on the four pillars of food security: availability (Section 5.2.2), access (Section 5.2.3), utilisation (Section 5.2.4), and stability (Section 5.2.5). It considers impacts on the food system from climate changes that are already taking place and how impacts are projected to occur in the future. See Supplementary Material Section SM5.2 for discussion of detection and attribution and improvement in projection methods.

5.2.1 Climate drivers important to food security

Climate drivers relevant to food security and food systems include temperature-related, precipitation-related, and integrated metrics that combine these and other variables. These are projected to affect many aspects of the food security pillars (FAO 2018b) (see Supplementary Material Table SM5.2 and Chapter 6 for assessment of observed and projected climate impacts). Climate drivers relevant to food production and availability may be categorised as modal climate changes (e.g., shifts in climate envelopes causing shifts in cropping varieties planted), seasonal changes (e.g., warming trends extending growing seasons), extreme events (e.g., high temperatures affecting critical growth periods, flooding/droughts), and atmospheric conditions (e.g., CO_2 concentrations, short-lived climate pollutants (SLCPs), and dust). Water resources for food production will be affected through changing rates of precipitation and evaporation, ground water levels, and dissolved oxygen content (Cruz-Blanco et al. 2015; Sepulcre-Canto et al. 2014; Huntington et al. 2017; Schmidtko et al. 2017). Potential changes in major modes of climate variability can also have widespread impacts such as occurred during late 2015 to early 2016 when a strong El Niño contributed to regional shifts in precipitation in the Sahel region. Significant drought across Ethiopia resulted in widespread crop failure and more than 10 million people in Ethiopia required food aid (U.S. Department of State 2016; Huntington et al. 2017) (see Figure 5.3).

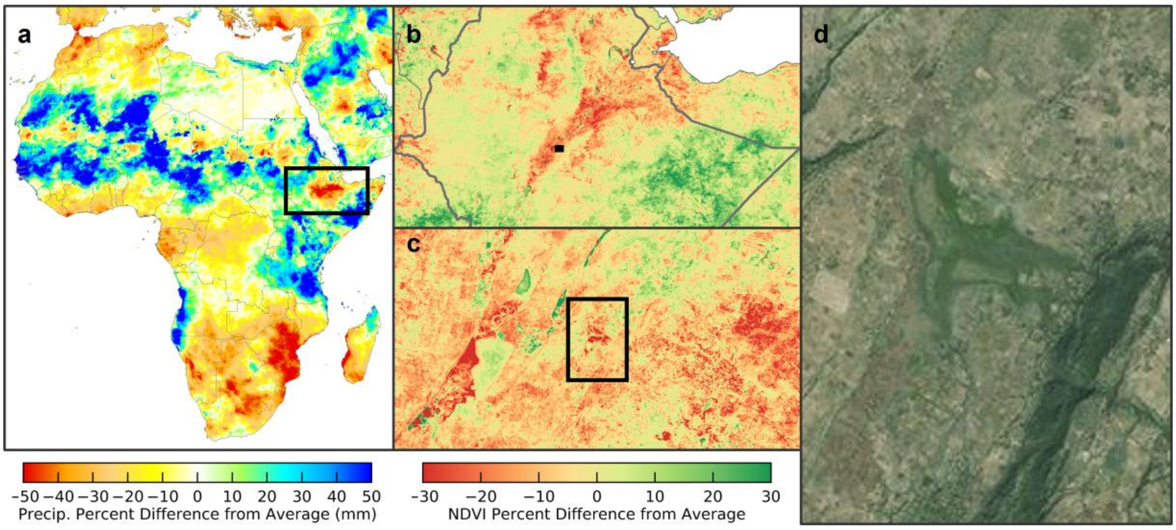


Figure 5.3 Precipitation anomaly and vegetation response in Eastern Africa. (a) Sep 2015–Feb 2016 Climate Hazards Group Infrared Precipitation with Station (CHIRPS) precipitation anomaly over Africa relative to the 1981–2010 average shows that large areas of Ethiopia received less than half of normal precipitation. Consequently, widespread impacts to agricultural productivity, especially within pastoral regions, were present across Ethiopia as evidenced by (d) reduced greenness in remote sensing images. (b) MODIS NDVI anomalies for Sep 2015–Feb 2016 relative to 2000–2015 average are shown for the inset box in (a). (c) Landsat NDVI anomalies for Sep 2015–Feb 2016 relative to 2000–2015 average are shown for the inset box in (b) (Huntington et al. 2017).

Other variables that affect agricultural production, processing, and/or transport are solar radiation, wind, humidity, and (in coastal areas) salinisation and storm surge (Mutahara et al. 2016; Myers et al. 2017). Extreme climate events resulting in inland and coastal flooding, can affect the ability of people to obtain and prepare food (Rao et al. 2016; FAO et al. 2018). For direct effects of atmospheric CO_2 concentrations on crop nutrient status see Section 5.2.4.2.

5.2.1.1 Short-lived climate pollutants

The important role of short-lived climate pollutants such as ozone and black carbon is increasingly emphasised since they affect agricultural production through direct effects on crops and indirect effects on climate (Emberson et al. 2018; Lal et al. 2017; Burney and Ramanathan 2014; Ghude et al. 2014) (see Chapters 2 and 4). Ozone causes damage to plants through damages to cellular metabolism that influence leaf-level physiology to whole-canopy and root-system processes and feedbacks; these impacts affect leaf-level photosynthesis senescence and carbon assimilation, as well as whole-canopy water and nutrient acquisition and ultimately crop growth and yield (Emberson et al. 2018). Using atmospheric chemistry and a global integrated assessment model, Chuwah et al. (2015) found that without a large decrease in air pollutant emissions, high ozone concentration could lead to an increase in crop damage of up to 20% in agricultural regions in 2050 compared to projections in which changes in ozone are not accounted for. Higher temperatures are associated with higher ozone concentrations; C3 crops are sensitive to ozone (e.g., soybeans, wheat, rice, oats, green beans, peppers, and some types of cottons) and C4 crops are moderately sensitive (Backlund et al. 2008).

Methane increases surface ozone which augments warming-induced losses and some quantitative analyses now include climate, long-lived (CO_2) and mulitple short-lived pollutants (CH_4, O_3) simultaneously (Shindell et al. 2017; Shindell 2016). Reduction of tropospheric ozone and black carbon can avoid premature deaths from outdoor air pollution and increases annual crop yields (Shindell et al. 2012). These actions plus methane reduction can influence climate on shorter time scales than those of carbon dioxide–reduction measures. Implementing them substantially reduces the risks of crossing the 2°C threshold and contributes to achievement of the SDGs (Haines et al. 2017; Shindell et al. 2017).

5.2.2 Climate change impacts on food availability

Climate change impacts food availability through its effect on the production of food and its storage, processing, distribution, and exchange.

5.2.2.1 Impacts on crop production

Observed impacts. Since AR5, there have been further studies that document impacts of climate change on crop production and related variables (See Supplementary Material Table SM5.3). There have been also a few studies that demonstrate a strengthening relationship between observed climate variables and crop yields that indicate future expected warming will have severe impacts on crop production (Mavromatis 2015; Innes et al. 2015). At the global scale, Iizumi et al. (2018) used a counterfactual analysis and found that climate change between 1981-2010 has decreased global mean yields of maize, wheat, and soybeans by 4.1, 1.8 and 4.5%, respectively, relative to preindustrial climate, even when CO_2 fertilisation and agronomic adjustments are considered. Uncertainties (90% probability interval) in the yield impacts are -8.5 to +.5% for maize, -7.5 to +4.3% for wheat, and -8.4 to -0.5% for soybeans. For rice, no significant impacts were detected. This study suggests that climate change has modulated recent yields on the global scale and led to production losses, and that adaptations to date have not been sufficient to offset the negative impacts of climate change, particularly at lower latitudes.

Dryland settlements are perceived as vulnerable to climate change with regard to food security, particularly in developing countries; such areas are known to have low capacities to cope effectively with decreasing crop yields (Shah et al. 2008; Nellemann et al. 2009). This is of concern because drylands constitute over 40% of the earth's land area, and are home to 2.5 billion people (FAO et al. 2011).

Australia. In Australia, declines in rainfall and rising daily maximum temperatures based on simulations of 50 sites caused water-limited yield potential to decline by 27% from 1990 to 2015, even though elevated atmospheric CO_2 concentrations had a positive effect (Hochman et al. 2017). In New South Wales, high-temperature episodes during the reproduction stage of crop growth were found to have negative effects on wheat yields, with combinations of low rainfall and high temperatures being the most detrimental (Innes et al. 2015).

Asia. There are numerous studies demonstrating that climate change is affecting agriculture and food security in Asia. Several studies with remote sensing and statistical data have examined rice areas in northeastern China, the northernmost region of rice cultivation, and found expansion over various time periods beginning in the 1980s, with most of the increase occurring after 2000 (Liu et al. 2014; Wang et al. 2014; Zhang et al. 2017). Rice yield increases have also been found over a similar period (Wang et al. 2014). Multiple factors, such as structural adjustment, scientific and technological progress, and government policies, along with regional warming (1.43°C in the past century) (Fenghua et al. 2006) have been put forward as contributing to the observed expanded rice areas and yield in the region. Shi et al. (2013) indicate that there is a partial match between climate change patterns and shifts in extent and location of the rice-cropping area (2000-2010).

There have also been documented changes in winter wheat phenology in Northwest China (He 2015). Consistent with this finding, dates of sowing and emergence of spring and winter wheat were delayed, dates of anthesis and maturity was advanced, and length of reproductive growth period was prolonged from 1981-2011 in a study looking at these crops across China (Liu et al. 2018b). Another study looking in Norwest China demonstrated that there have been changes in the phenology and productivity of spring cotton (Huang and Ji 2015). A study looking at wheat growth and yield in different climate zones of China from 1981-2009 found that impacts were positive in Northern China and negative in Southern China (Tao et al. 2014). Temperature increased across the zones while precipitation changes were not consistent (Tao et al. 2014).

Crop yield studies focusing on India have found that warming has reduced wheat yields by 5.2% from 1981 to 2009, despite adaptation (Gupta et al. 2017); that maximum daytime temperatures have risen along with some night-time temperatures (Jha and Tripathi 2017).

Agriculture in Pakistan has also been affected by climate change. From 1980 to 2014, spring maize growing periods have shifted an average of 4.6 days per decade earlier, while sowing of autumn maize has been delayed 3.0 days per decade (Abbas et al. 2017). A similar study with sunflower showed that increases in mean temperature from 1980 to 2016 were highly correlated with shifts in sowing, emergence, anthesis, and maturity for fall and spring crops (Tariq et al. 2018).

Mountain people in the Hindu-Kush Himalayan region encompassing parts of Pakistan, India, Nepal, and China, are particularly vulnerable to food insecurity related to climate change because of poor infrastructure, limited access to global markets, physical isolation, low productivity, and hazard exposure, including Glacial Lake Outburst Floods (GLOFs) (Rasul et al. 2019; Rasul 2010; Tiwari and Joshi 2012; Huddleston et al. 2003; Ward et al. 2013; FAO 2008; Nautiyal et al. 2007; Din et al. 2014). Surveys have been conducted to determine how climate-related changes have affected food security (Hussain et al. 2016; Shrestha and Nepal 2016) with results showing that the region is experiencing an increase in extremes, with farmers facing more frequent floods as well as prolonged droughts with ensuing negative impacts on agricultural yields and increases in food insecurity (Hussain et al. 2016; Manzoor et al. 2013).

South America. In another mountainous region, the Andes, inhabitants are also beginning to experience changes in the timing, severity, and patterns of the annual weather cycle. Data collected through participatory workshops, semi-structured interviews with agronomists, and qualitative fieldwork from 2012 to 2014 suggest that in Colomi, Bolivia climate change is affecting crop yields and causing farmers to alter the timing of planting, their soil management strategies, and the use and spatial distribution of crop varieties (Saxena et al. 2016). In Argentina, there has also been in increase in yield variability of maize and soybeans (Iizumi and Ramankutty 2016). These changes have had important implications for the agriculture, human health, and biodiversity of the region (Saxena et al. 2016).

Africa. In recent years, yields of staple crops such as maize, wheat, sorghum, and fruit crops, such as mangoes, have decreased across Africa, widening food insecurity gaps (Ketiem et al. 2017). In Nigeria, there have been reports of climate change having impacts on the livelihoods of arable crop farmers (Abiona et al. 2016; Ifeanyi-obi et al. 2016; Onyeneke 2018). The Sahel region of Cameroon has experienced an increasing level of malnutrition, partly due to the impact of climate change since harsh climatic conditions leading to extreme drought have a negative influence on agriculture (Chabejong 2016).

Utilising farmer interviews in Abia State, Nigeria, researchers found that virtually all responders agreed that the climate was changing in their area (Ifeanyi-obi et al. 2016). With regard to management responses, a survey of farmers from Anambra State, Nigeria showed that farmers are adapting to climate change by utilising such techniques as mixed cropping systems, crop rotation, fertiliser application (Onyeneke et al. 2018). In Ebonyi State, Nigeria, Eze (2017) interviewed 160 women cassava farmers and found the major climate change risks in production to be severity of high temperature stress, variability in relative humidity, and flood frequency.

Europe. The impacts of climate change are varied across the continent. Moore and Lobell (2015) showed that climate trends are affecting European crop yields, with long-term temperature and precipitation trends since 1989 reducing continent-wide wheat and barley yields by 2.5% and 3.8%, respectively, and having slightly increased maize and sugar beet yields. Though these aggregate affects appear small, the impacts are not evenly distributed. In cooler regions such as the United Kingdom and Ireland, the effect of increased warming has been ameliorated by an increase in rainfall. Warmer regions, such as Southern Europe, have suffered more from the warming; in Italy this effect has been amplified by a drying, leading to yield declines of 5% or greater.

Another study examining the impacts of recent climate trends on cereals in Greece showed that crops are clearly responding to changes in climate – and demonstrated via statistical analysis that significant impacts on wheat and barley production are expected at the end of the twenty-first century (Mavromatis 2015). In the Czech Republic, a study documented positive long-term impacts of recent warming on yields of fruiting vegetables (cucumbers and tomatoes) (from 4.9 to 12% per 1°C increase in local temperature) but decreases in yield stability of traditionally grown root vegetables in the warmest areas of the country (Potopová et al. 2017). A study in Hungary also indicated the increasingly negative impacts of temperature on crops and indicated that a warming climate is at least partially responsible for the stagnation or reduction in crop yields since the mid-1980s in Eastern Europe (Pinke and Lövei 2017).

In summary, climate change is already affecting some aspects of food security *(high confidence)*. Recent studies in both large-scale and smallholder farming systems document declines in crop productivity related to rising temperatures and changes in precipitation. Evidence for climate change impacts (e.g., declines and stagnation in yields, changes in sowing and harvest dates, increased infestation of pests and diseases, and declining viability of some crop varieties) is emerging from detection and attribution studies and indigenous and local knowledge in Australia, Europe, Asia, Africa, North America, and South America (*medium evidence, robust agreement*).

Projected impacts. Climate change effects have been studied on a global scale following a variety of methodologies that have recently been compared (Lobell and Asseng 2017; Zhao et al. 2017a; Liu et al. 2016). Approaches to study global and local changes include global gridded crop model simulations (e.g., (Deryng et al. 2014)), point-based crop model simulations (e.g., (Asseng et al. 2015)), analysis of point-based observations in the field (e.g., (Zhao et al. 2016)), and temperature-yield regression models (e.g., (Auffhammer and Schlenker 2014)). For an evaluation of model skills see e.g., used in AgMIP see Müller et al. (2017b).

Results from Zhao et al. (2017a) across different methods consistently showed negative temperature impacts on crop yield at the global scale, generally underpinned by similar impacts at country and site scales. A limitation of Zhao et al. (2017a) is that it is based on the assumption that yield responses to temperature increase are linear, while yield response differs depending on growing season temperature level. Iizumi et al. (2017) showed that the projected global mean yields of maize and soybean at the end of this century do decrease monotonically with warming, whereas those of rice and wheat increase with warming and level off at a warming of about 3°C (2091–2100 relative to 1850–1900).

Empirical statistical models have been applied widely to different cropping systems, at multiple scales. Analyses using statistical models for maize and wheat tested with global climate model scenarios found that the RCP4.5 scenario reduced the size of average yield impacts, risk of major slowdowns, and exposure to critical heat extremes compared to RCP8.5 in the latter decades of the 21st century (Tebaldi and Lobell 2018). Impacts on crops grown in the tropics are projected to be more negative than in mid- to high-latitudes as stated in AR5 and confirmed by recent studies (e.g., (Levis et al. 2018)). These projected negative effects in the tropics are especially pronounced under conditions of explicit nitrogen stress (Figure 5.4) (Rosenzweig et al. 2014).

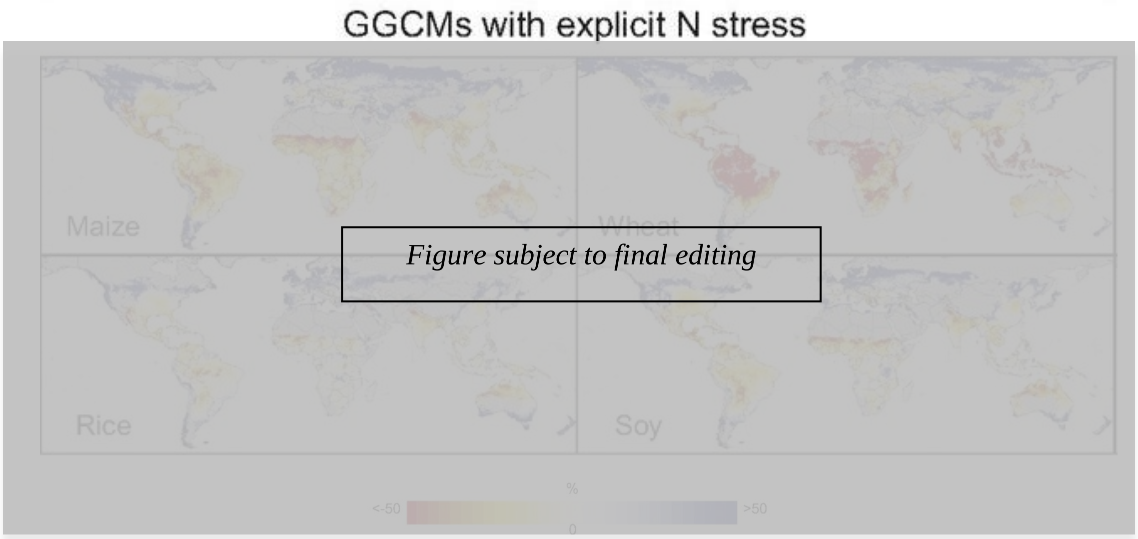


Figure 5.4 Median yield changes (%) for RCP8.5 (2070–2099 in comparison to 1980–2010 baseline) with CO_2 effects and explicit nitrogen stress over five GCMs x four Global Gridded Crop Models (GGCMs) for rainfed maize, wheat, rice, and soy (20 ensemble members from EPIC, GEPIC, pDSSAT, and PEGASUS; except for rice which has 15). Gray areas indicate historical areas with little to no yield capacity. All models use a 0.5° grid, but there are differences in grid cells simulated to represent agricultural land. While some models simulated all land areas, others simulated only potential suitable cropland area according to evolving climatic conditions; others utilised historical harvested areas in 2000 according to various data sources (Rosenzweig et al. 2014).

Reyer et al. (2017b) examined biophysical impacts in five world regions under different warming scenarios - 1, 1.5, 2, and 4 °C warming. For the Middle East and Northern Africa region a significant correlation between crop yield decrease and temperature increase was found, regardless of whether the effects of CO_2 fertilisation or adaptation measures are taken into account (Waha et al. 2017). For Latin America and the Caribbean the relationship between temperature and crop yield changes was only significant when the effect of CO_2 fertilisation is considered (Reyer et al. 2017a).

A review of recent scientific literature found that projected yield loss for West Africa depends on the degree of wetter or drier conditions and elevated CO_2 concentrations (Sultan and Gaetani 2016). Faye et al. (2018b) in a crop modelling study with RCPs 4.5 and 8.5 found that climate change could have limited effects on peanut yield in Senegal due to the effect of elevated CO_2 concentrations.

Crop productivity changes for 1.5°C and 2.0°C. The IPCC Special Report on Global Warming of 1.5°C found that climate-related risks to food security are projected to increase with global warming of 1.5°C and increase further with 2°C (IPCC 2018b). These findings are based among others on Schleussner et al. (2018); Rosenzweig et al. (2018a); Betts et al. (2018), Parkes et al. (2018) and Faye et al. (2018a). The importance of assumptions about CO_2 fertilisation was found to be significant by Ren et al. (2018) and Tebaldi and Lobell (2018)

AgMIP coordinated global and regional assessment (CGRA) results confirm that at the global scale, there are mixed results of positive and negative changes in simulated wheat and maize yields, with declines in some breadbasket regions, at both 1.5°C and 2.0°C (Rosenzweig et al. 2018a). In conjunction with price changes from the global economics models, productivity declines in the Punjab, Pakistan resulted in an increase in vulnerable households and poverty rate (Rosenzweig et al. 2018a).

Crop suitability. Another method of assessing the effects of climate change on crop yields that combined observations of current maximum-attainable yield with climate analogues also found strong reductions in attainable yields across a large fraction of current cropland by 2050 (Pugh et al. 2016).

However, the study found the projected total land area in 2050, including regions not currently used for crops, climatically suitable for a high attainable yield similar to today. This indicates that large shifts in land-use patterns and crop choice will likely be necessary to sustain production growth and keep pace with current trajectories of demand.

Fruits and vegetables. Understanding the full range of climate impacts on fruits and vegetables is important for projecting future food security, especially related to dietary diversity and healthy diets. However, studies for vegetables are very limited (Bisbis et al. 2018). Of the 174 studies considered in a recent review only 14 described results of field or greenhouse experiments studying impacts of increased temperatures on yields of different root and leafy vegetables, tomatoes and legumes (Scheelbeek et al. 2018). Bisbis et al. (2018) found similar effects for vegetables as have been found for grain crops, that is, the effect of increased CO_2 on vegetables is mostly beneficial for production, but may alter internal product quality, or result in photosynthetic down-regulation. Heat stress reduces fruit set of fruiting vegetables, and speeds up development of annual vegetables, shortening their time for photoassimilation. Yield losses and impaired product quality result, thereby increasing food loss and waste. On the other hand, a longer growing season due to warmer temperatures enables a greater number of plantings and can contribute to greater annual yields. However, some vegetables, such as cauliflower and asparagus, need a period of cold accumulation to produce a harvest and warmer winters may not provide those requirements.

For vegetables growing in higher baseline temperatures (>20°C), mean yield declines caused by 4°C warming were 31.5%; for vegetables growing in cooler environments (</= 20°C), yield declines caused by 4°C were much less, on the order of ~5% (Scheelbeek et al. 2018). Rippke et al. (2016) found that 30–60% of the common bean growing area and 20–40% of the banana growing areas in Africa will lose viability in 2078–2098 with a global temperature increase of 2.6°C and 4°C respectively. Tripathi et al. (2016) found fruits and vegetable production to be highly vulnerable to climate change at their reproductive stages and also due to potential for greater disease pressure.

In summary, studies assessed find that climate change will increasingly be detrimental to crop productivity as levels of warming progress *(high confidence)*. Impacts will vary depending on CO_2 concentrations, fertility levels, and region. Productivity of major commodity crops as well as crops such as millet and sorghum yields will be affected. Studies on fruits and vegetables find similar effects to those projected for grain crops in regard to temperature and CO_2 effects. Total land area climatically suitable for high attainable yield, including regions not currently used for crops, will be similar in 2050 to today.

5.2.2.2 Impacts on livestock production systems

Livestock systems are impacted by climate change mainly through increasing temperatures and precipitation variation, as well as atmospheric carbon dioxide (CO_2) concentration and a combination of these factors. Temperature affects most of the critical factors of livestock production, such as water availability, animal production and reproduction, and animal health (mostly through heat stress) (Figure 5.5). Livestock diseases are mostly affected by increases in temperature and precipitation variation (Rojas-Downing et al. 2017). Impacts of climate change on livestock productivity, particularly of mixed and extensive systems, are strongly linked to impacts on rangelands and pastures, which include the effects of increasing CO_2 on their biomass and nutritional quality. This is critical considering the very large areas concerned and the number of vulnerable people affected (Steinfeld 2010; Morton 2007). Pasture quality and quantity are mainly affected through increases in temperature and CO_2, and precipitation variation.

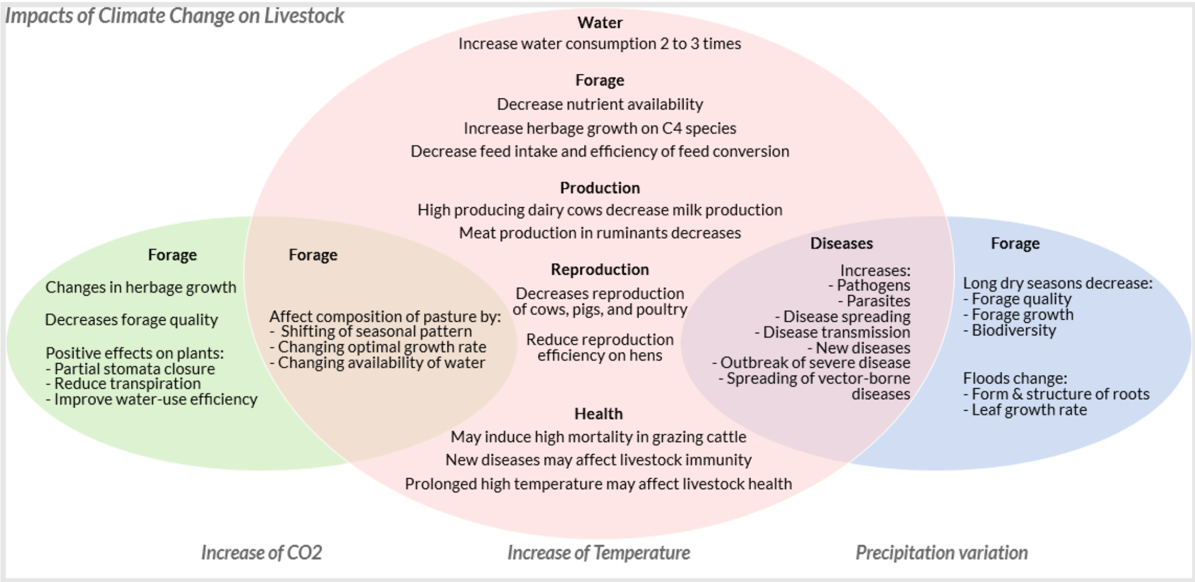


Figure 5.5 Impacts of climate change on livestock, based on (Rojas-Downing et al. 2017)

Among livestock systems, pastoral systems are particularly vulnerable to climate change (Dasgupta et al. 2014) (see Section 5.2.2.6 for impacts on smallholder systems that combine livestock and crops). Industrial systems will suffer most from indirect impacts leading to rises in the costs of water, feeding, housing, transport and the destruction of infrastructure due to extreme events, as well as an increasing volatility of the price of feedstuff which increases the level of uncertainty in production (Rivera-Ferre et al. 2016b; Lopez-i-Gelats 2014). Mixed systems and industrial or landless livestock systems could encounter several risk factors mainly due to the variability of grain availability and cost, and low adaptability of animal genotypes (Nardone et al. 2010).

Considering the diverse typologies of animal production, from grazing to industrial, Rivera-Ferre et al. (2016b) distinguished impacts of climate change on livestock between those related to extreme events and those related to more gradual changes in the average of climate-related variables. Considering vulnerabilities, they grouped the impacts as those impacting the animal directly, such as heat and cold stress, water stress, physical damage during extremes; and others impacting their environment, such as modification in the geographical distribution of vector-borne diseases, location, quality and quantity of feed and water and destruction of livestock farming infrastructures.

With severe negative impacts due to drought and high frequency of extreme events, the average gain of productivity might be cancelled by the volatility induced by increasing variability in the weather. For instance, semiarid and arid pasture will likely have reduced livestock productivity, while nutritional quality will be affected by CO_2 fertilisation (Schmidhuber and Tubiello 2007).

Observed impacts. Pastoralism is practiced in more than 75% of countries by between 200 and 500 million people, including nomadic communities, transhumant herders, and agro-pastoralists (McGahey et al. 2014). Observed impacts in pastoral systems reported in the literature include decreasing rangelands, decreasing mobility, decreasing livestock number, poor animal health, overgrazing, land degradation, decreasing productivity, decreasing access to water and feed, and increasing conflicts for the access to pasture land (López-i-Gelats et al. 2016; Batima et al. 2008; Njiru 2012; Fjelde and von Uexkull 2012; Raleigh and Kniveton 2012; Egeru 2016) (*high confidence*).

Pastoral systems in different regions have been affected differently. For instance, in China changes in precipitation were a more important factor in nomadic migration than temperature (Pei and Zhang 2014). There is some evidence that recent years have already seen an increase in grassland fires in parts of China and tropical Asia (IPCC 2012). In Mongolia, grassland productivity has declined by 20-30% over the latter half of the 20th centrury, and ewe average weight reduced by 4 kg on an annual basis, or about 8% since 1980 (Batima et al. 2008). Substantial decline in cattle herd sizes can be due to increased mortality and forced off-take (Megersa et al. 2014). Important but less studied is the impact of the interaction of grazing patterns with climate change on grassland composition. (Spence et al. 2014) showed that climate change effects on Mongolia mountain steppe could be contingent on land use.

Conflicts due to resource scarcity (as well as other socio-political factors (Benjaminsen et al. 2012)) aggravated by climate change has differentiated impact on women. In Turkana, female-headed households have lower access to decision-making on resource use and allocation, investment and planning (Omolo 2011), increasing their vulnerability (Section 5.1.3, Gender Box in Chapter 7).

Non-climate drivers add vulnerability of pastoral systems to climate change (McKune and Silva 2013). For instance, during environmental disasters, livestock holders have been shown to be more vulnerable to food insecurity than their crop-producing counterparts because of limited economic access to food and unfavorable market exchange rates (Nori et al. 2005). Sami reindeers in Finland showed reduced freedom of action in response to climate change due to loss of habitat, increased predation, and presence of economic and legal constraints) (Tyler et al. 2007; Pape and Löffler 2012). In Tibet, emergency aid has provided shelters and privatised communally owned rangeland, which have increased the vulnerability of pastoralists to climate change (Yeh et al. 2014; Næss 2013).

Projected impacts. The impacts of climate change on global rangelands and livestock have received comparatively less attention than the impacts on crop production. Projected impacts on grazing systems include changes in herbage growth (due to changes in atmospheric CO_2 concentrations and rainfall and temperature regimes) and changes in the composition of pastures and in herbage quality, as well as direct impacts on livestock (Herrero et al. 2016b). Droughts and high temperatures in grasslands can also be a predisposing factor for fire occurrence (IPCC 2012).

Net primary productivity, soil organic carbon, and length of growing period. There are large uncertainties related to grasslands and grazing lands (Erb et al. 2016), especially in regard to net primary productivity (NPP) (Fetzel et al. 2017; Chen et al. 2018). Boone et al. (2017) estimated that the mean global annual net primary production (NPP) in rangelands may decline by 10 g C m^{-2} yr^{-1} in 2050 under RCP 8.5, but herbaceous NPP is likely to increase slightly (i.e., average of 3 g C m^{-2} yr^{-1}) (Figure 5.6). Results of a similar magnitude were obtained by Havlík et al. (2015), using EPIC and LPJmL on a global basis (Rojas-Downing et al. 2017). According to Rojas-Downing et al. (2017), an increase of 2°C is estimate to negatively impact pasture and livestock production in arid and semiarid regions and positively impact humid temperate regions.

Boone et al. (2017) identified significant regional heterogeneity in responses, with large increases in annual productivity projected in northern regions (e.g., a 21% increase in productivity in the US and Canada) and large declines in western Africa (-46% in sub-Saharan western Africa) and Australia (-17%). Regarding the length of growing period (LGP, average number of growing days per year) (Herrero et al. 2016b) projected reductions in the lower latitudes due to changes in rainfall patterns and increases in temperatures, which indicate increasing limitations of water. They identified 35°C as a critical threshold for rangeland vegetation and heat tolerance in some livestock species.

Figure 5.6 Ensemble simulation results for projected annual net primary productivity of rangelands as simulated in 2000 (top) and their change in 2050 (bottom) under emissions scenario RCP 8.5, with plant responses enhanced by CO_2 fertilisation. Results from RCP 4.5 and 8.5, with and without positive effects of atmospheric CO_2 on plant production, differed considerably in magnitude but had similar spatial patterns, and so results from RCP 8.5 with increasing production are portrayed spatially here and in other figures. Scale bar labels and the stretch applied to colors are based on the spatial mean value plus or minus two standard deviations (Boone et al. 2017).

Rangeland composition. According to Boone et al. (2017), the composition of rangelands is projected to change as well (see Chapter 3). Bare ground cover is projected to increase, averaging 2.4% across rangelands, with increases projected for the eastern Great Plains, eastern Australia, parts of southern Africa, and the southern Tibetan Plateau. Herbaceous cover declines are projected in the Tibetan Plateau, the eastern Great Plains, and scattered parts of the Southern Hemisphere. Shrub cover is projected to decline in eastern Australia, parts of southern Africa, the Middle East, the Tibetan Plateau, and the eastern Great Plains. Shrub cover could also increase in much of the Arctic and some parts of Africa. In mesic and semi-arid savannahs south of the Sahara, both shrub and tree cover are projected to increase, albeit at lower productivity and standing biomass. Rangelands in western and southwestern parts of the Isfahan province in Iran were found to be more vulnerable to future drying–warming conditions (Saki et al. 2018; Jaberalansar et al. 2017).

Soil degradation and expanding woody cover suggest that climate-vegetation-soil feedbacks catalysing shifts toward less productive, possibly stable states (Ravi et al. 2010) may threaten mesic and semi-arid savannahs south of the Sahara (see Chapter 3 and 4). This will also change their suitability for grazing different animal species; switches from cattle, which mainly consume herbaceous plants, to goats or camels are likely to occur as increases in shrubland occur.

Direct and indirect effects on livestock. Direct impacts of climate change in mixed and extensive production systems are linked to increased water and temperature stress on the animals potentially leading to animal morbidity, mortality and distress sales. Most livestock species have comfort zones between 10^{o}C–30^{o}C, and at temperatures above this animals reduce their feed intake 3–5% per additional degree of temperature (NRC 1981). In addition to reducing animal production, higher temperatures negatively affect fertility (HLPE 2012).

Indirect impacts to mixed and extensive systems are mostly related to the impacts on the feed base, whether pastures or crops, leading to increased variability and sometimes reductions in availability and quality of the feed for the animals (Rivera-Ferre et al. 2016b). Reduced forage quality can increase CH_4 emissions per unit of gross energy consumed. Increased risk of animal diseases is also an important impact to all production systems (Bett et al. 2017). These depend on the geographical region, land use type, disease characteristics, and animal susceptibility (Thornton et al. 2009). Also important is the interaction of grazing intensity with climate change. Pfeiffer et al. (2019) estimated that in a scenario of mean annual precipitation below 500 mm increasing grazing intensity reduced rangeland productivity and increased annual grass abundance.

Pastoral systems. In Kenya, some 1.8 million extra cattle could be lost by 2030 because of increased drought frequency, the value of the lost animals and production foregone amounting to USD 630 million (Herrero et al. 2010). Martin et al. (2014) assessed impacts of changing precipitation regimes to identify limits of tolerance beyond which pastoral livelihoods could not be secured and found that reduced mean annual precipitation had always negative effects as opposed to increased rainfall variability. Similarly, Martin et al. (2016) found that drought effects on pastoralists in High Atlas in Morocco depended on income needs and mobility options (see Section 5.2.2.6 for additional information about impacts on smallholder farmers).

In summary, observed impacts in pastoral systems include changes in pasture productivity, lower animal growth rates and productivity, damaged reproductive functions, increased pests and diseases, and loss of biodiversity (*high confidence*). Livestock systems are projected to be adversely affected by rising temperatures, depending on the extent of changes in pasture and feed quality, spread of diseases, and water resource availability (*high confidence*). Impacts will differ for different livestock systems and for different regions (*high confidence*). Vulnerability of pastoral systems to climate change is very high (*high confidence*), and mixed systems and industrial or landless livestock systems could encounter several risk factors mainly due to variability of grain availability and cost, and low adaptability of animal genotypes. Pastoral system vulnerability is exacerbated by non-climate factors (land tenure issues, sedentarisation programs, changes in traditional institutions, invasive species, lack of markets, and conflicts) (*high confidence*).

5.2.2.3 Impacts on pests and diseases

Climate change is changing the dynamics of pests and diseases of both crops and livestock. The nature and magnitude of future changes is likely to depend on local agro-ecological and management context. This is because of the many biological and ecological mechanisms by which climate change can affect the distribution, population size, and impacts of pests and diseases on food production (Canto et al. 2009; Gale et al. 2009; Thomson et al. 2010; Pangga et al. 2011; Juroszek and von Tiedemann 2013; Bett et al. 2017).

These mechanisms include changes in host susceptibility due to CO_2 concentration effects on crop composition and climate stresses; changes in the biology of pests and diseases or their vectors (e.g., more generational cycles, changes in selection pressure driving evolution); mismatches in timing between pests or vectors and their 'natural enemies'; changes in survival or persistence of pests or disease pathogens (e.g., changes in crop architecture driven by CO_2 fertilisation and increased temperature, providing a more favourable environment for persistence of pathogens like fungi), and changes in pest distributions as their "climate envelopes" shift. Such processes may affect pathogens, and their vectors, as well as plant, invertebrate and vertebrate pests. (Latham et al. 2015) .

Furthermore, changes in diseases and their management, as well as changing habitat suitability for pests and diseases in the matrix surrounding agricultural fields, have the ability to reduce or exacerbate impacts (Bebber 2015). For example, changes in water storage and irrigation to adapt to rainfall variation have the potential to enhance disease vector populations and disease occurrence (Bett et al. 2017).

There is *robust evidence* that pests and diseases have already responded to climate change (Bebber et al. 2014), and many studies have now built predictive models based on current incidence of pests, diseases or vectors that indicate how they may respond in future (e.g., (Caminade et al. 2015; Kim et al. 2015; Kim and Cho 2016; Samy and Peterson 2016; Yan et al. 2017)). Warren et al. (2018) estimate that about 50% of insects, which are often pests or disease vectors, will change ranges by about 50% by 2100 under current GHG emissions trajectories. These changes will lead to crop losses due to changes in insect pests (Deutsch et al. 2018) and weed pressure (Ziska et al. 2018), and thus affect pest and disease management at the farm level (Waryszak et al. 2018). For example, Samy and Peterson (2016) modelled Blue-tongue virus (BTV), which is spread by biting *Culicodes* midges, finding that the distribution of BTV is likely to be extended, particularly in central Africa, the US, and western Russia.

There is some evidence (*medium confidence*) that exposure will, on average, increase (Bebber and Gurr 2015; Yan et al. 2017), although there are a few examples where changing stresses may limit the range of a vector. There is also a general expectation that perturbations may increase the likelihood of pest and disease outbreaks by disturbing processes that may currently be at some quasi-equilibrium (Canto et al. 2009; Thomson et al. 2010; Pangga et al. 2011). However, in some places, and for some diseases, risks may decrease as well as increase (e.g., drying out may reduce the ability of fungi to survive) (Kim et al. 2015; Skelsey and Newton 2015), or Tsetse fly's range may decrease (Terblanche et al. 2008; Thornton et al. 2009) .

Pests, diseases, and vectors for both crop and livestock diseases are likely to be altered by climate change (*high confidence*). Such changes are likely to depend on specifics of the local context, including management, but perturbed agroecosystems are more likely, on theoretical grounds, to be subject to pest and disease outbreaks (*low confidence*). Whilst specific changes in pest and disease pressure will vary with geography, farming system, pest/pathogen – increasing in some situations decreasing in others – there is robust evidence, with *high agreement*, that pest and disease pressures are likely to change; such uncertainty requires robust strategies for pest and disease mitigation.

5.2.2.4 Impacts on pollinators

Pollinators play a key role on food security globally (Garibaldi et al. 2016). Pollinator-dependent crops contribute up to 35% of global crop production volume and are important contributors to healthy human diets and nutrition (IPBES 2016). On a global basis, some 1500 crops require pollination (typically by insects, birds and bats) (Klein et al. 2007). Their importance to nutritional security is therefore perhaps under-rated by valuation methodologies, which, nonetheless, include estimates of the global value of pollination services at over USD 225 billion (2010 prices) (Hanley et al. 2015). As with other ecosystem processes affected by climate change (e.g., changes in pests and diseases), how complex systems respond is highly context-dependent. Thus, predicting the effects of climate on pollination services is difficult (Tylianakis et al. 2008; Schweiger et al. 2010) and uncertain, although there is *limited evidence* that impacts are occurring already (Section 5.2.2.4), and *medium evidence* that there will be an effect.

Pollination services arise from a mutualistic interaction between an animal and a plant – which can be disrupted by climate's impacts on one or the other or both (Memmott et al. 2007). Disruption can occur through changes in species' ranges or by changes in timing of growth stages (Settele et al.

2016). For example, if plant development responds to different cues (e.g., day length) from insects (e.g., temperature), the emergence of insects may not match the flowering times of the plants, causing a reduction in pollination. Climate change will affect pollinator ranges depending on species, life-history, dispersal ability and location. Warren et al. (2018) estimate that under a 3.2ºC warming scenario, the existing range of about 49% of insects will be reduced by half by 2100, suggesting either significant range changes (if dispersal occurs) or extinctions (if it does not). However, in principle, ecosystem changes caused by invasions, in some cases, could compensate for the decoupling generated between native pollinators and pollinated species (Schweiger et al. 2010).

Other impacts include changes in distribution and virulence of pathogens affecting pollinators, such as the fungus *Nosema cerana*, which can develop at a higher temperature range than the less-virulent *Nosema apis*; increased mortality of pollinators due to higher frequency of extreme weather events; food shortage for pollinators due to reduction of flowering length and intensity; and aggravation of other threats, such as habitat loss and fragmentation (González-Varo et al. 2013; Goulson et al. 2015; Le Conte and Navajas 2008; Menzel et al. 2006; Walther et al. 2009; IPBES, 2016). The increase in atmospheric CO_2 is also reducing the protein content of pollen, with potential impact on pollination population biology (Ziska et al. 2016).

In summary, as with other complex agroecosystem processes affected by climate change (e.g., changes in pests and diseases), how pollination services respond will be highly context-dependent. Thus, predicting the effects of climate on pollination services is difficult and uncertain, although there is *medium evidence* that there will be an effect.

5.2.2.5 Impacts on aquaculture

This report focuses on land-based aquaculture; for assessment of impacts on marine fisheries both natural and farmed see the IPCC Special Report on the Ocean and Cryosphere in a Changing Climate (SROCC, forthcoming).

Aquaculture will be affected by both direct and indirect climate change drivers, both in the short and the long-term. Barange et al. (2018) provides some examples of short-term loss of production or infrastructure due to extreme events such as floods, increased risk of diseases, toxic algae and parasites; and decreased productivity due to suboptimal farming conditions; and long-term impacts may include scarcity of wild seed, limited access to freshwater for farming due to reduced precipitation, limited access to feeds from marine and terrestrial sources, decreased productivity due to suboptimal farming conditions, eutrophication and other perturbations.

FAO (2014a) assessed the vulnerability of aquaculture stakeholders to non-climate change drivers which add to climate change hazards. Vulnerability arises from discrimination in access to inputs and decision-making; conflicts; infrastructure damage; and dependence on global markets and international pressures. Other non-climate drivers identified by McClanahan et al. (2015), which add vulnerability to fisheries for food security include: declining fishery resources; a North–South divide in investment; changing consumption patterns; increasing reliance on fishery resources for coastal communities; and inescapable poverty traps creating by low net resource productivity and few alternatives. In areas where vulnerability to climate change is heightened, increased exposure to climate change variables and impacts is likely to exacerbate current inequalities in the societies concerned, penalising further already disadvantaged groups such as migrant fishers (e.g. Lake Chad) or women (e.g. employees in Chile's processing industry) (FAO 2014a).

In many countries the projected declines co-occur across both marine fisheries and agricultural crops (Blanchard et al. 2017), both of which will impact the aquaculture and livestock sectors (See Supplementary Material Figure SM5.1). Countries with low Human Development Index, trade opportunities and aquaculture technologies are likely to face greater challenges. These cross-sectoral impacts point to the need for a more holistic account of the inter-connected vulnerabilities of food systems to climate and global change.

5.2.2.6 Impacts on smallholder farming systems

New work has developed farming system approaches that take into account both biophysical and economic processes affected by climate change and multiple activities. Farm households in the developing world often rely on a complex mix of crops, livestock, aquaculture, and non-agricultural activities for their livelihoods (Rosenzweig and Hillel 2015; Antle et al. 2015). Across the world, smallholder farmers are considered to be disproportionately vulnerable to climate change because changes in temperature, rainfall and the frequency or intensity of extreme weather events directly affect their crop and animal productivity as well as their household's food security, income and well-being (Vignola et al. 2015; Harvey et al. 2014b). For example, smallholder farmers in the Philippines, whose survival and livelihood largely depend on the environment, constantly face risks and bear the impacts of the changing climate (Peria et al. 2016).

Smallholder farming systems have been recognised as highly vulnerable to climate change (Morton, 2007) because they are highly dependent on agriculture and livestock for their livelihood (Dasgupta et al. 2014) (*high confidence*). In Zimbawe, farmers were found vulnerable due to their marginal location, low levels of technology, and lack of other essential farming resources. Farmers observed high frequency and severity of drought, excessive precipitation, drying up of rivers, dams and wells, and changes in timing and pattern of seasons as evidence of climate change, and indicated that prolonged wet, hot, and dry weather conditions resulted in crop damage, death of livestock, soil erosion, bush fires, poor plant germination, pests, lower incomes, and deterioration of infrastructure (Mutekwa 2009).

In Madagascar, Harvey et al. (2014b) conducted surveyed 600 small farmers and found that chronic food insecurity, physical isolation and lack of access to formal safety nets increased Malagasy farmers' vulnerability to any shocks to their agricultural system, particularly extreme events. In Chitwan, Nepal, occurrence of extreme events and increased variability in temperature has increased the vulnerability of crops to biotic and abiotic stresses and altered the timing of agricultural operations; thereby affecting crop production (Paudel et al. 2014). In Lesotho, a study on subsistence farming found that food crops were the most vulnerable to weather, followed by soil and livestock. Climate variables of major concern were hail, drought and dry spells which reduced crop yields. In the Peruvean Altiplan, Sietz et al. (2012) evaluate smallholders' vulnerability to weather extremes with regard to food security and found the relevance of resource scarcity (livestock, land area), diversification of activities (lack of alternative income, education deprivation) and income restrictions (harvest failure risk) in shaping vulnerability of smallholders. See Section 5.2.2.6 for observed impacts on smallholder pastoral systems.

Projected impacts. By including regional economic models, integrated methods take into account the potential for yield declines to raise prices and thus livelihoods (up to a certain point) in some climate change scenarios. Regional economic models of farming systems can be used to examine the potential for switching to other crops and livestock, as well as the role that non-farm income can play in adaptation (Valdivia et al. 2015; Antle et al. 2015). On the other hand, lost income for smallholders from climate change-related declines, for example in coffee production, can decrease their food security (Hannah et al. 2017).

Farming system methods developed by AgMIP have been used in regional integrated assessments in Sub-Saharan Africa (Kihara et al. 2015), West Africa (Adiku et al. 2015); East Africa (Rao et al. 2015), South Africa (Beletse et al. 2015), Zimbabwe (Masikati et al. 2015), South Asia (McDermid et al. 2015), Pakistan (Ahmad et al. 2015), the Indo-Gangetic Basin (Subash et al. 2015), Tamil Nadu

(Ponnusamy et al. 2015) and Sri Lanka (Zubair et al. 2015). The assessments found that climate change adds pressure to smallholder farmers across Sub-Saharan Africa and South Asia, with winners and losers within each area studied. Temperatures are expected to increase in all locations, and rainfall decreases are projected for the western portion of West Africa and Southern Africa, while increases in rainfall are projected for eastern West Africa and all study regions of South Asia. The studies project that climate change will lead to yield decreases in most study regions except South India and areas in central Kenya, as detrimental temperature effects overcome the positive effects of CO_2. These studies use AgMIP representative agricultural pathways (RAPs) as a way to involve stakeholders in regional planning and climate resilience (Valdivia et al. 2015). RAPs are consistent with and complement the RCP/SSP approaches for use in agricultural model intercomparisons, improvement, and impact assessments

New methods have been developed for improving analysis of climate change impacts and adaptation options for the livestock component of smallholder farming systems in Zimbabwe (Descheemaeker et al. 2018). These methods utilised disaggregated climate scenarios, as well as differentiating farms with larger stocking rates compared to less densely stocked farms. By disaggregating climate scenarios, impacts, and smallholder farmer attributes, such assessments can more effectively inform decision-making towards climate change adaptation.

In Central Asia, a study using the bio-economic farm model (BEFM) found large differences in projected climate change impact ranging from positive income gains in large-scale commercial farms in contrast to negative impacts in small-scale farms (Bobojonov and Aw-Hassan 2014). Negative impacts may be exacerbated if irrigation water availability declines due to climate change and increased water demand in upstream regions. In Iran, changes in rainfall and water endowments are projected to significantly impact crop yield and water requirements, as well as income and welfare of farm families (Karimi et al. 2018).

Climate change impacts on food, feed and cash crops other than cereals, often grown in smallholder systems or family farms are less often studied, although impacts can be substantial. For example, areas suitable for growing coffee are expected to decrease by 21% in Ethiopia with global warming of 2.4°C (Moat et al. 2017) and more than 90% in Nicaragua (Läderach et al. 2017) with 2.2°C local temperature increase.

Climate change can modify the relationship between crops and livestock in the landscape, affecting mixed crop-livestock systems in many places. Where crop production will become marginal, livestock may provide an alternative to cropping. Such transitions could occur in up to 3% of the total area of Africa, largely as a result of increases in the probability of season failure in the drier mixed crop–livestock systems of the continent (Thornton et al. 2014).

In Mexico, subsistence agriculture is expected to be the most vulnerable to climate change, due to its intermittent production and reliance on maize and beans (Monterroso et al. 2014). Overall, a decrease in suitability and yield is expected in Mexico and Central America for beans, coffee, maize, plantain and rice (Donatti et al. 2018). Municipalities with a high proportional area under subsistence crops in Central America tend to have less resources to promote innovation and action for adaptation (Bouroncle et al. 2017). .

In summary, smallholder farmers are especially vulnerable to climate change because their livelihoods often depend primarily on agriculture. Further, smallholder farmers often suffer from chronic food insecurity (*high confidence*). Climate change is projected to exacerbate risks of pests and diseases and extreme weather events in smallholder farming systems.

5.2.3 Climate change impacts on access

Access to food involves the ability to obtain food, including the ability to purchase food at affordable prices.

5.2.3.1 Impacts on prices and risk of hunger

A protocol-based analysis based on AgMIP methods tested a combination of RCPs and SSPs to provide a range of projections for prices, risk of hunger, and land use change (Figure 5.7 and Supplementary Material Table SM5.4.) (Hasegawa et al. 2018). Previous studies have found that decreased agricultural productivity will depress agricultural supply, leading to price increases. Despite different economic models with various representations of the global food system (Valin et al. 2014; Robinson et al. 2014; Nelson et al. 2013; Schmitz et al. 2014), as well as having represented the SSPs in different ways (i.e., technological change, land-use policies, sustainable diets, etc. (Stehfest et al. 2019; Hasegawa et al. 2018)), the ensemble of participating models projected a 1-29% cereal price increase in 2050 across SSPs 1, 2, and 3 due to climate change (RCP 6.0), which would impact consumers globally through higher food prices; regional effects will vary. The median cereal price increase was 7%, given current projections of demand. In all cases (across SSPs and global economic models), prices are projected to increase for rice and coarse grains, with only one instance of a price decline (-1%) observed for wheat in SSP1, with price increases projected in all other cases. Animal-sourced foods (ASFs) are also projected to see price increases (1%), but the range of projected price changes are about half those of cereals, highlighting that the climate impacts on ASFs is indirect through the cost and availability of feed, and that there is significant scope for feed substitution within the livestock sector.

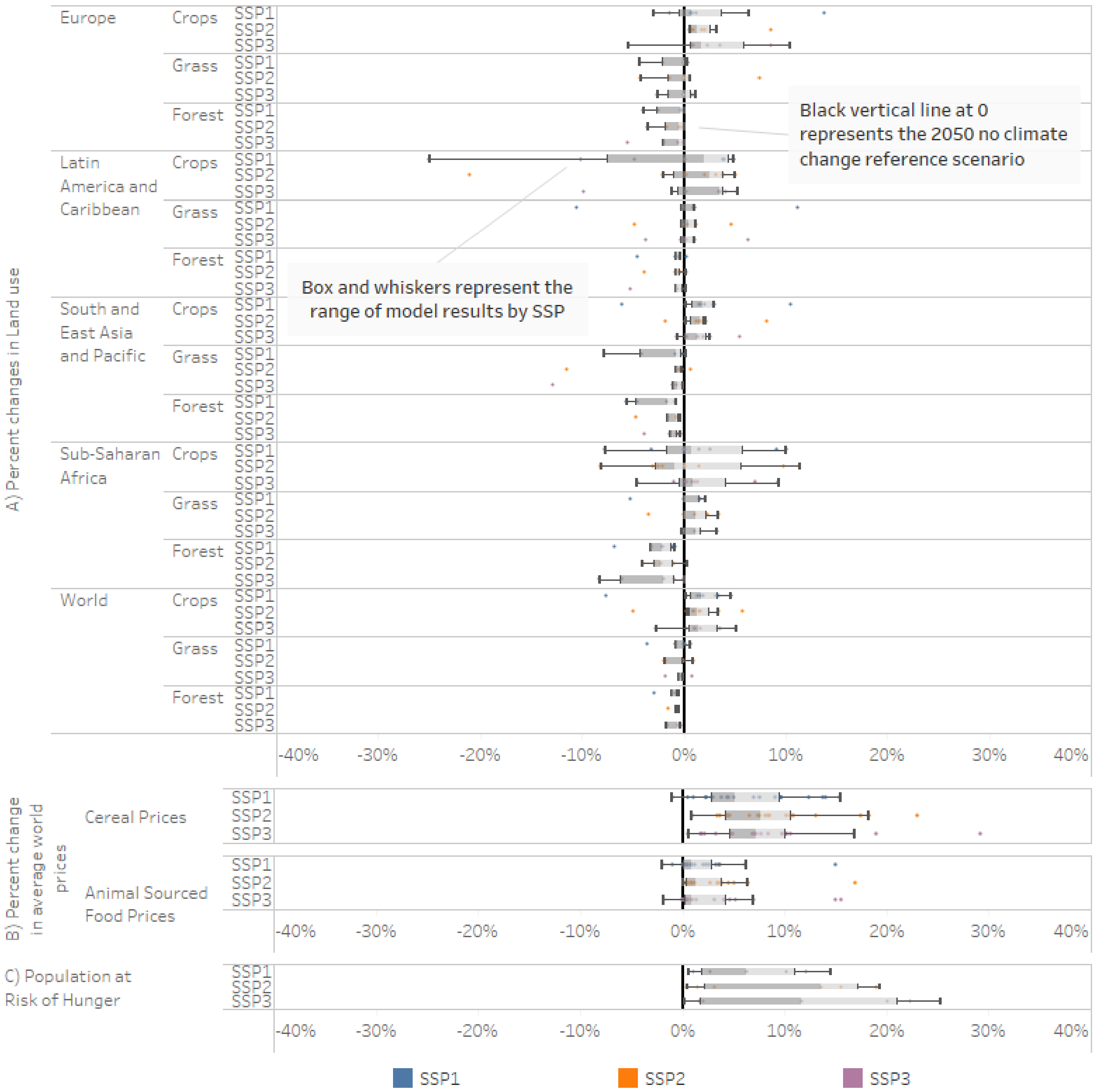


Figure 5.7. Implications of climate change by 2050 on land-use, selected agricultural commodity prices, and the population at risk of hunger based on AgMIP Global Economic Model analysis. (A) Projected % change in land-use by 2050 by land type (cropland, grassland, and forest) and SSP. (B) Projected % changes in average world prices by 2050 for cereals (rice, wheat, and coarse grains) and animal sourced foods (ruminant meat, monogastric, and dairy) by SSP. (C) Percentage change by 2050 in the global population at risk of hunger by SSP.

Declining food availability caused by climate change is likely to lead to increasing food cost impacting consumers globally through higher prices and reduced purchasing power, with low-income consumers particularly at risk from higher food prices (Nelson et al. 2010; Springmann et al. 2016a; Nelson et al. 2018). Higher prices depress consumer demand, which in turn will not only reduce energy intake (calories) globally (Hasegawa et al. 2015; Nelson et al. 2010; Springmann et al. 2016a; Hasegawa et al. 2018), but will also likely lead to less healthy diets with lower availability of key micronutrients (Nelson et al. 2018) and increase diet-related mortality in lower and middle-income countries (Springmann et al. 2016a). These changes will slow progress towards the eradication of malnutrition in all its forms.

The extent that reduced energy intake leads to a heightened risk of hunger varies by global economic model. However, all models project an increase in the risk of hunger, with the median projection of an increase in the population at risk of insufficient energy intake by 6, 14, and 12 % in 2050 for SSPs 1, 2 and 3 respectively compared to a no climate change reference scenario. This median percentage increase would be the equivalent of 8, 24, and 80 million (full range 1-183 million) additional people at risk of hunger due to climate change (Hasegawa et al. 2018).

5.2.3.2 Impacts on land use

Climate change is likely to lead to changes in land use globally (Nelson et al. 2014; Schmitz et al. 2014; Wiebe et al. 2015). Hasegawa et al. (2018) found that declining agricultural productivity broadly leads to the need for additional cropland, with 7 of 8 models projecting increasing cropland and the median increase by 2050 projected across all models of 2 % compared to a no climate change reference (Figure 5.7). Not all regions will respond to climate impacts equally, with more uncertainty on regional land-use change across the model ensemble than the global totals might suggest. For example, the median land-use change for Latin America is an increase of cropland by 3 %, but the range across the model ensemble is significant, with 3 models projecting declines in cropland (-25 –-1 %) compared to the 5 models projecting cropland increase (0 – 5 %). For further discussion on land use change and food security see Section 5.6.

5.2.4 Climate change impacts on food utilisation

Food utilisation involves nutrient composition of food, its preparation, and overall state of health. Food safety and quality affects food utilisation.

5.2.4.1 Impacts on food safety and human health

Climate change can influence food safety through changing the population dynamics of contaminating organisms due to, for example, changes in temperature and precipitation patterns, and also humidity, increased frequency and intensity of extreme weather events, and changes in contaminant transport pathways. Changes in food and farming systems, e.g., intensification to maintain supply under climate change, may also increase vulnerabilities as the climate changes (Tirado et al. 2010). Climate-related changes in the biology of contaminating organisms include changing the activity of mycotoxin-producing fungi, changing the activity of micro-organisms in aquatic food chains that cause disease (e.g., dinoflagellates, bacteria like *Vibrio*), and increasingly heavy rainfall and floods causing contamination of pastures with enteric microbes (like *Salmonella*) that can enter the human food chain. Degradation and spoilage of products in storage and transport can also be affected by changing humidity and temperature outside of cold chains, notably from microbial decay but also from potential changes in the population dynamics of stored product pests (e.g., mites, beetles, moths) (Moses et al. 2015).

Mycotoxin-producing fungi occur in specific conditions of temperature and humidity, so climate change will affect their range, increasing risks in some areas (such as mid-temperate latitudes) and reducing them in others (e.g., the tropics) (Paterson and Lima 2010). There is *robust evidence* from process-based models of particular species (*Aspergillus*/Aflatoxin B1, *Fusarium*/deoxynivalenol) with projections of future climate that show that aflatoxin contamination of maize in southern Europe will increase significantly (Battilani et al. 2016), and deoxynivalenol contamination of wheat in north-west Europe will increase by up to 3 times (van der Fels-Klerx et al. 2012b,a). Whilst the downscaled climate models make any specific projection for a given geography uncertain (Van der Fels-Klerx et al. 2013), experimental evidence on the small scale suggests that the combination of rising CO_2 levels, affecting physiological processes in photosynthetic organisms, and temperature changes, can be significantly greater than temperature alone (Medina et al. 2014). Risks related to aflatoxins are likely to change, but detailed projections are difficult because they depend on local conditions (Vaughan et al. 2016).

Foodborne pathogens in the terrestrial environment typically come from enteric contamination (from humans or animals), and can be spread by wind (blowing contaminated soil) or flooding – the incidence of both of which are likely to increase with climate change (Hellberg and Chu 2016). Furthermore, water stored for irrigation, which may be increased in some regions as an adaptation strategy, can become an important route for the spread of pathogens (as well as other pollutants); contaminated water and diarrheal diseases are acute threats to food security (Bond et al. 2018). Whilst there is little direct evidence (in terms of modelled projections) the results of a range of reviews, as well as expert groups, suggest that risks from foodborne pathogens are likely to increase through multiple mechanisms (Tirado et al. 2010; van der Spiegel et al. 2012; Liu et al. 2013; Kirezieva et al. 2015; Hellberg and Chu 2016).

An additional route to climate change impacts on human health can arise from the changing biology of plants altering human exposure levels. This may include climate changing how crops sequester heavy metals (Rajkumar et al. 2013), or how they respond to changing pest pressure (e.g., cassava produces hydrogen cyanide as a defence against herbivore attack).

All of these factors will lead to regional differences regarding food safety impacts (Paterson and Lima 2011). For instance, in Europe it is expected that most important food safety-related impacts will be mycotoxins formed on plant products in the field or during storage; residues of pesticides in plant products affected by changes in pest pressure; trace elements and/or heavy metals in plant products depending on changes in abundance and availability in soils; polycyclic aromatic hydrocarbons in foods following changes in long-range atmospheric transport and deposition; and presence of pathogenic bacteria in foods following more frequent extreme weather, such as flooding and heat waves (Miraglia et al. 2009).

In summary, there is *medium evidence,* with *high agreement* that food utilisation via changes in food safety (and potentially food access from food loss) will be impacted by climate change, mostly by increasing risks, but there is *low confidence,* exactly how they may change for any given place.

5.2.4.2 Impacts on food quality

There are two main routes by which food quality may change. First, the direct effects of climate change on plant and animal biology, such as through changing temperatures changing the basic metabolism of plants. Secondly, by increasing carbon dioxide's effect on biology through CO_2 fertilisation.

Direct effects on plant and animal biology. Climate affects a range of biological processes, including the metabolic rate in plants and ectothermic animals. Changing these processes can change growth rates, and therefore yields, but can also cause organisms to change relative investments in growth vs reproduction, and therefore change the nutrients assimilated. This may decrease protein and mineral nutrient concentrations, as well as alter lipid composition (DaMatta et al. 2010). For example, apples in Japan have been exposed to higher temperatures over 3–4 decades and have responded by blooming earlier. This has led to changes in acidity, firmness, and water content, reducing quality (Sugiura et al. 2013). In other fruit, such as grapes, warming-induced changes in sugar composition affect both colour and aroma (Mira de Orduña 2010). Changing heat stress in poultry can affect yield as well as meat quality (by altering fat deposition and chemical constituents), shell quality of eggs, and immune systems (Lara and Rostagno 2013).

Effects of rising CO_2 concentrations. Climate change is being driven by rising concentrations of carbon dioxide and other greenhouse gases in the atmosphere. As plants use CO_2 in photosynthesis to form sugar, rising CO_2 levels, all things being equal, enhances the process unless limited by water or nitrogen availability. This is known as "CO_2 fertilisation". Furthermore, increasing CO_2 allows the stomata to be open for a shorter period for gas exchange, reducing water loss through transpiration. These two factors affect the metabolism of plants, and, as with changing temperatures, affects plant growth rates, yields and their nutritional quality. Studies of these effects include meta-analyses, modelling, and small-scale experiments (Franzaring et al. 2013; Mishra and Agrawal 2014; Myers et al. 2014; Ishigooka et al. 2017; Zhu et al. 2018; Loladze 2014; Yu et al. 2014)

In regard to nutrient quality, a meta-analysis from seven Free-Air Carbon dioxide Enrichment (FACE), (with elevated atmospheric CO_2 concentration of 546–586 ppm) experiments (Myers et al. 2014), found that wheat grains had 9.3% lower zinc (CI5.9–12.7%), 5.1% lower iron (CI 3.7–6.5%) and 6.3% lower protein (CI 5.2–7.5%), and rice grains had 7.8% lower protein content (CI 6.8–8.9%). Changes in nutrient concentration in field pea, soybean and C4 crops such as sorghum and maize were small or insignificant. Zhu et al. (2018) report a meta-analysis of FACE trials on a range of rice cultivars. They show that protein declines by an average of 10% under elevated CO_2, iron and zinc decline by 8% and 5% respectively. Furthermore, a range of vitamins show large declines across all rice cultivars, including B1 (-17%), B2 (-17%), B5 (-13%) and B9 (-30%), whereas Vitamin E increased. As rice underpins the diets of many of the world's poorest people in low-income countries, especially in Asia, Zhu et al. (2018) estimate that these changes under high CO_2 may affect the nutrient status of about 600 million people.

Decreases in protein concentration with elevated CO_2 are related to reduced nitrogen concentration possibly caused by nitrogen uptake not keeping up with biomass growth, an effect called 'carbohydrate dilution' or 'growth dilution', and by inhibition of photorespiration which can provide much of the energy used for assimilating nitrate into proteins (Bahrami et al. 2017). Other mechanisms have also been postulated (Feng et al. 2015; Bloom et al. 2014; Taub and Wang 2008). Together, the impacts on protein availability may take as many as 150 million people into protein deficiency by 2050 (Medek et al. 2017). Legume and vegetable yields increased with elevated CO_2 concentration of 250 ppm above ambient by 22% (CI 11.6–32.5%), with a stronger effect on leafy vegetables than on legumes and no impact for changes in iron, vitamin C or flavonoid concentration (Scheelbeek et al. 2018).

Increasing concentrations of atmospheric CO_2 lower the content of zinc and other nutrients in important food crops. Dietary deficiencies of zinc and iron are a substantial global public health problem (Myers et al. 2014). An estimated two billion people suffer these deficiencies (FAO 2013a), causing a loss of 63 million life-years annually (Myers et al. 2014). Most of these people depend on C3 grain legumes as their primary dietary source of zinc and iron. Zinc deficiency is currently responsible for large burdens of disease globally, and the populations who are at highest risk of zinc deficiency receive most of their dietary zinc from crops (Myers et al. 2015). The total number of people estimated to be placed at new risk of zinc deficiency by 2050 is 138 million. The people likely to be most affected live in Africa and South Asia, with nearly 48 million residing in India alone. Differences between cultivars of a single crop suggest that breeding for decreased sensitivity to atmospheric CO_2 concentration could partly address these new challenges to global health (Myers et al. 2014).

In summary, while increased CO_2 is projected to be beneficial for crop productivity at lower temperature increases, it is projected to lower nutritional quality (e.g., less protein, zinc, and iron) *(high confidence)*.

5.2.5 Climate change impacts on food stability

Food stability is related to people's ability to access and use food in a steady way, so that there not intervening periods of hunger. Increasing extreme events associated with climate change can disrupt food stability. (See Section 5.8.1 for assessment of food price spikes.)

5.2.5.1 Impacts of extreme events

FAO et al. (2018) conducted an analysis of prevalence of undernourishment (PoU) and found that in 2017, the average of the prevalence of undernourishment (PoU) was 15.4% for all countries exposed to climate extremes (See Supplementary Material Figure SM5.2). At the same time, the PoU was 20% for countries that additionally show high vulnerability of agriculture production/yields to climate variability, or 22.4% for countries with high PoU vulnerability to severe drought. When there is both high vulnerability of agriculture production/yields and high PoU sensitivity to severe drought, the PoU is 9.8 points higher (25.2%). These vulnerabilities were found to be higher when countries had a high dependence on agriculture as measured by the number of people employed in the sector. Bangkok experienced severe flooding in 2011-2012 with large-scale disruption of the national food supply chains since they were centrally organised in the capital city (Allen et al. 2017).

The IPCC projects that frequency, duration, and intensity of some extreme events will increase in the coming decades (IPCC 2018a, 2012). To test these effects on food security, Tigchelaar et al. (2018) showed rising instability in global grain trade and international grain prices, affecting especially the about 800 million people living in extreme poverty who are most vulnerable to food price spikes (see Section 5.8.1). They used global datasets of maize production and climate variability combined with future temperature projections to quantify how yield variability will change in the world's major maize-producing and -exporting countries under 2°C and 4°C of global warming.

Tesfaye et al. (2017) projected that the extent of heat-stressed areas in South Asia could increase by up to 12% in 2030 and 21% in 2050 relative to the baseline (1950–2000). Another recent study found that drier regions are projected to dry earlier, more severely and to a greater extent than humid regions, with the population of sub-Saharan Africa most vulnerable (Lickley and Solomon 2018).

5.2.5.2 Food aid

Food aid plays an important role in providing food security and saving lives after climate disasters. In 2015, 14.5 million people were assisted through disaster-risk reduction, climate change and/or resilience building activities (WFP 2018). However, there is no agreement on how to better use emergency food aid, since it can come with unintended consequences for individuals, groups, regions, and countries (Barrett 2006). These may include negative dependency of food recipients (Lentz et al. 2005) or price increases, among others.

Some authors state that tied food aid provided as "in kind" by the donor country hampers local food production (Clay 2006), although others found no evidence of this (Ferrière and Suwa-Eisenmann 2015). Untied cash aid can be used to buy food locally or in neighbouring countries, which is cheaper and can contribute to improving the livelihoods of local farmers (Clay 2006).

Ahlgren et al. (2014) found that food aid dependence of Marshall Islands due to climate change impacts can result in poor health outcomes due to the poor nutritional quality of food aid, which may result in future increases of chronic diseases. In this regard, Mary et al. (2018) showed that nutrition-sensitive aid can reduce the prevalence of undernourishment.

In summary, based on AR5 and SR15 assessments that the likelihood that extreme weather will increase, (e.g., increases in heatwaves, droughts, inland and coastal flooding due to sea level rise depending on region) in both frequency and magnitude, decreases in food stability and thus increases in food insecurity will likely rise as well *(medium evidence, high agreement)*.

5.3 Adaptation options, challenges, and opportunities

This section assesses the large body of literature on food system adaptation to climate change, including increasing extreme events, within a framework of autonomous, incremental, and transformational adaptation. It focuses primarily on regional and local considerations and adaptation options for both the supply side (production, storage, transport, processing, and trade) and the demand side (consumption and diets) of the food system. Agroecological, social, and cultural contexts are considered throughout. Finally, the section assesses the role of institutional measures at global, regional (multiple countries), national, and local scales and capacity-building.

5.3.1 Challenges and opportunities

By formulating effective adaptation strategies, it is possible to reduce or even avoid some of the negative impacts of climate change on food security (See Section 5.2). However, if unabated climate change continues, limits to adaptation will be reached (SR15). In the food system, adaptation actions involve any activities designed to reduce vulnerability and enhance resilience of the system to climate change. In some areas, expanded climate envelopes will alter agro-ecological zones, with opportunity for expansion towards higher latitudes and altitudes, soil and water resources permitting (Rosenzweig and Hillel 2015).

More extreme climatic events are projected to lead to more agro-meteorological disasters with associated economic and social losses. There are many options for adapting the food system to extreme events reported in IPCC (2012), highlighting measures that reduce exposure and vulnerability and increase resilience, even though risks cannot fully be eliminated (IPCC 2012). Adaptation responses to extreme events aim to minimise damages, modify threats, prevent adverse impacts, or share losses, thus making the system more resilient (Harvey et al. 2014a).

With current and projected climate change (higher temperature, changes in precipitation, flooding and extremes events), achieving adaptation will require both technological (e.g., recovering and improving orphan crops, new cultivars from breeding or biotechnology) and non-technological (e.g., market, land management, diet change) solutions. Climate interacts with other factors such as market supplies over longer distances and policy drivers (Mbow et al. 2008; Howden et al. 2007), as well as local agricultural productivity to determine access to food locally.

Given the site-specific nature of climate change impacts on food system components together with wide variation in agroecosystems types and management, and socio-economic conditions, it is widely understood that adaptation strategies are linked to environmental and cultural contexts at the regional and local levels (*high confidence*). Developing systemic resilience that integrates climate drivers with social and economic drivers would reduce the impact on food security, particularly in developing countries. For example, in Africa, improving food security requires evolving food systems to be highly climate resilient, while supporting the need for increasing yield to feed the growing population (Mbow et al. 2014b) (Box 5.2).

Adaptation involves producing more food where needed, moderating demand, reducing waste, and improving governance (Godfray and Garnett 2014) (see Section 5.6 for the significant synergies between adaptation and mitigation through specific practices, actions and strategies.).

Box 5.2 Sustainable solutions for food systems and climate change in Africa

Climate change, land use change, and food security are important aspects of sustainability policies in Africa. According to the McKinsey Global Institute (2010), Africa has around 60% of the global uncultivated arable land; thus the continent has a high potential for transformative change in food production. With short and long-term climate change impacts combined with local poverty conditions, land degradation and poor farming practices, Africa cannot grow enough food to feed its rapidly growing population. Sustainable improvement of productivity is essential, even as the impacts of climate change on food security in Africa are projected to be plural and severe.

Sustainable Land Management (SLM) of farming systems is important to address climate change while dealing with these daunting food security needs and the necessity to improve access to nutritious food to maintain healthy and active lives in Africa (AGRA 2017). SLM has functions beyond the production of food, such as delivery of water, protection against disease (especially zoonotic diseases), the delivery of energy, fibre and building materials.

Commodity-based systems—driven by external markets—are increasing in Africa (cotton, cocoa, coffee, oil palm, groundnuts) with important impacts on the use of land and climate. Land degradation, decreasing water resources, loss of biodiversity, excessive use of synthetic fertilisers and pesticides are some of the environmental challenges that influence preparedness to adapt to climate change (Pretty and Bharucha 2015).

A balanced strategy on African agriculture can be based on SLM and multifunctional land use approaches combining food production, cash crops, ecosystem services, biodiversity conservation, and ecosystem services delivery, and indigenous and local knowledge.

Thus, sustainable food systems in Africa entail multiple dimensions as shown in Figure 5.7.

Figure 5.7 Factors influencing sustainable food systems in Africa

With rapid urbanisation, it is important to used combined land goals (e.g., zero-carbon energy, smart irrigation systems, and climate-resilient agriculture) to minimise the negative side effects of climate change while securing quality food for a growing population.

Building resilience into productivity and production can be based on simultaneous attention to the following five overarching issues:

1) Closing yield gaps through adapted cultivars, sustainable land management, that combine production and preservation of ecosystems essential functions such as sustainable

intensification approaches based on conservation agriculture and community-based adaptation with functioning support services and market access (Mbow et al. 2014a).

2) Identifying Sustainable Land Management practices (agroecology, agroforestry, etc.) addressing different ecosystem services (food production, biodiversity, reduction of GHG emissions, soil carbon sequestration) for improved land-based climate change adaptation and mitigation (Sanz et al. 2017; Francis 2016).

3) Paying attention to the food-energy-water nexus, especially water use and reutilisation efficiency but also management of rain water (Albrecht et al. 2018).

4) Implementing institutional designs focused on youth, women through new economic models that help access credit and loans to support policies that balance cash and food crops.

5) Build on and use of local knowledge, culture and traditions while seeking innovations for food waste reduction and transformation of agricultural products.

These aspects suppose both incremental and transformational adaptation that may stem from better infrastructure (storage and food processing), adoption of harvest and post-harvest technologies that minimise food waste, and development of new opportunities for farmers to respond to environmental, economic and social shocks that affect their livelihoods (Morton 2017).

Agriculture in Africa offers a unique opportunity for merging adaption to and mitigation of climate change with sustainable production to ensure food security (CCAFS 2012; FAO 2012). Initiatives throughout the food system on both the supply and demand sides can lead to positive outcomes.

5.3.2 Adaptation framing and key concepts

5.3.2.1 Autonomous, incremental, and transformational adaptation

Framing of adaptation in this section categorises and assesses adaptation measures as autonomous, incremental, and transformational (See Glossary and Table 5.3). Adaptation responses can be reactive or anticipatory.

Autonomous. Autonomous adaptation in food systems does not constitute a conscious response to climatic stimuli but is triggered by changes in agroecosystems, markets, or welfare changes. It is also referred to as spontaneous adaptation (IPCC 2007). Examples of autonomous adaptation of rural populations have been documented in the Sahel (IRD 2017). In India, farmers are changing sowing and harvesting timing, cultivating short duration varieties, inter-cropping, changing cropping patterns, investing in irrigation, and establishing agroforestry. These are considered as passive responses or autonomous adaptation, because they do not acknowledge that these steps are taken in response to perceived climatic changes (Tripathi and Mishra 2017).

Incremental. Incremental adaptation maintains the essence and integrity of a system or process at a given scale (Park et al. 2012). Incremental adaptation focuses on improvements to existing resources and management practices. The central aim of incremental adaptation is to maintain the essence and integrity of a system or process at a given scale (IPCC 2014a).

Transformational. Transformational adaptation changes the fundamental attributes of a socio-ecological system either in anticipation of or in response to climate change and its impacts (IPCC 2014a). Transformational adaptation seeks alternative livelihoods and land use strategies needed to develop new farming systems (Termeer et al. 2016). For example, limitations in incremental adaptation among smallholder rice farmers in Northwest Costa Rica led to a shift from rice to sugarcane production due to decreasing market access and water scarcity (Warner et al. 2015). Migration from the Oldman River Basin has been described as a transformational adaption to climate change in the Canadian agriculture sector (Hadarits et al. 2017). If high-end scenarios of climate change eventuate, the food security of farmers and consumers will depend on how transformational change in food systems is managed. An integrated framework of adaptive transition – management of socio-technical transitions and adaptation to socio-ecological changes – may help build transformational adaptive capacity (Mockshell and Kamanda 2018; Pant et al. 2015). Rippke et al. (2016) has suggested overlapping phases of adaptation needed to support transformational change in Africa.

Table 5.3 Synthesis of food security related adaptation options to address various climate risks (IPCC 2014b; Vermeulen et al. 2013, 2018; Burnham and Ma 2016; Bhatta and Aggarwal 2016)

Key climate drivers and risks	**Incremental adaptation**	**Transformational adaptation**	**Enabling conditions**
Extreme events and short-term climate variability Stress on water resources, drought stress, dry spells, heat extremes, flooding, shorter rainy seasons, pests	- Change in variety, water management, water harvesting, supplemental irrigation during dry spells, - Planting dates, pest control, feed banks, - Transhumance, Other sources of revenue (e.g. charcoal, wild fruits, wood, temporary work) - Soil management, composting,	- Early Warning Systems - Use of planning and prediction at seasonal to intra-seasonal climate risk to transition to a food safer condition. - Abandonment of monoculture, diversification - Crop and livestock insurance - Alternate cropping, intercropping -Erosion control	- Establishment of climate services - Integrated water management policies, integrated land and water governance - Seed banks, seed sovereignty and seed distribution policies - Capacity building and extension programs
Warming trend, drying trend Reduced crop productivity due to persistent heat, long drought cycles, deforestation and land degradation with strong adverse effects on food production and nutrition quality, increased pest and disease damage	- Strategies to reduce effects of recurring food challenges - Sustainable intensification, agroforestry, conservation agriculture, SLM - Adoption of existing drought-tolerant crop and livestock species - Counter season crop production, - Livestock fattening - New ecosystem-based adaptation (e.g. bee keeping, woodlots)	- Climate services for new agricultural programs, e.g., sustainable irrigation districts) - New technology, e.g., new farming systems, new crops and livestock breeds - Switches between cropping and transhumant livelihoods, replacement of pasture or forest to irrigated/rainfed crops - Shifting to small ruminants or drought resistant livestock or fish	- Climate information in local development policies. - Stallholders' access to credit and production resources, - National food security program based on increased productivity, diversification, transformation and trade - Strengthening (budget, capacities, expertise) of local and national institutions to support agriculture and

	- Farmers management of natural resources - Labor redistribution (e.g., mining, development projects, urban migration) - Adjustments to markets and trade pathways already in place	farming Food storage infrastructures, food transformation - Changes in cropping area, land rehabilitation (enclosures, afforestation) perennial farming - New markets and trade pathways	livestock breeding - Devolution to local communities, women empowerment, market opportunities - Incentives for establishing new markets and trade pathways

5.3.2.2 *Risk management*

Climate risks affect all pillars of food security, particularly stability because extreme events lead to strong variation to food access. The notion of risk is widely treated in IPCC reports (IPCC 2014c) (see also Chapter 7 in this report). With food systems, many risks co-occur or reinforce each other and this can limit effective adaptation planning as they require a comprehensive and dynamic policy approach covering a range of drivers and scales. For example, from the understanding by farmers of change in risk profiles to the establishment of efficient markets that facilitate response strategies will require more than systemic reviews of risk factors (Howden et al. 2007).

Integration of Climate Change Adaptation (CCA) and Disaster Risk Reduction (DRR) helps to minimise the overlap and duplication of projects and programs (Nalau et al. 2016). Recently, countries started integrating the concept of DRR and CCA. For instance, The Philippines has introduced legislation calling for CCA and DRR integration as current policy instruments were largely unsuccessful in combining agencies and experts across the two areas (Leon and Pittock 2016).

Studies reveal that the amplitude of interannual growing-season temperature variability is in general larger than that of long-term temperature change in many locations. Responding better to seasonal climate-induced food supply shocks therefore increases society's capability to adapt to climate change. Given these backgrounds, seasonal crop forecasting and early response recommendations, based on seasonal climate forecasts, are emerging to strengthen existing operational systems for agricultural monitoring and forecasting (FAO 2016a; Ceglar et al. 2018; Iizumi et al. 2018).

While adaptation and mitigation measures are intended to reduce the risk from climate change impacts in food systems, they can also be sources of risk themselves (e.g. investment risk, political risk) (IPCC 2014b). Climate-related hazards are a necessary element of risks related to climate impacts but may have little or nothing to do with risks related to some climate policies/responses

Adoption of agroecological practices could provide resilience for future shocks, spread farmer risk and mitigate the impact of droughts (Niles et al. 2018) (See Section 5.3.2.3). Traditionally, risk management is performed through multifunctional landscape approaches in which resource utilisation is planned across wide areas and local agreements on resource access. Multifunctionality permits vulnerable communities to access various resources at various times and under various risk conditions (Minang et al. 2015).

In many countries, governmental compensation for crop-failure and financial losses are used to protect against risk of severe yield reductions. Both public and private sector groups develop insurance markets and improve and disseminate index-based weather insurance programs.

Catastrophe bonds, microfinance, disaster contingency funds, and cash transfers are other available mechanisms for risk management.

In summary, risk management can be accomplished through agro-ecological landscape approaches and risk sharing and transfer mechanisms such as development of insurance markets and improve index-based weather insurance programs *(high confidence)*.

5.3.2.3 Role of agroecology and diversification

Agro-ecological systems are integrated land-use systems that maintain species diversity in a range of productive niches. Diversified cropping systems and practicing traditional agro-ecosystems of crop production where a wide range of crop varieties are grown in various spatial and temporal arrangements, are less vulnerable to catastrophic loss (Zhu et al. 2011). The use of local genetic diversity, soil organic matter enhancement, multiple-cropping or poly-culture systems, and home gardening, agro-ecological approaches can build resilience against extreme climate events (Altieri and Koohafkan 2008). However, Nie et al. (2016) argued that while integrated crop-livestock systems present some opportunities such as control of weeds, pests and diseases, and environmental benefits, there are some challenges, including yield reduction, difficulty in pasture-cropping, grazing, and groundcover maintenance in high rainfall zones, and development of persistent weeds and pests. Adaptation measures based on agroecology entail enhancement of agrobiodiversity; improvement of ecological processes and delivery of ecosystem services. They also entail strengthening of local communities and recognition of the role and value of indigenous and local knowledge. Such practices can enhance the sustainability and resilience of agricultural systems by buffering climate extremes, reducing degradation of soils, and reversing unsustainable use of resources; outbreak of pests and diseases and consequently increase yield without damaging biodiversity. Increasing and conserving biological diversity such as soil microorganisms can promote high crop yields and sustain the environment (Schmitz et al 2015; Bhattacharyya et al 2016; Garibaldi et al 2017).

Diversification of many components of the food system is a key element for increasing performance and efficiency that may translate into increased resilience and reduced risks (integrated land management systems, agrobiodiversity, indigenous and local knowledge, local food systems, dietary diversity, the sustainable use of indigenous fruits, neglected and underutilised crops as a food source) *(medium confidence)* (Makate et al. 2016; Lin 2011; Awodoyin et al. 2015).

The more diverse the food systems are, the more resilient they are in enhancing food security in the face of biotic and abiotic stresses. Diverse production systems are important for providing regulatory ecosystem services such as nutrient cycling, carbon sequestration, soil erosion control, reduction of GHG emissions and control of hydrological processes (Chivenge et al. 2015). Further options for adapting to change in both mean climate and extreme events are livelihood diversification (Michael 2017; Ford et al. 2015), and production diversity (Sibhatu et al. 2015).

Crop diversification, maintaining local genetic diversity, animal integration, soil organic matter management, water conservation, and harvesting the role of microbial assemblages. These types of farm management significantly affect communities in soil, plant structure, and crop growth in terms of number, type, and abundance of species (Morrison-Whittle et al. 2017). Complementary strategies towards sustainable agriculture (ecological intensification, strengthening existing diverse farming systems and investment in ecological infrastructure) also address important drivers of pollinator decline (IPBES 2016).

Evidence also shows that, together with other factors, on-farm agricultural diversity can translate into dietary diversity at the farm level and beyond (Pimbert and Lemke 2018; Kumar et al. 2015; Sibhatu et al. 2015). Dietary diversity is important but not enough as an adaptation option, but results in positive health outcomes by increasing the variety of healthy products in people's diets and reducing exposure to unhealthy environments.

Locally developed seeds and the concept of seed sovereignty can both help protect local agrobiodiversity and can often be more climate resilient than generic commercial varieties (Wattnem 2016; Coomes et al., 2015; van Niekerk and Wynberg 2017; Vasconcelos et al. 2013). Seed exchange networks and banks protect local agrobiodiversity and landraces, and can provide crucial lifelines when crop harvests fail (Coomes et al. 2015; van Niekerk and Wynberg 2017; Vasconcelos et al. 2013).

Related to locally developed seeds, neglected and underutilised species (NUS) can play a key role in increasing dietary diversity (*high confidence*) (Baldermann et al. 2016; van der Merwe et al. 2016; Kahane et al. 2013; Muhanji et al. 2011) (see Box 5.3). These species can also improve nutritional and economic security of excluded social groups, such as tribals (Nandal and Bhardwaj 2014; Ghosh-Jerath et al. 2015), indigent (Kucich and Wicht 2016) or rural populations (Ngadze et al. 2017).

Box 5.3 Climate change and indigenous food systems in the Hindu-Kush Himalayan Region

Diversification of production systems through promotion of Neglected and Underutilised Species (NUS; also known as understudied, neglected, orphan, lost or disadvantaged crops) offers adaptation opportunities to climate change, particularly in mountains. Neglected and Underutilised Species (NUS) have a potential to improve food security and at the same time help protect and conserve traditional knowledge and biodiversity. Scaling-up NUS requires training farmers and other stakeholders on ways to adopt adequate crop management, quality seed, select varieties, farming systems, soil management, development of new products, and market opportunities (Padulosi et al. 2013). Farmers in the Rasuwa district, in the mid-hills of Nepal, prefer to cultivate local bean, barley, millet and local maize, rather than commodity crops because they are more tolerant to water stress and extremely cold conditions (Adhikari et al. 2017). Farmers in the high-altitude cold climate of Nepal prefer local barley with its short growing period because of a shorter growing window. Buckwheat is commonly grown in the Hindu-Kush Himalayan (HKH) region mainly because it grows fast and suppresses weeds. In Pakistan, quinoa (*Chenopodium quinoa*) grew and produced well under saline and marginal soil where other crops would not grow (Adhikari et al. 2017).

At the same time, in many parts of the HKH region, a substantial proportion of the population is facing malnutrition. Various factors are responsible for this, and lack of diversity in food and nutrition resulting from production and consumption of few crops is one of them. In the past, food baskets in this region consisted of many different edible plant species, many of which are now neglected and underutilised. This is because almost all the efforts of the Green Revolution after 1960 focused on major crops. Four crops viz. rice, wheat, maize and potato account for about 60% of global plant-derived energy supply (Padulosi et al. 2013).

While the Green Revolution technologies substantially increased the yield of few crops and allowed countries to reduce hunger, they also resulted in inappropriate and excessive use of agrochemicals, inefficient water use, loss of beneficial biodiversity, water and soil pollution and significantly reduced crop and varietal diversity. With farming systems moving away from subsistence-based to commercial farming, farmers are also reluctant to grow these local crops because of low return, poor market value and lack of knowledge about their nutritional environmental value.

However, transition from traditional diets based on local foods to a commercial crop-based diet with high fats, salt, sugar and processed foods, increased the incidence of non-communicable diseases, such as diabetes, obesity, heart diseases and certain types of cancer (Abarca-Gómez et al. 2017; NCD-

RisC 2016b, 2017b). This 'hidden hunger' – enough calories, but insufficient vitamins - is increasingly evident in mountainous communities including the HKH region.

Internationally, there is rising interest nowadays on NUS, not only because they present tremendous opportunities for fighting poverty, hunger and malnutrition, but also because of their role in mitigating climate risk in agricultural production systems. NUS play an important role in mountain agro-ecosystems because mountain agriculture is generally low-input agriculture, for which many NUS are well adapted.

In the HKH region, mountains are agro-ecologically suitable for cultivation of traditional food crops, such as barley, millet, sorghum, buckwheat, bean, grams, taro, yam and a vast range of wild fruits, vegetables and medicinal plants. In one study carried out in two villages of mid-hills in Nepal, Khanal et al. (2015) reported 52 indigenous crop species belonging to 27 families with their various uses. Farming communities continue to grow various indigenous crops, albeit in marginal land, because of their value on traditional food and associated culture. Nepal Agricultural Research Council (NARC) has identified a list of indigenous crops based on their nutritional, medicinal, cultural and other values.

Many indigenous crops supply essential micronutrients to the human body, and need to be conserved in mountain food systems. Farmers in HKH region are cultivating and maintaining various indigenous crops such as amaranthus, barley, black gram, horse gram, olarum, yam, rayo, sesame, niger, etc. because of their nutritional value. Most of these indigenous crops are comparable with commercial cereals in terms of dietary energy and protein content, but are also rich in micronutrients. For example, pearl millet has higher content of calcium, iron, zinc, fiboflavin and folic acid than rice or maize (Adhikari et al. 2017).

NUS can provide both climate resilience and more options for dietary diversity to the farming communities of mountain ecosystems. Some of these indigenous crops have high medical importance. For example, mountain people in the HKH region have been using *jammun* (i.e., *Syzygium cumini*) to treat diabetes. In the Gilgit-Baltistan province of Pakistan, realising the importance of sea-buckthorn for nutritional and medicinal purposes, local communities have expanded its cultivation to larger areas. Many of these crops can be cultivated in marginal and/or fallow land which otherwise remains fallow. Most of these species are drought resistant and can be easily grown in rainfed conditions in non-irrigated land.

Dietary diversity has also been correlated (*medium evidence, medium agreement*) to agricultural diversity in small-holder and subsistence farms (Ayenew et al. 2018; Jones et al. 2014; Jones 2017; Pimbert and Lemke 2018), including both crops and animals, and has been proposed as a strategy to reduce micronutrient malnutrition in developing countries (Tontisirin et al. 2002). In this regard, the capacity of subsistence farming to supply essential nutrients in reasonable balance to the people dependent on them has been considered as a means of overcoming their nutrient limitations in sound agronomic and sustainable ways (Graham et al. 2007).

Ecosystem-based adaptation (EbA). EbA is a set of nature-based methods addressing climate change adaptation and food security by strengthening and conserving natural functions, goods and services that benefit to people. EbA approaches to address food security provide co-benefits such as contributions to health and improved diet, sustainable land management, economic revenue and water security. EbA practices can reduce greenhouse gas emissions and increase carbon storage (USAID 2017).

For example, agroforestry systems can contribute to improving food productivity while enhancing biodiversity conservation, ecological balance and restoration under changing climate conditions

(Mbow et al. 2014a; Paudela et al. 2017; Newaj et al. 2016; Altieri et al. 2015). Agroforestry systems have been shown to reduce erosion through their canopy cover and their contribution to the micro-climate and erosion control (Sida et al. 2018). Adoption of conservation farming practices such as removing weeds from and dredging irrigation canals, draining and levelling land, and using organic fertilisation were among the popular conservation practices in small-scale paddy rice farming community of northern Iran (Ashoori and Sadegh 2016).

Adaptation potential of ecologically-intensive systems includes also forests and rivers ecosystems, where improved resources management such as soil conservation, water cycling and agro-biodiversity support the function of food production affected by severe climate change (Muthee et al. 2017). The use of non-crop plant resources in agro-ecosystems (permaculture, perennial polyculture) can improve ecosystem conservation and may lead to increased crop productivity (Balzan et al. 2016; Crews et al. 2018; Toensmeier 2016).

In summary, increasing the resilience of the food system through agroecology and diversification is an effective way to achieve climate change adaptation (*robust evidence, high agreement)*. Diversification in the food system is a key adaptation strategy to reduce risks (e.g., implementation of integrated production systems at landscape scales, broad-based genetic resources, and heterogeneous diets) (*medium confidence*).

5.3.2.4 Role of cultural values

Food production and consumption are strongy influenced by cultures and beliefs. Culture, values and norms are primary factors in most climate change and food system policies. The benefits of integrating cultural beliefs and indigenous and local knowledge (ILK) into formal climate change mitigation and adaptation strategies can add value to the development of sustainable climate change that are rich in local aspirations, and planned with and for local people (Nyong et al. 2007).

Cultural dimensions are important in understanding how societies establish food production systems and respond to climate change, since they help to explain differences in responses across populations to the same environmental risks (Adger et al. 2013). There is an inherent adaptability of indigenous people who are particularly connected to land use, developed for many centuries to produce specific solutions to particular climate change challenges. Acknowledging that indigenous cultures across the world are supporting many string strategies and beliefs that offer sustainable systems with pragmatic solutions will help move forward the food and climate sustainability policies. For instance, in the Sahel, the local populations have developed and implemented various adaptation strategies that sustain their resilience despite many threats (Nyong et al. 2007). There is an increased consideration of these local knowledge and cultural values and norms in the design and implementation of modern mitigation and adaptation strategies.

There are some entrenched cultural beliefs and values that may be barriers to climate change adaptation. For instance, culture has been shown to be a major barrier to adaptation for the Fulbe ethnic group of Burkina Faso (Nielsen and Reenberg 2010). Thus, it is important to understand how beliefs, values, practices and habits interact with the behaviour of individuals and collectivities that have to confront climate change (Heyd and Thomas 2008). Granderson (2014) suggests that making sense of climate change and its responses at the community level demands attention to the cultural and political processes that shape how risk is conceived, prioritised and managed. For a discussion of gender issues related to climate change, see Section 5.2.

Culturally sensitive risk analysis can deliver a better understanding of what climate change means for society (O'Brien and Wolf 2010; Persson et al. 2015) and thus, how to better adapt. Murphy et al. (2016) stated that culture and beliefs play an important role in adaptive capacity but that they are not static. In the work done by Elum et al. (2017) in South Africa about farmers perception of climate change, they concluded that perceptions and beliefs often have negative effects on adaptation options.

Culture is a key issue in food systems and the relation of people with nature. Food is an intrinsically cultural process: food production shapes landscapes, which are in turn linked to cultural heritages and identities (Koohafkan and Altieri 2011; Fuller and Qingwen 2013), and food consumption has a strong cultural dimension. The loss of subsistence practices in modern cultures and its related indigenous and local knowledge, has resulted in a loss of valuable adaptive capacities (Hernández-Morcillo et al. 2014). This is so because these systems are often characterised by livelihood strategies linked to the management of natural resources that that have been evolved to reduce overall vulnerability to climate shocks ('adaptive strategies') and to manage their impacts ex-post ('coping strategies') (Morton 2007; López-i-Gelats et al. 2016).

5.3.3 Supply-side adaptation

Supply-side adaptation takes place in the production (of crops, livestock, and aquaculture), storage, transport, processing, and trade of food.

5.3.3.1 Crop production

There are many current agricultural management practices that can be optimised and scaled up to advance adaptation. Among the often-studied adaptation options include increased soil organic matter, improved cropland management, increased food productivity, prevention and reversal of soil erosion (see Chapter 6 for evaluation of these practices in regard to desertification and land degradation). Many analyses have demonstrated the effectiveness of soil management and changing sowing date, crop type or variety (Waongo et al. 2015; Bodin et al. 2016; Teixeira et al. 2017; Waha et al. 2013; Zimmermann et al. 2017; Chalise and Naranpanawa 2016; Moniruzzaman 2015; Sanz et al. 2017). Biophysical adaptation options also include pest and disease management (Lamichhane et al. 2015) and water management (Palmer et al. 2015; Korbeľová and Kohnová 2017).

In Africa, Scheba (2017) found that conservation agriculture techniques were embedded in an agriculture setting based on local traditional knowledge, including crop rotation, no or minimum tillage, mulching, and cover crops. Cover cropping and no-tillage also improved soil health in a highly commercialised arid irrigated system in California's San Joaquin Valley, US (Mitchell et al. 2017). Biofertilisers can enhance rice yields (Kantachote et al. 2016), and Amanullah and Khalid (2016) found that manure and biofertiliser improve maize productivity under semi-arid conditions.

Adaptation also involves use of current genetic resources as well as breeding programs for both crops and livestock. More drought, flood and heat-resistant crop varieties (Atlin et al. 2017; Mickelbart et al. 2015; Singh et al. 2017) and improved nutrient and water use efficiency, including overabundance as well as water quality (such as salinity) (Bond et al. 2018) are aspects to factor in to the design of adaptation measures. Both availability and adoption of these varieties is a possible path of adaptation and can be facilitated by new outreach policy and capacity building.

Water management is another key area for adaptation. Increasing water availability and reliability of water for agricultural production using different techniques of water harvesting, storage, and its judicious utilisation through farm ponds, dams, and community tanks in rainfed agriculture areas have been presented by Rao et al. (2017) and Rivera-Ferre et al. (2016a). In addition, improved drainage systems (Thiel et al. 2015), and Alternate Wetting and Drying (AWD) techniques for rice cultivation (Howell et al. 2015; Rahman and Bulbul 2015) have been proposed. Efficient irrigation systems have been also analysed and proposed by (Jägermeyr et al. 2016) Naresh et al (2017) (Gunarathna et al.

2017; Chartzoulakis and Bertaki 2015). Recent innovation includes using farming system with low usage of water such as drip-irrigation or hydroponic systems mostly in urban farming.

5.3.3.2 Livestock production systems

Considering the benefits of higher temperature in temperate climate and the increase of pasture with incremental warming in some humid and temperate grasslands, as well as potential negative effects, can be useful in planning adaptation strategies to future climate change. Rivera-Ferre et al. (2016b) characterize adaptation for different livestock systems as managerial, technical, behavioural and policy-related options. Managerial included production adjustments (e.g., intensification, integration with crops, shifting from grazing to browsing species, multispecies herds, mobility, soil and nutrient management, water management, pasture management, corralling, feed and food storage, farm diversification or cooling systems); and changes in labor allocation (diversifying livelihoods, shifting to irrigated farming, labor flexibility). Technological options included breeding strategies and information technology research. Behavioral options are linked to cultural patterns and included encouraging social collaboration and reciprocity, e.g., livestock loans, communal planning, food exchanges. and information sharing. Policy options are discussed in Section 5.7 and Chapter 7.

5.3.3.3 Aquaculture, fisheries, and agriculture interactions

Options may include livelihood diversification within and across sectors of fisheries, aquaculture and agriculture. Thus, adaptation options need to provide management approaches and policies that build the livelihood asset base, reducing vulnerability to multiple stressors with a multi-sector perspective (Badjeck et al. 2010). In Bangladesh fishing pressure on post-larval prawns has increased as displaced farmers have shifted to fishing following salt-water intrusion of agricultural land (Ahmed et al. 2013). In West Africa, strategies to cope with sudden shifts in fisheries are wider-reaching and have included turning to seafood import (Gephart et al. 2017) or terrestrial food production including farming and bush-meat hunting on land (Brashares et al. 2004). Proposed actions for adaptation include effective governance, improved management and conservation, efforts to maximise societal and environmental benefits from trade, increased equitability of distribution and innovation in food production, and the continued development of low-input and low-impact aquaculture (FAO 2018c).

Particular adaptation strategies proposed by FAO (2014a) include diverse and flexible livelihood strategies, such as introduction of fish ponds in areas susceptible to intermittent flood/drought periods; flood-friendly small-scale homestead bamboo pens with trap doors allowing seasonal floods to occur without loss of stocked fish; cage fish aquaculture development using plankton feed in reservoirs created by dam building; supporting the transition to different species, polyculture and integrated systems, allowing for diversified and more resilient systems; promotion of rice–fish farming systems reducing overall water needs and providing integrated pest management; and supporting transitions to alternative livelihoods.

Risk reduction initiatives include innovative weather-based insurance schemes being tested for applicability in aquaculture and fisheries and climate risk assessments introduced for integrated coastal zone management. For aquaculture’s contribution to building resilient food systems, Troell et al. (2014) found that aquaculture could potentially enhance resilience through improved resource use efficiencies and increased diversification of farmed species, locales of production, and feeding strategies. Yet, given its high reliance on terrestrial crops and wild fish for feeds, its dependence on freshwater and land for culture sites and its environmental impacts reduce this potential. For instance, the increase in aquaculture worldwide may enhance land competition for feed crops, increasing price levels and volatility and worsening food insecurity among the most vulnerable populations.

5.3.3.4 Transport and storage

Fewer studies have been done on adaptation of food system transport and storage compared to the many studies on adaptation to climate in food production.

Transport. One transport example is found in Bangkok. Between mid-November 2011 and early January 2012, Bangkok, the capital city of Thailand, faced its most dramatic flood in approximately 70 years with most transport networks cut-off or destroyed. This caused large-scale disruption of the national food supply chains since they were centrally organised in the capital city (Allen et al. 2017). From this experience, the construction and management of 'climate-proof' rural roads and transport networks is argued as one the most important adaptation strategies for climate change and food security in Thailand (Rattanachot et al. 2015).

Similarly in Africa, it has been shown that enhanced transportation networks combined with other measures could reduce the impact of climate change on food and nutrition security (Brown et al. 2017b). This suggests that strengthening infrastructure and logistics for transport would significantly enhance resilience to climate change while improving food and nutrition security in developing counties.

Storage. Storage refers to both structures and technologies for storing seed as well as produce. Predominant storage methods used in Uganda are single-layer woven polypropylene bags (popularly called "kavera" locally), chemical insecticides and granaries. Evidence from Omotilewa et al. (2018) showed that the introduction of new storage technology called Purdue Improved Crop Storage (PICS) could contribute to climate change adaptation. PICS is a chemical-free airtight triple-layered technology consisting of two high-density polyethylene inner liners and one outer layer of woven polypropylene bag. Its adoption has increased the number of households planting hybrid maize varieties that are more susceptible to insect pests in storage than traditional lower-yielding varieties. Such innovations could help to protect crops more safely and for longer periods from postharvest insect pests that are projected to increase as result of climate change, thus contributing to food security. In the Indo-Gangetic Plains many different storage structures based on ILK provide reliable and low-cost options made of local materials. For example, elevated grain stores protect from floods, but also provide for air circulation to prevent rot and to control insects and other vermin (Rivera-Ferre et al. 2013).

5.3.3.5 Trade and processing

Adaptation measures are also being considered in trade, processing, and packaging, other components of the food system. These will enable availability, stability, and safety of food under changing climate conditions.

Trade. Brooks & Matthews (2015) found that food trade increases the availability of food by enabling products to flow from surplus to deficit areas, raises incomes and favors access to food, improves utilisation by increasing the diversity of national diets while pooling production risks across individual markets to maintain stability.

Processing. Growth of spoilage bacteria of red meat and poultry during storage due to increasing temperature has been demonstrated by European Food Safety Authority (EFSA Panel on Biological Hazards 2016). In a recent experiment conducted on the optimisation of processing conditions of Chinese traditional smoke-cured bacon (Larou), Liu et al. (2018a) showed that the use of new natural coating solution composed of lysozyme, sodium alginate, and chitosan during storage period resulted in 99.69% rate of reducing deterioration after 30-day storage. Also, the use of High Hydrostatic Pressure (HHP) technology to inactivate pathogenic, spoilage microorganisms and enzymes with little or no effects on the nutritional and sensory quality of foods have been described by Wang et al.

(2016) and Ali et al. (2018) as new advances in processing and packaging fruits, vegetables, meats, seafood, dairy, and egg products.

In summary, there are many practices that can be optimised and scaled up to advance supply-side adaptation. On-farm adaptation options include increased soil organic matter and erosion control in cropland, improved livestock and grazing land management, and transition to different species, polyculture and integrated systems in aquaculture. Crop and livestock genetic improvements include tolerance to heat, drought, and pests and diseases. Food transport, storage, trade, and processing will likely play increasingly important roles in adapting to climate change-induced food insecurity.

5.3.4 Demand-side adaptation

Adaptation in the demand side of the food system involves consumption practices, diets, and reducing food loss and waste. Recent studies showed that supply-side adaptation measures alone will not be sufficient to sustainably achieve food security under climate change (Springmann et al. 2018b; Swinburn et al. 2019; Bajželj et al. 2014). As noted by Godfray (2015), people with higher income demand more varied diets, and typically ones that are richer in meat and other food types that require more resources to produce. Therefore, both supply-side (production, processing, transport, trade, etc.) and demand-side solutions (changing diets, food loss and waste reduction, etc.) can be effective in adapting to climate change (Creutzig et al. 2016) (see Section **Error! Reference source not found.** or food loss and waste).

The implications of dietary choice can have severe consequences for land. For example, Alexander et al. 2016, found that if every country were to adopt the UK's 2011 average diet and meat consumption, 95% of global habitable land area would be needed for agriculture – up from 50% of land currently used. For the average US diet, 178% of global land would be needed (relative to 2011) (Alexander et al. 2016); and for "business as usual" dietary trends and existing rates of improvement in yields, 55% more land would be needed above baseline (2009) (Bajželj et al. 2014). Changing dietary habits has been suggested as an effective food route to affect land use (Beheshti et al. 2017) and promote adaptation to climate change through food demand.

Most literature has focused on demand-side options that analyse the effects on climate change mitigation by dietary changes. Little focus has been brought on demand-side adaptation measures to adjust the demand to the food challenges related to drivers such as market, climate change, inputs limitations (e.g., fossil fuels, nitrogen, phosphorus), food access, and quality. Adding to that, the high cost of nutritious foods contributes to a higher risk of overweight and obesity (FAO 2018d). Adaptation measures relate also to the implications of easy access to inexpensive, high-calorie, low-nutrition foods which have been shown to lead to malnutrition (Section 5.1). Therefore, adaptation related to diet may be weighed against the negative side-effects on health of current food choices.

Reduction in the demand for animal-based food products and increasing proportions of plant-based foods in diets, particularly pulses and nuts; and replacing red meat with other more-efficient protein sources are demand-side adaptation measures (Machovina et al. 2015) (see also Section 5.5.2). For example, replacing beef in the US diet with poultry can meet caloric and protein demands of about 120 to 140 million additional people consuming the average American diet (Shepon et al. 2016). Similar suggestions are made for adopting the benefits of moving to plant-based protein, such as beans (Harwatt et al. 2017).

The main reason why reducing meat consumption is an adaptation measure is because it reduces pressure on land and water and thus our vulnerability to climate change and inputs limitations (Vanham et al. 2013). For animal feed, ruminants can have positive ecological effects (species diversity, soil carbon) if they are fed extensively on existing grasslands. Similarly, reducing waste at all points along the entire food chain is a significant opportunity for improving demand-side adaptation measures (Godfray 2015).

It is important to highlight the opportunities for improving the feed-to-meat conversion considered as a form of food loss. However, the unique capacity of ruminants to produce high-quality food from low-quality forage, in particular from landscapes that cannot be cropped and from cellulosic biomass that humans cannot digest could be seen as an effective way to improve the feed:meat ratio (Cawthorn and Hoffman 2015).

In summary, there is potential for demand-side adaptation, such as adoption of diets low in animal-sourced products, in conjunction with reduction in food loss and waste to contribute to reduction in food demand, land sparing, and thus need for adaptation.

5.3.5 Institutional measures

To facilitate the scaling up of adaptation throughout the food system, institutional measures are needed at global, regional, national, and local levels (See Section 5.7). Institutional aspects including policies and laws depend on scale and context. International institutions (financial and policies) are driving many aspects of global food systems (e.g., UN agencies, international private sector agribusinesses and retailers). Many others operate at local level and strongly influence livelihoods and markets of smallholder farmers. Hence, differentiation in the roles of the organisations, their missions and outcomes related to food and climate change action need to be clearly mapped and understood.

Awareness about the institutional context within which adaptation planning decisions are made is essential for the usability of climate change projection (Lorenz 2017) (Chapter 7 SRCCL). In the planning and operational process of food production, handling and consumption, the environment benefits and climate change goals can be mainstreamed under sustainable management approaches that favor alternative solutions for inputs, energy consumption, transformation and diet. For instance, land use planning would guide current and future decision making and planners in exploring uncertainty to increase the resilience of communities (Berke & Stevens 2016). One of the important policy implications for enhanced food security are the trade-offs between agricultural production and environmental concerns, including the asserted need for global land use expansion, biodiversity and ecological restoration (See Section 5.6) (Meyfroidt 2017).

There are a number of adaptation options in agriculture in the form of policy, planning, governance and institutions (Lorenz 2017). For example, early spatial planning action is crucial to guide decision-making processes and foster resilience in highly uncertain future climate change (Brunner and Grêt-Regamey, 2016). Institutions may develop new capacities to empower value chain actors take climate change into account as they develop quality products, promote adoption of improved diet for healthier lifestyles, aid the improvement of livelihoods of communities, and further socioeconomic development (Sehmi et al. 2016). Other adaptation policies include property rights and land tenure security as legal and institutional reforms to ensure transparency and access to land that could stimulate adaptation to climate change Antwi-Agyei et al. (2015).

5.3.5.1 Global initiatives

Climate change poses serious wide-ranging risks, requiring a broader approach in fighting the phenomenon. The United Nations Framework Convention on Climate Change (UNFCCC) and its annual Conferences of the Parties (COPs) has been instrumental in ensuring international cooperation in the field of tackling the impacts of climate change in a broader framework (Clémençon 2016). The National Adaptation Plan (NAP) program under the UNFCCC, was established to: identify vulnerable regions; assess the impacts of climate change on food security; and prioritise adaptation measures for implementation to increase resilience. The National Adaptation Programs of Action (NAPAs) was also established to support least-developed countries (LDCs) address their particular challenges in adaptation, to enhance food security among other priorities. The Paris Agreement (UNFCCC 2015) is a major victory for small island states and vulnerable nations that face climate change-related impacts of floods and droughts resulting in food security challenges. Adaptation and mitigation targets set by the parties through their nationally determined commitments (NDCs) are reviewed internationally to ensure consistency and progress towards actions (Falkner 2016).

The Food and Agriculture Organization of the United Nations (FAO) also plays a significant role in designing and coordinating national policies to increase adaptation and food security. The five key strategic objectives of FAO (Help eliminate hunger, food insecurity and malnutrition; Make agriculture, forestry and fisheries more productive and sustainable; Reduce rural poverty; Enable inclusive and efficient agricultural and food systems; and increase the resilience of livelihoods to climate threats) (FAO 2018e), all relate to building resilience and increasing global adaptation to climate variability.

In support of the Paris Agreement, FAO launched a global policy, "Tracking Adaptation" with the aim of monitoring the adaptation processes and outcomes of the parties to increase food security, and make available technical information for evaluation by stakeholders. In response to the estimated world population of 9.7 billion by 2050, FAO adopted the Climate Smart Agriculture (CSA) approach to increase global food security without compromising environmental quality (See Section 5.6). FAO supports governments at the national level to plan CSA programs and to seek climate finance to fund their adaptation programs.

The Global Commission on Adaptation, co-managed by World Resources Institute (WRI) and the Global Center on Adaptation, seeks to accelerate adaptation action by elevating the political visibility of adaptation and focusing on concrete solutions (Global Commission on Adaptation 2019). The Commission works to demonstrate that adaptation is a cornerstone of better development, and can help improve lives, reduce poverty, protect the environment, and enhance resilience around the world. The Commission is led by Ban Ki-moon, 8th Secretary-General of the United Nations, Bill Gates, co-chair of the Bill & Melinda Gates Foundation, and Kristalina Georgieva, CEO, World Bank. It is convened by 17 countries and guided by 28 commissioners. A global network of research partners and advisors provide scientific, economic, and policy analysis.

5.3.5.2 National policies

The successful development of food systems under climate change conditions requires a national-level management that involves the cooperation of a number of institutions and governance entities to enable more sustainable and beneficial production and consumption practices.

For example, Nepal has developed a novel multi-level institutional partnership, under the Local Adaptation Plan of Action (LAPA), which is an institutional innovation that aims to better integrate local adaptation planning processes and institutions into national adaptation processes. That includes collaboration with farmers and other non-governmental organisations (Chhetri et al. 2012). By combining conventional technological innovation process with the tacit knowledge of farmers, this new alliance has been instrumental in the innovation of location-specific technologies thereby facilitating the adoption of technologies in a more efficient manner.

National Adaptation Planning of Indonesia was officially launched in 2014 and was an important basis for ministries and local governments to mainstream climate change adaptation into their respective sectoral and local development plans Kawanishi et al. (2016). Crop land use policy to switch from crops that are highly impacted by climate change to those that are less vulnerable were suggested for improving climate change adaptation policy processes and outcomes in Nepal (Chalise and Naranpanawa 2016). Enhancement of representation, democratic and inclusive governance, as well as equity and fairness for improving climate change adaptation policy processes and outcomes in Nepal were also suggested as intuitional measure by Ojha et al. (2015). Further, food, nutrition, and health policy adaptation options such as social safety nets and social protection have been implemented in India, Pakistan, Middle East and North Africa (Devereux 2015; Mumtaz and Whiteford 2017; Narayanan and Gerber 2017).

Financial incentives policies at the national scale used as adaptation options include taxes and subsidies; index-based weather insurance schemes; and catastrophe bonds (Zilberman et al. 2018; Linnerooth-Bayer and Hochrainer-Stigler 2015; Ruiter et al. 2017; Campillo et al. 2017). Microfinance, disaster contingency funds, and cash transfers are other mechanisms (Ozaki 2016; Kabir et al. 2016).

5.3.5.3 Community-based adaptation

Community-based adaptation (CBA) builds on social organisational capacities and resources to addressing food security and climate change. CBA represents bottom-up approaches and localised adaptation measures where social dynamics serve as the power to respond to the impacts of climate change (Ayers and Forsyth 2009). It identifies, assists, and implements development activities that strengthen the capacity of local people to adapt to living in a riskier and less predictable climate, while ensuring their food security.

Klenk et al. (2017) found that mobilisation of local knowledge can inform adaptation decision-making and may facilitate greater flexibility in government-funded research. As an example, rural innovation in terrace agriculture developed on the basis of a local coping mechanism and adopted by peasant farmers in Latin America may serve as an adaptation option to climate change (Bocco and Napoletano, 2017). Clemens et al. (2015) indicated that learning alliances provided social learning and knowledge-sharing in Vietnam through an open dialogue platform that provided incentives and horizontal exchange of ideas.

Community-based adaptation generates strategies through participatory processes, involving local stakeholders and development and disaster risk–reduction practitioners. Fostering collaboration and community stewardship is central to the success of CBA (Scott et al. 2017). Preparedness behaviours that are encouraged include social connectedness, education, training, and messaging; CBA also can encompass beliefs that might improve household preparedness to climate disaster risk (Thomas et al. 2015). Reliance on social networks, social groups connectivities, or moral economies reflect the importance of collaboration within communities (Reuter 2018; Schramski et al. 2017).

Yet, community-based adaptation also needs to consider methods that engage with the drivers of vulnerability as part of community-based approaches, particularly questions of power, culture, identity and practice (Ensor et al. 2018). The goal is to avoid maladaptation or exacerbation of existing inequalities within the communities (Buggy and McNamara 2016). For example, in the Pacific Islands, elements considered in a CBA plan included people’s development aspirations; immediate economic, social and environmental benefits; dynamics of village governance, social rules and protocols; and traditional forms of knowledge that could inform sustainable solutions (Remling and Veitayaki 2016).

With these considerations, community-based adaptation can help to link local adaptation with international development and climate change policies (Forsyth 2013). In developing CBA programs, barriers exist that may hinder implementation. These include poor coordination within and between organisations implementing adaptation options, poor skills, poor knowledge about climate change, and inadequate communication among stakeholders (Spires et al. 2014). A rights-based approach has been suggested to address issues of equality, transparency, accountability and empowerment in adaptation to climate change (Ensor et al. 2015).

In summary, institutional measures, including risk management, policies, and planning at global, national, and local scales can support adaptation. Advance planning and focus on institutions can aid in guiding decision-making processes and foster resilience. There is evidence that institutional measures can support the scaling up of adaptation and thus there is reason to believe that systemic resilience is achievable.

5.3.6 Tools and finance

5.3.6.1 Early Warning Systems

Many countries and regions in the world have adopted early warning systems (EWS) to cope with climate variability and change as it helps to reduce interruptions and improve response times before and after extreme weather events (Ibrahim and Kruczkiewicz 2016). The Early Warning and Early Action (EW/EA) framework has been implemented in West Africa (Red Cross 2011) and Mozambique (DKNC 2012). Bangladesh has constructed cyclone shelters where cyclone warnings are disseminated and responses organised (Mallick et al. 2013). In Benin, a Standard Operating Procedure is used to issue early warnings through the UNDP Climate Information and Early Warning Systems Project (UNDP 2016).

However, there are some barriers to building effective early warning systems in Africa, such as lack of reliable data and distribution systems, lack of credibility, and limited relationships with media and government agencies (UNDP 2016). Mainstreaming early warning systems in adaptation planning could present a significant opportunity for climate disaster risk reduction (Zia and Wagner 2015). Enenkel et al. (2015) suggested that the use of smartphone applications that concentrate on food and nutrition security could help with more frequent and effective monitoring of food prices, availability of fertilisers and drought-resistant seeds, and could help to turn data streams into useful information for decision support and resilience building.

GIS and remote sensing technology are used for monitoring and risk quantification for broad-spectrum stresses such as drought, heat, cold, salinity, flooding, and pests (Skakun et al. 2017; Senay et al. 2015; Hossain et al. 2015; Brown 2016), while site-specific applications, such as drones, for nutrient management, precision fertilisers, and residue management can help devise context-specific adaptations (Campbell et al. 2016; Baker et al. 2016). Systematic monitoring and remote sensing options, as argued by Aghakouchak et al. (2015), showed that satellite observations provide opportunities to improve early drought warning. Waldner et al. (2015) found that cropland mapping allows strategic food and nutrition security monitoring and climate modelling.

Access to a wide range of adaptation technologies for precipitation change is important, such as rainwater harvesting, wastewater treatment, stormwater management and bioswales, water demand reduction, water-use efficiency, water recycling and reuse, aquifer recharge, inter-basin water transfer, desalination, and surface-water storage (ADB 2014).

5.3.6.2 Financial resources

Financial instruments such as micro-insurance, index-based insurance, provision of post-disaster finances for recovery and pre-disaster payment are fundamental means to reduce lower and medium level risks (Linnerooth-Bayer and Hochrainer-Stigler 2014). Fenton & Paavola, 2015; Dowla, 2018). Hammill et al. (2010) found that microfinance services (MFS) are especially helpful for the poor. MFS can provide poor people with the means to diversify, accumulate and manage the assets needed to become less susceptible to shocks and stresses. As a result, MFS plays an important role in vulnerability reduction and climate change adaptation among some of the poor. The provision of small-scale financial products to low-income and otherwise disadvantaged groups by financial institutions can serve as adaptation to climate change. Access to finance in the context of climate change adaptation that focuses on poor households and women in particular is bringing encouraging results (Agrawala and Carraro 2010).

In summary, effective adaptation strategies can reduce the negative impacts of climate change. Food security under changing climate conditions depends on adaptation throughout the entire food system – production, supply chain, and consumption/demand, as well as reduction of food loss and waste. Adaptation can be autonomous, incremental, or transformative, and can reduce vulnerability and enhance resilience. Local food systems are embedded in culture, beliefs and values, and indigenous and local knowledge can contribute to enhancing food system resilience to climate change *(high confidence)*. Institutional and capacity-building measures are needed to scale up adaptation measures across local, national, regional, and global scales.

5.4 Impacts of food systems on climate change

5.4.1 Greenhouse gas emissions from food systems

This chapter assesses the contributions of the entire food system to greenhouse gas (GHG) emissions. Food systems emissions include CO_2 and non-CO_2 gases, specifically those generated from: i) crop and livestock activities within the farm gate (Table 5.4, category 'Agriculture'); ii) land use and land use change dynamics associated with agriculture (Table 5.4, category 'Land Use'); and iii) food processing, retail and consumption patterns, including upstream and downstream processes such as manufacture of chemical fertilisers and fuel (Table 5.4, category 'Beyond Farm Gate'). The first two categories comprise emissions reported by countries in the AFOLU (Agriculture, Forestry, and Other Land Use) sectors of national GHG inventories; the latter comprises emissions reported in other sectors of the inventory, as appropriate, for instance, industrial processes, energy use, and food loss and waste.

The first two components (agriculture and land use) identified above are well quantified and supported by an ample body of literature (Smith et al. 2014). During the period 2007-2016, global agricultural non-CO_2 emissions from crop and livestock activities within the farm gate were 6.2 ± 1.9 Gt CO_2-eq yr^{-1} during 2007-2016, with methane (142 ± 43 Mt CH_4 yr^{-1}, or 4.1 ± 1.2 Gt CO_2-eq yr^{-1}) contributing in CO2eq about twice as much as nitrous oxide (8.3 ± 2.3 Mt N_2O yr^{-1}, or 2.1 ± 0.6 Gt CO_2-eq yr^{-1}) to this total (see Table 2.2 in Chapter 2). Emissions from land use associated with agriculture in some regions, such as from deforestation and peatland degradation (both processes involved in preparing land for agricultural use), added globally during the same period another 4.8 ± 2.4 Gt CO_2-eq yr^{-1} (see Chapter 2). These estimates are associated with uncertainties of about 30% (agriculture) and 50% (land use), as per IPCC AR5 (Smith et al. 2014).

Agriculture activities within the farm gate and associated land use dynamics are therefore responsible for about 11.0 ± 3.1 Gt CO_2-eq yr^{-1}, or some 20% of total anthropogenic emissions (Table 5.4), consistent with post-AR5 findings (e.g., Tubiello et al. (2015)). In terms of individual gases, the contributions of agriculture to total emissions by gas are significantly larger. For instance, over the period 2010-2016, methane gas emissions within the farm gate represented about half of the total CH_4 emitted by all sectors, while nitrous dioxide gas emissions within the farm gate represented about three-quarters of the total N_2O emitted by all sectors (Tubiello 2019). In terms of carbon, CO_2

emissions from deforestation and peatland degradation linked to agriculture contributed about 10% of the CO_2 emitted by all sectors in 2017 (Le Quéré et al. 2018).

Food systems emissions beyond the farm gate, such as those upstream from manufacturing of fertilisers, or downstream such as food processing, transport and retail, and food consumption, generally add to emissions from agriculture and land use, but their estimation is very uncertain due to lack of sufficient studies. The IPCC AR5 (Fischedick et al. 2014) provided some information on these other food system components, noting that emissions beyond the farm gate in developed countries may equal those within the farm gate, and cited one study estimating world total food system emissions to be up to 30% of total anthropogenic emissions (Garnett 2011). More recently, Poore and Nemecek (2018), by looking at a database of farms and using a combination of modelling approaches across relevant processes, estimated a total contribution of food systems around 26% of total anthropogenic emissions. Total emissions from food systems may thus account for 25-30% of total GHG emissions (*medium confidence*).

Based on the available literature, a break-down of individual contributions of food systems emissions is show in Table 5.4, between those from agriculture within the farm gate (10-12%) (*high confidence*); emissions from land use and land use change dynamics such as deforestation and peatland degradation, which are associated with agriculture in many regions (8-10%) (*high confidence*); and those from food supply chain activities past the farm gate, such as storage, processing, transport, and retail (5-10%) *(limited evidence, medium agreement).* Note that the corresponding lower range of emissions past the farm gate, i.e., 2.5 Gt CO_2-eq yr^{-1} (Table 5.4), is consistent with recent estimates made by Poore and Nemecek (2018). Contributions from food loss and waste are implicitly included in these estimates of total emissions from food systems (See Section 5.5.2.5). They may account for 8–10% of total GHG emissions from agriculture and land use (FAO 2013b) (*low confidence*).

Table 5.4 GHG emissions (Gt CO_2eq yr^{-1}) from the food system and their contribution (%) to total anthopogenic emissions. Mean of 2007-2016 period.

Food system component	Emissions (Gt CO_2eq yr^{-1})	Share in mean total emissions (%)
Agriculture	6.2 ± 1.9 [a]	10-12%
Land use	4.8 ± 2.4 [a]	8-10%
Beyond farm gate	3.8 ± 1.3 [b]	5-10%
Food system (Total)	14.8 ± 3.4	25-30%

Notes: Food system emissions are estimated by combining emissions data from a) FAOSTAT (2018) and US EPA (See also Chapter 2) and b) Garnett (2011) and Poore and Nemecek (2018). Percentage shares were computed by using a total emissions value for the period 2007-2016 of nearly 51 Gt CO_2-eq yr^{-1} (See Chapter 2). GWP values used are those , and by using GWP values of the IPCC AR5 with no climate feedback (GWP-CH4=28; GWP-N2O=265)..

5.4.2 Greenhouse gas emissions from croplands and soils

Since AR5, a few studies have quantified separate contributions of crops and soils on the one hand, and livestock on the other, to the total emissions from agriculture and associated land use. For instance, Carlson et al. (2017) estimated emissions from cropland to be in the range of 2–3 $GtCO_2$-eq

yr^{-1}, including methane emissions from rice, CO_2 emissions from peatland cultivation, and N_2O emissions from fertiliser applications. Data from FAOSTAT (2018), recomputed to use AR5 GWP values, indicated that cropland emissions from these categories were 3.6 ± 1.2 Gt CO_2-eq yr^{-1} over the period 2010–2016; two-thirds of this were related to peatland degradation, followed by N_2O emissions from synthetic fertilisers and methane emissions from paddy rice fields (Tubiello 2019). These figures are a subset of the total emissions from agriculture and land use reported in Table 5.4. Asia, especially India, China and Indonesia accounted for roughly 50% of global emissions from croplands. Figure 5.8 shows the spatial distribution of emissions from cropland according to Carlson et al. (2017), not including emissions related to deforestation or changes in soil carbon.

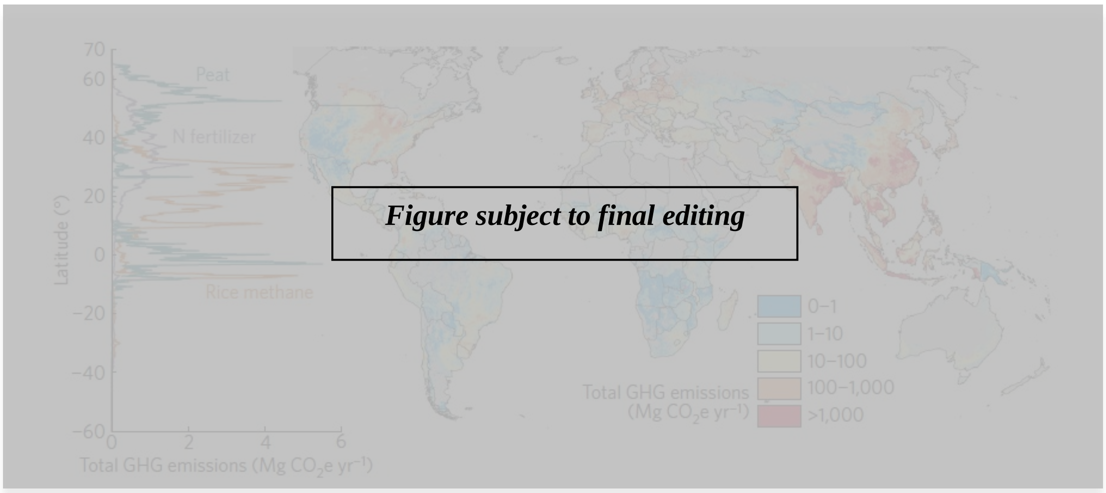


Figure 5.8 Cropland GHGs consist of CH_4 from rice cultivation, CO_2, N_2O, and CH_4 from peatland draining, and N_2O from N fertiliser application. Total emissions from each grid cell are concentrated in Asia, and are distinct from patterns of production intensity (Carlson et al. 2017).

5.4.3 Greenhouse gas emissions from livestock

Emissions from livestock include non-CO_2 gases from enteric fermentation from ruminant animals and from anaerobic fermentation in manure management processes, as well as non-CO_2 gases from manure deposited on pastures (Smith et al. 2014). Estimates after the AR5 include those from Herrero et al. (2016), who quantified non-CO_2 emissions from livestock to be in the range of 2.0–3.6 $GtCO_2$-eq yr^{-1}, with enteric fermentation from ruminants being the main contributor. FAOSTAT (2018) estimates of these emissions, renormalized to AR5 GWP values, were 4.1 ± 1.2 Gt CO_2-eq yr^{-1} over the period 2010–2016.

These estimates of livestock emissions are for those generated within the farm gate. Adding emissions from relevant land use change, energy use, and transportation processes, FAO (2014a) and Gerber et al. (2013) estimated livestock emissions of up to 5.3 ±1.6 $GtCO_2$-eq yr^{-1} circa the year 2010 (data from original papers, but scaled to SAR global warming potential (GWP) values for methane, for comparability with previous results).

All estimates agree that cattle are the main source of global livestock emissions (65–77%). Livestock in low and middle-income countries contribute 70% of the emissions from ruminants and 53% from monogastric livestock (animals without ruminant digestion processes such as sheep, goats, pigs, and poultry), and these are expected to increase as demand for livestock products increases in these countries (Figure 5.9). In contrast to the increasing trend in absolute GHG emissions, GHG emissions intensities, defined as GHG emissions per unit produced, have declined globally and are about 60% lower today than in the 1960s. This is largely due to improved meat and milk productivity of cattle breeds (FAOSTAT 2018; Davis et al. 2015).

Still, products like red meat remain the most inefficient in terms of emissions per kg of protein produced in comparison to milk, pork, eggs and all crop products (IPCC 2014b). Yet, the functional unit used in these measurements is higly relevant and may produce different results (Salou et al. 2017). For instance, metrics based on products tend to rate intensive livestock systems as efficient, while metrics based on area or resources used tend to rate extensive system as efficient (Garnett 2011). In ruminant dairy systems, less intensified farms show higher emissions if expressed by product, and lower emissions if expressed by Utilizable Agricultural Land (Gutiérrez-Peña et al. 2019; Salvador et al. 2017; Salou et al. 2017).

Furthermore, if other variables are used in the analysis of GHG emissions of different ruminant production systems, such as human-edible grains used to feed animals instead of crop waste and pastures of marginal lands, or C sequestration in pasture systems in degraded lands, then the GHG emissions of extensive systems are reduced. Reductions of 26 and 43% have been shown in small ruminants, such as sheep and goats (Gutiérrez-Peña et al. 2019; Salvador et al. 2017; Batalla et al. 2015; Petersen et al. 2013). In this regard, depending on what the main challenge is in different regions (e.g., undernourishment, overconsumption, natural resources degradation), different metrics could be used as reference. Other metrics that consider nutrient density have been proposed because they provide potential for addressing both mitigation and health targets (Doran-Browne et al. 2015).

Uncertainty in worldwide livestock population numbers remain the main source of variation in total emissions of the livestock sector, while at the animal level, feed intake, diet regime, and nutritional composition are the main sources of variation through their impacts on enteric fermentation and manure N excretion.

Increases in economies of scale linked to increased efficiencies and decreased emission intensities may lead to more emissions, rather than less, an observed dynamic referred to by economists as a ‘rebound effect.’ This is because increased efficiency allows production processes to be performed using fewer resources and often at lower cost. This in turn influences consumer behaviour and product use, increasing demand and leading to increased production. In this way, the expected gains from new technologies that increase the efficiency of resource use may be reduced (i.e., increase in the total production of livestock despite increased efficiency of production due to increased demand for meat sold at lower prices). Thus, in order for the livestock sector to provide a contribution to GHG mitigation, reduction in emissions intensities need to be accompanied by appropriate governance and incentive mechanisms to avoid rebound effects, such as limits on total production.

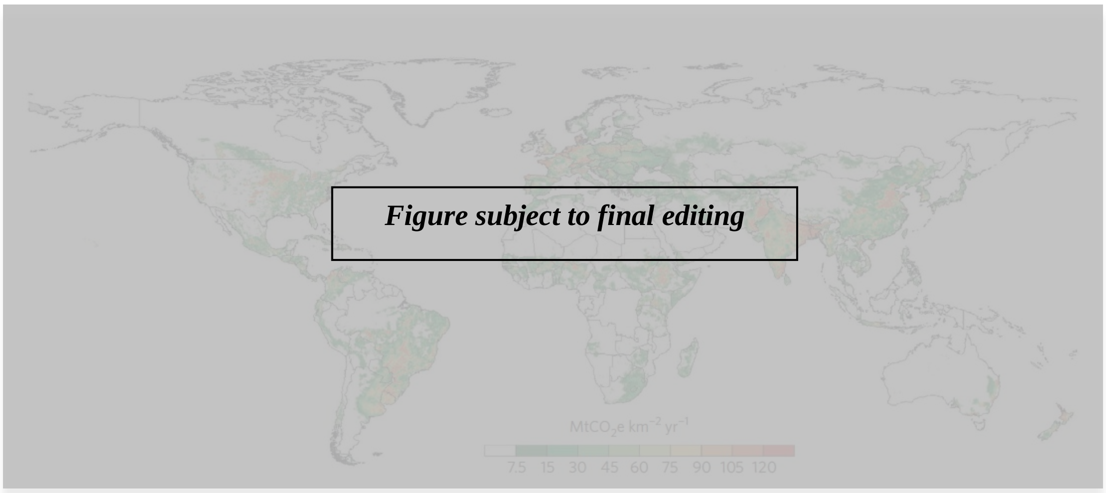


Figure 5.9 Global GHG emissions from livestock for 1995-2005 (Herrero et al. 2016a)

Variation in estimates of N_2O emissions are due to differing a) climate regimes, b) soil types, and c) N transformation pathways (Charles et al. 2017; Fitton et al. 2017). It was recently suggested that

N_2O soil emissions linked to livestock through manure applications could be 20%-40% lower than previously estimated in some regions, for instance in Sub-Saharan Africa and Eastern Europe (Gerber et al. 2016) and from smallholder systems in East Africa (Pelster et al. 2017). Herrero et al. (2016a) estimated global livestock enteric methane to range from 1.6–2.7 Gt CO_2-eq, depending on assumptions of body weight and animal diet.

5.4.4 Greenhouse gas emissions from aquaculture

Emissions from aquaculture and fisheries may represent some 10% of total agriculture emissions, or about 0.58 Gt CO_2-eq yr-1 (Barange et al. 2018), with two-thirds being non-CO_2 emissions from aquaculture (Hu et al. 2013; Yang et al. 2015) and the rest due to fuel use in fishing vessels. They were not included in Table 5.4 under agriculture emissions, as these estimates are not included in national GHG inventories and global numbers are small as well as uncertain.

Methodologies to measure aquaculture emissions are still being developed (Vasanth et al. 2016). N_2O emissions from aquaculture are partly linked to fertiliser use for feed as well as aquatic plant growth, and depend on the temperature of water as well as on fish production (Paudel et al. 2015). Hu et al. (2012) estimated the global N_2O emissions from aquaculture in 2009 to be 0.028 Gt CO_2-eq yr^{-1}, but could increase to 0.114 Gt CO_2-eq yr^{-1} (that is 5.72% of anthropogenic N_2O–N emissions) by 2030 for an estimated 7.10% annual growth rate of the aquaculture industry. Numbers estimated by Williams and Crutzen (2010) were around 0.036 Gt CO_2-eq yr^{-1}, and suggested that this may rise to more than 0.179 Gt CO_2-eq yr^{-1} within 20 years for an estimated annual growth of 8.7%. (Barange et al. 2018) assessed the contribution of aquaculture to climate change as 0.38 Gt CO_2-eq yr^{-1} in 2010, around 7% of those from agriculture.

CO_2 emissions coming from the processing and transport of feed for fish raised in aquaculture, and also the emissions associated with the manufacturing of floating cultivation devices (e.g., rafts or floating fish-farms), connecting or mooring devices, artificial fishing banks or reefs, and feeding devices (as well as their energy consumption) may be considered within the emissions from the food system. Indeed, most of the GHG emissions from aquaculture are associated with the production of raw feed materials and secondarily, with the transport of raw materials to mills and finished feed to farms (Barange et al. 2018).

5.4.5 Greenhouse gas emissions from inputs, processing, storage, and transport

Apart from emissions from agricultural activities within the farm gate, food systems also generate emissions from the pre- and post-production stages in the form of input manufacturing (fertilisers, pesticides, feed production) and processing, storage, refrigeration, retail, waste disposal, food service, and transport. The total contribution of these combined activities outside the farm gate is not well documented. Based on information reported in the AR5 (Fischedick et al. 2014), we estimated their total contribution to be roughly 15% of total anthropogenic emissions (Table 5.4). There is no post-AR5 assessment at the global level in terms of absolute emissions. Rather, several studies have recently investigated how the combined emissions within and outside the farm gate are embedded in food products and thus associated with specific dietary choices (see next section). Below important components of food systems emissions beyond the farm gate are discussed based on recent literature.

Refrigerated trucks, trailers, shipping containers, warehouses, and retail displays that are vital parts of food supply chains all require energy and are direct sources of GHG emissions. Upstream emissions in terms of feed and fertiliser manufacture and downstream emissions (transport, refrigeration) in intensive livestock production (dairy, beef, pork) can account for up to 24–32% of total livestock emissions, with the higher fractions corresponding to commodities produced by monogastric animals

(Weiss and Leip 2012). The proportion of upstream/downstream emissions fall significantly for less-intensive and more-localised production systems (Mottet et al. 2017a).

Transport and processing. Recent globalisation of agriculture has promoted industrial agriculture and encouraged value-added processing and more distant transport of agricultural commodities, all leading to increased GHG emissions. Although greenhouse gas-intensive, food transportation plays an important role in food chains: it delivers food from producers to consumers at various distances, particularly to feed people in food-shortage zones from food-surplus zones. (See Section **Error! eference source not found.** for assessment of local food production.)

To some extent, processing is necessary in order to make food supplies more stable, safe, long-lived, and in some cases, nutritious (FAO 2007). Agricultural production within the farm gate may contribute 80–86% of total food-related emissions in many countries, with emissions from other processes such as processing and transport being small (Vermeulen et al. 2012). However, in net food-importing countries where consumption of processed food is common, emissions from other parts of the food life cycle generated in other locations are much higher (Green et al. 2015).

A study conducted by Wakeland et al. (2012) in the US found that the transportation-related carbon footprint varies from a few percent to more than half of the total carbon footprint associated with food production, distribution, and storage. Most of the GHGs emitted from food processing are a result of the use of electricity, natural gas, coal, diesel, gasoline or other energy sources. Cookers, boilers, and furnaces emit carbon dioxide, and wastewater emits methane and nitrous oxide. The most energy-intensive processing is wet milling of maize, which requires 15% of total US food industry energy (Bernstein et al. 2008); processing of sugar and oils also requires large amounts of energy.

5.4.6 Greenhouse gas emissions associated with different diets

There is now an extensive literature on the relationship between food products and emissions, although the focus of the studies has been on high-income countries. Godfray et al. (2018) updated Nelson et al. (2016), a previous systematic review of the literature on environmental impacts associated with food, and concluded that higher consumption of animal-based foods was associated with higher estimated environmental impacts, whereas increased consumption of plant-based foods was associated with estimated lower environmental impact. Assessment of individual foods within these broader categories showed that meat – sometimes specified as ruminant meat (mainly beef) – was consistently identified as the single food with the greatest impact on the environment, most often in terms of GHG emissions and/or land use per unit commodity. Similar hierarchies, linked to well-known energy losses along trophic chains, from roots to beef were found in another recent review focussing exclusively on GHG emissions (Clune et al. 2017), and one on life-cycle assessments (Poore and Nemecek 2018). Poore and Nemecek (2018) amassed an extensive database that specifies both the hierarchy of emissions intensities and the variance with the production context (i.e., by country and farming system).

The emissions intensities of red meat mean that its production has a disproportionate impact on total emissions (Godfray et al. 2018). For example, in the US 4% of food sold (by weight) is beef, which accounts for 36% of food-related emissions (Heller and Keoleian 2015). Food-related emissions are therefore very sensitive to the amount and type of meat consumed. However, 100 g of beef has twice as much protein as the equivalent in cooked weight of beans, for example, and 2.5 times more iron. One can ingest only about 2.5 kg of food per day and not all food items are as dense in nutrition.

There is therefore *robust evidence with high agreement* that the mixture of foods eaten can have a highly significant impact on per capita carbon emissions, driven particularly through the amount of (especially grain-fed) livestock and products.

Given the rising costs of malnutrition in all its forms, a legitimate question is often asked: would a diet that promotes health through good nutrition also be one that mitigates GHG emissions? Whilst sustainable diets need not necessarily provide more nutrition, there is certainly significant overlap between those that are healthier (e.g., via eating more plant-based material and less livestock-based material), and eating the appropriate level of calories. In their systematic review, Nelson et al. (2016) conclude that, in general, a dietary pattern that is higher in plant-based foods, such as vegetables, fruits, whole grains, legumes, nuts, and seeds, and lower in animal-based foods is more health-promoting and is associated with lesser environmental impact (GHG emissions and energy, land, and water use) than is the current average “meat-based” diet.

Recent FAO projections of food and agriculture to 2050 under alternative scenarios characterised by different degrees of sustainability, provide global-scale evidence that rebalancing diets is key to increasing the overall sustainability of food and agricultural systems world-wide. A 15% reduction of animal products in the diets of high-income countries by 2050 would contribute to containing the need to expand agricultural output due to upward global demographic trends. Not only would GHG emissions and the pressure on land and water be significantly reduced but the potential for low-income countries to increase the intake of animal-based food, with beneficial nutritional outcomes, could be enhanced (FAO 2018a). Given that higher-income countries typically have higher emissions per capita, results are particularly applicable in such places.

However, Springmann et al. (2018a) found that there are locally applicable upper bounds to the footprint of diets around the world, and for lower-income countries undergoing a nutrition transition, adopting “Westernised” consumption patterns (over consumption, large amounts of livestock produce, sugar and fat), even if in culturally applicable local contexts, would increase emissions. The global mitigation potential of healthy but low-emissions diets is discussed in detail in Section **Error! eference source not found.**.

In summary, food system emissions are growing globally due to increasing population, income, and demand for animal-sourced products (*high confidence*). Diets are changing on average toward greater consumption of animal-based foods, vegetable oils and sugar/sweeteners (*high confidence*) (see also Chapter 2), with GHG emissions increasing due to greater amounts of animal-based products in diets (*robust evidence, medium agreement*).

5.5 Mitigation options, challenges, and opportunities

The IPCC AR5 WG III concluded that mitigation in agriculture, forestry, and land use (AFOLU) is key to limit climate change in the 21st century, in terms of mitigation of non-CO_2 GHGs, which are predominately emitted in AFOLU, as well as in terms of land-based carbon sequestration. Wollenberg et al. (2016) highlighted the need to include agricultural emissions explicitly in national mitigation targets and plans, as a necessary strategy to meet the 2°C goal of the Paris Agreement. This chapter expands on these key findings to document how mitigation in the entire food system, from farm ate to consumer, can contribute to reaching the stated global mitigation goals, but in a context of improved food security and nutrition. To put the range of mitigation potential of food systems in context, it is worth noting that emissions from crop and livestock are expected to increase by 30-40% from present to 2050, under business-as-usual scenarios that include efficiency improvements as well as dietary changes linked to increased income per capita (FAO 2018a; Tubiello et al. 2014). Using current emissions estimates in this chapter and Chapter 2, these increases translate into projected GHG emissions from agriculture of 8-9 Gt CO_2eq yr^{-1} by 2050 (*medium confidence*).

The AR5 ranked mitigation measures from simple mechanisms such as improved crop and livestock management (Smith et al. 2014) to more complex carbon dioxide reduction interventions, such as afforestation, soil carbon storage and biomass energy projects with carbon capture and storage (BECCS). The AR5 WGIII AFOLU chapter (Smith et al. 2014) identified two primary categories of mitigation pathways from the food system:

Supply side: Emissions from agricultural soils, land use change, land management, and crop and livestock practices can be reduced and terrestrial carbon stocks can be increased by increased production efficiencies and carbon sequestration in soils and biomass, while emissions from energy use at all stages of the food system can be reduced through improvements in energy efficiency and fossil fuel substitution with carbon-free sources including biomass.

Demand side: GHG emissions could be mitigated by changes in diet, reduction in food loss and waste, and changes in wood consumption for cooking.

In this chapter, supply-side mitigation practices include land use change and carbon sequestration in soils and biomass in both crop and livestock systems. Cropping systems practices include improved land and fertiliser management, land restoration, biochar applications, breeding for larger root systems, and bridging yield gaps (Dooley and Stabinsky 2018). Options for mitigation in livestock systems include better manure management, improved grazing land management, and better feeding practices for animals. Agroforestry also is a supply-side mitigation practice. Improving efficiency in supply chains is a supply-side mitigation measure.

Demand-side mitigation practices include dietary changes that lead to reduction of GHG emissions from production and changes in land use that sequester carbon. Reduction of food loss and waste can contribute to mitigation of GHGs on both the supply and demand sides. See Section 5.7 and Chapter 7 for the enabling conditions needed to ensure that these food system measures would deliver their potential mitigation outcomes.

5.5.1 Supply-side mitigation options

The IPCC AR5 identified options for GHG mitigation in agriculture including cropland management, restoration of organic soils, grazing land management and livestock, with a total mitigation potential of 1.5–4.0 $GtCO_2$-eq yr^{-1} by 2030 (compared to baseline emissions in the same year), at carbon prices from 20 to 100 USD/tCO_2-eq (Smith et al. 2014). Reductions in GHG emissions intensity (emissions per unit product) from livestock and animal products can also be a means to achieve reductions in absolute emissions in specific contexts and with appropriate governance (*medium confidence*). Agroforestry mitigation practices include rotational woodlots, long-term fallow, and integrated land use.

Emissions from food systems can be reduced significantly by the implementation of practices that reduce carbon dioxide, methane, and nitrous oxide emissions from agricultural activities related to the production of crops, livestock, and aquaculture. These include implementation of more sustainable and efficient crop and livestock production practices aimed at reducing the amount of land needed per output (reductions in GHG emissions intensity from livestock and animal production can support reductions in absolute emissions if total production is constrained), bridging yield gaps, implementing better feeding practices for animals and fish in aquaculture, and better manure management (FAO 2019a). Practices that promote soil improvements and carbon sequestration can also play an important role. In the South America region, reduction of deforestation, restoration of degraded pasture areas, and adoption of agroforestry and no-till agricultural techniques play a major role in the nation's voluntary commitments to reduce GHG emissions in the country's mitigation activities (Box 5.4).

The importance of supply-side mitigation options is that these can be directly applied by food system actors (farmers, processors, retailers, etc.) and can contribute to improved livelihoods and income generation. Recognising and empowering farming system actors with the right incentives and governance systems will be crucial to increasing the adoption rates of effective mitigation practices and to build convincing cases for enabling GHG mitigation (Section 5.7 and Chapter 7).

Box 5.4 Towards sustainable intensification in South America region

Reconciling the increasing global food demand with limited land resources and low environmental impact is a major global challenge (FAO 2018a; Godfray and Garnett 2014; Yao et al. 2017). South America has been a significant contributor of the world's agricultural production growth in the last three decades (OECD and FAO 2015), driven partly by increased export opportunities for specific commodities, mainly soybeans and meat (poultry, beef and pork).

Agricultural expansion, however, has driven profound landscape transformations in the region, particularly between the 1970s and early 2000s, contributing to increased deforestation rates and associated GHG emissions. High rates of native vegetation conversion were found in Argentina, Bolivia, Brazil, Colombia, Ecuador, Paraguay and Peru (FAO 2016b; Graesser et al. 2015), threatening ecologically important biomes, such as the Amazon, the savannas (Cerrado, Chacos and Lannos), the Atlantic Rainforest, the Caatinga, and the Yungas. The Amazon biome is a particularly sensitive biome as it provides crucial ecosystem services including biodiversity, hydrological processes (through evapotranspiration, cloud formation, and precipitation), and biogeochemical cycles (including carbon) (Bogaerts et al. 2017; Fearnside 2015; Beuchle et al. 2015; Grecchi et al. 2014; Celentano et al. 2017; Soares-Filho et al. 2014; Nogueira et al. 2018). Further, deforestation associated with commodity exports has not led to inclusive socioeconomic development, but rather has exacerbated social inequality and created more challenging living conditions for lower-income people (Celentano et al. 2017); nor has it avoided increased hunger of local populations in the last few years (FAO 2018b).

In the mid-2000s, governments, food industries, NGOs, and international programs joined forces to put in place important initiatives to respond to the growing concerns about the environmental impacts of agricultural expansion in the region (Negra et al. 2014; Finer et al. 2018). Brazil led regional action by launching the Interministerial Plan of Action for Prevention and Control of Deforestation of the Legal Amazon[2] (PPCDAm), associated with development of a real-time deforestation warning system. Further, Brazil built capacity to respond to alerts by coordinated efforts of ministries, the federal police, the army and public prosecution (Negra et al. 2014; Finer et al. 2018).

Other countries in the region have also launched similar strategies, including a zero-deforestation plan in Paraguay in 2004 (Gasparri and de Waroux 2015), and no-deforestation zones in Argentina in 2007 (Garcia Collazo et al. 2013). Peru also developed the National System of Monitoring and Control, led by the National Forest Service and Wildlife Authority (SERFOR), to provide information and coordinate response to deforestation events, and Colombia started producing quarterly warning reports on active fronts of deforestation in the country (Finer et al. 2018).

Engagement of the food industry and NGOs, particularly through the Soy Moratorium (from 2006) and Beef Moratorium (from 2009) also contributed effectively to keep deforestation at low historical rates in the regions where they were implemented (Nepstad et al. 2014; Gibbs et al. 2015). In 2012, Brazil also created the national land registry system (SICAR), a georeferenced database, which allows monitoring of farms' environmental liability in order to grant access to rural credit. Besides the

[2] FOOTNOTE: The Legal Amazon is a Brazilian region of 501.6 Mha (ca. 59% of the Brazilian territory) that contains all the Amazon but also 40% of the Cerrado and 40% of the Pantanal biomes, with a total population of 25.47 million inhabitants.

governmental schemes, funding agencies and the Amazon Fund provide financial resources to assist smallholder farmers to comply with environmental regulations (Jung et al. 2017).

Nevertheless, Azevedo et al. (2017) argue that the full potential of these financial incentives has not been achieved, due to weak enforcement mechanisms and limited supporting public policies. Agricultural expansion and intensification have complex interactions with deforestation. While mechanisms have been implemented in the region to protect native forests and ecosystems, control of deforestation rates require stronger governance of natural resources (Ceddia et al. 2013; Oliveira and Hecht 2016), including monitoring programs to evaluate fully the results of land use policies in the region.

Public and private sector actions resulted in a reduction of the Brazilian legal Amazon deforestation rate from 2.78 Mha yr^{-1} in 2004, to about 0.75 Mha yr^{-1} (ca. 0.15%) in 2009 (INPE 2015), oscillating from 0.46 Mha and 0.79 Mha (2016) since then (INPE 2018; Boucher and Chi 2018). The governmental forest protection scheme was also expanded to other biomes. As a result, the Brazilian Cerrado deforestation was effectively reduced from 2.9 Mha yr^{-1} in 2004 to an average of 0.71 Mha yr^{-1} in 2016-2017 (INPE 2018).

Overall, deforestation rates in South America have declined significantly, with current deforestation rates being about half of rates in the early 2000s (FAOSTAT 2018). However, inconsistent conservation policies across countries (Gibbs et al. 2015) and recent hiccups (Curtis et al. 2018) indicate that deforestation control still requires stronger reinforcement mechanisms (Tollefson 2018). Further, there are important spill-over effects that need coordinated international governance. Curtis et al. (2018) and Dou et al. (2018) point out that, although the Amazon deforestation rate decreased in Brazil, it has increased in other regions, particularly in Southern Asia, and in other countries in South America, resulting in nearly constant deforestation rates worldwide.

Despite the reduced expansion rates into forest land, agricultural production continues to rise steadily in South America, relying on increasing productivity and substitution of extensive pastureland by crops. The average soybean and maize productivity in the region increased from 1.8 and 2.0 t ha^{-1} in 1990 to 3.0 and 5.0 t ha^{-1}, respectively, in 2015 (FAOSTAT 2018). Yet, higher crop productivity was not enough to meet growing demand for cereals and oilseeds and cultivation continued to expand, mainly on grasslands (Richards 2015). The reconciliation of this expansion with higher demand for meat and dairy products was carried out through the intensification of livestock systems (Martha et al. 2012). Nevertheless, direct and indirect deforestation still occurs, and recently deforestation rates have increased (INPE 2018), albeit they remain far smaller than observed in the 2000-2010 period.

The effort towards sustainable intensification has also been incorporated in agricultural policies. In Brazil, for instance, the reduction of deforestation, the restoration of degraded pasture areas, the adoption of integrated agroforestry systems[3] and no-till agricultural techniques play a major role in the nation's voluntary commitments to reduce GHG emissions in the country's NAMAs (Mozzer 2011) and NDCs (Silva Oliveira et al. 2017; Rochedo et al. 2018). Such commitment under the UNFCCC is operationalised through the Low Carbon Agriculture Plan (ABC)[4], which is based on low interest credit for investment in sustainable agricultural technologies (Mozzer 2011). Direct pasture restoration and integrated systems reduce area requirements (Strassburg et al. 2014), and increase organic matter (Gil et al. 2015; Bungenstab 2012; Maia et al. 2009), contributing to overall life cycle

[3] FOOTNOTE: Integrated agroforestry systems are agricultural systems that strategically integrate two or more components among crops, livestock and forestry. The activities can be in consortium, succession or rotation in order to achieve overall synergy.

[4] FOOTNOTE: ABC - *Agricultura de Baixo Carbono* in Portuguese.

emissions reduction (Cardoso et al. 2016; de Oliveira Silva et al. 2016). Also, increased adoption of supplementation and feedlots, often based on agro-industrial co-products and agricultural crop residues are central to improve productivity and increase climate resilience of livestock systems (Mottet et al. 2017a; van Zanten et al. 2018).

Despite providing clear environmental and socio-economic co-benefits, including improved resource productivity, socio-environmental sustainability and higher economic competitiveness, implementation of the Brazilian Low Carbon Agriculture Plan is behind schedule (Köberle et al. 2016). Structural inefficiencies related to the allocation and distribution of resources need to be addressed to put the plan on track to meet its emissions reduction targets. Monitoring and verification are fundamental tools to guarantee the successful implementation of the plan.

Overall, historical data and projections show that South America is one of the regions of the world with the highest potential to increase crop and livestock production in the coming decades in a sustainable manner (Cohn et al. 2014), increasing food supply to more densely populated regions in Asia, Middle East and Europe. However, a great and coordinated effort is required from governments, industry, traders, scientists and the international community to improve planning, monitoring and innovation to guarantee sustainable intensification of its agricultural systems, contribution to GHG mitigation, and conservation of the surrounding environment (Negra et al. 2014; Curtis et al. 2018; Lambin et al. 2018).

5.5.1.1 Greenhouse gas mitigation in croplands and soils

The mitigation potential of agricultural soils, cropland and grazing land management has been the subject of much research and was thoroughly summarised in the AR5 (Smith et al. 2014) (See also Chapter 2 Section 2.5.1 and Chapter 6 Section 6.3.1). Key mitigation pathways are related to practices reducing nitrous oxide emissions from fertiliser applications, reducing methane emissions from paddy rice, reducing both gases through livestock manure management and applications, and sequestering carbon or reducing its losses, with practices for improving grassland and cropland management identified as the largest mitigation opportunities. Better monitoring reporting and verification (MRV) systems are currently needed for reducing uncertainties and better quantifying the actual mitigation outcomes of these activities.

New work since AR5 has focused on identifying pathways for the reductions of GHG emissions from agriculture to help meet Paris Agreement goals (Paustian et al. 2016; Wollenberg et al. 2016). Altieri and Nicholls (2017) have characterised mitigation potentials from traditional agriculture. Zomer et al. (2017) have updated previous estimates of global carbon sequestration potential in cropland soils. Mayer et al. (2018) converted soil carbon sequestration potential through agricultural land management into avoided temperature reductions. Fujisaki et al. (2018) identify drivers to increase soil organic carbon in tropical soils. For discussion of integrated practices such as sustainable intensification, conservation agriculture and agroecology, see Section 5.6.4.

Paustian et al. (2016) developed a decision-tree for facilitating implementation of mitigation practices on cropland and described the features of key practices. They observed that most individual mitigation practices will have a small effect per unit of land, and hence they need to be combined and applied at large scales for their impact to be significant. Examples included aggregation of cropland practices (e.g., organic amendments, improved crop rotations and nutrient management and reduced tillage) and grazing land practices (e.g., grazing management, nutrient and fire management and species introduction) that could increase net soil C stocks while reducing emissions of N_2O and CH_4. However, it is well-known that the portion of projected mitigation from soil C stock increase (about 90% of the total technical potential) is impermanent, i.e., it would be effective for only 20–30 years due to saturation of the soil capacity to sequester carbon, whereas non-CO_2 emission reductions could continue indefinitely. “Technical potential” is the maximum amount of GHG mitigation achievable through technology diffusion.

Biochar application and management towards enhanced root systems are mitigation options that have been highlighted in recent literature (Dooley and Stabinsky 2018; Hawken 2017; Paustian et al. 2016; Woolf et al. 2010; Lenton 2010).

5.5.1.2 Greenhouse gas mitigation in livestock systems

The technical options for mitigating GHG emissions in the livestock sector have been the subject of recent reviews (Mottet et al., 2017b; Hristov et al. 2013a,b; Smithers 2015; Herrero et al. 2016a; Rivera-Ferre et al. 2016b) (Figure 5.11). They can be classified as either targeting reductions in enteric methane; reductions in nitrous oxide through manure management; sequestering carbon in pastures; implementation of best animal husbandry and management practices, which would have an effect on most GHG; and land use practices that also help sequester carbon. Excluding land use practices, these options have a technical mitigation potential ranging 0.2-2.4 $GtCO_2$-eq yr^{-1} (Herrero et al. 2016a; FAO 2007). See also Chapters 2 and 6 in this report.

The opportunities for carbon sequestration in grasslands and rangelands may be significant (Conant 2010), for instance, through changes in grazing intensity or manure recycling aimed at maintaining grassland productivity (Hirata et al. 2013). Recent studies have questioned the economic potential of such practices, i.e., whether they could implement at scale for economic gain (Garnett et al. 2017; Herrero et al. 2016a; Henderson et al. 2015). For instance, Henderson et al. (2015) found economic potentials below 200 $MtCO_2$-eq yr^{-1}. Carbon sequestration can occur in situations where grasslands are highly degraded (Garnett 2016). Carbon sequestration linked to livestock management could thus be considered as a co-benefit of well-managed grasslands, as well as a mitigation practice.

Different production systems will require different strategies, including the assessment of impacts on food security, and this has been the subject of significant research (e.g., Rivera-Ferre et al. 2016b). Livestock systems are heterogeneous in terms of their agro-ecological orientation (arid, humid or temperate/highland locations), livestock species (cattle, sheep, goats, pigs, poultry and others), structure (grazing only, mixed-crop-livestock systems, industrial systems, feedlots and others), level of intensification, and resource endowment (Robinson 2011).

The implementation of strategies presented in **Error! Reference source not found.** builds on this ifferentiation, providing more depth compared to the previous AR5 analysis. Manure management strategies are more applicable in confined systems, where manure can be easily collected, such as in pigs and poultry systems or in smallholder mixed crop–livestock systems. More intensive systems, with strong market orientation, such as dairy in the US, can implement a range of sophisticated practices like feed additives and vaccines, while many market-oriented dairy systems in tropical regions can improve feed digestibility by improving forage quality and adding larger quantities of concentrate to the rations. Many of these strategies can be implemented as packages in different systems, thus maximising the synergies between different options (Mottet et al. 2017b).

See the Supplementary Material Section SM5.5 for detailed description of livestock mitigation strategies; synergies and trade-offs with other mitigation and adaptation options are discussed in Section 5.6.

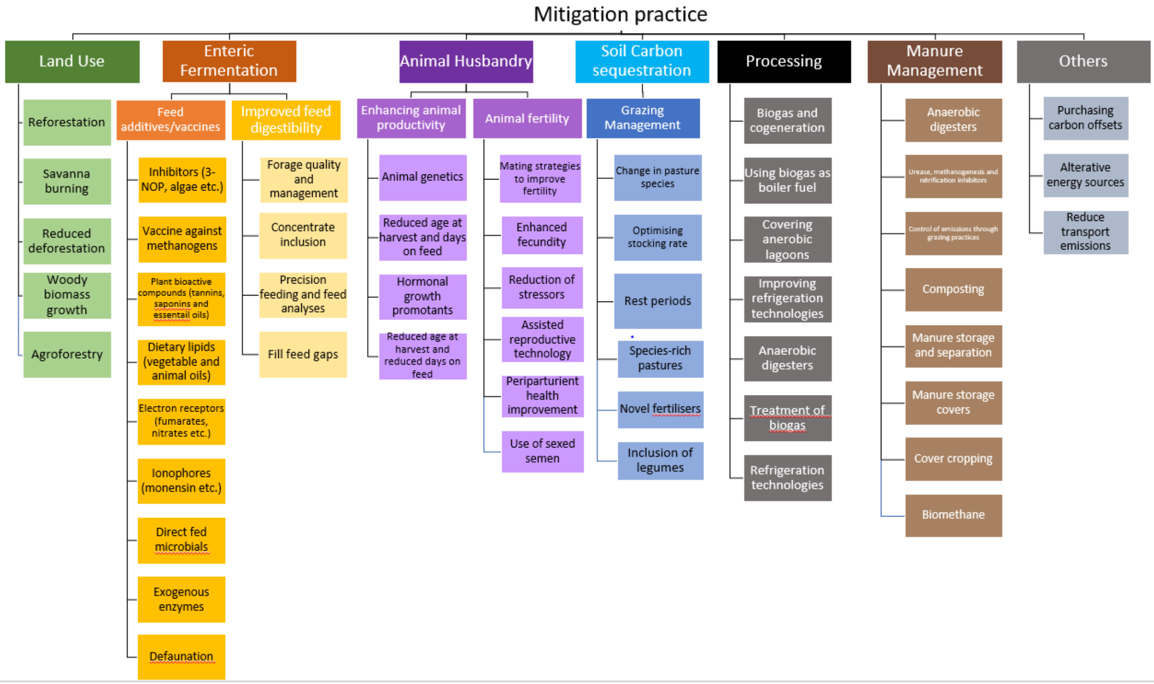


Figure 5.10 Technical supply-side mitigation practices in the livestock sector (adapted from Hristov et al. 2013b; Herrero et al. 2016b; Smith et al. 2014)

5.5.1.3 Greenhouse gas mitigation in agroforestry

Agroforestry can curb GHG emissions of CO_2, CH_4, and N_2O in agricultural systems in both developed and developing countries (See Glossary for definition) (see Chapter 2, Section 2.5.1 and Figure 2.24). Soil carbon sequestration, together with biological N fixation, improved land health and underlying ecosystem services may be enhanced through agricultural lands management practices used by large-scale and smallholder farmers, such as incorporation of trees within farms or in hedges (manure addition, green manures, cover crops, etc.), whilst promoting greater soil organic matter and nutrients (and thus soil organic carbon) content and improve soil structure (Mbow et al. 2014b) (Table 5.5). The tree cover increases the microbial activity of the soil and increases the productivity of the grass under cover. CO_2 emissions are furthermore lessened indirectly, through lower rates of erosion due to better soil structure and more plant cover in diversified farming systems than in monocultures. There is great potential for increasing above ground and soil C stocks, reducing soil erosion and degradation, and mitigating GHG emissions.

These practices can improve food security through increases in productivity and stability since they contribute to increased soil quality and water-holding capacity. Agroforestry provides economic, ecological, and social stability through diversification of species and products. On the other hand, trade-offs are possible when cropland is taken out of production mainly as a mitigation strategy.

Table 5.5 Carbon sequestration potential for agroforestry (Mbow et al. 2014b)

Source	**Carbon sequestration (tCO_2 km^{-2} yr^{-1}) (range)**	**C stock (tCO_2 km^{-2}) (range)**	**Max rotation period (years)**
Dominant parklands	183 (73–293)	12,257 (2,091–25,983)	50
Rotational woodlots[a]	1,431 (807–2,128)	6,789 (4,257–9,358)	5

Tree planting-windrows-home gardens	220.2 (146–293)	6,973 (-)	25
Long term fallows, regrowth of woodlands in abandoned farms[b]	822 (80–2,128)	5,761 (-)	25
Integrated land use	1,145 (367–2,458)	28,589 (4,404–83,676)	50
Soil carbon	330 (91–587)	33,286 (4,771–110,100)	-

[a] May be classified as forestry on forest land, depending on the spatial and temporal characteristics of these activities.
[b] This is potentially not agro-forestry, but forestry following abandonment of agricultural land.

Meta-analyses have been done on carbon budgets in agroforestry systems (Zomer et al. 2016; Chatterjee et al. 2018). In a review of 42 studies, (Ramachandran Nair et al. 2009) estimated C sequestration potentials of differing agroforestry systems. These include sequestration rates in ranging from 954 (semi-arid); to 1,431 (temperate); 2,238 (sub-humid) and 3,670 tCO_2 km^{-2} yr^{-1} (humid). The global technical potential for agroforestry is 0.1–5.7 Gt CO2e yr-1 (Griscom et al. 2017; Zomer et al. 2016; Dickie et al. 2014) (see Chapter 2, Section 2.5.1). Agroforestry-based carbon sequestration can be used to offset N_2O and CO_2 emissions from soils and increase methane sink strength compared to annual cropping systems (Rosenstock et al. 2014).

Agroforestry systems with perennial crops, such as coffee and cacao, may be more important carbon sinks than those that combine trees with annual crops. Brandt et al. (2018) showed that farms in semi-arid region (300–600 mm precipitation) were increasing in tree cover due to natural regeneration and that the increased application of agroforestry systems were supporting production and reducing GHG emissions.

5.5.1.4 Integrated approaches to crop and livestock mitigation

Livestock mitigation in a circular economy. Novel technologies for increasing the integration of components in the food system are being devised to reduce GHG emissions. These include strategies that help decoupling livestock from land use. Work by van Zanten et al. (2018) shows that 7–23 g of animal protein per capita per day could be produced without livestock competing for vital arable land. This would imply a contraction of in the land area utilised by the livestock sector, but also a more efficient use of resources, and would lead to land sparing and overall emissions reductions.

Pikaar et al. (2018) demonstrated the technical feasibility of producing microbial protein as a feedstuff from sewage that could replace use of feed crops such as soybean. The technical potential of this novel practice could replace 10–19% of the feed protein required, and would reduce cropland demand and associated emissions by 6-7%. These practices are, however, not economically feasible nor easily upscalable in most systems. Nonetheless, significant progress in Japan and South Korea in the reduction and use of food waste to increase efficiencies in livestock food chains has been achieved, indicating a possible pathway to progress elsewhere (FAO 2017; zu Ermgassen et al. 2016). Better understanding of biomass and food and feed wastes, value chains, and identification of mechanisms for reducing the transport and processing costs of these materials is required to facilitate larger-scale implementation.

Waste streams into energy. Waste streams from manure and food waste can be used for energy generation and thus reduction in overall GHG emissions in terms of recovered methane (for instance through anaerobic digestion) production (De Clercq et al. 2016) or for the production of microbial protein (Pikaar et al. 2018). Second-generation biorefineries, once the underlying technology is improved, may enable the generation of hydro-carbon from agricultural residues, grass, and woody biomass in ways that do not compete with food and can generate, along with biofuel, high-value products such as plastics (Nguyen et al. 2017). Second-generation energy biomass from residues may constitute a complementary income source for farmers that can increase their incentive to produce. Technologies include CHP (combined heat and power) or gas turbines, and fuel types such as bio-diesel, bio-pyrolysis (i.e., high temperature chemical transformation of organic material in the absence of oxygen), torrefaction of biomass, production of cellulosic bio-ethanol and of bio-alcohols produced by other means than fermentation, and the production of methane by anaerobic fermentation. (Nguyen et al. 2017).

Technology for reducing fossil fuel inputs. Besides biomass and bioenergy, other forms of renewable energy substitution for fossil fuels (e.g., wind, solar, geothermal, hydro) are already being applied on farms and throughout the supply chain. Energy efficiency measures are being developed for refrigeration, conservation tillage, precision farming (e.g., fertiliser and chemical application and precision irrigation.

Novel technologies. Measures that can reduce livestock emissions given continued research and development include methane and nitrification inhibitors, methane vaccines, targeted breeding of lower-emitting animals, and genetically modified grasses with higher sugar content. New strategies to reduce methanogenesis include supplementing animal diets with antimethanogenic agents (e.g., 3-NOP, algae, chemical inhibitors such as chloroform) or supplementing with electron acceptors (e.g., nitrate) or dietary lipids. These could potentially contribute, once economically feasible at scale, to significant reductions of methane emissions from ruminant livestock. A well-tested compound is 3-nitrooxypropanol (3-NOP), which was shown to decrease methane by up to 40% when incorporated in diets for ruminants (Hristov et al. 2015).

Whilst these strategies may become very effective at reducing methane, they can be expensive and also impact on animal performance and/or welfare (Llonch et al. 2017). The use of novel fertilisers and/or plant species that secrete biological nitrification inhibitors also have the potential to significantly reduce N_2O emissions from agricultural soils (Subbarao et al. 2009; Rose et al. 2018).

Economic mitigation potentials of crop and livestock sectors. Despite the large technical mitigation potential of the agriculture sector in terms of crop and livestock activities, its economic potential is relatively small in the short term (2030) and at modest carbon prices (less than USD 20 tC^{-1}). For crop and soil management practices, it is estimated that 1.0–1.5 $GtCO_2$-eq yr^{-1} could be a feasible mitigation target at a carbon price of USD 20/tonne of carbon (Frank et al. 2018, 2017; Griscom et al. 2016; Smith et al. 2013; Wollenberg et al. 2016). For the livestock sector, these estimates range from 0.12–0.25 $GtCO_2$-eq yr^{-1} at similar carbon prices (Herrero et al. 2016c; Henderson et al. 2017). But care in needed in comparing crop and livestock economic mitigation potentials due to differing assumptions.

Frank et al. (2018) recently estimated that the economic mitigation potential of non-CO_2 emissions from agriculture and livestock to 2030 could be up to four times higher than indicated in the AR5, if structural options such as switching livestock species from ruminants to monogastrics, or allowing for flexibility to relocate production to more efficient regions were implemented, at the same time as the technical options such as those described above. At higher carbon prices (i.e., at about USD 100tC^{-1}), they found a mitigation potential of supply-side measures of 2.6 $GtCO_2$-eq yr^{-1}.

In this scenario, technical options would account for 38% of the abatement, while another 38% would be obtained through structural changes, and a further 24% would be obtained through shifts in consumption caused by food price increases. Key to the achievement of this mitigation potential lay in the livestock sector, as reductions in livestock consumption, structural changes and implementation of technologies in the sector had some of the highest impacts. Regions with the highest mitigation potentials were Latin America, China and Sub-Saharan Africa. The large-scale implementability of such proposed sweeping changes in livestock types and production systems is likely very limited as well as constrained by long-established socio-economic, traditional and cultural habits, requiring significant incentives to generate change.

In summary, supply-side practices can contribute to climate change mitigation by reducing crop and livestock emissions, sequestering carbon in soils and biomass, and by decreasing emissions intensity within sustainable production systems *(high confidence)*. The AR5 estimated the total economic mitigation potential of crop and livestock activities as 1.5–4.0 $GtCO_2$-eq yr^{-1} by 2030 at prices ranging from 20-100 USD/tCO_2eq *(high confidence)*. Options with large potential for GHG mitigation in cropping systems include soil carbon sequestration (at decreasing rates over time), reductions in N_2O emissions from fertilisers, reductions in CH_4 emissions from paddy rice, and bridging of yield gaps. Options with large potential for mitigation in livestock systems include better grazing land management, with increased net primary production and soil carbon stocks, improved manure management, and higher-quality feed. Reductions in GHG emissions intensity (emissions per unit product) from livestock can support reductions in absolute emissions, provided appropriate governance structures to limit total production isare implemented at the same time *(medium confidence)*.

5.5.1.5 Greenhouse gas mitigation in aquaculture

Barange et al. (2018) provide a synthesis of effective options for GHG emissions reduction in aquaculture including reduction of emissions from production of feed material, replacement of fish-based feed ingredients with crop-based ingredients; reduction of emissions from feed mill energy use, improvement of feed conversion rates, improvement of input use efficiency, shift of energy supply (from high-carbon fossil fuels to low-carbon fossil fuels or renewables), and improvement of fish health. Conversion of 25% of total aquaculture area to integrated aquaculture-agriculture ponds (greening aquaculture) has the potential to sequester 95.4 millions tonnes carbon per year (Ahmed et al. 2017).

Proposed mitigation in aquaculture includes avoided deforestation. By halting annual mangrove deforestation in Indonesia, associated total emissions would be reduced 10-31% of estimated annual emissions from land use sector at present (Murdiyarso et al. 2015). Globally, 25% mangrove regeneration could sequester 0.54–0.65 millions tonnes carbon per year (Ahmed et al. 2017) of which 0.17-0.21 could be through integrated or organic shrimp culture (Ahmed et al. 2018).

5.5.1.6 Cellular agriculture

The technology for growing muscle tissue in culture from animal stem cells to produce meat, i.e., “cultured” or “synthetic” or “in vitro” or “hydroponic” meat could in theory be constructed with different characteristics and be produced faster and more efficiently than traditional meat (Kadim et al. 2015). Cultured meat (CM) is part of so-called cellular agriculture, which includes production of milk, egg white and leather from industrial cell cultivation (Stephens et al. 2018). CM is produced from muscle cells extracted from living animals, isolation of adult skeletal muscle stem cells (myosatellite cells), placement in a culture medium which allow their differentiation into myoblasts and then, through another medium, generation of myocytes which coalesce into myotubes and grow into strands in a stirred-tank bioreactor (Mattick et al. 2015). Current technology enables the creation of beef hamburgers, nuggets, steak chips or similar products from meat of other animals, including wild species, although production currently is far from being economically feasible. Nonetheless, by allowing bioengineering from the manipulation of the stem cells and nutritive culture, CM allows for reduction of harmful fatty acids, with advantages such as reduced GHG emissions, mostly indirectly through reduced land use (Bhat et al. 2015; Kumar et al. 2017b).

Tuomisto and de Mattos (2011) made optimistic technological assumptions, relying on cyanobacteria hydrolysate nutrient source, and produced the lowest estimates on energy and land use. Tuomisto and de Mattos (2011) conducted a lifecycle assessment that indicates that cultured meat could have less than 60% of energy use and 1% of land use of beef production and it would have lower GHG emissions than pork and poultry as well. Newer estimates (Alexander et al. 2017; Mattick et al. 2015) indicate a trade-off between industrial energy consumption and agricultural land requirements of conventional and cultured meat and possibly higher GWP than pork or poultry due to higher energy use. The change in proportion of CO_2 vs CH_4 could have important implications in climate change projections and, depending on decarbonisation of the energy sources and climate change targets, cultured meat may be even more detrimental than exclusive beef production (Lynch 2019).

Overall, as argued by Stephens et al. (2018), cultured meat is an "as-yet undefined ontological object" and, although marketing targets people who appreciate meat but are concerned with animal welfare and environmental impacts, its market is largely unknown (Bhat et al. 2015; Slade 2018). In this context it will face the competition of imitation meat (meat analogues from vegetal protein) and insect-derived products, which have been evaluated as more environmentally friendly (Alexander et al. 2017) and it may be considered as being an option for a limited resource world, rather than a mainstream solution. Besides, as commercial production process is still largely undefined, its actual contribution to climate change mitigation and food security is largely uncertain and challenges are not negligible. Finally, it is important to understand the systemic nature of these challenges and evaluate their social impacts on rural populations due to transforming animal agriculture into an industrialised activity and its possible rebound effects on food security, which are still understudied in the literature.

Studies are needed to improve quantification of mitigation options for supply chain activities.

5.5.2 Demand-side mitigation options

Although population growth is one of the drivers of global food demand and the resulting environmental burden, demand-side management of the food system could be one of the solutions to curb climate change. Avoiding food waste during consumption, reducing over-consumption, and changing dietary preferences can contribute significantly to provide healthy diets for all, as well as reduce the environmental footprint of the food system. The number of studies addressing this issue have increased in the last few years (see also Chapter 2). (See Section 5.6 for synergies and trade-offs with health and Section 5.7 for discussion of Just Transitions).

5.5.2.1 Mitigation potential of different diets

A systematic review found that higher consumption of animal-based foods was associated with higher estimated environmental impact, whereas increased consumption of plant-based foods was associated with an estimated lower environmental impact (Nelson et al. 2016). Assessment of individual foods within these broader categories showed that meat – especially ruminant meat (beef and lamb) – was consistently identified as the single food with the greatest impact on the environment, on a global basis, most often in terms of GHG emissions and/or land use.

Figure 5.12 shows the technical mitigation potentials of some scenarios of alternative diets examined in the literature. Stehfest et al. (2009) were among the first to examine these questions. They found that under the most extreme scenario, where no animal products are consumed at all, adequate food production in 2050 could be achieved on less land than is currently used, allowing considerable forest regeneration, and reducing land-based greenhouse gas emissions to one third of the reference "business-as-usual" case for 2050, a reduction of 7.8 Gt CO_2-eq yr^{-1}. Springmann et al. (2016b) recently estimated similar emissions reduction potential of 8 Gt CO_2-eq yr^{-1} from a vegan diet without animal-sourced foods. This defines the upper bound of the technical mitigation potential of demand side measures.

Herrero et al. (2016a) reviewed available options, with a specific focus on livestock products, assessing technical mitigation potential across a range of scenarios, including "No animal products", "No meat", "No ruminant meat", and "Healthy diet" (reduced meat consumption). With regard to 'credible low-meat diets,' where reduction in animal protein intake was compensated by higher intake of pulses, emissions reductions by 2050 could be in the 4.3–6.4 Gt CO_2-eq yr^{-1}, compared to a business-as-usual scenario. Of this technical potential, 1–2 $GtCO_2$-eq yr^{-1} come from reductions of mostly non-CO_2 GHG within the farm gate, while the remainder was linked to carbon sequestration on agricultural lands no longer needed for livestock production. When the transition to a low-meat diet reduces the agricultural area required, land is abandoned and the re-growing vegetation can take up carbon until a new equilibrium is reached. This is known as the land-sparing effect.

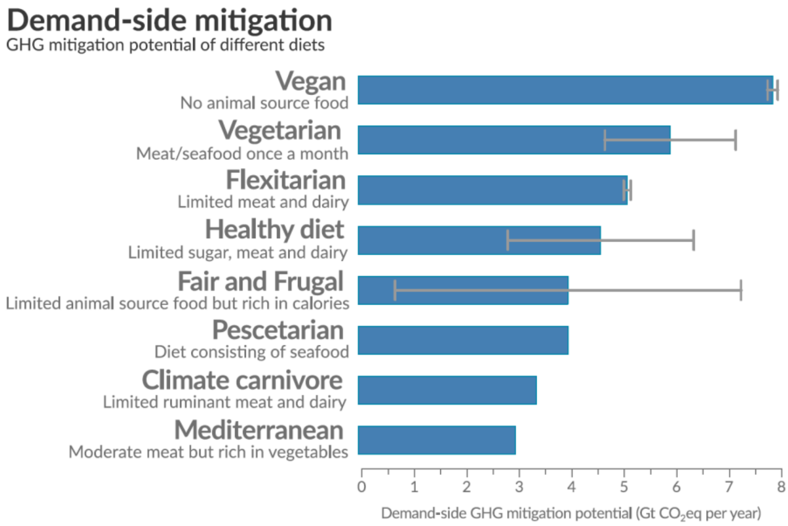


Figure 5.12 Technical mitigation potential of changing diets by 2050 according to a range of scenarios examined in the literature. Estimates are technical potential only, and include additional effects of carbon sequestration from land-sparing. Data without error bars are from one study only.

All diets need to provide a full complement of nutritional quality, including micronutrients (FAO et al. 2018)
Vegan: Completely plant-based (Springmann et al. 2016; Stehfest et al. 2009)
Vegetarian: Grains, vegetables, fruits, sugars, oils, eggs and dairy, and generally at most one serving per month of meat or seafood (Springmann et al. 2016; Tilman and Clark 2014; Stehfest et al. 2009)
Flexitarian: 75% of meat and dairy replaced by cereals and pulses; at least 500 g per day fruits and vegetables; at least 100 g per day of plant-based protein sources; modest amounts of animal-based proteins and limited amounts of red meat (one portion per week), refined sugar (less than 5% of total energy), vegetable oils high in saturated fat, and starchy foods with relatively high glycaemic index (Springmann et al. 2018; Hedenus et al. 2014)
Healthy diet: Based on global dietary guidelines for consumption of red meat, sugar, fruits and vegetables, and total energy intake (Springmann et al. 2018; Bajželj et al. 2014)
Fair and Frugal: Global daily per-capita calorie intake of 2800 kcal/cap/day (11.7 MJ/cap/day), paired with relatively low level of animal products (Smith et al. 2013)
Pescetarian: Vegetarian diet that includes seafood (Tilman and Clark 2014)
Climate carnivore: 75% of ruminant meat and dairy replaced by other meat (Hedenus et al. 2014)
Mediterranean: Vegetables, fruits, grains, sugars, oils, eggs, dairy, seafood, moderate amounts of poultry, pork, lamb and beef (Tilman and Clark 2014)

Other studies have found similar results for potential mitigation linked to diets. For instance, Smith et al. (2013) analysed a dietary change scenario that assumed a convergence towards a global daily per-capita calorie intake of 2800 kcal cap^{-1} day^{-1} (11.7 MJ cap^{-1} day^{-1}), paired with a relatively low level of animal product supply, estimated technical mitigation potential in the range 0.7–7.3 Gt CO_2-eq yr^{-1} for additional variants including low or high-yielding bioenergy, 4.6 Gt CO_2-eq yr^{-1} if spare land is afforested.

Bajželj et al. (2014) developed different scenarios of farm systems change, waste management, and dietary change on GHG emissions coupled to land use. Their dietary scenarios were based on target kilocalorie consumption levels and reductions in animal product consumption. Their scenarios were "Healthy Diet"; Healthy Diet with 2500 kcal cap^{-1} day^{-1} in 2050; corresponded to technical mitigation potentials in the range 5.8 and 6.4 Gt CO_2-eq yr^{-1}.

Hedenus et al. (2014) explored further dietary variants based on the type of livestock product. "Climate Carnivore", in which 75% of the baseline-consumption of ruminant meat and dairy was replaced by pork and poultry meat, and "Flexitarian", in which 75% of the baseline-consumption of meat and dairy was replaced by pulses and cereal products. Their estimates of technical mitigation potentials by 2050 ranged 3.4- 5.2 Gt CO_2-eq yr^{-1}, the high end achieved under the Flexitarian. Finally, Tilman and Clark (2014) used stylised diets as variants that included "Pescetarian", "Mediterranean", "Vegetarian", compared to a reference diet, and estimated technical mitigation potentials within the farm gate of 1.2-2.3 Gt CO_2-eq yr^{-1}, with additional mitigation from carbon sequestration on spared land ranging 1.8–2.4 Gt CO_2-eq yr^{-1}.

Studies have defined dietary mitigation potential as, for example, 20kg per person per week CO_2-eq for Mediterranean diet, vs 13kg per person per week CO_2-eq for vegan (Castañé and Antón 2017). Rosi et al. (2017) developed seven-day diets in Italy for about 150 people defined as omnivore 4.0 ± 1.0; ovo-lacto-veggie 2.6 ± 0.6; and vegan 2.3 ± 0.5 kg CO_2-eq per capita per day.

Importantly, many more studies that compute the economic and calorie costs of these scenarios are needed. Herrero et al. (2016a) estimated that once considerations of economic and calorie costs of their diet-based solutions were included, the technical range of 4.3-6.4 Gt CO_2-eq yr^{-1} in 2050 was reduced to 1.8-3.4 Gt CO_2-eq yr^{-1} when implementing a GHG tax ranging from 20-100 USD/tCO_2. While caloric costs where low below 20 USD/tCO_2, they ranged from 27-190 kcal cap^{-1} day^{-1} under the higher economic potential, thus indicating possible negative trade-offs with food security.

In summary, demand-side changes in food choices and consumption can help to achieve global GHG mitigation targets *(high confidence)*. Low-carbon diets on average tend to be healthier and have smaller land footprints. By 2050, technical mitigation potential of dietary changes range from 2.7–6.4 GtCO_2-eq yr^{-1} for assessed diets *(high confidence)*. At the same time, the economic potential of such solutions is lower, ranging from 1.8-3.4 Gt CO_2-eq yr^{-1} at prices of 20-100 USD/tCO_2, with caloric costs up to 190 kcal cap^{-1} day^{-1}. The feasibility of how to create economically viable transitions to more sustainable and healthy diets that also respect food security requirements needs to be addressed in future research.

5.5.2.2 Role of dietary preferences

Food preference is an inherently cultural dimension that can ease or hinder transformations to food systems that contribute to climate change mitigation. Consumer choice and dietary preferences are guided by social, cultural, environmental, and traditional factors as well as economic growth. The food consumed by a given group conveys cultural significance about social hierarchy, social systems and human-environment relationships (Herforth and Ahmed 2015).

As suggested by Springmann et al. (2018a), per capita dietary emissions will translate into different realised diets, according to regional contexts including cultural and gendered norms (e.g, among some groups, eating meat is perceived as more masculine (Ruby and Heine 2011). In some cases, women and men have different preferences in terms of food, with women reporting eating healthier food (Imamura et al. 2015; Kiefer et al. 2005; Fagerli and Wandel 1999): these studies found that men tend to eat more meat, while women eat more vegetables, fruits and dairy products (Kanter and Caballero 2012).

Food preferences can change over time, with the nutrition transition from traditional diets to high-meat, high-sugar, high-saturated fat diets being a clear example of significant changes occurring in a short period of time. Meat consumption per capita consistently responds to income with a saturating trend at high income levels (Sans and Combris 2015; Vranken et al. 2014). Some emerging economies have rapidly increased demand for beef, leading to pressure on natural resources (Bowles et al. 2019). In another example, by reducing beef consumption between 2005 and 2014, Americans avoided approximately 271 million metric tonnes of emissions (CO_2eq) (NRDC 2017). See Section **Error! eference source not found.** for quantitative analysis. Attending farmers markets or buying directly from local producers has been shown to change worldviews (Kerton and Sinclair 2010), and food habits towards healthier diets (Pascucci et al. 2011) can be advanced through active learning (Milestad et al. 2010).

Regarding the options to reduce meat intake in developed countries, research shows that there is an apparent sympathy of consumers for meat reduction due to environmental impacts (Dagevos and Voordouw 2013) which has not been exploited. Social factors that influence reducing meat consumption in New Zealand include the need for better education or information dispersal regarding perceived barriers to producing meat-reduced/less meals; ensuring there is sensory or aesthetic appeal; and placing emphasis on human health or nutritional benefits (Tucker 2018).

Different and complementary strategies can be used in parallel for different consumer's profiles to facilitate step-by-step changes in the amounts and the sources of protein consumed. In the Netherlands, a nationwide sample of 1083 consumers were used to study their dietary choices toward smaller portions of meat, smaller portions using meat raised in a more sustainable manner, smaller portions and eating more vegetable protein, and meatless meals with or without meat substitutes. Results showed that strategies to change meat eating frequencies and meat portion sizes appeared to overlap and that these strategies can be applied to address consumers in terms of their own preferences (de Boer et al. 2014).

5.5.2.3 Uncertainties in demand-side mitigation potential

Both reducing ruminant meat consumption and increasing its efficiency are often identified as main options to reduce greenhouse gases emissions (GHGE) and to lessen pressure on land (Westhoek et al. 2014) (See Section 5.6 for synergies and trade-offs with health and Section 5.7 for discussion of Just Transitions). However, analysing ruminant meat production is highly complex because of the extreme heterogeneity of production systems and due to the numerous products and services associated with ruminants (Gerber et al. 2015). See Supplementary Material Section SM5.5 for further discussion of uncertainties in estimates of livestock mitigation technical potential. Further, current market mechanisms are regarded as insufficient to decrease consumption or increase efficiency, and governmental intervention is often suggested to encourage mitigation in both the supply-side and demand-side of the food system (See Section 5.7) (Wirsenius et al. 2011; Henderson et al. 2018).

Minimising GHG emissions through mathematical programming with near-minimal acceptability constraints can be understood as a reference or technical potential for mitigation through diet shifts. In this context (Macdiarmid et al. 2012) found up to 36% reduction in emissions in UK with similar diet costs applying fixed lifecycle analyses (LCA) carbon footprints (i.e., no rebound effects considered). Westhoek et al. (2014) found 25-40 % in emissions by halving meat, dairy and eggs intake in EU, applying standard IPCC fixed emission intensity factors. Uncertainty about the consequences of on-the-ground implementation of policies towards low ruminant meat consumption in the food system and their externalities remain noteworthy.

Often, all emissions are allocated to only to human edible meat and the boundaries are set only within the farm gate (Henderson et al. 2018; Gerber et al. 2013). However, less than 50% of slaughtered cattle weight is human edible meat, and 1-10% of the mass is lost or incinerated, depending on specified risk materials legislation. The remaining mass provide inputs to multiple industries e.g. clothing, furniture, vehicle coating materials, biofuel, gelatine, soap, cosmetics, chemical and pharmaceutical industrial supplies, pet feed ingredients and fertilisers (Marti et al. 2011; Mogensen et al. 2016; Sousa et al. 2017). This makes ruminant meat production one of the most complex problems for LCA in the food system (Place and Mitloehner 2012; de Boer et al. 2011). There are only a few examples taking into account slaughter byproducts e.g., Mogensen et al. (2016).

5.5.2.4 Insect-based diets

Edible insects are, in general, rich in protein, fat, and energy and can be a significant source of vitamins and minerals (Rumpold and Schlüter 2015). Approximately 1,900 insect species are eaten worldwide, mainly in developing countries (van Huis 2013). The development of safe rearing and effective processing methods are mandatory for utilisation of insects in food and feed. Some insect species can be grown on organic side streams, reducing environmental contamination and transforming waste into high-protein feed. Insects are principally considered as meat substitutes, but worldwide meat substitute consumption is still very low, principally due to differences in food culture, and will require transition phases such as powdered forms (Megido et al. 2016; Smetana et al. 2015). Wider consumer acceptability will relate to pricing, perceived environmental benefits, and the development of tasty insect-derived protein products (van Huis et al. 2015; van Huis 2013). Clearly increasing share of insect-derived protein has the potential to reduce GHG emissions otherwise associated with livestock production. No study to date however has quantified such potential.

5.5.2.5 Food loss and waste, food security, and land use

Food loss and waste impacts food security by reducing global and local food availability, limiting food access due to increase in food price and decrease of producer income, and affecting future food production due to unstainable use of natural resources (HLPE 2014). Food loss is defined as the reduction of edible food during production, postharvest, and processing, whereas food discarded by consumers is considered as food waste (FAO 2011b). Combined food loss and waste amount to a third of global food production *(high confidence)*. During 2010-2016, global food loss and waste equalled 8–10% of total GHG emissions *(medium confidence);* and cost about USD 1 trillion per year (FAO 2014b) *(low confidence).*

A large share of produced food is lost in developing countries due to poor infrastructure, while a large share of produced food is wasted in developed countries (Godfray et al. 2010). Changing consumer behaviour to reduce per capita overconsumption offers substantial potential to improve food security by avoiding related health burdens (Alexander et al. 2017; Smith 2013) and reduce emissions associated with the extra food (Godfray et al. 2010). In 2007, around 20% of the food produced went to waste in Europe and North America, while around 30% of the food produced was lost in sub-Saharan Africa (FAO 2011b). During the last 50 years, the global food loss and waste increased from around 540 Mt in 1961 to 1630 Mt in 2011 (Porter et al. 2016).

In 2011, food loss and waste resulted in about 8–10% of the total anthropogenic greenhouse gas emissions of the entire food system. The mitigation potential of reduced food loss and waste from a full life-cycle perspective, i.e., considering both food supply chain activities and land use change, was estimated as 4.4 Gt CO_2-eq yr^{-1} (FAO 2015a, 2013b). At a global scale, loss and waste of milk, poultry meat, pig meat, sheep meat, and potatoes are associated with 3% of the global agricultural N_2O emissions (more than 200 Gg N_2O-N yr^{-1} or 0.06 Gt CO_2-eq yr^{-1}) in 2009 (Reay et al. 2012). For the United States, 35% of energy use, 34% of blue water use, 34% of GHG emissions, 31% of land use, and 35% of fertiliser use related to an individual's food-related resource consumption were accounted for as food waste and loss in 2010 (Birney et al. 2017).

Similar to food waste, overconsumption, defined as food consumption in excess of nutrient requirements, leads to GHG emissions (Alexander et al. 2017). In Australia for example, overconsumption accounts for about 33% GHGs associated with food (Hadjikakou 2017). In addition to GHG emissions, overconsumption also can lead to severe health conditions such as obesity or diabetes. Over-eating was found to be at least as large a contributor to food system losses (Alexander et al. 2017). Similarly, food system losses associated with consuming resource-intensive animal-based products instead of nutritionally-comparable plant-based alternatives are defined as 'opportunity food losses.' These were estimated to be 96, 90, 75, 50, and 40% for beef, pork, dairy, poultry, and eggs, respectively, in the US (Shepon et al. 2018).

Avoiding food loss and waste will contribute to reducing emissions from the agriculture sector. By 2050, agricultural GHG emissions associated with production of food that might be wasted may increase to 1.9–2.5 Gt CO_2-eq yr^{-1} (Hiç et al. 2016). When land use change for agriculture expansion is also considered, halving food loss and waste reduces the global need for cropland area by around 14% and GHG emissions from agriculture and land use change by 22–28% (4.5 Gt CO_2-eq yr^{-1}) compared to the baseline scenarios by 2050 (Bajželj et al. 2014). The GHG emissions mitigation potential of food loss and waste reduction would further increase when life cycle analysis accounts for emissions throughout food loss and waste through all food system activities.

Reducing food loss and waste to zero might not be feasible. Therefore, appropriate options for the prevention and management of food waste can be deployed to reduce food loss and waste and to minimise its environmental consequences. Papargyropoulou et al. (2014) proposed the 3Rs (i.e., reduction, recovery and recycle) options to prevent and manage food loss and waste. A wide range of approaches across the food supply chain is available to reduce food loss and waste, consisting of technical and non-technical solutions (Lipinski et al. 2013). However, technical solutions (e.g., improved harvesting techniques, on-farm storage, infrastructure, packaging to keep food fresher for longer, etc.) include additional costs (Rosegrant et al. 2015) and may have impacts on local environments (FAO 2018b). Additionally, all parts of food supply chains need to become efficient to achieve the full reduction potential of food loss and waste (Lipinski et al. 2013).

Together with technical solutions, approaches (i.e, non- technical solutions) to changes in behaviours and attitudes of a wide range of stakeholders across the food system will play an important role in reducing food loss and waste. Food loss and waste can be recovered by distributing food surplus to groups affected by food poverty or converting food waste to animal feed (Vandermeersch et al. 2014). Unavoidable food waste can also be recycled to produce energy based on biological, thermal and thermochemical technologies (Pham et al. 2015). Additionally, strategies for reducing food loss and waste also need to consider gender dynamics with participation of females throughout the food supply chain (FAO 2018f).

In summary, reduction of food loss and waste can be considered as a climate change mitigation measure that provides synergies with food security and land use (*robust evidence, medium agreement*,). Reducing food loss and waste reduces agricultural GHG emissions and the need for agricultural expansion for producing excess food. Technical options for reduction of food loss and waste include improved harvesting techniques, on-farm storage, infrastructure, and packaging. However, the beneficial effects of reducing food loss and waste will vary between producers and consumers, and across regions. Causes of food loss (e.g., lack of refrigeration) and waste (e.g., behaviour) differ substantially in developed and developing countries *(robust evidence, medium agreement)*. Additionally, food loss and waste cannot be avoided completely.

5.5.2.6 Shortening supply chains

Encouraging consumption of locally produced food and enhancing efficiency of food processing and transportation can in some cases minimise food loss, contribute to food security, and reduce GHG emissions associated with energy consumption and food loss. For example, Michalský and Hooda (2015), through a quantitative assessment of GHG emissions of selected fruits and vegetables in the UK, reported that increased local production offers considerable emissions savings. They also highlighted that when imports are necessary, importing from Europe instead of the Global South can contribute to considerable GHG emissions savings. Similar results were found by Audsley et al. (2010), with exceptions for some foods, such as tomatoes, peppers or sheep and goat meat. Similarly, a study in India shows that long and fragmented supply chains, which lead to disrupted price signals, unequal power relations perverse incentives and long transport time, could be a key barrier to reducing post-harvest losses (CIPHET 2007).

In other cases, environmental benefits associated with local food can be offset by inefficient production systems with high emission intensity and resource needs, e.g., water, due to local conditions. For example, vegetables produced in open fields can have much lower GHG emissions than locally produced vegetables from heated greenhouses (Theurl et al. 2014). Whether locally grown food has a lower carbon footprint depends on the on-farm emissions intensity as well as the transport emissions. In some cases, imported food may have a lower carbon footprint than locally grown food because some distant countries can produce food at much lower emissions intensity. For example, Avetisyan et al. (2014) reported that regional variation of emission intensities associated with production of ruminant products have large implications for emissions associated with local food. They showed that consumption of local livestock products can reduce emissions due to short supply chains in countries with low emission intensities; however, this might not be the case in countries with high emission intensities.

In addition to improving emission intensity, efficient distribution systems for local food are needed for lowering carbon footprints (Newman et al. 2013). Emissions associated with food transport depend on the mode of transport, for example, emissions are lower for rail rather than truck (Brodt et al. 2013). Tobarra et al. (2018) reported that emissions saving from local food may vary across seasons and regions of import. They highlighted that in Spain local production of fruits and vegetables can reduce emissions associated with imports from Africa but imports from France and Portugal can save emissions in comparison to production in Spain. Additionally, local production of seasonal products in Spain reduces emissions, while imports of out-of-season products can save emissions rather than producing them locally.

In summary, consuming locally grown foods can reduce GHG emissions, if they are grown efficiently *(high confidence)*. The emissions reduction potential varies by region and season. Whether food with shorter supply chains has a lower carbon footprint depends on both the on-farm emissions intensity as well as the transport emissions. In some cases, imported food may have a lower carbon footprint because some distant agricultural regions can produce food at lower emissions intensities.

5.6 Mitigation, Adaptation, Food Security, and Land Use – Synergies, Trade-Offs, and Co-Benefits

Food systems will need to adapt to changing climates and also to reduce their greenhouse gas (GHG) emissions and sequester carbon if Paris Agreement goals are to be met (Springmann et al. 2018a; van Vuuren et al. 2014). The synergies and trade-offs between the food system mitigation and adaptation options described in Sections 5.3 and 5.5 are of increasing importance in both scientific and policy communities because of the necessity to ensure food security, i.e., provision of nutritious food for the growing population, while responding to climate change (Rosenzweig and Hillel 2015) A special challenge involves interactions between land-based non-food system mitigation, such as negative emissions technologies, and food security. Response options for the food system have synergies and trade-offs between climate change mitigation and adaptation (Figure 5.13; Chapter 6).

Tirado et al. (2013) suggest an integrated approach to address the impacts of climate change to food security that considers a combination of nutrition-sensitive adaptation and mitigation measures, climate-resilient and nutrition-sensitive agricultural development, social protection, improved maternal and child care and health, nutrition-sensitive risk reduction and management, community development measures, nutrition-smart investments, increased policy coherence, and institutional and cross-sectoral collaboration. These measures are a meansto achieve both short-term and long-term benefits in poor and marginalised groups.

This section assesses the synergies and trade-offs for land-based atmospheric carbon dioxide removal measures, effects of mitigation measures on food prices, and links between dietary choices and human health. It then evaluates a range of integrated agricultural systems and practices that combine mitigation and adaptation measures, including the role of agricultural intensification. The roles of women's empowerment and urban agriculture are examined, as well as interactions between SDG2 (Zero Hunger) and SDG 13 (Climate Action).

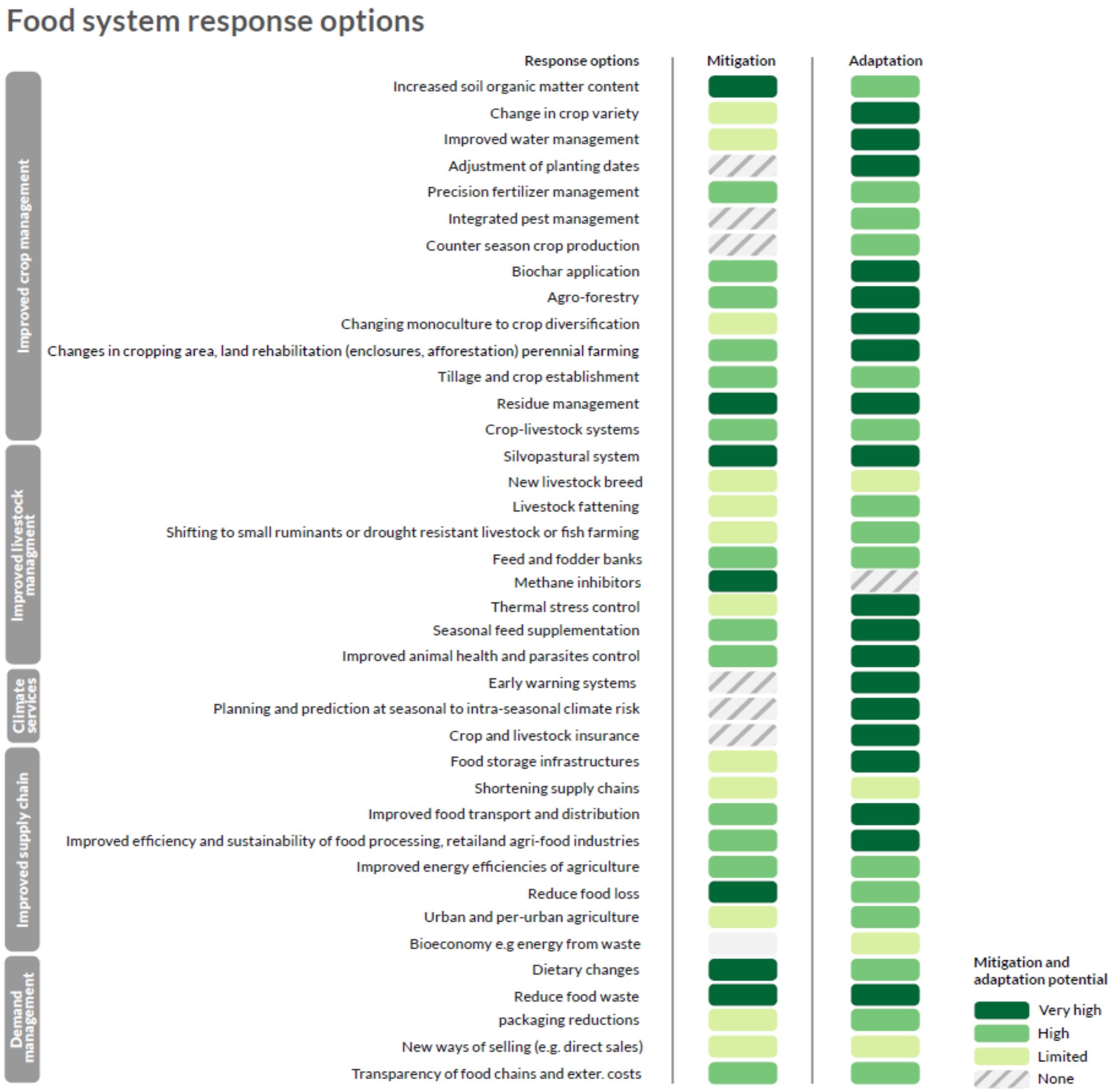


Figure 5.13 Response options related to food system and their potential impacts on mitigation and adaptation. Many response options offer significant potential for both mitigation and adaptation.

5.6.1 Land-based carbon dioxide removal (CDR) and bioenergy

Large-scale deployment of negative emission technologies (NETs) in emission scenarios has been identified as necessary for avoiding unacceptable climate change (IPCC 2018b). Among the available NETs, carbon dioxide removal (CDR) technologies are receiving increasing attention. Land-based CDRs include afforestation and reforestation (AR), sustainable forest management, biomass energy with carbon capture and storage (BECCS), and biochar (BC) production (Minx et al. 2018). Most of the literature on global land-based mitigation potential relies on CDRs, particularly on BECCS, as a major mitigation action (Kraxner et al. 2014; Larkin et al. 2018; Rogelj et al. 2018, 2015, 2011). BECCS is not yet deployable at a commercial scale, as it faces challenges similar to fossil fuel carbon capture and storage (CCS) (Fuss et al. 2016; Vaughan and Gough 2016; Nemet et al. 2018). Regardless, the effectiveness of large-scale BECCS to meet Paris Agreement goals has been questioned and other pathways to mitigation have been proposed (Anderson and Peters 2016; van Vuuren et al. 2017, 2018; Grubler et al. 2018; Vaughan and Gough 2016).

Atmospheric CO_2 removal by storage in vegetation depends on achieving net organic carbon accumulation in plant biomass over decadal time scales (Kemper 2015) and, after plant tissue decay, in soil organic matter (Del Grosso et al. 2019). AR, BECCS and BC differ in the use and storage of plant biomass. In BECCS, biomass carbon from plants is used in industrial processes (e.g., for electricity, hydrogen, ethanol, and biogas generation), releasing CO_2, which is then captured and geologically stored (Greenberg et al. 2017; Minx et al. 2018).

Afforestation and reforestation result in long-term carbon storage in above and belowground plant biomass on previously unforested areas, and is effective as a carbon sink during the AR establishment period, in contrast to thousands of years for geological C storage (Smith et al. 2016).

Biochar is produced from controlled thermal decomposition of biomass in absence of oxygen (pyrolysis), a process that also yields combustible oil and combustible gas in different proportions. Biochar is a very stable carbon form, with storage on centennial timescales (Lehmann et al. 2006) (See also Chapter 4). Incorporated in soils, some authors suggest it may lead to improved water-holding capacity, nutrient retention, and microbial processes (Lehmann et al. 2015). There is however considerable uncertainty about the benefits and risks of this practice (The Royal Society 2018).

Land-based CDRs require high biomass-producing crops. Since not all plant biomass is harvested (e.g., roots and harvesting losses), it can produce co-benefits related to soil carbon sequestration, crop productivity, crop quality, as well improvements in air quality, but the overall benefits strongly depend on the previous land use and soil management practices (Smith et al. 2016; Wood et al. 2018). In addition, CDR effectiveness varies widely depending on type of biomass, crop productivity, and emissions offset in the energy system. Importantly, its mitigation benefits can be easily lost due to land-use change interactions (Harper et al. 2018; Fuss et al. 2018; Daioglou et al. 2019).

Major common challenges of implementing these large-scale CDR solutions, as needed to stabilise global temperature "well-below" 2°C by the end of the century, are the large investments and the associated significant changes in land use required. Most of the existing scenarios estimate the global area required for BECCS alone in the range of 109-990 Mha (IPCC 2018a), most commonly around 380–700 Mha (Smith et al. 2016), reaching rates of net area expansion rates up to 23.7 M ha yr^{-1} (IPCC 2018b). The upper limit implies unprecedented rates of area expansion for crops and forestry observed historically, for instance as reported by FAO since 1961 (FAOSTAT 2018). By comparison, the sum of recent worldwide rates of expansion in harvested area of soybean and sugarcane has not exceeded 3.5 M ha yr^{-1} on average. Even at this rate, they have been the source of major concerns for their possible negative environmental and food security impacts (Boerema et al. 2016; Popp et al. 2014).

Most land area available for CDR is currently pasture, estimated at 3,300 Mha globally (FAOSTAT 2018). However, there is *low confidence* about how much low-productivity land is actually available for CDR (Lambin et al. 2013; Gibbs and Salmon 2015). There is also *low confidence* and *low agreement* if the transition to BECCS will take place directly on low-productivity grasslands (Johansson and Azar 2007), and uncertainty on the governance mechanisms required to avoid unwanted spill-over effects, for instance causing additional deforestation (Keles et al. 2018).

Further, grasslands and rangelands may often occur in marginal areas, in which case they may be exposed to climate risks, including periodic flooding. Grasslands and especially rangelands and savannahs tend to predominate in less-developed regions, often bordering areas of natural vegetation with little infrastructure available for transport and processing of large quantities of CDR-generated biomass (O'Mara 2012; Beringer et al. 2011; Haberl et al. 2010; Magdoff 2007).

CDR-driven reductions in available pastureland area is a scenario of constant or increasing global animal protein output as proposed by (Searchinger et al. 2018). However, despite the recent reduction in meat consumption in western countries, this will require productivity improvements (Cohn et al. 2014; Strassburg et al. 2014). It would also result in lower emission intensities and create conditions for increased soil carbon stocks (de Oliveira Silva et al. 2016; Searchinger et al. 2018; Soussana et al.

2019, 2013). At the same time, food security may be threatened if land-based mitigation displaced crops elsewhere, especially if to regions of lower productivity potential, higher climatic risk, and higher vulnerability.

There is *low agreement* about what are the more competitive regions of the world for CDRs. Smith et al. (2016) and Vaughan et al. (2018) identify as candidates relatively poor countries in Latin America, Africa and Asia (except China and India). Others indicate those regions may be more competitive for food production, placing Europe as a major BECCS exporter (Muratori et al. 2016). Economically feasible CDR investments are forecast to be directed to regions with high biomass production potential, demand for extra energy production, low leakage potential for deforestation and low competition for food production (Vaughan et al. 2018). Latin America and Africa, for instance, although having high biomass production potential, still have low domestic energy consumption (589 and 673 MTOE – 24.7 and 28.2 EJ, respectively), with about 30% of primary energy from renewable sources (reaching 50% in Brazil), mainly hydropower and traditional biomass.

There is *high confidence* that deployment of BECCS will require ambitious investments and policy interventions (Peters and Geden 2017) with strong regulation and governance of bioenergy production to ensure protection of forests, maintain food security and enhance climate benefits (Burns and Nicholson 2017; Vaughan et al. 2018; Muratori et al. 2016), and that such conditions may be challenging for developing countries. Increased value of bioenergy puts pressure on land, ecosystem services, and the prices of agricultural commodities, including food (*high confidence*).

There is *medium confidence* for the impact of CDR technologies on increased food prices and reduced food security, as these depend on several assumptions. Nevertheless, those impacts could be strong, with food prices doubling under certain scenario combinations (Popp et al. 2017). The impacts of land-mitigation policies on the reduction of dietary energy availability alone, i.e., without climate change impacts, is estimated at over 100 kcal.person^{-1} day^{-1} by 2050, with highest regional impacts in south Asia and sub-Saharan Africa (Hasegawa et al. 2018) (See Section 5.2). However, only limited pilot BECCS projects have been implemented to date (Lenzi et al. 2018). Integrated assessment models (IAMs) use theoretical data based on high-level studies and limited regional data from the few on-the-ground BECCS projects.

Furthermore, it has been suggested that several BECCS IAM scenarios rely on unrealistic assumptions regarding regional climate, soils and infrastructure suitability (Anderson and Peters 2016), as well as international bioenergy trade (Lamers et al. 2011). Current global IAMs usually consider major trends in production potential and projected demand, overlooking major challenges for the development of a reliable international market. Such a market will have to be created from scratch and overcome a series of constraints, including trade barriers, logistics, and supply chains, as well as social, ecological and economic impacts (Matzenberger et al. 2015).

In summary, there is *high agreement* that better assessment of BECCS mitigation potential would need to be based on increased regional, bottom-up studies of biomass potentials, socio-economic consequences (including on food security), and environmental impacts in order to develop more realistic estimates (IPCC 2018a).

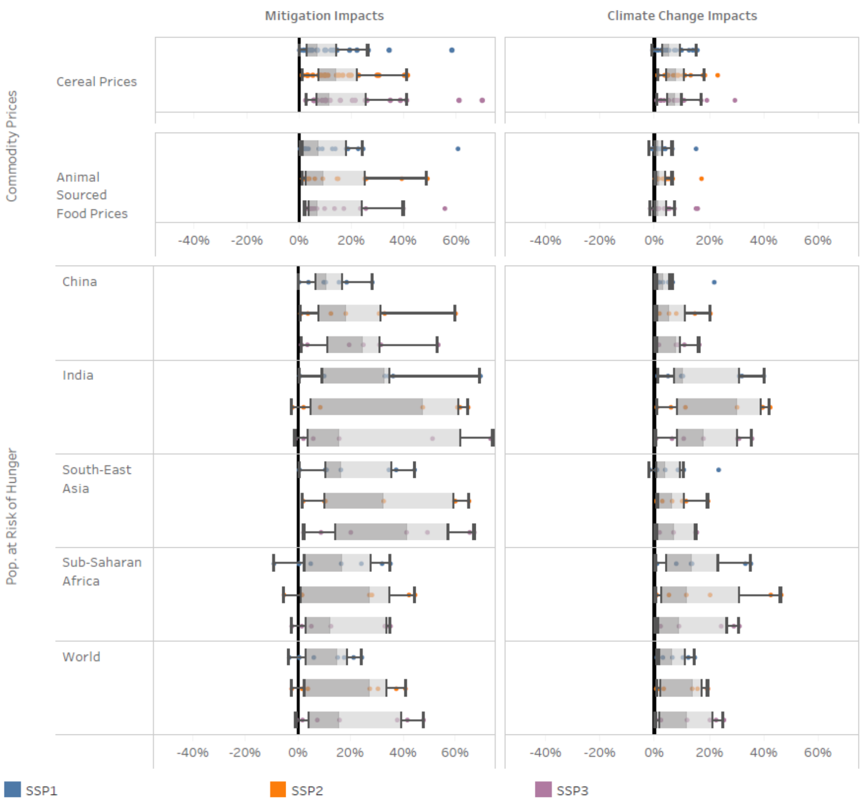
5.6.2 Mitigation, food prices, and food security

Food prices are the result of supply, demand and trade relations. Earlier studies (e.g., (Nelson et al. 2009)) showed that recent climate impacts that reduced crop productivity led to higher prices and increased trade of commodities between regions, with asymmetric impacts on producers and consumers. In terms of published scenario analyses, the most affected regions tend to be Sub-Saharan Africa and parts of Asia, but there is significant heterogeneity in results between countries. Relocation of production to less affected areas buffers these impacts to a certain extent, as well as potential for improvements in food production technologies (Hasegawa et al. 2018; van Meijl et al. 2017; Wiebe et al. 2015; Lotze-Campen et al. 2014; Valin et al. 2014; Robinson et al. 2014).

A newer, less studied impact of climate change on prices and their impacts on food security is the level of land-based mitigation necessary to stabilise global temperature. Hasegawa et al. (2018), using an ensemble of seven global economic models across a range of greenhouse gas emissions pathways and socioeconomic trajectories, suggested that the level of mitigation effort needed to reduce emissions can have a more significant impact on prices than the climate impacts themselves on reduced crop yields (Figure 5.14). This occurs because in the models, taxing GHG emissions leads to higher crop and livestock prices, while land-based mitigation leads to less land availability for food production, potentially lower food supply, and therefore food price increases.

Price increases in turn lead to reduced consumption, especially by vulnerable groups, or to shifts towards cheaper food, which are often less nutritious. This leads to significant increases in the number of malnourished people. Frank et al. (2017) and Fujimori et al. (2017) arrived at the same conclusions for the 1.5°C mitigation scenario using the IAM Globiom and ensembles of AgMIP global economic models. While the magnitude of the response differs between models, the results are consistent between them. In contrast, a study based on five global agro-economic models highlights that the global food prices may not increase much when the required land for bioenergy is accessible on the margin of current cropland, or the feedstock does not have a direct completion with agricultural land (Lotze-Campen et al. 2014).

These studies highlight the need for careful design of emissions mitigation policies in upcoming decades—for example, targeted schemes encouraging more productive and resilient agricultural production systems and the importance of incorporating complementary policies (such as safety-net programmes for poverty alleviation) that compensate or counteract the impacts of the climate change mitigation policies on vulnerable regions (Hasegawa et al. 2018). Fujimori et al. (2018) showed how an inclusive policy design can avoid adverse side-effects on food security through international aid, bioenergy taxes, or domestic reallocation of income. These strategies can shield impoverished and vulnerable people from the additional risk of hunger that would be caused by the economic effects of policies narrowly focussing on climate objectives only.



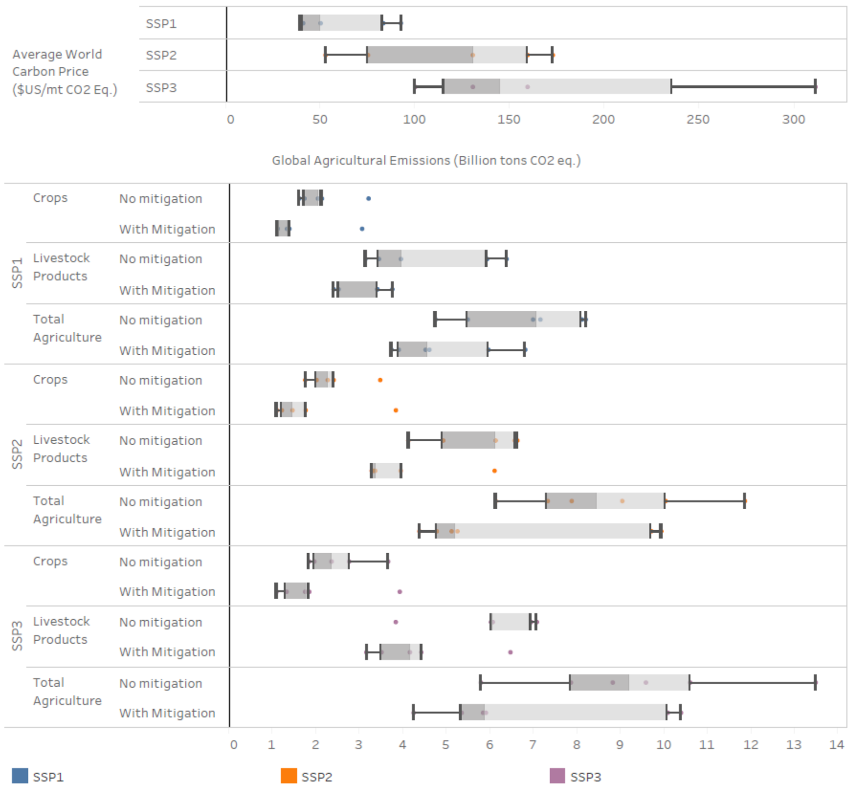


Figure 5.14 Regional impacts of climate change and mitigation on food price (top), population (pop) at risk of hunger or undernourishment (middle), greenhouse gas emissions (bottom) in 2050 under different socio-economic scenarios (SSP1, SSP2 and SSP33). Values indicate changes from no climate change and no climate change mitigation scenario. MAgPIE, a global land use allocation model, is excluded due to

inelastic food demand. The value of India includes that of Other Asia in MAGNET, a global general equilibrium model (Hasegawa et al. 2018)

In summary, food security will be threatened through increasing numbers of malnourished people if land-based mitigation raises prices, unless other policy mechanisms reduce its impact (*high confidence*). Inclusive policy design can avoid adverse side-effects on food security by shielding vulnerable people from the additional risk of hunger that would be caused by the economic effects of policies narrowly focusing on climate objectives (*medium confidence*).

5.6.3 Environmental and health effects of adopting healthy and sustainable diets

Two key questions arise from the potentially significant mitigation potential of dietary change: 1) Are 'low-GHG emission diets' likely to be beneficial for health? and 2) Would changing diets at scale provide substantial benefits? In short, what are the likely synergies and trade-offs between low-GHG emissions diets and food security, health, and climate change? See Supplementary Material Section SM5.6 for further discussion.

Are "low GHG emission diets" healthy? Consistent evidence indicates that, in general, a dietary pattern that is higher in plant-based foods, such as vegetables, fruits, whole grains, legumes, nuts, and seeds, and lower in animal-based foods, is more health-promoting and associated with lower environmental impact (GHG emissions and energy, land, and water use) than either the current global average diets (Swinburn et al. 2019; Willett et al. 2019; Springmann et al. 2016b), or the current average US diet (Nelson et al. 2016). Another study (Van Mierlo et al. 2017) showed that nutritionally-equivalent diets can substitute plant-based foods for meat and provide reductions in GHG emissions.

There are several studies that estimate health adequacy and sustainability and conclude that healthy sustainable diets are possible. These include global studies (e.g., (Willett et al. 2019; Swinburn et al. 2019)), as well as localised studies (e.g., (Van Dooren et al. 2014). For example, halving consumption of meat, dairy products and eggs in the European Union would achieve a 40% reduction in ammonia emissions, 25–40% reduction in non-CO_2 GHG emissions (primarily from agriculture) and 23% per capita less use of cropland for food production, with dietary changes lowering health risks (Westhoek et al. 2014). In China, diets were designed that could meet dietary guidelines while creating significant reductions in GHG emissions (between 5% and 28%, depending on scenario) (Song et al. 2017). Changing diets can also reduce non-dietary related health issues caused by emissions of air pollutants; for example, specific changes in diets were assessed for their potential to mitigate PM2.5 in China (Zhao et al. 2017b).

A range of studies are starting to estimate both health and environmental benefits from dietary shifts. For example, (Farchi et al. 2017) estimate health (colorectal cancer, cardio-vascular disease) and GHG reductions of "Mediterranean" diets, low in meat, in Italy, finding the potential to reduce deaths from colorectal cancer of 7-10% and CVD from 9-10%, and potential savings of up to 263 CO_2-eq.person^{-1}.yr^{-1}. In the US, Hallström et al. (2017) found that adoption of healthier diets (consistent with dietary guidelines, and reducing amounts of red and processed meats) could reduce relative risk of coronary heart disease, colorectal cancer, and type 2 diabetes by 20–45%, US health care costs by USD 77–93 billion per year, and direct GHG emissions by 222–826 kg CO_2-eq/capita yr^{-1} (69–84 kg from the health care system, 153–742 kg from the food system). Broadly similar conclusions were found for the Netherlands (Biesbroek et al. 2014); and the UK (Friel et al. 2009; Milner et al. 2015).

Whilst for any given disease, there are a range of factors, including diet, that can affect it, and evidence is stronger for some diseases than others, a recent review found that an overall trend toward increased cancer risk was associated with unhealthy dietary patterns, suggesting that diet-related choices could significantly affect the risk of cancer (Grosso et al. 2017). Tilman and Clark (2014) found significant benefits in terms of reductions in relative risk of key diseases: type 2 diabetes, cancer, coronary mortality and all causes of mortality (Figure 5.11).

5.6.3.1 Can dietary shifts provide significant benefits?

Many studies now indicate that dietary shifts can significantly reduce GHG emissions. For instance, several studies highlight that if current dietary trends are maintained, this could lead to emissions from agriculture of approximately 20 Gt CO_2-eq yr^{-1} by 2050, creating significant mitigation potential (Pradhan et al. 2013b; Bajželj et al. 2014; Hedenus et al. 2014; Bryngelsson et al. 2017). Additionally in the US, a shift in consumption towards a broadly healthier diet, combined with meeting the USDA and Environmental Protection Agency's 2030 food loss and waste reduction goals, could increase *per capita* food-related energy use by 12%, decrease blue water consumption by 4%, decrease green water use by 23%, decrease GHG emissions from food production by 11%, decrease GHG emissions from landfills by 20%, decrease land use by 32%, and increase fertiliser use by 12% (Birney et al. 2017). This study, however, does not account for all potential routes to emissions, ignoring, for example, fertiliser use in feed production. Similar studies have been conducted, for China (Li et al. 2016), where adoption of healthier diets and technology improvements have the potential to reduce food systems GHG emissions by >40% relative to those in 2010; and India (Green et al. 2017; Vetter et al. 2017), where alternative diet scenarios can affect emissions from the food system by -20 to +15%.

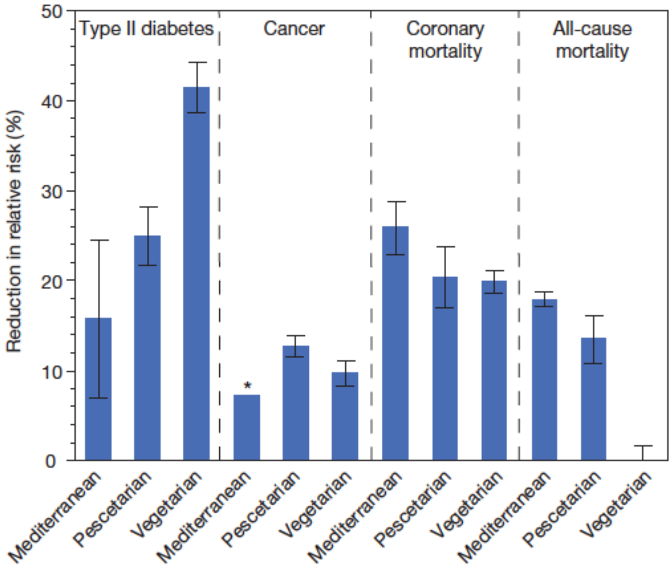


Figure 5.11 Diet and health effects of different consumption scenarios (Tilman and Clark 2014) (* reflects data from a single study, hence no error bars)

Springmann et al (2018a) modelled the role of technology, waste reduction and dietary change in living within planetary boundaries (Rockström et al. 2009), with the climate change boundary being a 66% chance of limiting warming to less than 2°C. They found that all are necessary for achievement of a sustainable food system. Their principal conclusion is that only by adopting a "flexitarian diet", as a global average, would climate change be limited to under two degrees. Their definition of a flexitarian diet is fruits and vegetables, plant-based proteins, modest amounts of animal-based proteins, and limited amounts of red meat, refined sugar, saturated fats, and starchy foods.

Healthy and sustainable diets address both health and environmental concerns (Springmann et al. 2018b). There is high agreement that there are significant opportunities to achieve both objectives simultaneously. Contrasting results of marginal GHG emissions, i.e variations in emissions as a result of variation in one or more dietary components, are found when comparing low to high emissions self-selected diets (diets freely chosen by consumers). (Vieux et al. 2013) found self-selected healthier diets with higher amounts of plant-based food products did not result in lower emissions, while (Rose et al. 2019) found that the lowest emission diets analysed were lower in meat but higher in oil, refined grains and added sugar. (Vieux et al. 2018) concluded that setting nutritional goals with no consideration for the environment may increase GHG emissions (GHGE).

Tukker et al. (2011) also found a slight increase in emissions by shifting diets towards the European dietary guidelines, even with lower meat consumption. Heller and Keoleian (2015) found a 12% increase in GHGE when shifting to iso-caloric diets, i.e., diets with the same caloric intake of diets currently consumed, following the US guidelines and a 1% decrease in GHGE when adjusting caloric intake to recommended levels for moderate activity. There is scarce information on the marginal GHGE that would be associated with following dietary guidelines in developing countries.

Some studies have found a modest mitigation potential of diet shifts when economic and biophysical systems effects are taken into account in association with current dietary guidelines. Tukker et al. (2011), considering economic rebound effects of diet shifts (i.e., part of the gains would be lost due to increased use at lower prices), found maximum changes in emissions of the EU food system of 8% (less than 2% of total EU emissions) when reducing meat consumption by 40 to 58%. Using an economic optimisation model for studying carbon taxation in food but with adjustments of agricultural production systems and commodity markets in Europe (Zech and Schneider 2019) found a reduction of 0.41% in GHG emissions at a tax level of 50 $USDt^{-1}CO_2eq$. They estimate a leakage of 43% of the greenhouse gas emissions reduced by domestic consumption, (i.e., although reducing emissions due to reducing consumption, around 43% of the emissions would not be reduced because part of the production would be directed to exports).

Studying optimised beef production systems intensification technologies in a scenario of no grasslands area expansion (de Oliveira Silva et al. 2016) found marginal GHG emissions to be negligible in response to beef demand in the Brazilian Cerrado. This was because reducing productivity would lead to increased emission intensities, cancelling out the effect of reduced consumption.

In summary, there is significant potential mitigation *(high confidence)* arising from the adoption of diets in line with dietary recommendations made on the basis of health. These are broadly similar across most countries. These are typically capped by at the number of calories and higher in plant-based foods, such as vegetables, fruits, whole grains, legumes, nuts and seeds, and lower in animal-sourced foods, fats and sugar. Such diets have the potential to be both more sustainable and healthy than alternative diets (but healthy diets are not necessarily sustainable and vice versa). The extent to which the mitigation potential of dietary choices can be realised requires both climate change and health being considered together. Socio-economic (prices, rebound effects), political, and cultural contexts would require significant consideration to enable this mitigation potential to be realised.

5.6.4 Sustainable integrated agricultural systems

A range of integrated agricultural systems are being tested to evaluate synergies between mitigation and adaptation and lead to low-carbon and climate-resilient pathways for sustainable food security and ecosystem health *(robust evidence, medium agreement)*. Integration refers to the use of practices that enhance an agroecosystem's mitigation, resilience, and sustainability functions. These systems follow holistic approaches with the objective of achieving biophysical, socio-cultural, and economic benefits from land management systems (Sanz et al. 2017). These integrated systems may include agroecology (FAO et al. 2018; Altieri et al. 2015), climate smart agriculture (FAO 2011c; Lipper et al. 2014; Aggarwal et al. 2018), conservation agriculture (Aryal et al. 2016; Sapkota et al. 2015), and sustainable intensification (FAO 2011d; Godfray 2015), among others.

Many of these systems are complementary in some of their practices, although they tend to be based on diffèrent narratives (Wezel et al. 2015; Lampkin et al. 2015; Pimbert 2015). They have been tested in various production systems around the world (Dinesh et al. 2017; Jat et al. 2016; Sapkota et al. 2015; Neufeldt et al. 2013). Many technical innovations, e.g., precision nutrient management (Sapkota et al. 2014) and precision water management (Jat et al. 2015), can lead to both adaptation and mitigation outcomes and even synergies; although negative adaptation and mitigation outcomes (i.e., trade-offs) are often overlooked. Adaptation potential of ecologically intensive systems includes crop diversification, maintaining local genetic diversity, animal integration, soil organic management, water conservation and harvesting the role of microbial assemblages (See Section 5.3). Technical innovations may encompass not only inputs reduction, but complete redesign of agricultural systems (Altieri et al. 2017) and how knowledge is generated (Levidow et al. 2014), including social and political transformations.

5.6.4.1 Agroecology

Agroecology (see Glossary) (Francis et al. 2003; Gliessman and Engles 2014; Gliessman 2018), provides knowledge for their design and management, including social, economic, political, and cultural dimensions (Dumont et al. 2016). It started with a focus at the farm level but has expanded to include the range of food system activities (Benkeblia 2018). Agroecology builds systems resilience through knowledge-intensive practices relying on traditional farming systems and co-generation of new insights and information with stakeholders through participatory action research (Menéndez et al. 2013). It provides a multidimensional view of food systems within ecosystems, building on indigenous and local knowledge (ILK) and co-evolving with the experiences of local people, available natural resources, access to these resources, and ability to share and pass on knowledge among communities and generations, emphasising the inter-relatedness of all agroecosystem components and the complex dynamics of ecological processes (Vandermeer 1995).

At the farm level, agroecological practices recycle biomass and regenerate soil biotic activities. They strive to attain balance in nutrient flows to secure favorable soil and plant growth conditions, minimise loss of water and nutrients, and improve use of solar radiation. Practices include efficient microclimate management, soil cover, appropriate planting time and genetic diversity. They seek to promote ecological processes and services such as nutrient cycling, balanced predator/prey interactions, competition, symbiosis, and successional changes. The overall goal is to benefit human and non-human communities in the ecological sphere, with fewer negative environmental or social impacts and fewer external inputs (Vandermeer et al. 1998; Altieri et al. 1998). From a food system focus, agroecology provides management options in terms of commercialisation and consumption through the promotion of short food chains and healthy diets (Pimbert and Lemke 2018; Loconto et al. 2018).

Agroecology has been proposed as a key set of practices in building climate resilience (FAO et al. 2018; Altieri et al. 2015). These can enhance on-farm diversity (of genes, species, and ecosystems) through a landscape approach (FAO 2018g). Outcomes include soil conservation and restoration and thus soil carbon sequestration, reduction of the use of mineral and chemical fertilisers, watershed protection, promotion of local food systems, waste reduction, and fair access to healthy food through nutritious and diversified diets (Pimbert and Lemke 2018; Kremen et al. 2012; Goh 2011; Gliessman and Engles 2014).

A principle agroecology is to contribute to food production by smallholder farmers (Altieri 2002). Since climatic events can severely impact smallholder farmers, there is a need to better understand the heterogeneity of small-scale agriculture in order to consider the diversity of strategies that traditional farmers have used and still use to deal with climatic variability. In Africa, many smallholder farmers cope with and even prepare for climate extremes, minimising crop failure through a series of agroecological practices (e.g., biodiversification, soil management, and water harvesting) (Mbow et al. 2014a). Resilience to extreme climate events is also linked to on-farm biodiversity, a typical feature of traditional farming systems (Altieri and Nicholls 2017).

Critiques of agroecology refer to its explicit exclusion of modern biotechnology (Kershen 2013) and the assumption that smallholder farmers are a uniform unit with no heterogeneity in power (and thus gender) relationships (Neira and Montiel 2013; Siliprandi and Zuluaga Sánchez 2014).

5.6.4.2 Climate-smart agriculture

'Climate-smart agriculture' (CSA) is an approach developed to tackle current food security and climate change challenges in a joint and synergistic fashion (Lipper et al. 2014; Aggarwal et al. 2018; FAO 2013c). CSA is designed to be a pathway towards development and food security built on three pillars: increasing productivity and incomes, enhancing resilience of livelihoods and ecosystems and reducing, and removing GHG emissions from the atmosphere (FAO 2013c). Climate-smart agricultural systems are integrated approaches to the closely linked challenges of food security, development, and climate change adaptation/mitigation to enable countries to identify options with maximum benefits and those where trade-offs need management.

Many agricultural practices and technologies already provide proven benefits to farmers' food security, resilience and productivity (Dhanush and Vermeulen 2016). In many cases these can be made implemented by changing the suites of management practices. For example, enhancing soil organic matter to improve water-holding capacity of agricultural landscapes also sequesters carbon. In annual cropping systems, changes from conventional tillage practices to minimum tillage can convert the system from one that either provides only adaptation or mitigation benefits or neither types of benefits to one that provides both adaptation and mitigation benefits (Sapkota et al. 2017a; Harvey et al. 2014a).

Increasing food production by using more fertilisers in agricultural fields could maintain crop yield in the face of climate change, but may result in greater overall GHG emissions. But increasing or maintaining the same level of yield by increasing nutrient-use-efficiency through adoption of better fertiliser management practices could contribute to both food security and climate change mitigation (Sapkota et al. 2017a).

Mixed farming systems integrating crops, livestock, fisheries and agro-forestry could maintain crop yield in the face of climate change, help the system to adapt to climatic risk, and minimise GHG emissions by increasingly improving the nutrient flow in the system (Mbow et al. 2014a; Newaj et al. 2016; Bioversity International 2016). Such systems can help diversify production and/or incomes and support efficient and timely use of inputs thus contributing to increased resilience, but require local seed and input systems and extension services. Recent whole farm modelling exercises have shown the economic and environmental (reduced GH emissions, reduced land use) benefits of integrated crop-livestock systems. Gil et al. (2018) compared different soy-livestock systems across multiple economic and environmental indicators, including climate resilience. However it is important to note that potential benefits are very context specific.

Although climate-smart agriculture involves a holistic approach, some argue that it narrowly focuses on technical aspects at the production level (Taylor 2018; Newell and Taylor 2018). Studying barriers to the adoption and diffusion of technological innovations for climate-smart agriculture in Europe, Long et al. (2016) found that there was incompatibility between existing policies and climate-smart agriculture objectives, including barriers to the adoption of technological innovations.

Climate-smart agricultural systems recognise that the implementation of the potential options will be shaped by specific country contexts and capacities, as well as enabled by access to better information, aligned policies, coordinated institutional arrangements and flexible incentives and financing mechanisms (Aggarwal et al. 2018). Attention to underlying socio-economic factors that affect adoption of practices and access to technologies is crucial for enhancing biophysical processes, increasing productivity, and reducing GHG emissions at scale. The Government of India, for example, has started a program of climate resilient villages (CRV) as a learning platform to design, implement, evaluate and promote various climate-smart agricultural interventions, with the goal of ensuring enabling mechanisms at the community level (Srinivasa Rao et al. 2016).

5.6.4.3 Conservation agriculture

Conservation agriculture (CA) is based on the principles of minimum soil disturbance and permanent soil cover combined with appropriate crop rotation (Jat et al. 2014; FAO 2011e). CA has been shown to respond with positive benefits to smallholder farmers under both economic and environmental pressures (Sapkota et al. 2017a, 2015). This agricultural production system uses a body of soil and residues management practices that control erosion (Blanco Sepúlveda and Aguilar Carrillo 2016) and at the same time to improve soil quality, by increasing organic matter content and improving porosity, structural stability, infiltration and water retention (Sapkota et al. 2017a, 2015; Govaerts et al. 2009)

Intensive agriculture during the second half of the 20th century led to soil degradation and loss of natural resources and contributed to climate change. Sustainable soil management practices can address both food security and climate change challenges faced by these agricultural systems. For example, sequestration of soil organic carbon (SOC) is an important strategy to improve soil quality and to mitigation of climate change (Lal 2004). CA has been reported to increase farm productivity by reducing costs of production (Aryal et al. 2015; Sapkota et al. 2015; Indoria et al. 2017) as well as to reduce GHG emission (Pratibha et al. 2016).

CA brings favourable changes in soil properties that affect the delivery of nature's contribution to people (NCPs) or ecosystem services including climate regulation through carbon sequestration and GHG emissions (Palm et al. 2013; Sapkota et al. 2017a). However, by analysing datasets for soil carbon in the tropics, Powlson et al. (2014, 2016) argued that the rate of SOC increase and resulting GHG mitigation in CA systems, from zero-tillage in particular, has been overstated (see also Chapter 2).

However, there is unanimous agreement that the gain in SOC and its contribution to GHG mitigation by CA in any given soil is largely determined by the quantity of organic matter returned to the soil (Giller et al. 2009; Virto et al. 2011; Sapkota et al. 2017b). Thus, a careful analysis of the production system is necessary to minimise the trade-offs among the multiple use of residues, especially where residues remain an integral part of livestock feeding (Sapkota et al. 2017b). Similarly, replacing mono-cropping systems with more diversified cropping systems and agroforestry as well as afforestation and deforestation can buffer temperatures as well as increase carbon storage (Mbow et al. 2014a; Bioversity International 2016), and provide diversified and healthy diets in the face of climate change.

CA adoption in Africa has been low despite more than three decades of implementation (Giller et al. 2009), although there is promising uptake recently in east and southern Africa. This calls for a better understanding of the social and institutional aspects around CA adoption. Brown et al. (2017a) found that institutional and community constraints hampered the use of financial, physical, human and informational resources to implement CA programs.

Gender is another variable to consider since at intra-household level, decision-making and benefits distribution. CA interventions have implications for labour requirements, labour allocation, and investment decisions, all of which impacting the roles of men and women (Farnworth et al. 2016) (see also Section 5.1.3). For example, in the global South, CA generally reduces labour and production costs and generally leads to increased returns to family labour (Aryal et al. 2015) although a gender shift of the labour burden to women have also been described (Giller et al. 2009).

5.6.4.4 Sustainable intensification

The need to produce about 50% more food by 2050, required to feed the increasing world population (FAO 2018a) may come at the price of significant increases in GHG emissions and environmental impacts, including loss of biodiversity. For instance, land conversion for agriculture is responsible for an estimated 8-10% of all anthropogenic GHG emissions currently (see Section 5.4). Recent calls for sustainable intensification (SI) are based on the premise that damage to the environment through extensification outweighs benefits of extra food produced on new lands (Godfray 2015). However, increasing net production area by restoring already degraded land may contribute to increased production on the one hand and increase carbon sequestration on the other (Jat et al. 2016), thereby contributing to both increased agricultural production and improved natural capital outcomes (Pretty et al. 2018).

Sustainable intensification is a goal but does not specify *a priori* how it could be attained, e.g., which agricultural techniques to deploy (Garnett et al. 2013). It can be combined with selected other improved management practices, e.g., conservation agriculture (see above) or agroforestry, with additional economic, ecosystem services, and carbon benefits. Sustainable intensification , by improving nutrient-, water- and other input-use-efficiency, not only helps to close yield gaps and contribute to food security (Garnett et al. 2013), but also reduces the loss of such production inputs and associated emissions (Sapkota et al. 2017c; Wollenberg et al. 2016). Closing yield gaps is a way to become more efficient in use of land per unit production. Currently, most regions in Africa and South Asia have attained less than 40% of their potential crop production (Pradhan et al. 2015). Integrated farming systems (e.g., mixed crop-livestock, crop-aquaculture) are strategies to produce more products per unit land, which in regard to food security, becomes highly relevant.

Sustainable intensification acknowledges that enhanced productivity needs to be accompanied by maintenance of other ecosystem services and enhanced resilience to shocks (Vanlauwe et al. 2014). SI in intensively farmed areas may require a reduction in production in favour of increasing sustainability in the broad sense (Buckwell et al. 2014) (see Cross-Chapter Box 6: Agricultural Intensification). Hence, moving towards sustainability may imply lower yield growth rates than those maximally attainable in such situations. For areas that contain valuable natural ecosystems, such as the primary forest in the Congo basin, intensification of agriculture is one of the pillars of the strategy to conserve forest (Vanlauwe et al. 2014). Intensification in agriculture is recognised as one of the pathways to meet food security and climate change adaptation and mitigation goals (Sapkota et al. 2017c).

However, SI does not always confer co-benefits in terms of food security and climate change adaption/mitigation. For example, in the case of Vietnam, intensified production of rice and pigs reduced GHG emissions in the short term through land sparing, but after two decades, the emissions associated with higher inputs were likely to outweigh the savings from land sparing (Thu Thuy et al. 2009). Intensification needs to be sustainable in all components of food system by curbing agricultural sprawl, rebuilding soils, restoring degraded lands, reducing agricultural pollution, increasing water use efficiency, and decreasing the use of external inputs (Cook et al, 2015).

A study conducted by Palm et al. (2010) in sub-Saharan Africa, reported that at low population densities and high land availability, food security and climate mitigation goals can be met with intensification scenarios, resulting in surplus crop area for reforestation. In contrast, for high population density and small farm sizes, attaining food security and reducing GHG emissions require use of more mineral fertilisers to make land available for reforestation. However, some forms of intensification in drylands can increase rather than reduce vulnerability due to adverse effects such as environmental degradation and increased social inequity (Robinson et al. 2015).

Sustainable intensification has been critiqued for considering food security only from the supply side, whereas global food security requires attention to all aspects of food system, including access, utilisation, and stability (Godfray 2015). Further, adoption of high-input forms of agriculture under the guise of simultaneously improving yields and environmental performance will attract more investment leading to higher rate of adoption but with the environmental component of SI quickly abandoned (Godfray 2015). Where adopted, SI needs to engage with the sustainable development agenda to (i) identify SI agricultural practices that strengthen rural communities, improve smallholder livelihoods and employment, and avoid negative social and cultural impacts, including loss of land tenure and forced migration; (ii) invest in the social, financial, natural, and physical capital needed to facilitate SI implementation; and (iii) develop mechanisms to pay poor farmers for undertaking sustainability measures (e.g., GHG emissions mitigation or biodiversity protection) that may carry economic costs (Garnett et al. 2013).

In summary, integrated agricultural systems and practices can enhance food system resilience to climate change and reduce GHG emissions, while helping to achieve sustainability (*high confidence*).

Cross-Chapter Box 6: Agricultural intensification: land sparing, land sharing and sustainability

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Introduction

The projected demand for more food, fuel and fibre for a growing human population necessitates intensification of current land use to avoid conversion of additional land to agriculture and potentially allow the sparing of land to provide other ecosystem services, including carbon sequestration, production of biomass for energy, and the protection of biodiversity (Benton et al. 2018; Garnett et al. 2013). Land use intensity may be defined in terms of three components; (i) intensity of system inputs (land/soil, capital, labour, knowledge, nutrients and other chemicals), (ii) intensity of system outputs (yield per unit land area or per specific input) and (iii) the impacts of land use on ecosystem services such as changes in soil carbon or biodiversity (Erb et al. 2013). Intensified land use can lead to ecological damage as well as degradation of soil resulting in a loss of function which underpins many ecosystem services (Wilhelm and Smith 2018); (Smith et al. 2016). Therefore, there is a risk that increased agricultural intensification could deliver short-term production goals at the expense of future productive potential, jeopardising long term food security (Tilman et al. 2011).

Agroecosystems which maintain or improve the natural and human capital and services they provide may be defined as sustainable systems, while those which deplete these assets as unsustainable (Pretty and Bharucha 2014). Producing more food, fuel and fibre without the conversion of additional non-agricultural land while simultaneously reducing environmental impacts requires what has been termed sustainable intensification (Godfray et al. 2010; FAO 2011e); see glossary and Cross-Chapter Box 6, Figure 1). Sustainable intensification (SI) may be achieved through a wide variety of means; from improved nutrient and water use efficiency via plant and animal breeding programs, to the implementation of integrated soil fertility and pest management practices, as well as by smarter land use allocation at a larger spatial scale: for example, matching land use to the context and specific capabilities of the land (Benton et al. 2018). However, implementation of SI is broader than simply increasing the technical efficiency of agriculture ("doing more with less"); it sometimes may require a reduction of yields to raise sustainability, and successful implementation can be dependent on place and scale. (Pretty et al. 2018), following (Hill 1985), highlights three elements to SI: (i) increasing efficiency, (ii) substitution of less beneficial or efficient practices for better ones, and (iii) system redesign to adopt new practices and farming systems (see Cross-Chapter Box 6, Table 1).

Under a land sparing strategy, intensification of land use in some areas, generating higher productivity per unit area of land, can allow other land to provide other ecosystem services such as increased carbon sequestration and the conservation of natural ecosystems and biodiversity (Balmford et al. 2018; Strassburg et al. 2014). Conversely under a land sharing strategy less, or no, land is set aside, but lower levels of intensification are applied to agricultural land, providing a combination of provisioning and other functions such as biodiversity conservation from the same land (Green et al. 2005). The two approaches are not mutually exclusive and the suitability of their application is generally system-, scale- and/or location specific (Fischer et al. 2014). One crucial issue for the success of a land sparing strategy is that spared land is protected from further conversion: as the profits from the intensively managed land increase, there is an incentive for conversion of additional land for production (Byerlee et al. 2014). Furthermore, it is implicit that there are limits to the SI of land at a local and also planetary boundary level (Rockström et al. 2009). These may relate to the "health" of soil, the presence of supporting services, such as pollination, local limits to water availability, or limits on air quality. This implies that it may not be possible to meet demand "sustainably" if demand exceeds local and global limits. There are no single global solutions to these challenges and specific in situ responses for different farming systems and locations are required. Bajželj et al. (2014) showed that implementation of SI, primarily through yield gap closure, had better environmental outcomes compared with business as usual trajectories. However, SI alone will not be able to deliver the necessary environmental outcomes from the food system – dietary change and reduced food waste are also required (Springmann et al. 2018a; Bajželj et al. 2014).

Cross-Chapter Box 6, Table 1 Approaches to sustainable intensification of agriculture (Pretty et al. 2018; Hill 1985)

Approach	Sub-category	Examples/notes
Improving efficiency	Precision agriculture	High and low-technology options to optimise resource use.
	Genetic improvements	Improved resource use efficiency through crop or livestock breeding.
	Irrigation technology	Increase production in areas currently limited by precipitation (sustainable water supply required).
	Organisational scale-up	Increasing farm organisational scale (e.g. co-operative schemes) can increase efficiency via facilitation of mechanisation and precision techniques.
Substitution	Green fertiliser	Replacing chemical fertiliser with green manures, compost (including vermicompost), biosolids and digestate (by product of anaerobic digestion) to maintain and improve soil fertility.
	Biological control	Pest control through encouraging natural predators.
	Alternative crops	Replacement of annual with perennial crops reducing the need for soil disturbance and reducing erosion.
	Premium products	Increase farm-level income for less output by producing a premium product.
System redesign	System diversification	Implementation of alternative farming systems: organic, agroforestry and intercropping (including the use of legumes).
	Pest management	Implementing integrated pest and weed management to reduce the quantities of inputs required.
	Nutrient management	Implementing integrated nutrient management by using crop and soil specific nutrient management – guided by soil testing.
	Knowledge transfer	Using knowledge sharing and technology platforms to accelerate the uptake of good agricultural practices.

Improved efficiency – example of precision agriculture

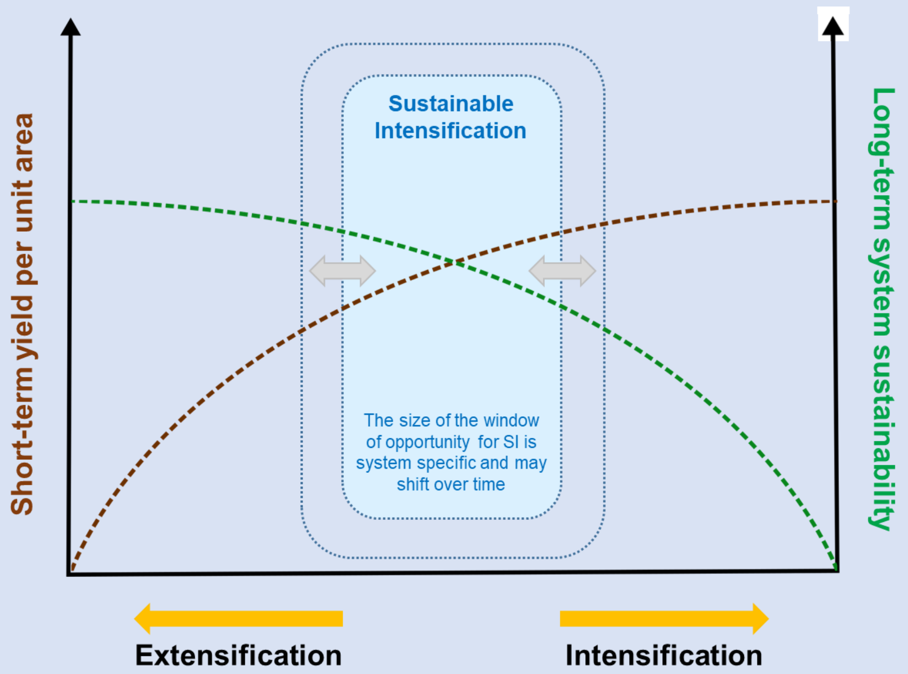
Precision farming usually refers to optimising production in fields through site-specific choices of crop varieties, agrochemical application, precise water management (e.g. in given areas or threshold moistures) and management of crops at a small scale (or livestock as individuals) (Hedley 2015). Precision agriculture has the potential to achieve higher yields in a more efficient and sustainable manner compared with traditional low-precision methods.

Precision agriculture

Precision agriculture is a technologically advanced approach that uses continual monitoring of crop and livestock performance to actively inform management practices. Precise monitoring of crop performance over the course of the growing season will enable farmers to economise on their inputs in terms of water, nutrients and pest management. Therefore, it can contribute to both the food security (by maintaining yields), sustainability (by reducing unnecessary inputs) and land sparing goals associated with SI. The site-specific management of weeds allows a more efficient application of herbicide to specific weed patches within crops (Jensen et al. 2012). Such precision weed control has resulted in herbicide savings of 19 – 22% for winter oilseed rape, 46 – 57% for sugar beet and 60 -77% for winter wheat production (Gutjahr and Gerhards 2010). The use of on-farm sensors for real time management of crop and livestock performance can enhance farm efficiency (Aqeel-Ur-Rehman et al. 2014). Mapping soil nutrition status can allow for more targeted and therefore effective nutrient management practices (Hedley 2015). Using wireless sensors to monitor environmental conditions such as soil moisture, has the potential to allow more efficient crop irrigation (Srbinovska et al. 2015). Controlled traffic farming, where farm machinery is confined to permanent tracks, using automatic steering and satellite guidance, increases yields by minimising soil compaction. However, barriers to the uptake of many of these high-tech precision agriculture technologies remain. In what is described as the 'implementation problem', despite the potential to collect vast quantities of data on crop or livestock performance, applying these data to inform management decisions remains a challenge (Lindblom et al. 2017).

Low-tech precision agriculture

The principle of precision agriculture can be applied equally to low capital-input farming, in the form of low-tech precision agriculture (Conway 2013). The principle is the same but instead of adopting capital-heavy equipment (such as sensor technology connected to the 'internet of things', or large machinery and expensive inputs), farmers use knowledge and experience and innovative approaches often re-purposed, such as a bottle cap as a fertiliser measure for each plant, applied by hand (Mondal and Basu 2009). This type of precision agriculture is particularly relevant to small-scale farming in the global South, where capital investment is major limiting factor. For example, the application of a simple seed priming technique resulted in a 20 to 30% increase in yields of pearl millet and sorghum in semi-arid West Africa (Aune et al. 2017). Low-tech precision agriculture has the potential to increase the economic return per unit land area while also creating new employment opportunities.



Cross-Chapter Box 6, Figure 1 There is a need to balance increasing demands for food, fuel and fibre with long-term sustainability of land use. Sustainable intensification can, in theory, offer a window of opportunity for the intensification of land use without causing degradation. This potentially allows the sparing of land to provide other ecosystem services, including carbon sequestration and the protection of biodiversity. However, the potential for SI is system specific and may change through time (indicated by grey arrows). Current practice may already be outside of this window and be unsustainable in terms of negative impacts on the long-term sustainability of the system

Sustainable intensification through farming system redesign

SI requires equal weight to be placed on the sustainability and intensification components (Benton 2016; Garnett et al. 2013), Cross-Chapter Box 6, Figure 1 outlines the trade-offs which SI necessitates between the intensity of land use against long-term sustainability. One approach to this challenge is through farming system redesign including increased diversification.

Diversification of intensively managed systems

Incorporating higher levels of plant diversity in agroecosystems can improve the sustainability of farming systems (Isbell et al. 2017). Where intensive land use has led to land degradation more diverse land use systems such as intercropping can provide a more sustainable land use option with co-benefits for food security, adaptation and mitigation objectives. For example, in temperate regions, highly productive agricultural grasslands used to produce meat and dairy products are characterised by monoculture pastures with high agrochemical inputs. Multi-species grasslands may provide a route to SI, as even a modest increase in species richness in intensively managed grasslands can result in higher forage yields without increased inputs, such as chemical fertiliser (Finn et al. 2013; Sanderson et al. 2013; Tilman et al. 2011). Recent evidence also indicates multispecies grasslands have greater resilience to drought, indicating co-benefits for adaptation (Hofer et al. 2016; Haughey et al. 2018).

Diversification of production systems

Agroforestry systems (see glossary) can promote regional food security and provide many additional ecosystem services when compared with monoculture crop systems. Co-benefits for mitigation and adaptation include increased carbon sequestration in soils and biomass, improved water and nutrient use efficiency and the creation of favourable micro-climates (Waldron et al. 2017). Silvopasture systems, which combine grazing of livestock and forestry, are particularly useful in reducing land degradation where the risk of soil erosion is high (Murgueitio et al. 2011). Crop and livestock systems can also be combined to provide multiple services. Perennial wheat derivatives produced both high quality forage and substantial volumes of cereal grains (Newell and Hayes 2017), and show promise for integrating cereal and livestock production while sequestering soil carbon (Ryan et al. 2018). A key feature of diverse production systems is the provision of multiple income streams for farming households, providing much needed economic resilience in the face of fluctuation of crop yields and prices.

Landscape Approaches

The land sparing and land sharing approaches which may be used to implement SI are inherently “landscape approaches” (e.g., (Hodgson et al. 2010)). While the term landscape is by no means precise (Englund et al. 2017), landscape approaches, focused for example at catchment scale, are generally agreed to be the best way to tackle competing demands for land (e.g. Sayer et al. 2013), and are the appropriate scale at which to focus the implementation of sustainable intensification. The landscape approach allots land to various uses – cropping, intensive and extensive grazing, forestry, mining, conservation, recreation, urban, industry, infrastructure – through a planning process that seeks to balance conservation and production objectives. With respect to SI, a landscape approach is pertinent to achieving potential benefits for biodiversity conservation, ensuring that land “spared” through SI remains protected, and that adverse impacts of agriculture on conservation land are minimised. Depending on the land governance mechanisms applied in the jurisdiction, different approaches will be appropriate/required. However, benefits are only assured if land use restrictions are devised and enforced.

Summary

Intensification needs to be achieved sustainably, necessitating a balance between productivity today and future potential (*high agreement, medium evidence*). Improving efficiency of agriculture systems can increase production per unit of land through greater resource use efficiency. To achieve SI some intensively managed agricultural systems may have to be diversified as they cannot be further intensified without land degradation. A combination of land sparing and sharing options can be utilised to achieve SI – their application is most likely to succeed if applied using a landscape approach.

5.6.5 Role of urban agriculture

Cities are an important actor in the food system in regard to both demand for food for urban dwellers and production of food in urban and peri-urban areas (see also cross-chapter box 4: Climate Change and Urbanisation in Chapter 2). Both the demand side and supply side roles are important relative to climate change mitigation and adaptation strategies. Urban areas are home to more than half of the world's population, and a minimal proportion of the production; thus, they are important drivers for the development of the complex food systems in place today, in regard to supply chains and dietary preferences.

The increasing separation of urban and rural populations with regard to territory and culture is one of the factors favouring the nutrition transition towards urban diets (Weber and Matthews 2008; Neira et al. 2016). These are primarily based on a high diversity of food products, independent of season and local production, and on the extension of the distances that food travels between production and consumption The transition of traditional diets to more homogeneous diets has also become tied to consumption of animal protein, which has increased GHG emissions globally (see also Section 5.4.6).

Cities are becoming key actors in developing strategies of mitigation to climate change, in their food procurement and in sustainable urban food policies alike (McPhearson et al. 2018). These are being developed by big and medium-sized cities in the world, often integrated within climate change policies (Moragues et al. 2013; Calori and Magarini 2015). A review conducted of 100 cities across the world shows that urban food consumption is one of the largest sources of urban material flows, urban carbon footprint, and land footprint (Goldstein et al. 2017). Additionally, the urban poor have limited capacity to adapt to climate-related impacts, which place their food security at risk under climate change (Dubbeling and de Zeeuw 2011).

Urban and peri-urban areas. In 2010, around 14% of the global population was nourished by food grown in urban and peri-urban areas (Kriewald et al.). A review study on sub-Saharan Africa shows that urban and peri-urban agriculture contributes to climate change adaptation and mitigation (Lwasa et al. 2014, 2015). Urban and peri-urban agriculture reduces food carbon footprint by avoiding long distance food transport and limits GHG emissions by recycling organic waste and wastewater that would otherwise releases methane from landfill and dumping sites (Lwasa et al. 2014). Urban and peri-urban agriculture also contributes in adapting to climate change including extreme events, by reducing urban heat island effect, increasing water infiltration and slowing down run-offs to prevent flooding, etc. (Lwasa et al. 2014, 2015; Kumar et al. 2017a). For example, a scenario analysis shows that urban gardens reduce the surface temperature up to 10°C in comparison to the temperature without vegetation (Tsilini et al. 2015). Urban agriculture can also improve biodiversity and strengthen associated ecosystem services (Lin et al. 2015).

Urban and peri-urban agriculture is exposed to climate risks and urban growth that may undermine its long-term potential to address urban food security (Padgham et al. 2015). Therefore, there is a need to better understand the impact of urban sprawl on peri-urban agriculture; the contribution of urban and peri-urban agriculture to food self-sufficiency of cities; the risks posed by pollutants from urban areas to agriculture and vice-versa; the global and regional extent of urban agriculture; and the role that urban agriculture could play in climate resilience and abating malnutrition (Mok et al. 2014; Hamilton et al. 2014). Globally, urban sprawl is projected to consume 1.8–2.4% and 5% of the current cultivated land by 2030 and 2050 respectively, leading to crop calorie loss of 3–4% and 6–7%, respectively (Pradhan et al. 2014; Bren d'Amour et al. 2017). Kriewald et al. shows that the urban growth has the largest impacts in most of the sub-continents (e.g., Western, Middle, and Eastern Africa) while climate change will mostly reduce potential of urban and peri-urban agriculture in Southern Europe and Northern Africa.

In summary, urban and peri-urban agriculture can contribute to improving urban food security, reducing greenhouse gas emissions, and adapting to climate change impacts *(robust evidence, medium agreement).*

5.6.6 Links to the Sustainable Development Goals

In 2015, the Sustainable Development Goals (SDGs) and the Paris Agreement were two global major international policies adopted by all countries to guide the world to overall sustainability, within the 2030 Sustainable Development Agenda and UNFCCC processes respectively. The 2030 Sustainable Development agenda includes 17 goals and 169 targets, including zero hunger, sustainable agriculture and climate action (United Nations 2015).

This section focuses on intra- and inter-linkages of SDG 2 and SDG 13 based on the official SDG indicators (Figure 5.16), showing the current conditions (see (Roy et al. 2018) and Chapter 7 for further discussion). The second goal (Zero Hunger – SDG 2) aims to end hunger and all forms of malnutrition by 2030 and commits to universal access to safe, nutritious and sufficient food at all times of the year. SDG 13 (Climate Action) calls for urgent action to combat climate change and its impacts. Integrating the SDGs into the global food system can provide opportunities for mitigation and adaptation and enhancement of food security.

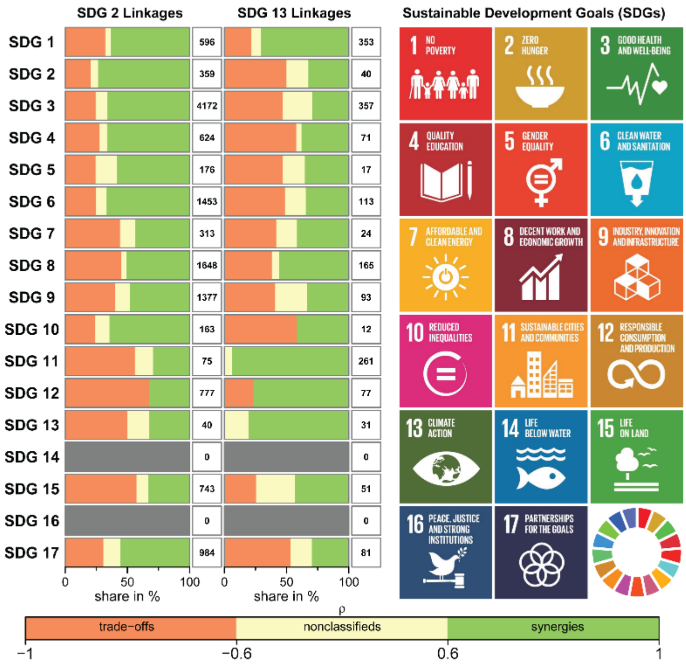


Figure 5.16. Intra and inter-linkages for SDG 2 (*Zero hunger*) and SDG 13 (*Climate action*) at the global level using the official indicators of Sustainable Development Goals that consists data for 122 indicators for a total of 227 countries between the years 1983 and 2016 (United Nations Statistics Division 2016) and applying a statistical approach (Pradhan et al. 2017). Pradhan et al. (2017) defined synergy and trade-offs as significant positive (ρ > 0.6, red bar) and negative (ρ < -0.6, green bar) spearman correlation between SDG indicators, respectively. The ρ between 0.6 and -0.6 is considered as nonclassifieds (yellow bar). The correlation between unique pairs of indicator time-series is carried based on country data, e.g., betweem "prevalence of undernourishment" (an indicator for SDG 2.1) and "maternalmortality ratio" (an indicator for SDG 3.1). The data pairs can belong to the same goal or to two distinct goals. At the global level, intra-linkages of SDGs are quantified by the percentage of synergies, trade-offs, and nonclassifieds of indicator pairs belonging to the same SDG (here, SDG 2 and SDG 13) for all the countries. Similarly, SDG interlinkages are estimated by the percentage of synergies, trade-offs, and nonclassifieds between indicator pairs that fall into two distinct goals for all the countries. The grey bar shows insufficient data for analysis. The number of data pair used for the analysis is presented in the grey box.

Ensuring food security (SDG 2) shows positive relations (synergies) with most goals (Pradhan et al. 2017; International Council for Science (ICSU) 2017), but has trade-offs with SDG 12 (*Responsible Consumption and Production*) and SDG 15 (*Life on Land*) under current development paradigms

(Pradhan et al. 2017). Sustainable transformation of traditional consumption and production approaches can overcome these trade-offs based on several innovative methods (Shove et al. 2012). For example, sustainable intensification and reduction of food waste can minimise the observed negative relations between SDG 2 and other goals (Obersteiner et al. 2016) (see also Cross-Chapter Box 6 and Section 5.5.2). Achieving the target 12.3 of SDG 12 (Responsible Consumption and Production) "by 2030, to halve per capita global food waste at the retail and consumer levels and reduce food losses along production and supply chains, including post-harvest losses" will contribute to climate change mitigation.

Doubling productivity of smallholder farmers and halving food loss and waste by 2030 are targets of SDG 2 and SDG 12, respectively (United Nations Statistics Division 2016). Agroforestry that promotes biodiversity and sustainable land management also contributes to food security (Montagnini and Metzel 2017). Land restoration and protection (SDG 15) can increase crop productivity (SDG 2) (Wolff et al. 2018). Similarly, efficient irrigation practices can reduce water demand for agriculture that could improve health of the freshwater ecosystem (SDG 6 and SDG 15) without reducing food production (Jägermeyr et al. 2017).

Climate action (SDG 13) shows negative relations (trade-offs) with most goals and antagonistic to the 2030 development agenda under the current development paradigm (Figure 5.16) (Lusseau and Mancini 2019; Pradhan 2019). The targets for SDG 13 have a high focus on climate change adaptation and the data for the SDG 13 indicators are limitedly available. SDG 13 shares two indicators with SDG 1 and SDG 11 (United Nations 2017) and therefore, has mainly positive linkages with these two goals. Trade-offs was observed between SDG 2 and SDG 13 for around 50% of the case (Pradhan et al. 2017).

Transformation from current development paradigms and breaking of these lock-in effects can protect climate and achieve food security in future. Sustainable agriculture practices can provide climate change adaptation and mitigation synergies, linking SDG 2 and SDG 13 more positively (International Council for Science (ICSU) 2017). IPCC highlights that most of the current observed trade-offs between SDG 13 and other SDGs can be converted into synergies based on various mitigation options that can be deployed to limit the global warming well below 1.5°C (IPCC 2018b).

In summary, there are fundamental synergies that can facilitate the joint implementation of strategies to achieve SDGs and climate action, with particular reference to those climate response strategies related to both supply side (production and supply chains) and demand side (consumption and dietary choices) described in this chapter (*high agreement and medium evidence*).

5.7 Enabling conditions and knowledge gaps

To achieve mitigation and adaptation to climate change in food systems, enabling conditions are needed to scale up the adoption of effective strategies (such as those described in Sections 5.3 to 5.6 and Chapter 6). These enabling conditions include multi-level governance and multi-sector institutions (Supplementary Material Section SM5.7) and multiple policy pathways (Section 5.7.1, 5.7.2). In this regard, the subnational level is gaining relevance both in food systems and climate change. Just Transitions are needed to address both climate change and food security (Section 5.7.3). Mobilisation of knowledge, education, and capacity will be required (Section 5.7.4) to fill knowledge gaps (Section 5.7.5).

Effective governance of food systems and climate change requires the establishment of institutions responsible for coordinating among multiple sectors (education, agriculture, environment, welfare, consumption, economic, health), levels (local, regional, national, global) and actors (governments, CSO, public sector, private sector, international bodies). Positive outcomes will be enegenderd by participation, learning, flexibility, and cooperation. See Supplementary Material Section SM5.7 for further discussion.

5.7.1 Enabling policy environments

The scope for responses to make sustainable land use inclusive of climate change mitigation and adaptation, and the policies to implement them, are covered in detail in Chapters 6 and 7. Here we highlight some of the major policy areas that have shaped the food system, and might be able to shape responses in future. Although two families of policy – agriculture and trade – have been instrumental in shaping the food system in the past (and potentially have led to conditions that increase climate vulnerability) (Benton and Bailey 2019), a much wider family of policy instruments can be deployed to reconfigure the food system to deliver healthy diets in a sustainable way.

5.7.1.1 Agriculture and trade policy

Agriculture. The thrust of agricultural policies over the last 50 years has been to increase productivity, even if at the expense of environmental sustainability (Benton and Bailey 2019). For example, in 2007-9 46% of OECD support for agriculture was based on measures of output (price support or payments based on yields), 37% of support was based on the current or historical area planted, herd size (or correlated measures of the notional costs of farming), and 13% was payments linked to input prices. In a similar vein, non-OECD countries have promoted productivity growth for their agricultural sectors.

Trade. Along with agricultural policy to grow productivity, the development of frameworks to liberalise trade (such as the General Agreement on Tariffs and Trade (GATT) Uruguay Round, now incorporated into the World Trade Organisation) have been essential in stimulating the growth of a globalised food system. Almost every country has a reliance on trade to fulfil some or all of its local food needs, and trade networks have grown to be highly complex (Puma et al. 2015; MacDonald et al. 2015; Fader et al. 2013; Ercsey-Ravasz et al. 2012). This is because many countries lack the capacity to produce sufficient food due to climatic conditions, soil quality, water constraints, and availability of farmland (FAO 2015b). In a world of liberalised trade, using comparative advantage to maximise production in high-yielding commodities, exporting excess production, and importing supplies of other goods supports economic growth.

City states as well as many small island states, do not have adequate farmland to feed their populations, while sub-Saharan African countries are projected experience high population growth as well as to be negatively impacted by climate change, and thus will likely find it difficult to produce all of their own food supplies (Agarwal et al. 2002). One study estimates that some 66 countries are currently incapable of being self-sufficient in food (Pradhan et al. 2014). Estimates of the proportion of people relying on trade for basic food security vary from ~16% to ~22% (Fader et al. 2013; Pradhan et al. 2014), with this figure rising to between 1.5 and 6 billion people by 2050, depending on dietary shifts, agricultural gains, and climate impacts (Pradhan et al. 2014).

Global trade is therefore essential for achieving food and nutrition security under climate change because it provides a mechanism for enhancing the efficiency of supply chains, reducing the vulnerability of food availability to changes in local weather, and moving production from areas of surplus to areas of deficit (FAO 2018d). However, the benefits of trade will only be realised if trade is managed in ways that maximise broadened access to new markets while minimising the risks of increased exposure to international competition and market volatility (Challinor et al. 2018; Brown et al. 2017b).

As described in Section 5.8.1, trade acts to buffer exposure to climate risks when the market works well. Under certain conditions – such as shocks, or the perception of a shock, coupled with a lack of food stocks or lack of transparency about stocks (Challinor et al. 2018; Marchand et al. 2016) – the market can fail and trade can expose countries to food price shocks.

Furthermore, Clapp (2016) showed that trade, often supported by high levels of subsidy support to agriculture in some countries, can depress world prices and reduce incomes for other agricultural exporters. Lower food prices that result from subsidy support may benefit urban consumers in importing countries, but at the same time they may hurt farmers' incomes in those same countries. The outmigration of smallholder farmers from the agriculture sector across the Global South is significantly attributed to these trade patterns of cheap food imports (Wittman 2011; McMichael 2014; Akram-Lodhi and others 2013). Food production and trade cartels, as well as financial speculation on food futures markets, affect low-income market-dependent populations.

Food sovereignty is a framing developed to conceptualise these issues (Reuter 2015). They directly relate to the ability of local communities and nations to build their food systems, based among other aspects, on diversified crops and indigenous and local knowledge. If a country enters international markets by growing more commodity crops and reducing local crop varieties, it may get economic benefits, but may also expose itself to climate risks and food insecurity by increasing reliance on trade, which may be increasingly disrupted by climate risks. These include a local lack of resilience from reduced diversity of products, but also exposure to food price spikes, which can become amplified by market mechanisms such as speculation.

In summary, countries must determine the balance between locally produced vs imported food (and feed) such that it both minimises climate risks and ensures sustainable food security. There is *medium evidence* that trade has positive benefits but also creates exposure to risks (Section 5.3).

5.7.1.2 Scope for expanded policies

There are a range of ways that policy can intervene to stimulate change in the food system – through agriculture, research and development, food standards, manufacture and storage, changing the food environment and access to food, changing practices to encourage or discourage trade (Table 5.6). Novel incentives can stimulate the market, for example, through reduction in waste or changes in diets to gain benefits from a health or sustainability direction. Different contexts with different needs will require different set of policies at local, regional and national levels. See Supplementary Material Section SM5.7 for further discussion on expanded policies.

Table 5.6 Potential policy "families" for food-related adaptation and mitigation of climate change. The column "scale" refers to scale of implementation: International (I), national (N), sub-national-regional (R), and local (L).

Family	Sub-family	Scale	Interventions	Examples
Supply-side efficiency	Increasing agricultural efficiency and yields	I, N	Agricultural R&D	Investment in research, innovation, knowledge exchange, e.g., on genetics, yield gaps, resilience
		I, N	Supporting precision agriculture	Agricultural engineering, robotics, big data, remote sensing, inputs
		I, N	Sustainable intensification	Soils, nutrients, capital, labour (see

			projects	Cross-Chapter Box 6)
		N, R	Improving farmer training and knowledge sharing	Extension services, online access, field schools, farmer-to-farmer networks (CABI 2019)
	Land use planning	N, R, L	Land use planning for ecosystem services (remote sensing, indigenous and local knowledge	Zoning, protected area networks, multifunctional landscapes, "land sparing" (see Cross-Chapter Box 6; Benton et al. 2018; Jones et al. 2013)
		N, R, L	Conservation agriculture programs	Soil and water erosion control, soil quality improvement (Conservation Evidence 2019)
		N	Payment for ecosystem services	Incentives for farmers/landowners to choose lower-profit but environmentally benign resource use, e.g., Los Negros Valley in Bolivia (Ezzine-de-Blas et al. 2016)
	Market approaches	I, N	Mandated carbon cost reporting in supply chains; public/private incentivised insurance products	Carbon and natural capital accounts (CDP 2019), crop insurance (Müller et al. 2017a)
	Trade	I	Liberalising trade flows; green trade	Reduction in GHG emissions from supply chains (Neumayer 2001)
Raising profitability and quality	Stimulating markets for premium goods	N, R	Sustainable farming standards, agroecology projects, local food movements	Regional policy development, public procurement of sustainable food (Mairie de Paris 2015)
Modifying demand	Reducing food waste	I, N, L	Regulations, taxes	'Pay-As-You-Throw (PAYT)' schemes; EU Landfill Directives; Japan Food Waste Recycling Law 2008; South Africa Draft Waste Classification and Management Regulations 2010 (Chalak et al. 2016)
		I, N, L	Awareness campaigns, education	FAO Global Initiative on Food Loss and Waste Reduction (FAO 2019b)
		I, N	Funding for reducing food waste	Research and investment for shelf life, processing, packaging, cold storage (MOFPI 2019)
		I, N, L	Circular economy using waste as inputs	Biofuels, distribution of excess food to charities (Baglioni et al. 2017)
	Reducing consumptio	I, N, L	Carbon pricing for selected food commodities	Food prices reflective of GHG gas emissions throughout production and

	n of carbon-intensive food			supply chain (Springmann et al. 2017; Hasegawa et al. 2018)
		I, N, L	Changing food choice through education	Nutritional and portion-size labelling, 'nudge' strategies (positive reinforcement, indirect suggestion) (Arno and Thomas 2016)
		I, N, L	Changing food choices through money transfers	Unconditional cash transfers; e-vouchers exchanged for set quantity or value of specific, pre-selected goods (Fenn 2018)
		N, L	Changing food environments through planning	Farmers markets, community food production, addressing 'food deserts' (Ross et al. 2014)
	Combining carbon and health objectives	I, N, L	Changing subsidies, standards, regulations to healthier and more sustainably produced foods	USDA's "Smart Snacks for School" regulation mandating nutritional guidelines (USDA 2016) Incentivising production via subsidies (direct to producer based on output or indirect via subsidising inputs)
		N	Preventative vs curative public health care incentives	Health insurance cost reductions for healthy and sustainable diets
		I, N, L	Food system labelling	Organic certification, nutrition labels, blockchain ledgers (Chadwick 2017)
		N, L	Education and awareness campaigns	School curricula; public awareness campaigns
		N, L	Investment in disruptive technologies (e.g., cultured meat)	Tax breaks for R&D, industrial strategies (European Union 2018)
		N, L	Public procurement	For health: Public Procurement of Food for Health (Caldeira et al. 2017) For environment: Paris Sustainable Food Plan 2015-2020 Public Procurement Code (Mairie de Paris 2015)

In summary, although agriculture is often thought to be shaped predominantly by agriculture and trade policies, there are over twenty families of policy areas that can shape agricultural production directly or indirectly (through environmental regulations or through markets, including by shaping consumer behaviour). Thus, delivering outcomes promoting climate change adaptation and mitigation can arise from policies across many departments, if suitably designed and aligned.

5.7.1.3 Health-related policies and cost savings

The co-benefits arising from mitigating climate change through changing dietary patterns, and thus demand, have potentially important economic impacts (*high confidence*). The gross value added from agriculture to the global economy (GVA) was USD 1.9tn (in 2013 (FAO 2015c)), from a global agriculture economy (GDP) of USD 2.7tn (in 2016). In 2013, the FAO estimated an annual cost of USD 3.5tn for malnutrition (FAO 2013a).

However, this is likely to be an underestimate of the economic health costs of current food systems for several reasons: (1) Lack of data – for example there is little robust data in the UK on the prevalence of malnutrition in the general population (beyond estimates of obesity and surveys of malnourishment of patients in hospital and care homes, from which estimates over 3 million people in the UK are undernourished (BAPEN 2012)); (2) Lack of robust methodology to determine, for example, the exact relationship between overconsumption of poor diets, obesity and non-communicable diseases like diabetes, cardio-vascular disease, a range of cancers or Alzheimer's disease (Pedditizi et al. 2016), (3) Unequal healthcare spending around the world.

In the US, the economic cost of diabetes, a disease strongly associated with obesity and affecting about 23 million Americans, is estimated at USD 327bn in 2017 (American Diabetes Association 2018), with direct healthcare costs of USD 9,600 per person. By 2025, it is estimated that globally there will be over 700 million people with diabetes (NCD-RisC 2016b), over 30 times the number in the US. Even if a global average cost of diabetes per capita were a quarter of that in the US, the total economic cost of diabetes would be approximately the same as global agricultural GDP. Finally, (4) the role of agriculture in causing ill-health beyond dietary health, such as through degrading air quality (e.g., (Paulot and Jacob 2014)).

Whilst data of the healthcare costs associated with the food system and diets are scattered and the proportion of costs directly attributable to diets and food consumption is uncertain, there is potential for more preventative healthcare systems to save significant costs that could incentivise agricultural business models to change what is grown, and how. The potential of moving towards more preventative healthcare is widely discussed in the health economics literature, particularly in order to reduce the life-style-related (including dietary-related) disease component in aging populations (e.g., (Bloom et al. 2015)).

5.7.1.4 Multiple policy pathways

As discussed in more detail in Chapters 6 and 7, there is a wide potential suite of interventions and policies that can potentially enhance the adaptation of food systems to climate change, as well as enhance the mitigation potential of food systems on climate change. There is an increasing number of studies that argue that the key to sustainable land management is not in land management practices but in the factors that determine the demand for products from land (such as food). Public health policy therefore has the potential to affect dietary choice and thus the demand for different amounts of, and types of, food.

Obersteiner et al. (2016) show that increasing the average price of food is an important policy lever that, by reducing demand, reduces food waste, pressure on land and water, impacts on biodiversity and through reducing emissions, mitigates climate change and potentially helps to achieve multiple SDGs. Whilst such policy responses – such as a carbon tax applied to goods including food – has the potential to be regressive, affecting the poor differentially (Frank et al. 2017; Hasegawa et al. 2018;

Kehlbacher et al. 2016), and increasing food insecurity – further development of social safety nets can help to avoid the regressive nature (Hasegawa et al. 2018). Hasegawa et al. (2018) point out that such safety nets for vulnerable populations could be funded from the revenues arising from a carbon tax.

The evidence suggests, as with SR15 (IPCC 2018a) and its multiple pathways to climate change solutions, that there is no single solution that will address the problems of food and climate change, but instead there is a need to deploy many solutions simultaneously adapted to the needs and options available in a given context. For example, Springmann et al. (2018a) indicate that maintaining the food system within planetary boundaries at mid-century, including equitable climate, requires increasing the production (and resilience) of agricultural outputs (i.e., closing yield gaps), reducing waste, and changes in diets towards ones often described as flexitarian (low-meat dietary patterns that are in line with available evidence on healthy eating). Such changes can have significant co-benefits for public health, as well as facing significant challenges to ensure equity (in terms of affordability for those in poverty).

Significant changes in the food system require them to be acceptable to the public ("public license"), or they will be rejected. Focus groups with members of the public around the world, on the issue of changing diets, have shown that there is a general belief that the government plays a key role in leading efforts for change in consumption patterns (Wellesley et al. 2015). If governments are not leading on an issue, or indicating the need for it through leading public dialogue, it signals to their citizens that the issue is unimportant or undeserving of concern

In summary, there is significant potential *(high confidence)* that, through aligning multiple policy goals, multiple benefits can be realised that positively impact public health, mitigation and adaptation (e.g. adoption of healthier diets, reduction in waste, reduction in environmental impact). These benefits may not occur without the alignment across multiple policy areas *(high confidence)*.

5.7.2 Enablers for changing markets and trade

"Demand" for food is not an exogenous variable to the food system but is shaped crucially by its ability to produce, market, and supply food of different types and prices. These market dynamics can be influenced by a variety of factors beyond consumer preferences (e.g., corporate power and marketing, transparency, the food environment more generally), and the ability to reshape the market can also depend on its internal resilience and/or external shocks (Challinor et al. 2018; Oliver et al. 2018)).

5.7.2.1 Capital markets

Two areas are often discussed in regard to role of capital markets in shaping the food system. First, investment in disruptive technologies might stimulate climate-smart food systems (WEF/McKinsey & Company 2018; Bailey and Wellesley 2017), including alternative proteins, such as laboratory or "clean meat" (which has significant ability to impact on land use requirements) (Alexander et al. 2017) (See Section 5.5.1.6). An innovation environment through which disruptive technology can emerge typically requires the support of public policy, whether in directly financing small and emerging enterprises, or funding research and development via reducing tax burdens.

Second, widespread adoption of (and perhaps underpinned by regulation for) natural capital accounting as well as financial accounting are needed. Investors can then be aware of the risk exposure of institutions, which can undermine sustainability through externalising costs onto the environment. The prime example of this in the realm of climate change is the Carbon Disclosure Project, with around 2500 companies voluntarily disclosing their carbon footprint, representing nearly 60% of the world's market capital (CDP 2018).

5.7.2.2 Insurance and re-insurance

The insurance industry can incentivise actors' behaviour towards greater climate mitigation or adaptation, including building resilience. For example, Lloyd's of London analysed the implications of extreme weather for the insurance market, and conclude that the insurance industry needs to examine their exposure to risks through the food supply chain and develop innovative risk-sharing products can make an important contribution to resilience of the global food system (Lloyd 2015).

Many of these potential areas for enabling healthy and sustainable food systems are also knowledge gaps, in that whilst the levers are widely known, their efficacy and the ability to scale-up, in any given context, are poorly understood.

5.7.3 Just transitions to sustainability

Research is limited on how land use transitions would proceed from ruminant production to other socio-ecological farming systems. Ruminants have been associated with humans since the early development of agriculture, and the role of ruminants in many agricultural systems and smallholder communities is substantial. Ruminant production systems have been adapted to a wide range of socioeconomic and environmental conditions in crop, forestry, and food processing settings (Čolović et al. 2019), bioenergy production (de Souza et al. 2019), and food waste recycling (Westendorf 2000). Pasture cultivation in succession to crops is recognised as important to management of pest and diseases cycles and to improve soil carbon stocks and soil quality (Carvalho and Dedieu 2014). Grazing livestock is important as a reserve of food and economic stocks for some smallholders (Ouma et al. 2003).

Possible land-use options for transitions away from livestock production in a range of systems include (a) retain land but reduce investments to run a more extensive production system; (b) change land use by adopting a different production activity; (c) abandon land (or part of the farm) to allow secondary vegetation regrowth (Carvalho et al. 2019; Laue and Arima 2016); and (d) invest in afforestation or reforestation (Baynes et al. 2017). The extensification option could lead to increases rather than decreases in GHG emissions related to reduction in beef consumption. Large-scale abandonment, afforestation, or reforestation would probably have more positive environmental outcomes, but could result in economic and social issues that would require governmental subsidies to avoid decline and migration in some regions (Henderson et al. 2018). Alternative economic use of land, such as bioenergy production, could balance the negative socioeconomic impact of reducing beef output, reduce the tax values needed to reduce consumption, and avoid extensification of ruminant production systems (Wirsenius et al. 2011). However, the analysis of the transition of land use for ruminants to other agricultural production systems is still a literature gap (Cross-Chapter Box 7: Bioenergy and BECCS in mitigation scenarios, in Chapter 6).

Finally, it is important to recognise that, while energy alternatives produce the same function for the consumer, it is questionable that providing the same nutritional value through an optimised mix of dietary ingredients provides the same utility for humans. Food has a central role in human pleasure, socialisation, cultural identity, and health (Röös et al. 2017), including some of the most vulnerable groups, so just transitions and their costs need to be taken into account. Pilot projects are important to provide greater insights for large-scale policy design, implementation, and enforcement.

In summary, more research is needed on how land use transitions would proceed from ruminant production to other farming systems and affect the farmers and other food system actors involved. There is *limited evidence* on what the decisions of farmers under lower beef demand would be.

5.7.4 Mobilising knowledge

Addressing climate change-related challenges and ensuring food security requires all types of knowledge (formal/non-formal, scientific/indigenous, women, youth, technological). Miles et al. (2017) stated that a research and policy feedback that allows transitions to sustainable food systems must have at first a whole system approach. Currently, in transmitting knowledge for food security and land sustainability under climate change there are three major approaches: (1) public technology transfer with demonstration (extension agents); (2) public and private advisory services (for intensification techniques) and; (3) non-formal education with many different variants such as farmers field schools, rural resource centers; facilitation extension where front-line agents primarily work as "knowledge brokers" in facilitating the teaching–learning process among all types of farmers (including women and rural young people), or farmer-to-farmer, where farmers act themselves as knowledge transfer and sharing actors through peer processes.

5.7.4.1 Indigenous and local knowledge

Recent discourse has a strong orientation towards scaling-up innovation and adoption by local farmers. However, autonomous adaptation, indigenous knowledge and local knowledge are both important for agricultural adaptation (Biggs et al. 2013) (See Section 5.3). These involve the promotion of farmer participation in governance structures, research, and the design of systems for the generation and dissemination of knowledge and technology, so that farmers needs and knowledge can be taken into consideration. Klenk et al. (2017) found that mobilisation of local knowledge can inform adaptation decision-making and may facilitate greater flexibility in government-funded research. As an example, rural innovation in terrace agriculture developed on the basis of a local coping mechanism and adopted by peasant farmers in Latin America may serve as an adaptation option or starting place for learning about climate change responses(Bocco and Napoletano 2017). Clemens et al. (2015) found that an open dialogue platform enabled horizontal exchange of ideas and alliances for social learning and knowledge-sharing in Vietnam. Improving local technologies in a participatory manner, through on-farm experimentation, farmer-to-farmer exchange, consideration of women and youths, is also relevant in mobilising knowledge and technologies.

5.7.4.2 Citizen science

Citizen science has been tested as a useful tool with potential for biodiversity conservation (Schmitz et al. 2015) and mobilising knowledge from society. In food systems, knowledge-holders (e.g., farmers and pastoralists) are trained to gather scientific data in order to promote conservation and resource management (Fulton et al. 2019) or to conserve and use traditional knowledge in developed countries relevant to climate change adaptation and mitigation through the use of ICT (Calvet-Mir et al. 2018).

5.7.4.3 Capacity building and education

Mobilising knowledge may also require significant efforts on capacity building and education to scale up food system responses to climate change. This may involve increasing the capacity of farmers to manage current climate risks and to mitigate and adapt in their local contexts, and of citizens and consumers to understand the links between food demand and climate change emissions and impacts, as well as policy makers to take a systemic view of the issues. Capacity building may also require institutional change. For example, alignment of policies towards sustainable and healthy food systems may require building institutional capacity across policy silos.

As a tool for societal transformation, education is a powerful strategy to accelerate changes in the way we produce and consume food. Education refers to early learning and life-long acquisition of skills for higher awareness and actions for solving food system challenges (FAO 2005). Education also entails, vocational training, research and institutional strengthening (Hollinger 2015). Educational focus changes according to the supply side (e.g., crop selection, input resource management, yield improvement, and diversification) and the demand since (nutrition and dietary health implications). Education on food loss and waste spans both the supply and demand sides.

In developing countries, extension learning such as Farmer Field Schools – also known as Rural Resources Centers – are established to promote experiential learning on improved production and food transformation (FAO 2016c). While in developed countries, mass education campaigns are rising to reduce food waste, improved diets or acceptable food, and ultimately changes the structure of food industries that is based on the large-scale food products (Heller 2019; UNCCD 2017).

The design of new education modules from primary to secondary to tertiary education could help create new jobs in the realm of sustainability (e.g., certification programs). For example, one area could be educating managers of recycling programs for food-efficient cities where food and organic waste are recycled to fertilisers (Jara-Samaniego et al. 2017). Research and education need to be coordinated so that knowledge gaps can be filled and greater trust established in shifting behavior of individuals from conventional options to more sustainable ones. Education campaigns can also influence policy and legislation, and help to advance successful outcomes for climate change mitigation and adaptation in regard to supply-side innovations, technologies, trade, and investment, and demand-side evolution of food choices for health and sustainability, and greater gender equality throughout the entire food system. (Heller 2019).

5.7.5 Knowledge gaps and key research areas

Knowledge gaps around options and solutions and their (co-)benefits and trade-offs are increasingly important now that implementation of mitigation and adaptation measures is scaling up.

Research is needed on how a changing climate and interventions to respond to it will affect all aspects of food security, including access, utilisation and stability, not just availability. Knowledge gaps across all the food security pillars are one of the barriers hindering mitigation and adaptation to climate change in the food system and its capacity to deliver food security. The key areas for climate change, food systems, and food security research are enlisted below.

5.7.5.1 Impacts and adaptation

Climate Services (Food availability). Agriculture and food security is a priority area for the Global Framework for Climate Services (GFCS) a program of the World Meteorological Organization (WMO). The GFCS enables vulnerable sectors and populations to better manage climate variability and adapt to climate change (Hansen et al. 2018). Global precipitation datasets and remote sensing technologies can be used to detect local to regional anomalies in precipitation as a tool for devising early-warning systems for drought-related impacts, such as famine (Huntington et al. 2017). Crop model improvements are needed for evapotranspiration to guide crop water management in future climate (Cammarano et al. 2016).

Crop and livestock genetics (Food availability, utilisation). Advances in plant breeding are crucial for enhancing food security under changing climate for a wide variety of crops including fruits and vegetables as well as staples. Genetics improvement is needed in order to breed crops and livestock that can both reduce greenhouse gas emissions, increase drought and heat tolerance (e.g., rice), and enhance nutrition and food security (Nankishore and Farrell 2016; Kole et al. 2015). Many of these characteristics already exist in traditional varieties, including orphan crops and autochthonous breeds, so research in needed to recuperate such varieties and evaluate their potential for adaptation and mitigation.

Phenomics-assisted breeding appears to be a promising tool for deciphering the stress responsiveness of crop and animal species (Papageorgiou 2017; Kole et al. 2015; Lopes et al. 2015; Boettcher et al. 2015). Initially discovered in bacteria and archaea, CRISPR–Cas9 is an adaptive immune system found in prokaryotes and since 2013 has been used as a genome editing tool in plants. The main use of CRISPR systems is to achieve improved yield performance, biofortification, biotic and abiotic stress tolerance, with rice (Oryza sativa) being the most studied crop (Gao 2018; Ricroch et al. 2017).

Climate impact models (Food availability). Understanding the full range of climate impacts on staple crops (especially those important in developing countries), fruits and vegetables is missing in the current climate impact models. Further, CO_2 effects on nutrition quality of different crops are just beginning to be parameterised in the models (Müller et al. 2014). Bridging these gap is essential for projecting future dietary diversity, healthy diets, and food security (Bisbis et al. 2018). Crop model improvements are needed for evapotranspiration to guide crop water management in future climate (Cammarano et al. 2016). Similarly, mores studies are needed to understand the impacts of climate change on global rangelands and livestock and aquaculture, which have received comparatively less attention than the impacts on crop production.

Resilience to extreme events (Food availability, access, utilisation, and stability). On the adaptation side, knowledge gaps include impacts of climate shocks (Rodríguez Osuna et al. 2014) as opposed to impacts of slow-onset climate change, how climate-related harvest failures in one continent may influence food security outcomes in others, impacts of climate change on fruits and vegetables and their nutrient contents.

5.7.5.2 Emissions and mitigation

GHG emissions inventory techniques (Food utilisation). Knowledge gaps include food consumption-based emissions at national scales, embedded emissions (overseas footprints) of food systems, comparison of GHG emissions per type of food systems (e.g., smallholder and large-scale commercial food system), and GHG emissions from land-based aquaculture. An additional knowledge gap is the need for more socio-economic assessments of the potential of various integrated practices to deliver the mitigation potential estimated from a biophysical perspective. While studies often project how much CO_2 could theoretically be sequestered in soil, for instance, there is not yet discussion of the potential for this to be effectively monitored, verified, and implemented, once barriers and incentives to adoption of the techniques, practices, and technologies are considered. Thus, future research needs fill the gaps on evaluation of climate actions in the food system.

Food supply chains (Food availability). The expansion of the cold chain into developing economies means increased energy consumption and GHG emissions at the consumer stages of the food system, but its net impact on GHG emissions for food systems as a whole is complex and uncertain (Heard and Miller 2016). Further understanding of negative side effects in intensive food processing systems is still needed.

Blockchains, as a distributed digital ledger technology which ensures transparency, traceability, and security, is showing promise for easing some global food supply chain management challenges, including the need for documentation of sustainability and the circular economy for stakeholders including governments, communities, and consumers to meet sustainability goals. Blockchain-led transformation of food supply chains is still in its early stages; research is needed on overcoming barriers to adoption (Tripoli and Schmidhuber 2018; Casado-Vara et al. 2018; Mao et al. 2018; Saberi et al. 2019).

5.7.5.3 Synergies and trade-offs

Supply-side and demand-side mitigation and adaptation (Food availability, utilisation). Knowledge gaps exit on potential and risk associated with novel mitigation technologies on supply side (e.g., inhibitors, targeted breeding, cellular agriculture, etc.). Additionally, most integrated assessment models (IAMs) currently have limited regional data on BECCS projects because of little BECCS implementation (Lenzi et al. 2018). Hence, several BECCS scenarios seem to rely on unrealistic assumptions regarding regional climate, soils and infrastructure suitability (Köberle et al. 2019) as well as trade of international trade of bioenergy (Lamers et al. 2011).

Areas for study include how to incentivise, regulate, and raise awareness on the co-benefits of healthy consumption patterns and climate change mitigation and adaptation; to improve access to healthy diets for vulnerable groups through food assistance programs; and to implement policies and campaigns to reduce food loss and food waste. Knowledge gaps also exist on the role of different policies, and underlying uncertainties, to promote changes in food habits towards climate resilience and healthy diets.

Food systems, land use change, and telecoupling (Food availability, access, utilisation). The analytical framework of telecoupling has recently been proposed to address this complexity, particularly the connections, flows, and feedbacks characterising food systems (Friis et al. 2016; Easter et al. 2018). For example, how will climate-induced shifts in livestock and crop diseases affect food production and consumption in the future. Investigating the social and ecological consequences of these changes will contribute to decision making under uncertainty in the future. Research areas include food systems and their boundaries, hierarchies, and scales through metabolism studies, political ecology and cultural anthropology.

Food-Energy-Water Nexus (Food availability, utilisation, stability). Emerging interdisciplinary science efforts are providing new understanding of the interdependence of food, energy, and water systems and these interdependencies are beginning to take into account climate change, food security, and AFOLU assessments (Scanlon et al. 2017; Liu et al. 2017). These science advances, in turn, provide critical information for coordinated management to improve the affordability, reliability, and environmental sustainability of food, energy, and water systems. Despite significant advances within the past decade, there are still many challenges for the scientific community. These include the need for interdisciplinary science related to the food-energy-water nexus; ground-based monitoring and modelling at local-to-regional scales (Van Gaelen et al. 2017); incorporating human and institutional behaviour in models; partnerships among universities, industry, and government to develop policy-relevant data; and systems modelling to evaluate trade-offs associated with food-energy-water decisions (Scanlon et al. 2017). However, the nexus approach, as a conceptual framework, requires the recognition that, although land and the goods and services it provides is finite, potential demand for the goods and services may be greater than the ability to supply them sustainably (Benton et al. 2018). By addressing demand-side issues, as well as supply-side efficiencies, it provides a potential route for minimising trade-offs for different goods and services (Benton et al. 2018) and (Section 5.6).

5.8 Future challenges to food security

A particular concern in regard to the future of food security is the potential for the impacts of increasing climate extremes on food production to contribute to multi-factored complex events such as food price spikes. In this section, we assess literature on food price spikes and potential strategies for increasing resilience to such occurrences. We then assess the potential for such food system events to affect migration and conflict.

5.8.1 Food price spikes

Under average conditions, global food system markets may function well, and equilibrium approaches can estimate demand and supply with some confidence; however, if there is a significant shock, the market can fail to smoothly link demand and supply through price, and a range of factors can act to amplify the effects of the shock, and transmit it across the world (Box 5.5). Given the potential for shocks driven by changing patterns of extreme weather to increase with climate change, there is the potential for market volatility to disrupt food supply through creating food price spikes. This potential is exacerbated by the interconnectedness of the food system (Puma et al. 2015) with other sectors (i.e., the food system depends on water, energy, transport, etc.) (Homer-Dixon et al. 2015), so the impact of shocks can propagate across sectors and geographies (Homer-Dixon et al. 2015). There is also less spare land globally than there has been in the past, such that if prices spike, there are fewer options to bring new production on stream (Marianela et al. 2016).

Increasing extreme weather events can disrupt production and transport logistics. For example, in 2012 the US Corn Belt suffered a widespread drought; US corn yield declined 16% compared to 2011 and 25% compared to 2009. A record yield loss of 2016 in French that is attributed to a conjunction of abnormal warmness in late autumn and abnormal wet in the following spring (Ben-Ari et al. 2018) is another well-documented example. To the extent that such supply shocks are associated with climate change, they may become more frequent and contribute to greater instability in agricultural markets in the future. Furthermore, analogue conditions of past extremes might create significantly greater impacts in a warmer world. A study simulating analogous conditions to the Dustbowl drought in today’s agriculture suggests that Dust-Bowl-type droughts today would have unprecedented consequences, with yield losses about 50% larger than the severe drought of 2012 (Glotter and Elliott 2016). Damages at these extremes are highly sensitive to temperature, worsening by about 25% with each degree centigrade of warming. By mid-century, over 80% of summers are projected to have average temperatures that are likely to exceed the hottest summer in the Dustbowl years (1936) (Glotter and Elliott 2016).

How a shortfall in production – or an interruption in trade due to an event affecting a logistics choke-point (Wellesley et al. 2017) – of any given magnitude may create impacts depends on many interacting factors (Homer-Dixon et al. 015; Tadasse et al. 2016; Challinor et al. 2018). The principal route is by affecting agricultural commodity markets, which respond to a perturbation through multiple routes as in Figure 5.17**Error! Reference source not found.** This includes pressures from ther sectors (such as if biofuels policy is incentivising crops for the production of ethanol, as happened in 2007–2008). The market response can be amplified by poor policies, setting up trade and non-trade barriers to exports, from countries seeking to ensure their local food security (Bailey et al. 2015). Furthermore, the perception of problems can fuel panic buying on the markets that in turn drives up prices.

Thus, the impact of an extreme weather event on markets has both a *trigger* component (the event) and a risk *perception* component (Challinor et al. 2016, 2018). Through commodity markets, prices change across the world because almost every country depends, to a greater or lesser extent, on trade to fulfil local needs. Commodity prices can also affect local market prices by altering input prices, changing the cost of food aid, and through spill-over effects; for example, in 2007–2008 the grain affected by extreme weather was wheat, but there was a significant price spike in rice markets (Dawe 2010).

As discussed by Bailey et al. (2015), there are a range of adaptation measures that can be put in place to reduce the impact of climate-related production shortfalls. These include (a) ensuring transparency of public and private stocks, as well as improved seasonal forecasting to signal forthcoming yield shortfalls (FAO 2016a; Ceglar et al. 2018; Iizumi et al. 2018), (b) building real or virtual stockholdings, (c) increasing local productivity and diversity (as a hedge against a reliance on trade) and (d) ensuring smoother market responses, through, for example, avoiding the imposition of export bans.

In summary, given the likelihood that extreme weather will increase, in both frequency and magnitude (Hansen et al. 2012; Coumou et al. 2014; Mann et al. 2017; Bailey et al. 2015), and the current state of global and cross-sectoral interconnectedness, the food system is at increasing risk of disruption (*medium evidence, medium agreement*), with large uncertainty about how this could manifest. There is therefore a need to build resilience into international trade as well as local supplies.

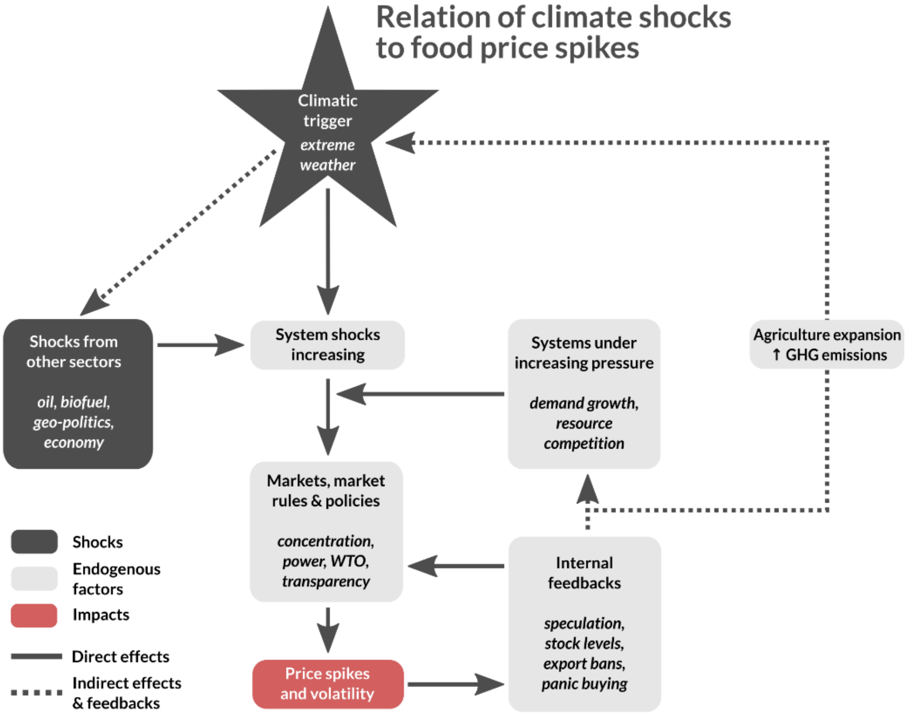


Figure 5.17 Underlying processes that affect the development of a food price spike in agricultural commodity markets (Challinor et al. 2018)

Box 5.5 Market drivers and the consequences of extreme weather in 2010-2011

The 2010–2011 food price spike was initially triggered by the exceptional heat in summer 2010, with an extent from Europe to the Ukraine and Western Russia (Barriopedro et al. 2011; Watanabe et al. 2013; Hoag 2014). The heatwave in Russia was extreme in both temperature (over 40°C) and duration (from July to mid-August in 2010). This reduced wheat yields by approximately one third (Wegren 2011; Marchand et al. 2016). Simultaneously, in the Indus Valley in Pakistan, unprecedented rainfall led to flooding, affecting the lives and livelihoods of 20 million people. There is evidence that these effects were both linked and made more likely through climate change (Mann et al. 2017).

In response to its shortfall in yields, Russia imposed an export ban in order to maintain local food supplies. Other countries responded in a largely uncoordinated ways, each of them driven by internal politics as well as national self-interests (Jones and Hiller 2017). Overall, these measures led to rapid price rises on the global markets (Welton 2011), partly through panic buying, but also through financial speculation (Spratt 2013).

Analysis of responses to higher food prices in the developing world showed that lower-income groups responded by taking on more employment, reducing food intake, limiting expenditures, spending savings (if available), and participating in demonstrations. People often identified their problems as stemming from collusion between powerful incumbent interests (e.g., of politicians and big business) and disregard for the poor (Hossain and Green 2011). This politicised social response helped spark food-related civil protest, including riots, across a range of countries in 2010–2011 (Natalini et al. 2017). In Pakistan, food price rises were exacerbated by the economic impacts of the floods, and which further contributed to food-related riots in 2010.

Price spikes also impact on food security in the developed world. In the UK, global commodity price inflation influenced local food prices, increasing food-price inflation by ~5 times at the end of 2010. Comparing household purchases over the five year period from 2007 to 2011 showed that the amount of food bought declined, on average, by 4.2%, whilst paying 12% more for it. The lowest income decile spent 17% more by 2011 than they did in 2007 (Holding et al. 2013; Tadasse et al. 2016). Consumers also saved money by trading down for cheaper alternatives. For the poorest, in the extreme situation, food became unaffordable: the Trussell Trust, a charity supplying emergency food handouts for people in crisis, noted a 50% increase in handouts in 2010.

5.8.2 Migration and conflict

Since the IPCC AR5 (Porter et al. 2014; Cramer et al. 2014), new work has advanced multi-factor methodological issues related to migration and conflict (e.g., Kelley et al. 2015, 2017; Werrell et al. 2015; Challinor et al. 2018; Pasini et al. 2018). These in particular have addressed systemic risks to food security that result from cascading impacts triggered by droughts and floods and how these are related to a broad range of societal influences.

Climate variability and extremes have short-, medium- and long-term impacts on livelihoods and livelihood assets – especially of the poor – contributing to greater risk of food insecurity and malnutrition (FAO et al. 2018). Drought threatens local food security and nutrition and aggravates humanitarian conditions, which can trigger large-scale human displacement and create a breeding ground for conflict (Maystadt and Ecker 2014). There is *medium agreement* that existing patterns of conflict could be reinforced under climate change affecting food security and livelihood opportunities, for example, already fragile regions with ethnic divides such as North and Central Africa as well as Central Asia (Buhaug 2016; Schleussner et al. 2016) (Box 5.6).

Challinor et al. (2018) have developed a typology for transboundary and transboundary risk transmission that distinguishes the roles of climate and social and economic systems. To understand these complex interactions, they recommend a combination of methods that include expert judgement; interactive scenario building; global systems science and big data; and innovative use of climate and integrated assessment models; and social science techniques (e.g., surveys, interviews, and focus groups).

5.8.2.1 Migration

There has been a surge in international migration in recent years, with around five million people migrating permanently in 2016 (OECD 2017). Though the initial driver of migration may differ across populations, countries and contexts, migrants tend to seek the same fundamental objective: to provide security and adequate living conditions for their families and themselves. Food insecurity is a critical ‘push’ factor driving international migration, along with conflict, income inequality, and population growth. The act of migration itself causes food insecurity, given the lack of income opportunities and adverse conditions compounded by conflict situations.

Warner et al. (2012) found the interrelationships between changing rainfall patterns, food and livelihood security in eight countries in Asia, Africa and Latin America. Several studies in Africa have found that persistent droughts and land degradation contributed to both seasonal and permanent migration (Gray 2011; Gray and Mueller 2012; Hummel 2015; Henry et al. 2004; Folami and Folami 2013), worsening contextual vulnerability conditions of different households (Dasgupta et al. 2014).

Dependency on rainfed agriculture is from 13% in Mexico to more than 30% in Guatemala, Honduras, and Nicaragua, suggesting a high degree of sensitivity to climate variability and change, and undermined food security (Warner et al. 2009). Studies have demonstrated that Mexican migration (Feng et al. 2010; Nawrotzki et al. 2013) and Central American migration (WFP 2017) fluctuate in response to climate variability. The food system is heavily dependent on maize and bean production and long-term climate change and variability significantly affect the productivity of these crops and the livelihoods of smallholder farmers (WFP 2017). In rural Ecuador, adverse environmental conditions prompt out-migration, although households respond to these challenges in diverse ways resulting in complex migratory responses (Gray and Bilsborrow 2013).

Migration patterns have been linked to heat stress in Pakistan (Mueller et al. 2014) and climate variability in the Sundarbans due to decline in food security (Guha and Roy 2016). In Bangladesh, the impacts of climate change have been on the rise throughout the last three decades with increasing migration, mostly of men leaving women and children to cope with increasing effects of natural disasters (Rabbani et al. 2015).

Box 5.6 Migration in the Pacific region: Impacts of climate change on food security

Climate change-induced displacement and migration in the Pacific has received wide attention in the scientific discourse (Fröhlich and Klepp 2019). The processes of climate change and their effects in the region have serious implications for Pacific Island nations as they influence the environments that are their ‘life-support systems’ (Campbell 2014). Climate variability poses significant threats to both agricultural production and food security. Rising temperatures and reductions in groundwater availability, as well as increasing frequency and severity of disaster events translate into substantial impacts on food security causing human displacement, a trend that will be aggravated by future climate impacts (ADB 2017). Declining soil productivity, groundwater depletion, and non-availability of freshwater threatens agricultural production in many remote atolls.

Many countries in the Pacific devote a large share of available land area to agricultural production. For example, more than 60% of land area is cultivated in the Marshall Islands and Tuvalu and more than 40% in Kiribati and Tonga. With few options to expand agricultural area, the projected impacts of climate change on food production are of particular concern (ADB 2013, 2017). The degradation of available land area for traditional agriculture, adverse disruptions of agricultural productivity and diminishing livelihood opportunities through climate change impacts leads to increasing poverty and food insecurity, incentivising migration to urban agglomerations (ADB 2017; FAO et al. 2018).

Campbell (2014) describe the trends that lead to migration. First, climate change, including sea level rise, affect communities’ land security, which is the physical presence on which to live and sustain livelihoods. Second, they impinge on livelihood security (especially food security) of island communities where the productivity of both subsistence and commercial food production systems is reduced. Third, the effects of climate change are especially severe on small-island environments since they result in declining ecological habitat. The effects on island systems are mostly manifested in atolls through erosion and inundation, and on human populations through migration. Population growth and scenarios of climate change is *likely* to further induce food stress as impacts unfolds in coming decades (Campbell 2015).

While the populations of several islands and island groups in the Pacific (e.g., Tuvalu, Carteret Islands, and Kiribati) have been perceived as the first probable victims of rising seas so that their inhabitants would become, and in some quarters already are seen to be, the first 'environmental' or 'climate change refugees,' migration patterns vary. Especially in small islands, the range and nature of the interactions among economic, social, and/or political drivers are complex. For example, in the Maldives, Stojanov et al. (2017) show that while collective perceptions support climate change impacts as being one of the key factors prompting migration, individual perceptions give more credence to other cultural, religious, economic or social factors.

In the Pacific, Tuvalu has long been a prime candidate to disappear due to rising sea level, forcing human migration. However, results of a recent study (Kench et al. 2018) challenge perceptions of island loss in Tuvalu, reporting despite sea level rise, a net increase in land area of 73.5 ha. The findings suggest that islands are dynamic features likely to persist as habitation sites over the next century, presenting opportunities for adaptation that embrace the heterogeneity of island types and processes. Farbotko (2010) and Farbotko and Lazrus (2012) present Tuvalu as a site of 'wishful sinking,' in the climate change discourse. These authors argue that representations of Tuvalu as a laboratory for global climate change migration are visualisations by non-locals.

In Nanumea (Tuvalu), forced displacements and voluntary migrations are complex decisions made by individuals, families and communities in response to discourses on risk, deteriorating infrastructure and other economic and social pressures (Marino and Lazrus 2015). In many atoll nations in western Pacific, migration has increasingly become a sustainable livelihood strategy, irrespective of climate change (Connell 2015).

In Lamen Bay, Vanuatu, migration is both a cause and consequence of local vulnerabilities. While migration provides an opportunity for households to meet their immediate economic needs, it limits the ability of the community to foster longer-term economic development. At the same time, migration adversely affects the ability of the community to maintain food security due to lost labour and changing attitudes towards traditional ways of life among community members (Craven 2015).

Small islands are very sensitive to climate change impacts (*high confidence*) (Nurse et al. 2014) and impacted by multiple climatic stressors (see (IPCC 2018a) and SROCC). Food security in the Pacific, especially in Micronesia, has worsened in the past half century and climate change is *likely* to further hamper local food production, especially in low-lying atolls (Connell 2016) Migration in small islands (internally and internationally) occurs for multiple reasons and purposes, mostly for better livelihood opportunities (*high confidence*).

Beyond sea level rise, effects of increasing frequency and intensity of extreme events such as severe tropical cyclones are *likely* to affect human migration in the Pacific (Connell 2015; Krishnapillai and Gavenda 2014; Charan et al. 2017; Krishnapillai 2017). On Yap Island, extreme weather events are affecting every aspect of atoll communities' existence, mainly due to islands' small size, their low elevation, and extensive coastal areas (Krishnapillai 2018). Displaced atoll communities on Yap Island grow a variety of nutritious vegetables and use alternative crop production methods such as small-plot intensive farming, raised bed gardening, as part of a community-based adaptation program (Krishnapillai and Gavenda 2014; Krishnapillai 2018).

Recurrences of natural disasters and crises threaten food security through impacts on traditional agriculture, causing the forced migration and displacement of coastal communities to highlands in search of better living conditions. Although considerable differences occur in the physical manifestations of severe storms, such climate stressors threaten the life-support systems of many atoll communities (Campbell et al. 2014). Failure of these systems resulting from climate disasters propel vulnerable atoll communities into poverty traps, and low adaptive capacity could eventually force these communities to migrate.

5.8.2.2 Conflict

While climate change will not alone cause conflict, it is often acknowledged as having the potential to exacerbate or catalyse conflict in conjunction with other factors. Increased resource competition can aggravate the potential for migration to lead to conflict. When populations continue to increase, competition for resources will also increase, and resources will become even scarcer due to climate change (Hendrix and Glaser 2007). In agriculture-dependent communities in low-income contexts, droughts have been found to increase the likelihood of violence and prolonged conflict at the local level, which eventually pose a threat to societal stability and peace (FAO et al. 2017). In contrast, conflicts can also have diverging effects on agriculture due to land abandonment, resulting in forest growth, or agriculture expansion causing deforestation, e.g., in Colombia (Landholm et al. 2019).

Several studies have explored the causal links among climate change, drought, impacts on agricultural production, livelihoods, and civil unrest in Syria from 2007-2010, but without agreement as to the role played by climate in subsequent migration (Kelley et al. 2015, 2017; Challinor et al. 2018; Selby et al. 2017; Hendrix 2018). Contributing factors that have been examined include rainfall deficits, population growth, agricultural policies, and influx of refugees that had placed burdens on the region's water resources (Kelley et al. 2015). Drought may have played a role as a trigger, as this drought was the longest and the most intense in the last 900 years (Cook et al. 2016; Mathbout et al. 2018). Some studies linked the drought to widespread crop failure, but the climate hypothesis has been contested (Selby et al. 2017; Hendrix 2018). Recent evidence shows that the severe drought triggered agricultural collapse and displacement of rural farm families with approximately 300,000 families going to Damascus, Aleppo and other cities (Kelley et al. 2017).

Persistent drought in Morocco during the early 1980s resulted in food riots and contributed to an economic collapse (El-Said and Harrigan 2014). A drought in Somalia that fuelled conflict through livestock price changes, establishing livestock markets as the primary channel of impact (Maystadt and Ecker 2014). Cattle raiding as a normal means of restocking during drought in the Great Horn of Africa led to conflict (ICPAC and WFP 2017) whereas a region-wide drought in northern Mali in 2012 wiped out thousands of livestock and devastated the livelihoods of pastoralists, in turn swelling the ranks of armed rebel factions and forcing others to steal and loot for survival (Breisinger et al. 2015).

On the other hand, inter-annual adjustments in international trade can play an important role in shifting supplies from food surplus regions to regions facing food deficits which emerge as a consequence of extreme weather events, civil strife, and/or other disruptions (Baldos and Hertel 2015). A more freely functioning global trading system is tested for its ability to deliver improved long run food security in 2050.

In summary, given increasing extreme events and global and cross-sectoral interconnectedness, the food system is at increasing risk of disruption, e.g., via migration and conflict (*high confidence*). {5.2.3, 5.2.4}

Frequently Asked Questions

FAQ 5.1 How does climate change affect food security?

Climate change negatively affects all four pillars of food security: availability, access, utilisation and stability. Food availability may be reduced by negative climate change impacts on productivity of crops, livestock and fish, for instance due to increases in temperature and changes in rainfall patterns. Productivity is also negatively affected by increased pests and diseases, as well as changing distributions of pollinators under climate change. Food access and its stability may be affected through disruption of markets, prices, infrastructure, transport, manufacture, and retail, as well as direct and indirect changes in income and food purchasing power of low-income consumers. Food utilisation may be directly affected by climate change due to increases in mycotoxins in food and feed with rising temperatures and increased frequencies of extreme events, and indirectly through effects on health. Elevated atmospheric CO_2 concentrations can increase yields at lower temperature increases, but tend to decrease protein content in many crops, reducing their nutritional values. Extreme events, e.g., flooding, will affect the stability of food supply directly through disruption of transport and markets.

FAQ 5.2 How can changing diets help address climate change?

Agricultural activities emit substantial amounts of greenhouse gases (GHGs). Food supply chains activities past the farm gate (e.g., transportation, storage, packaging) also emit GHGs, for instance due to consumption of energy. GHG emissions from food production vary across food types. Producing animal-sourced food (i.e., meat and dairy) emits larger amount of GHGs than growing crops, especially in intensive, industrial livestock systems. This is mainly true for commodities produced by ruminant livestock such as cattle, due to enteric fermentation processes that are large emitters of methane. Changing diets towards a lower share of animal-sourced food, once implemented at scale, reduces the need to raise livestock and changes crop production from animal feed to human food. This reduces the need for agricultural land compared to present and thus generates changes in the current food system. From field to consumer this would reduce overall GHG emissions. Changes in consumer behaviour beyond dietary changes can also have, at scale, effects on overall GHG emissions from food systems. consuming regional and seasonal food can reduce GHG emissions, if they are grown efficiently.

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Supplementary Material

Section SM5.1

Table SM5.1 A gendered approach to understanding how climate change affects dimensions of food security across pastoral and agro-pastoral livestock-holders (adapted from McKune et al. (2015); Ongoro and Ogara (2012) and Fratkin et al. (2004). ↑ increased, ↓ decreased

Group	Livelihoods	Health	Nutrition
Pastoral	↑ time demand on *women* and *girls* for water, fuel collection ↑ time demand on *men* to seek out water sources with herd ↑ *men* exposure to attacks from other groups ↑ *men* migration resulting in ↑ *women* workload ↑ productive and reproductive demands on *women* ↓ financial autonomy of *women* due to liquidation of small animal assets ↑ women poverty due to livestock losses of men	↑ disease risk due to proximity of *women's* work to disease agents ↑ *children* health and growth due to reduced milk consumption ↑ *women* and *girls* exposure to insecurity and dangers when looking for water ↑ *women* and *children* vulnerability to water-borne diseases ↑ vulnerability to *maternal mortality* due to ↑ fertility due to sedentarisation ↓ mental and emotional health due to increased stress/loss of social support for both *men and women* ↑ vulnerability of newly sedentarized households, particularly *women*	↑ undernutrition of *men and women* due to ↓ availability of plant and animal foods ↑ undernutrition of *men and women* due to separation of from milk-producing animals ↑ undernutrition in *men and women* due to unfavorable trade-offs in diet between animal products and grains ↑ risk of food insecurity I *men and women* due to ↓ production of livestock and ↑ prices
Agro-pastoral	↑ time demand on *women* due to migration of men for herding or wage labor ↓ financial autonomy of *women* due to liquidation of small animal assets ↑ constraints on herd management due to shifts in responsibilities ↑ susceptibility to market	Earlier weaning, shortened birth intervals, and risk of *maternal depletion* ↑ incidence of anemia and stunting in *children* ↑ susceptibility to infectious diseases that are sensitive to climate change in both *men and women* ↑ *child* mortality rates	↑ exposure of *men and women* to foods that have become spoiled Less varied and less nutritious diets for *men and women* ↑ malnutrition, including overnutrition, in *men and women*

	fluctuations		

1 **Section SM5.2**

2 Table SM5.2 Impacts of selected climate drivers on food security pillars.

Food security pillar	Driver of climate change	Process	Impact	Reference
Availability	Increase in temperature	• Increased water demand • Increased heat and drought stress • Shorter growing period • More frequent heat wave • Terminal heat • Reduced grain filling period • Decreased soil fertility • Land degradation • Higher pre-harvest loss due to disease and pest attack • Negative effects on physiological processes	Decreased crop yield and animal performance	Zhao et al. (2017) Asseng et al. (2015) Myers et al. (2017) Ovalle-Rivera et al. (2015) Rosenzweig et al. (2014) Medina et al. (2017) Paterson and Lima (2011) Schlenker and Roberts (2009)
	CO_2 concentration	• Increased photosynthesis in C3 crops • Increased water use efficiency	Increased crop yield	Franzaring et al. (2013) Mishra and Agrawal (2014) Myers et al. (2014) Ishigooka et al. (2017) Zhu et al. (2018) Loladze (2014) Yu et al. (2014)
	Precipitation (untimely, erratic, decreased)	• Drought and heat stress • Crop failure • Land degradation • Reduced soil fertility	Decreased crop yield and pasture stocking rates and animal performance	Leng and Hall (2019) Zscheischler et al. (2018) Meng et al. (2016) Zimmerman et al. (2017) FAO et al. (2018)

	Extreme events (drought, flood, cyclones etc.)	• Decrease in organic matter • Soil erosion • Crop failure • Disruption of distribution and exchange	Decreased crop yield Increased livestock mortality Decreased distribution and exchange	Leng and Hall (2019) Rivera-Ferre (2014)
Access	Increase in Temperature	• Increase in price • Loss of agricultural income • Disproportionate impact on low-income consumers	Increased food price and reduced purchasing power	Morris et al. (2017) Vermeulen et al. (2012) Abid et al. (2016) Harvey et al. (2014) UNCCD (2017)
	Precipitation (untimely, erratic, decreased)	• Low yield, price increase • Loss of agricultural income due to reduced yield and productivity • Decrease in barley yield • Inability to invest in adaptation and diversification measures to endure price rises	Increased food price and reduced purchasing power	FAO (2016) Kelley et al. (2015) Morris et al. (2017) Vermeulen et al. (2012) Abid et al. (2016) Harvey et al. (2014) UNCCD (2017)
	Extreme Events (drought, flood, cyclones etc.)	• Price increase due to low yield or sporadic crop failure • Loss of agricultural income	Increased food price and reduced purchasing power	Valin et al. (2014) Robinson et al. (2014) Nelson et al. (2013) Schmitz et al. (2014)
Utilization	Increase in Temperature	• Decreased in nutritional content • Increased mycotoxins • Reduced water quantity and	Reduced quality	Tirado and Meerman (2012) Aberman and Tirado (2014) Thompson et al. (2012)

		quality to prepare food • Negative impact on food safety • Higher post-harvest loss both in quantity and quality		
	CO_2 Concentration	• Decreased protein content • Less zinc content • Less iron content • Increased biomass but reduced multiple nutrients • Less radiation interception and less biomass production	Reduced quality	Myers et al. (2014) Smith et al. (2017) Myers et al. (2015) Medek et al. (2017) Bahrami et al. (2017) Rosenzweig and Hillel (2015)
	Extreme Events (drought, flood, cyclones etc.)	• Adverse weather affects food storage and distribution	Reduced quality	Wellesley et al. (2017) Thompson et al. (2012)
Stability	Increase in Temperature	• Disruption of food supply	Fluctuation in production, supply and price	Allen et al. (2017) Tigchelaar et al. (2018)
	Precipitation (untimely, erratic, decreased)	• Disruption of food supply • Yield variability • Fluctuation in yield, supply and price • Crop failure due to extreme drought	Fluctuation in production, supply and price	Schmidhuber and Tubiello (2007) Kelley et al. (2015) Selby et al. (2017) Kelley et al. (2017) Medina-Elizalde and Rohling (2012)

	Extreme Events (drought, flood, cyclones etc.)	• Impacts on world market export prices that carry through to domestic consumer prices • Widespread crop failure contributing to migration and conflict • Disruption of food supply due to civil disturbance and social tension	Fluctuation in production, supply and price	Kelley et al. (2015) Willenbockel (2012) Hendrix (2018) Selby et al. (2017) Kelley et al. (2017)

Detection and attribution methods

Observed impacts of climate change on food security have been noted as a cause of concern (HLPE 2012) and assessed in AR5 (Porter et al. 2014; Cramer et al. 2014) and SR15 (IPCC 2018). Assessing evidence for detection and attribution of observed climate change impacts on the food system remains a challenge because agriculture is a managed system with practices changing over time. Using AR5 and SR15 findings that observed climate changes attributable to human influence include rising temperatures, increases in the intensity and frequency of hot days and nights, more areas with increases than decreases in the frequency, intensity, and or amount of heavy precipitation, and drying trends in some regions especially in the Mediterranean region (including southern Europe, northern Africa and the Near East), we assess recent studies of observed climate change impacts on the food system that utilise IPCC attribution methods (Hegerl et al. 2010), as well as others that depend on local knowledge from the developing world.

New work has addressed observed climate effects on expanded aspects of the food system, including pastoral systems (Rasul et al. 2019; Abiona et al. 2016), pests, diseases, and pollinators (Bebber et al. 2014; Schweiger et al. 2010), and adaptation (Li et al. 2017) (see Section 5.3). Surveys of farmer perceptions of climate changes and their impacts are being increasingly utilised in developing countries for example (Hussain et al. 2016) (Ifeanyi-obi et al. 2016; Onyeneke 2018).

Improvements in projection methods since AR5

Since AR5, methods for assessment of future climate change impacts on food systems have improved in several areas, providing new insights. These methods include greater number of ensembles of multiple climate, crop, and economic models, with improved characterisation of uncertainty (Wiebe et al. 2015); further comparison of results from process-based crop models and statistical models (Zhao et al. 2017); advances in regional integrated assessments (Rosenzweig and Hillel 2015), and new coordinated global and regional studies (Rosenzweig et al. 2017; Ruane et al. 2018). Temperature response functions in crop models have been improved (Wang et al. 2017).

Expanded meta-analyses of free-air carbon dioxide experiments (FACE) have examined effects of high CO_2 on crop nutrients not just on yield (Smith and Myers 2018; Zhu et al. 2018) (Section 5.2.4.2). Recent reviews have confirmed that higher CO_2 concentrations increase crop growth and yield, especially in crops with C3 photosynthetic pathways, but realisation of these direct CO_2 effects depends on nutrient and water availability (Lombardozzi et al. 2018; Toreti et al.; Uddin et al. 2018) *(high confidence)*. New work has considered future impacts of farming systems, extreme events, fruits and vegetables, rangelands and livestock, and aquaculture, as well as food safety, pests and diseases, and food quality (Section 5.2).

However, several sources of uncertainty exist in projection of climate change crop impacts, partly stemming from differences between the models and methods utilised, sparse observations related to current climate trends, and other agro-ecosystem responses (e.g., to CO_2 effects) (Mistry et al. 2017; Li et al. 2015; Bassu et al. 2014; Asseng et al. 2013). The uncertainty in climate simulations is generally larger than, or sometimes comparable to, the uncertainty in crop simulations using a single model (Iizumi et al. 2011), but is less than crop model uncertainty when multiple crop models are used as in AgMIP (Rosenzweig et al. 2014b) and CO_2 is considered (Hasegawa et al. 2018; Müller et al. 2014; Asseng et al. 2013).

Most of the work on projected impacts on climate change impacts on crops continues to focus on the major commodities-wheat, maize, rice, and soybean-while areas still lagging are multi-model ensemble approaches for livestock and fruits and vegetables. While the current reliance on the four major commodities makes assessment of climate change impacts on them important, there is a growing recognition that more than caloric intake is required to achieve food security for all and that assessments need to take into account how climate change will affect the 2 billion malnourished people in the current climate and food system.

Table SM5.3 Observed climate change impacts on crop production, data sources, and detection and attribution methods

Climate observations	Climate data source	Observed impacts	Impact method/source	Time period	Region	Detection &Attribution method	Reference	Continent
Warming temperatures	Chinese Meteorological Administration	If 1980 variety was still grown, maize yield would stagnate or decrease; due to adoption of maize varieties with long growth period yield increased by 7-17% per decade.	China Agricultural Database	1980-2009	Heilongjiang Province, Northeast China	Single step attribution	(Meng et al. 2014)	Asia
Warming temperatures	Chinese Meteorological Administration	Changes in winter wheat phenology; observed dates of sowing, emergence, and beginning of winter dormancy were delayed by 1.2, 1.3, and 1.2 days per decade. Dates of regrowth after dormancy, anthesis, and maturity advanced 2.0, 3.7, and 3.1 days per decade. Growth duration, overwintering period, and vegetation phase shortened by 4.3, 3.1, and 5.0 days per decade.	Local agro-meterological experimental stations maintained by Chinese Meterological Administration	1981-2009	Loess Plateau, Northwest China	Single step attribution	(He 2015)	Asia

Warming temperatures	Central China Meteorological Agency	Advance in sowing and phenological stages advanced by 23-26 days	Agrometeorological experimental station Wulanwusu, China	1981-2010	Northwest China	Statistical relationships for cotton phenologies, seed cotton yields, and climate parameters using Pearson correlation analysis.	(Huang and Ji 2015)	Asia
Warming temperatures	China Meteorological Administration	Changes in temperature, precipitation and solar radiation in past three decades and increased wheat yield in northern China by 0.9-12.9%; reduced wheat yield in southern China by 1.2-10.2 %.	China Meteorological Administration	1981-2009	China	Correlations between annual yields with climate variables. Partial correlations with detrended yields and climate variables.	(Tao et al. 2014)	Asia
Warming temperatures	Pakistan Meteorological Department	Change in phenology of sunflowers. Sowing dates for spring sunflowers 3.4-9.3 days per decade earlier. Sowing dates for autumn sunflower delayed by 2.7-8.4 days per decade.	Punjab Agriculture Department	1980-2016	Punjab, Pakistan	Single step attribution	(Tariq et al. 2018)	Asia

Warming temperatures	Pakistan Meteorological Department	Change in phenology in maize. Sowing dates for spring maize 3.5-5.5 days per decade earlier. Sowing dates for autumn maize 1.5-4.2 days per decade later.	Punjab Agriculture Department	1980-2014	Pujab, Pakistan	Single step attribution	(Abbas et al. 2017)	Asia
Increases in max and min temperatures	India Meteorological Department (IMD)	Reduced wheat yields by 5.2% . 1 degree C increase in maximum temperature lowers yields by 2.3% while same increase in minimum temperature lowers yields by 3.6%.	Indian Harvest Database Centre of Monitoring the Indian Economy (CMIE) and Directorate of Economics, Ministry of Agriculture.	1981-2009	India	Regression analysis between temperature and yield.	(Gupta et al. 2017)	Asia
Reduced rainfall and rising temperatures	Australian Bureau of Meteorology	Stagnated wheat yields. Declines in water-limited yield potential.	Agricultural Commodity Statistics	1965-2015	Australia	Single step attribution	(Hochman et al. 2017)	Australia

Increases in temperature and drought	Czech Hydrometerological Institute (CHMI), 268 climatological stations, and 774 rain gauge stations	Long-term impacts on fruiting vegetables (+4.9 to 12.2% per degree C) but decreases in stability of tradionally grown root vegetables in warmest areas of country.	Database of 12 field-grown vegetables at district level as reported by Czech Statistical Office.	1961-2014	Czech Republic	Associative pattern attribution	(Potopová et al. 2017)	Europe
Long-term temperature and precipitation trends	Precipitation: 1900-2008 Gridded Monthly Time Series Version 2.01. Available at: http://climate.geog.udel.edu/~climate/.	Wheat and barley yields declined by 2.5% and 3.8%, and maize and sugar beet yields have increased due to temperature and precipitation changes.	EU Farm Accountancy Data Network (FADN)	1989-2009	Europe	Associative pattern attribution	(Moore and Lobell 2015)	Europe

Notes: See Hegerl et al. (2010) for full definitions of attribution methods: Single Step: where a model(s) is run with and without a single variable of interest (i.e., temperature) and results compared to observed changes within a system; Multi-Step: Through processes modelling and/or a statistical link, a change in climate is linked to a variable of interest, and then that variable of interest is linked to an observed change; Associative Pattern: Involves the synthesis of multiple observations – and demonstrates a pattern of strong association between these changes and changes in temperatures due to anthropogenic forcing.

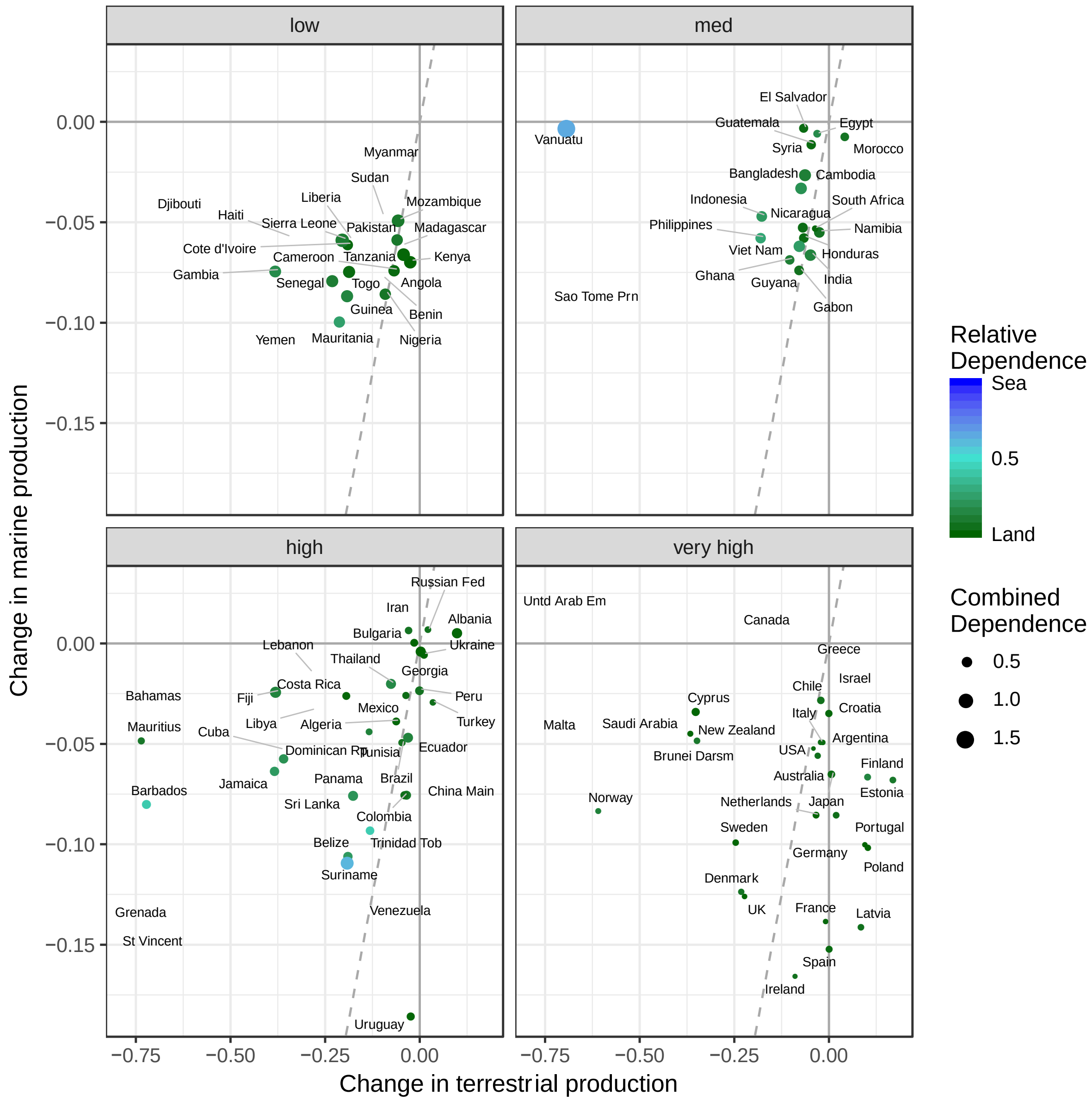

Figure SM5.1 Climate change impacts and adaptive capacity by continent across land and sea. Vulnerability of societies to climate change impacts in fisheries and agriculture under RCP6.0. Changes in marine fisheries (Tittensor 2017) and terrestrial crop production (Rosenzweig et al. 2014b) **are expressed as $\log_{10}$(projected/baseline) production, where a value below zero indicates decreases and above are increases. Fisheries and agriculture dependency estimates calculated from employment, economy and food security. Circle size represents total dependency on both sectors and green to blue colour scale reflects the balance between land and sea with white indicative of equal dependence. The dependence indices were calculated using publicly available online data from FAO, the World Bank and a recent compilations of fisheries employment data (Teh and Sumaila 2013). Each panel a-d) represents the four Human Development Index (HDI) categories (low, medium, high and very high) and open diamonds indicate no data for agricultural and fisheries dependency. Modified from: Blanchard et al. 2017.**

Table SM5.4 Models included in Hasegawa et al. (2018)

Model	Reference
AIM/CGE	(Fujimori et al. 2012)
CAPRI	(Britz and Witzke 2014)
GCAM	(Kyle et al. 2011; Wise and Calvin 2011)
GLOBIOM	(Havlik et al. 2014)
IMAGE 3.0	(Stehfest et al. 2014)
IMPACT 3	(Robinson et al. 2015)
MAGNET	(Woltjer et al. 2014)
MAgPIE	(Lotze-Campen et al. 2008; Popp et al. 2014)

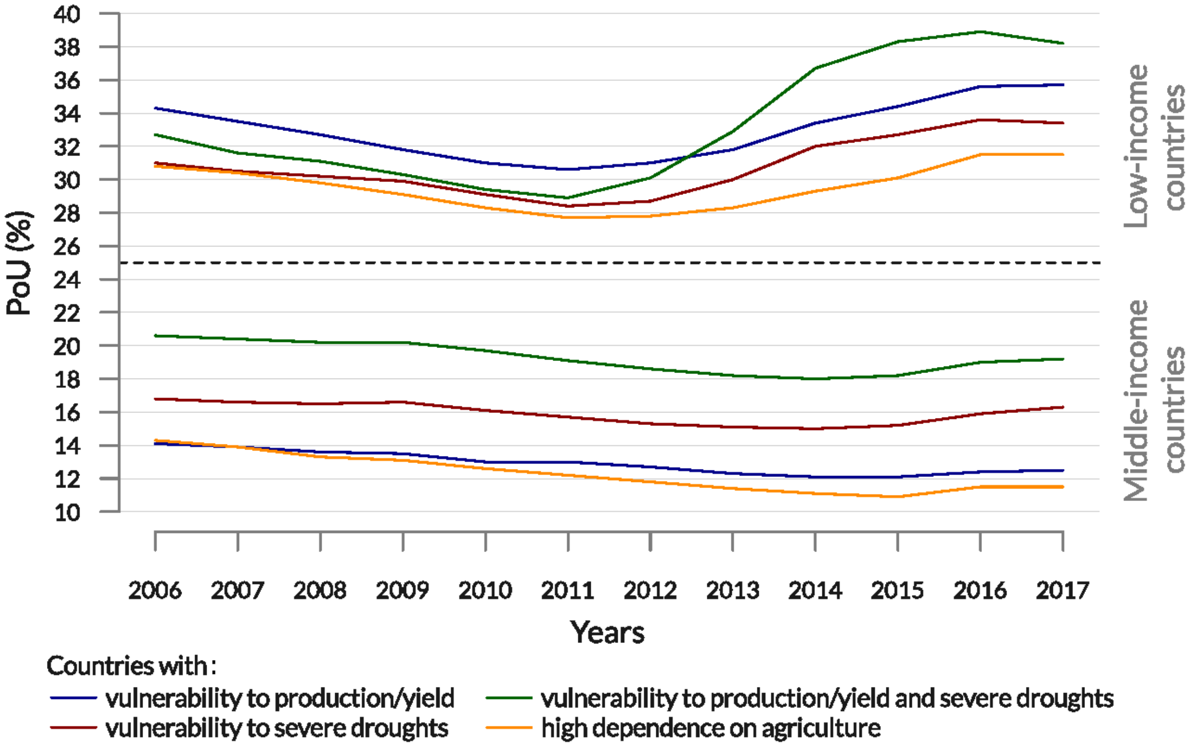

Figure SM5.2 Undernourishment is higher when exposure to climate extremes is compounded by high levels of vulnerability in agriculture (FAO et al. 2018).

Section SM5.5

Livestock mitigation strategies

Intensification of animal diets. It is well established that appropriate diet regimes may contribute to reduce the amount of GHG produced per unit of animal product (Gerber et al. 2013b), which, within the appropriate implementation including governance, may lead to mitigation of absolute emissions. This increased efficiency can be achieved through improved supplementation practices or through land use management with practices like improved pasture management, including grazing rotation, fertiliser applications, soil pH modification, development of fodder banks, improved pasture species, use of legumes and other high protein feeds, the use of improved crop by-products and novel feeds (i.e., black soldier fly meal, industrially produced microbial protein (Pikaar et al. 2018).

When done through increased feeding of grains, transition to improved diets shifts the contributions of different GHG gases to the total emissions. This is due to the fact that the proportion of methane to total emissions is reduced (due to lower roughage intake), while the proportion of emissions associated with feed manufacture (energy and land use change) increases. Therefore, CO_2 emissions from land use change increase while methane emissions per unit of output decrease (Gill et al. 2010). As a consequence, the quantified benefits of a given strategy wil also depend on the assumed GWP of methane.

Of the available livestock GHG mitigation options, improved feeding systems are relatively easy to implement at the farm level. A prerequisite for these options to work is that the livestock systems need to be geared towards market-oriented production, as otherwise there is little incentive to improve feeding systems. This in turn implies that costs and benefits to farmers are appropriate to incentivise specific management changes and also assess the impact that market-orientation may have in some societies, such as pastoralists (López-i-Gelats et al. 2016). Examples of where this option could be applicable are smallholder dairy-crop mixed systems in Africa and Asia, dual-purpose and dairy production in Latin America and beef cattle operations, where significant mitigation opportunities exist. Other mitigation options include manipulation of rumen microflora, breeding for lower methane production, and the use of feed additives (Hristov et al. 2013).

The largest GHG efficiency gaps are observed in livestock systems where the quality of the diet is the poorest (i.e., grassland-based and some arid and humid mixed systems in the developing world). The highest marginal gains of improving animal diets through simple feeding practices, both biologically and economically, are in these systems (FAO, 2013; Herrero et al. 2013).

Control of animal numbers, shifts in breeds, and improved management. Increases in animal numbers are one of the biggest factors contributing directly to GHG emissions (Tubiello, 2019). Regions with intensive animal production, such as concentrated animal feeding operations (CAFOs), can control animal numbers, conduct breeding programs for efficient animals, and improve feeding management. In the developing world, many low-producing animals could be replaced by fewer but better-fed cross-bred animals of a higher potential, with improved grazing management (i.e., attention to feed, herbage availability, and allowances) playing an important role. In both developed and developing countries these practices are able to reduce total emissions while maintaining or increasing the supply of livestock products.

However, attention must be paid to synergies and trade-offs between livelihoods and specific mitigation strategies, such as controlling animal numbers, recognising the multiple objectives that livestock raising may contribute to within specific settings, especially in low-input systems. Improvements in animal health can also significantly reduce emissions intensity by improved yields and fertility per animal and reductions in mortality (ADAS 2015).

Changes in livestock species. Switching species to better suit particular environments is a strategy that could yield higher productivity per animal for the resources available. At the same time, structural changes in the livestock sector from beef to sheeps and goats, or mainly from ruminants to monogastrics (e.g., from beef to pig or poultry production) could lead to reduced methane emissions and higher efficiency gains. Assessment done using integrated assessment models (IAMs) have shown that these practices could lead to reductions in land use change and its associated emissions (Havlik et al. 2014; Frank et al. 2018).

Managing nitrous oxide emissions from manure. In the developing world, large amounts of nutrients are lost due to poor manure management. In currently adopted feeding systems, large amounts of nutrients and carbon are lost in connection with manure storage (e.g., Herrero et al. 2013). In many places pig manure is not recycled; considered a waste, it is often discharged to water bodies or left to accumulate unused. Yet these farming systems can be highly N and P limited. This practice creates serious problems especially in urban and peri-urban systems by contributing to water and air pollution. Research in intensive African ruminant livestock systems, for instance, has shown that up to 70% of the manure N can be lost within six months of excretion when manure is poorly managed (Tittonell et al. 2009).

Options to manage emissions in the livestock sector are not easy to design because they require systems thinking and awareness of key driving factors in different livestock systems. Reducing N emissions starts with feeding livestock balanced diets so that excreta are not rich in labile N, which is easily lost as ammonia and enters the N cascade (Bouwman et al. 2013). In intensive systems, mineral N can be captured effectively using bedding material, which has been increasingly excluded from livestock facilities to reduce operational costs.

Manure is increasingly handled as slurry in tanks or anaerobic lagoons, which may reduce direct nitrous oxide emissions during storage but can increase methane and ammonia loss and also increase the risk of emissions during land spreading (Velthof and Mosquera 2011). However, optimising land spreading of manures (in terms of timing or placement) to maximise N and P replacement value can minimise ammonia losses while also displacing mineral fertiliser (Bourdin et al. 2014).

In intensive systems, emissions of ammonia and nitrous oxide can be managed by spatially shifting livestock pens or the facilities where they overnight. Other options in more-intensive grazing systems may include nitrification inhibitors, stand-off pads, delayed manure spreading collected in milking sheds, although the fate of the full applied N and its partitioning between direct and indirect emissions as a result of the specific option chosen must be evaluated (e.g., Lam et al., 2017)

Uncertainties in demand-side technical mitigation potential

There are several unresolved issues regarding modelling and quantification of marginal emissions identified in the literature. Diet shift studies often focus on beef production emission intensities, although the cattle industry in many locations includes both meat and dairy production; these activities may be integrated in different types of farming systems (Flysjö et al. 2012) with significantly lower emission intensities (Gerber et al. 2013a; Flysjö et al. 2012). Links between ruminant meat production, the dairy sector (primarily cows and goats), and wool production in sheep are often overlooked in diet shift studies. FAOStat 2017 data indicate there are 278 million dairy cows worldwide, which make significant contributions to meat production (304 million head slaughtered per year) by providing calves (lactating cows must calve to produce milk) and dairy cows (replacements by younger females).

Attributional LCA values are often applied to diet shifts studies, overlooking the feedback loop (rebound effect) of demand on production system emission intensities. There are a few examples of consequential analysis of diet shifts (Tukker et al. 2011) (de Oliveira Silva et al. 2016) (Zech and Schneider 2019), reporting modest potential for mitigation (i.e., from 0-8%) but each of them emphasise only one particular aspect of diet shifts. Further, the application of those models to different regions of the world may require further development.

Current attributional LCA studies present inconsistencies related to the definition of system boundaries, allocation of co-products (including dairy), method of attribution of land use change, and pasture productivity effects on soil carbon stocks (Lynch 2019) (Yan et al. 2011; Dudley et al. 2014). Major differences in the results are due to how land use change affects emissions and soil carbon stocks,, particularly when addressing developing countries where deforestation and intensification can both take place at the same time. Deforestation-related emissions have been attributed to first land use (Bustamante et al. 2012), the activities under a given amortization time (Persson et al. 2014), change in total land covered by the activity (Gerber et al. 2013a), or the missed potential carbon sink, i.e., the opportunity for natural vegetation recovery (Schmidinger and Stehfest 2012) (Schmidt et al. 2015).

Also, variation in soil carbon stocks is not considered in most studies, while a few account for variations up to 0.3 m soil depth, and very rarely consider 1.0 m soil depth for estimating soil carbon variation. Overlooking soil carbon at deeper soil layers largely contributes to underestimating the environmental benefits of transition to more productive systems. Time considerations in soil carbon stocks dynamics also vary among studies, with some applying a standard 20-year equilibrium time instantaneously and others using dynamic (discrete or continuous) models.

The type of food replacement is another major source of uncertainty in calculating the impact of dietary changes (Smetana et al. 2015). Nutritional replacement with animal-based protein candidates such as chicken, eggs, pork, fish, and insects is likely to vary widely in different geographical contexts. While chicken and soybean are currently dominating international trade of protein sources (FAOStat), legumes, pulses, seaweed, and yeast-derived foods are being tested as ingredients by the food industry.

In regard to food quality, reducing meat consumption may lower the iron and zinc nutritional status of certain vulnerable groups. For example, in Europe 22% of preschool children, 25% of pregnant women, and 19% of nonpregnant women already have anemia (WHO, 2008). Reductions in red meat consumption also may have food safety implications. Substituting meat with poultry or seafood might increase foodborne illnesses, whereas replacement with pulses and vegetables would reduce them (Lake et al., 2012).

GHG emissions associated with food preparation and food waste are usually unaccounted for in diet shift studies with rare exceptions (Corrado et al. 2019). Dietary supplements (vitamin, minerals and amino acids) are highly recommended for low-meat diets, but they are not considered in GHG mitigation studies of diet shifts, mostly because of lack of LCA data for supplements (Corrado et al. 2019).

The varying proportions of CO_2, CH_4, and N_2O contributions to ruminant-related emissions, with a high proportion of the short-lived methane, make interpretation sensitive to the global warming metrics adopted (Reisinger and Clark 2018) (Lynch 2019). As more intensive systems or other diet alternatives would alter the relative contributions to food of these gases, the choice of metric often changes the ranking of mitigation options (Lynch and Pierrehumbert 2019)(Garnett, 2011). Most projections related to diet shifts do not account for the potential of methane inhibitors, non-symbiotic nitrogen fixation, advances in livestock and forage genetics, and other emerging technologies in the livestock sector, some of which are close to market launch (Jayanegara et al. 2018).

In a systems view, dairy and wool production can be affected if reductions in ruminant meat demand take place. While beef production sytems are often characterised by low energy and protein efficiency, milk production is as efficient energetically as egg production and second after eggs in protein conversion efficiency among animal-based proteins (Eshel et al. 2016).

In summary, systems level analyses revealed wide variation in mitigation estimates of diet shifts, in part due to differing accounting for the main interactions. There is *robust evidence* that diet shifts can mitigate GHG emissions but *low agreement* on how much could be achieved and what would be the effectiveness of interventions to promote diet shifts. In high-income industrialised countries, there is scope for reducing consumption of livestock produce with tangible environmental benefits; in developing countries, high meat-based diets are less prevalent and scope for reductions may be more limited, but there are options for encouraging nutritition transitions towards healthy diets.

Section SM5.6

Global meat consumption

The issue of global meat consumption as a driver of GHG emission, can be weighed against the requirements of healthy diet. Healthy and sustainable diets are high in coarse grains, pulses, fruits and vegetables, and nuts and seeds; low in energy-intensive animal-sourced and discretionary foods (such as sugary beverages and fats); and have a carbohydrate threshold. Based on the potential impact of suboptimal diets on non-communicable diseases (NCD) mortality and morbidity, the World Health Organization (WHO) and the EAT-LANCET report (Willett et al. 2019) highlighted the need for improving diets across nations and made recommendations on how to balance nutrition to prevent malnutrition. The source of protein is not limited to meat; it is found in fish, vegetable and insects. The range of options in balancing protein sources runs primarily into cultural resistance, food habits, economic conditions and the social and economic factors influencing how the food system affects climate and land.

Most recent analyses, like the EAT-LANCET (Willett et al. 2019) work, show that reductions in consumption, especially of red meat, apply to over-consumers, while scope remains for growth in consumption in Low- and Middle-Income Countries (LMICs).

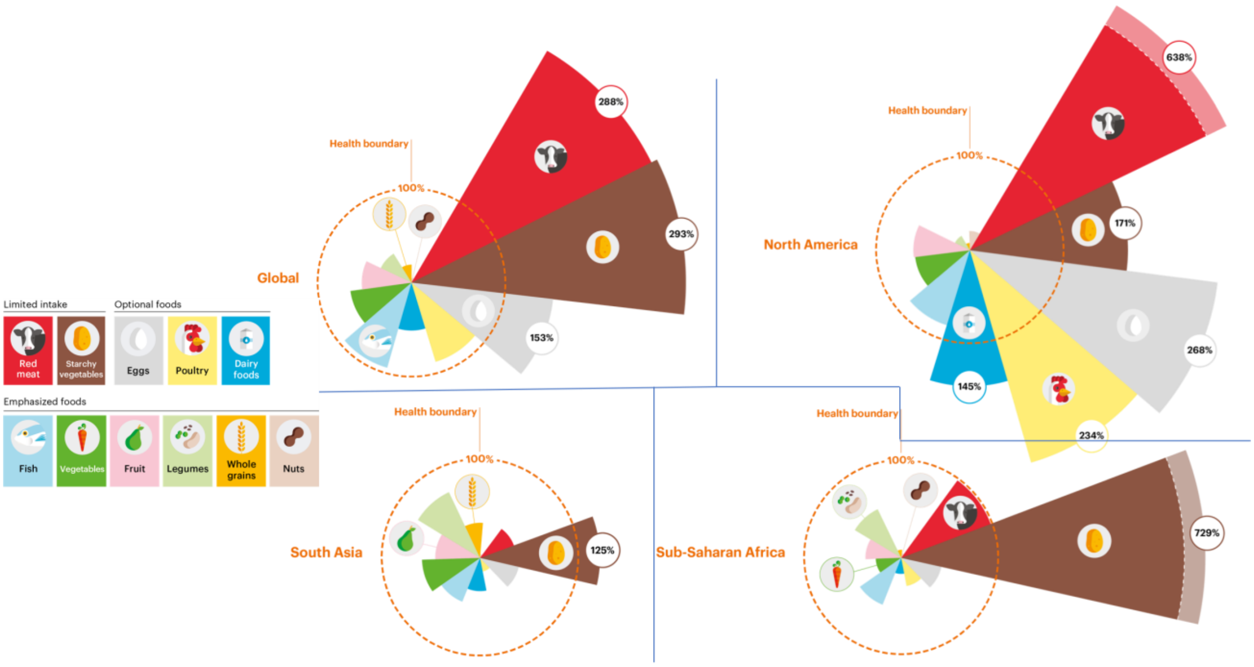

Figure SM5.3 The "diet gap" between current dietary patterns and intakes of food in the planetary health diet (Willett et al. 2019).

From the climate and land perspectives, there is a difference between red meat production and other meat production (Willett et al. 2019). The impacts of meat production will depend on resource use intensity to produce meat calories, the land and climate footprints of the processing and supply chains, and the scale of the production systems (i.e., livestock on crop by-products vs. pasture vs. intensive grain-fed) (Willett et al. 2019). Hence, the question is not about eating less meat for everyone, but to adopt sustainable supply and consumption practices across a broad range of food systems.

The biggest challenge to achieve changes in meat consumption is on how to start a transition that has increasing diversity of food sources with lower land and water requirements and GHG emissions. This could be a gradual transition that recognises the need for just transitions for people whose livelihoods depend on (red) meat production. In this regard, all parts of the food system, including production, trade, and consumption, play important roles.

Section SM5.7

Governance

Governance of climate change and governance of food systems have been developed independently of each other. This section highlights the main characteristics of food and climate governance and assesses what options may exist for establishing arrangements that link the two. See Chapter 7 for important characteristics of governance and institutions; here we describe those relevant for enhancing the interactions between climate change and food systems.

In the governance of climate change, Huitema et al. (2016) highlighted differences between mitigation and adaptation. Mitigation often requires global agreements and national policies while adaptation requires local and regional considerations. However, in the case of food systems this difference does not apply, because mitigation measures also require local actions (e.g., at the farm level), while adaptation actions may also require measures at global and national levels (such as emergency food aid for climate disasters and food safety nets).

Governance of food systems holds particular challenges because it is only recently that a systems approach has been embraced by policy-makers. (Rivera-Ferre et al. 2013) proposed principles for food systems management considering them as complex socioecological systems (SES) including: learning, flexibility, adaptation, participation, diversity enhancement, and precaution. These principles are part of the framework of adaptive governance (see Chapter 7). Termeer et al. (2018) developed a diagnostic framework with five principles to assess governance options appropriate to food systems: 1) system-based problem framing; 2) connectivity across boundaries to span siloed governance structures and include non-state actors; 3) adaptability to flexibly respond to inherent uncertainties and volatility; 4) inclusiveness to facilitate support and legitimacy; and 5) transformative capacity to overcome path dependencies and create conditions to foster structural change.

Both the food and climate systems require integrated governance and institutions (*high confidence*). These need to span government levels and actors across a wide range of sectors including agriculture, environment, economic development, health, education, and welfare (Misselhorn et al. 2012). For climate and food system management, the creation of government entities or ministerial units responsible for coordinating among these ministries (horizontal coordination) and for cutting across different administrative levels (vertical coordination) have been proposed (Orr et al. 2017).

However, integration is not easy. Termeer et al. (2018) analysed three South African governance arrangements that explicitly aim for a holistic system-based approach. They found that they were not delivering the expected outcomes due to reversion to technical one-dimensional problem framing. Issues included dominance of single departments, limited attention to monitoring and flexible responses, and exclusion of those most affected by food insecurity. Newell et al. (2018) analysed the governance process of climate smart agriculture (CSA) from global to local scales for Kenya and found a triple disconnect between global, national, and local scales. Different levels of authority and actors imposed their own framing of CSA, and how to implement it.. As a result of the competition among different actors, siloed policy practices were reproduced.

Food systems governance must also include governance of the resources needed to produce food, which vary from land tenure (see chapter 7) and seed sovereignty (see Chapter 6), to other resources such as soil fertility. Montanarella and Vargas (2012) proposed a supranational structure to guarantee soil conservation on all continents, such as the Global Soil Partnership. This can also apply for the governance of food and climate systems.

Polycentric and multiscalar governance structures have been proposed for coping with climate change to address both mitigation and adaptation (Ostrom 2010), and were suggested by Rivera-Ferre et al.

(2013) for food systems. A polycentric approach provides more opportunities for experimentation and learning across levels (Cole 2015), entails many policy experiments from which policymakers at various levels of governance can learn (Ostrom 2010), and contributes to building trust among stakeholders (e.g., nation states, public and private sectors, civil society). Polycentric approaches have been suggested for the Sustainable Development Goals (SDGs) (Monkelbaan 2019).

Another governance option suggested for the SDGs (Monkelbaan 2019) are already implemented in global atmospheric and marine agreements (e.g., the Montreal protocol (De Búrca et al. 2014; Armeni 2015) is global experimentalist governance). Global experimentalist governance is an institutionalised process of participatory and multilevel collective problem-solving, in which the problems (and the means of addressing them) are framed in an open-ended way, and subjected to periodic revision by peer review in the light of locally generated knowledge (De Búrca et al. 2014), This favours learning, participation and cooperation (Armeni 2015). This form of governance can establish processes that enable unimagined alternatives.

Institutions

As Candel (2014) highlighted, based on a systematic review of food security governance focused on hunger, global governance of food security is lacking because there is no institution with a mandate to address concerns across sectors and levels. No international organisation deals with food security in a holistic and inclusive manner. This results in overlapping (often conflicting) norms, rules and negotiations that generate a "regime complex" (Margulis 2013), particularly in regard to agriculture and food, international trade and human rights (e.g. UN Committee of World Food Security (CFS), WTO, G8, G20). In climate change governance there are also multiple overlapping institutions with often-conflicting rules and actors (Keohane and Victor 2011).

New multi-stakeholder governance arrangements are emerging, such as the Global Agenda for Sustainable Livestock (Breeman et al. 2015) and the CFS (Duncan 2015). Also relevant in food systems and climate change governance is that food security governance is spread across domains, sectors and spatial scales (global, regional, national, local, community, household, or individual) with a lack of coherency and coordination across multiple scales (*high confidence*). Thus, a major challenge is to coordinate all these domains, sectors and scales.

It is important to consider the variety of actors involved in food security governance at all levels (international bodies, civil society organisations (CSOs), nation states, public sector groups, and private sector entities), with different agendas and values. But new in this regard is the participation of CSOs that can provide the policy-making process with bottom-up knowledge to identify food insecurity issues and locally relevant responses. CSOs can also contribute to multi-sector and multi-scalar approaches by bridging government agencies and levels (Candel 2014). Thus, to facilitate coordination and coherence, new adaptive governance enables interactions across multiple levels and scales (Pereira and Ruysenaar 2012) and the use of "boundary organisations" (Candel 2014). To address different narratives regarding food security (Rivera-Ferre 2012; Lang and Barling 2012), a first step is to agree on basic principles and values (Margulis 2013).

In this regard, an opportunity to address food systems governance challenges arises within the UN Committee on World Food Security (CFS), where diverse actors, voices and narratives are integrated in the global food security governance. As a point of departure, the CFS could provide the platform to develop global experimentalist governance in food sytems (Duncan 2015; Duncan and Barling 2012) providing a combination of bottom-up and top-down initiatives (Lambek 2019). However, the existence of overlapping structures with different focuses on food security and power may hinder the potential of this institution. (Margulis and Duncan 2016).

Mainstreaming of collaborative and more inclusive modes of governance, such as those displayed at the CFS, are needed to effectively address thehe impacts of a changing planet on food systems (Barling and Duncan 2015) and improve the balance of sustainable production and food consumption. Despite improvements in global food security, food systems and climate governance, the main focus is still on food security as undernutrition. New challenges will arise from the increasing evidence of the burden of obesity, for which other institutions, focused on nutrition, will be needed. The new Global Strategy Framework for Food Security and Nutrition (Committee on World Food Security 2017) of the CFS provides a new overarching framework for food security and nutrition strategies, policies and actions that includes environmental concerns within a food system approach and a broad vision of food and nutrition security. This framework fits within the "governance through goals" provided by the SDGs (Biermann et al. 2017).

Both in climate change and food systems, the sub-national governance at the level of cities and communities is also becoming relevant in terms of responses (*high evidence, high agreement*). From a climate change perspective (see Chapter 7 for more examples) transnational municipal networks, particularly transnational municipal climate networks, have played a key role in climate change mitigation and have potential to facilitate adaptation (Fünfgeld 2015; Busch et al. 2018; Rosenzweig et al. 2018). Efficient food systems require subnational governments to include food policy councils (Feenstra 2002; Schiff 2008) and cities networks to address food systems challenges (e.g., Sustainable Food Cities in the UK or Agroecological Cities in Spain). Transition Towns are engaged in common principles towards sustainable development, including food systems transformation for food security (Sage 2014), health and well-being (Richardson et al. 2012), and climate change (Taylor Aiken 2015).

Scope for expanded policies

The interaction of production-based support through agricultural policy, coupled with agricultural research investment and the development of frameworks to liberalise trade has led to a range of consequences for global and local food systems. Together, these policies have shaped the food system and incentivised global intensification of agriculture, and significant gains in global production. However, jointly they have also incentivised a concentration on a small number of energy-dense commodity crops grown at large scales (*high confidence*) (just eight crops supply 75% of the world's consumed calories (West et al. 2014)). The production of these commodity crops underpin global dietary transitions, leading to dietary homogenisation (based primarily on starchy grains/tubers, vegetable oil, sugar and livestock produce) (Khoury et al. 2014).

Global intensification of agriculture, as well as increasing the supply of affordable calories, has impacted soil, water, air quality and biodiversity in major and negative ways (Dalin et al. 2017; Tamea et al. 2016; Newbold et al. 2015; García-Ruiz et al. 2015; Amundson et al. 2015; Paulot and Jacob 2014). Importantly in the context of this report, a narrow focus on productivity has led to a food system that emits a large proportion of GHGs (Section 5.4), is fragile in the face of climate shocks (Section 5.3) and from which food is used inefficiently (through waste and over-consumption, Section 5.5.2.5). Mitigation of climate change, as well as adaptation, can then arise from a transformation of the food system to one that provides nutrition and health (Willett et al. 2019; Springmann et al. 2018b,a; Godfray et al. 2018; Ramankutty et al. 2018; Chaudhary et al. 2018). There is therefore *medium confidence*, that continued focus on the past drivers of the food system will be detrimental for climate change and food security.

Addressing this challenge requires action across the food system to enhance synergies and co-benefits and minimise trade-offs among multiple objectives of food security, adaptation and mitigation (Sapkota et al. 2017; Palm et al. 2010; Jat et al. 2016; Sapkota et al. 2015) (Section 5.6), as well as broader environmental goods exemplified by the SDG framework such as water, air-quality, soil health and biodiversity (Obersteiner et al. 2016; Pradhan et al. 2017). In short, this requires greater policy alignment and coherence between traditionally separate policy domains to recognise the systemic nature of the problem. For example, aligning the policy goals of sustainable land management for the purposes of managing both food security and biodiversity (Meyfroidt 2017; Wittman et al. 2017), or public health and agricultural policies (Thow et al. 2018) that can drive mitigation, as well as the enabling conditions of land rights, tenure and ownership. Significant co-benefits can arise from integrated food systems policies, as well as integrated approaches to generating evidence to underpin coherent policy, exemplified, for example, by the EU's integrated research and innovation strategy "Food2030" that aligns agriculture, environment, nutrition and research policy (European Commission 2018).

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Chapter 6: Interlinkages between Desertification, Land Degradation, Food Security and GHG fluxes: synergies, trade-offs and Integrated Response Options

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Executive summary

The land challenges, in the context of this report, are climate change mitigation, adaptation, desertification, land degradation, and food security. The chapter also discusses implications for Nature's Contributions to People (NCP), including biodiversity and water, and sustainable development, by assessing intersections with the Sustainable Development Goals (SDGs). The chapter assesses response options that could be used to address these challenges. These response options were derived from the previous chapters and fall into three broad categories (land management, value chain, and risk management).

The land challenges faced today vary across regions; climate change will increase challenges in the future, while socioeconomic development could either increase or decrease challenges (*high confidence*). Increases in biophysical impacts from climate change can worsen desertification, land degradation, and food insecurity (*high confidence*). Additional pressures from socioeconomic development could further exacerbate these challenges; however, the effects are scenario dependent. Scenarios with increases in income and reduced pressures on land can lead to reductions in food insecurity; however, all assessed scenarios result in increases in water demand and water scarcity (*medium confidence*). {6.1}

The applicability and efficacy of response options are region and context specific; while many value chain and risk management options are potentially broadly applicable, many land management options are applicable on less than 50% of the ice-free land surface (*high confidence*). Response options are limited by land type, bioclimatic region, or local food system context (*high confidence*). Some response options produce adverse side-effects only in certain regions or contexts; for example, response options that use freshwater may have no adverse side effects in regions where water is plentiful, but large adverse side effects in regions where water is scarce (*high confidence*). Response options with biophysical climate effects (e.g., afforestation, reforestation) may have different effects on local climate depending on where they are implemented (*medium confidence*). Regions with more challenges have fewer response options available for implementation (*medium confidence*). {6.1, 6.2, 6.3, 6.4}

Nine options deliver medium to large benefits for all five land challenges (*high confidence*). The options with medium to large benefits for all challenges are increased food productivity, improved cropland management, improved grazing land management, improved livestock management, agroforestry, improved forest management, increased soil organic carbon content, fire management and reduced post-harvest losses. A further two options, dietary change and reduced food waste, have no global estimates for adaptation but have medium to large benefits for all other challenges (*high confidence*). {6.3, 6.4}

Five options have large mitigation potential (> 3 $GtCO_2e\ yr^{-1}$) without adverse impacts on the other challenges (*high confidence*). These are increased food productivity, reduced deforestation and degradation, increased soil organic carbon content, fire management and reduced post-harvest losses. Two further options with large mitigation potential, dietary change and reduced food waste, have no global estimates for adaptation but show no negative impacts across the other challenges. Five options: improved cropland management, improved grazing land managements, agroforestry,

integrated water management and forest management, have moderate mitigation potential, with no adverse impacts on the other challenges (*high confidence*). {6.3.6}

Sixteen response options have large adaptation potential (>25 million people benefit), without adverse side-effects on other land challenges (*high confidence*). These are increased food productivity, improved cropland management, agroforestry, agricultural diversification, improved forest management, increased soil organic carbon content, reduced landslides and natural hazards, restoration and reduced conversion of coastal wetlands, reduced post-harvest losses, sustainable sourcing, management of supply chains, improved food processing and retailing, improved energy use in food systems, livelihood diversification, use of local seeds, and disaster risk management (*high confidence*). Some options (such as enhanced urban food systems or management of urban sprawl) may not provide large global benefits but may have significant positive local effects without adverse effects (*high confidence*). {6.3, 6.4}

Seventeen of 40 options deliver co-benefits or no adverse side-effects for the full range of NCPs and SDGs; only three options (afforestation, bioenergy and BECCS and some types of risk sharing instruments, such as insurance) have potentially adverse side-effects for five or more NCPs or SDGs (*medium confidence*). The 17 options with co-benefits and no adverse side-effects include most agriculture- and soil-based land management options, many ecosystem-based land management options, improved forest management, reduced post-harvest losses, sustainable sourcing, improved energy use in food systems, and livelihood diversification (*medium confidence*). Some of the synergies between response options and SDGs include positive poverty reduction impacts from activities like improved water management or improved management of supply chains. Examples of synergies between response options and NCPs include positive impacts on habitat maintenance from activities like invasive species management and agricultural diversification. However, many of these synergies are not automatic, and are dependent on well-implemented activities requiring institutional and enabling conditions for success. {6.4}

Most response options can be applied without competing for available land; however, seven options result in competition for land (*medium confidence*). A large number of response options do not require dedicated land, including several land management options, all value chain options, and all risk management options. Four options could greatly increase competition for land if applied at scale: afforestation, reforestation, and land used to provide feedstock for BECCS and biochar, with three further options: reduced grassland conversion to croplands, restoration and reduced conversion of peatlands and restoration and reduced conversion of coastal wetlands having smaller or variable impacts on competition for land. Other options such as reduced deforestation and degradation, restrict land conversion for other options and uses. Expansion of the current area of managed land into natural ecosystems could have negative consequences for other land challenges, lead to the loss of biodiversity, and adversely affect a range of NCPs (*high confidence*). {6.3.6, 6.4}

Some options, such as bioenergy and BECCS, are scale dependent. The climate change mitigation potential for bioenergy and BECCS is large (up to 11 $GtCO_2$ yr^{-1}); however, the effects of bioenergy production on land degradation, food insecurity, water scarcity, GHG emissions, and other environmental goals are scale and context specific (*high confidence*). These effects depend on the scale of deployment, initial land use, land type, bioenergy feedstock, initial carbon stocks, climatic region and management regime (*high confidence*). Large areas of monoculture bioenergy crops that displace other land uses can result in land competition, with adverse effects for food production, food consumption, and thus food security, as well as adverse effects for land degradation, biodiversity, and water scarcity (*medium confidence*). However, integration of bioenergy into sustainably managed agricultural landscapes can ameliorate these challenges (*medium confidence*). {6.2, 6.3, 6.4, Cross-Chapter Box 7 in this Chapter}

Response options are interlinked; some options (e.g., land sparing and sustainable land management options) can enhance the co-benefits or increase the potential for other options (*medium confidence*). Some response options can be more effective when applied together (*medium confidence)*; for example, dietary change and waste reduction expand the potential to apply other options by freeing as much as 5.8 Mkm^2 (0.8-2.4 Mkm^2 for dietary change; ~2 Mkm^2 for reduced post-harvest losses, and 1.4 Mkm^2 for reduced food waste) of land (*low confidence*). Integrated water management and increased soil organic carbon can increase food productivity in some circumstances. {6.4}

Other response options (e.g., options that require land) may conflict; as a result, the potentials for response options are not all additive, and a total potential from the land is currently unknown (*high confidence*). Combining some sets of options (e.g., those that compete for land) may mean that maximum potentials cannot be realised, for example reforestation, afforestation, and bioenergy and BECCS all compete for the same finite land resource so the combined potential is much lower than the sum of potentials of each individual option calculated in the absence of alternative uses of the land (*high confidence*). Given the interlinkages among response options and that mitigation potentials for individual options assume that they are applied to all suitable land, the total mitigation potential is much lower than the sum of the mitigation potential of the individual response options (*high confidence*). {6.4}

The feasibility of response options, including those with multiple co-benefits, is limited due to economic, technological, institutional, socio-cultural, environmental and geophysical barriers (*high confidence*). A number of response options (e.g., most agriculture-based land management options, forest management, reforestation and restoration) have already been implemented widely to date (*high confidence*). There is robust evidence that many other response options can deliver co-benefits across the range of land challenges, yet these are not being implemented. This limited application is evidence that multiple barriers to implementation of response options exist (*high confidence*). {6.3, 6.4}

Coordinated action is required across a range of actors, including business, consumers, land managers, indigenous and local communities and policymakers to create enabling conditions for adoption of response options (*high confidence*). The response options assessed face a variety of barriers to implementation (economic, technological, institutional, socio-cultural, environmental and geophysical) that require action across multiple actors to overcome (*high confidence*). There are a variety of response options available at different scales that could form portfolios of measures applied by different stakeholders from farm to international scales. For example, agricultural diversification and use of local seeds by smallholders can be particularly useful poverty reduction and biodiversity conservation measures, but are only successful when higher scales, such as national and international markets and supply-chains, also value these goods in trade regimes, and consumers see the benefits of purchasing these goods. However, the land and food sectors face particular challenges of institutional fragmentation, and often suffer from a lack of engagement between stakeholders at different scales (*medium confidence*). {6.3, 6.4}

Delayed action will result in an increased need for response to land challenges and a decreased potential for land-based response options due to climate change and other pressures (*high confidence*). For example, failure to mitigate climate change will increase requirements for adaptation and may reduce the efficacy of future land-based mitigation options (*high confidence*). The potential for some land management options decreases as climate change increases; for example, climate alters the sink capacity for soil and vegetation carbon sequestration, reducing the potential for increased soil organic carbon (*high confidence*). Other options (e.g., reduced deforestation and degradation) prevent further detrimental effects to the land surface; delaying these options could lead to increased deforestation, conversion, or degradation, serving as increased sources of GHGs and having concomitant negative impacts on NCPs (*medium confidence*). Carbon dioxide removal (CDR) options, like reforestation, afforestation, bioenergy and BECCS, are used to compensate for unavoidable emissions in other sectors; delayed action will result in larger and more rapid deployment later (*high confidence*). Some response options will not be possible if action is delayed too long; for example, peatland restoration might not be possible after certain thresholds of degradation have been exceeded, meaning that peatlands could not be restored in certain locations (*medium confidence*) {6.2, 6.3, 6.4}.

Early action, however, has challenges including technological readiness, upscaling, and institutional barriers (*high confidence*). Some of the response options have technological barriers that may limit their wide-scale application in the near-term (*high confidence*). Some response options, e.g., BECCS, have only been implemented at small-scale demonstration facilities; challenges exist with upscaling these options to the levels discussed in this Chapter (*medium confidence*). Economic and institutional barriers, including governance, financial incentives and financial resources, limit the near-term adoption of many response options, and 'policy lags', by which implementation is delayed by the slowness of the policy implementation cycle, are significant across many options (*medium confidence*). Even some actions that initially seemed like 'easy wins' have been challenging to implement, with stalled policies for REDD+ providing clear examples of how response options need sufficient funding, institutional support, local buy-in, and clear metrics for success, among other necessary enabling conditions. {6.2, 6.4}

Some response options reduce the consequences of land challenges, but do not address underlying drivers (*high confidence*). For example, management of urban sprawl can help reduce the environmental impact of urban systems; however, such management does not address the socioeconomic and demographic changes driving the expansion of urban areas. By failing to address the underlying drivers, there is a potential for the challenge to re-emerge in the future (*high confidence*). {6.4}

Many response options have been practiced in many regions for many years; however, there is limited knowledge of the efficacy and broader implications of other response options (*high confidence*). For the response options with a large evidence base and ample experience, further implementation and upscaling would carry little risk of adverse side-effects (*high confidence*). However, for other options, the risks are larger as the knowledge gaps are greater; for example, uncertainty in the economic and social aspects of many land response options hampers the ability to predict their effects (*medium confidence*). Furthermore, Integrated Assessment Models, like those used to develop the pathways in the IPCC Special Report on Global Warming of 1.5°C (SR15), omit many of these response options and do not assess implications for all land challenges (*high confidence*). {6.4}

6.1 Introduction

6.1.1 Context of this chapter

This chapter focuses on the interlinkages between response options[1] to deliver climate mitigation and adaptation, to address desertification and land degradation, and to enhance food security, and also assesses reported impacts on Nature's Contributions to People (NCP) and contributions to the UN Sustainable Development Goals (SDG). By identifying which options provide the most co-benefits with the fewest adverse side-effects, this chapter aims to provide *integrated response options* that could co-deliver across the range of challenges. This chapter *does not consider*, in isolation, response options that affect only one of climate mitigation, adaptation, desertification, land degradation, or food security, since these are the subjects of Chapters 2–5; this chapter *considers only* interlinkages between response options and two or more of these challenges in the land sector.

Since the aim is to assess and provide guidance on integrated response options, each response option is first described and categorised drawing on previous chapters 2-5 (Section 6.2), and their impact on climate mitigation / adaptation, desertification, land degradation, and food security are quantified (Section 6.3). The feasibility of each response option, respect to costs, barriers, saturation and reversibility is then assessed (Section 6.4.1), before considering their sensitivity to future climate change (Section 6.4.2).

The *co-benefits* and *adverse side-effects*[2] of each integrated response option across the five land challenges, and their impacts on the NCP and the SDG, are then assessed in Section 6.4.3. In section 6.4.4, the spatial applicability of these integrated response options is assessed in relation to the location of the challenges with the aim of identifying which options have the greatest potential to co-deliver across the challenges, and the contexts and circumstances in which they do so. Interlinkages among response options and challenges in future scenarios are also assessed in 6.4.4. Finally, section 6.4.5 discusses the potential consequences of delayed action.

In providing this evidence-based assessment, drawing on the relevant literature, this chapter does not assess the merits of policies to deliver these integrated response options - Chapter 7 assesses the various policy options currently available to deliver these interventions - rather this chapter provides an assessment of the integrated response options and their ability to co-deliver across the multiple challenges addressed in this Special Report.

6.1.2 Framing social challenges and acknowledging enabling factors

In this section we outline the approach used in assessing the evidence for interactions between response options to deliver climate mitigation and adaptation, to prevent desertification and land degradation, and to enhance food security. Overall, while defining and presenting the response options to meet these goals is the primary goal of this chapter, we note that these options must not be considered only as technological interventions, or one-off actions. Rather, they need to be understood as responses to socio-ecological challenges whose success will largely depend on external enabling factors. There have been many previous efforts at compiling positive response options that meet numerous Sustainable Development Goals, but which have not resulted in major shifts in

[1] FOOTNOTE: Many of the response options considered are *sustainable land management* options, but several response options are not based on land management, for example those based on value chain management and governance and risk management options

[2] FOOTNOTE: We use the IPCC AR5 WGIII definitions of co-benefits and adverse side-effect – see glossary. Co-benefits and adverse side-effects can be biophysical and/or socio-economic in nature, and all are assessed as far as the literature allows.

implementation; for example, online databases of multiple response options for sustainable land management (SLM), adaptation, and other objectives have been compiled by many donor agencies, including World Overview of Conservation Approaches and Technologies (WOCAT), Climate Adapt, and the Adaptation Knowledge Portal, (Schwilch et al.,2012)[3]. Yet clearly barriers to adoption remain, or these actions would have been more widely used by now. Much of the scientific literature on barriers to implementing response options focuses on the individual and household level, and discusses limits to adoption, often primarily identified as economic factors (Nigussie et al. 2017; Dallimer et al. 2018). While a useful approach, such studies often are unable to account for the larger enabling factors that might assist in more wide scale implementation (chapter 7 discusses these governance factors and barriers to be overcome in more detail).

Instead, this chapter proposes that each response option identified and assessed needs to be understood as an intervention within complex socio-ecological systems (SES) (introduced in Chapter 1). In this understanding, physical changes affect human decision-making over land and risk management options, as do economics, policies, and cultural factors, which in turn may drive additional ecological change (Rawlins et al.,2010). This co-evolution of responses within an SES provides a more nuanced understanding of the dynamics between drivers of change and impacts of interventions. Thus, in discussions of the 40 specific response options in this chapter, it must be kept in mind that all need to be contextualised within the specific SES in which they are deployed (see Figure 6.1). Framing response options within SESs also recognises the interactions *between* different response options. However, a major problem within SESs is that the choice and use of different response options requires knowledge of the problems they are aimed at solving, which may be unclear, contested, or not shared equally among stakeholders (Carmenta et al., 2017). Drivers of environmental change often have primarily social or economic rather than technological roots, which requires acknowledgement that response options that do not aim at reducing the drivers of change may thus be less successful (Schwilch et al., 2014).

[3] FOOTNOTE: E.g. see https://qcat.wocat.net/en/wocat/; https://climate-adapt.eea.europa.eu; https://www4.unfccc.int/sites/NWPStaging/Pages/Home.aspx

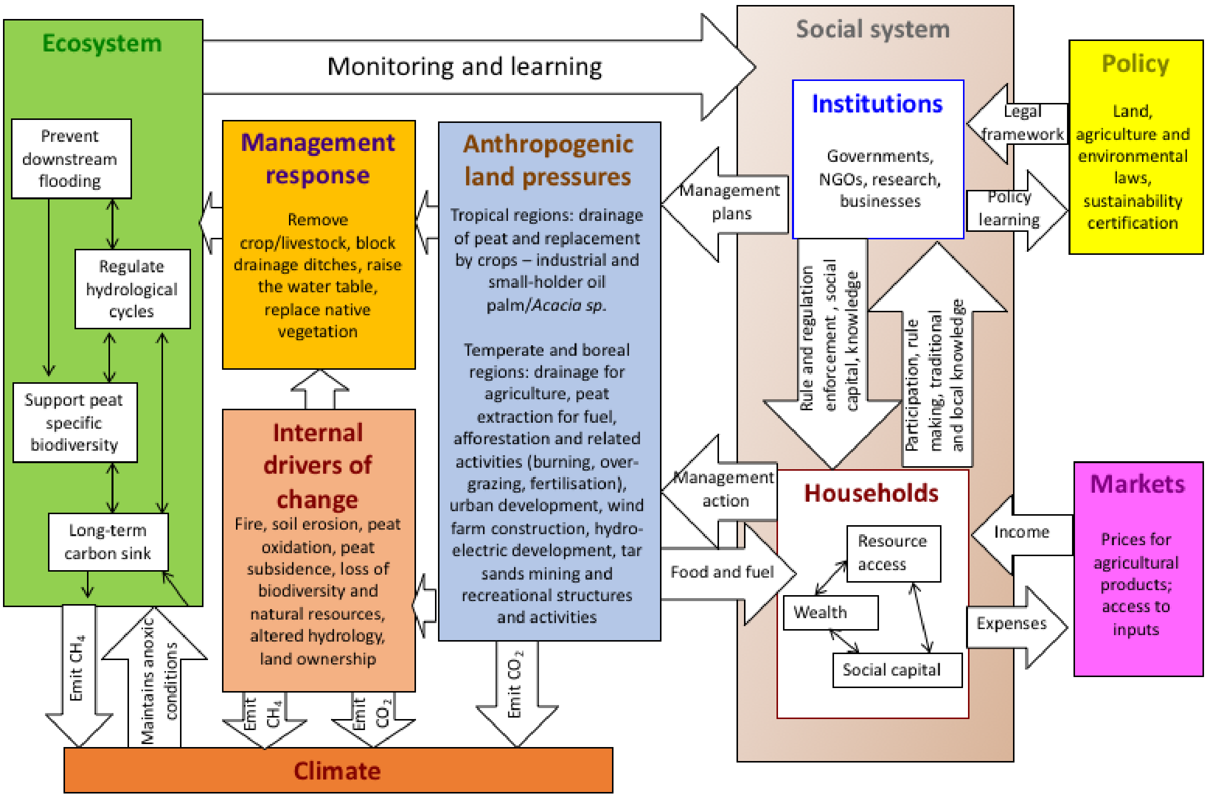

Figure 6.1 Model to represent a social-ecological system of one of the integrated response options in this chapter, using restoration and reduced impact of peatlands as an example. The boxes show systems (ecosystem, social system), external and internal drivers of change and the management response – here enacting the response option. Unless included in the internal drivers of change box, all other drivers of change are external (e.g. climate, policy, markets, anthropogenic land pressures). The arrows represent how the systems can influence each other, with key drivers of impact written in the arrow in the direction of effect.

Response options must also account for the uneven distribution of impacts among populations of both environmental change and intervention responses to this change. Understanding the integrated response options available in a given context requires an understanding of the specificities of social vulnerability, adaptive capacity, and institutional support to assist communities, households and regions to reach their capabilities and achievement of the SDG and other social and land management goals. Vulnerability reflects how assets are distributed within and among communities, shaped by factors that are not easily overcome with technical solutions, including inequality and marginalisation, poverty, and access to resources (Adger et al. 2004; Hallegate et al 2016). Understanding why some people are vulnerable and what structural factors perpetuate this vulnerability requires attention to both micro and meso scales (Tschakert et al. 2013). These vulnerabilities create barriers to adoption of even low-cost high-return response options, such as soil carbon management, that may seem obviously beneficial to implement (Mutoko et al. 2014; Cavanagh et al. 2017). Thus, assessment of the differentiated vulnerabilities that may prevent response option adoption needs to be considered as part of any package of interventions.

Adaptive capacity relates to the ability of institutions or people to modify or change characteristics or behaviour so as to cope better with existing or anticipated external stresses (Moss et al. 2001; Brenkert and Malone 2005; Brooks et al. 2005). Adaptive capacity reflects institutional and policy support networks, and has often been associated at the national level with strong developments in the fields of economics, education, health, and governance and political rights (Smit et al. 2001). Areas with low adaptive capacity, as reflected in low Human Development Index scores, might constrain the ability of communities to implement response options (section 6.4.4.1 and Figure 6.7).

Further, while environmental changes like land degradation have obvious social and cultural impacts, as discussed in the preceding chapters, so do response options, and thus careful thought is needed about what impacts are expected and what trade-offs are acceptable. One potential way to assess the impact of response interventions relates to the idea of capabilities, a concept first proposed by economist Amartya Sen (Sen 1992). Understanding capability as the "freedom to achieve well-being" frames a problem as being a matter of facilitating what people aspire to do and be, rather than telling them to achieve a standardised or predetermined outcome (Nussbaum and Sen 1993). Thus a capability approach is generally a more flexible and multi-purpose framework, appropriate to an SES understanding because of its open-ended approach (Bockstael and Berkes 2017). Thus, one question for any decision-maker approaching schematics of response options is to determine which response options lead to increased or decreased capabilities for the stakeholders who are the objects of the interventions, given the context of the SES in which the response option will be implemented.

Section 6.4.3 examines some of the capabilities that are reflected in the UN Sustainable Development Goals (SDG), such as gender equity and education, and assesses how each of the 40 response options may affect those goals, either positively or negatively, through a review of the available literature.

6.1.2.1 Enabling conditions

Response options are not implemented in a vacuum and rely on knowledge production and socio-economic and cultural strategies and approaches embedded within them to be successful. For example, it is well known that "Weak grassroots institutions characterised by low capacity, failure to exploit collective capital and poor knowledge sharing and access to information, are common barriers to sustainable land management and improved food security" (Oloo and Omondi 2017). Achieving broad goals such as reduced poverty or sustainable land management requires conducive enabling conditions, such as attention to gender issues and the involvement of stakeholders, like indigenous peoples, as well as attention to governance, including adaptive governance, stakeholder engagement, and institutional facilitation (see section 6.4.4.3). These enabling conditions – such as gender-sensitive programming or community-based solutions - are not categorised as individual response options in subsequent sections of this chapter because they are conditions that can potentially help improve *all* response options when used in tandem to produce more sustainable outcomes. Chapter 7 picks up on these themes and discusses the ways various policies to implement response options have tried to minimise unwanted social and economic impacts on participants in more depth, through deeper analysis of concepts such as citizen science and adaptive governance. Here we simply note the importance of assessing the contexts within which response options will be delivered, as no two situations are the same, and no single response option is likely to be a 'silver bullet' to solve all land-climate problems, as each option comes with potential challenges and trade-offs (section 6.2), barriers to implementation (section 6.4.1), interactions with other sectors of society (section 6.4.3), and potential environmental limitations (section 6.4.4).

6.1.3 Challenges and response options in current and historical interventions

Land-based systems are exposed to multiple overlapping challenges including climate change (adaptation and mitigation), desertification (Chapter 3), land degradation (Chapter 4) and food insecurity (Chapter 5), as well as loss of biodiversity, ground water stress (from over-abstraction) and water quality. The spatial distribution of these individual land-based challenges is shown in Figure 6.2, based on recent studies and using the following indicators:

- Desertification attributed to land use is estimated from vegetation remote sensing (Figure 3.7c), mean annual change in NDVImax < -0.001 (between 1982-2015) in dryland areas (Aridity Index > 0.65), noting however that desertification has multiple causes (Chapter 3);
- Land degradation (see Chapter 4) is based on a soil erosion (Borrelli et al. 2017) proxy (annual erosion rate of 3 t ha^{-1} or above);
- The climate change challenge for adaptation is based on a dissimilarity index of monthly means of temperature and precipitation between current and end of century scenarios (dissimilarity index equal to 0.7 or above, Netzel and Stepinski 2018), noting however that rapid warming could occur in all land regions (Chapter 2);
- The food security challenge is estimated as the prevalence of chronic undernourishment (higher or equal to 5%) by country in 2015 (FAO 2017a), noting however that food security has several dimensions (see Chapter 5);
- The biodiversity challenge uses threatened terrestrial biodiversity hotspots (areas where exceptional concentrations of endemic species are undergoing exceptional loss of habitat, (Mittermeier et al. 2011), noting however that biodiversity concerns more than just threatened endemic species;
- The groundwater stress challenge is estimated as groundwater abstraction over recharge ratios above one (Gassert et al. 2014) in agricultural areas (croplands and villages);
- The water quality challenge is estimated as critical loads (higher or equal to 1000 kg N km^{-2} or 50 kg P km^{-2}) of nitrogen (N) and phosphorus (P) (Xie and Ringler 2017)

Overlapping land-based challenges affect all land use categories: croplands, rangelands, semi-natural forests, villages, dense settlements, wild forests and sparse trees and barren lands. These land use categories can be defined as anthropogenic biomes, or anthromes, and their global distribution was mapped by Ellis and Ramankutty (2008) (Figure 6.2).

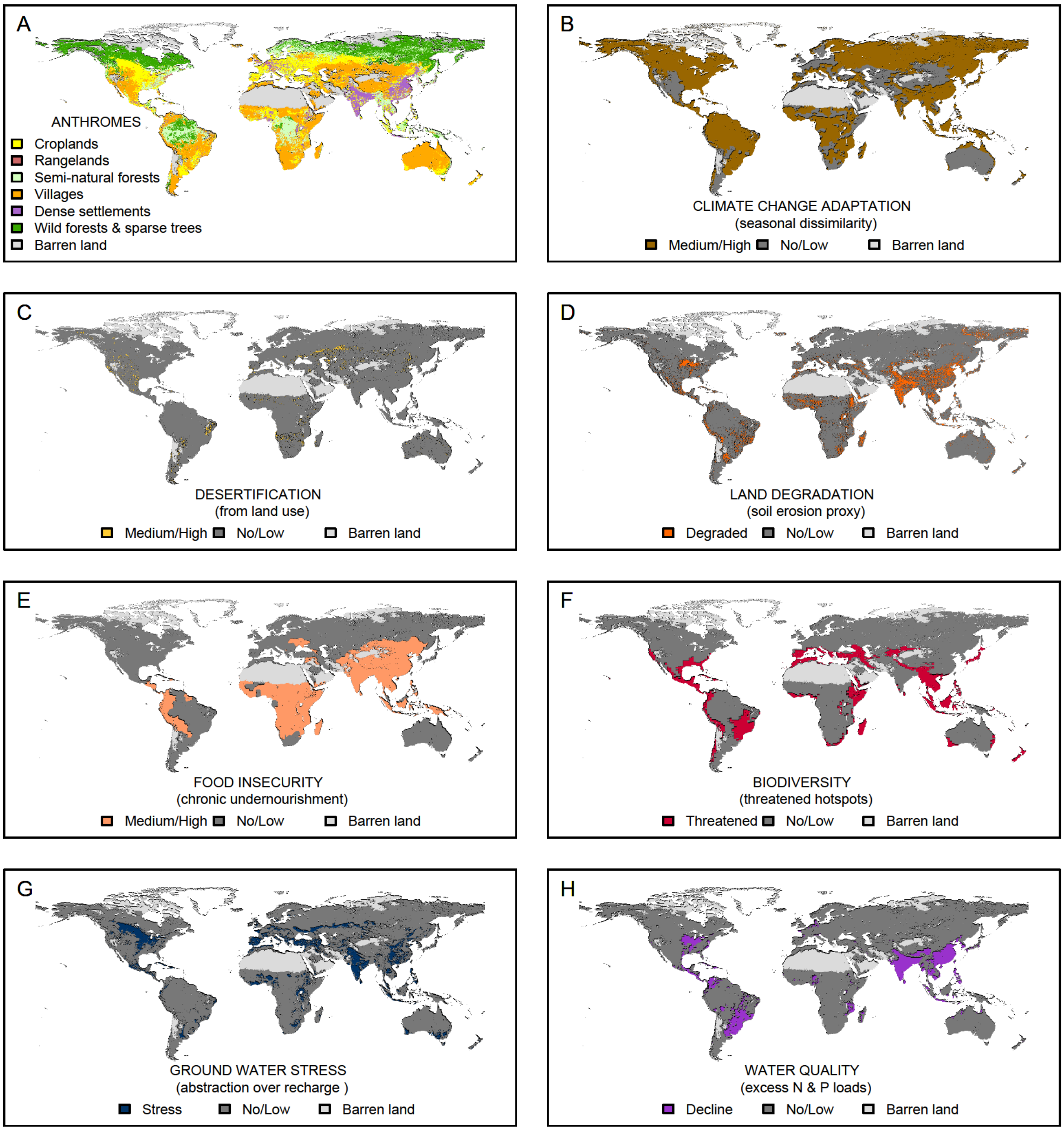

Figure 6.2 Global distributions of land use types and individual land-based challenges. A, land use types (or anthromes, after Ellis and Ramankutty 2008); B, climate change adaptation challenge (estimated from dissimilarity between current and end of century climate scenarios, Netzel and Stepinski 2018); C, desertification challenge (after Chapter 3, Figure 3.7c); D, land degradation challenge (estimated from a soil erosion proxy, one indicator of land degradation Borrelli et al. 2017); E, food security challenge (estimated from chronic undernourishment, a component of food security, FAO 2017a); F, biodiversity challenge (estimated from threatened biodiversity hotspots, a component of biodiversity, Mittermeier et al. 2011); G, groundwater stress challenge (estimated from water over-abstraction, Gassert et al. 2014); H, water quality challenge (estimated from critical N and P loads of water systems, Xie and Ringler 2017).

Table 6.1 Global area of land use types (or anthromes) and current percentage area exposure to individual (overlapping) land-based challenges. See Figure 6.2 and text for further details on criteria for individual challenges.

Land use type (anthrome[a])	Anthrome area	Climate change adaptation (dissimilarity index proxy)[b]	Land degradation (soil erosion proxy)[c]	Desertification (ascribed to land use)[d]	Food security (chronic under nourishment)[e]	Biodiversity (threatened hotspot)[f]	Ground water stress (over abstraction)[g]	Water quality (critical N-P loads)[h]
	% of ice-free land area [1]	*% anthrome area exposed to an individual challenge*						
Dense settlement	1	76	20	3	30	32	-	30
Village	5	70	49	3	78	28	77	59
Cropland	13	68	21	7	28	27	65	20
Rangeland	26	46	14	7	43	21	-	10
Semi-natural forests	14	91	17	0.7	-	21	-	7
Wild forests and sparse trees	17	98	4	0.5	-	2	-	0.3
Barren	19	53	6	0.9	2	4	-	0.4
***Organic soils**	4	95	10	2	9	13	-	6
***Coastal wetlands**	0.6	74	11	2	24	33	-	26
All anthromes	100	69	13	3.2	20	15	12	10

[a] Ellis and Ramankutty (2008) - the global ice-free land area is estimated at 134 Mkm^2; [b] Borrelli et al. 2017; [c] Netzel and Stepinski 2018; [d] From Figure 3.7c, chapter 3; [e] FAO 2017a; [f] Mittermeier et al. 2011; [g] Gassert et al 2014; [h] Xie and Ringler 2017

The majority of the global population is concentrated in dense settlements and villages accounting for less than 7% of the global ice-free land area, while croplands and rangelands use 39% of land. The remainder of the ice-free land area (more than half) is used by semi-natural forests, by wild forests and sparse trees and by barren lands (Table 6.1).

Land use types (or anthromes) are exposed to multiple overlapping challenges. Climate change could induce rapid warming in all land areas (see Chapter 2). In close to 70% of the ice-free land area, the climate change adaptation challenge could be reinforced by a strong dissimilarity between end of century and current temperature and precipitation seasonal cycles (Netzel and Stepinski 2018). Chronic undernourishment (a component of food insecurity) is concentrated in 20% of global ice-free land area. Severe soil erosion (a proxy of land degradation) and desertification from land use affect 13 and 3% of ice-free land area, respectively. Both groundwater stress and severe water quality decline (12 and 10% of ice-free land area, respectively) contribute to the water challenge. Threatened biodiversity hot-spots (15% of ice-free land area) are significant for the biodiversity challenge (Table 6.1).

Since land-based challenges overlap, part of the ice-free land area is exposed to combinations of two or more challenges. For instance, land degradation (severe soil erosion) or desertification from land use and food insecurity (chronic undernourishment) are combined with a strong climate change adaptation challenge (dissimilarity in seasonal cycles) in 4.5% of the ice-free land area (Figure 6.3).

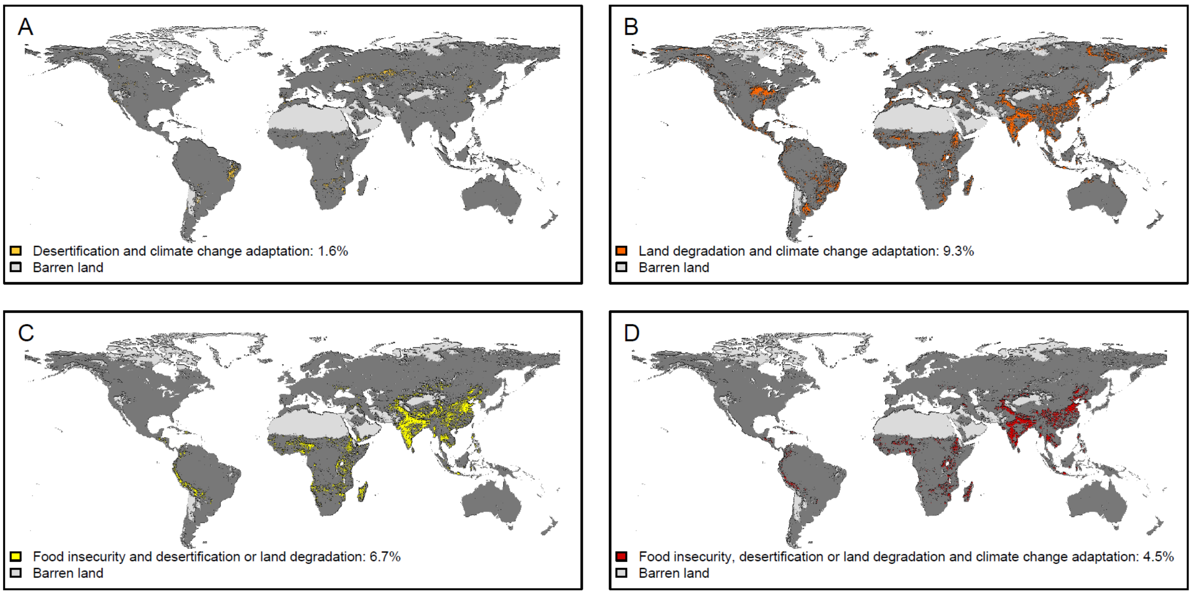

Figure 6.3 Example of overlap between land challenges. A. Overlap between the desertification (from land use) challenge and the climate change adaptation (strong dissimilarity in seasonal cycles) challenge. B. Overlap between the land degradation (soil erosion proxy) challenge and the climate change adaptation challenge. C. Overlap between the desertification or land degradation challenges and the food insecurity (chronic undernourishment) challenge. D. Overlap between challenges shown in C and the climate change adaptation challenge. For challenges definitions, see text; references as in Figure 6.2.

The global distribution of land area by the number of overlapping land challenges (Figure 6.4) shows: the least exposure to land challenges in barren lands; less frequent exposure to two or more challenges in wild forests than in semi-natural forests; more frequent exposure to two or more challenges in agricultural anthromes (croplands and rangelands) and dense settlements than in forests; most frequent exposure to 3 or more challenges in villages compared to other land use types. Therefore, land use types intensively used by humans are, on average, exposed to a larger number of challenges than land use types (or anthromes) least exposed to human use.

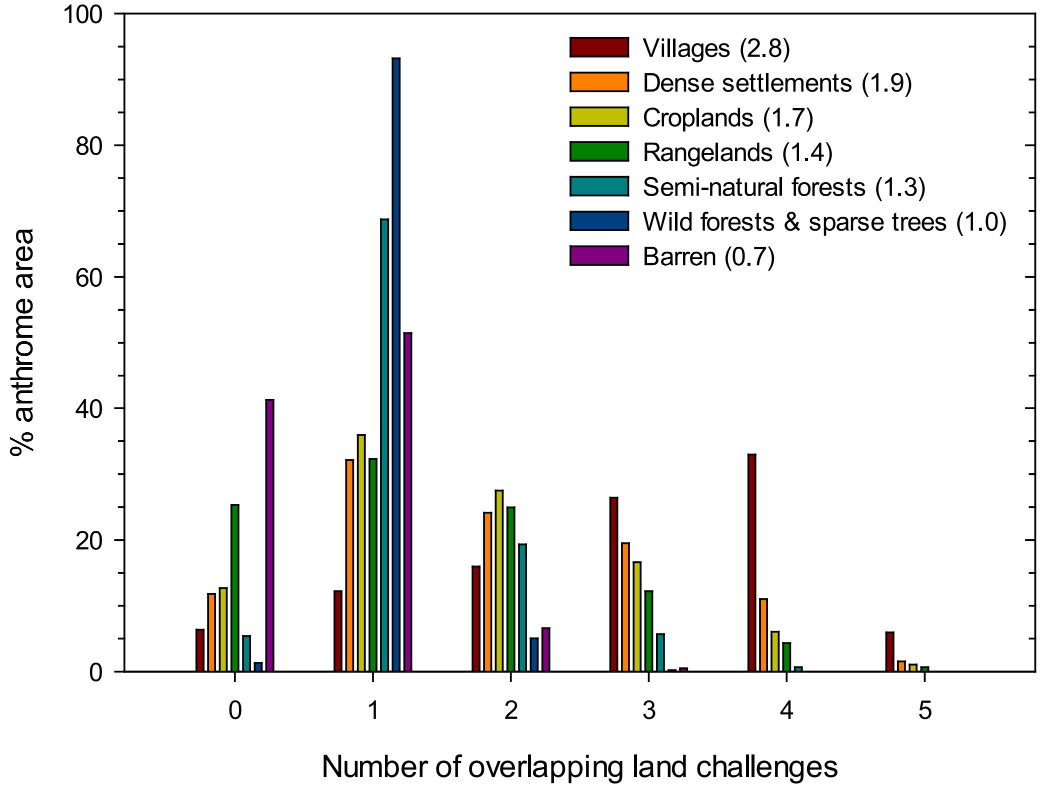

Figure 6.4 Percentage distribution of land use type (or anthrome) area by number of overlapping land challenges for the villages, dense settlements, croplands, rangelands, semi-natural forests, wild forests & sparse trees and barren land use types. Values in brackets show the mean number of land challenges per land use type. Land challenges include desertification (from land use), land degradation (soil erosion proxy), climate change adaptation (seasonal dissimilarity proxy), food security (chronic undernourishment), biodiversity (threatened hot spots), groundwater stress (over abstraction) and water quality (critical N and P loads).

Case studies located in different world regions are presented for each anthrome, in order to provide historical context on the interlinkages between multiple challenges and responses (Box 6.1: A to E). Taken together, these case studies illustrate the large contrast across anthromes in land-based interventions and the way these interventions respond to combinations of challenges.

Box 6.1 Case studies by anthrome type showing historical interlinkages between land-based challenges and the development of local responses

A. Croplands. Land degradation, groundwater stress and food insecurity: soil and water conservation measures in the Tigray region of Ethiopia

In northern Ethiopia, the Tigray region is a drought-prone area that has been subjected to severe land degradation (Frankl et al. 2013) and to recurrent drought and famine during 1888–1892, 1973–1974 and 1984–1985 (Gebremeskel et al. 2018). The prevalence of stunting and being underweight among children under five years is high (Busse et al. 2017) and the region was again exposed to a severe drought during the strong El Niño event of 2015–2016. Croplands are the dominant land-use type, with approximately 90% of the households relying on small-scale plough-based cultivation. Gullies affect nearly all slopes and frequently exceed 2 m in depth and 5 m in top width. Landsat imagery shows that cropland area peaked in 1984–1986 and increased erosion rates in the 1980s and 1990s caused the drainage density and volume to peak in 1994 (Frankl et al. 2013). Since ca. 2000, the large-scale implementation of Soil and Water Conservation (SWC) measures, integrated catchment management, conservation agriculture and indigenous tree regeneration has started to yield positive effects on the vegetation cover and led to the stabilisation of about 25% of the gullies by 2010 (Frankl et al. 2013). Since 1991, farmers have provided labour for SWC in January as a free service for 20 consecutive working days, followed by food for work for the remaining days of the dry season. Most of the degraded landscapes have been restored, with positive impacts over the last two decades on soil fertility, water availability and crop productivity. However, misuse of fertilisers, low survival of tree seedlings and lack of income from exclosures may affect the sustainability of these land restoration measures (Gebremeskel et al. 2018).

B. Rangelands. Biodiversity hotspot, land degradation and climate change: pasture intensification in the Cerrados of Brazil

Cerrados are a tropical savannah ecoregion in Brazil corresponding to a biodiversity hot spot with less than 2% of its region protected in national parks and conservation areas (Cava et al. 2018). Extensive cattle ranching (limited mechanisation, low use of fertiliser and seed inputs) has led to pasture expansion, including clearing forests to secure properties rights, occurring mainly over 1950–1975 (Martha et al. 2012). Despite observed productivity gains made over the last three decades (Martha et al. 2012), more than half of the pasture area is degraded to some extent and challenges remain to reverse grassland degradation while accommodating growing demand and simultaneously avoiding the conversion of natural habitats (de Oliveira Silva et al. 2018). The largest share of production is on unfertilised pastures, often sown with perennial forage grasses of African origin, mainly *Brachiaria* spp. (Cardoso et al. 2016). This initial intensification era was partly at the expense of significant uncontrolled deforestation and average animal stocking rates remained well below the potential carrying capacity (Strassburg et al. 2014). Changes in land use are difficult to reverse since pasture abandonment does not lead to the spontaneous restoration of old-growth savannah (Cava et al. 2018); moreover pasture to crop conversion is frequent, supporting close to half of cropland expansion in Mato Grosso state over 2000–2013 (Cohn et al. 2016). Pasture intensification through liming, fertilisation and controlled grazing could increase soil organic carbon and reduce net GHG emission intensity per unit meat product, but only at increased investment cost per unit of area (de Oliveira Silva et al. 2017). Scenarios projecting a decoupling between deforestation and increased pasture intensification, provide the basis for a Nationally Determined Contribution (NDC) of Brazil that is potentially consistent with accommodating an upward trend in livestock production to meet increasing demand (de Oliveira Silva et al. 2018). Deforestation in Brazil has declined significantly between 2004 and 2014 in the national inventory but recent data and analyses suggest that the decrease in deforestation and the resulting GHG emissions reductions have slowed down or even stopped (UNEP 2017).

C. Semi-natural forests. Biodiversity hotspot, land degradation, climate change and food insecurity: restoration and resilience of tropical forests in Indonesia

During the last two decades, forest cover in Indonesia declined by 150,000 km^2 in the period 1990-2000 (Stibig et al. 2014) and approximately 158,000 km^2 in the period 2000–2012 (Hansen et al. 2013a), most of which was converted to agricultural lands (e.g., oil palm, pulpwood plantations). According to recent estimates, deforestation in Indonesia mainly concerns primary forests, including intact and degraded forests, thus leading to biodiversity loss and reduced carbon sequestration potentials (e.g., Margono et al. 2014). For example, Graham et al. (2017) estimated that the following strategies to reduce deforestation and degradation may cost-effectively increase carbon sequestration and reduce carbon emissions in 30 years: reforestation (3.54 Gt CO_2), limiting the expansion of oil palm and timber plantations into forest (3.07 Gt CO_2 and 3.05 Gt CO_2, respectively), reducing illegal logging (2.34 Gt CO_2), and halting illegal forest loss in Protected Areas (1.52 Gt CO_2) at a total cost of 15.7 USD tC^{-1}. The importance of forest mitigation in Indonesia is indicated by the NDC, where between half and two-thirds of the 2030 emission target relative to business-as-usual scenario is from reducing deforestation, forest degradation, peatland drainage and fires (Grassi et al. 2017). Avoiding deforestation and reforestation could have multiple co-benefits by improving biodiversity conservation and employment opportunities, while reducing illegal logging in protected areas. However, these options could also have adverse side-effects if they deprive local communities of access to natural resources (Graham et al. 2017). The adoption of the Roundtable on Sustainable Palm Oil certification in oil palm plantations reduced deforestation rates by approximately 33% in the period 2001–2015 (co-benefits with mitigation), and fire rates much more than for non-certified plantations (Carlson et al. 2018). However, given that large-scale oil palm plantations are one of largest drivers of deforestation in Indonesia, objective information on the baseline trajectory for land clearance for oil palm is needed to further assess commitments, regulations and transparency in plantation development (Gaveau et al. 2016). For adaptation options, the community forestry scheme "Hutan Desa" (Village Forest) in Sumatra and Kalimantan helped to avoid deforestation (co-benefits with mitigation) by between 0.6 and 0.9 ha km^{-2} in Sumatra and 0.6 and 0.8 ha km^{-2} in Kalimantan in the period 2012–2016; Santika et al. 2017), improve local livelihood options, and restore degraded ecosystems (positive side-effects for NCP provision) (e.g., Pohnan et al. 2015). Finally, the establishment of Ecosystem Restoration Concessions in Indonesia (covering more than 5.5 thousand km^2 of forests now, and 16 thousand km^2 allocated for the future) facilitates the planting of commercial timber species (co-benefits with mitigation), while assisting natural regeneration, preserving important habitats and species, and improving local well-being and incomes (positive side-effects for Nature's Contributions to People provision), at relatively lower costs compared with timber concessions (Silalahi et al. 2017).

D. Villages. Land degradation, groundwater overuse, climate change and food insecurity: climate smart villages in India

Indian agriculture, which includes both monsoon-dependent rainfed (58%) and irrigated agriculture, is exposed to climate variability and change. Over the past years, the frequency of droughts, cyclones, and hailstorms has increased, with severe droughts in 8 of 15 years between 2002 and 2017 (Srinivasa Rao et al. 2016; Mujumdar et al. 2017). Such droughts result in large yield declines for major crops like wheat in the Indo-Gangetic plain (Zhang et al. 2017). The development of a submersible pump technology in the 1990s, combined with public policies that provide farmers free electricity for groundwater irrigation, resulted in a dramatic increase in irrigated agriculture (Shah et al. 2012). This shift has led to increased dependence on irrigation from groundwater and induced a groundwater crisis, with large impacts on socio-ecosystems. An increasing number of farmers report bore-well failures either due to excessive pumping of an existing well or a lack of water in new wells. The decrease in the groundwater table level has suppressed the recharge of river beds, turning permanent rivers into ephemeral streams (Srinivasan et al. 2015). Wells have recently been drilled in upland areas, where groundwater irrigation is also increasing (Robert et al. 2017). Additional challenges are declining soil organic matter and fertility under monocultures and rice/wheat systems. Unoccupied land is scarce, meaning that the potential for expanding the area farmed is very limited (Aggarwal et al. 2018). In rural areas, diets are deficient in protein, dietary fibre and iron, and mainly comprised of cereals and pulses grown and/or procured through welfare programs (Vatsala et al. 2017). Cultivators are often indebted and suicide rates are much higher than the national average, especially for those strongly indebted (Merriott 2016). Widespread use of diesel pumps for irrigation, especially for paddies, high use of inorganic fertilisers and crop residue burning lead to high GHG emissions (Aggarwal et al. 2018). The Climate-Smart Village (CSV) approach aims at increasing farm yield, income, input use efficiency (water, nutrients, and energy) and reducing GHG emissions (Aggarwal et al. 2018). Climate-smart agriculture interventions are considered in a broad sense by including practices, technologies, climate information services, insurance, institutions, policies, and finance. Options differ based on the CSV site, its agro-ecological characteristics, level of development, and the capacity and interest of farmers and the local government (Aggarwal et al. 2018). Selected interventions included crop diversification, conservation agriculture (minimum tillage, residue retention, laser levelling), improved varieties, weather-based insurance, agro-advisory services, precision agriculture and agroforestry. Farmers' cooperatives were established to hire farm machinery, secure government credit for inputs, and share experiences and knowledge. Tillage practices and residue incorporation increased rice–wheat yields by 5–37%, increased income by 28–40%, reduced GHG emissions by 16–25%, and increased water-use efficiency by 30% (Jat et al., 2014). The resulting portfolio of options proposed by the CSV approach has been integrated with the agricultural development strategy of some states like Haryana.

E. Dense settlements. Climate change and food: green infrastructures

Extreme heat events have led to particularly high rates of mortality and morbidity in cities as urban populations are pushed beyond their adaptive capacities, leading to an increase in mortality rates of 30–130% in major cities in developed countries (Norton et al. 2015). Increased mortality and morbidity from extreme heat events are exacerbated in urban populations by the urban heat island effect (Gabriel and Endlicher 2011; Schatz and Kucharik 2015), which can be limited by developing green infrastructure in cities. Urban green infrastructure includes public and private green spaces, including remnant native vegetation, parks, private gardens, golf courses, street trees, urban farming and more engineered options such as green roofs, green walls, biofilters and raingardens (Norton et al. 2015). Increasing the amount of vegetation, or green infrastructure, in a city is one way to help reduce urban air temperature maxima and variation. Increasing vegetation by 10% in Melbourne, Australia was estimated to reduce daytime urban surface temperatures by approximately 1°C during extreme heat events (Coutts and Harris 2013). Urban farming (a type of urban green infrastructure) is largely driven by the desire to reconnect food production and consumption (Whittinghill and Rowe 2012; see Chapter 5). Even though urban farming can only meet a very small share of the overall urban food demand, it provides fresh and local food, especially perishable fruits and crops that are usually shipped from far and sold at high prices (Thomaier et al. 2015). Food-producing urban gardens and farms are often started by grassroots initiatives (Ercilla-Montserrat et al. 2019) that occupy vacant urban spaces. In recent years, a growing number of urban farming projects (termed Zero-Acreage farming, or Z-farming, Thomaier et al. 2015) were established in and on existing buildings, using rooftop spaces or abandoned buildings through contracts between food businesses and building owners. Almost all Z-farms are located in cities with more than 150,000 inhabitants, with a majority in North American cities such as New York City, Chicago and Toronto (Thomaier et al. 2015). They depend on the availability of vacant buildings and roof tops thereby competing with other uses, such as roof-based solar systems. Urban farming, however, has potentially high levels of soil pollution and air pollutants, which may lead to crop contamination and health risks. These adverse effects could be reduced on rooftops (Harada et al. 2019) or in controlled environments.

6.1.4 Challenges represented in future scenarios

In this section, the evolution of several challenges (climate change, mitigation, adaptation, desertification, land degradation, food insecurity, biodiversity and water) in the future are assessed, focusing on global analyses. The effect of response options on these land challenges in the future is discussed in Section 6.4.4. Where possible, studies quantifying these challenges in the Shared Socio-economic Pathways (SSPs) (Chapter 1; Cross-Chapter Box 1: Scenarios, Chapter 1; Cross-Chapter Box 9: Illustrative Climate and Land Pathways, in this chapter; O'Neill et al. 2014), as these studies can be used to assess which future scenarios could experience multiple challenges in the future.

Climate change: Absent any additional efforts to mitigate, global mean temperature rise is expected to increase by anywhere from 2°C to 7.8°C in 2100 relative to the 1850-1900 reference period (Clarke et al. 2014a; Chapter 2). The level of warming varies depending on the climate model (Collins et al. 2013), uncertainties in the Earth system (Clarke et al. 2014), and socioeconomic/technological assumptions (Clarke et al. 2014a; Riahi et al. 2017) Warming over land is 1.2 to 1.4 times higher than global mean temperature rise; warming in the arctic region is 2.4 to 2.6 times higher than warming in the tropics (Collins et al. 2013). Increases in global mean temperature are accompanied by increases in global precipitation; however, the effect varies across regions with some regions projected to see increases in precipitation and others to see decreases (Collins et al. 2013; Chapter 2). Additionally, climate change also has implications for extreme events (e.g., drought, heat waves, etc.), freshwater availability, and other aspects of the terrestrial system (Chapter 2).

Mitigation: Challenges to mitigation depend on the underlying emissions and "mitigative capacity", including technology availability, policy institutions, and financial resources (O'Neill et al. 2014b). Challenges to mitigation are high in the SSP3 and SSP5, medium in SSP2, and low in SSP1 and SSP4 (O'Neill et al. 2014b, 2017; Riahi et al. 2017a).

Adaptation: Challenges to adaptation depend on climate risk and adaptive capacity, including technology availability, effectiveness of institutions, and financial resources (O'Neill et al. 2014b). Challenges to adaptation are high in the SSP3 and SSP4, medium in SSP2, and low in SSP1 and SSP5 (O'Neill et al. 2014b, 2017; Riahi et al. 2017a).

Desertification: The combination of climate and land use changes can lead to decreases in soil cover in drylands (Chapter 3). Population living in drylands is expected to increase by 43% in the SSP2-Baseline, due to both population increases and an expansion of dryland area (UNCCD 2017).

Land degradation: Future changes in land use and climate have implications for land degradation, including impacts on soil erosion, vegetation, fire, and coastal erosion (Chapter 4; Scholes et al. 2018). For example, soil organic carbon is expected to decline by 99 $GtCO_2e$ in 2050 in an SSP2-Baseline scenario, due to both land management and expansion in agricultural area (Brink et al. 2018).

Food insecurity: Food insecurity in future scenarios varies significantly, depending on socio-economic development and study. For example, the population at risk of hunger ranges from 0 to 800 million in 2050 (Hasegawa et al. 2015a; Ringler et al. 2016; Fujimori et al. 2018b; Hasegawa et al. 2018; Fujimori et al. 2018a; Baldos and Hertel 2015) and 0–600 million in 2100 (Hasegawa et al. 2015a). Food prices in 2100 in non-mitigation scenarios range from 0.9 to about 2 times their 2005 values (Hasegawa et al. 2015a; Calvin et al. 2014a; Popp et al. 2017). Food insecurity depends on both income and food prices (Fujimori et al. 2018b). Higher income (e.g., SSP1, SSP5), higher yields (e.g., SSP1, SSP5), and less meat intensive diets (e.g., SSP1) tend to result in reduced food insecurity (Hasegawa et al. 2018; Fujimori et al. 2018b).

Biodiversity: Future species extinction rates vary from modest declines to 100-fold increases from 20th century rates, depending on the species (e.g., plants, vertebrates, invertebrates, birds, fish, corals), the degree of land-use change, the level of climate change, and assumptions about migration (Pereira et al.,2010). Mean species abundance (MSA) is also estimated to decline in the future by 10–20% in 2050 (Vuuren et al., 2015; Pereira et al. 2010). Scenarios with greater cropland expansion lead to larger declines in MSA (UNCCD 2017) and species richness (Newbold et al., 2015).

Water stress: Changes in both water supply (due to climate change) and water demand (due to socioeconomic development) in the future have implications for water stress. Water withdrawals for irrigation increase from about 2500 $km^3 yr^{-1}$ in 2005 to between 2900 and 9000 $km^3 yr^{-1}$ at the end of the century (Chaturvedi et al. 2013; Kim et al. 2016; Bonsch et al.,2015; Wada and Bierkens 2014; Graham et al. 2018; Hejazi et al. 2014); total water withdrawals at the end of the century range from 5000 to 13000 $km^3 yr^{-1}$ (Wada and Bierkens 2014a; Hejazi et al. 2014a; Graham et al. 2018; Kim et al. 2016). The magnitude of change in both irrigation and total water withdrawals depend on population, income, and technology (Hejazi et al. 2014a; Graham et al. 2018a). The combined effect of changes in water supply and water demand will lead to an increase of between 1 and 6 billion people living in water stressed areas (Schlosser et al. 2014; Hanasaki et al. 2013a; Hejazi et al. 2014c). Changes in water quality are not assessed here but could be important (Liu et al. 2017).

Scenarios with Multiple Challenges: Table 6.2 summarises the challenges across the five SSP Baseline scenarios.

Table 6.2: Assessment of future challenges to climate change, mitigation, adaptation, desertification, land degradation, food insecurity, water stress, and biodiversity in the SSP Baseline scenarios

SSP	Summary of Challenges
SSP1	The SSP1 (van Vuuren et al. 2017b) has low challenges to mitigation and adaptation. The resulting Baseline scenario includes: • Continued, but moderate, *climate change*: global mean temperature increases by 3 to 3.5°C in

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	2100 (Riahi et al. 2017a; Huppmann et al. 2018), • Low levels of *food insecurity*: malnourishment is eliminated by 2050 (Hasegawa et al. 2015b), • Declines in *biodiversity*: biodiversity loss increases from 34% in 2010 to 38% in 2100 (UNCCD 2017), and • High *water stress*: global water withdrawals decline slightly from the baseline in 2071-2100, but ~2.6 billion people live in water stressed areas (Hanasaki et al. 2013b). Additionally, this scenario is likely to have lower challenges related to desertification, land degradation, and biodiversity loss than the SSP2 as it has lower population, lower land use change and lower climate change (Riahi et al. 2017a).
SSP2	The SSP2 (Fricko et al. 2017) is a scenario with medium challenges to mitigation and high challenges to adaptation. The resulting Baseline scenario includes: • Continued *climate change*: global mean temperature increases by 3.8 to 4.3°C in 2100 (Fricko et al. 2017; Riahi et al. 2017a; Huppmann et al. 2018), • Increased challenges related to *desertification*: the population living in drylands is expected to increase by 43% in 2050 (UNCCD 2017), • Increased *land degradation*: soil organic carbon is expected to decline by 99 $GtCO_2e$ in 2050 (Brink et al. 2018), • Low levels of *food insecurity*: malnourishment is eliminated by 2100 (Hasegawa et al. 2015b), • Declines in *biodiversity*: biodiversity loss increases from 34% in 2010 to 43% in 2100 (UNCCD 2017), and • High *water stress*: global water withdrawals nearly doubles from the baseline in 2071-2100, with ~4 billion people living in water stressed areas (Hanasaki et al. 2013).
SSP3	The SSP3 (Fujimori et al.,2017) is a scenario with high challenges to mitigation and high challenges to adaptation. The resulting Baseline scenario includes: • Continued *climate change*: global mean temperature increases by 4 to 4.8°C in 2100 (Riahi et al. 2017a; Huppmann et al. 2018), • High levels of *food insecurity*: about 600 million malnourished in 2100 (Hasegawa et al. 2015b), • Declines in *biodiversity*: biodiversity loss increases from 34% in 2010 to 46% in 2100 (UNCCD 2017), and • High *water stress*: global water withdrawals more than double from the baseline in 2071-2100, with ~5.5 billion people living in water stressed areas (Hanasaki et al. 2013). Additionally, this scenario is likely to have higher challenges to desertification, land degradation, and biodiversity loss than the SSP2 as it has higher population, higher land use change and higher climate change (Riahi et al. 2017a).
SSP4	The SSP4 (Calvin et al. 2017a) has high challenges to adaptation but low challenges to mitigation. The resulting Baseline scenario includes: • Continued *climate change*: global mean temperature increases by 3.4 to 3.8°C in 2100 (Calvin et al. 2017b; Riahi et al. 2017a; Huppmann et al. 2018), • High levels of *food insecurity*: about 400 million malnourished in 2100 (Hasegawa et al. 2015b), and • High *water stress*: about 3.5 billion people live in water stressed areas in 2100 (Hanasaki et al. 2013). Additionally, this scenario is likely to have similar effects on biodiversity loss as the SSP2 as it has similar land use change and similar climate change (Riahi et al. 2017a).
SSP5	The SSP5 (Kriegler et al. 2017) has high challenges to mitigation but low challenges to adaptation. The resulting Baseline scenario includes: • Continued *climate change*: global mean temperature increases by 4.6 to 5.4°C in 2100 (Kriegler et al. 2017; Riahi et al. 2017a; Huppmann et al. 2018), • Low levels of *food insecurity*: malnourishment is eliminated by 2050 (Hasegawa et al. 2015b), and • Increased water use and water scarcity: global water withdrawals increase by ~80% in 2071-

	2100 with nearly 50% of the population living in water stressed areas (Hanasaki et al. 2013b). Additionally, this scenario is likely to have higher effects on biodiversity loss as the SSP2 as it has similar land use change and higher climate change (Riahi et al. 2017a).

6.2 Response options, co-benefits and adverse side-effects across the land challenges

This section describes the integrated response options available to address the land challenges of climate change mitigation, climate change adaptation, desertification, land degradation and food security. These can be categorised into options that rely on a) land management, b) value chain management and c) risk management (Figure 6.5). The land management integrated response options can be grouped according to those that are applied in agriculture, in forests, on soils, in other/all ecosystems and those that are applied specifically for carbon dioxide removal (CDR). The value chain management integrated response options can be categorised as those based demand management and supply management. The risk management options are grouped together (Figure 6.5).

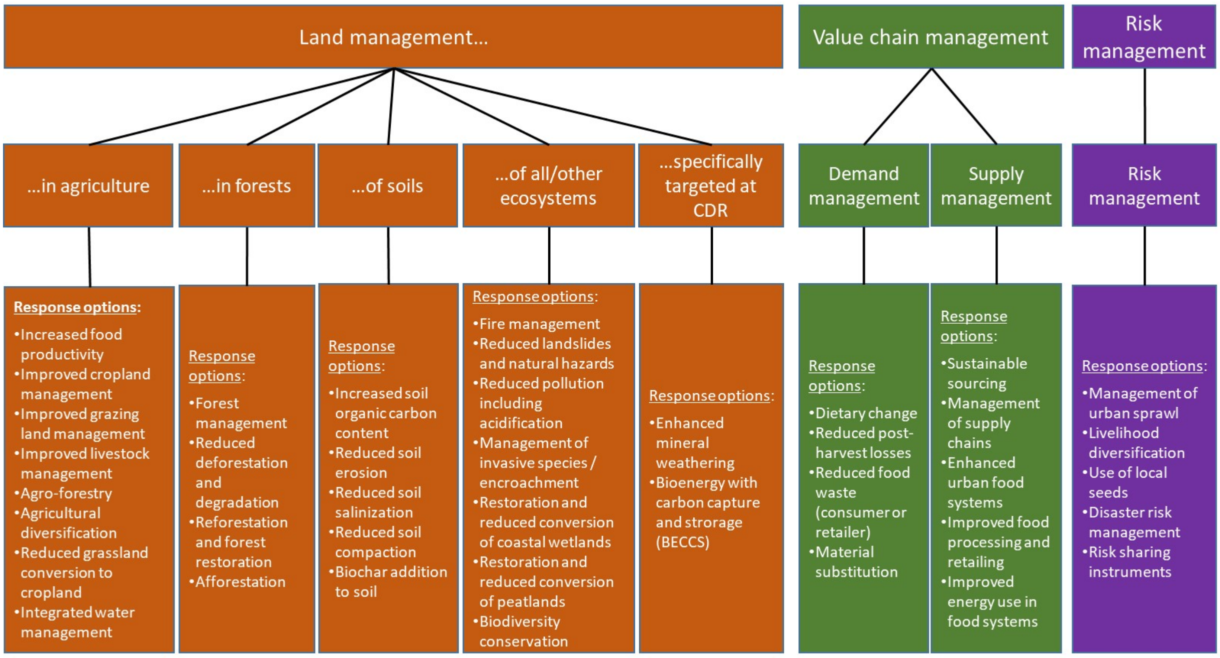

Figure 6.5 Broad categorisation of response options categorised into three main classes and eight sub-classes.

Note that the integrated response options are not mutually exclusive (e.g. cropland management might also increase soil organic matter stocks), and a number of the integrated response options are comprised of a number of practices (e.g., improved cropland management is a collection of practices consisting of a) management of the crop: including high input carbon practices, e. g., improved crop varieties, crop rotation, use of cover crops, perennial cropping systems, agricultural biotechnology; b) nutrient management: including optimised fertiliser application rate, fertiliser type [organic and mineral], timing, precision application, inhibitors; c) reduced tillage intensity and residue retention; d) improved water management: including drainage of waterlogged mineral soils and irrigation of crops in arid / semi-arid conditions; and e) improved rice management: including water management such as mid-season drainage and improved fertilisation and residue management in paddy rice systems).

In this section we deal only with integrated response options, not the policies that are currently / could be implemented to enable their application; that is the subject of Chapter 7. Also note that enabling conditions such as indigenous and local knowledge, gender issues, governance etc. are not categorised as integrated response options (see Section 6.1.2). Some suggested methods to address land challenges are better described as *overarching frameworks* than as integrated response options. For example, *climate smart agriculture* is a collection of integrated response options aimed at delivering mitigation and adaptation in agriculture, including improved cropland management, grazing land management and livestock management. Table 6.3 shows how a number of overarching frameworks are comprised of a range of integrated response options.

Similarly, policy goals, such as *land degradation neutrality* (discussed further in Chapter 7), are not considered as integrated response options. For this reason, *land degradation neutrality*, and overarching frameworks, such as those described in Table 6.3 do not appear as response options in the following sections, but the component integrated response options that contribute to these policy goals or over-arching frameworks are addressed in detail.

Table 6.3 Examples of overarching frameworks that consist of a range of response options, showing how various response options contribute to the overarching frameworks

Framework (definition used)	Nature based solutions (IUCN)	Agro-ecology (FAO)	Climate smart agriculture (FAO)	Ecosystem based adaptation (CBD)	Conservation agriculture (FAO)	Community based adaptation (IIED)	Integrated landscape management including integrated coastal zone management (FAO)	Precision agriculture (FAO)	Sustainable forest management (UN)	Sustainable intensification (Cross Chapter Box 5; Chapter 5)	Organic agriculture (FAO)
<u>Response options based on land management</u>											
Increased food productivity			x		x		x	x		x	
Improved cropland management		x	x		x	x	x	x		x	X
Improved grazing land management		x	x	x		x	x			x	X
Improved livestock management		x	x			x	x			x	X
Agroforestry		x	x	x		x	x			x	X
Agricultural diversification		x	x				x			x	X
Reduced grassland conversion to cropland		x		x		x	x				
Integrated water management	x	x	x	x	x	x	x	x		x	X
Improved forest management	x			x		x	x		x		
Reduced deforestation and degradation		x		x		x	x				
Reforestation and forest restoration	x	x		x		x	x		x		
Afforestation				x		x	x				
Increased soil organic carbon content		x	x	x	x		x			x	X

Reduced soil erosion		x	x	x	x		x			x	X
Reduced soil salinisation		x	x	x	x		x	x		x	X
Reduced soil compaction		x	x	x	x		x			x	X
Biochar addition to soil		x	x								
Fire management		x	x	x		x	x		x		
Reduced landslides and natural hazards		x	x	x		x	x				
Reduced pollution including acidification							x	x		x	X
Management of invasive species / encroachment	x	x		x		x	x		x		X
Restoration and reduced conversion of coastal wetlands		x		x		x	x				
Restoration and reduced conversion of peatlands		x	x	x		x	x				
Biodiversity conservation	x	x	x	x	x	x	x		x	x	
Enhanced weathering of minerals											
Bioenergy and BECCS							x				
<u>Response options based on value chain management</u>											
Dietary change		x									x
Reduced post-harvest losses		x	x			x		x			x
Reduced food waste (consumer or retailer)		x									
Material substitution											
Sustainable sourcing		x	x			x	x				x
Management of supply chains		x	x								
Enhanced urban food systems		x	x			x	x	x		x	x
Improved food processing and retailing		x									
Improved energy use in food systems		x	x		x			x		x	
<u>Response options based on risk management</u>											
Management of urban sprawl				x		x	x				
Livelihood diversification		x	x	x		x	x	x			
Use of local seeds	x	x	x	x		x	x				
Disaster risk management	x			x		x	x				x
Risk sharing instruments										x	

The SR15 considered a range of response options (from a mitigation / adaptation perspective only). Table 6.4 shows how the SR15 options map on to the response options considered in this report (SRCCL). Note that this report excludes most of the energy-related options from SR15, as well as green infrastructure and sustainable aquaculture.

Table 6.4 Mapping of response options considered in this report (SRCCL) and SR15

SRCCL Response Option or Options	SR15 Response Option or Options
Afforestation	Afforestation
Reforestation and forest restoration	Reforestation and reduced land degradation and forest restoration

Agricultural diversification	Mixed crop-livestock systems
Agroforestry	Agroforestry and silviculture
Biochar addition to soil	Biochar
Biodiversity conservation	Biodiversity conservation
Bioenergy and BECCS	Biomass use for energy production with carbon capture and sequestration (BECCS) (through combustion, gasification, or fermentation)
Dietary change	Dietary changes, reducing meat consumption
Disaster risk management	Climate services
	Community-based adaptation
Enhanced urban food systems	Urban and peri-urban agriculture and forestry
Enhanced weathering of minerals	Mineralisation of atmospheric CO2 through enhanced weathering of rocks
Fire management	Fire management and (ecological) pest control
Improved forest management	Forest management
Improved cropland management	Methane reductions in rice paddies
Improved cropland management	Nitrogen pollution reductions, e.g., by fertiliser reduction, increasing nitrogen fertiliser efficiency, sustainable fertilisers
	Precision agriculture
	Conservation agriculture
Improved food processing and retailing	
Improved grazing land management	Livestock and grazing management, for example, methane and ammonia reductions in ruminants through feeding management or feed additives, or manure management for local biogas production to replace traditional biomass use
Improved livestock management	
	Manure management
Increased energy efficiency in food systems	
Increased food productivity	Increasing agricultural productivity
Increased soil organic carbon content	Changing agricultural practices enhancing soil carbon
	Soil carbon enhancement, enhancing carbon sequestration in biota and soils, e.g. with plants with high carbon sequestration potential (also AFOLU measure)
Integrated water management	Irrigation efficiency
Livelihood diversification	
Management of invasive species / encroachment	
Management of supply chains	
Management of urban sprawl	Urban ecosystem services
	climate resilient land use
Material substitution	Material substitution of fossil CO_2 with bio-CO_2 in industrial application (e.g. the beverage industry)
	Carbon Capture and Usage – CCU; bioplastics (bio-based materials replacing fossil fuel uses as feedstock in the production of chemicals and polymers), carbon fibre
Reduced soil erosion	
Reduced soil compaction	
Reduced deforestation	Reduced deforestation, forest protection, avoided forest conversion
Reduced food waste (consumer or retailer)	Reduction of food waste (incl. reuse of food processing waste for fodder)
Reduced grassland conversion to cropland	
Reduced landslides and natural hazards	
Reduced pollution including acidification	Reduced air pollution
Reduced post-harvest losses	

Reduced soil salinisation	
Restoration and reduced conversion of coastal wetlands	Managing coastal stress Restoration of wetlands (e.g., coastal and peat-land restoration, blue carbon) and wetlands management
Restoration and reduced conversion of peatlands	
Risk sharing instruments	Risk sharing
Sustainable sourcing	
Use of local seeds	

Before providing the quantitative assessment of the impacts of each response option in addressing mitigation, adaptation, desertification, land degradation and food security in section 6.3, the integrated response options are descried in section 6.2.1 and any context specificities in the effects are noted.

6.2.1 Integrated response options based on land management

6.2.1.1 Integrated response options based on land management in agriculture

Integrated response options based on land management in agriculture are described in Table 6.5, which also notes any context specificities in the effects of the response options and provides the evidence base.

6.2.1.2 Integrated response options based on land management in forests

Integrated response options based on land management in forests are described in Table 6.6, which also notes any context specificities in the effects of the response options and provides the evidence base.

6.2.1.3 Integrated response options based on land management of soils

Integrated response options based on land management of soils are described in Table 6.7, which also notes any context specificities in the effects of the response options and provides the evidence base.

6.2.1.4 Integrated response options based on land management of all/other ecosystems

Integrated response options based on land management in all/other ecosystems are described in Table 6.8, which also notes any context specificities in the effects of the response options and provides the evidence base.

6.2.1.5 Integrated response options based on land management specifically for carbon dioxide removal (CDR)

Integrated response options based on land management specifically for CDR are described in Table 6.9, which also notes any context specificities in the effects of the response options and provides the evidence base.

Table 6.5 Integrated response options based on land management in agriculture

Integrated response option	Description	Context and caveats	Supporting evidence
Increased food productivity	Increased food productivity arises when the output of food commodities increases per unit of input, e.g. per unit of land or water. It can be realised through many other interventions such as improved cropland, grazing land and livestock management.	Many interventions to increase food production, particularly those predicated on very large inputs of agro-chemicals, have a wide range of negative externalities leading to the proposal of sustainable intensification as a mechanism to deliver future increases in productivity that avoid these adverse outcomes. Intensification through additional input of N fertiliser, for example, would result in negative impacts on climate, soil, water and air pollution. Similarly, if implemented in a way that over-exploits the land significant negative impacts would occur, but if achieved through sustainable intensification, and used to spare land, it could reduce the pressure on land.	Cross-Chapter Box 6 on Agricultural Intensification, Chapter 5; Chapter 3; Burney et al. 2010; Foley et al. 2011; Garnett et al. 2013; Godfray et al. 2010; Lal 2016; Lamb et al. 2016; Lobell et al 2008.; Shcherbak et al. 2014; Smith et al. 2013; Tilman et al. 2014; Scholes et al. 2018; Balmford et al. 2018
Improved cropland management	Improved cropland management is a collection of practices consisting of a) *management of the crop*: including high input carbon practices, for example, improved crop varieties, crop rotation, use of cover crops, perennial cropping systems, integrated production systems, crop diversification, agricultural biotechnology, b) *nutrient management*: including optimised fertiliser application rate, fertiliser type (organic manures, compost and mineral), timing, precision application, nitrification inhibitors, c) *reduced tillage intensity and residue retention*, d) *improved water management*: including drainage of waterlogged mineral soils and irrigation of crops in arid / semi-arid conditions, e) *improved rice management*: including water management such as mid-season drainage and improved fertilisation and residue management in paddy rice systems, and f) *biochar application*.	Improved cropland management can reduce greenhouse gas emissions and create soil carbon sinks, though if poorly implemented, it could increase N_2O and CH_4 emissions from N fertilisers, crop residues and organic amendments. It can improve resilience of food crop production systems to climate change and can be used to tackle desertification and land degradation by improving sustainable land management. It can also contribute to food security by closing crop yield gaps to increase food productivity.	Chapter 4; Chapter 3; Chapter 2; Chapter 5; Bryan et al. 2009; Chen et al. 2010; Labrière et al. 2015; Lal 2011; Poeplau and Don 2015; Porter et al. 2014a; Smith et al. 2014b; Smith 2008; Tilman et al. 2011
Improved grazing land	Improved grazing land management is a collection of practices consisting of a) *management of vegetation*: including improved grass varieties / sward composition, deep rooting grasses,	Improved grazing land management can increase soil carbon sinks, reduce greenhouse gas emissions, improve the resilience of grazing lands to future	Chapter 2; Chapter 3; Chapter 4; Chapter 5; Section 6.3; Archer et al. 2011; Briske et al. 2015;

management	increased productivity, and nutrient management, b) *animal management*: including appropriate stocking densities fit to carrying capacity, fodder banks, and fodder diversification, and c) *fire management*: improved use of fire for sustainable grassland management, including fire prevention and improved prescribed burning (see also fire management as a separate response option; Table 6.8).	climate change, help reduce desertification and land degradation by optimising stocking density and reducing overgrazing, and can enhance food security through improved productivity.	Conant et al. 2017; Herrero et al. 2016; Porter et al. 2014a; Schwilch et al. 2014; Smith et al. 2014b; Tighe et al. 2012
Improved livestock management	Improved livestock management is a collection of practices consisting of a) *improved feed and dietary additives* (e.g., bioactive compounds, fats), used to increase productivity and reduce emissions from enteric fermentation; b) *breeding* (e.g., breeds with higher productivity or reduced emissions from enteric fermentation), c) *herd management,* including decreasing neo-natal mortality, improving sanitary conditions, animal health and herd renewal, and diversifying animal species, d) *emerging technologies* (of which some are not legally authorised in several countries) such as propionate enhancers, nitrate and sulphate supplements, archaea inhibitors and archaeal vaccines, methanotrophs, acetogens, defaunation of the rumen, bacteriophages and probiotics, ionophores / antibiotics; and e) *improved manure management,* including manipulation of bedding and storage conditions, anaerobic digesters; biofilters, dietary change and additives, soil-applied and animal-fed nitrification inhibitors, urease inhibitors, fertiliser type, rate and timing, manipulation of manure application practices, and grazing management.	Improved livestock management can reduce greenhouse gas emissions, particularly from enteric methane and manure management. It can improve the resilience of livestock production systems to climate change by breeding better adapted livestock. It can help with desertification and land degradation, e.g. through use of more efficient and adapted breeds to allow reduced stocking densities. Improved livestock sector productivity can also increase food production.	Chapter 2; Chapter 3; Chapter 4; Chapter 5; Archer et al. 2011; Herrero et al. 2016; Miao et al. 2015; Porter et al. 2014a; Rojas-Downing et al. 2017; Smith et al. 2008, 2014b; Squires et al. 2005; Tighe et al. 2012
Agroforestry	Agroforestry involves the deliberate planting of trees in croplands and silvo-pastoral systems.	Agroforestry sequesters carbon in vegetation and soils. The use of leguminous trees can enhance biological N fixation and resilience to climate change. Soil improvement and the provision of perennial vegetation can help to address desertification and land degradation. Agroforestry can increase agricultural productivity, with benefits for food security. Additionally, agroforestry can enable payments to farmers for ecosystem services and reduce vulnerability to climate shocks.	Antwi-Agyei et al. 2014; Mbow et al. 2014a; Mutuo et al. 2005; Rosenstock et al. 2014; Sain et al. 2017; Sida et al. 2018; Vignola et al. 2015; Yirdaw et al. 2017 Benjamin et. al. 2018; Guo et al. 2018; Herder et al. 2017; Mosquera-Losada et al. 2018; Nair et al. 2014; Ram et al. 2017; Santiago-Freijanes et. al. 2018;

Agricultural diversification	Agricultural diversification includes a set of agricultural practices and products obtained in the field that aim to improve the resilience of farmers to climate variability and climate change and to economic risks posed by fluctuating market forces. In general, the agricultural system is shifted from one based on low-value agricultural commodities to one that is more diverse, composed of a basket of higher value-added products.	Agricultural diversification is targeted at adaptation but could also deliver a small carbon sink, depending on how it is implemented. It could reduce pressure on land, benefiting desertification, land degradation, food security and household income. However, the potential to achieve household food security is influenced by the market orientation of a household, livestock ownership, non-agricultural employment opportunities, and available land resources.	Birthal et al. 2015; Campbell et al. 2014; Cohn et al. 2017; Lambin and Meyfroidt 2011; Lipper et al. 2014; Massawe et al. 2016; Pellegrini and Tasciotti 2014; Waha et al. 2018
Reduced grassland conversion to cropland	Grasslands can be converted to croplands by ploughing of grassland and seeding with crops. Since croplands have a lower soil carbon content than grasslands and are also more prone to erosion than grasslands, reducing conversion of grassland to croplands will prevent soil carbon losses by oxidation and soil loss through erosion. These processes can be reduced if the rate of grassland conversion to cropland is reduced.	Stabilising soils by retaining grass cover also improves resilience, benefiting adaptation, desertification and land degradation. Since conversion of grassland to cropland usually occurs to remedy food security challenges, food security could be adversely affected, since more land is required to produce human food from livestock products on grassland than from crops on cropland.	Chapter 3; Chapter 4; Chapter 5; Clark and Tilman 2017; Lal 2001a; de Ruiter et al. 2017; Poore & Nemecek, 2018
Integrated water management	Integrated water management is the process of creating holistic strategies to promote integrated, efficient, equitable and sustainable use of water for agroecosystems. It includes a collection of practices including water-use efficiency and irrigation in arid/semi-arid areas, improvement of soil health through increases in soil organic matter content, and improved cropland management, agroforestry and conservation agriculture. Increasing water availability, and reliability of water for agricultural production, can be achieved by using different techniques of water harvesting, storage, and its judicious utilisation through farm ponds, dams, and community tanks in rainfed agriculture areas can benefit adaptation.	These practices can reduce aquifer and surface water depletion, and prevent over extraction, and the management of climate risks. Many technical innovations, e.g., precision water management, can have benefits for both adaptation and mitigation, although trade-offs are possible. Maintaining the same level of yield through use of site-specific water management-based approach could have benefits for both food security and mitigation.	Chapter 3; Chapter 4; Chapter 5; Brindha and Pavelic 2016; Jat et al. 2016; Jiang 2015; Keesstra et al. 2018; Liu et al. 2017; Nejad 2013; Rao et al. 2017; Shaw et al. 2014; Sapkota et al. 2017; Scott et al. 2011; Waldron et al. 2017

Table 6.6 Integrated response options based on land management in forests

Integrated response option	Description	Context and caveats	Supporting evidence
Improved forest management	Improved forest management refers to management interventions in forests for the purpose of climate change mitigation. It includes a wide variety of practices affecting the growth of trees and the biomass removed, including improved regeneration (natural or artificial) and a better schedule, intensity and execution of operations (thinning, selective logging, final cut; reduced impact logging, etc.). Sustainable forest management is the stewardship and use of forests and forest lands in a way, and at a rate, that maintains their biodiversity, productivity, regeneration capacity, vitality and their potential to fulfil, now and in the future, relevant ecological, economic and social functions, at local, national, and global levels, and that does not cause damage to other ecosystems.	Sustainable forest management can enhance the carbon stock in biomass, dead organic matter, and soil – while providing wood-based products to reduce emissions in other sectors through material and energy substitution. A trade-off exists between different management strategies: higher harvest decreases the carbon in the forest biomass in the short term but increases the carbon in wood products and the potential for substitution effects. Sustainable forest management, also through close-to-nature silvicultural techniques, can potentially offer many co-benefits in terms of climate change mitigation, adaptation, biodiversity conservation, microclimatic regulation, soil erosion protection, coastal area protection and water and flood regulation. Forest management strategies aimed at increasing the biomass stock levels may have adverse side-effects, such as decreasing the stand-level structural complexity, biodiversity and resilience to natural disasters. Forest management also affects albedo and evapotranspiration.	Chapter 2; Chapter 4; D'Amato et al. 2011; Dooley and Kartha 2018a; Ellison et al. 2017; Erb et al. 2017; Grassi et al. 2018; Griscom et al. 2017a; Jantz et al. 2014; Kurz et al. 2016; Locatelli 2011; Luyssaert et al. 2018; Nabuurs et al. 2017; Naudts et al. 2016; Putz et al. 2012; Seidl et al. 2014; Smith et al. 2014a; Smyth et al. 2014; Stanturf et al. 2015; Forest Europe 2016 Pingoud et al. 2018
Reduced deforestation and degradation	Reduced deforestation and forest degradation includes conservation of existing carbon pools in forest vegetation and soil by controlling the drivers of deforestation (i.e., commercial and subsistence agriculture, mining, urban expansion) and forest degradation (i.e., overharvesting including fuelwood collection, poor harvesting practices, overgrazing, pest outbreaks, and extreme wildfires), also through establishing protected areas, improving law enforcement, forest governance and land tenure, supporting community forest management and introducing forest certification.	Reducing deforestation and degradation is a major strategy to reduce global GHG emissions. The combination of reduced GHG emissions and biophysical effects results in a large climate mitigation effect, with benefits also at local level. Reduced deforestation preserves biodiversity and ecosystem services more efficiently and at lower costs than afforestation/reforestation. Efforts to reduce deforestation and forest degradation may have potential adverse side-effects, for example, reducing availability of land for farming, restricting the rights and access of local people to forest resources (e.g. firewood), or increasing the dependence of local people to insecure external funding.	Chapter 2; Alkama and Cescatti 2016; Baccini et al. 2017; Barlow et al. 2016; Bayrak et al. 2016; Caplow et al. 2011; Curtis et al. 2018; Dooley and Kartha 2018; Griscom et al. 2017a; Hansen et al. 2013b; Hosonuma et al. 2012; Houghton et al. 2015; Lewis et al. 2015; Pelletier et al. 2016; Rey Benayas et al. 2009
Reforestation	Reforestation is the conversion to forest of land that has	Reforestation is similar to afforestation with respect to the co-	Chapter 2; Dooley and

and forest restoration	previously contained forests but that has been converted to some other use. Forest restoration refers to practices aimed at regaining ecological integrity in a deforested or degraded forest landscape. As such, it could fall under reforestation if it were re-establishing trees where they have been lost, or under forest management if it were restoring forests where not all trees have been lost. For practical reasons, here forest restoration is treated together with reforestation.	benefits and adverse side-effects among climate change mitigation, adaptation, desertification, land degradation and food security (see row on Afforestation below). Forest restoration can increase terrestrial carbon stocks in deforested or degraded forest landscapes and can offer many co-benefits in terms of increased resilience of forests to climate change, enhanced connectivity between forest areas and conservation of biodiversity hotspots. Forest restoration may threaten livelihoods and local access to land if subsistence agriculture is targeted.	Kartha 2018; Ellison et al. 2017; Locatelli 2011; Locatelli et al. 2015a; Smith et al. 2014b; Stanturf et al. 2015
Afforestation	Afforestation is the conversion to forest of land that historically have not contained forests (see also reforestation).	Afforestation increases terrestrial carbon stocks but can also change the physical properties of land surfaces, such as surface albedo and evapotranspiration with implications for local and global climate. In the tropics, enhanced evapotranspiration cools surface temperatures, reinforcing the climate benefits of CO_2 sequestration in trees. At high latitudes and in areas affected by seasonal snow cover, the decrease in surface albedo after afforestation becomes dominant and causes an annual average warming that counteracts carbon benefits. Net biophysical effects on regional climate from afforestation is seasonal and can reduce the frequency of climate extremes, such as heat waves, improving adaptation to climate change and reducing the vulnerability of people and ecosystems. Afforestation helps to address land degradation and desertification, as forests tend to maintain water quality by reducing runoff, trapping sediments and nutrients, and improving groundwater recharge. However, food security could be hampered since an increase in global forest area can increase food prices through land competition. Other adverse side-effects occur when afforestation is based on non-native species, especially with the risks related to the spread of exotic fast-growing tree species. For example, exotic species can upset the balance of evapotranspiration regimes, with negative impacts on water availability, particularly in dry regions.	Chapter 2; Chapter 3; Chapter 4; Chapter 5; Alkama and Cescatti 2016; Arora and Montenegro 2011; Bonan 2008; Boysen et al. 2017; Brundu and Richardson 2016; Cherubini et al. 2017; Ciais et al. 2013; Ellison et al. 2017; Findell et al. 2017; Idris Medugu et al. 2010; Kongsager et al. 2016; Kreidenweis et al. 2016a; Lejeune et al 2018.; Li et al. 2015; Locatelli et al. 2015a; Perugini et al. 2017; Salvati et al. 2014; Smith et al. 2013, 2014b; Trabucco et al. 2008;

1 **Table 6.7 Integrated response options based on land management of soils**

Integrated response option	**Description**	**Context and caveats**	**Supporting evidence**
Increased soil organic carbon content	Practices that increase soil organic matter content include a) *land use change* to an ecosystem with higher equilibrium soil carbon levels (e.g. from cropland to forest), b) *management of the vegetation*: including high input carbon practices, for example, improved varieties, rotations and cover crops, perennial cropping systems, biotechnology to increase inputs and recalcitrance of below ground carbon, c) *nutrient management and organic material input* to increase carbon returns to the soil: including optimised fertiliser and organic material application rate, type, timing and precision application, d) *reduced tillage intensity and residue retention*, and e) *improved water management*: including irrigation in arid / semi-arid conditions.	Increasing soil carbon stocks removes CO_2 from the atmosphere and increases the water holding capacity of the soil thereby conferring resilience to climate change and enhancing adaptation capacity. It is a key strategy for addressing both desertification and land degradation. There is some evidence that crop yields and yield stability increase by increased organic matter content, though some studies show equivocal impacts. Some practices to increase soil organic matter stocks vary in their efficacy. For example, the impact of no till farming and conservation agriculture on soil carbon stocks is often positive, but can be neutral or even negative, depending on the amount of crop residues returned to the soil. If soil organic carbon stocks were increased by increasing fertiliser inputs to increase productivity, emissions of nitrous oxide from fertiliser use could offset any climate benefits arising from carbon sinks. Similarly, if any yield penalty is incurred from practices aimed at increasing soil organic carbon stocks (e.g. through extensification), emissions could be increased through indirect land use change, and there could also be adverse side-effects on food security.	Bestelmeyer and Briske 2012; Cheesman et al. 2016; Frank et al. 2017; Gao et al. 2018; Keesstra et al 2016.; Lal 2016, 2006; Lambin and Meyfroidt 2011; de Moraes Sá et al. 2017; Palm et al. 2014; Pan et al. 2009; Paustian et al. 2016; Powlson et al. 2014, 2016, Smith et al. 2013, 2016a, 2014b; Soussana et al. 2019a; Steinbach et al 2006.; VandenBygaart 2016; Hijbeek et al., 2017; Schjønning et al., 2018;
Reduced soil erosion	Soil erosion is the removal of soil from the land surface by water, wind or tillage, which occurs worldwide but it is particularly severe in Asia, Latin America and the Caribbean, and the Near East and North Africa. Soil erosion management includes conservation practices (e.g., the use of minimum tillage or zero tillage, crop rotations and cover crops, rational grazing systems), engineering-like practices (e.g., construction of terraces and contour cropping for controlling water erosion), or forest barriers and strip	The fate of eroded soil carbon is uncertain, with some studies indicating a net source of CO_2 to the atmosphere and others suggesting a net sink. Reduced soil erosion has benefits for adaptation as it reduces vulnerability of soils to loss under climate extremes, increasing resilience to climate change. Some management practices implemented to control erosion, such as increasing ground cover, can reduce the vulnerability of soils to degradation / landslides, and prevention of soil erosion is a key measure used to tackle desertification. Because it protects the capacity of land to produce food, it also contributes positively to food security.	Chapter 3; Chen 2017; Derpsch et al. 2010; FAO and ITPS 2015; FAO 2015; Garbrecht et al. 2015; Jacinthe and Lal 2001; de Moraes Sá et al. 2017; Poeplau and Don 2015; Smith et al. 2001; Stallard 1998; Lal and Moldenhauer 1987; Van Oost et al. 2007; Lugato et al. 2016; Smith et al. 2005; Lal 2001a

	cultivation for controlling wind erosion. In eroded soils, the advance of erosion gullies and sand dunes can be limited by increasing plant cover, among other practices.		
Reduced soil salinisation	Soil salinisation is a major process of land degradation that decreases soil fertility and affects agricultural production, aquaculture and forestry. It is a significant component of desertification processes in drylands. Practices to reduce soil salinisation include improvement of water management (e.g., water-use efficiency and irrigation/drainage technology in arid/semi-arid areas, surface and groundwater management), improvement of soil health (through increase in soil organic matter content) and improved cropland, grazing land and livestock management, agroforestry and conservation agriculture.	Techniques to prevent and reverse soil salinisation may have small benefits for mitigation by enhancing carbon sinks. These techniques may benefit adaptation and food security by maintaining existing crop systems and closing yield gaps for rainfed crops. These techniques are central to reducing desertification and land degradation, since soil salinisation is a primary driver of both.	Section 3.5; Chapter 4; Chapter 5; Baumhardt et al. 2015; Dagar et al. 2016a; Datta et al. 2000; DERM 2011; Evans and Sadler 2008; He et al. 2015; D'Odorico et al. 2013; Prathapar 1988; Qadir et al. 2013; Rengasamy 2006; Singh 2009; UNCTAD 2011; Wong et al. 2010
Reduced soil compaction	Reduced soil compaction mainly includes agricultural techniques (e.g. crop rotations, control of livestock density) and control of agricultural traffic.	Techniques to reduce soil compaction have variable impacts on GHG emissions but may benefit adaptation by improving soil climatic resilience. Since soil compaction is a driver of both desertification and land degradation, a reduction of soil compaction could benefit both. It could also help close yield gaps in rainfed crops.	Chamen et al. 2015; Epron et al. 2016; ITPS-FAO 2015; Hamza and Anderson 2005; Soane and van Ouwerkerk 1994; Tullberg et al. 2018
Biochar addition to soil	The use of biochar, a solid product of the pyrolysis process, as a soil amendment increases the water-holding capacity of soil. It may therefore provide better access to water and nutrients for crops and other vegetation types (so can form part of cropland, grazing land and improved forest management).	The use of biochar increases carbon stocks in the soil. It can enhance yields in the tropics (but less so in temperate regions), thereby benefiting both adaptation and food security. Since it can improve soil water holding capacity and nutrient use efficiency, and can ameliorate heavy metal pollution and other impacts, it can benefit desertification and land degradation. The positive impacts could be tempered by additional pressure on land if large quantities of biomass are required as feedstock for biochar production.	Chapter 2; Chapter 3; Chapter 4; Chapter 5; Jeffery et al. 2017; Smith 2016; Sohi 2012; Woolf et al. 2010

Table 6.8 Integrated response options based on land management of all/other ecosystems

Integrated response option	Description	Context and caveats	Supporting evidence
Fire management	Fire management is a land management option aimed at safeguarding life, property and resources through the prevention, detection, control, restriction and suppression of fire in forest and other vegetation. It includes the improved use of fire for sustainable forestry management, including wildfire prevention and prescribed burning. Prescribed burning is used to reduce the risk of large, uncontrollable fires in forest areas, and controlled burning is among the most effective and economic methods of reducing fire danger and stimulating natural reforestation under the forest canopy and after clear felling.	The frequency and severity of large wildfires have increased around the globe in recent decades, which has impacted forest carbon budgets. Fire can cause various greenhouse gas emissions such as CO_2, CH_4, and N_2O, and others such as CO, volatile organic carbon, and smoke aerosols. Fire management can reduce GHG emissions and can reduce haze pollution, which has significant health and economic impacts. Fire management helps to prevent soil erosion and land degradation and is used in rangelands to conserve biodiversity and to enhance forage quality.	Chapter 2; Cross-Chapter Box 3 on fire and climate; Esteves et al. 2012; FAO 2006; Lin et al. 2017; O'Mara 2012; Rulli et al. 2006; Scasta et al. 2016; Seidl et al. 2014; Smith et al. 2014b; Tacconi 2016; Valendik et al. 2011; Westerling et al. 2006; Whitehead et al. 2008; Yong and Peh 2016
Reduced landslides and natural hazards	Landslides are mainly triggered by human activity (e.g. legal and illegal mining, fire, deforestation) in combination with climate. Management of landslides and natural hazards (e.g. floods, storm surges, droughts) is based on vegetation management (e.g. afforestation) and engineering works (e.g. dams, terraces, stabilisation and filling of erosion gullies).	Management of landslides and natural hazards is important for adaptation and is a very important intervention for managing land degradation, since landslides and natural hazards are among the most severe degradation processes. In countries where mountain slopes are planted with food crops, reduced landslides will help deliver benefits for food security. Most deaths caused due to different disasters have occurred in developing countries, in which poverty, poor education and health facilities, and other aspects of development increase exposure, vulnerability and risk.	IPCC AR5 WG2 Chapter 14; Arnáez J et al. 2015; Campbell 2015; ITPS-FAO 2015; Gariano and Guzzetti 2016; Mal et al. 2018
Reduced pollution including acidification	Management of air pollution is connected to climate change by emission sources of air polluting materials and their impacts on climate, human health, and ecosystems, including agriculture. Acid deposition is one of the many consequences of air pollution, harming trees and other vegetation, as well as being a significant driver of land degradation. Practices that reduce acid deposition include prevention of emissions of nitrogen oxides (NO_x) and sulphur dioxide (SO_2), which also reduce GHG emissions and	There are a few potential adverse side effects of reduction in air pollution to carbon sequestration in terrestrial ecosystems, because some forms of air pollutants can enhance crop productivity by increasing diffuse sunlight, compared to direct sunlight. Reactive N deposition could also enhance CO_2 uptake in boreal forests and increase soil carbon pools to some extent. Air pollutants have different impacts on climate depending primarily on the composition, with some aerosols (and clouds seeded by them) increasing the reflection of solar radiation to space leading to net cooling, while others (e.g. black carbon and tropospheric ozone) having a net warming effect. Therefore, control of these different pollutants will have both positive and negative impacts on climate mitigation.	Chapter 2; Anderson et al. 2017; Chum et al. 2013; Carter et al. 2015; Coakley; Maaroufi et al. 2015; Markandya et al. 2018; Melamed and Schmale 2016; Mostofa et al 2016.; Nemet et al. 2010; Ramanathan et al. 2001; Seinfeld and

	other Short-Lived Climate Pollutants (SLCPs). Reductions of SLCPs reduce warming in the near term and the overall rate of warming, which can be crucial for plants that are sensitive to even small increases in temperature. Management of harmful air pollutants such as fine particulate matter (PM2.5) and ozone (O_3) also mitigate the impacts of incomplete fossil fuel combustion and GHG emissions. In addition, management of pollutants such as tropospheric O_3 has beneficial impacts on food production, since O_3 decreases crop production. Control of urban and industrial air pollution would also mitigate the harmful effects of pollution and provide adaptation co-benefits *via* improved human health. Management of pollution contributes to aquatic ecosystem conservation since controlling air pollution, rising atmospheric CO_2 concentrations, acid deposition, and industrial waste will reduce acidification of marine and freshwater ecosystems.		Pandis; Smith et al. 2015b; UNEP 2017; Wild et al. 2012 UNEP and WMO 2011; Xu & Ramanathan, 2017; Xu et al., 2013
Management of invasive species / encroachment	Agriculture and forests can be diverse but often, much of the diversity is non-native. Invasive species in different biomes have been introduced intentionally or unintentionally through export of ornamental plants or animals, and through the promotion of modern agriculture and forestry. Non-native species tend to be more numerous in larger than in smaller human-modified landscapes (e.g. over 50% of species in an urbanised area or extensive agricultural fields can be non-native). Invasive alien species in the United States cause major environmental damage amounting to almost USD120 billion yr^{-1}. There are approximately 50,000 foreign species and the number is increasing. About 42% of the species on the Threatened or Endangered species lists are at risk primarily because of alien-invasive species. Invasive species can be managed	Exotic species are used in forestry where local indigenous forests cannot produce the type, quantity and quality of forest products required. Planted forests of exotic tree species make significant contributions to the economy and provide multiple products and Nature's Contributions to People. In general, exotic species are selected to have higher growth rates than native species and produce more wood per unit of area and time. In 2015, the total area of planted forest with non-native tree species was estimated to around 0.5 Mkm^2. Introduced species were dominant in South America, Oceania and Eastern and Southern Africa, where industrial forestry is dominant. The use of exotic tree species has played an important role in the production of roundwood, fibre, firewood and other forest products. The challenge is to manage existing and future plantation forests of alien trees to maximise current benefits, while minimising present and future risks and negative impacts, and without compromising future benefits. In many countries or regions, non-native trees planted for production or other purposes often lead to sharp conflicts of interest when they become invasive, and to negative impacts on Nature's Contributions to People and nature conservation.	Brundu and Richardson 2016; Cossalter and Pye-Smith 2003; Dresner et al. 2015; Payn et al. 2015; Pimentel et al. 2005; Vilà et al. 2011

	through manual clearance of invasive species, while in some areas, natural enemies of the invasive species are introduced to control them.		
Restoration and reduced conversion of coastal wetlands	Coastal wetland restoration involves restoring degraded / damaged coastal wetlands including mangroves, salt marshes and seagrass ecosystems.	Coastal wetland restoration and avoided coastal wetland impacts have the capacity to increase carbon sinks and can provide benefits by regulating water flow and preventing downstream flooding. Coastal wetlands provide a natural defence against coastal flooding and storm surges by dissipating wave energy, reducing erosion and by helping to stabilise shore sediments. Since large areas of global coastal wetlands are degraded, restoration could provide benefits land degradation. Since some areas of coastal wetlands are used for food production, restoration could displace food production and damage local food supply (Section 6.3.4), though some forms (e.g. mangrove restoration) can improve local fisheries.	Griscom et al. 2017a; Lotze et al. 2006; Munang et al. 2014; Naylor et al. 2000
Restoration and reduced conversion of peatlands	Peatland restoration involves restoring degraded / damaged peatlands which both increases carbon sinks, but also avoids ongoing CO_2 emissions from degraded peatlands, so it both prevents future emissions and creates a sink, as well as protecting biodiversity.	Avoided peat impacts and peatland restoration can provide significant mitigation, though restoration can lead to an increase in methane emissions, particularly in nutrient rich fens. There may also be benefits for climate adaptation by regulating water flow and preventing downstream flooding. Considering that large areas of global peatlands are degraded, peatland restoration is a key tool in addressing land degradation. Since large areas of tropical peatlands and some northern peatlands have been drained and cleared for food production, their restoration could displace food production and damage local food supply, potentially leading to adverse impacts on food security locally, though the global impact would be limited due to the relatively small areas affected.	Griscom et al. 2017a; Jauhiainen et al. 2008; Limpens et al. 2008; Munang et al. 2014
Biodiversity conservation	Biodiversity conservation refers to practices aiming at maintaining components of biological diversity. It includes conservation of ecosystems and natural habitats, maintenance and recovery of viable populations of species in their natural surroundings (*in-situ* conservation) and, in the case of domesticated or cultivated species, in the surroundings where they have developed their distinctive properties outside their natural habitats (*ex-situ* conservation). Examples of biodiversity conservation measures are establishment of protected areas to achieve specific conservation objectives, preservation of biodiversity hotspots, land management to recover natural habitats, interventions to expand	Biodiversity conservation measures interact with the climate system through many complex processes, which can have either positive or negative impacts. For example, establishment of protected areas can increase carbon storage in vegetation and soil, and tree planting to promote species richness and natural habitats can enhance carbon uptake capacity of ecosystems. Management of wild animals can influence climate *via* emissions of GHGs (from anaerobic fermentation of plant materials in the rumen), impacts on vegetation (*via* foraging), changes in fire frequency (as grazers lower grass and vegetation densities as potential fuels), and nutrient cycling and transport (by adding nutrients to soils). Conserving and restoring megafauna in northern regions also prevents thawing of permafrost and reduces woody encroachment, thus avoiding methane emissions and increases in albedo. Defaunation affects carbon storage in tropical forests and savannahs. In the tropics, the loss of mega-faunal frugivores is estimated be responsible for up to 10% reduction in carbon storage of global tropical forests. Frugivore	Bello et al. 2015; Campbell et al. 2008; Cromsigt et al. 2018; Kapos et al. 2008; Osuri et al. 2016; Schmitz et al. 2018a; Secretariat of the Convention on Biological Diversity 2008

	or control selective plant or animal species in productive lands or rangelands (e.g., rewilding).	rewilding programmes in the tropics are seen as carbon sequestration options that can be equally effective as tree planting schemes. Biodiversity conservation measures generally favour adaptation, but can interact with food security, land degradation or desertification. Protected areas for biodiversity reduce the land available for food production, and abundancies in some species like large animals can influence land degradation processes by grazing, trampling and compacting soil surfaces, thereby altering surface temperatures and chemical reactions affecting sediment and carbon retention.	

Table 6.9 Integrated response options based on land management specifically for CDR

Integrated response option	Description	Context and caveats	Supporting evidence
Enhanced weathering of minerals	The enhanced weathering of minerals that naturally absorb CO_2 from the atmosphere has been proposed as a CDR technology with a large mitigation potential. The rocks are ground to increase the surface area and the ground minerals are then applied to the land where they absorb atmospheric CO_2.	Enhanced mineral weathering can remove atmospheric CO_2. Since ground minerals can increase pH, there could be some benefits for efforts to prevent or reverse land degradation where acidification is the driver of degradation. Since increasing soil pH in acidified soils can increase productivity, the same effect could provide some benefit for food security. Minerals used for enhanced weathering need to be mined, and mining has large impacts locally, though the total area mined is likely to be small on the global scale.	Lenton 2010; Schuiling and Krijgsman 2006; Smith et al. 2016a; Taylor et al. 2016a; Beerling et al. 2018
Bioenergy and BECCS	Bioenergy production can mitigate climate change by delivering an energy service, therefore avoiding combustion of fossil energy. It is the most common renewable energy source used today in the world and has a large potential for future deployment (see Cross-Chapter Box 7 on bioenergy in this chapter). BECCS entails the use of bioenergy technologies (e.g. bioelectricity or biofuels) in combination with CO_2 capture and storage (see also Glossary). BECCS simultaneously provides energy and can reduce atmospheric CO_2 concentrations (see Chapter 2 and Cross-Chapter Box 7 on bioenergy in this chapter) for a discussion of potentials and atmospheric	Bioenergy and BECCS can compete for land and water with other uses. Increased use of bioenergy and BECCS can result in large expansion of cropland area, significant deforestation, and increased irrigation water use and water scarcity. Large-scale use of bioenergy can result in increased food prices and can lead to an increase in the population at risk of hunger. As a result of these effects, large-scale bioenergy and BECCS can have negative impacts for food security. Interlinkages of bioenergy and BECCS with climate change adaptation, land degradation, desertification, and biodiversity are highly dependent on local factors such as the type of energy crop, management practice, and previous land use. For example, intensive agricultural practices aiming to achieve high crop yields, as is the case for some bioenergy systems, may have significant effects on soil health, including depletion of soil organic matter, resulting in negative impacts on land degradation and desertification. However, with low inputs of fossil fuels and chemicals, limited irrigation, heat/drought tolerant species, using marginal land, biofuel programs can be beneficial to future adaptation of ecosystems. Planting bioenergy crops, like perennial grasses, on degraded land can increase soil carbon and ecosystem quality (including biodiversity), thereby helping to preserve soil quality, reverse land degradation, prevent desertification processes, and reduce food insecurity. These effects depend on the scale of deployment, the feedstock, the prior land use, and which other response options are included (see Section 6.4.4.2). Large-scale production of	Cross-Chapter Box 7 on Bioenergy in this chapter; IPCC SR15; Chapter 2; Chapter 4; Section 6.4; Chapter 7; Baker et al. 2019a; Calvin et al. 2014c; Chaturvedi et al. 2013; Chum et al. 2011; Clarke et al. 2014a; Correa et al. 2017; Creutzig et al. 2015; Dasgupta et al. 2014; Don et al. 2012; Edelenbosch et al. 2017; Edenhofer et al. 2011; FAO 2011; Favero and Mendelsohn 2014; Fujimori et al. 2018a; Fuss et al. 2016, 2018a; Hejazi et al. 2015a; Kemper 2015; Kline et al. 2017; Lal 2014; Lotze-Campen et al. 2013; Mello et al. 2014b; Muratori et al. 2016; Noble et al. 2014; Obersteiner et al. 2016a; Popp et al. 2011c, 2014a, 2017; Riahi et al. 2017a; Robertson et al. 2017b; Sánchez et al. 2017; Searchinger et al. 2018; Sims et al. 2014; Slade et al. 2014; Smith

	effects); thus, BECCS is considered a CDR technology. While several BECCS demonstration projects exist, it has yet to be deployed at scale. Bioenergy and BECCS are widely-used in many future scenarios as a climate change mitigation option in the energy and transport sector, especially those scenarios aimed at a stabilisation of global climate at 2°C or less above pre-industrial levels.	bioenergy can require significant amounts of land, increasing potential pressures for land conversion and land degradation. Low levels of bioenergy deployment require less land, leading to smaller effects on forest cover and food prices; however, these land requirements could still be substantial. In terms of feedstocks, some feedstocks, grown in some regions, may not need irrigation, and thus would not compete for water with food crops. Additionally, the use of residues or microalgae could limit competition for land and biodiversity loss; however, residues could result in land degradation or decreased soil organic carbon. Whether woody bioenergy results in increased competition for land or not is disputed in the literature, with some studies suggesting reduced competition and others suggesting enhanced. One study noted that this effect changes over time, with complementarity between woody bioenergy and forest carbon sequestration in the near-term, but increased competition for land with afforestation/reforestation in the long-term. Additionally, woody bioenergy could also result in land degradation.	et al. 2016a; Torvanger 2018; van Vuuren et al. 2011, 2015b, 2016; Wise et al. 2015; Tian et al. 2018;

6.2.2 Integrated response options based on value chain management

6.2.2.1 Integrated response options based on value chain management through demand management

Integrated response options based on value chain management through demand management are described in Table 6.10, which also notes any context specificities in the effects of the response options and provides the evidence base.

6.2.2.2 Integrated response options based on value chain management through supply management

Integrated response options based on value chain management through supply management are described in Table 6.11, which also notes any context specificities in the effects of the response options and provides the evidence base.

6.2.3 Integrated response options based on risk management

6.2.3.1 Risk management options

Integrated response options based on risk management described in Table 6.12, which also notes any context specificities in the effects of the response options and provides the evidence base.

1 **Table 6.10 Integrated response options based on value chain management through demand management**

Integrated response option	Description	Context and caveats	Supporting evidence
Dietary change	Sustainable healthy diets represent a range of dietary changes to improve human diets, to make them healthy in terms of the nutrition delivered, and also (economically, environmentally and socially) sustainable. A "contract and converge" model of transition to sustainable healthy diets would involve a reduction in overconsumption (particularly of livestock products) in over-consuming populations, with increased consumption of some food groups in populations where minimum nutritional needs are not met. Such a conversion could result in a decline in undernourishment, as well as reduction in the risk of morbidity and mortality due to over-consumption.	A dietary shift away from meat can reduce greenhouse gas emissions, reduce cropland and pasture requirements, enhance biodiversity protection, and reduce mitigation costs. Additionally, dietary change can both increase potential for other land-based response options and reduce the need for them by freeing land. By decreasing pressure on land, demand reduction through dietary change could also allow for decreased production intensity, which could reduce soil erosion and provide benefits to a range of other environmental indicators such as deforestation and decreased use of fertiliser (N and P), pesticides, water and energy, leading to potential benefits for adaptation, desertification, and land degradation.	Chapter 5; Section 6.4.4.2; Aleksandrowicz et al. 2016a; Bajželj et al. 2014; Bonsch et al. 2016; Erb et al. 2016; Godfray et al. 2010; Haberl et al. 2011; Havlík et al. 2014; Muller et al. 2017a; Smith et al. 2013; Springmann et al. 2018a; Stehfest et al. 2009; Tilman and Clark 2014; Wu et al. 2019
Reduced post-harvest losses	Approximately one-third of the food produced for human consumption is wasted in post-production operations. Most of these losses are due to poor storage management. Post-harvest food losses underlie the food system's failure to equitably enable accessible and affordable food in all countries. Reduced post-harvest food losses can improve food security in developing countries (while food loss in developed countries mostly occurs at the retail/consumer stage). The key drivers for post-harvest waste in developing countries are structural and infrastructure deficiencies. Thus, reducing food waste at the post-harvest stage requires responses that process,	Differences exist between farm food waste reduction technologies between small-scale agricultural systems and large-scale agricultural systems. A suite of options includes farm level storage facilities, trade or exchange processing technologies including food drying, onsite farm processing for value addition, and improved seed systems. For large scale agri-food systems, options include cold chains for preservation, processing for value addition and linkages to value chains that absorb the harvests almost instantly into the supply chain. In addition to the specific options to reduce food loss and waste, there are more systemic possibilities related to food systems. Improving and expanding the 'dry chain' can significantly reduce food losses at the household level. Dry chains are analogous to the	Chapter 5; Ansah et al. 2017; Bajželj et al. 2014; Billen et al. 2018; Bradford et al. 2018; Chaboud and Daviron 2017; Göbel et al. 2015; Gustavsson et al. 2011; Hengsdijk and de Boer 2017; Hodges et al. 2011; Ingram et al. 2016;

	preserve and, where appropriate, redistribute food to where it can be consumed immediately.	cold chain and refers to the 'initial dehydration of durable commodities to levels preventing fungal growth' followed by storage in moisture-proof containers. Regional and local food systems are now being promoted to enable production, distribution, access and affordability of food. Reducing post-harvest losses has the potential to reduce emissions and could simultaneously reduce food costs and increase availability. The perishability and safety of fresh foods are highly susceptible to temperature increase.	Kissinger et al. 2018; Kumar and Kalita 2017; Ritzema et al. 2017; Sheahan and Barrett 2017; Wilhelm et al. 2016
Reduced food waste (consumer or retailer)	Since approximately 9-30% of all food is wasted, reducing food waste can reduce pressure on land (see also reducing post-harvest losses).	Reducing food waste could lead to a reduction in cropland area and GHG emissions, resulting in benefits for mitigation. By decreasing pressure on land, food waste reduction could allow for decreased production intensity, which could reduce soil erosion and provide benefits to a range of other environmental indicators such as deforestation and decreases in use of fertiliser (N and P), pesticides, water and energy, leading to potential benefits for adaptation, desertification, and land degradation.	Alexander et al. 2016; Bajželj et al. 2014; Gustavsson et al. 2011; Kummu et al. 2012a; Muller et al. 2017a; Smith et al. 2013; Vermeulen et al. 2012b
Material substitution	Material substitution involves the use of wood or agricultural biomass (e.g. straw bales) instead of fossil fuel-based materials (e.g. concrete, iron, steel, aluminium) for building, textiles or other applications.	Material substitution reduces carbon emissions both because the biomass sequesters carbon in materials while re-growth of forests can lead to continued sequestration, and because it reduces the demand for fossil fuels, delivering a benefit for mitigation. However, a potential trade-off exists between conserving carbon stocks and using forests for wood products. If the use of material for substitution was large enough to result in increased forest area, then the adverse side-effects for adaptation and food security would be similar to that of reforestation and afforestation. In addition, some studies indicate that wooden buildings, if properly constructed, could reduce fire risk compared to steel, creating a co-benefit for adaptation. The effects of material substitution on land degradation depend on management practice; some forms of logging can lead to increased land degradation. Long-term forest management with carbon storage in long-lived products also results in atmospheric carbon dioxide (CO_2) removal.	Chapter 4; Dugan et al. 2018; Eriksson et al. 2012; Gustavsson et al. 2006; Kauppi et al. 2018; Leskinen et al. 2018; McLaren 2012; Oliver and Morecroft 2014; Ramage et al. 2017; Sathre and O'Connor 2010; Smyth et al. 2014; Kurz et al. 2016; Miner 2010; Iordan et al. 2018

Table 6.11 Integrated response options based on value chain management through supply management

Integrated response option	Description	Context and caveats	Supporting evidence
Sustainable sourcing	Sustainable sourcing includes approaches to ensure that the production of goods is done in a sustainable way, such as through low-impact agriculture, zero-deforestation supply chains, or sustainably harvested forest products. Currently around 8% of global forest area has been certified in some manner, and 25% of global industrial roundwood comes from certified forests. Sustainable sourcing also aims to enabling producers to increase their percentage of the final value of commodities. Adding value to products requires improved innovation, coordination and efficiency in the food supply chain, as well as labelling to ensure consumer demands. As such, sustainable sourcing is an approach that combines both supply and demand-side management. Promoting sustainable and value-added products can reduce the need for compensatory extensification of agricultural areas and is a specific commitment of some sourcing programs (such as forest certification programs). Table 7.3 (Chapter 7) provides examples of the many sustainable sourcing programs now available globally.	Sustainable sourcing is expanding but accounts for only a small fraction of overall food and material production; many staple food crops do not have strong sustainability standards. Sustainable sourcing provides potential benefits for both climate mitigation and climate adaptation by reducing drivers of unsustainable land management, and by diversifying and increasing flexibility in the food system to climate stressors and shocks. Sustainable sourcing can lower expenditures of food processors and retailers by reducing losses. Adding value to products can extend a producer's marketing season and provide unique opportunities to capture niche markets thereby increasing their adaptive capacity to climate change. Sustainable sourcing can also provide significant benefits for food security, while simultaneously creating economic alternatives for the poor. Sustainable sourcing programmes often also have positive impacts on the overall efficiency of the food supply chain and can create closer and more direct links between producers and consumers. In some cases, processing of value-added products could lead to higher emissions or demand of resources in the food system, potentially leading to small adverse impacts on land degradation and desertification challenges.	Chapter 2; Chapter 3; Chapter 5; Section 6.4; Accorsi et al. 2017; Bajželj et al. 2014; Bustamante et al. 2014a; Clark and Tilman 2017; Garnett 2011; Godfray et al. 2010a; Hertel 2015; Ingram et al. 2016a; James and James 2010a; Muller et al. 2017a; Tilman and Clark 2014a; Springer et al. 2015; Tayleur et al. 2017
Management of supply chains	Management of supply chains include a set of polycentric governance processes focused on improving efficiency and sustainability across the supply chain for each product, to reduce climate risk and profitably reduce emissions. Trade-driven food supply chains are becoming increasingly complex and contributing to emissions. Improved management of supply chains can include both: 1) better food transport and increasing the economic value or reduce risks	Successful implementation of supply chain management practices is dependent on organisational capacity, the agility and flexibility of business strategies, the strengthening of public-private policies and effectiveness of supply-chain governance. Existing practices include a) greening supply chains (e.g. utilising products and services with a reduced impact on the environment and human health), b) adoption of specific sustainability instruments among agri-food companies (e.g. eco-	Chapter 5; Barthel and Isendahl 2013; Haggblade et al. 2017; Lewis and Witham 2012; Michelini et al. 2018; Minot 2014; Mundler and Rumpus 2012; Tadasse et al.

	of commodities through production processes (e.g., packaging, processing, cooling, drying, extracting) and 2) improved policies for stability of food supply, as globalised food systems and commodity markets are vulnerable to food price volatility. The 2007-2008 food price shocks negatively affected food security for millions, most severely in Sub-Saharan Africa. Increasing the stability of food supply chains is a key goal to increase food security, given that climate change threatens to lead to more production shocks in the future.	innovation practices), c) adopting emission accounting tools (e.g. carbon and water foot-printing), and d) implementing "demand forecasting" strategies (e.g. changes in consumer preference for 'green' products). In terms of food supply, measures to improve stability in traded markets can include: 1) financial and trade policies, such as reductions on food taxes and import tariffs; 2) shortening food supply chains (SFSCs); 3) increasing food production; 4) designing alternative distribution networks; 5) increasing food market transparency and reducing speculation in futures markets; 6) increasing storage options; and 7) increasing subsidies and food-based safety nets.	2016; Wheeler and von Braun 2013; Wilhelm et al. 2016; Wodon and Zaman 2010; The World Bank 2011
Enhanced urban food systems	Urban areas are becoming the principal territories for intervention in improving food access through innovative strategies that aim to reduce hunger and improve livelihoods. Interventions include Urban and Peri-urban Agriculture and Forestry and local food policy and planning initiatives such as Food Policy Councils and city-region-wide regional food strategies. Such systems have demonstrated inter-linkages of the city and its citizens with surrounding rural areas to create sustainable, and more nutritious food supplies for the city, while improving the health status of urban dwellers, reducing pollution levels, adapting to and mitigating climate change, and stimulating economic development. Options include support for urban and peri-urban agriculture, green infrastructure (e.g., green roofs), local markets, enhanced social (food) safety nets and development of alternative food sources and technologies, such as vertical farming.	Urban territorial areas have a potential to reduce GHG emissions through improved food systems to reduce vehicle miles of food transportation, localised carbon capture and food waste reduction. The benefits of Urban food forests that are intentionally planted woody perennial food producing species, are also cited for their carbon sequestration potentials. However, new urban food systems may have diverse unexpected adverse side-effects with climate systems, such as lower efficiencies in food supply and higher costs than modern large-scale agriculture. Diversifying markets, considering value added products in the food supply system may help to improve food security by increasing its economic performance and revenues to local farmers.	Akhtar et al. 2016; Benis and Ferrão 2017; Brinkley et al. 2013, 2016; Chappell et al. 2016; Goldstein et al. 2016; Kowalski and Conway 2018; Lee-Smith 2010; Barthel and Isendahl 2013; Lwasa et al. 2014, 2015; Revi et al. 2014; Specht et al. 2014; Tao et al. 2015; UPAF (date)
Improved food processing and retailing	Improved food processing and retailing involves several practices related to a) greening supply chains (e.g., utilising products and services with a reduced impact on the environment and human health), b) adoption of specific sustainability instruments among agri-food companies (e.g., eco-innovation practices), c) adopting emission accounting tools (e.g., carbon and water foot-printing), d) implementing "demand forecasting" strategies (e.g., changes in consumer	Improved food processing and retailing can provide benefits for climate mitigation since GHG-friendly foods can reduce agri-food GHG emissions from transportation, waste and energy use. In cases where climate extremes and natural disasters disrupt supply chain networks, improved food processing and retailing can benefit climate adaptation by buffering the impacts of changing temperature and rainfall patterns on upstream agricultural production. It can provide benefits for food security	Chapter 2; Chapter 5; Avetisyan et al. 2014; Garnett et al. 2013; Godfray et al. 2010; Mohammadi et al. 2014; Porter et al. 2016; Ridoutt et al. 2016; Song et al. 2017

	preference for 'green' products) and, e) supporting polycentric supply-chain governance processes.	by supporting healthier diets and reducing food loss and waste. Successful implementation is dependent on organisational capacity, the agility and flexibility of business strategies, the strengthening of public-private policies and effectiveness of supply-chain governance.	
Improved energy use in food systems	Energy efficiency of agriculture can be improved to reduce the dependency on non-renewable energy sources. This can be realised either by decreased energy inputs, or through increased outputs per unit of input. In some countries, managerial inefficiency (rather than a technology gap) is the main source for energy efficiency loss. Heterogenous patterns of energy efficiency exist at the national scale and promoting energy efficient technologies along with managerial capacity development can reduce the gap and provide large benefits for climate adaptation. Improvements in carbon monitoring and calculation techniques such as the foot-printing of agricultural products can enhance energy efficiency transition management and uptake in agricultural enterprises.	Transformation to low carbon technologies such as renewable energy and energy efficiency can offer opportunities for significant climate change mitigation by providing a substitute to transport fuel (for example) that could benefit marginal agricultural resources, while simultaneously contributing to long term economic growth. In poorer nations, increased energy efficiency in agricultural value-added production, in particular, can provide large mitigation benefits. Under certain scenarios, the efficiency of agricultural systems can stagnate and could exert pressure on grasslands and rangelands, thereby impacting land degradation and desertification. Rebound effects can also occur, with adverse impacts on emissions.	Al-Mansour F and Jejcic V 2017; Baptista et al. 2013; Gunatilake et al. 2014; Begum et al. 2015; Jebli and Youssef 2017; van Vuuren et al. 2017b

Table 6.12 Integrated response options based on risk management

Integrated response option	Description	Context and caveats	Supporting evidence
Management of urban sprawl	Unplanned urbanisation leading to sprawl and extensification of cities along the rural-urban fringe has been identified as a driver of forest and agricultural land loss and a threat to food production around cities. It has been estimated that urban expansion will result in a 1.8–2.4% loss of global croplands by 2030. This rapid urban expansion is especially strong in new emerging towns and cities in Asia and Africa. Policies to prevent such urbanisation have included integrated land use planning, agricultural zoning ordinances and agricultural districts, urban redevelopment, arable land reclamation, and transfer/purchase of development rights or easements.	The prevention of uncontrolled urban sprawl may provide adaptation co-benefits, but adverse side effects for adaptation might arise due to restricted ability of people to move in response to climate change.	Barbero-Sierra et al. 2013a; Bren d'Amour et al. 2016; Cai et al. 2013; Chen 2007; Francis et al. 2012a; Gibson et al. 2015; Lee et al. 2015; Qian et al. 2015; Shen et al. 2017; Tan et al. 2009
Livelihood diversification	When households' livelihoods depend on a small number of sources of income without much diversification, and when those income sources are in fields that are highly climate dependent, like agriculture and fishing, this dependence can put food security and livelihoods at risk. Livelihood diversification (drawing from a portfolio of dissimilar sources of livelihood as a tool to spread risk) has been identified as one option to increase incomes and reduce poverty, increase food security, and promote climate resilience and risk reduction.	Livelihood diversification offers benefits for desertification and land degradation, particularly through non-traditional crops or trees in agroforestry systems which improve soil. Livelihood diversification may increase on-farm biodiversity due to these investments in more ecosystem-mimicking production systems, like agroforestry and polycultures. Diversification into non-agricultural fields, such as wage labour or trading, is increasingly favoured by farmers as a low-cost strategy, particularly to respond to increasing climate risks.	Adger 1999; Ahmed and Stepp 2016a; Antwi-Agyei et al. 2014; Barrett et al. 2001; Berman et al. 2012; Bryceson 1999; DiGiano and Racelis 2012; Ellis 1998, 2008; Ngigi et al. 2017; Rakodi 1999; Thornton and Herrero 2014; Little et al. 2001
Use of local seeds	Using local seeds (also called seed sovereignty) refers to use of non-improved, non-commercial seeds varieties. These can be used and stored by local farmers as low-cost inputs and can often help contribute to the conservation of local varieties and land races, increasing local biodiversity. Many local seeds also require no pesticide or fertiliser use, leading to less land degradation in their use.	Use of local seeds is important in the many parts of the developing world that do not rely on commercial seed inputs. Promotion of local seed saving initiatives can include seed networks, banks and exchanges, and non-commercial open source plant breeding. These locally developed seeds can both help protect local agrobiodiversity and can often be more climate resilient than generic commercial varieties, although	Bowman 2015; Campbell and Veteto 2015; Coomes et al. 2015; Kloppenberg 2010; Luby et al. 2015; van Niekerk and Wynberg 2017; Patnaik et al. 2017; Reisman 2017;

		the impacts on food security and overall land degradation are inconclusive.	Vasconcelos et al. 2013; Wattnem 2016
Disaster risk management	Disaster risk management encompasses many approaches to try to reduce the consequences of climate and weather-related disasters and events on socio-economic systems. The Hyogo Plan of Action is a UN framework for nations to build resilience to disasters through effective integration of disaster risk considerations into sustainable development policies. For example, in Vietnam a national strategy on disasters based on Hyogo has introduced the concept of a "four-on-the-spot" approach for disaster risk management of: proactive prevention; timely response; quick and effective recovery; and sustainable development. Other widespread approaches to disaster risk management include using early warning systems that can encompass 1) education systems; 2) hazard and risk maps; 3) hydrological and meteorological monitoring (such as flood forecasting or extreme weather warnings); and 4) communications systems to pass on information to enable action. These approaches have long been considered to reduce the risk of household asset damage during one-off climate events and are increasingly being combined with climate adaptation policies.	Community-based disaster risk management has been pointed to as one of the most successful ways to ensure information reaches people, who need to be participants in risk reduction. Effective disaster risk management approaches must be 'end-to-end,' both reaching communities at risk and supporting and empowering vulnerable communities to take appropriate action. The most effective early warning systems are not simply technical systems of information dissemination, but utilise and develop community capacities, create local ownership of the system, and are based on a shared understanding of needs and purpose. Tapping into existing traditional or local knowledge has also been recommended for disaster risk management approaches to reducing vulnerability.	Ajibade and McBean 2014; Alessa et al. 2016; Bouwer et al. 2014; Carreño et al. 2007; Cools et al. 2016; Djalante et al. 2012; Garschagen 2016; Maskrey 2011; Mercer 2010; Sternberg and Batbuyan 2013; Thomalla et al. 2006; Vogel and O'Brien 2006; Schipper and Pelling 2006
Risk sharing instruments	Risk sharing instruments can encompass a variety of approaches. Intra-household risk pooling is a common strategy in rural communities, such as through extended family financial transfers; one study found 65% of poor households in Jamaica report receiving transfers, and such transfers can account for up to 75% of household income or more after crisis events. Community rotating credit associations (ROSCAs) have long been used for general risk pooling and can be a source of financing to cope with climate variability as well. Credit services have been shown to be important for adaptation actions and risk reduction. Insurance of various kinds is also a form of risk pooling. Commercial crop insurance is one of the most widely used risk-hedging	Locally developed risk pooling measures show general positive impacts on household livelihoods. However, more commercial approaches have mixed effects. Commercial crop insurance is highly subsidised in much of the developed world. Index insurance programmes have often failed to attract sufficient buyers or have remained financially unfeasible for commercial insurance sellers. The overall impact of index insurance on food production supply and access has also not been assessed. Traditional crop insurance has generally been seen as positive for food security as it leads to expansion of agricultural production areas and increased food supply. However, insurance may also 'mask' truly risky agriculture and prevent farmers from seeking less risky	Akter et al. 2016; Annan and Schlenker 2015; Claassen et al. 2011; Fenton et al. 2017; Giné et al. 2008; Goodwin and Smith 2003; Hammill et al. 2008; Havemenn and Muccione 2011; Jaworski 2016; Meze-Hausken et al. 2009; Morduch and Sharma 2002; Bhattamishra and Barrett 2010; Peterson 2012;

	financial vehicles, and can involve both traditional indemnity-based insurance that reimburses clients for estimated financial losses from shortfalls, or index insurance that pays out the value of an index (such as weather events) rather than actual losses; the former is more common for large farms in the developed world and the latter for smaller non-commercial farms in developing countries.	production strategies. Insurance can also provide perverse incentives for farmers to bring additional lands into crop production, leading to greater risk of degradation.	Sanderson et al. 2013; Skees and Collier 2012; Smith and Glauber 2012

Cross-Chapter Box 7: Bioenergy and Bioenergy with Carbon Dioxide Capture and Storage (BECCS) in mitigation scenarios

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Bioenergy and BECCS potential

Using biomass to produce heat, electricity and transport fuels (bioenergy) instead of coal, oil, and natural gas can reduce GHG emissions. Combining biomass conversion technologies with systems that capture CO_2 and inject it into geological formations (bioenergy with carbon dioxide capture and storage (BECCS)) can deliver net negative emissions. The net climate effects of bioenergy and BECCS depend on the magnitude of bioenergy supply chain emissions and land/climate interactions, described further below.

Biomass in 2013 contributed ~60 EJ (10%) to global primary energy[4] (WBA 2016). In 2011, the IPCC *Special Report on Renewable Energy Sources* concluded that biomass supply for energy could reach 100-300 EJ yr^{-1} by 2050 with the caveat that the technical potential[5] cannot be determined precisely while societal preferences are unclear; that deployment depends on "factors that are inherently uncertain"; and that biomass use could evolve in a "sustainable" or "unsustainable" way depending on the governance context (IPCC, 2011). The IPCC WGIII AR5 report noted, in addition, that high deployment levels would require extensive use of technologies able to convert lignocellulosic biomass such as forest wood, agricultural residues, and lignocellulosic crops. The SR15 noted that high levels of bioenergy deployment may result in adverse side-effects for food security, ecosystems, biodiversity, water use, and nutrients (de Coninck et al. 2018).

Although estimates of potential are uncertain, there is *high confidence* that the most important factors determining future biomass supply are land availability and land productivity. These factors are in turn determined by competing uses of land and a myriad of environmental and economic considerations (Dornburg et al. 2010; Batidzirai et al. 2012; Erb et al. 2012; Slade 2014, Searle and Malins 2014). Overlaying estimates of technical potential with such considerations invariably results in a smaller estimate. Recent studies that have attempted to do this estimate that 50-244 EJ biomass could be produced on 0.1-13 Mkm^2 (Fuss et al. 2018a; Schueler et al. 2016; Searle and Malins 2014; IPCC SR15; Wu et al. 2019; Heck et al. 2018; de Coninck et al. 2018). While preferences concerning economic, social and environmental objectives vary geographically and over time, studies commonly estimate "sustainable" potentials by introducing restrictions intended to protect environmental values and avoid negative effects on poor and vulnerable segments in societies.

Estimates of global geological CO_2 storage capacity are large – ranging from 1680 $GtCO_2$ to 24000 $GtCO_2$ (McCollum et al. 2014) – however the potential of BECCS may be significantly constrained by socio-political and technical and geographical considerations, including limits to knowledge and experience (Chapter 6, 7).

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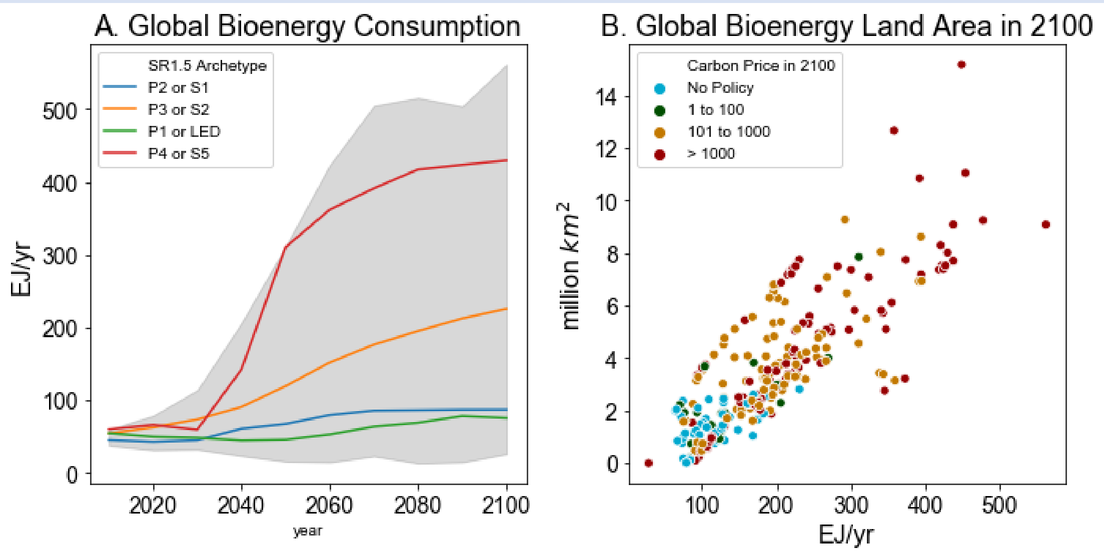
[4] FOOTNOTE: Of this, more than half was traditional biomass, predominately used for cooking and heating in developing regions, bioelectricity accounted for ~1.7EJ, and transport biofuels for 3.19EJ. (Cross-Chapter Box 12 on Traditional Biomass, Chapter 7)

[5] FOOTNOTE: The future availability of biomass is usually discussed in terms of a hierarchy of potentials: theoretical>technical>economic. Caution is required, however, as these terms are not always defined consistently and estimates depend on the specific definitions and calculation methodologies.

Bioenergy and BECCS use in mitigation scenarios

Most mitigation scenarios include substantial deployment of bioenergy technologies (Clarke et al. 2014; Fuss et al. 2014; IPCC SR15). Across all scenarios, the amount of bioenergy and BECCS ranges from 0 EJ yr^{-1} to 561 EJ yr^{-1} in 2100 (Cross-Chapter Box 7 Figure 1, left panel). Notably, all 1.5°C pathways include bioenergy, requiring as much as 7 Mkm^2 to be dedicated to the production of energy crops in 2050 (Rogelj et al. 2018a). If BECCS is excluded as a mitigation option, studies indicate that more biomass may be required in order to substitute for a greater proportion of fossil fuels (Muratori et al. 2016; Rose et al. 2014a).

Different Integrated Assessment Models (IAMs) use alternative approaches to land allocation when determining where and how much biomass is used, with some relying on economic approaches and some relying on rule-based approaches (Popp et al. 2014b). Despite these differences a consistent finding across models is that increasing biomass supply to the extent necessary to support deep decarbonisation is likely to involve substantial land use change (Popp et al. 2017) (Cross-Chapter Box 9). In model runs, bioenergy deployment and the consequent demand for biomass and land, is influenced by assumptions around the price of bioenergy, the yield of bioenergy crops, the cost of production (including the costs of fertiliser and irrigation if used), the demand for land for other uses, and the inclusion of policies (e.g., subsidies, taxes, constraints) that may alter land use or bioenergy demand. In general, higher carbon prices result in greater bioenergy deployment (Cross-Chapter Box 7 Figure 1, right panel) and a larger percentage of BECCS. Other factors can also strongly influence bioenergy use, including the cost and availability of fossil fuels (Calvin et al. 2016a), socioeconomics (Popp et al. 2017), and policy (Calvin et al. 2014a; Reilly et al. 2012a).



Cross-Chapter Box 7 Figure 1: Global bioenergy consumption in IAM scenarios. Data is from an update of the IAMC Scenario Explorer developed for the SR15 (Huppmann et al. 2018; Rogelj et al. 2018a). The left panel shows bioenergy deployment over time for the entire scenario database (grey areas) and the four illustrative pathways from SR15 (Rogelj et al. 2018a). The right panel shows global land area for energy crops in 2100 versus total global bioenergy consumption in 2100; colours indicate the carbon price in 2100 (in 2010 USD per tCO_2). Note that this figure includes 409 scenarios, many of which exceed 1.5ºC.

Co-benefits, adverse side effect, and risks associated with bioenergy

The production and use of biomass for bioenergy can have co-benefits, adverse side effects, and risks for land degradation, food insecurity, GHG emissions, and other environmental goals. These impacts are context specific and depend on the scale of deployment, initial land use, land type, bioenergy feedstock, initial carbon stocks, climatic region and management regime (Qin et al. 2016; Del Grosso et al. 2014; Alexander et al. 2015; Popp et al. 2017; Davis et al. 2013a; Mello et al. 2014b; Hudiburg et al. 2015; Carvalho et al. 2016; Silva-Olaya et al. 2017; Whitaker et al. 2018; Robledo-Abad et al. 2017; Jans et al. 2018).

Synergistic outcomes with bioenergy are possible, for example, strategic integration of perennial bioenergy crops with conventional crops can provide multiple production and environmental benefits including management of dryland salinity, enhanced biocontrol and biodiversity, and reduced eutrophication (Davis et al. 2013b; Larsen et al. 2017; Cacho et al. 2018; Odgaard et al. 2019). Additionally, planting perennial bioenergy crops on low carbon soil could enhance soil carbon sequestration (Bárcena et al. 2014; Schröder et al. 2018; Walter et al. 2015; Robertson et al. 2017a; Rowe et al. 2016; Chadwick et al. 2014; Immerzeel et al. 2014; Del Grosso et al. 2014; Mello et al. 2014c; Whitaker et al. 2018). However, large-scale expansion of bioenergy may also result in increased competition for land (DeCicco 2013; Humpenöder et al. 2018a; Bonsch et al. 2016; Harris et al. 2015; Richards et al. 2017; Ahlgren et al. 2017; Bárcena et al. 2014), increased greenhouse gas emissions from land use change and land management, loss in biodiversity, and nutrient leakage (Harris et al. 2018; Harper et al. 2018; Popp et al. 2011c; Wiloso et al. 2016; Behrman et al. 2015; Valdez et al. 2017; Hof et al. 2018). If biomass crops are planted on land with a high carbon stock, the carbon loss due to land conversion may take decades to over a century to be compensated by either fossil fuel substitution or CCS (Harper et al. 2018). Competition for land may be experienced locally or regionally and is one of the determinants of food prices, food security (Popp et al. 2014a; Bailey 2013; Pahl-Wostl et al. 2018; Rulli et al. 2016; Yamagata et al. 2018; Franz et al. 2017; Kline et al. 2017; Schröder et al. 2018) and water availability (Rulli et al. 2016; Bonsch et al. 2015b; Pahl-Wostl et al. 2018; Bailey 2013; Chang et al. 2016; Bárcena et al. 2014).

Experience in countries at quite different levels of economic development (Brazil, Malawi and Sweden) has shown that persistent efforts over several decades to combine improved technical standards and management approaches with strong governance and coherent policies, can facilitate long-term investment in more sustainable production and sourcing of liquid biofuels (Johnson and Silveira 2014). For woody biomass, combining effective governance with active forest management over long time periods can enhance substitution-sequestration co-benefits, such as in Sweden where bioenergy has tripled during the last 40 years (currently providing about 25% of total energy supply) while forest carbon stocks have continued to grow (Lundmark et al. 2014). A variety of approaches are available at landscape level and in national and regional policies to better reconcile food security, bioenergy and ecosystem services, although more empirical evidence is needed (Mudombi et al. 2018; Manning et al. 2015; Kline et al. 2017; Maltsoglou et al. 2014; Lamers et al.).

Thus, while there is *high confidence* that the technical potential for bioenergy and BECCS is large, there is also *very high confidence* that this potential is reduced when environmental, social and economic constraints are considered. The effects of bioenergy production on land degradation, water scarcity, biodiversity loss, and food insecurity are scale and context specific (*high confidence*). Large areas of monoculture bioenergy crops that displace other land uses can exacerbate these challenges, while integration into sustainably managed agricultural landscapes can ameliorate them (*medium confidence*).

Inventory reporting for BECCS and bioenergy

One of the complications in in assessing the total GHG flux associated with bioenergy under UNFCCC reporting protocols is that fluxes from different aspects of bioenergy life cycle are reported in different sectors and are not linked. In the energy sector, bioenergy is treated as carbon neutral at the point of biomass combustion because all change in land carbon stocks due to biomass harvest or land use change related to bioenergy are reported under AFOLU sector. Use of fertilisers is captured in the Agriculture sector, while fluxes related to transport/conversion and removals due to CCS are reported in the energy sector. IAMs follow a similar reporting convention. Thus, the whole life cycle

GHG effects of bioenergy systems are not readily observed in national GHG inventories or modelled emissions estimates (see also IPCC 2006; SR15 Chapter 2 Technical Annex; Chapter 2).

Bioenergy in this report

Bioenergy and BECCS are discussed throughout this special report. Chapter 1 provides an introduction to bioenergy and BECCS and its links to land and climate. Chapter 2 discusses mitigation potential, land requirements and biophysical climate implications. Chapter 4 includes a discussion of the threats and opportunities with respect to land degradation. Chapter 5 discusses linkages between bioenergy and BECCS and food security. Chapter 6 synthesises the co-benefits and adverse side-effects for mitigation, adaptation, desertification, land degradation, and food security, as well as barriers to implementation (e.g., cost, technological readiness, etc.). Chapter 7 includes a discussion of risk, policy, governance, and decision-making with respect to bioenergy and BECCS.

6.3 Potentials for addressing the land challenges

In this section, we assess how each of the integrated response options described in Section 6.2 address the land challenges of climate change mitigation (6.3.1), climate change adaptation (6.3.2), desertification (6.3.3), land degradation (6.3.4), and food security (6.3.5). The quantified potentials across all of mitigation, adaptation, desertification, land degradation and food security are summarised and categorised for comparison in section 6.3.6.

6.3.1 Potential of the integrated response options for delivering mitigation

In this section, the impacts of integrated response options on climate change mitigation are assessed.

6.3.1.1 Integrated response options based on land management

In this section, the impacts on climate change mitigation of integrated response options based on land management are assessed. Some of the caveats of these potential mitigation studies are discussed in Chapter 2 and section 6.2.1.

6.3.1.1.1 Integrated response options based on land management in agriculture

Increasing the productivity of land used for food production can deliver significant mitigation by avoiding emissions that would occur if increased food demand were met through expansion of the agricultural land area (Burney et al., 2010). If pursued through increased agrochemical inputs, numerous adverse impacts on greenhouse gas emissions (and other environmental sustainability) can occur (Table 6.5), but if pursued through sustainable intensification, increased food productivity could provide high levels of mitigation. For example, yield improvement has been estimated to have contributed to emissions savings of >13 $GtCO_2$ yr^{-1} since 1961 (Burney et al., 2010; Table 6.13). This can also reduce the greenhouse gas intensity of products (Bennetzen et al., 2016) which means a smaller environmental footprint of production, since demand can be met using less land and/or with fewer animals.

Improved cropland management could provide moderate levels of mitigation (1.4-2.3 $GtCO_2e$ yr^{-1}; Smith et al. 2008, 2014c; Pradhan et al., 2013; Table 6.13). The lower estimate of potential is from Pradhan et al. (2013) for decreasing emissions intensity, and the upper end of technical potential is estimated by adding technical potentials for cropland management (about 1.4 $GtCO_2e$ yr^{-1}), rice management (about 0.2 $GtCO_2e$ yr^{-1}) and restoration of degraded land (about 0.7 $GtCO_2e$ yr^{-1}) from Smith et al. (2008) and Smith et al. (2014c). Note that much of this potential arises from soil carbon sequestration so there is an overlap with that response option (see 6.3.1.1.3).

Grazing lands can store large stocks of carbon in soil and root biomass compartments (Conant and Paustian 2002; O'Mara 2012; Zhou et al. 2017). The global mitigation potential is moderate (1.4–1.8 $GtCO_2$ yr^{-1}), with the lower value in the range for technical potential taken from Smith et al. (2008)

which includes only grassland management measures, and the upper value in the range from Herrero et al. (2016), which includes also indirect effects and some components of livestock management, and soil carbon sequestration, so there is overlap with these response options (see below and 6.3.1.1.3). Conant et al. (2005) caution that increases in soil carbon stocks could be offset by increases in N_2O fluxes.

The mitigation potential of improved livestock management is also moderate (0.2–1.8 $GtCO_2e\ yr^{-1}$; Smith et al. (2008) including only direct livestock measures; Herrero et al. (2016) include also indirect effects, and some components of grazing land management and soil carbon sequestration) to high (6.13 Gt CO2e yr^{-1}; Pradhan et al., 2013; Table 6.13). There is an overlap with other response options (see above and 6.3.1.1.3).

Zomer et al. (2017) reported that the trees agroforestry landscapes have increased carbon stock by 7.33 $GtCO_2$ between 2000–2010, which is equivalent to 0.7 $GtCO_2\ yr^{-1}$. Estimates of global potential range from 0.1 $GtCO_2\ yr^{-1}$ to 5.7 $GtCO_2\ yr^{-1}$ (from an optimum implantation scenario of Hawken, 2014), based on an assessment of all values in Griscom et al. (2017a), Hawken (2014), Zomer et al 2016., and Dickie et al. (2014) (Table 6.13).

Agricultural diversification mainly aims at increasing climate resilience, but it may have a small (but globally unquantified) mitigation potential as a function of type of crop, fertiliser management, tillage system, and soil type (Campbell et al. 2014; Cohn et al. 2017).

Reducing conversion of grassland to cropland could provide significant climate mitigation by retaining soil carbon stocks that might otherwise be lost. When grasslands are converted to croplands, they lose about 36% of their soil organic carbon stocks after 20 years (Poeplau et al. 2011). Assuming an average starting soil organic carbon stock of grasslands of 115 t C ha^{-1} (Poeplau et al. 2011), this is equivalent to a loss of 41.5 t C ha^{-1} on conversion to cropland. Mean annual global cropland conversion rates (1961–2003) have been around 47000 $km^2\ yr^{-1}$ (Krause et al. 2017), or 940000 km^2 over a 20 year period. The equivalent loss of soil organic carbon over 20 years would therefore be 14 Gt CO_2e = 0.7 Gt $CO_2\ yr^{-1}$. Griscom et al. (2017a) estimate a cost-effective mitigation potential of 0.03 Gt $CO_2\ yr^{-1}$ (Table 6.13).

Integrated water management provides moderate benefits for climate mitigation due to interactions with other land management strategies. For example, promoting soil carbon conservation (e.g. reduced tillage) can improve the water retention capacity of soils. Jat et al. (2015) found that improved tillage practices and residue incorporation increased water-use efficiency by 30%, rice–wheat yields by 5–37%, income by 28–40% and reduced GHG emission by 16–25%. While irrigated agriculture accounts for only 20% of the total cultivated land, the energy consumption from groundwater irrigation is significant. However, current estimates of mitigation potential are limited to reductions in greenhouse gas emissions mainly in cropland and rice cultivation (Chapter 2; Table 6.13; Smith et al. 2008, 2014c). Li et al. (2006) estimated a 0.52-0.72 $GtCO_2\ yr^{-1}$ reduction using the alternate wetting and drying technique. Current estimates of N_2O release from terrestrial soils and wetlands accounts for 10-15% of anthropogenically fixed nitrogen on the Earth System (Wang et al. 2017).

Table 6.13 summarises the mitigation potentials for agricultural response options, with confidence estimates based on the thresholds outlined in Table 6.53 in section 6.3.6, and indicative (not exhaustive) references upon which the evidence in based.

Table 6.13 Mitigation effects of response options based on land management in agriculture

Integrated response option	Potential	Confidence	Citation
Increased food productivity	>13 $GtCO_2e\ yr^{-1}$	Low confidence	Chapter 5; Burney et al. 2010

Improved cropland management[a]	1.4-2.3 $GtCO_2e$ yr^{-1}	Medium confidence	Chapter 2; Chapter 5; Smith et al. 2008, 2014c; Pradhan et al., 2013;
Improved grazing land management[a]	1.4-1.8 $GtCO_2e$ yr^{-1}	Medium confidence	Chapter 2; Chapter 5; Conant et al. 2017; Herrero et al. 2016; Smith et al. 2008, 2014c
Improved livestock management[a]	0.2-2.4 $GtCO_2e$ yr^{-1}	Medium confidence	Chapter 2; Chapter 5; Smith et al. 2008, 2014c; Herrero et al. 2016
Agroforestry	0.1-5.7 Gt CO_2e yr^{-1}	Medium confidence	Chapter 2; Griscom et al. 2017a; Zomer et al. 2016; Dickie et al. 2014; Hawken 2014;
Agricultural diversification	> 0	Low confidence	Campbell et al. 2014; Cohn et al. 2017
Reduced grassland conversion to cropland	0.03-0.7 Gt CO_2e yr^{-1}	Low confidence	Note high value not shown in Chapter 2; Calculated from values in Krause et al. 2017 and Poeplau et al. 2011; Griscom et al. 2017
Integrated water management	0.1-0.72 Gt CO_2 yr^{-1}	Low confidence	IPCC 2014; Smith et al. 2008, 2014b; Howell et al. 2015; Li et al. 2006; Rahman and Bulbul 2015

[a] Note that Chapter 2 reports mitigation potential for subcategories within this response option and not the combined total reported here.

6.3.1.1.2 Integrated response options based on land management in forests

Improved forest management could potentially contribute to moderate mitigation benefits globally, up to about 2 Gt CO_2e yr^{-1} (Chapter 2, Table 6.14). For managed forests, the most effective forest carbon mitigation strategy is the one that, through increasing biomass productivity, optimises the carbon stocks (in forests and in long-lived products) as well as the wood substitution effects for a given time frame (Smyth et al. 2014; Grassi et al. 2018; Nabuurs et al. 2007; Lewis et al. 2019) (Kurz et al. 2016; Erb et al. 2018). Estimates of the mitigation potential vary also depending on the counterfactual, such as business-as-usual management (e.g. Grassi et al. 2018) or other scenarios. Climate change will affect the mitigation potential of forest management due to an increase in extreme events like fires, insects and pathogens (Seidl et al. 2017). More detailed estimates are available at regional or biome level. For instance, according to Nabuurs et al. (2017), the implementation of Climate-Smart Forestry (a combination of improved forest management, expansion of forest areas, energy substitution, establishment of forest reserves, etc.) in the European Union has the potential to contribute to an additional 0.4 Gt CO_2 yr^{-1} mitigation by 2050. Sustainable forest management is often associated with a number of co-benefits for adaptation, ecosystem services, biodiversity conservation, microclimatic regulation, soil erosion protection, coastal area protection and water and flood regulation (Locatelli 2011). Forest management mitigation measures are more likely to be long-lasting if integrated into adaptation measures for communities and ecosystems, for example, through landscape management (Locatelli et al. 2011). Adoption of reduced-impact logging and wood processing technologies along with financial incentives can reduce forest fires, forest degradation, maintain timber production, and retain carbon stocks (Sasaki et al. 2016). Forest certification may support sustainable forest management, helping to prevent forest degradation and over-logging (Rametsteiner and Simula 2003). Community forest management has proven a viable model for sustainable forestry, including for carbon sequestration (Chhatre and Agrawal 2009, Chapter 7, section 7.6.4).

Reducing deforestation and forest degradation rates represents one of the most effective and robust options for climate change mitigation, with large mitigation benefits globally (Chapter 2, Chapter 4, Table 6.14). Because of the combined climate impacts of GHGs and biophysical effects, reducing deforestation in the tropics has a major climate mitigation effect, with benefits at local levels too

(Chapter 2, Alkama and Cescatti 2016). Reduced deforestation and forest degradation typically lead to large co-benefits for other ecosystem services (Table 6.14).

A large range of estimates exist in the scientific literature for the mitigation potential of reforestation and forest restoration, and they sometimes overlap with estimates for afforestation. At global level the overall potential for these options is large, reaching about 10 $GtCO_2$ yr^{-1} (Chapter 2, Table 6.14). The greatest potential for these options is in tropical and subtropical climate (Houghton and Nassikas 2018). Furthermore, climate change mitigation benefits of afforestation, reforestation and forest restoration are reduced at high latitudes owing to the surface albedo feedback (see Chapter 2).

Table 6.14 summarises the mitigation potentials for forest response options, with confidence estimates based on the thresholds outlined in Table 6.53 in section 6.3.6, and indicative (not exhaustive) references upon which the evidence in based.

Table 6.14 Mitigation effects of response options based on land management in forests

Integrated response option	Potential	Confidence	Citation
Improved forest management	0.4-2.1 Gt CO_2e yr^{-1}	Medium confidence	Chapter 2; Griscom 2017; Sasaki et al. 2016
Reduced deforestation and degradation	0.4-5.8 Gt CO_2e yr^{-1}	High confidence	Chapter 2; Houghton & Nassikas 2018; Griscom 2017; Baccini 2017; Hawken 2017; Houghton et al 2015; Smith et al. 2014a
Reforestation and forest restoration	1.5-10.1 Gt CO_2e yr^{-1}	Medium confidence	Chapter 2; Dooley and Kartha 2018a; Hawken 2017; Houghton & Nassikas 2018; Griscom 2017. Estimates partially overlapping with Afforestation.
Afforestation	0.5-8.9 Gt CO_2e yr^{-1}	Medium confidence	Chapter 2; Fuss et al. 2018; Hawken 2017; Kreidenweis et al. 2016; Lenton 2010. Estimates partially overlapping with Reforestation.

6.3.1.1.3 Integrated response options based on land management of soils

The global mitigation potential for increasing soil organic matter stocks in mineral soils is estimated to be in the range of 1.3–5.1 $GtCO_2e$ yr^{-1}, though the full literature range is wider (Smith et al. 2008; Smith 2016; Fuss et al 2018.; Sanderman et al. 2017; Sommer & Bossio 2014; Lal 2004; Lal et al. 2010; Table 6.15).

The management and control of erosion may prevent losses of organic carbon in water- or wind-transported sediments, but since the final fate of eroded material is still debated, ranging from a source of 1.36–3.67 $GtCO_2$ yr^{-1} (Jacinthe and Lal 2001; Lal et al., 2004) to a sink of 0.44–3.67 $GtCO_2$ yr^{-1} (Stallard 1998; Smith et al. 2001, 2005; Van Oost et al. 2007; Table 6.15), the overall impact of erosion control on mitigation is context-specific and at the uncertain at the global level (Hoffmann et al., 2013).

Salt-affected soils are highly constrained environments that require permanent prevention of salinisation. Their mitigation potential is likely to be small (Wong et al. 2010; UNCTAD 2011; Dagar et al. 2016b).

Soil compaction prevention could reduce N_2O emissions by minimising anoxic conditions favourable for denitrification (Mbow et al. 2010), but its carbon sequestration potential depends on crop management and the global mitigation potential, though globally unquantified, is likely to be small (Chamen et al. 2015; Epron et al. 2016; Tullberg et al. 2018; Table 6.15).

For biochar, a global analysis of technical potential, in which biomass supply constraints were applied to protect against food insecurity, loss of habitat and land degradation, estimated technical potential abatement of 3.7–6.6 $GtCO_2e\ yr^{-1}$ (including 2.6–4.6 $GtCO_2e\ yr^{-1}$ carbon stabilisation). Considering all published estimates by Woolf et al. (2010), Smith (2016), Fuss et al. (2018b), Griscom et al. (2017), Hawken (2017), Paustian et al. (2016), Powell & Lenton (2012), Dickie et al. (2014), Lenton (2010), Lenton (2014), Roberts et al. (2010), Pratt & Moran (2010) and IPCC (2018), the low value for the range of potentials of 0.03 $GtCO_2e\ yr^{-1}$ is for the "plausible" scenario of Hawken, 2017; Table 6.15). Fuss et al. (2018) propose a range of 0.5–2 $GtCO_2e\ yr^{-1}$ as the sustainable potential for negative emissions through biochar, similar to the range proposed by Smith (2016) and IPCC (2018).

Table 6.15 summarises the mitigation potentials for soil-based response options, with confidence estimates based on the thresholds outlined in Table 6.53 in section 6.3.6, and indicative (not exhaustive) references upon which the evidence in based.

Table 6.15 Mitigation effects of response options based on land management of soils

Integrated response option	Potential	Confidence	Citation
Increased soil organic carbon content	0.4-8.6 $GtCO_2e\ yr^{-1}$	High confidence	Chapter 2; McLaren 2012; Poeplau and Don 2015; Conant et al. 2017; Dickie et al. 2014; Frank et al. 2017; Fuss et al. 2018b; Griscom et al. 2017; Herrero et al. 2016; Paustian et al. 2016; Powlson et al. 2014; Sanderman et al. 2017; Smith 2016b; Zomer et al. 2016; Hawken 2017; Henderson et al. 2015; Lal 2004; Lal et al. 2010; Sommer & Bossio 2014;
Reduced soil erosion	Source of 1.36-3.67 to sink of 0.44-3.67 Gt $CO_2e\ yr^{-1}$	Low confidence	Chapter 2; Jacinthe and Lal 2001; Smith et al. 2001, 2005; Stallard 1998; Van Oost et al. 2007; Lal et al., 2004; Stallard, 1998
Reduced soil salinisation	>0	Low confidence	Dagar et al. 2016b; UNCTAD 2011; Wong et al. 2010
Reduced soil compaction	>0	Low confidence	Chamen et al. 2015b; Epron et al. 2016; Tullberg et al. 2018b
Biochar addition to soil	0.03-6.6 $GtCO_2e\ yr^{-1}$	Medium confidence	Chapter 2; IPCC 2018; Fuss et al. 2018b; Griscom et al. 2017a; Lenton 2010; Paustian et al. 2016; Smith 2016; Woolf et al. 2010; Dickie et al. 2014; Hawken 2017; Lenton 2014; Powell & Lenton 2012; Pratt & Moran 2010; Roberts et al 2010;

6.3.1.1.4 Integrated response options based on land management in all/other ecosystems

For fire management, total emissions from fires have been in the order of 8.1 $GtCO_2e\ yr^{-1}$ for the period 1997-2016 (see Chapter 2, Cross-Chapter Box 3) and there are important synergies between air pollution and climate change control policies. Reduction in fire CO_2 emissions due to fire suppression and landscape fragmentation associated with increases in population density is calculated to enhance land carbon uptake by 0.48 Gt CO_2e yr^{-1} for the 1960–2009 period (Arora and Melton 2018; Table 6.16).

Management of landslides and natural hazards is a key climate adaptation option but due to limited global areas vulnerable to landslides and natural hazards, its mitigation potential is likely to be modest (Noble et al. 2015).

In terms of management of pollution, including acidification, UNEP and WMO (2011) and Shindell et al. (2012) identified measures targeting reduction in SLCP emissions that reduce projected global mean warming about 0.5°C by 2050. Bala et al. (2013) reported that a recent coupled modelling study showed N deposition and elevated CO_2 could have a synergistic effect, which could explain 47% of terrestrial carbon uptake in the 1990s. Estimates of global terrestrial carbon uptake due to current N deposition ranges between 0.55 and 1.28 $GtCO_2$ yr^{-1} (de Vries et al. 2006; de Vries et al. 2009; Bala et al. 2013; Zaehle and Dalmonech 2011; Table 6.16).

There are no global data on the impacts of management of invasive species / encroachment on mitigation.

Coastal wetland restoration could provide high levels of climate mitigation, with avoided coastal wetland impacts and coastal wetland restoration estimated to deliver 0.3-3.1 $GtCO_2e$ yr^{-1} in total when considering all global estimates from Griscom et al. (2017a), Hawken (2017), Pendleton et al. (2012), Howard et al. (2017) and Donato et al. (2010) (Table 6.16).

Peatland restoration could provide moderate levels of climate mitigation, with avoided peat impacts and peat restoration estimated to deliver 0.6-2 $GtCO_2e$ yr^{-1} from all global estimates published in Griscom et al. (2017a), Hawken (2017), Hooijer et al. (2010), Couwenberg et al. (2010) and Joosten and Couwenberg (2008), though there could be an increase in methane emissions after restoration (Jauhiainen et al. 2008; Table 6.16).

Mitigation potential from biodiversity conservation varies depending on the type of intervention and specific context. Protected areas are estimated to store over 300 Gt carbon, roughly corresponding to 15% of terrestrial carbon stocks (Kapos et al. 2008; Campbell et al. 2008). At global level, the potential mitigation resulting from protection of these areas for the period 2005-2095 is on average about 0.9 $GtCO_2$-eq. yr^{-1} relative to a reference scenario (Calvin et al. 2014a). The potential effects on the carbon cycle of management of wild animal species are case context dependent. For example, moose browsing in boreal forests can decrease the carbon uptake of ecosystems by up to 75% (Schmitz et al. 2018b), and reducing moose density through active population management in Canada is estimated to be a carbon sink equivalent to about 0.37 Gt CO_2e yr^{-1} (Schmitz et al. 2014).

Table 6.16 summarises the mitigation potentials for land management response options in all/other ecosystems, with confidence estimates based on the thresholds outlined in Table 6.53 in section 6.3.6, and indicative (not exhaustive) references upon which the evidence in based.

Table 6.16 Mitigation effects of response options based on land management in all/other ecosystems

Integrated response option	**Potential**	**Confidence**	**Citation**
Fire management	0.48-8.1 $GtCO_2e$ yr^{-1}	Medium confidence	Chapter 2, Cross-Chapter Box 3 on Fire (Chapter 2); Arora and Melton 2018; Tacconi 2016
Reduced landslides and natural hazards	>0	Low confidence	
Reduced pollution including acidification	1) Reduce projected warming ~0.5ºC by 2050; 2) Reduce terrestrial C uptake 0.55-1.28 $GtCO_2e$ yr^{-1}	1) and 2) Medium confidence	1) Shindell et al., 2012; UNEP and WMO, 2011; 2) Bala et al. 2013

Management of invasive species / encroachment	No global estimates	No evidence	
Restoration and reduced conversion of coastal wetlands	0.3-3.1 $GtCO_2e\ yr^{-1}$	Medium confidence	Chapter 2; Griscom et al. 2017a;Donato et al. 2010; Hawken 2017; Howard et al. 2017; Pendleton et al. 2012;
Restoration and reduced conversion of peatlands	0.6-2 $GtCO_2e\ yr^{-1}$	Medium confidence	Chapter 2; Couwenberg et al. 2010; Griscom et al. 2017a; Hooijer et al. 2010; Joosten and Couwenberg 2008; Hawken 2017;
Biodiversity conservation	~0.9 $GtCO_2$-e yr^{-1}	Low confidence	Chapter 2; Calvin et al. 2014c; Schmitz et al. 2014

6.3.1.1.5 Integrated response options based on land management specifically for CDR

Enhanced mineral weathering provides substantial climate mitigation, with a global mitigation potential in the region of about 0.5–4 $GtCO_2e\ yr^{-1}$ (Beerling et al 2018.; Lenton 2010; Smith et al. 2016c; Taylor et al. 2016; Table 6.17).

The mitigation potential for bioenergy and BECCS derived from bottom-up models is large (IPCC SR15; Chapter 2; Cross-Chapter Box 7 on Bioenergy in this chapter), with technical potential estimated at 100-300 EJ yr^{-1} (IPCC 2011; Cross-Chapter Box 7 in this chapter) or up to ~11 $GtCO_2\ yr^{-1}$ (Chapter 2). These estimates, however, exclude N_2O associated with fertiliser application and land-use change emissions. Those effects are included in the modelled scenarios using bioenergy and BECCS, with the sign and magnitude depending on where the bioenergy is grown (Wise et al. 2015), at what scale, and whether N fertiliser is used.

Table 6.17 summarises the mitigation potentials for land management options specifically for CDR, with confidence estimates based on the thresholds outlined in Table 6.53 in section 6.3.6, and indicative (not exhaustive) references upon which the evidence in based.

Table 6.17 Mitigation effects of response options based on land management specifically for CDR

Integrated response option	**Potential**	**Confidence**	**Citation**
Enhanced weathering of minerals	0.5-4 $GtCO_2\ yr^{-1}$	Medium confidence	Chapter 2; Beerling et al.; Lenton 2010; Smith et al. 2016c; Taylor et al. 2016b
Bioenergy and BECCS	0.4-11.3 $GtCO_2\ yr^{-1}$	Medium confidence	Chapter 2; IPCC SR15; Fuss et al. 2018b; Lenton 2014; McLaren 2012; Lenton 2010; Powell and Lenton 2012

6.3.1.2 Integrated response options based on value chain management

In this section, the impacts on climate change mitigation of integrated response options based on value chain management are assessed.

6.3.1.2.1 Integrated response options based on value chain management through demand management

Dietary change and waste reduction can provide large benefits for mitigation, with potentials of 0.7-8 $GtCO_2\ yr^{-1}$ for both (Bajželj et al. 2014; Stehfest et al. 2009; Tilman and Clark 2014b; Aleksandrowicz et al. 2016; Herrero et al. 2016; Springmann et al. 2016; Smith et al. 2013; Dickie et al. 2014;Popp et al. 2010; Hawken 2017; Hedenus (2014)). Estimates for food waste reduction

(Hawken 2017; Hic et al. 2016; Dickie et al. 2014) (Bajželj et al. 2014) include both consumer / retailed waste and post-harvest losses (Table 6.18).

Some studies indicate that material substitution has the potential for significant mitigation, with one study estimating a 14–31% reduction in global CO_2 emissions (Oliver et al. 2014); other studies suggest more modest potential (Gustavsson et al. 2006; Table 6.18).

Table 6.18 summarises the mitigation potentials for demand management options, with confidence estimates based on the thresholds outlined in Table 6.53 in section 6.3.6, and indicative (not exhaustive) references upon which the evidence in based.

Table 6.18 Mitigation effects of response options based on demand management

Integrated response option	Potential	Confidence	Citation
Dietary change	0.7 to 8 $GtCO_2\ yr^{-1}$	High confidence	Chapter 2; Chapter 5; Bajželj et al. 2014; Herrero et al. 2016; Smith et al. 2013; Springmann et al. 2016, 2018b; Stehfest et al. 2009; Tilman and Clark 2014b; Dickie et al. 2014; Hawken 2017; Hedenus 2014; Popp et al. 2010;
Reduced post-harvest losses	4.5 $GtCO_2\ yr^{-1}$	High confidence	Chapter 5; Bajželj et al. 2014
Reduced food waste (consumer or retailer)	0.8 to 4.5 $GtCO_2\ yr^{-1}$	High confidence	Chapter 5; Bajželj et al. 2014; Dickie et al. 2014; Hawken 2017; Hic et al. 2016
Material substitution	0.25 to 1 $GtCO_2\ yr^{-1}$	Medium confidence	Chapter 2; Dugan et al. 2018; Gustavsson et al. 2006; Leskinen et al. 2018; McLaren 2012; Sathre and O'Connor 2010; Miner 2010;Kauppi 2001; Smyth et al. 2016

6.3.1.2.2 Integrated response options based on value chain management through supply management

While sustainable sourcing presumably delivers a mitigation benefit, there are no global estimates of potential. Palm oil production alone is estimated to contribute 0.038 to 0.045 GtC yr^{-1}, and Indonesian palm oil expansion contributed up to 9% of tropical land use change carbon emissions in the 2000s (Carlson and Curran 2013), however, the mitigation benefit of sustainable sourcing of palm oil has not been quantified. There are no estimates of the mitigation potential for urban food systems.

Efficient use of energy and resources in food transport and distribution contribute to a reduction in GHG emissions, estimated to be 1% of global CO_2 emissions (James and James 2010; Vermeulen et al. 2012). Given that global CO_2 emissions in 2017 were 37 $GtCO_2$, this equates to 0.37 $GtCO_2\ yr^{-1}$ (covering food transport and distribution, improved efficiency of food processing and retailing, and improved energy efficiency; Table 6.19).

Table 6.19 summarises the mitigation potentials for supply management options, with confidence estimates based on the thresholds outlined in Table 6.53 in section 6.3.6, and indicative (not exhaustive) references upon which the evidence in based.

Table 6.19 Mitigation effects of response options based on supply management

Integrated response option	Potential	Confidence	Citation
Sustainable sourcing	No global estimates	No evidence	
Management of supply chains	No global estimates	No evidence	
Enhanced urban food systems	No global estimates	No evidence	
Improved food processing and retailing	See improved energy efficiency		
Improved energy use in food	0.37 $GtCO_2\ yr^{-1}$	Low confidence	James and James

systems			2010b; Vermeulen et al. 2012b

6.3.1.3 Integrated response options based on risk management

In this section, the impacts on climate change mitigation of integrated response options based on risk management are assessed. In general, because these options are focused on adaptation and other benefits, the mitigation benefits are modest, and mostly unquantified.

Extensive and less dense urban development tends to have higher energy usage, particularly from transport (Liu et al. 2015), such that a 10% reduction of very low density urban fabrics is correlated with 9% fewer emissions per capita in Europe (Baur et al. 2015). However, the exact contribution to mitigation from the prevention of land conversion in particular has not been well quantified (Thornbush et al. 2013). Suggestions from select studies in the US are that biomass decreases by half in cases of conversion from forest to urban land uses (Briber et al. 2015), and a study in Bangkok found a decline by half in carbon sinks in the urban area in the past 30 years (Ali et al. 2018).

There is no literature specifically on linkages between livelihood diversification and climate mitigation benefits, although some forms of diversification that include agroforestry would likely result in increased carbon sinks (Altieri et al. 2015; Descheemaeker et al. 2016). There is no literature exploring linkages between local seeds and GHG emission reductions, although use of local seeds likely reduces emissions associated with transport for commercial seeds, though the impact has not been quantified.

While disaster risk management can presumably have mitigation co-benefits, as it can help reduce food loss on-farm (e.g. crops destroyed before harvest or avoided animal deaths during droughts and floods meaning reduced production losses and wasted emissions), there is no quantified global estimate for this potential.

Risk sharing instruments could have some mitigation co-benefits if they buffer household losses and reduce the need to expand agricultural lands after experiencing risks. However, the overall impacts of these are unknown. Further, commercial insurance may induce producers to bring additional land into crop production, particularly marginal or land with other risks that may be more environmentally sensitive (Claassen et al. 2011). Policies to deny crop insurance to farmers who have converted grasslands in the US resulted in a 9% drop in conversion, which likely has positive mitigation impacts (Claassen et al. 2011). Estimates of emissions from cropland conversion in the US in 2016 were 23.8 Mt CO_2e, only some of which could be attributed to insurance as a driver.

Table 6.20 summarises the mitigation potentials for risk management options, with confidence estimates based on the thresholds outlined in Table 6.53 in section 6.3.6, and indicative (not exhaustive) references upon which the evidence is based.

Table 6.20 Mitigation effects of response options based on risk management

Integrated response option	Potential	Confidence	Citation
Management of urban sprawl	No global estimates	No evidence	
Livelihood diversification	No global estimates	No evidence	
Use of local seeds	No global estimates	No evidence	
Disaster risk management	No global estimates	No evidence	
Risk sharing instruments	->0.024 GtCO_2e yr^{-1} for crop insurance; likely some benefits for other risk sharing instruments	Low confidence	Claussen et al 2011; EPA 2018

6.3.2 Potential of the integrated response options for delivering adaptation

In this section, the impacts of integrated response options on climate change adaptation are assessed.

6.3.2.1 Integrated response options based on land management

In this section, the impacts on climate change adaptation of integrated response options based on land management are assessed.

6.3.2.1.1 Integrated response options based on land management in agriculture

Increasing food productivity by practices such as sustainable intensification improves farm incomes and allows households to build assets for use in times of stress, thereby improving resilience (Campbell et al. 2014). By reducing pressure on land and increasing food production, increased food productivity could be beneficial for adaptation (Chapter 2; Section 6.3; Campbell et al. 2014). Pretty et al. (2018) report that 163 million farms occupying 4.53 Mkm^2 have passed a redesign threshold for application of sustainable intensification, suggesting the minimum number of people benefiting from increased productivity and adaptation benefits under sustainable intensification is >163 million, with the total likely to be far higher (Table 6.21).

Improved cropland management is a key climate adaptation option, potentially affecting more than 25 million people, including a wide range of technological decisions by farmers. Actions towards adaptation fall into two broad overlapping areas: (1) accelerated adaptation to progressive climate change over decadal time scales, for example integrated packages of technology, agronomy and policy options for farmers and food systems, including changing planting dates and zones, tillage systems, crop types and varieties, and (2) better management of agricultural risks associated with increasing climate variability and extreme events, for example improved climate information services and safety nets (Vermeulen et al. 2012b; Challinor et al. 2014; Lipper et al. 2014; Lobell 2014). In the same way, improved livestock management is another technological adaptation option potentially benefiting 1–25 million people. Crop and animal diversification are considered the most promising adaptation measures (Porter et al. 2014; Rojas-Downing et al. 2017a). In grasslands and rangelands, regulation of stocking rates, grazing field dimensions, establishment of exclosures and locations of drinking fountains and feeders are strategic decisions by farmers to improve grazing management (Taboada et al. 2011; Mekuria and Aynekulu 2013; Porter et al. 2014).

Around 30% of the world's rural population use trees across 46% of all agricultural landscapes (Lasco et al. 2014), meaning that up to 2.3 billion people benefit from agroforestry, globally (Table 6.21).

Agricultural diversification is key to achieve climatic resilience (Campbell et al. 2014; Cohn et al. 2017). Crop diversification is one important adaptation option to progressive climate change (Vermeulen et al. 2012) and it can improve resilience by engendering a greater ability to suppress pest outbreaks and dampen pathogen transmission, as well as by buffering crop production from the effects of greater climate variability and extreme events (Lin 2011).

Reduced conversion of grassland to cropland may lead to adaptation benefits by stabilising soils in the face of extreme climatic events (Lal 2001b), thereby increasing resilience, but since it would likely have a negative impact on food production / security (since croplands produce more food per unit area than grasslands), the wider adaptation impacts would likely be negative. However, there is no literature quantifying the global impact of avoidance of conversion of grassland to cropland on adaptation.

Integrated water management provides large co-benefits for adaptation (Dillon and Arshad 2016) by improving the resilience of food crop production systems to future climate change (Chapter 2; Table 6.7; Porter et al. 2014). Improving irrigation systems and integrated water resource management, such as enhancing urban and rural water supplies and reducing water evaporation losses (Dillon and

Arshad 2016), are significant options for enhancing climate adaptation. Many technical innovations (e.g., precision water management) can lead to beneficial adaptation outcomes by increasing water availability and the reliability of agricultural production, using different techniques of water harvesting, storage, and its judicious utilisation through farm ponds, dams, and community tanks in rainfed agriculture areas. Integrated water management response options that use freshwater would be expected to have few adverse side effects in regions where water is plentiful, but large adverse side effects in regions where water is scarce (Grey and Sadoff 2007; Liu et al. 2017; Scott et al. 2011).

Table 6.21 summarises the potentials for adaptation for agricultural response options, with confidence estimates based on the thresholds outlined in Table 6.53 in section 6.3.6, and indicative (not exhaustive) references upon which the evidence in based.

Table 6.21 Adaptation effects of response options based on land management in agriculture

Integrated response option	Potential	Confidence	Citation
Increased food productivity	>163 million people	Medium confidence	Pretty et al. 2018
Improved cropland management	>25 million people	Low confidence	Challinor et al. 2014; Lipper et al. 2014; Lobell 2014; Vermeulen et al. 2012b
Improved grazing land management	1-25 million people	Low confidence	Porter et al. 2014
Improved livestock management	1-25 million people	Low confidence	Porter et al. 2014; Rojas-Downing et al. 2017
Agroforestry	2300 million people	Medium confidence	Lasco et al. 2014
Agricultural diversification	>25 million people	Low confidence	Campbell et al. 2014; Cohn et al. 2017; Vermeulen et al. 2012b
Reduced grassland conversion to cropland	No global estimates	No evidence	
Integrated water management	250 million people	Low confidence	Dillon and Arshad 2016; Liu et al. 2017

6.3.2.1.2 Integrated response options based on land management in forestry

Improved forest management positively impacts adaptation through limiting the negative effects associated with pollution (of air and fresh water), infections and other diseases, exposure to extreme weather events and natural disasters, and poverty (e.g., Smith et al. 2014c). There is high agreement on the fact that reduced deforestation and forest degradation positively impact adaptation and resilience of coupled human-natural systems. Based on the number of people affected by natural disasters (CRED 2015), the number of people depending to varying degrees on forests for their livelihoods (World Bank et al. 2009) and the current deforestation rate (Keenan et al. 2015), the estimated global potential effect for adaptation is large positive for improved forest management, and moderate positive for reduced deforestation when cumulated till the end of the century (Table 6.22). The uncertainty of these global estimates is high, e.g. the impact of reduced deforestation may be higher when the large biophysical impacts on the water cycle (and thus drought) from deforestation (e.g. Alkama & Cescatti 2016, etc) are taken into account (see Chapter 2).

More robust qualitative and some quantitative estimates are available at local and regional level. According to Karjalainen et al. (2009), reducing deforestation and habitat alteration contributes to limiting infectious diseases such as malaria in Africa, Asia, and Latin America, thus lowering the expenses associated with healthcare treatments. Bhattacharjee and Behera (2017) found that human lives lost due to floods increase with reducing forest cover and increasing deforestation rates in India. In addition, maintaining forest cover in urban contexts reduces air pollution and therefore avoids mortality of about one person per year per city in US, and up to 7.6 people per year in New York City

(Nowak et al. 2014). There is also evidence that reducing deforestation and degradation in mangrove plantations potentially improves soil stabilisation, and attenuates the impact of tropical cyclones and typhoons along the coastal areas in South and Southeast Asia (Chow 2018). At local scale, co-benefits between REDD+ and adaptation of local communities can potentially be substantial (Long 2013; Morita & Matsumoto 2017), even if often difficult to quantify, and not explicitly acknowledged (McElwee et al. 2017b).

Forest restoration may facilitate the adaptation and resilience of forests to climate change by enhancing connectivity between forest areas and conserving biodiversity hotspots (Locatelli et al. 2011, 2015c; Ellison et al. 2017; Dooley and Kartha 2018b). Furthermore, forest restoration may improve ecosystem functionality and services, provide microclimatic regulation for people and crops, wood and fodder as safety nets, soil erosion protection and soil fertility enhancement for agricultural resilience, coastal area protection, water and flood regulation (Locatelli et al. 2015c).

Afforestation and reforestation are important climate change adaptation response options (Reyer et al. 2009; Ellison et al. 2017a; Locatelli et al. 2015c), and can potentially help a large proportion of the global population to adapt to climate change and to associated natural disasters (Table 6.22). For example, trees general mitigate summer mean warming and temperature extremes (Findell et al. 2017; Sonntag et al. 2016).

Table 6.22 summarises the potentials for adaptation for forest response options, with confidence estimates based on the thresholds outlined in Table 6.53 in section 6.3.6, and indicative (not exhaustive) references upon which the evidence in based.

Table 6.22 Adaptation effects of response options based on land management in forests

Integrated response option	Potential	Confidence	Citation
Improved forest management	> 25 million people	Low confidence	CRED 2015; World Bank et al. 2009
Reduced deforestation and degradation	1-25 million people	Low confidence	CRED 2015; Keenan et al. 2015; World Bank et al. 2009. The estimates consider a cumulated effect till the end of the century.
Reforestation and forest restoration	See afforestation		
Afforestation	> 25 million people	Medium confidence	Griscom et al. 2017a; Reyer et al. 2009; Smith et al. 2014b; Sonntag et al. 2016. CRED 2015; World Bank, FAO, and IFAD, 2009. The estimates consider a cumulated effect till the end of the century.

6.3.2.1.3 Integrated response options based on land management of soils

Soil organic carbon increase is promoted as an action for climate change adaptation. Since increasing soil organic matter content is a measure to address land degradation (see Section 6.2.1), and restoring degraded land helps to improve resilience to climate change, soil carbon increase is an important option for climate change adaptation. With around 120 thousand km^2 lost to degradation every year, and over 3.2 billion people negatively impacted by land degradation globally (IPBES 2018), practices designed to increase soil organic carbon have a large potential to address adaptation challenges (Table 6.23).

Since soil erosion control prevents land degradation and desertification, it improves the resilience of agriculture to climate change and increases food production (Lal 1998; IPBES 2018), though the global number of people benefiting from improved resilience to climate change has not been reported in the literature. Using figures from (FAO et al. 2015), Scholes et al. (2018) estimates that land losses due to erosion are equivalent to 1.5 Mkm^2 of land used for crop production to 2050, or 45 thousand km^2 yr^{-1} (Foley et al. 2011). Control of soil erosion (water and wind) could benefit 11 Mkm^2 of degraded land (Lal 2014), and improve the resilience of at least some of the 3.2 billion people affected by land degradation (IPBES 2018), suggesting positive impacts on adaptation. Management of erosion is an important climate change adaptation measure, since it reduces the vulnerability of soils to loss under climate extremes, thereby increasing resilience to climate change (Garbrecht et al. 2015).

Prevention and/or reversion of topsoil salinisation may require a combined management of groundwater, irrigation techniques, drainage, mulching and vegetation, with all of these considered relevant for adaptation (Qadir et al. 2013; UNCTAD 2011; Dagar et al. 2016b). Taking into account the widespread diffusion of salinity problems, many people can benefit from its implementation by farmers. The relation between compaction prevention and/or reversion and climate adaption is less evident, and can be related to better hydrological soil functioning (Chamen et al. 2015; Epron et al. 2016; Tullberg et al. 2018b).

Biochar has potential to benefit climate adaptation by improving the resilience of food crop production systems to future climate change by increasing yield in some regions and improving water holding capacity (Chapter 2; Section 6.4; Woolf et al. 2010; Sohi 2012). By increasing yield by 25% in the tropics (Jeffery et al. 2017), this could increase food production for 3.2 billion people affected by land degradation (IPBES 2018), thereby potentially improving their resilience to climate change shocks (Table 6.23). A requirement for large areas of land to provide feedstock for biochar could adversely impact adaptation, though the impact has not been quantified globally.

Table 6.23 summarises the potentials for adaptation for soil-based response options, with confidence estimates based on the thresholds outlined in Table 6.53 in section 6.3.6, and indicative (not exhaustive) references upon which the evidence in based.

Table 6.23 Adaptation effects of response options based on land management of soils

Integrated response option	Potential	Confidence	Citation
Increased soil organic carbon content	Up to 3200 million people	Low confidence	Scholes et al. 2018
Reduced soil erosion	Up to 3200 million people	Low confidence	Scholes et al. 2018
Reduced soil salinisation	1-25 million people	Low confidence	Dagar et al. 2016b; Qadir et al. 2013b; UNCTAD 2011
Reduced soil compaction	<1 million people	Low confidence	Chamen et al. 2015c; Epron et al. 2016; Tullberg et al. 2018b
Biochar addition to soil	Up to 3200 million people; but potential negative (unquantified) impacts from land required from feedstocks	Low confidence	Jeffery et al. 2017

6.3.2.1.4 Integrated response options based on land management across all/other ecosystems

For fire management, Doerr et al. (2016) showed the number of people killed by wildfire was 1940, and the total number of people affected was 5.8 million from 1984 to 2013, globally. Johnston et al. (2012) showed the average mortality attributable to landscape fire smoke exposure was 339 thousand deaths annually. The regions most affected were sub-Saharan Africa (157 thousand) and Southeast Asia (110 thousand). Estimated annual mortality during La Niña was 262 thousand, compared with around 100 thousand excess deaths across Indonesia, Malaysia and Singapore (Table 6.24).

Management of landslides and natural hazards are usually listed among planned adaptation options in mountainous and sloped hilly areas, where uncontrolled runoff and avalanches may cause climatic disasters, affecting millions of people from both urban and rural areas. Landslide control requires both increasing plant cover and engineering practices (see Table 6.8).

For management of pollution, including acidification, Anenberg et al. (2012) estimated that, for PM2.5 and ozone, respectively, fully implementing reduction measures could reduce global population-weighted average surface concentrations by 23–34% and 7–17% and avoid 0.6–4.4 and 0.04–0.52 million annual premature deaths globally in 2030. UNEP and WMO (2011) considered emission control measures to reduce ozone and black carbon (BC) and estimated that 2.4 million annual premature deaths (with a range of 0.7 to 4.6 million) from outdoor air pollution could be avoided. West et al. (2013) estimated global GHG mitigation brings co-benefits for air quality and would avoid 0.5±0.2, 1.3±0.5, and 2.2±0.8 million premature deaths in 2030, 2050, and 2100, respectively.

There are no global data on the impacts of management of invasive species / encroachment on adaptation.

Coastal wetlands provide a natural defence against coastal flooding and storm surges by dissipating wave energy, reducing erosion and by helping to stabilise shore sediments, so restoration may provide significant benefits for adaptation. The Ramsar Convention on Wetlands covers 1.5 Mkm^2 across 1674 sites Keddy et al. (2009). Coastal floods currently affect 93–310 million people (in 2010) globally, and this could rise to 600 million people in 2100 with sea level rise, unless adaptation measures are taken (Hinkel et al. 2014). The proportion of the flood-prone population that could avoid these impacts through restoration of coastal wetlands has not been quantified, but this sets an upper limit.

Avoided peat impacts and peatland restoration can help to regulate water flow and prevent downstream flooding (Munang et al. 2014), but the global potential (in terms of number of people who could avoid flooding through peatland restoration) has not been quantified.

There are no global estimates about the potential of biodiversity conservation to improve the adaptation and resilience of local communities to climate change, in terms of reducing the number of people affected by natural disasters. Nevertheless, it is widely recognised that biodiversity, ecosystem health and resilience improves the adaptation potential (Jones et al. 2012). For example, tree species mixture improves the resistance of stands to natural disturbances, such as drought, fires, and windstorms (Jactel et al. 2017), as well as stability against landslides (Kobayashi and Mori 2017). Moreover, Protected Areas play a key role for improving adaptation (Watson et al. 2014; Lopoukhine et al. 2012), through reducing water flow, stabilising rock movements, creating physical barriers to coastal erosion, improving resistance to fires, and buffering storm damages (Dudley et al. 2010). 33 out of 105 of the largest urban areas worldwide rely on protected areas for some, or all, of their drinking water (Secretariat of the Convention on Biological Diversity 2008), indicating that many millions are likely benefit from conservation practices.

Table 6.24 summarises the potentials for adaptation for soil-based response options, with confidence estimates based on the thresholds outlined in Table 6.53 in section 6.3.6, and indicative (not exhaustive) references upon which the evidence in based.

Table 6.24 Adaptation effects of response options based on land management of soils

Integrated response option	Potential	Confidence	Citation
Fire management	> 5.8 million people affected by wildfire; max. 0.5 million deaths per year by smoke	Medium confidence	Doerr and Santín 2016; Johnston et al. 2012; Shannon et al., 2016
Reduced landslides and natural hazards	>25 million people	Low confidence	Arnáez J et al. 2015; Gariano and Guzzetti 2016
Reduced pollution	Prevent 0.5–4.6 million annual	Medium	Anenberg et al. 2012;

including acidification	premature deaths globally	confidence	Shindell et al.; West et al. 2013; UNEP & WMO, 2011;
Management of invasive species / encroachment	No global estimates	No evidence	
Restoration and reduced conversion of coastal wetlands	up to 93-310 million people	Low confidence	Hinkel et al. 2014
Restoration and reduced conversion of peatlands	No global estimates	No evidence	
Biodiversity conservation	Likely many millions	Low confidence	CBD, 2008

6.3.2.1.5 Integrated response options based on land management specifically for CDR

Enhanced weathering of minerals has been proposed as a mechanism of improving soil health and food security (Beerling et al. 2018), but there is no literature estimating the global adaptation benefits.

Large-scale bioenergy and BECCS can require substantial amounts of cropland (Popp et al. 2017; Calvin et al. 2014a; Smith et al. 2016c), forestland (Baker et al. 2019b; Favero and Mendelsohn 2017), and water (Chaturvedi et al. 2013; Smith et al. 2016; Fuss et al. 2018; Popp et al. 2011; Hejazi et al. 2015b) suggesting that bioenergy and BECCS could have adverse side-effects for adaptation. In some contexts, e.g., low inputs of fossil fuels and chemicals, limited irrigation, heat/drought tolerant species, and using marginal land, bioenergy can have co-benefits for adaptation (Dasgupta et al. 2014; Noble et al. 2014). However, no studies were found that quantify the magnitude of the effect.

Table 6.25 summarises the impacts on adaptation of land management response options specifically for CDR, with confidence estimates based on the thresholds outlined in Table 6.53 in section 6.3.6, and indicative (not exhaustive) references upon which the evidence in based.

Table 6.25 Adaptation effects of response options based on land management specifically for CDR

Integrated response option	**Potential**	**Confidence**	**Citation**
Enhanced weathering of minerals	No global estimates	No evidence	
Bioenergy and BECCS	Potentially large negative consequences	Low confidence	Fuss et al. 2018b; Muller et al. 2017b; Smith et al.

6.3.2.2 Integrated response options based on value chain management

In this section, the impacts on climate change adaptation of integrated response options based on value chain management are assessed.

6.3.2.2.1 Integrated response options based on value chain management through demand management

Decreases in pressure on land and decreases in production intensity associated with sustainable healthy diets or reduced food waste could also benefit adaptation; however, the size of this effect is not well quantified (Muller et al. 2017b).

Reducing food waste losses can relieve pressure on the global freshwater resource, thereby aiding adaptation. Food losses account for 215 km^3 yr^{-1} of freshwater resources, which (Kummu et al. 2012) report to be about 12–15% of the global consumptive water use. Given that 35% of the global population is living under high water stress or shortage (Kummu et al. 2010), reducing food waste could benefit 320–400 million people (12–15% of the 2681 million people affected by water stress / shortage).

While no studies report quantitative estimates of the effect of material substitution on adaptation, the effects are expected to be similar to reforestation and afforestation if the amount of material substitution leads to an increase in forest area. Additionally, some studies indicate that wooden buildings, if properly constructed, could reduce fire risk compared to steel, which softens when burned (Gustavsson et al. 2006; Ramage et al. 2017a).

Table 6.26 summarises the impacts on adaptation of demand management options, with confidence estimates based on the thresholds outlined in Table 6.53 in section 6.3.6, and indicative (not exhaustive) references upon which the evidence in based.

Table 6.26 Adaptation effects of response options based on demand management

Integrated response option	Potential	Confidence	Citation
Dietary change	No global estimates	No evidence	Muller et al. 2017b
Reduced post-harvest losses	320-400 million people	Medium confidence	Kummu et al. 2012a
Reduced food waste (consumer or retailer)	No global estimates	No evidence	Muller et al. 2017b
Material substitution	No global estimates	No evidence	

6.3.2.2.2 Integrated response options based on value chain management through supply management

It is estimated that 500 million smallholder farmers depend on agricultural businesses in developing countries (World Bank, 2017), meaning that better promotion of value-added products and improved efficiency and sustainability of food processing and retailing could potentially help up to 500 million people to adapt to climate change. However, figures on how sustainable sourcing in general could help farmers and forest management is mostly unquantified. More than 1 million farmers have currently been certified through various schemes (Tayleur et al. 2017), but how much this has helped them prepare for adaptation is unknown.

Management of supply chains has the potential to reduce vulnerability to price volatility. Consumers in lower income countries are most affected by price volatility, with sub-Saharan Africa and South Asia at highest risk (Regmi and Meade 2013; Fujimori et al. 2018a). However, understanding of the stability of food supply is one of the weakest links in global food system research (Wheeler and von Braun 2013) as instability is driven by a confluence of factors (Headey and Fan 2008). Food price spikes in 2007 increased the number of people under the poverty line by between 100 million people (Ivanic and Martin 2008) and 450 million people (Brinkman et al. 2009), and caused welfare losses of 3% or more for poor households in many countries (Zezza et al. 2009). Food price stabilisation by China, India and Indonesia alone in 2007/2008 led to reduced staple food price for 2 billion people (Timmer 2009). Presumably, spending less on food frees up money for other activities, including adaptation, but it is unknown how much (Zezza et al. 2009; Ziervogel and Ericksen 2010). One example of reduction in staple food price costs to consumers in Bangladesh from food stability policies saved rural households USD887 million total (Torlesse et al. 2003b). Food supply stability through improved supply chains also potentially reduces conflicts (by avoiding food price riots, which occurred in countries with over 100 million total in population in 2007/2008), and thus increases adaptation capacity (Raleigh et al. 2015a).

There are no global estimates of the contribution of improved food transport and distribution, or of urban food systems, in contributing to adaptation, but since the urban population in 2018 was 4.2 billion people, this sets the upper limit on those that could benefit.

Given that 65% (760 million) of poor working adults make a living through agriculture, increased energy efficiency in agriculture could benefit this 760 million people.

Table 6.27 summarises the impacts on adaptation of supply management options, with confidence estimates based on the thresholds outlined in Table 6.53 in section 6.3.6, and indicative (not exhaustive) references upon which the evidence in based.

Table 6.27 Adaptation effects of response options based on demand management

Integrated response option	Potential	Confidence	Citation
Sustainable sourcing	> 1 million	Low confidence	Tayleur et al. 2017
Management of supply chains	>100 million	Medium confidence	Ivanic and Martin 2008; Timmer 2009; Vermeulen et al. 2012b; Campbell et al. 2016;
Enhanced urban food systems	No global estimates	No evidence	
Improved food processing and retailing	500 million people	Low confidence	World Bank 2017
Improved energy use in food systems	760 million	Low confidence	World Bank 2017

6.3.2.3 Integrated response options based on risk management

In this section, the impacts on climate change adaptation of integrated response options based on risk management are assessed.

Reducing urban sprawl is likely to provide adaptation co-benefits *via* improved human health (Frumkin 2002; Anderson 2017), as sprawl contributes to reduced physical activity, worse air pollution, and exacerbation of urban heat island effects and extreme heat waves (Stone et al. 2010). The most sprawling cities in the US have experienced extreme heat waves more than double those of denser cities, and "urban albedo and vegetation enhancement strategies have significant potential to reduce heat-related health impacts" (Stone et al. 2010). Other adaption co-benefits are less well understood. There are likely to be cost savings from managing planning growth (one study found 2% savings in metropolitan budgets, which can be then spent on adaptation planning) (Deal and Schunk 2004).

Diversification is a major adaptation strategy and form of risk management, as it can help households smooth out income fluctuations and provide a broader range of options for the future (Osbahr et al. 2008; Adger et al. 2011; Thornton and Herrero 2014). Surveys of farmers in climate variable areas find that livelihood diversification is increasingly favoured as an adaptation option (Bryan et al. 2013), although it is not always successful, since it can increase exposure to climate variability (Adger et al. 2011). There are over 570 million small farms in the world (Lowder et al. 2016), and many millions of smallholder agriculturalists already practice livelihood diversification by engaging in multiple forms of off-farm income (Rigg 2006). It is not clear, however, how many farmers have not yet practiced diversification and thus how many would be helped by supporting this response option.

Currently, millions of farmers still rely to some degree on local seeds. Use of local seeds can facilitate adaptation for many smallholders, as moving to use of commercial seeds can increase costs for farmers (Howard 2015). Seed networks and banks protect local agrobiodiversity and landraces, which are important to facilitate adaptation, as local landraces may be resilient to some forms of climate change (Coomes et al. 2015a; van Niekerk and Wynberg 2017a; Vasconcelos et al. 2013).

Disaster risk management is an essential part of adaptation strategies. The Famine Early Warning System funded by the USAID has operated across 3 continents since the 1980s, and many millions of people across 34 countries have access to early information on drought. Such information can assist communities and households in adapting to onset conditions (Hillbruner and Moloney 2012).

However, concerns have been raised as to how many people are actually reached by disaster risk management and early warning systems; for example, less than 50% of respondents in Bangladesh had heard a cyclone warning before it hit, even though an early warning system existed (Mahmud and Prowse 2012). Further, there are concerns that current early warning systems "tend to focus on response and recovery rather than on addressing livelihood issues as part of the process of reducing underlying risk factors," (Birkmann et al. 2015a), leading to less adaptation potential being realised.

Local risk sharing instruments like rotating credit or loan groups can help buffer farmers against climate impacts and help facilitate adaptation. Both index and commercial crop insurance offers some potential for adaptation, as it provides a means of buffering and transferring weather risk, saving farmers the cost of crop losses (Meze-Hausken et al. 2009; Patt et al. 2010). However, overly subsidised insurance can undermine the market's role in pricing risks and thus depress more rapid adaptation strategies (Skees and Collier 2012; Jaworski 2016) and increase the riskiness of decision-making (McLeman and Smit 2006). For example, availability of crop insurance was observed to reduce farm-level diversification in the US, a factor cited as increasing adaptive capacity (Sanderson et al. 2013b) and crop insurance-holding soybean farmers in the US have been less likely to adapt to extreme weather events than those not holding insurance (Annan and Schlenker 2015). It is unclear how many people worldwide use insurance as an adaptation strategy; (Platteau et al. 2017) suggest less than 30% of smallholders take out any form of insurance), but it is likely in the millions.

Table 6.28 summarises the impacts on adaptation of risk management options, with confidence estimates based on the thresholds outlined in Table 6.53 in section 6.3.6, and indicative (not exhaustive) references upon which the evidence in based.

Table 6.28 Adaptation effects of response options based on risk management

Integrated response option	Potential	Confidence	Citation
Management of urban sprawl	Unquantified but likely to be many millions	Low confidence	Stone et al. 2010
Livelihood diversification	>100 million likely	Low confidence	Morton 2007; Rigg 2006
Use of local seeds	Unquantified but likely to be many millions	Low confidence	Louwaars 2002; Santilli 2012
Disaster risk management	>100 million	High confidence	Hillbruner and Moloney 2012
Risk sharing instruments	Unquantified but likely to be several million	Low confidence	Platteau et al. 2017

6.3.3 Potential of the integrated response options for addressing desertification

In this section, the impacts of integrated response options on desertification are assessed.

6.3.3.1 Integrated response options based on land management

In this section, the impacts on desertification of integrated response options based on land management are assessed.

6.3.3.1.1 Integrated response options based on land management in agriculture

Burney et al. (2010) estimated that an additional global cropland area of 11.11–15.14 Mkm^2 would have been needed if productivity had not increased between 1961 and 2000. Given that agricultural expansion is a main driver of desertification (FAO et al. 2015), increased food productivity could have prevented up to 11.11–15.14 Mkm^2 from exploitation and desertification (Table 6.10).

Improved cropland, livestock and grazing land management are strategic options aiming at prevention of desertification, and may include crop and animal selection, optimised stocking rates, changed tillage and/or cover crops, to land use shifting from cropland to rangeland, in general targeting increases in ground cover by vegetation, and protection against wind erosion (Schwilch et al. 2014; Bestelmeyer et al. 2015). Considering the widespread distribution of deserts and desertified lands globally, more than 10 Mkm^2 could benefit from improved management techniques.

Agroforestry can help stabilise soils to prevent desertification (Section 6.3.2.1.1), so given that there are is around 10 Mkm^2 of land with more than 10% tree cover (Garrity, 2012), agroforestry could benefit up to 10 Mkm^2 of land.

Agricultural diversification to prevent desertification may include the use of crops with manures, legumes, fodder legumes and cover crops combined with conservation tillage systems (Schwilch et al. 2014). These practices can be considered to be part of improved crop management options (see above) and aim at increasing ground coverage by vegetation and controlling wind erosion losses.

Since shifting from grassland to the annual cultivation of crops increases erosion and soil loss, there are significant benefits for desertification control, by stabilising soils in arid areas (Chapter 3). Cropland expansion during 1985 to 2005 was 359 thousand km^2, or 17.4 thousand $km^2\ yr^{-1}$ (Foley et al. 2011). Not all of this expansion will be from grasslands or in desertified areas, but this value sets the maximum contribution of prevention of conversion of grasslands to croplands, a small global benefit for desertification control (Table 6.10).

Integrated water management strategies such as water-use efficiency and irrigation, improve soil health through increase in soil organic matter content, thereby delivering benefits for prevention or reversal of desertification (Chapter 3; Baumhardt et al. 2015; Datta et al. 2000; Evans and Sadler 2008; He et al. 2015). Climate change will amplify existing stress on water availability and on agricultural systems, particularly in semi-arid environments (AR5; Chapter 3). In 2011, semiarid ecosystems in the southern hemisphere contributed 51% of the global net carbon sink (Poulter et al., 2014). These results suggest that arid ecosystems could be an important global carbon sink, depending on soil water availability.

Table 6.29 summarises the impacts on desertification of agricultural options, with confidence estimates based on the thresholds outlined in Table 6.53 in section 6.3.6, and indicative (not exhaustive) references upon which the evidence in based.

Table 6.29 Effects on desertification of response options in agriculture

Integrated response option	Potential	Confidence	Citation
Increased food productivity	11.1-15.1 Mkm^2	Low confidence	Burney et al. 2010
Improved cropland management	10 Mkm^2	Low confidence	Schwilch et al. 2014
Improved grazing land management	0.5-3 Mkm^2	Low confidence	Schwilch et al. 2014
Improved livestock management	0.5-3 Mkm^2	Low confidence	Miao et al. 2015; Squires and Karami 2005
Agroforestry	10 Mkm^2 (with >10% tree cover)	Medium confidence	Garrity (2012)
Agricultural diversification	0.5-3 Mkm^2	Low confidence	Lambin and Meyfroidt 2011; Schwilch et al. 2014
Reduced grassland conversion to cropland	up to 17.4 thousand $km^2\ yr^{-1}$	Low confidence	Foley et al. 2011
Integrated water management	10 thousand km^2	Low confidence	Pierzynski et al., 2017; UNCCD, 2011

6.3.3.1.2 Integrated response options based on land management in forestry

Forests are important to help to stabilise land and regulate water and microclimate (Locatelli et al. 2015c). Based on the extent of dry forest at risk of desertification (Núñez et al. 2010; Bastin et al.

2017), the estimated global potential effect for avoided desertification is large for both improved forest management and for reduced deforestation and forest degradation when cumulated for at least 20 years (Table 6.30). The uncertainty of these global estimates is high. More robust qualitative and some quantitative estimates are available at regional level. For example, it has been simulated that human activity (i.e., land management) contributed to 26% of the total land reverted from desertification in Northern China between 1981 and 2010 (Xu et al. 2018). In Thailand, it was found that the desertification risk is reduced when the land use is changed from bare lands to agricultural lands and forests, and from non-forests to forests; conversely, the desertification risk increases when converting forests and denuded forests to bare lands (Wijitkosum 2016).

Afforestation, reforestation and forest restoration are land management response options that are used to prevent desertification. Forests tend to maintain water and soil quality by reducing runoff and trapping sediments and nutrients (Idris Medugu et al. 2010a; Salvati et al. 2014a), but planting of non-native species in semi-arid regions can deplete soil water resources if they have high evapotranspiration rates (Feng et al.; Yang et al.). Afforestation and reforestation programmes can be deployed over large areas of the Earth, so can create synergies in areas prone to desertification. Global estimates of land potentially available for afforestation are up to 25.8 Mkm^2 by the end of the century, depending on a variety of assumptions on socio-economic developments and climate policies (Griscom et al. 2017; Kreidenweis et al. 2016a; Popp et al. 2017). The higher end of this range is achieved under the assumption of a globally uniform reward for carbon uptake in the terrestrial biosphere, and it is halved by considering tropical and subtropical areas only to minimise albedo feedbacks (Kreidenweis et al. 2016a). When safeguards are introduced (e.g., excluding existing cropland for food security, boreal areas, etc.), the area available declines to about 6.8 Mkm^2 (95% confidence interval of 2.3 and 11.25 Mkm^2), of which about 4.72 Mkm^2 is in the tropics and 2.06 Mkm^2 is in temperate regions (Griscom et al. 2017a; Table 6.30).

Table 6.30 summarises the impacts on desertification of forestry options, with confidence estimates based on the thresholds outlined in Table 6.53 in section 6.3.6, and indicative (not exhaustive) references upon which the evidence in based.

Table 6.30 Effects on desertification of response options in forests

Integrated response option	Potential	Confidence	Citation
Improved forest management	> 3 Mkm^2	Low confidence	Bastin et al. 2017; Núñez et al. 2010
Reduced deforestation and degradation	> 3 Mkm^2 (effects cumulated for at least 20 years)	Low confidence	Bastin et al. 2017; Keenan et al. 2015; Núñez et al. 2010
Reforestation and forest restoration	See afforestation		
Afforestation	2-25.8 Mkm^2 by the end of the century	Medium confidence	Griscom et al. 2017a; Kreidenweis et al. 2016a; Popp et al. 2017

6.3.3.1.3 Integrated response options based on land management of soils

With over 2.7 billion people affected globally by desertification (IPBES 2018), practices to increase soil organic carbon content are proposed as actions to address desertification, and could be applied to an estimated 11.37 Mkm^2 of desertified soils (Lal 2001a; Table 6.31).

Control of soil erosion could have large benefits for desertification control. Using figures from (FAO et al. 2015), Scholes et al. (2018) estimated that land losses due to erosion to 2050 are equivalent to 1.5 Mkm^2 of land from crop production, or 45 thousand km^2 yr^{-1} (Foley et al. 2011), so soil erosion control could benefit up to 1.50 Mkm^2 of land in the coming decades. Lal (2001a) estimated that desertification control (using soil erosion control as one intervention) could benefit 11.37 Mkm^2 of desertified land globally (Table 6.10).

Oldeman et al. (1991) estimated the global extent soil affected by salinisation is 0.77 Mkm^2 yr^{-1}, which sets the upper limit on the area that could benefit from measures to address soil salinisation (Table 6.31).

In degraded arid grasslands, shrublands and rangelands, desertification can be reversed by alleviation of soil compaction through installation of enclosures and removal of domestic livestock (Allington et al. 2010), but there are no global estimates of potential (Tale 6.31).

Biochar could potentially deliver benefits in efforts to address desertification though improving water holding capacity (Woolf et al. 2010; Sohi 2012), but the global effect is not quantified.

Table 6.31 summarises the impacts on desertification of soil-based options, with confidence estimates based on the thresholds outlined in Table 6.53 in section 6.3.6, and indicative (not exhaustive) references upon which the evidence in based.

Table 6.31 Effects on desertification of land management of soils

Integrated response option	Potential	Confidence	Citation
Increased soil organic carbon content	Up to 11.37 Mkm^2	Medium confidence	Lal 2001a
Reduced soil erosion	Up to 11.37 Mkm^2	Medium confidence	Lal 2001a
Reduced soil salinisation	0.77 Mkm^2 yr^{-1}	Medium confidence	Oldeman et al. 1991
Reduced soil compaction	No global estimates	No evidence	FAO and ITPS 2015; Hamza and Anderson 2005b
Biochar addition to soil	No global estimates	No evidence	

6.3.3.1.4 Integrated response options based on land management across all/other ecosystems

For fire management, Arora and Melton (2018) estimated, using models and GFED4.1s0 data, that burned area over the 1997–2014 period was 4.834-4.855 Mkm^2 yr^{-1}. Randerson et al. (2012) estimated small fires increased total burned area globally by 35% from 3.45 to 4.64 Mkm^2 yr^{-1} during the period 2001–2010. Tansey et al. (2004) estimated over 3.5 Mkm^2 yr^{-1} of burned areas were detected in the year 2000 (Table 6.32).

Although slope and slope aspect are predictive factors of desertification occurrence, the factors with the greatest influence are land cover factors, such as normalised difference vegetation index (NDVI) and rangeland classes (Djeddaoui et al. 2017). Therefore, prevention of landslides and natural hazards exert indirect influence on the occurrence of desertification.

The global extent of chemical soil degradation (salinisation, pollution, and acidification) is about 1.03 Mkm^2 yr^{-1} (Oldeman et al. 1991), giving the maximum extent of land that could benefit from the management of pollution and acidification.

There are no global data on the impacts of management of invasive species / encroachment on desertification, though the impact is presumed to be positive. There are no studies examining the potential role of restoration and avoided conversion of coastal wetlands on desertification.

There are no impacts of peatland restoration for prevention of desertification, as peatlands occur in wet areas and deserts in arid areas, so they are not connected.

For management of pollution, including acidification, Oldeman et al. (1991) estimated global extent of chemical soil degradation, with 0.77 Mkm^2 yr^{-1} affected by salinisation, 0.21 Mkm^2 yr^{-1} affected by pollution, and 0.06 Mkm^2 yr^{-1} affected by pollution (total: 1.03 Mkm^2 yr^{-1}), so this is the area that could potentially benefit from pollution management measures.

Biodiversity conservation measures can interact with desertification, but the literature contains no global estimates of potential.

Table 6.32 summarises the impacts on desertification of options on all/other ecosystems, with confidence estimates based on the thresholds outlined in Table 6.53 in section 6.3.6, and indicative (not exhaustive) references upon which the evidence in based.

Table 6.32 Effects on desertification of response options on all/other ecosystems

Integrated response option	Potential	Confidence	Citation
Fire management	Up to 3.5-4.9 $Mkm^2\ yr^{-1}$	Medium confidence	Arora and Melton 2018; Randerson et al. 2012; Tansey et al. 2004
Reduced landslides and natural hazards	>0	Low confidence	Djeddaoui et al.; Noble et al. 2014
Reduced pollution including acidification	1.03 $Mkm^2\ yr^{-1}$	Low confidence	Oldeman et al. 1991
Management of invasive species / encroachment	No global estimates	No evidence	
Restoration and reduced conversion of coastal wetlands	No global estimates	No evidence	
Restoration and reduced conversion of peatlands	No impact		
Biodiversity conservation	No global estimates	No evidence	

6.3.3.1.5 Integrated response options based on land management specifically for CDR

While spreading of crushed minerals onto land as part of enhanced weathering may provide soil / plant nutrients in nutrient-depleted soils (Beerling et al. 2018), there is no literature reporting on the potential global impacts of this in addressing desertification.

Large-scale production of bioenergy can require significant amounts of land (Smith et al. 2016d; Clarke and Jiang 2014a; Popp et al. 2017), with as much as 15 Mkm^2 in 2100 in 2°C scenarios (Popp et al. 2017), increasing pressures for desertification (Table 6.33).

Table 6.33 summarises the impacts on desertification of options specifically for CDR, with confidence estimates based on the thresholds outlined in Table 6.53 in section 6.3.6, and indicative (not exhaustive) references upon which the evidence in based.

Table 6.33 Effects on desertification of response options specifically for CDR

Integrated response option	Potential	Confidence	Citation
Enhanced weathering of minerals	No global estimates	No evidence	
Bioenergy and BECCS	Negative impact on up to 15 Mkm^2	Medium confidence	Clarke et al. 2014a; Popp et al. 2017; Smith et al.

6.3.3.2 Integrated response options based on value chain management

In this section, the impacts on desertification of integrated response options based on value chain management are assessed.

6.3.3.2.1 Integrated response options based on value chain management through demand management

Dietary change and waste reduction both result in decreased cropland and pasture extent (Bajželj et al. 2014a; Stehfest et al. 2009; Tilman and Clark 2014), reducing the pressure for desertification (Table 6.34).

Reduced post-harvest losses could spare 1.98 Mkm^2 of cropland globally (Kummu et al. 2012). Not all of this land could be subject to desertification pressure, so this represents that maximum area that could be relieved from desertification pressure by reduction of post-harvest losses. No studies were found linking material substitution to desertification.

Table 6.34 summarises the impacts on desertification of demand management options, with confidence estimates based on the thresholds outlined in Table 6.53 in section 6.3.6, and indicative (not exhaustive) references upon which the evidence in based.

Table 6.34 Effects on desertification of response options based on demand management

Integrated response option	Potential	Confidence	Citation
Dietary change	0.80-5 Mkm^2	Low confidence	Alexander et al. 2016; Bajželj et al. 2014; Stehfest et al. 2009; Tilman and Clark 2014a
Reduced post-harvest losses	<1.98 Mkm^2	Low confidence	Kummu et al. 2012a
Reduced food waste (consumer or retailer)	1.4 Mkm^2	Low confidence	Bajželj et al. 2014
Material substitution	No global estimates	No evidence	

6.3.3.2.2 Integrated response options based on value chain management through supply management

There are no global estimates of the impact on desertification of sustainable sourcing, management of supply chains, enhanced urban food systems, improved food processing, or improved energy use in agriculture.

Table 6.35 summarises the impacts on desertification of supply management options, with confidence estimates based on the thresholds outlined in Table 6.53 in section 6.3.6, and indicative (not exhaustive) references upon which the evidence in based.

Table 6.35 Effects on desertification of response options based on supply management

Integrated response option	Potential	Confidence	Citation
Sustainable sourcing	No global estimates	No evidence	
Management of supply chains	No global estimates	No evidence	
Enhanced urban food systems	No global estimates	No evidence	
Improved food processing and retailing	No global estimates	No evidence	
Improved energy use in food systems	No global estimates	No evidence	

6.3.3.3 Integrated response options based on risk management

In this section, the impacts on desertification of integrated response options based on risk management are assessed.

There are regional case studies of urban sprawl contributing to desertification in Mediterranean climates in particular (Barbero-Sierra et al. 2013b; Stellmes et al. 2013), but no global figures.

Diversification may deliver some benefits for addressing desertification when it involves greater use of tree crops that may reduce the need for tillage (Antwi-Agyei et al. 2014). Many anti-desertification programmes call for diversification (Stringer et al. 2009), but there is little evidence on how many households had done so (Herrmann and Hutchinson 2005). There are no numbers for global impacts.

The literature is unclear on whether the use of local seeds has any relationship to desertification, although some local seeds are likely more adapted to arid climates and less likely to degrade land than commercial introduced varieties (Mousseau 2015). Some anti-desertification programmes have also shown more success using local seed varieties (Bassoum and Ghiggi 2010; Nunes et al. 2016).

Some disaster risk management approaches can have impacts on reducing desertification, like the Global Drought Early Warning System (GDEWS) (currently in development), which will monitor precipitation, soil moisture, evapotranspiration, river flows, groundwater, agricultural productivity and natural ecosystem health. It may have some potential co-benefits to reduce desertification (Pozzi et al. 2013). However, there are no figures yet for how much land area will be covered by such early warning systems.

Risk sharing instruments, like pooling labour or credit, could help communities invest in anti-desertification actions, but evidence is missing. Commercial crop insurance is likely to deliver no co-benefits for prevention and reversal of desertification, as evidence suggests that subsidised insurance, in particular, can increase crop production in marginal lands. Crop insurance could have been responsible for shifting up to 0.9% of rangelands to cropland in the Upper US Midwest (Claassen et al. 2011).

Table 6.36 summarises the impact on desertification for options based on risk management, with confidence estimates based on the thresholds outlined in Table 6.53 in section 6.3.6, and indicative (not exhaustive) references upon which the evidence in based.

Table 6.36 Effects on desertification of response options based on risk management

Integrated response option	Potential	Confidence	Citation
Management of urban sprawl	>5 thousand km^2	Low confidence	Barbero-Sierra et al. 2013b
Livelihood diversification	No global estimates	Low confidence	Herrmann and Hutchinson 2005
Use of local seeds	No global estimates	No evidence	
Disaster risk management	No global estimates	No evidence	Pozzi et al. 2013a
Risk sharing instruments	Likely negative impacts but not quantified	Low confidence	Claassen et al. 2011

6.3.4 Potential of the integrated response options for addressing land degradation

In this section, the impacts of integrated response options on land degradation are assessed.

6.3.4.1 Integrated response options based on land management

In this section, the impacts on land degradation of integrated response options based on land management are assessed.

6.3.4.1.1 Integrated response options based on land management in agriculture

Burney et al. (2010) estimated that an additional global cropland area of 11.11–15.14 Mkm^2 would have been needed if productivity had not increased between 1961 and 2000. As for desertification, given that agricultural expansion is a main driver of land degradation (FAO and ITPS 2015), increased food productivity has prevented up to 11.11–15.14 Mkm^2 from exploitation and land degradation (Table 6.37).

Land degradation can be addressed by the implementation of improved cropland, livestock and grazing land management practices, such as those outlined in the recently published Voluntary Guidelines for Sustainable Soil Management (FAO 2017b). Each one could potentially affect extensive surfaces, not less than 10 Mkm^2. The Guidelines include a list of practices aiming at minimising soil erosion, enhancing soil organic matter content, fostering soil nutrient balance and cycles, preventing, minimising and mitigating soil salinisation and alkalinisation, soil contamination, soil acidification, and soil sealing, soil compaction, and improving soil water management. Land cover and land cover change are key factors and indicators of land degradation. In many drylands, land cover is threatened by overgrazing, so management of stocking rate and grazing can help to prevent the advance of land degradation Smith et al. (2016c).

Agroforestry can help stabilise soils to prevent land degradation, so given that there are is around 10 Mkm^2 of land with more than 10% tree cover (Garrity, 2012), agroforestry could benefit up to 10 Mkm^2 of land.

Agricultural diversification usually aims at increasing climate and food security resilience, such as under "climate smart agriculture" approaches (Lipper et al. 2014). Both objectives are closely related to land degradation prevention, potentially affecting 1–5 Mkm^2.

Shifting from grassland to tilled crops increases erosion and soil loss, so there are significant benefits for addressing land degradation, by stabilising degraded soils (Chapter 3). Since cropland expansion during 1985 to 2005 was 17.4 thousand km^2 yr^{-1} (Foley et al.,2009), and not all of this expansion will be from grasslands or degraded land, the maximum contribution of prevention of conversion of grasslands to croplands is 17.4 thousand km^2 yr^{-1}, a small global benefit for control of land degradation (Tale 6.37).

Most land degradation processes that are sensitive to climate change pressures (e.g. erosion, decline in soil organic matter, salinisation, waterlogging, drying of wet ecosystems) can benefit from integrated water management. Integrated water management options include management to reduce aquifer and surface water depletion, and to prevent over extraction, and provide direct co-benefits for prevention of land degradation. Land management practices implemented for climate change mitigation may also affect water resources. Globally, water erosion is estimated to result in the loss of 23–42 MtN and 14.6–26.4 MtP annually (Pierzynski et al., 2017). Forests influence the storage and flow of water in watersheds (Eisenbies et al. 2007) and are therefore important for regulating how climate change will impact landscapes.

Table 6.37 summarises the impact on land degradation of options in agriculture, with confidence estimates based on the thresholds outlined in Table 6.53 in section 6.3.6, and indicative (not exhaustive) references upon which the evidence in based.

Table 6.37 Effects on land degradation of response options in agriculture

Integrated response option	Potential	Confidence	Citation
Increased food productivity	11.11-15.14 Mkm^2	Medium confidence	Burney et al. 2010
Improved cropland management	10 Mkm^2	Low confidence	Lal 2015; Smith et al. 2016c
Improved grazing land management	10 Mkm^2	Low confidence	Smith et al. 2016c
Improved livestock management	10 Mkm^2	Low confidence	Lal 2015; Smith et al. 2016c
Agroforestry	10 Mkm^2 (with >10% tree cover)	Medium confidence	Garrity 2012
Agricultural diversification	1-5 Mkm^2	Medium confidence	Lambin and Meyfroidt 2011
Reduced grassland conversion to cropland	Up to 17.4 thousand km^2 yr^{-1}	Low confidence	Foley et al. 2011
Integrated water management	0.01 Mkm^2	Medium confidence	Pierzynski et al., 2017; UNCCD, 2011

6.3.4.1.2 Integrated response options based on land management in forestry

Based on the extent of forest exposed to degradation (Gibbs and Salmon 2015), the estimated global potential effect for reducing land degradation, e.g. through reduced soil erosion (Borrelli et al. 2017), is large for both improved forest management and for reduced deforestation and forest degradation when cumulated for at least 20 years (Table 6.38) The uncertainty of these global estimates is high. More robust qualitative and some quantitative estimates are available at regional level. For example, in Indonesia, Santika et al. (2017) demonstrated that reduced deforestation (Sumatra and Kalimantan islands) contributed to reduce significantly land degradation.

Forest restoration is a key option to achieve the overarching frameworks to reduce land degradation at global scale, such as for example, Zero Net Land Degradation (ZNLD; UNCCD 2012) and Land Degradation Neutrality (LDN), not only in drylands (Safriel 2017). Indeed, it has been estimated that more than 20 Mkm^2 are suitable for forest and landscape restoration, of which 15 Mkm^2 may be devoted to mosaic restoration (UNCCD 2012). Moreover, the Bonn Challenge aims to restore 1.5 Mkm^2 of deforested and degraded land by 2020, and 3.5 Mkm^2 by 2030 (http://www.bonnchallenge.org/content/challenge). Under a restoration and protection scenario (implementing restoration targets), Wolff et al. (2018) simulated that there will be a global increase in net tree cover of about 4 Mkm^2 by 2050 (Table 6.38). At local level, Brazil's Atlantic Restoration Pact aims to restore 0.15 Mkm^2 of forest areas in 40 years (Melo et al. 2013). The Y Ikatu Xingu campaign (launched in 2004) aims to contain deforestation and degradation processes by reversing the liability of 3 thousand km^2 in the Xingu Basin, Brazil (Durigan et al. 2013).

Afforestation and reforestation are land management options frequently used to address land degradation (see Section 6.3.3.1.2 for details; Table 6.38).

Table 6.38 summarises the impact on land degradation of options in forestry, with confidence estimates based on the thresholds outlined in Table 6.53 in section 6.3.6, and indicative (not exhaustive) references upon which the evidence in based.

Table 6.38 Effects on land degradation of response options in forestry

Integrated response option	Potential	Confidence	Citation
Improved forest management	> 3 Mkm^2	Low confidence	Gibbs and Salmon 2015
Reduced deforestation and degradation	> 3 Mkm^2 (effects cumulated for at least 20 years)	Low confidence	Gibbs and Salmon 2015; Keenan et al. 2015
Reforestation and forest restoration	20 Mkm^2 suitable for restoration > 3 Mkm^2 by 2050 (net increase in tree cover for forest restoration)	Medium confidence	UNCCD 2012; Wolff et al. 2018
Afforestation	2-25.8 Mkm^2 by the end of the century	Low confidence	Griscom et al. 2017a; Kreidenweis et al. 2016a; Popp et al. 2017

6.3.4.1.3 Integrated response options based on land management of soils

Increasing soil organic matter content is a measure to address land degradation. With around 120 thousand km^2 lost to degradation every year, and over 3.2 billion people negatively impacted by land degradation globally (IPBES 2018), practices designed to increase soil organic carbon have a large potential to address land degradation, estimated to affect over 11 Mkm^2 globally (Lal, 2004; Table 6.39).

Control of soil erosion could have large benefits for addressing land degradation. Soil erosion control could benefit up to 1.50 Mkm^2 of land to 2050 (IPBES 2018). Lal (2004) suggested interventions to prevent wind and water erosion (two of the four main interventions proposed to address land degradation), could restore 11 Mkm^2 of degraded and desertified soils globally (Table 6.39).

Oldeman et al. (1991) estimated the global extent soil affected by salinisation is 0.77 Mkm^2 yr^{-1}, which sets the upper limit on the area that could benefit from measures to address soil salinisation

(Table 6.39). The global extent of chemical soil degradation (salinisation, pollution, and acidification) is about 1.03 Mkm^2 (Oldeman et al. 1991) giving the maximum extent of land that could benefit from the management of pollution and acidification.

Biochar could provide moderate benefits for the prevention or reversal of land degradation, by improving water holding capacity, improving nutrient use efficiency, managing heavy metal pollution and other co-benefits (Sohi 2012), though the global effects are not quantified.

Table 6.39 summarises the impact on land degradation of soil-based options, with confidence estimates based on the thresholds outlined in Table 6.53 in section 6.3.6, and indicative (not exhaustive) references upon which the evidence in based.

Table 6.39 Effects on land degradation of soil-based response options

Integrated response option	Potential	Confidence	Citation
Increased soil organic carbon content	11 Mkm^2	Medium confidence	Lal 2004
Reduced soil erosion	11 Mkm^2	Medium confidence	Lal 2004
Reduced soil salinisation	0.77 Mkm^2 yr^{-1}	Medium confidence	Qadir et al. 2013a; FAO 2016;
Reduced soil compaction	10 Mkm^2	Low confidence	FAO and ITPS 2015; Hamza and Anderson 2005a
Biochar addition to soil	Positive but not quantified globally	Low confidence	Chapter 4

6.3.4.1.4 Integrated response options based on land management across all/other ecosystems

For fire management, details of estimates of the impact of wildfires (and thereby the potential impact of their suppression) are given in Section 6.3.3.1.4 (Table 6.40).

Management of landslides and natural hazards aims at controlling a severe land degradation process affecting sloped and hilly areas, many of them with poor rural inhabitants (FAO et al. 2015; Gariano and Guzzetti 2016b), but the global potential has not been quantified.

There are no global data on the impacts of management of invasive species / encroachment on land degradation, though the impact is presumed to be positive.

Since large areas of coastal wetlands are degraded, restoration could potentially deliver moderate benefits for addressing land degradation, with 0.29 Mkm^2 globally considered feasible for restoration (Griscom et al. 2017a; Table 6.40).

Considering that large areas (0.46 Mkm^2) of global peatlands are degraded and considered suitable for restoration (Griscom et al. 2017), peatland restoration could deliver moderate benefits for addressing land degradation (Table 6.40).

There are no global estimates of the effects of biodiversity conservation on reducing degraded lands. However, at local scale, biodiversity conservation programmes have been demonstrated to stimulate gain of forest cover over large areas over the last three decades (e.g. in China; Zhang et al. 2013). Management of wild animals can influence land degradation processes by grazing, trampling and compacting soil surfaces, thereby altering surface temperatures and chemical reactions affecting sediment and carbon retention (Cromsigt et al. 2018).

Table 6.40 summarises the impact on land degradation of options in all/other ecosystems, with confidence estimates based on the thresholds outlined in Table 6.53 in section 6.3.6, and indicative (not exhaustive) references upon which the evidence in based.

Table 6.40 Effects on land degradation of response options in all/other ecosystems

Integrated response option	Potential	Confidence	Citation
Fire management	Up to 3.5-4.9 $Mkm^2\ yr^{-1}$	Medium confidence	Arora and Melton 2018; Randerson et al. 2012; Tansey et al. 2004
Reduced landslides and natural hazards	1-5 Mkm^2	Low confidence	FAO and ITPS 2015; Gariano and Guzzetti 2016
Reduced pollution including acidification	~1.03 Mkm^2	Low confidence	Oldeman et al. 1991
Management of invasive species / encroachment	No global estimates	No evidence	
Restoration and reduced conversion of coastal wetlands	0.29 Mkm^2	Medium confidence	Griscom et al. 2017a
Restoration and reduced conversion of peatlands	0.46 Mkm^2	Medium confidence	Griscom et al. 2017a
Biodiversity conservation	No global estimates	No evidence	

6.3.4.1.5 Integrated response options based on land management specifically for CDR

While spreading of crushed minerals onto land as part of enhanced weathering can provide soil / plant nutrients in nutrient-depleted soils, can increase soil organic carbon stocks and can help to replenish eroded soil (Beerling et al. 2018), there is no literature on the global potential for addressing land degradation.

Large-scale production of bioenergy can require significant amounts of land (Smith et al. 2016c; Clarke and Jiang 2014b; Popp et al. 2017), much as 15 Mkm^2 in 2°C scenarios (Popp et al. 2017), increasing pressures for land conversion and land degradation (Table 6.13). However, bioenergy production can either increase (Robertson et al. 2017c; Mello et al. 2014a) or decrease (FAO 2011; Lal 2014) soil organic matter, depending on where it is produced and how it is managed. These effects are not included in the quantification in Table 6.41.

Table 6.41 summarises the impact on land degradation of options specifically for CDR, with confidence estimates based on the thresholds outlined in Table 6.53 in section 6.3.6, and indicative (not exhaustive) references upon which the evidence in based.

Table 6.41 Effects on land degradation of response options specifically for CDR

Integrated response option	Potential	Confidence	Citation
Enhanced weathering of minerals	Positive but not quantified	Low confidence	Beerling et al. 2018
Bioenergy and BECCS	Negative impact on up to 15 Mkm^2	High confidence	Clarke et al. 2014a; Popp et al. 2017; Smith et al. 2016c

6.3.4.2 Integrated response options based on value chain management

In this section, the impacts on land degradation of integrated response options based on value change management are assessed.

6.3.4.2.1 Integrated response options based on value chain management through demand management

Dietary change and waste reduction both result in decreased cropland and pasture extent (Bajželj et al. 2014; Stehfest et al. 2009; Tilman and Clark 2014a), reducing the pressure for land degradation (Table 6.15). Reduced post-harvest losses could spare 1.98 Mkm^2 of cropland globally (Kummu et al. 2012) meaning that land degradation pressure could be relieved from this land area through reduction of post-harvest losses. The effects of material substitution on land degradation depend on management practice; some forms of logging can lead to increased land degradation (Chapter 4).

Table 6.42 summarises the impact on land degradation of demand management options, with confidence estimates based on the thresholds outlined in Table 6.53 in section 6.3.6, and indicative (not exhaustive) references upon which the evidence in based.

Table 6.42 Effects on land degradation of response options based on demand management

Integrated response option	Potential	Confidence	Citation
Dietary change	4-28 Mkm^2	High confidence	Alexander et al. 2016; Bajželj et al. 2014; Stehfest et al. 2009; Tilman and Clark 2014a
Reduced post-harvest losses	1.98 Mkm^2	Medium confidence	Kummu et al. 2012a
Reduced food waste (consumer or retailer)	7 Mkm^2	Medium confidence	Bajželj et al. 2014
Material substitution	No global estimates	No evidence	

6.3.4.2.2 Integrated response options based on value chain management through supply management

There are no global estimates of the impact on land degradation of enhanced urban food systems, improved food processing, retailing, or improved energy use in food systems.

There is evidence that sustainable sourcing could reduce land degradation, as the explicit goal of sustainable certification programs is often to reduce deforestation or other unsustainable land uses. Over 4 Mkm^2 of forests are certified for sustainable harvesting (PEFC/FSC 2018), although it is not clear if all these lands would be at risk of degradation without certification. While the food price instability of 2007/2008 increased financial investment in crop expansion (especially through so-called land grabbing), and thus better management of supply chains might have reduced this amount, no quantification of the total amount of land acquired, nor the possible impact of this crop expansion on degradation, has been recorded (McMichael and Schneider 2011a; McMichael 2012).

Table 6.43 summarises the impact on land degradation of supply management options, with confidence estimates based on the thresholds outlined in Table 6.53 in section 6.3.6, and indicative (not exhaustive) references upon which the evidence in based.

Table 6.43 Effects on land degradation of response options based on supply management

Integrated response option	Potential	Confidence	Citation
Sustainable sourcing	>4 Mkm^2	Low confidence	Auld et al. 2008
Management of supply chains	No global estimates	No evidence	
Enhanced urban food systems	No global estimates	No evidence	
Improved food processing and retailing	No global estimates	No evidence	
Improved energy use in food systems	No global estimates	No evidence	

6.3.4.3 Integrated response options based on risk management

In this section, the impacts on land degradation of integrated response options based on risk management are assessed.

Urban expansion has been identified as a major culprit in soil degradation in some countries; for example, urban expansion in China has now affected 0.2 Mkm^2, or almost one-sixth of the cultivated land total, causing an annual grain yield loss of up to 10 Mt, or around 5-6% of cropland production. Cropland production losses of 8-10% by 2030 are expected under model scenarios of urban expansion (Bren d'Amour et al. 2016). Pollution from urban development has included water and soil pollution from industry and wastes and sewage as well as acid deposition from increasing energy use in cities (Chen 2007a), all resulting in major losses to Nature's Contributions to People from urban conversion (Song and Deng 2015). Soil sealing from urban expansion is a major loss of soil productivity across many areas. The World Bank has estimated that new city dwellers in developing countries will require 160–500 m^2 per capita, converted from non-urban to urban land (Barbero-Sierra et al. 2013a; Angel et al 2005).

Degradation can be a driver leading to livelihood diversification (Batterbury 2001; Lestrelin and Giordano 2007). Diversification has the potential to deliver some reversal of land degradation, if diversification involves adding non-traditional crops or trees that may reduce the need for tillage (Antwi-Agyei et al. 2014). China's Sloping Land conversion programme has had livelihood diversification benefits and is said to have prevented degradation of 93 thousand km^2 of land (Liu et al. 2015). However, Warren (2002) provides conflicting evidence that more diverse-income households had increased degradation on their lands in Niger, and Palacios et al. (2013) associate landscape fragmentation with increased livelihood diversification in Mexico.

Use of local seeds may play a role in addressing land degradation due to the likelihood of local seeds being less dependent on inputs such as chemical fertilisers or mechanical tillage; for example, in India, local legumes are retained in seed networks while commercial crops like sorghum and rice dominate food markets (Reisman 2017a). However, there are no global figures.

Disaster Risk Management systems can have some positive impacts on prevention and reversal of land degradation, like the Global Drought Early Warning System (see section 6.3.3.3) (Pozzi et al. 2013).

Risk sharing instruments could have benefits for reduced degradation, but there are no global estimates. Commercial crop insurance is likely to deliver no co-benefits for prevention and reversal of degradation. One study found a 1% increase in farm receipts generated from subsidised farm programmes (including crop insurance and others) increased soil erosion by 0.3 t ha^{-1} (Goodwin and Smith 2003**)**. Wright and Wimberly (2013) found a 5310 km^2 decline in grasslands in the Upper Midwest of the US during 2006-2010 due to crop conversion driven by higher prices and access to insurance.

Table 6.44 summarises the impact on land degradation of risk management options, with confidence estimates based on the thresholds outlined in Table 6.53 in section 6.3.6, and indicative (not exhaustive) references upon which the evidence in based.

Table 6.44 Effects on land degradation of response options based on risk management

Integrated response option	Potential	Confidence	Citation
Management of urban sprawl	>0.2 Mkm^2	Medium confidence	Chen 2007b; Zhang 2000
Livelihood diversification	>0.1 Mkm^2	Low confidence	Liu and Lan 2015
Use of local seeds	No global estimates	No evidence	
Disaster risk management	No global estimates	No evidence	Pozzi et al. 2013
Risk sharing instruments	Variable, but negative impact on >5 thousand km^2 in Upper Midwest USA	Low confidence	Goodwin and Smith 2003; Wright and Wimberly 2013

6.3.5 Potential of the integrated response options for addressing food security

In this section, the impacts of integrated response options on food security are assessed.

6.3.5.1 Integrated response options based on land management

In this section, the impacts on food security of integrated response options based on land management are assessed.

6.3.5.1.1 Integrated response options based on land management in agriculture

Increased food productivity has fed many millions of people, who could not have otherwise been fed. Erisman et al. (2008) estimated that over 3 billion people worldwide could not have been fed without increased food productivity arising from N fertilisation (Table 6.45).

Improved cropland management to achieve food security aims at closing yield gaps by increasing use efficiency of essential inputs such as water and nutrients. Large production increases (45–70% for most crops) are possible from closing yield gaps to 100% of attainable yield, by increasing fertiliser use and irrigation, but overuse of nutrients could cause adverse environmental impacts (Mueller et al. 2012). This improvement can impact 1000 million people.

Improved grazing land management includes grasslands, rangelands and shrublands, and all sites on which pastoralism is practiced. In general terms, continuous grazing may cause severe damage to topsoil quality, through e.g. compaction. This damage may be reversed by short grazing exclusion periods under rotational grazing systems (Greenwood and McKenzie 2001; Drewry 2006; Taboada et al. 2011). Due to the widespread diffusion of pastoralism, improved grassland management may potentially affect more than 1000 million people, many of them under subsistence agricultural systems.

Meat, milk, eggs, and other animal products, including fish and other seafoods, will play an important role in achieving food security (Reynolds et al. 2015). Improved livestock management with different animal types and feeds may also impact one million people (Herrero et al. 2016). Ruminants are efficient converters of grass into human edible energy and protein and grassland-based food production can produce food with a comparable carbon footprint to mixed systems (O'Mara 2012b). However, in the future, livestock production will increasingly be affected by competition for natural resources, particularly land and water, competition between food and feed and by the need to operate in a carbon-constrained economy (Thornton et al. 2009a).

Currently, over 1.3 billion people are on degrading agricultural land, and the combined impacts of climate change and land degradation could reduce global food production by 10% by 2050. Since agroforestry could help to address land degradation, up to 1.3 billion people could benefit in terms of food security through agroforestry.

Agricultural diversification is not always economically viable; technological, biophysical, educational, and cultural barriers may emerge that limit the adoption of more diverse farming systems by farmers (Section 6.4.1). Nevertheless, diversification could benefit 1000 million people, many of them under subsistence agricultural systems (Birthal et al. 2015; Massawe et al. 2016; Waha et al. 2018).

Cropland expansion during 1985 to 2005 was 17 thousand km^2 yr^{-1} (Foley et al. 2005). Given that cropland productivity (global average of 250 kg protein ha^{-1} yr^{-1} for wheat; (Clark and Tilman 2017) is greater than that of grassland (global average of about 10 kg protein ha^{-1} yr^{-1} for beef/mutton; (Clark and Tilman 2017), prevention of this conversion to cropland would have led to a loss of about 0.4 Mt protein yr^{-1} globally. Given an average protein consumption in developing countries of 25.5 kg protein yr^{-1} (equivalent to 70g $person^{-1}$ day^{-1}; FAO, 2018), this is equivalent to the protein consumption of 16.4 million people each year (Table 6.45).

Integrated water management provides direct benefits to food security by improving agricultural productivity (Chapter 5; Tilman et al. 2011; Godfray and Garnett 2014), thereby potentially impacting the livelihood and well-being of >1000 million people (Campbell et al. 2016) affected by hunger and highly impacted by climate change. Increasing water availability and reliable supply of water for agricultural production using different techniques of water harvesting, storage, and its judicious utilisation through farm ponds, dams, and community tanks in rainfed agriculture areas have been presented by Rao (2017) and Rivera-Ferre et al. (2016).

Table 6.45 summarises the impact on food security of options in agriculture, with confidence estimates based on the thresholds outlined in Table 6.53 in section 6.3.6, and indicative (not exhaustive) references upon which the evidence in based.

Table 6.45 Effects on food security of response options in agriculture

Integrated response option	Potential	Confidence	Citation
Increased food productivity	3000 million people	High confidence	Erisman et al. 2008
Improved cropland management	>1000 million people	Low confidence	Campbell et al. 2014; Lipper et al. 2014
Improved grazing land management	>1000 million people	Low confidence	Herrero et al. 2016
Improved livestock management	>1000 million people	Low confidence	Herrero et al. 2016
Agroforestry	Up to 1300 million people	Low confidence	Sasha et al. 2018
Agricultural diversification	>1000 million people	Low confidence	Birthal et al. 2015; Massawe et al. 2016; Waha et al. 2018
Reduced grassland conversion to cropland	Negative impact on 16.4 million people	Low confidence	Clark and Tilman 2017; FAO, 2018
Integrated water management	>1000 million people	High confidence	Campbell et al. 2016

6.3.5.1.2 Integrated response options based on land management in forestry

Forests play a major role in providing food to local communities (non-timber forest products, mushrooms, fodder, fruits, berries etc.), and diversify daily diets directly or indirectly through improving productivity, hunting, diversifying tree-cropland-livestock systems, and grazing in forests. Based on the extent of forest contributing to food supply, considering the people undernourished (Rowland et al. 2017; FAO, IFAD, and WFP, 2013), and the annual deforestation rate (Keenan et al. 2015), the global potential to enhance food security is moderate for improved forest management and small for reduced deforestation (Table 6.46). The uncertainty of these global estimates is high. More robust qualitative and some quantitative estimates are available at regional level. For example, managed natural forests, shifting cultivation and agroforestry systems are demonstrated to be crucial to food security and nutrition for hundreds of million people in rural landscapes worldwide (Sunderland et al. 2013; Vira et al. 2015). According to Erb et al. (2016), deforestation would not be needed to feed the global population by 2050, in terms of quantity and quality of food. At local level, Cerri et al. (2018) suggested that reduced deforestation, along with integrated cropland-livestock management, would positively impact more than 120 million people in the Cerrado, Brazil. In Sub-Saharan Africa, where population and food demand are projected to continue to rise substantially, reduced deforestation may have strong positive effects on food security (Doelman et al. 2018).

Afforestation and reforestation negatively impact food security (Boysen et al. 2017b; Frank et al. 2017; Kreidenweis et al. 2016b). It is estimated that large-scale afforestation plans could cause increases in food prices of 80% by 2050 (Kreidenweis et al. 2016b), and more general mitigation measures in the AFOLU sector can translate into a rise in undernourishment of 80–300 million people (Frank et al. 2017) (Table 6.16). For reforestation, the potential adverse side-affects with food security are smaller than afforestation, because forest regrows on recently deforested areas, and its impact would be felt mainly through impeding possible expansion of agricultural areas. On a smaller scale, forested land also offers benefits in terms of food supply, especially when forest is established on degraded land, mangroves, and other land that cannot be used for agriculture. For example, food from forests represents a safety-net during times of food and income insecurity (Wunder et al., 2014), and wild harvested meat and freshwater fish provides 30-80% of protein intake from many rural communities (McIntyre et al., 2016; Nasi et al., 2011).

Table 6.46 summarises the impact on food security of options in forestry, with confidence estimates based on the thresholds outlined in Table 6.53 in section 6.3.6, and indicative (not exhaustive) references upon which the evidence in based.

Table 6.46 Effects on food security of response options in forestry

Integrated response option	Potential	Confidence	Citation
Improved forest management	Positive impact on < 100 million people	Low confidence	FAO, IFAD & WFP, 2013; Rowland et al. 2017
Reduced deforestation and degradation	Positive impact on < 1 million people	Low confidence	FAO, IFAD & WFP, 2013; Keenan et al. 2015; Rowland et al. 2017
Reforestation and forest restoration	See afforestation		
Afforestation	Negative impact on > 100 million people	Medium confidence	Boysen et al. 2017b; Frank et al. 2017; Kreidenweis et al. 2016b

6.3.5.1.3 Integrated response options based on land management of soils

Increasing soil organic matter stocks can increase yield and improve yield stability (Lal 2006b; Pan et al. 2009; Soussana et al. 2019), though this is not universally seen (Hijbeek et al., 2017). Lal (2006b) concludes that crop yields can be increased by 20–70 kg ha^{-1}, 10–50 kg ha^{-1} and 30–300 kg ha^{-1} for maize for wheat, rice and maize, respectively, for every 1 t C ha^{-1} increase in soil organic carbon in the root zone. Increasing soil organic carbon by 1 t C ha^{-1} could increase food grain production in developing countries by 32 Mt yr^{-1} (Lal 2006b). Frank et al. (2017) estimate that soil carbon sequestration could reduce calorie loss associated with agricultural mitigation measures by 65%, saving 60–225 million people from undernourishment compared to a baseline without soil carbon sequestration (Table 6.47).

Lal (1998) estimated the risks of global annual loss of food production due to accelerated erosion to be as high as 190 Mt yr^{-1} of cereals, 6 Mt yr^{-1} of soybean, 3 Mt yr^{-1} of pulses and 73 Mt yr^{-1} of roots and tubers. Considering only cereals, if we assume per-capita annual grain consumption in developing countries to be 300 kg yr^{-1} (estimated based on data included in Pradhan et al., 2013; FAO, 2018; FAO et al., 2018; and World Bank 2018a), the loss of 190 Mt yr^{-1} of cereals is equivalent to that consumed by 633 million people, annually (Table 6.47).

Though there are biophysical barriers, such as access to appropriate water sources and limited productivity of salt-tolerant crops, prevention / reversal of soil salinisation could benefit 1–100 million people (Qadir et al. 2013a). Soil compaction affects crop yields, so prevention of compaction could benefit an estimated 1–100 million people globally (Anderson and Peters 2016).

Biochar on balance, could provide moderate benefits for food security by improving yields by 25% in the tropics, but with more limited impacts in temperate regions (Jeffery et al. 2017), or through improved water holding capacity and nutrient use efficiency (Chapter 5; Sohi 2012). These benefits could, however, be tempered by additional pressure on land if large quantities of biomass are required as feedstock for biochar production, thereby causing potential conflicts with food security (Smith 2016b). Smith (2016b) estimated that 0.4–2.6 Mkm^2 of land would be required for biomass feedstock to deliver 2.57 $GtCO_2e$ yr^{-1} of CO_2 removal. If biomass production occupied 2.6 Mkm^2 of cropland, equivalent to around 20% of the global cropland area, this could potentially have a large effect on food security, although Woolf et al. (2010) argue that abandoned cropland could be used to supply biomass for biochar, thus avoiding competition with food production. Similarly, Woods et al (2015) estimate that 5-9 Mkm^2 of land is available for biomass production without compromising food security and biodiversity, considering marginal and degraded land and land released by pasture intensification (Table 6.47).

Table 6.47 summarises the impact on food security of soil-based options, with confidence estimates based on the thresholds outlined in Table 6.53 in section 6.3.6, and indicative (not exhaustive) references upon which the evidence in based.

Table 6.47 Effects on food security of soil-based response options

Integrated response option	Potential	Confidence	Citation
Increased soil organic carbon content	60-225 million people	Low confidence	Frank et al. 2017
Reduced soil erosion	633 million people yr^{-1}	Low confidence	FAO, 2018; FAO et al. 2018; Lal 1998; Pradhan et al. 2013; World Bank 2018a
Reduced soil salinisation	1-100 million people	Low confidence	Qadir et al. 2013a
Reduced soil compaction	1-100 million people	Low confidence	Anderson and Peters 2016
Biochar addition to soil	Range from positive impact in the tropics from biochar addition to soil to a maximum potential negative impact on >100 million people by worst-case conversion of 20% of global cropland	Low confidence	Jeffery et al. 2017; worse case negative impacts calculated from area values in Smith 2016b

6.3.5.1.4 Integrated response options based on land management across all/other ecosystems

FAO (2015) calculated that damage from forest fires between 2003 and 2013 impacted a total of 49 thousand km^2 of crops with the vast majority in Latin America. Based on the world cereal yield in 2013 reported by Word Bank (2018b) (3.8 t ha^{-1}), the loss of 49 thousand km^2 of crops is equivalent to 18.6 Mt yr^{-1} of cereals lost. Assuming annual grain consumption per capita to be 300 kg yr^{-1} (estimated based on data included in Pradhan et al., 2013; FAO, 2018; FAO et al., 2018; and World Bank 2018a), the loss of 18.6 Mt yr^{-1} would remove cereal crops equivalent to that consumed by 62 million people (Table 6.48).

Landslides and other natural hazards affect 1–100 Million people globally, so preventing them could provide food security benefits to this many people.

In terms of measures to tackle pollution, including acidification, Shindell et al. (2012) considered about 400 emission control measures to reduce ozone and black carbon (BC). This strategy increases annual crop yields by 30–135 Mt due to ozone reductions in 2030 and beyond. If annual grain consumption per capita is assumed as 300 kg yr^{-1} (estimated based on data included in Pradhan et al., 2013; FAO, 2018; FAO et al., 2018; and World Bank 2018a), increase in annual crop yields by 30–135 Mt feeds 100–450 million people.

There are no global data on the impacts of management of invasive species / encroachment on food security.

Since large areas of converted coastal wetlands are used for food production (e.g., mangroves converted for aquaculture; (Naylor et al. 2000b), restoration of coastal wetlands could displace food production and damage local food supply, potentially leading to adverse impacts on food security, though these effects are likely to be very small given that only 0.3% of human food comes from the oceans and other aquatic ecosystems (Pimentel 2006), and that the impacts could be offset by careful management, such as the careful siting of ponds within mangroves (Naylor et al. 2000b) (Table 6.46).

Around 14-20% (0.56–0.80 Mkm^2) of the global 4 Mkm^2 of peatlands are used for agriculture, mostly for meadows and pasture, meaning that if all of these peatlands were removed from production, 0.56–0.80 Mkm^2 of agricultural land would be lost. Assuming livestock production on this land (since it is mostly meadow and pasture) with a mean productivity of 9.8 kg protein ha^{-1} yr^{-1} (calculated from land footprint of beef/mutton in (Clark and Tilman 2017)), and average protein consumption in developing countries of 25.5 kg protein yr^{-1} (equivalent to 70g $person^{-1}$ day^{-1}; FAO, 2018), this would be equivalent to 21–31 million people no longer fed from this land (Table 6.46).

There are no global estimates on how biodiversity conservation improves nutrition (i.e. number of nourished people). Biodiversity, and its management, is crucial for improving sustainable and diversified diets (Global Panel on Agriculture and Food Systems for Nutrition 2016). Indirectly, the loss of pollinators (due to combined causes, including the loss of habitats and flowering species) would contribute to 1.42 million additional deaths per year from non-communicable and malnutrition-related diseases, and 27.0 million lost disability-adjusted life-years (DALYs) per year (Smith et al. 2015). However, at the same time, some options to preserve biodiversity, like protected areas, may potentially conflict with food production by local communities (Molotoks et al. 2017).

Table 6.48 summarises the impact on food security of response options in all/other ecosystems, with confidence estimates based on the thresholds outlined in Table 6.53 in section 6.3.6, and indicative (not exhaustive) references upon which the evidence in based.

Table 6.48 Effects on food security of response options in all/other ecosystems

Integrated response option	**Potential**	**Confidence**	**Citation**
Fire management	~62 million people	Low confidence	FAO 2015; FAO 2018; FAO et al. 2018; Pradhan et al. 2013; World Bank 2018a,b
Reduced landslides and natural hazards	1-100 million people	Low confidence	Campbell 2015
Reduced pollution including acidification	Increase annual crop yields 30-135 Mt globally; feeds 100-450 million people	Low confidence	Shindell et al. 2012; FAO, 2018; FAO et al., 2018; Pradhan et al. 2013; World Bank 2018a
Management of invasive species / encroachment	No global estimates	No evidence	
Restoration and reduced conversion of coastal wetlands	Very small negative impact but not quantified	Low confidence	
Restoration and reduced conversion of peatlands	Potential negative impact on 21-31 million people	Low confidence	Clark and Tilman 2017; FAO 2018
Biodiversity conservation	No global estimates	No evidence	

6.3.5.1.5 Integrated response options based on land management specifically for CDR

The spreading of crushed minerals on land as part of enhanced weathering on nutrient-depleted soils can potentially increase crop yield by replenishing plant available silicon, potassium and other plant nutrients (Beerling et al. 2018), but there are no estimates in the literature reporting the potential magnitude of this effect on global food production.

Competition for land between bioenergy and food crops can lead to adverse side-effects for food security. Many studies indicate that bioenergy could increase food prices (Calvin et al. 2014a; Popp et al. 2017; Wise et al. 2009a). Only three studies were found linking bioenergy to the population at risk of hunger; they estimate an increase in the population at risk of hunger of between 2 million and 150 million people (Table 6.49).

Table 6.49 summarises the impact on food security of response options specifically for CDR, with confidence estimates based on the thresholds outlined in Table 6.53 in section 6.3.6, and indicative (not exhaustive) references upon which the evidence in based.

Table 6.49 Effects on food security of response options specifically for CDR

Integrated response option	Potential	Confidence	Citation
Enhanced weathering of minerals	No global estimates	No evidence	
Bioenergy and BECCS	Negative impact on up to 150 million people	Medium confidence	Baldos and Hertel 2014a; Fujimori et al. 2018b

6.3.5.2 Integrated response options based on value chain management

In this section, the impacts on food security of integrated response options based on value change management are assessed.

6.3.5.2.1 Integrated response options based on value chain management through demand management

Dietary change can free up agricultural land for additional production (Bajželj et al. 2014; Stehfest et al. 2009; Tilman and Clark 2014b) and reduce the risk of some diseases (Tilman and Clark 2014b; Aleksandrowicz et al. 2016b) , with large positive impacts on food security (Table 6.50).

Kummu et al. (2012a) estimate that an additional billion people could be fed if food waste was halved globally. This includes both post-harvest losses and retail and consumer waste, and measures such as improved food transport and distribution could also contribute to this waste reduction (Table 6.50).

While no studies quantified the effect of material substitution on food security, the effects are expected to be similar to reforestation and afforestation if the amount of material substitution leads to an increase in forest area.

Table 6.50 summarises the impact on food security of demand management options, with confidence estimates based on the thresholds outlined in Table 6.53 in section 6.3.6, and indicative (not exhaustive) references upon which the evidence in based.

Table 6.50 Effects on food security of demand management options

Integrated response option	Potential	Confidence	Citation
Dietary change	821 million people	High confidence	Aleksandrowicz et al. 2016b; Tilman and Clark 2014b
Reduced post-harvest losses	1000 million people	Medium confidence	Kummu et al. 2012
Reduced food waste (consumer or retailer)	700-1000 million people	Medium confidence	FAO 2018; Kummu et al. 2012
Material substitution	No global estimates	No evidence	

6.3.5.2.2 Integrated response options based on value chain management through supply management

Since 810 million people are undernourished (FAO, 2018), this sets the maximum number of those that could potentially benefit from sustainable sourcing or better management of supply chains. Currently however, only 1 million people are estimated to benefit from sustainable sourcing (Tayleur et al. 2017). For the latter, food price spikes affect food security and health; there are clearly documented effects of stunting among young children as a result of the 2007/2008 food supply crisis (de Brauw 2011; Arndt et al. 2012; Brinkman et al. 2010; Darnton-Hill and Cogill 2010) with a 10% increase in wasting attributed to the crisis in South Asia (Vellakkal et al. 2015). There is conflicting evidence on the impacts of different food price stability options for supply chains, and little quantification (Byerlee et al. 2006; del Ninno et al. 2007; Alderman 2010; Braun et al. 2014). Reduction in staple food prices due to price stabilisation resulted in more expenditure on other foods and increased nutrition (e.g., oils, animal products), leading to a 10% reduction in malnutrition among children in one study (Torlesse et al. 2003a). Comparison of two African countries shows that protectionist policies (food price controls) and safety nets to reduce price instability resulted in a 20% decrease in risk of malnutrition (Nandy et al. 2016). Models using policies for food aid and domestic food reserves to achieve food supply and price stability showed the most effectiveness of all options in achieving climate mitigation and food security goals (e.g. more effective than carbon taxes) as they did not exacerbate food insecurity and did not reduce ambitions for achieving temperature goals (Fujimori et al. 2018a).

For urban food systems, increased food production in cities combined with governance systems for distribution and access can improve food security, with a potential to produce 30% of food consumed in cities. The urban population in 2018 was 4.2 billion people, so 30% represents 1230 million people who could benefit in terms of food security from improved urban food systems (Table 6.51).

It is estimated that 500 million smallholder farmers depend on agricultural businesses in developing countries (World Bank, 2017), which sets the maximum number of people who could benefit from improved efficiency and sustainability of food processing, retail and agri-food industries.

Up to 2500 million people could benefit from increased energy efficiency in agriculture, based on the estimated number of people worldwide lacking access to clean energy and instead relying on biomass fuels for their household energy needs (IEA, 2014).

Table 6.51 summarises the impact on food security of supply management options, with confidence estimates based on the thresholds outlined in Table 6.53 in section 6.3.6, and indicative (not exhaustive) references upon which the evidence in based.

Table 6.51 Effects on food security of supply management options

Integrated response option	Potential	Confidence	Citation
Sustainable sourcing	> 1 million people	Low confidence	Tayleur et al. 2017
Management of supply chains	> 1 million people	Low confidence	FAO 2018; Kummu et al. 2012
Enhanced urban food systems	Up to 1260 million people	Low confidence	Benis and Ferrão 2017b; Padgham et al.; Specht et al. 2014; de Zeeuw & Drechsel 2015;
Improved food processing and retailing	500 million people	Low confidence	World Bank 2017
Improved energy use in food systems	Up to 2500 million people	Low confidence	IEA 2014

6.3.5.3 Integrated response options based on risk management

In this section, the impacts on food security of integrated response options based on risk management are assessed.

Evidence in the US indicates ambiguous trends between sprawl and food security; on one hand, most urban expansion in the US has primarily been on lands of low and moderate soil productivity with only 6% of total urban land on highly productive soil. On the other hand, highly productive soils have experienced the highest rate of conversion of any soil type (Nizeyimana et al. 2001). Specific types of agriculture are often practiced in urban-influenced fringes, such as fruits, vegetables, and poultry and eggs in the US, the loss of which can have an impact on the types of nutritious foods available in urban areas (Francis et al. 2012b). China is also concerned with food security implications of urban sprawl, and a loss of 30 Mt of grain production from 1998–2003 in eastern China was attributed to urbanisation (Chen 2007b). However, overall global quantification has not been attempted.

Diversification is associated with increased welfare and incomes and decreased levels of poverty in several country studies (Arslan et al. 2018a; Asfaw et al. 2018). These are likely to have large food security benefits (Barrett et al. 2001; Niehof 2004), but there is little global quantification.

Local seed use can provide considerable benefits for food security because of the increased ability of farmers to revive and strengthen local food systems (McMichael and Schneider 2011b); studies have reported more diverse and healthy food in areas with strong food sovereignty networks (Coomes et al. 2015b; Bisht et al. 2018). Women in particular may benefit from seed banks for low value but nutritious crops (Patnaik et al. 2017). Many hundreds of millions of smallholders still rely on local seeds and they provision many hundreds of millions of consumers (Altieri et al. 2012a; McGuire and Sperling 2016), so keeping their ability to do so through seed sovereignty is important. However, there may be lower food yields from local and unimproved seeds, so the overall impact of local seed use on food security is ambiguous (McGuire and Sperling 2016).

Disaster risk management approaches can have important impacts on reducing food insecurity, and current systems for drought warning and other storms currently reach over 100 million people. When these early warning systems can help farmers harvest crops in advance of impending weather events or otherwise make agricultural decisions to prepare for adverse events, there are likely to be positive impacts on food security (Fakhruddin et al. 2015). Surveys with farmers reporting food insecurity from climate impacts have indicated their strong interest in having such early warning systems (Shisanya and Mafongoya 2016). Additionally, famine early warning systems have been successful in Sahelian Africa to alert authorities of impending food shortages so that food acquisition and transportation from outside the region can begin, potentially helping millions of people (Genesio et al. 2011; Hillbruner and Moloney 2012).

Risk sharing instruments are often aimed at sharing food supplies and reducing risk, and thus are likely to have important, but unquantified, benefits for food security. Crop insurance in particular has generally led to (modest) expansions in cultivated land area and increased food production (Claassen et al. 2011; Goodwin et al. 2004).

Table 6.52 summarises the impact on food security of risk management options, with confidence estimates based on the thresholds outlined in Table 6.53 in section 6.3.6, and indicative (not exhaustive) references upon which the evidence in based.

Table 6.52 Effects on food security of risk management options

Integrated response option	Potential	Confidence	Citation
Management of urban sprawl	>1 million likely	Low confidence	Bren d'Amour et al. 2016; Chen 2017
Livelihood diversification	>100 million	Low confidence	Morton 2007
Use of local seeds	>100 million	Low confidence	Altieri et al. 2012a
Disaster risk management	> 100 million	Medium confidence	Genesio et al. 2011; Hillbruner and Moloney 2012
Risk sharing instruments	>1 million likely	Low confidence	Claassen et al. 2011; Goodwin et al. 2004

6.3.6 Summarising the potential of the integrated response options across mitigation, adaptation, desertification land degradation and food security

Using the quantification provided in tables 6.13 to 6.52, the impacts are categorised as either positive or negative, and are designated as large, moderate and small according to the criteria given in Table 6.53[6].

[6] FOOTNOTE: Note that: 1) The response options often overlap, so are not additive. For example, increasing food productivity will involve changes to cropland, grazing land and livestock management, which in turn my include increasing soil carbon stocks. The response options cannot therefore be summed, nor regarded as entirely mutually exclusive interventions. 2) The efficacy of a response option for addressing the primary challenge for which it is implemented needs to be weighed against any co-benefits and adverse side-effects for the other challenges, e.g. if a response option has a major impact in addressing one challenge but results in relatively minor and manageable adverse-side effects for another challenge, it may remain a powerful response option despite the adverse side-effects, particularly if they can be minimised or managed. 3) Though the impacts of integrated response options have been quantified as far as possible in Section 6.3, there is no equivalence implied in terms benefits or adverse side-effects, either in number or in magnitude of the impact, i.e. one benefit *does not equal* one adverse side-effect. As a consequence: a) Large benefits for one challenge might outweigh relatively minor adverse side-effects in addressing another challenge, and b) Some response options may deliver mostly benefits with few adverse-side effects, but the benefits might be small in magnitude, i.e. the response options do no harm, but present only minor co-benefits. A number of benefits and adverse side-effects are context specific; the context specificity has been discussed in section 6.2 and is further examined Section 6.4.5.1.

Table 6.53 Key for criteria used to define magnitude of impact of each integrated response option

	Mitigation	Adaptation	Desertification	Land Degradation	Food
Large positive	More than 3 $GtCO_2$-eq yr^{-1}	Positively impacts more than around 25 million people	Positively impacts more than around 3 million km^2	Positively impacts more than around 3 million km^2	Positively impacts more than around 100 million people
Moderate positive	0.3 to 3 $GtCO_2$-eq	1 million to 25 million	0.5 to 3 million km^2	0.5 to 3 million km^2	1 million to 100 million
Small positive	>0	Under 1 million	>0	>0	Under 1 million
Negligible	0	No effect	No effect	No effect	No effect
Small negative	<0	Under 1 million	<0	<0	Under 1 million
Moderate negative	-0.3 to -3 $GtCO_2$-eq	1 million to 25 million	0.5 to 3 million km^2	0.5 to 3 million km^2	1 million to 100 million
Large negative	More than -3 $GtCO_2$-eq yr^{-1}	Negatively impacts more than around 25 million people	Negatively impacts more than around 3 million km^2	Negatively impacts more than around 3 million km^2	Negatively impacts more than around 100 million people

Note: All numbers are for global scale; all values are for technical potential. For mitigation, the target is set at around the level of large single mitigation measure (about 1 GtC yr^{-1} = 3.67 $GtCO_2$-eq yr^{-1}) (Pacala and Socolow 2004), with a combined target to meet 100 $GtCO_2$ in 2100, to go from baseline to 2˚C (Clarke and Jiang 2014b). For adaptation, numbers are set relative to the about 5 million lives lost per year attributable to climate change and a carbon-based economy, with 0.4 million per year attributable directly to climate change. This amounts to 100 million lives predicted to be lost between 2010 and 2030 due to climate change and a carbon-based economy (DARA 2012), with the largest category representing 25% of this total. For desertification and land degradation, categories are set relative to the 10–60 million km^2 of currently degraded land (Gibbs and Salmon 2015) with the largest category representing 30% of the lower estimate. For food security, categories are set relative to the roughly 800 million people currently undernourished (HLPE 2017) with the largest category representing around 12.5% of this total.

Tables 6.54 to 6.61 summarise the potentials of the integrated response options across mitigation, adaptation, desertification, land degradation and food security. Cell colours correspond to the large, moderate and small impact categories shown in Table 6.53.

As seen in tables 6.54 to 6.61, three response options across the 14 for which there are data for every land challenge: *increased food productivity, agroforestry* and *increased soil organic carbon content*, deliver large benefits across all five land challenges.

A further six response options: *improved cropland management, improved grazing land management, improved livestock management, agroforestry, fire management* and *reduced post-harvest losses*, deliver either large or moderate benefits for all land challenges.

Three additional response options: *dietary change, reduced food waste* and *reduced soil salinisation*, each missing data to assess global potential for just one of the land challenges, deliver large or moderate benefits to the four challenges for which there are global data.

Eight response options: *increased food productivity, reforestation and forest restoration, afforestation, increased soil organic carbon content, enhanced mineral weathering, dietary change,* *reduced post-harvest losses*, and *reduced food waste*, have large mitigation potential (>3 $GtCO_2e\ yr^{-1}$) without adverse impacts on other challenges.

Sixteen response options: *increased food productivity, improved cropland management, agroforestry, agricultural diversification, improved forest management, increased soil organic carbon content, reduced landslides and natural hazards, restoration and reduced conversion of coastal wetlands, reduced post-harvest losses, sustainable sourcing, management of supply chains, improved food processing and retailing, improved energy use in food systems, livelihood diversification, use of local seeds,* and *disaster risk management*, have large adaptation potential at global scale (positively affecting >25 million people) without adverse side-effects for other challenges.

Thirty-three of the 40 response options can be applied without requiring land use change and limiting available land. A large number of response options do not require dedicated land, including several land management options, all value chain options, and all risk management options. Four options, in particular, could greatly increase competition for land if applied at scale: *afforestation, reforestation,* and land used to provide feedstock for *bioenergy (with or without BECCS)* and *biochar*, with three further options: *reduced grassland conversion to croplands, restoration and reduced conversion of peatlands and restoration* and *reduced conversion of coastal wetlands* having smaller or variable impacts on competition for land. Other options such as *reduced deforestation and degradation*, restrict land conversion for other options and uses.

Some response options can be more effective when applied together; for example, dietary change and waste reduction expand the potential to apply other options by freeing as much as 25 Mkm^2 (4-25 Mkm^2 for dietary change; Alexander et al. 2016; Bajželj et al. 2014; Stehfest et al. 2009; Tilman and Clark 2014b and 7 Mkm^2 for reduced food waste; Bajželj et al. 2014).

In terms of the categories of response options, most agricultural land management response options (all except for reduced grassland conversion to cropland which potentially adversely affects food security), deliver benefits across the five land challenges (Table 6.54). Among the forest land management options, afforestation and reforestation have the potential to deliver large co-benefits across all land challenges except for food security, where these options provide a threat due to competition for land (Table 6.55). Among the soil-based response options, some global data are missing, but none except biochar shows any potential for negative impacts, with that potential negative impact arising from additional pressure on land if large quantities of biomass feedstock are required for biochar production (Table 6.56). Where global data exists, most response options in other/all ecosystems deliver benefits except for a potential moderate negative impact on food security by restoring peatlands currently used for agriculture (Table 6.57). Of the two response options specifically targeted at CDR, there are missing data for enhanced weathering of minerals for three of the challenges, but large-scale bioenergy and BECCS shows a potential large benefit for mitigation, but small to large adverse impacts on the other four land challenges (Table 6.58), mainly driven by increased pressure on land due to feedstock demand.

While data allow the impact of material substitution to be assessed only for mitigation, the three other demand-side response options: dietary change, reduced post-harvest losses and reduced food waste provide large or moderate benefits across all challenges for which data exist (Table 6.59). For none of the supply-side response options is data available to assess the impact on more than three of the land challenges, but there are large to moderate benefits for all those for which data are available (Table 6.60). Data are not available to assess the impact of risk management-based response options on all of the challenges, but there are small to large benefits for all of those for which data are available (Table 6.61).

Table 6.54 Summary of direction and size of impact of land management options in agriculture on mitigation, adaptation, desertification, land degradation and food security

Integrated response option	**Mitigation**	**Adaptation**	**Desertification**	**Land degradation**	**Food security**	**Context and evidence base for magnitude of effect**
Increased food productivity						These estimates assume that increased food production is implemented sustainably (e.g. through sustainable intensification: Garnett et al. 2013b; Pretty et al. 2018) rather than through increasing external inputs, which can have a range of negative impacts. Mitigation: ***Large benefits*** (Table 6.13). Adaptation: ***Large benefits*** (Chapter 2; Table 6.21; Campbell et al. 2014). Desertification: ***Large benefits*** (Chapter 3; Table 6.29; Dai 2010). Land degradation: ***Large benefits*** (Chapter 4; Table 6.37; Clay et al., 1995). Food security: ***Large benefits*** (Chapter 5; Table 6.45; Godfray et al. 2010b; Tilman et al. 2011; Godfray and Garnett 2014).
Improved cropland management						Mitigation: ***Moderate benefits*** by reducing greenhouse gas emissions and creating soil carbon sinks (Chapter 2; Table 6.13; Smith et al. 2008, 2014a). Adaptation: ***Large benefits*** by improving the resilience of food crop production systems to future climate change (Chapter 2; Table 6.21; Porter et al. 2014). Desertification: ***Large benefits*** by improving sustainable use of land in dry areas (Chapter 3; Table 6.29; Bryan et al. 2009b; Chen et al. 2010). Land degradation: ***Large benefits*** by forming a major component of sustainable land management (Chapter 4; Table 6.37; Labrière et al. 2015). Food security: ***Large benefits*** by improving agricultural productivity for food production (Chapter 5; Table 6.45; Porter et al. 2014).
Improved grazing land management						Mitigation: ***Moderate benefits*** by increasing soil carbon sinks and reducing greenhouse gas emissions (Chapter 2; Table 6.13; Herrero et al. 2016). Adaptation: ***Moderate benefits*** by improving the resilience of grazing lands to future climate change (Chapter 2; Table 6.21; Porter et al. 2014). Desertification: ***Moderate benefits*** by tackling overgrazing in dry areas to reduce desertification (Chapter 3; Table 6.29; Archer et al. 2011). Land degradation: ***Large benefits*** by optimising stocking density to reduce land degradation (Chapter 4; Table 6.37; Table 6.45; Tighe et al. 2012). Food security: ***Large benefits*** by improving livestock sector productivity to increase food production (Chapter 5; Table 6.45; Herrero et al. 2016).
Improved livestock management						Mitigation: ***Moderate benefits*** by reducing greenhouse gas emissions, particularly from enteric methane and manure management (Chapter 2; Table 6.13; Smith et al. 2008, 2014a). Adaptation: ***Moderate benefits*** by improving resilience of livestock production systems to climate change (Chapter 2; Table 6.21; Porter et al. 2014). Desertification: ***Moderate benefits*** by tackling overgrazing in dry areas (Chapter 3; Table 6.29; Archer et al. 2011). Land degradation: ***Large benefits*** by reducing overstocking which can reduce land degradation (Chapter 4; Table 6.37; Table 6.45; Tighe et al. 2012). Food security: ***Large benefits*** by improving livestock sector productivity to increase food production (Chapter 5; Table 6.45; Herrero et al. 2016).
Agroforestry						Mitigation: ***Moderate benefits*** by increasing carbon sinks in vegetation and soils (Chapter 2; Table 6.13; Delgado 2010; Mbow et al. 2014a; Griscom et al. 2017a). Adaptation: ***Large benefits*** by improving the resilience of agricultural lands to climate change (Chapter 2; Table 6.21; Mbow et al. 2014a). Desertification: ***Large benefits*** through e.g. provides perennial vegetation in dry areas (Chapter 3; Table 6.29; Nair et al. 2010; Lal 2001a). Land degradation: ***Large benefits*** by stabilising soils through perennial

						vegetation (Chapter 4; Table 6.37; Narain et al. 1997; Lal 2001a). Food production: ***Large benefits*** since well-planned agroforestry can enhance productivity (Chapter 5; Table 6.45; Bustamante et al. 2014b; Sasha et al., 2018).
Agricultural diversification						Agricultural diversification is a collection of practices aimed at deriving more crops or products per unit of area (e.g. intercropping) or unit of time (e.g. double cropping, ratoon crops etc.). Mitigation: ***Limited benefits*** (Table 6.13). Adaptation: ***Large benefits*** through improved household income (Pellegrini and Tasciotti 2014; Table 6.21). Desertification: ***Moderate benefits***, limited by global dryland cropped area (Table 6.29). Land degradation: ***Large benefits*** by reducing pressure on land (Table 6.37; Lambin and Meyfroidt 2011). Food security: ***Large benefits*** for food security by provision of more diverse foods (Chapter 5; Table 6.45; Birthal et al. 2015; Massawe et al. 2016; Waha et al. 2018).
Reduced grassland conversion to cropland		ND				Mitigation: ***Moderate benefits*** by retaining soil carbon stocks that might otherwise be lost. Historical losses of soil carbon have been on the order of 500 $GtCO_2$ (Table 6.13; Sanderman et al. 2017). Mean annual global cropland conversion rates (1961–2003) have been 0.36% per year (Krause et al. 2017), i.e. around 47 thousand km^2 yr^{-1} – so preventing conversion could potentially save moderate emissions of CO_2. Adaptation: No literature (Table 6.21). Desertification: ***Limited benefits*** by shifting from annual crops to permanent vegetation cover under grass in dry areas (Chapter 3; Table 6.29). Land degradation: ***Limited benefits*** by shifting from annual crops to permanent vegetation cover under grass (Chapter 4; Table 6.37). Food security: ***Moderate negative impacts***, since more land is required to produce human food from livestock products on grassland than from crops on cropland, meaning that a shift to grassland could reduce total productivity and threaten food security (Chapter 5; Table 6.45; Clark and Tilman 2017).
Integrated water management						Mitigation: M***oderate benefits*** by reducing greenhouse gas emissions mainly in cropland and rice cultivation (Chapter 2; Table 6.13; Smith et al. 2008, 2014a). Adaptation: ***Large benefits*** by improving the resilience of food crop production systems to future climate change (Chapter 2; Table 6.21; Porter et al. 2014). Desertification: ***Limited benefits*** by improving sustainable use of land in dry areas (Chapter 3; Table 6.29). Land degradation: ***Limited benefits*** by forming a major component of sustainable land and water management (Chapter 4; Table 6.37). Food security: ***Large benefits*** by improving agricultural productivity for food production (Chapter 5; Table 6.45; Tilman et al. 2011; Godfray and Garnett 2014).

Note: Cell colours correspond to the large, moderate and small categories shown in Table 6.53. Dark blue = large positive; mid-blue = moderate positive; light blue = small positive; no colour = no effect; light red = small negative; mid-red = moderate negative; dark red = large negative; green = variable; ND = no data.

Table 6.55 Summary of direction and size of impact of land management options in forests on mitigation, adaptation, desertification, land degradation and food security

Integrated response option	**Mitigation**	**Adaptation**	**Desertification**	**Land degradation**	**Food security**	**Context and evidence base for magnitude of effect**
Improved forest						Mitigation: ***Moderate benefits*** by conserving and enhancing carbon stocks in forests and long-lived products, through for example, selective logging (Table 6.14; Smith et al. 2014a). Adaptation: ***Large benefits***, including through improving ecosystem

management						functionality and services, with mostly qualitative evidence at global scale and more robust estimates at regional level and local scale (Table 6.22; Locatelli et al. 2015d). Desertification and land degradation: ***Large benefits*** by helping to stabilise land and regulate water and microclimate (Chapters 3 and 4; Tables 6.30 and 6.38; Locatelli et al. 2015d). Food security: ***Moderate benefits*** with mostly qualitative estimate at global level, by providing food to local communities, and diversify daily diets (Chapter 5; Table 6.46).
Reduced deforestation and degradation						Mitigation: ***Large benefits*** by maintaining carbon stocks in forest ecosystems (Chapter 2; Table 6.14). Adaptation ***Moderate benefits*** at global scale when effect is cumulated till the end of the century; local scale, co-benefits between REDD+ and adaptation of local communities can be more substantial (Long 2013; Morita & Matsumoto 2017), even if often difficult to quantify and not explicitly acknowledged (McElwee et al. 2017a; Table 6.22). Desertification and land degradation: ***Large benefits*** at global scale when effects are cumulated for at least 20 years, e.g. through reduced soil erosion (Borrelli et al. 2017; Tables 6.30 and 6.38). The uncertainty of these global estimates is high, while more robust qualitative and some quantitative estimates are available at regional level. Food security: ***Small benefits***; difficult to quantify at global level (Chapter 5; Table 6.46).
Reforestation and forest restoration						Mitigation: ***Large benefits*** by rebuilding the carbon stocks in forest ecosystems, although decreases in surface albedo can reduce the net climate benefits, particularly in areas affected by seasonal snow cover (Chapter 2; Table 6.14; Sonntag et al. 2016; Mahmood et al. 2014). Adaptation: ***Large benefits*** by provision of Nature's Contributions to People, including improving ecosystem functionality and services, providing microclimatic regulation for people and crops, wood and fodder as safety nets, soil erosion protection and soil fertility enhancement for agricultural resilience, coastal area protection, water and flood regulation (Locatelli et al. 2015d; Table 6.22). Desertification: ***Large benefits*** through restoring forest ecosystems in dryland areas (Chapter 3; Table 6.30; Idris Medugu et al. 2010a; Salvati et al. 2014b). Land degradation: ***Large benefits*** by re-establishment of perennial vegetation (Chapter 4; Table 6.38; Ellison et al. 2017b). Food security: ***Moderate negative impacts*** due to potential competition for land for food production (Chapter 5; Table 6.46; Frank et al. 2017).
Afforestation						Mitigation: ***Large benefits*** for mitigation (Chapter 2; Table 6.14), especially if it occurs in the tropics and in areas that are not significantly affected by seasonal snow cover. Adaptation: ***Large benefits*** on adaptation (Chapter 2; Table 6.22; Kongsager et al. 2016; Reyer et al. 2009). Desertification: ***Large benefits*** by providing perennial vegetation in dry areas to help control desertification (Chapter 3; Table 6.30; Idris Medugu et al. 2010a; Salvati et al. 2014b). Land degradation: ***Large benefits*** by stabilising soils through perennial vegetation (Chapter 4; Table 6.38; Lal 2001a). Food security: ***Large negative impacts*** due to competition for land for food production (Chapter 5; Table 6.46; Kreidenweis et al. 2016b; Smith et al. 2013b).

Note: Cell colours correspond to the large, moderate and small categories shown in Table 6.53. Dark blue = large positive; mid-blue = moderate positive; light blue = small positive; no colour = no effect; light red = small negative; mid-red = moderate negative; dark red = large negative; green = variable; ND = no data.

Table 6.56 Summary of direction and size of impact of soil-based land management options on mitigation, adaptation, desertification, land degradation and food security

Integrated response option	Mitigation	Adaptation	Desertification	Land degradation	Food security	Context and evidence base for magnitude of effect
Increased soil organic carbon content						Mitigation: ***Large benefits*** by creating soil carbon sinks (Table 6.15). Adaptation: ***Large benefits*** by improving resilience of food crop production systems to climate change (Chapter 2; Table 6.24; IPBES 2018). Desertification: ***Large benefits*** by improving soil health and sustainable use of land in dry areas (Chapter 3; Table 6.31; D'Odorico et al. 2013). Land degradation: ***Large benefits*** since it forms a major component of recommended practices for sustainable land management (Chapter 4; Table 6.39; Altieri and Nicholls 2017). Food security: ***Large benefits*** since it can increase yield and yield stability to enhance food production, though this is not always the case (Chapter 5; Table 6.47; Pan et al. 2009; Soussana et al. 2019; Hijbeek et al., 2017; Schjønning et al., 2018).
Reduced soil erosion						Mitigation: ***Large benefits or large negative impacts***, since the final fate of eroded material is still debated, at the global level it is debated whether it is a large source or a large sink (Chapter 2; Table 6.15; Hoffmann et al. 2013). Adaptation: ***Large benefits*** since soil erosion control prevents desertification (***large benefits***) and land degradation (***large benefits***), thereby improving the resilience of agriculture to climate change (Chapter 2, 3 and 4; Table 6.23, 6.30 and 6.39; Lal 1998; FAO and ITPS 2015). Food security: ***Large benefits*** mainly through the preservation of crop productivity (Chapter 5; Table 6.47; Lal 1998).
Reduced soil salinisation	ND					Techniques to prevent and reverse soil salinisation include groundwater management by drainage systems and/or crop rotation and use of amendments to alleviate soil sodicity. Mitigation: There are no studies to quantify the global impacts (Table 6.15). Adaptation: ***Moderate benefits*** by allowing existing crop systems to be maintained, reducing the need to abandon land (Table 6.23; UNCTAD 2011; Dagar et al. 2016b). Desertification and land degradation: ***Moderate benefits*** since soil salinisation is a main driver of both desertification and land degradation (Chapters 3 and 4; Tables 6.31 and 6.39; Rengasamy 2006; Dagar et al. 2016b). Food security: ***Moderate benefits*** by maintaining existing cropping systems and helping to close yield gaps in rainfed crops (Table 6.47).
Reduced soil compaction	ND		ND			Techniques to prevent and reverse soil compaction are based on the combination of suitable crop rotations, tillage and regulation of agricultural traffic (Hamza and Anderson 2005b). Mitigation: The global mitigation potential has not been quantified (Table 6.15; Chamen et al. 2015a; Epron et al. 2016; Tullberg et al. 2018b). Adaptation: ***Limited benefits*** by improving productivity but on relatively small global areas (Table 6.22). Desertification: no global data (Table 6.31). Land degradation: ***Large benefits*** since soil compaction is a main driver of land degradation (Table 6.39; FAO and ITPS 2015). Food security: ***Moderate benefits*** by helping to close yield gaps where compaction is a limiting factor (Table 6.47; Anderson and Peters 2016).
Biochar addition to soil		ND	ND			Mitigation: ***Large benefits*** by increasing recalcitrant carbon stocks in the soil (Chapter 2; Table 6.15; Smith 2016b; Fuss et al. 2018b; IPCC 2018). Adaptation: There are no global estimates of the impact of biochar on climate adaptation (Table 6.23). Desertification: There are no global estimates of the impact of biochar on desertification (Table 6.31). Land degradation: ***Limited benefits*** by improving the soil water holding capacity, nutrient use efficiency, and potentially ameliorating heavy metal pollution (Table 6.39; Sohi 2012). Food security: ***Limited benefits*** by increasing crop yields in the tropics (though not in temperate regions; Jeffery et al. 2017), but potentially ***Large negative impacts*** by creating additional pressure on land if large quantities of biomass

						feedstock are required for biochar production (Table 6.47).

<u>Note</u>: Cell colours correspond to the large, moderate and small categories shown in Table 6.53. Dark blue = large positive; mid-blue = moderate positive; light blue = small positive; no colour = no effect; light red = small negative; mid-red = moderate negative; dark red = large negative; green = variable; ND = no data.

Table 6.57 Summary of direction and size of impact of land management in all/other ecosystems on mitigation, adaptation, desertification, land degradation and food security

Integrated response option	Mitigation	Adaptation	Desertification	Land degradation	Food security	Context and evidence base for magnitude of effect
Fire management						<u>Mitigation</u>: ***Large benefits*** by reduced size, severity, and frequency of wildfires, thereby preventing emissions and preserving carbon stocks (Table 6.16; Chapter 2, Cross-Chapter Box 3; Arora and Melton 2018). <u>Adaptation</u>: ***Moderate benefits*** by reducing mortality attributable to landscape fire smoke exposure, fire management provides adaptation benefits (Table 6.24; Doerr and Santín 2016; Johnston et al. 2012; Shannon et al., 2016). <u>Desertification</u>: ***Large benefits*** since control of wildfires and long-term maintenance of tree stock density protects against soil erosion (Table 6.32; Neary et al. 2009a; Arora and Melton 2018). <u>Land degradation</u>: ***Large benefits*** by stabilising forest ecosystems (Table 6.40; Neary et al. 2009a; Arora and Melton 2018). <u>Food security</u>: ***Moderate benefits*** by maintaining forest food product availability and preventing fire expansion to agricultural land (Table 6.48; FAO 2015; FAO 2018; FAO et al., 2018; Pradhan et al., 2013; World Bank 2018a,b).
Reduced landslides and natural hazards						<u>Mitigation</u>: The prevention of landslides and natural hazards benefits mitigation, but because of the limited impact on GHG emissions and eventual preservation of topsoil carbon stores, the impact is estimated to be small globally (Table 6.16; IPCC AR5 WG2, Chapter 14). <u>Adaptation</u>: Provides structural/physical adaptations to climate change (Table 6.24; IPCC AR5 WG2, Chapter 14). <u>Desertification</u>: Due to the small global areas affected within global drylands, the benefits for desertification control are limited (Chapter 3; Table 6.32). <u>Land degradation</u>: Since landslides and natural hazards are among the most severe degradation processes, prevention will have a large positive impact on land degradation (Chapter 4; Table 6.40; FAO and ITPS 2015). <u>Food security</u>: In countries in which mountain slopes are cropped for food, such as in the Pacific Islands (Campbell 2015), the management and prevention of landslides can deliver benefits for food security, though the global areas are limited (Table 6.48).
Reduced pollution including acidification						<u>Mitigation</u>: ***Large benefits*** since measures to reduce emissions of Short-Lived Climate Pollutants (SLCPs) can slow projected global mean warming (UNEP and WMO 2011), with early intervention providing 0.5°C cooling by 2050 (Table 6.16; UNEP and WMO 2011). But ***moderate negative impacts*** are also possible since reduced reactive N deposition could decrease terrestrial carbon uptake (Table 6.16). <u>Adaptation</u>: ***Moderate benefits*** since controlling PM2.5 and ozone improves human health (Table 6.24; Anenberg et al. 2012). <u>Desertification:</u> ***Moderate benefits*** since salinisation, pollution, and acidification are stressors for desertification (Table 6.32; Oldeman et al. 1991). <u>Land degradation</u>: Moderate benefits since acid deposition is a significant driver of land degradation (Table 6.40; Oldeman et al. 1991; Smith et al. 2015). <u>Food security</u>: ***Large benefits*** since ozone is harmful to crops, so measures to reduce air pollution would be expected to increase crop production (Table 6.48; Shindell et al. 2012; Pradhan

						et al., 2013; FAO, 2018; FAO et al., 2018; World Bank 2018a).
Management of invasive species / encroachment	N D	N D	N D	N D	N D	There is no literature that assesses the global potential of management of invasive species on <u>mitigation, adaptation, desertification, land degradation</u> or on <u>food security</u> (Table 6.16; Table 6.24; Table 6.33; Table 6.40; Table 6.48).
Restoration and reduced conversion of coastal wetlands						<u>Mitigation</u>: ***Large benefits*** since coastal wetland restoration and avoided coastal wetland impacts deliver moderate carbon sinks by 2030 (Table 6.16; Griscom et al. 2017a). <u>Adaptation</u>: ***Large benefits*** by providing a natural defence against coastal flooding and storm surges by dissipating wave energy, reducing erosion and by helping to stabilise shore sediments (Table 6.24). <u>Desertification</u>: There is likely negligible impact of coastal wetland restoration for prevention of desertification (Table 6.32). <u>Land degradation</u>: Limited benefits since large areas of global coastal wetlands are degraded (Lotze et al. 2006; Griscom et al. 2017a; Table 6.40). <u>Food security</u>: ***Small benefits*** to ***small adverse impacts*** since large areas of converted coastal wetlands are used for food production (e.g. mangroves converted for aquaculture), restoration could displace food production and damage local food supply, though mangrove restoration can also restore local fisheries (Table 6.48; Naylor et al. 2000b).
Restoration and reduced conversion of peatlands		N D				<u>Mitigation</u>: ***Moderate benefits*** since avoided peat impacts and peat restoration deliver moderate carbon sinks by 2030 (Table 6.16; Griscom et al. 2017a), though there can be increases in methane emissions after restoration (Jauhiainen et al. 2008). <u>Adaptation</u>: Likely to be benefits by regulating water flow and preventing downstream flooding (Table 6.24; Munang et al. 2014a), but the global potential has not been quantified. <u>Desertification</u>: No impact since peatlands occur in wet areas and deserts in dry areas. <u>Land degradation</u>: ***Moderate benefits*** since large areas of global peatlands are degraded (Table 6.40; Griscom et al. 2017a). <u>Food security</u>: ***Moderate adverse impacts*** since restoration of large areas of tropical peatlands and some northern peatlands that have been drained and cleared for food production, could displace food production and damage local food supply (Table 6.48).
Biodiversity conservation			N D	N D	N D	<u>Mitigation</u>: ***Moderate benefits*** from carbon sequestration in protected areas (Table 6.16; Calvin et al. 2014a). <u>Adaptation</u>: ***Moderate benefits*** – likely many millions benefit adaptation and resilience of local communities to climate change (Table 6.24; CBD, 2008), though global potential is poorly quantified. <u>Desertification</u>: No global data (Table 6.32). <u>Land degradation</u>: No global data (Table 6.40). <u>Food security</u>: No global data (Table 6.48).

<u>Note</u>: Cell colours correspond to the large, moderate and small categories shown in Table 6.53. Dark blue = large positive; mid-blue = moderate positive; light blue = small positive; no colour = no effect; light red = small negative; mid-red = moderate negative; dark red = large negative; green = variable; ND = no data.

Table 6.58 Summary of direction and size of impact of land management options specifically for CDR on mitigation, adaptation, desertification, land degradation and food security

Integrated response option	**Mitigation**	**Adaptation**	**Desertification**	**Land degradation**	**Food security**	**Context and evidence base for magnitude of effect**
Enhanced		N	N		N	<u>Mitigation</u>: ***Moderate to large benefits*** by removing atmospheric CO_2 (Table 6.17; Lenton 2010; Smith et al. 2016b; Taylor et al.

weathering of minerals		D	D		D	2016b). <u>Adaptation</u>: There is no literature to assess the global impacts of enhanced mineral weathering on adaptation (Table 6.25) nor on <u>desertification</u> (Table 6.33). <u>Land degradation</u>: ***Limited benefits*** expected since ground minerals can increase pH where acidification is the driver of degradation (Table 6.41; Taylor et al. 2016b). <u>Food security</u>: Though there may be co-benefits for food production (Beerling et al. 2018), these have not been quantified globally (Table 6.49).
Bioenergy and BECCS						<u>Mitigation</u>: ***Large benefits*** of large-scale bioenergy and BECCS by potential to remove large quantities of CO_2 from the atmosphere (Table 6.17). <u>Adaptation</u>: ***Limited adverse impacts*** of large-scale bioenergy and BECCS by increasing pressure on land (Table 6.25). <u>Desertification</u>: Moderate adverse impacts of large-scale bioenergy and BECCS through increased pressure on land (Table 6.33). <u>Land degradation</u>: ***Large adverse impacts*** of large-scale bioenergy and BECCS through increased pressure on land (Table 6.41). <u>Food security</u>: ***Large adverse impacts*** of large-scale bioenergy and BECCS through increased competition for land for food (Table 6.49). These potentials and effects assume large areas of bioenergy crops resulting in large mitigation potentials (i.e. >3 $GtCO_2$ yr^{-1}). The sign and magnitude of the effects of bioenergy and BECCS depends on the scale of deployment, the type of bioenergy feedstock, which other response options are included, and where bioenergy is grown (including prior land use and indirect land use change emissions). For example, limiting bioenergy production to marginal lands or abandoned cropland would have negligible effects on biodiversity, food security, and potentially small co-benefits for land degradation; however, the benefits for mitigation would also be smaller (Cross-Chapter Box 7 on Bioenergy (Chapter 6); Table 6.13).

<u>Note</u>: Cell colours correspond to the large, moderate and small categories shown in Table 6.53. Dark blue = large positive; mid-blue = moderate positive; light blue = small positive; no colour = no effect; light red = small negative; mid-red = moderate negative; dark red = large negative; green = variable; ND = no data.

Table 6.59 Summary of direction and size of impact of demand management options on mitigation, adaptation, desertification, land degradation and food security

Integrated response option	**Mitigation**	**Adaptation**	**Desertification**	**Land degradation**	**Food security**	**Context and evidence base for magnitude of effect**
Dietary change		ND				<u>Mitigation</u>: ***Large benefits*** for mitigation by greatly reducing GHG emissions (Chapter 5; Table 6.18). <u>Adaptation</u>: While it would be expected to help with adaptation by reducing agricultural land area, there are no studies providing global quantifications (Table 6.26). <u>Desertification</u>: Potential ***moderate benefits*** by decreasing pressure on land (restricted by relatively limited global area; Table 6.34). <u>Land degradation</u>: ***Large benefits*** by decreasing pressure on land (Table 6.42). Food security: Large benefits by decreasing competition for land allowing more food to be produced from less land (Table 6.50).
Reduced post-harvest losses						<u>Mitigation</u>: ***Large benefits*** by reducing food sector GHG emissions and reducing area required to produce the same quantity of food (Table 6.18), though increased use of refrigeration could increase emissions from energy use. <u>Adaptation</u>: ***Large benefits*** by reducing pressure on land (Table 6.26). <u>Desertification</u> and <u>land degradation</u>: ***Moderate benefits*** for both by reducing pressure on land (Table 6.34; Table 6.42). <u>Food security</u>: ***Large benefits*** since most of the food wasted in developing countries arises from post-harvest losses (Chapter 5; Table 6.50; Ritzema et al. 2017).

Reduced food waste (consumer or retailer)		ND				Mitigation: ***Large benefits*** by reducing food sector GHG emissions and reducing area required to produce the same quantity of food (Table 6.18). Adaptation: While it would be expected to help with adaptation by reducing agricultural land area, there are no studies quantifying global adaptation impacts (Table 6.26). Desertification: ***Moderate benefits*** by reducing pressure on land (Table 6.34). Land degradation: ***Large benefits*** by reducing pressure on land (Table 6.42). Food security: ***Large benefits*** since 30% of all food produced globally is wasted (Table 6.50; Kummu et al. 2012).
Material substitution		ND	ND	ND	ND	Mitigation: ***Moderate benefits*** through long-lived carbon storage, and by substitution of materials with higher embedded GHG emissions (Table 6.18). No global studies available to assess the quantitative impact on adaptation, desertification, land degradation or food security (Table 6.26; Table 6.34; Table 6.42; Table 6.50).

Note: Cell colours correspond to the large, moderate and small categories shown in Table 6.53. Dark blue = large positive; mid-blue = moderate positive; light blue = small positive; no colour = no effect; light red = small negative; mid-red = moderate negative; dark red = large negative; green = variable; ND = no data.

Table 6.60 Summary of direction and size of impact of supply management options on mitigation, adaptation, desertification, land degradation and food security

Integrated response option	**Mitigation**	**Adaptation**	**Desertification**	**Land degradation**	**Food security**	**Context and evidence base for magnitude of effect**
Sustainable sourcing	ND		ND			Mitigation: No studies available to assess the global impact (Table 6.19). Adaptation: ***Moderate benefits*** by diversifying and increasing flexibility in the food system to climate stressors and shocks while simultaneously creating economic alternatives for the poor (thereby strengthening adaptive capacity) and lowering expenditures of food processors and retailers by reducing losses (Chapter 5; Table 6.27; Muller et al. 2017a). Desertification: No studies available to assess the global impact (Table 6.35; Table 6.43). Land degradation: Potentially ***large benefits***, as over 4 Mkm^2 currently certified for sustainable forest production, which could increase in future (Table 6.44). Food security: ***Moderate benefits*** by diversifying markets and developing value-added products in the food supply system, by increasing its economic performance and revenues to local farmers (Reidsma et al. 2010), by strengthening the capacity of food production chains to adapt to future markets and to improve income of smallholder farmers (Chapter 5; Table 6.51; Murthy and Madhava Naidu 2012). It may also provide more direct links between producers and consumers.
Management of supply chains	ND		ND	ND		Mitigation: There are no studies assessing the mitigation potential globally (Table 6.19). Adaptation: ***Large benefits*** by improving resilience to price increases or reducing volatility of production (Table 6.27; Fafchamps et al. 1998; Haggblade et al. 2017). Desertification and land degradation: No studies assessing global potential (Table 6.35; Table 6.43). Food security: ***Moderate benefits*** through helping to manage food price increases and volatility (Table 6.51; Vellakkal et al. 2015; Arndt et al. 2016).
Enhanced urban food	ND	ND	ND	ND		There are no studies that assess the global potential to contribute to mitigation, adaptation, desertification or land degradation (Table 6.19; Table 6.27; Table 6.35; Table 6.43). Food security: ***Large benefits*** by increasing food access to urban dwellers and

systems						shortening of supply chains (Chapter 5; Table 6.51; Chappell et al. 2016).
Improved food processing and retailing			ND	ND		<u>Mitigation</u>: ***Moderate benefits*** through reduced energy consumption, climate-friendly foods and reduced GHG emissions from transportation (Avetisyan et al. 2014), waste (Porter et al. 2016b), and energy use (Table 6.19; Mohammadi et al. 2014; Song et al. 2017). <u>Adaptation</u>: Large benefits among poor farmers through reduced costs and improved resilience (Table 6.27). <u>Desertification</u> and <u>land degradation</u>: There are no studies assessing global potential (Table 6.35; Table 6.43). <u>Food security</u>: ***Large benefits*** by supporting healthier diets and reducing food loss and waste (Chapter 5; Table 6.51; Garnett 2011).
Improved energy use in food systems			ND	ND		<u>Mitigation</u>: ***Moderate benefits*** by reducing GHG emissions through decreasing use of fossil fuels and energy-intensive products, though the emission reduction is not accounted for in the AFOLU sector (Table 6.19; Smith et al. 2014a; IPCC AR5 WG3 Chapter 11). <u>Adaptation</u>: ***Large benefits*** for small farmers by reducing costs and increasing their resilience to climate change (Table 6.27). <u>Desertification</u> and <u>land degradation</u>: There are no studies assessing global potential (Table 6.35; Table 6.43). <u>Food security</u>: ***Large benefits***, largely by improving efficiency for 2.5 million people still using traditional biomass for energy (Chapter 5; Table 6.51).

<u>Note</u>: Cell colours correspond to the large, moderate and small categories shown in Table 6.53. Dark blue = large positive; mid-blue = moderate positive; light blue = small positive; no colour = no effect; light red = small negative; mid-red = moderate negative; dark red = large negative; green = variable; ND = no data.

Table 6.61 Summary of direction and size of impact of risk management options on mitigation, adaptation, desertification, land degradation and food security

Integrated response option	**Mitigation**	**Adaptation**	**Desertification**	**Land degradation**	**Food security**	**Context and evidence base for magnitude of effect**
Management of urban sprawl	ND					<u>Mitigation</u>: There are no studies assessing the global potential (Table 6.20). <u>Adaptation</u>: ***Moderate benefits*** - though poorly quantified globally, likely to affect many millions of people (Table 6.28). <u>Desertification</u>: ***Limited benefits*** - though poorly quantified globally, 5000 km^2 is at risk from urban sprawl in Spain alone; Table 6.36). <u>Land degradation</u>: ***Limited benefits*** - though poorly quantified globally, urban sprawl effects millions of ha of land (Table 6.44). <u>Food security</u>: ***Moderate benefits*** estimated from impacts on food supply in models (Table 6.52; Bren d'Amour et al. 2016).
Livelihood diversification	ND					<u>Mitigation</u>: There are no studies assessing the global potential (Table 6.20). <u>Adaptation</u>: ***Large benefits*** through helping households to buffer income fluctuations and providing a broader range of options for the future (Table 6.28; Ahmed and Stepp 2016b; Thornton and Herrero 2014). <u>Desertification</u>: There are no studies assessing the global potential, although there are anecdotal reports of ***limited benefits*** from improved land management resulting from diversification (Batterbury 2001; Herrmann and Hutchinson 2005; Stringer et al. 2009) (Table 6.36). <u>Land degradation</u>: ***Limited benefits***, for example through improved land use mosaics (Ribeiro et al 2013), larger-scale adoption in China's Sloping Land Conversion program to diversify income and reduce degradation has impacted 0.1 Mkm^2 (Liu and Lan 2015; Table 6.44). <u>Food security</u>: ***Large benefits*** since many of the world's 700 million smallholders practice diversification, helping to provide economic access to food (Table 6.52; Morton 2007).

Use of local seeds	ND		ND	ND		<u>Mitigation</u>: There are no studies assessing the global potential (Table 6.19). <u>Adaptation</u>: ***Large benefits*** given that 60 to 100% of seeds used in various countries of the global South are likely local farmer-bred (non-commercial) seed and moving to the use of commercial seed would increase costs considerably for these farmers. Seed networks and banks protect local agrobiodiversity and landraces, which are important to facilitate adaptation, and can provide crucial lifelines when crop harvests fail (Table 6.28; Louwaars 2002; Howard 2015; Coomes et al. 2015b; van Niekerk and Wynberg 2017b; Vasconcelos et al. 2013; Reisman 2017). <u>Desertification</u> and <u>land degradation</u>: There are no studies assessing global potential (Table 6.36; Table 6.44). <u>Food security</u>: ***Large benefits*** since local seeds increases the ability of farmers to revive and strengthen local food systems; several studies have reported more diverse and healthy food in areas with strong food sovereignty networks (Table 6.52; Coomes et al. 2015b; Bisht et al. 2018).
Disaster risk management	ND		ND	ND		<u>Mitigation</u>: There are no studies to assess the global mitigation potential of different DRM approaches (Table 6.19). <u>Adaptation</u>: ***Large benefits*** due to widespread use of Early Warning Systems that reach hundreds of millions (Table 6.28; Hillbruner and Moloney 2012; Mahmud and Prowse 2012; Birkmann et al. 2015b). <u>Desertification</u> and <u>land degradation</u>. There are no studies assessing the global potential (Table 6.36; Table 6.44). <u>Food security</u>: ***Moderate benefits*** by helping farmers to harvest crops in advance of impending weather events or otherwise to make agricultural decisions to prepare for adverse events (Table 6.52; Fakhruddin et al. 2015; Genesio et al. 2011; Hillbruner and Moloney 2012).
Risk sharing instruments			ND			<u>Mitigation</u>: ***Variable impacts***- poor global coverage in the literature though studies from the US suggest a small increase in emissions from crop insurance and likely benefits from other risk sharing instruments(Table 6.20). <u>Adaptation</u>: ***Moderate benefits*** by buffering and transferring weather risk, saving farmers the cost of crop losses. However, overly subsidised insurance can undermine the market's role in pricing risks and thus depress more rapid adaptation strategies (Table 6.28; Meze-Hausken et al. 2009; Skees and Collier 2012; Jaworski 2016). <u>Desertification</u>: The impacts of risk sharing globally have not been quantified (Table 6.36). <u>Land degradation</u>: ***Variable impacts*** as evidence suggests that subsidised insurance in particular can increase crop production in marginal lands, and reforming this would lead to benefits (Table 6.44). <u>Food security</u>: ***Small*** to ***moderate benefits*** for food security, as risk sharing often promotes food supply sharing (Table 6.52).

<u>Note</u>: Cell colours correspond to the large, moderate and small categories shown in Table 6.53. Dark blue = large positive; mid-blue = moderate positive; light blue = small positive; no colour = no effect; light red = small negative; mid-red = moderate negative; dark red = large negative; green = variable; ND = no data.

6.4 Managing interactions and interlinkages

Having assessed the potential of each response option for contributing to addressing mitigation, adaptation, desertification, land degradation and food security is section 6.3, this section assesses the feasibility of each response option with respect to cost, barriers, and issues of saturation and reversibility (6.4.1), before assessing the sensitivity of the response options to future climate change (6.4.2) and examining the contribution of each response option to ecosystem services (classified according to Nature's Contribution to People (IPBES 2018) and to sustainable development (assessed against the UN Sustainable Development Goals) (6.4.3). Section 6.4.4 examines opportunities for implementation of integrated response options, paving the way to potential policies examined in Chapter 7, before the consequences of delayed action are assessed in section 6.4.5.

6.4.1 Feasibility of the integrated response options with respect to costs, barriers, saturation and reversibility

For each of the response options, Tables 6.62-6.69 summarise the feasibility with respect to saturation and reversibility and cost, technological, institutional, socio-cultural and environmental and geophysical barriers (the same barrier categories used in SR15).

Many land management options face issues of saturation and reversibility; however, these are not of concern for the value chain and risk management options. Reversibility is an issue for all options that increase terrestrial carbon stock, either through increased soil carbon or changes in land cover (e.g., reforestation, afforestation), since future changes in climate or land cover could result in reduced carbon storage (Smith 2013). In addition, the benefits of options that improve land management (e.g., improved cropland management, improved grazing management) will cease if the practice is halted, reversing any potential benefits.

The cost of the response options varies substantially, with some options having relatively low cost (e.g., the cost of agroforestry is less than USD 10 tCO_2e^{-1}) while others have much higher costs (e.g., the cost of BECCS could be as much as USD 250 tCO_2e^{-1}). In addition to cost, other economic barriers may prevent implementation; for example, agroforestry is a low- cost option (Smith et al. 2014a), but lack of reliable financial support could be a barrier (Hernandez-Morcillo et al. 2018). Additionally, there are a number of reasons why even no cost options are not adopted, including risk aversion, lack of information, market structure, externalities, and policies (Jaffe 2019).

Some of the response options have technological barriers that may limit their wide-scale application in the near-term. For example, BECCS has only been implemented at small-scale demonstration facilities (Kemper 2015a); challenges exist with upscaling these options to the levels discussed in this Chapter.

Many response options have institutional and socio-cultural barriers. Institutional barriers include governance, financial incentives and financial resources. For example, management of supply chains faces challenges related to political will within trade regimes, economic laissez-faire policies that discourage interventions in markets, and the difficulties of coordination across economic sectors (Poulton et al. 2006; Cohen et al. 2009; Gilbert 2012a). Implementation of other options, e.g., BECCS, is limited by the absence of financial incentives.

Options like dietary change face socio-cultural barriers; while diets have changed in the past, they are deeply culturally embedded and behaviour change is extremely difficult to effect, even when health benefits are well known (Macdiarmid et al. 2018). For some options, the specific barrier is dependent on the region. For example, barriers to reducing food waste in industrialised countries include inconvenience, lack of financial incentives, lack of public awareness, and low prioritisation (Kummu et al. 2012; Graham-Rowe et al. 2014). Barriers in developing countries include reliability of transportation networks, market reliability, education, technology, capacity, and infrastructure (Kummu et al. 2012).

Table 6.62 Feasibility of land management response options in agriculture, considering cost, technological, institutional, socio-cultural and environmental and geophysical barriers and saturation and reversibility. See also supplementary material.

Response option	Saturation	Reversibility	Cost	Technological	Institutional	Socio-cultural	Environmental and geophysical	Context and sources
Increased food productivity								Biophysical: only if limited by climatic and environmental factors. Sources: Barnes and Thomson 2014; Martin et al. 2015; Olesen and Bindi 2002; Pretty and Bharucha 2014; Schut et al. 2016
Improved cropland management								Institutional: only in some regions (e.g., poor sustainability frameworks). Sources: Bryan et al. 2009b; Bustamante et al. 2014b; Madlener et al. 2006; Reichardt et al. 2009; Roesch-McNally et al. 2017; Singh and Verma 2007; Smith et al. 2008, 2014a
Improved grazing land management								Institutional: only in some regions (e.g., need for extension services). Sources: Herrero et al. 2016; Singh and Verma 2007; Smith et al. 2008, 2015; McKinsey & Co., 2011; Ndoro et al., 2014;
Improved livestock management								Economic: improved productivity is cost negative, but others (e.g. dietary additives) are expensive. Institutional: only in some regions (e.g. need for extension services). Sources: Herrero et al. 2016; McKinsey and Company 2009; Rojas-Downing et al. 2017b; Smith et al. 2008; Thornton et al. 2009; Beauchemin et al., 2008; Ndoro et al., 2014;
Agroforestry								Economic: low cost but may lack reliable financial support. Institutional: only in some regions (e.g., seed availability). Sources: Lillesø et al. 2011; Meijer et al. 2015; Sileshi et al. 2008; Smith et al. 2007, 2014a
Agricultural diversification								More support from extension services, access to inputs and markets, economic incentives for producing a certain crop or livestock product, research and investments focused on adapted varieties and climatic resilient systems, a combination of agricultural and non-agricultural activities (e.g., off farm jobs) are all important interventions aimed at overcoming barriers to agricultural diversification. Sources: Ahmed and Stepp 2016b; Barnes et al. 2015; Barnett and Palutikof 2015; Martin and Lorenzen 2016; Roesch-McNally et al. 2016; Waha et al. 2018
Reduced grassland conversion to cropland								Economics: Avoiding conversion is low cost, but there may be significant opportunity costs associated with foregone production of crops. Institutional: only in some regions (e.g., poor governance to prevent conversion)
Integrated water								Institutional: effective implementation is dependent on the adoption of a combination of 'hard',

management								infrastructural, and 'soft' institutional measures. <u>Socio-cultural</u>: Education can be a barrier and some strategies (e.g. site-specific water management, drip irrigation) can be expensive. Cultural / behavioural barriers are likely to be small. <u>Sources</u>: Dresner et al. 2015; Erwin 2009; Lotze et al. 2006; Thornton et al. 2009

<u>Note</u>: For saturation and reversibility, a blue cell indicates that these issues are not important, and a red cell indicates that saturation and reversibility are concerns. For the cost column, a blue cell indicates low cost (< USD10 tCO_2e^{-1} or < USD20 ha^{-1}), a yellow cell indicates medium cost (USD10-USD100 tCO_2e^{-1} or USD20-USD100 ha^{-1}), and a red cell indicates high cost (>USD100 tCO_2e^{-1} or USD200 ha^{-1}). The cost thresholds in USD tCO_2e^{-1} are from Griscom et al. (2017a); thresholds in USD ha^{-1} are chosen to be comparable, but precise conversions will depend on the response option. For the technological, institutional, socio-cultural and environmental and geophysical barriers, dark blue indicates high current feasibility (no barriers), mid-blue indicates medium current feasibility (moderate barriers) and light blue indicates low current feasibility (large barriers). Green represents variable barriers.

Response option	**Saturation**	**Reversibility**	**Cost**	**Technological**	**Institutional**	**Socio-cultural**	**Environmental and geophysical**	**Context and sources**
Improved forest management								Sources: Seidl et al. 2014
Reduced deforestation and degradation								<u>Economic</u>: requires transaction and administration costs <u>Sources</u>: Kindermann et al. 2008; Overmars et al. 2014; Busch and Engelmann 2017;
Reforestation and forest restoration								<u>Sources</u>: Strengers et al. 2008
Afforestation								<u>Sources</u>: Idris Medugu et al. 2010a; Kreidenweis et al. 2016b

Table 6.63 Feasibility of land management response options in forests, considering cost, technological, institutional, socio-cultural and environmental and geophysical barriers and saturation and reversibility. See also supplementary material.

<u>Note</u>: See footnotes for Table 6.62.

Table 6.64 Feasibility of land management response options for soils, considering cost, technological, institutional, socio-cultural and environmental and geophysical barriers and saturation and reversibility. See also supplementary material.

Response option	Saturation	Reversibility	Cost	Technological	Institutional	Socio-cultural	Environmental and geophysical	Context and sources
Increased soil organic carbon content								Institutional: only in some regions (e.g., lack of institutional capacity). Sources: Smith et al. 2008; McKinsey and Company 2009; Baveye et al. 2018; Bustamante et al. 2014b; Reichardt et al. 2009; Smith 2006; Smith et al. 2007; Wollenberg et al. 2016
Reduced soil erosion								Sources: Haregeweyn et al. 2015
Reduced soil salinisation								Barriers depend on how salinisation and sodification are implemented. Sources: Bhattacharyya et al. 2015; CGIAR 2016; Dagar et al. 2016b; Evans and Sadler 2008; Greene et al. 2016; Machado and Serralheiro 2017
Reduced soil compaction								Sources: Antille et al. 2016; Chamen et al. 2015a
Biochar addition to soil								Saturation and reversibility issues lower than for soil organic carbon. Economics: In general, biochar has high costs. However, a small amount of biochar potential could be available at negative cost, and some at low cost, depending on markets for the biochar as a soil amendment. Institutional: only in some regions (e.g., lack of quality standards). Sources: Chapter 4; Dickinson et al. 2014; Guo et al. 2016; Meyer et al. 2011; Shackley et al. 2011; Woolf et al. 2010

Note: See footnotes for Table 6.62.

Table 6.65 Feasibility of land management response options in any/other ecosystems, considering cost, technological, institutional, socio-cultural and environmental and geophysical barriers and saturation and reversibility. See also supplementary material.

Response option	Saturation	Reversibility	Cost	Technological	Institutional	Socio-cultural	Environmental and geophysical	Context and sources
Fire management								Economic: the cost of its implementation is moderate, since it requires constant maintenance, and can be excessive for some local communities. Sources: Freeman et al. 2017; Hurteau et al. 2014; North et al. 2015
Reduced landslides and								Sources: Gill and Malamud 2017; Maes et al. 2017; Noble et al. 2014

natural hazards								
Reduced pollution including acidification								Sources: Begum et al., 2011; Shah et al., 2018; Yamineva & Romppanen, 2017; WMO 2015
Management of invasive species / encroachment								Technological: in the case of natural enemies. Socio-cultural: Education can be a barrier, where populations are unaware of the damage caused by the invasive species, but cultural / behavioural barriers are likely to be small. Institutional: where agricultural extension and advice services are poorly developed. Source: Dresner et al. 2015
Restoration and reduced conversion of coastal wetlands								Economic: can be cost-effective at scale. Institutional: only in some regions (e.g., poor governance of wetland use). Socio-cultural: educational barriers (e.g., lack of knowledge of impact of wetland conversion), though cultural / behavioural barriers are likely to be small. Sources: Erwin 2009; Lotze et al. 2006
Restoration and reduced conversion of peatlands								Institutional: only in some regions (e.g., lack of inputs). Sources: Bonn et al. 2014; Worrall et al. 2009
Biodiversity conservation								Economic: While protected areas and other forms of biodiversity conservation can be cost-effective, they are often underfunded relative to needs. Institutional: There have been challenges in getting systematic conservation planning to happen, due to institutional fragmentation and overlapping mandates. Socio-cultural: Despite the fact that biodiversity conservation may provide co-benefits like water or carbon protection, local populations often have had social and cultural conflicts with protected areas and other forms of exclusionary biodiversity conservation that are imposed in a top-down fashion or which restrict livelihood options. Sources: Emerton et al. 2006; Hill et al. 2015; Langford et al. 2011; Larsen et al. 2012; Schleicher 2018; Wei et al. 2018; Wilkie et al. 2001

Note: See footnotes for Table 6.62.

Table 6.66 Feasibility of land management response options specifically for CDR, considering cost, technological, institutional, socio-cultural and environmental and geophysical barriers and saturation and reversibility. See also supplementary material.

Response option	**Saturation**	**Reversibility**	**Cost**	**Technological**	**Institutional**	**Socio-cultural**	**Environmental and geophysical**	**Context and sources**
Enhanced								Permanence not an issue on the decadal timescales. Institutional: only in some regions (e.g., lack of infrastructure

weathering of minerals								for this new technology). Socio-cultural: could occur in some regions, for example, due to minerals lying under undisturbed natural areas where mining might generate public acceptance issues. Sources: Renforth et al. 2012; Smith et al. 2016b; Taylor et al. 2016b
Bioenergy and BECCS								Economic: while most estimates indicate the cost of BECCS as less than USD200 tCO_2^{-1}, there is significant uncertainty. Technological: while there are a few small BECCS demonstration facilities, BECCS has not been implemented at scale. Sources: IPCC SR15; Chapter 7; Kemper 2015; Sanchez and Kammen 2016; Vaughan and Gough 2016

Note: See footnotes for Table 6.62.

Table 6.67 Feasibility of demand management response options, considering economic, technological, institutional, socio-cultural and environmental and geophysical barriers and saturation and reversibility. See also supplementary material.

Response option	Saturation	Reversibility	Cost	Technological	Institutional	Socio-cultural	Environmental and geophysical	Context and sources
Dietary change								Institutional: only in some regions (e.g., poorly developed dietary health advice). Sources: Hearn et al. 1998; Lock et al. 2005; Macdiarmid et al. 2018; Wardle et al. 2000
Reduced post-harvest losses								
Reduced food waste (consumer or retailer)								Specific barriers differ between developed and developing countries. Sources: Graham-Rowe et al. 2014; Kummu et al. 2012; Diaz-Ruiz et al. 2018;
Material substitution								Sources: Gustavsson et al. 2006; Ramage et al. 2017

Note: See footnotes for Table 6.62.

Table 6.68 Feasibility of supply management response options, considering cost, technological, institutional, socio-cultural and environmental and geophysical barriers and saturation and reversibility. See also supplementary material.

Response option	Saturation	Reversibility	Cost	Technological	Institutional	Socio-cultural	Environmental and geophysical	Context and sources
Sustainable sourcing								Economic: the cost of certification and sustainable sourcing can lead to higher production costs. Institutional: there are some barriers to adopting sustainable sourcing in terms of getting governments on board with market-based policies. Socio-cultural: barriers include consumers unfamiliar with sustainably sourced goods. Sources: Capone et al. 2014; Ingram et al. 2016b
Management of supply chains								Economic: Supply chain management and management of price volatility faces challenges from businesses in terms of economic costs of change. Technological: barriers like supply chain tracking. Institutional: barriers like political will against government action in markets. Sources: Cohen et al. 2009; Gilbert 2012; Poulton et al. 2006
Enhanced urban food systems								
Improved food processing and retailing								Economic: The implementation of strategies to improve the efficiency and sustainability of retail and agri-food industries can be expensive. Institutional: Successful implementation is dependent on organisational capacity, the agility and flexibility of business strategies, the strengthening of public-private policies and effectiveness of supply-chain governance.
Improved energy use in food systems								Sources: Baudron et al. 2015; Vlontzos et al. 2014

Note: See footnotes for Table 6.62.

Table 6.69 Feasibility of risk management response options, considering cost, technological, institutional, socio-cultural and environmental and geophysical barriers and saturation and reversibility. See also supplementary material.

Response option	Saturation	Reversibility	Cost	Technological	Institutional	Socio-cultural	Environmental and geophysical	Context and sources
Management of urban sprawl								There are economic and political forces that benefit from less-regulated urban development. Sources: Tan et al. 2009

Livelihood diversification								Economic: Expanded diversification can cost additional financial resources. Socio-cultural: problems with adoption of new or unfamiliar crops and livelihoods. Sources: Ahmed and Stepp 2016b; Berman et al. 2012; Ngigi et al. 2017
Use of local seeds								Economic: Local seeds are highly cost effective, and do not require new technology. Institutional: barriers from agronomy departments and businesses promoting commercial seeds. Socio-cultural: preferences for some non-local seed sourced crops. Sources: Reisman 2017; Timmermann and Robaey 2016
Disaster risk management								Economic: DRM systems can be initially costly, but usually pay for themselves over time. Institutional: some barriers in terms of getting initial support and will behind new systems. Sources: Birkmann et al. 2015b; Hallegatte 2012
Risk sharing instruments								There are few barriers to risk sharing instruments, as they are often low cost and low technology. Socio-cultural: some barriers to instruments like crop insurance, which some farmers in developing countries are not familiar with. Sources: Goodwin and Smith 2013

Note: See footnotes for Table 6.62.

6.4.2 Sensitivity of the Integrated Response Options to climate change impacts

With continued increases in warming, there are risks to the efficacy of some of the response options due to future climate change impacts, such as increased climate variability and extreme events. While many of the response options can help increase capacity to deliver adaptation benefits (section 6.3.2), beyond certain thresholds of climate impacts they may be less effective or increasingly risky options. This requires that some response options need to anticipate these climate impacts in their implementation. We outline some of these impacts below.

Agriculture response options: Increased food productivity as a response option is highly sensitive to climate change impacts. Chapter 5 (section 5.2.3.1) notes that global mean yields of some crops (maize and soybean) decrease with warming, while others (rice and wheat) increase with warming, up to a threshold of 3ºC. Similarly, improved cropland management response options that rely on crop diversification or improved varieties may face challenges in efficacy from production declines. Improved grazing land management may continue to be feasible as a response option in the future under climate change in northern regions but will likely become more difficult in tropical regions and Australia as temperature rises will reduce the carrying capacity of lands (section 5.2.3.2; Nardone et al. 2010). Improved livestock management also faces numerous challenges, particularly related to stresses on animals from temperatures, water, and diseases; overall, livestock numbers are projected to decline 7.5–9.6% by 2050 (section 5.2.3.2; Rivera-Ferre et al. 2016; Boone et al. 2018). Pastoralists may also be less likely to implement improved measures due to other risks and vulnerabilities under climate change (Thornton et al. 2009a).

The impact of climate change on agroforestry is more difficult to model than single crops in process-based crop models, as agroforestry systems are far more complex (Luedeling et al. 2014); thus, it is unknown how the efficacy of this response option might be impacted. Agricultural diversification has been promoted as an adaptive strategy to climate impacts, given that diversity is known to increase resiliency of agricultural and natural systems, such as in resistance to increased pests or diseases; it also can provide diversified income portfolios when some crops may become sensitive to climate events (Bradshaw et al. 2004; Lin 2011). Diversified farms are expected to increase in Africa by 2060 as specialised farms with single crops face challenges under climate change (Seo 2010). However, it is not known if these options and advantages of diversification have a temperature threshold beyond which they are less effective.

Reduced grassland conversion is not likely to be affected as a response option *per se* since it is directed at conserving natural grassland areas, but these areas may face increased pressures for conversion if farmers experience crop failures under climate change and need to extensify holdings to make up for losses. Lobell et al. (2013) have estimated the impacts of investment decisions to adapt to the effects of climate change on crop yields to 2050 and find that cropland will expand over 23% more land area (over 3 Mkm^2), mostly in Latin America and Sub-Saharan Africa.

Integrated water management to improve water availability and reliability of water for agricultural production is likely to become more challenging in future scenarios of water declines, which are likely to be regionally uneven (section 2.5, 6.4.4).

Forest response options: The availability of improved forest management as a response option can be impacted by climate-induced changes, including increased diseases, pests and fires (Section 4.5.1.2; Dale et al. 2001; Logan et al. 2003). These impacts will affect reforestation and afforestation response options as well. Locatelli et al. (2015d) note that climate changes will influence seedling establishment, tree growth and mortality, and the presence of invasive species and/or pests; these can be buffered with modified silvicultural practices including species selection (Pawson et al. 2013). Climate changes can also alter the sink capacity for vegetation carbon sequestration, reducing the potential for REDD, reforestation and afforestation (Bonan 2008b; Mahli et al. 2002).

Soil management: Climate changes can alter the sink capacity for soil carbon sequestration, reducing the potential for increased soil organic carbon as an option. Projected climate changes can reduce soil resilience to extreme weather, pests and biological invasion, environmental pollutants and other pressures, making reduced soil erosion and reduced soil compaction as response options harder to achieve (Smith et al. 2015). Climate change will likely increase demand for irrigation in dryland areas, which can increase risks of salinisation, diminishing the effectiveness of this response (Smith et al. 2015). Biochar additions to soil may be affected by future climatic changes, such as rising soil temperatures, but little is known given that most research on the subject is from laboratory and not *in situ* field experiments, and there are wide estimates of the stability and residence times of biochar from this literature (Gurwick et al. 2013).

Other ecosystem management: Fire management is likely to become more challenging in a changing climate; some studies suggest an 50% increase in fire occurrence by end of the century in circumboreal forests (Flannigan et al. 2009). Landslide risks are related to climate through total rainfall, rainfall intensity, air temperature and the general weather system (Gariano and Guzzetti 2016a); thus reduced landslides and natural hazards as a response option will be made more difficult by increasing storms and seasonality of rainfall events projected for many areas of the world. Reduced pollution is likely less affected by climate change and can continue to be an option despite increasing temperatures.

Conversely, some invasive species may thrive under climate change, such as moving to new areas or being less susceptible to control protocols (Hellmann et al. 2008). Conversion of coastal wetlands will be more difficult to halt if loss of productive land elsewhere encourages development on these lands, but coastal wetlands will likely adapt to increased CO_2 and higher sea levels through sediment accretion, which will also enhance their capacity to act as carbon sinks (Duarte et al. 2013). While subarctic peatlands are at risk due to warming, these are not the main peatlands that are at risk form agricultural conversion (Tarnocai 2006); these peatlands, such as those in the tropics, may be more vulnerable in hotter scenarios to water table alterations and fire risk (Gorham 1991). Biodiversity conservation, such as through protected areas or corridors, may be threatened by increased land expansion under agriculture in climate change scenarios, including the newly available land in northern climates that may become agriculturally suited (Gimona et al. 2012), lessening the effectiveness of this response option.

CDR: The efficacy of enhanced weathering is not likely to be affected by future climate changes. On the other hand, climate change will affect the productivity of bioenergy crops (Cronin et al. 2018), influencing the mitigation potential of bioenergy and BECCS (Calvin et al. 2013a; Kyle et al. 2014). There is uncertainty in the sign and magnitude of the effect of climate change on bioenergy crop yields. As a result, there is uncertainty in whether climate change will increase or decrease the potential of bioenergy and BECCS.

Demand management of value chains: For most response options in demand side management, the tools are generally not made more difficult by future climate changes. For example, dietary change is not likely to be affected by climate change, and in fact, the opposite is more likely; that diets will shift in response to climate change impacts as reflected in high prices for some staple grains and meats, the productivity of which may be reduced (Tigchelaar et al. 2018). However, there is some indication that fruit and vegetable production will also be reduced in future scenarios, making healthier diets potentially harder to achieve in some regions (Springmann et al. 2016). Reduced post-harvest losses and reduced food waste may become an even more important option if water or heat stresses under climate change reduce overall harvests. Material substitution does have risks related to the availability of products if there are declines in the growth of forest and other biomass in certain future scenarios over time, although some evidence indicates that biomass may increase in the short-term with limited warming (Boisvenue and Running 2006).

Supply management of value chains: Sustainable sourcing relies on being able to produce consumer goods sustainably (palm oil, timber, cocoa, etc), and these may be at risk; for example, areas suitable for oil palm production are estimated to decrease by 75% by 2100 (Paterson et al. 2017). Improved management of supply chains is likely to increase in importance as a tool to manage food security, given that climate change threatens to lead to more production shocks in the future (Baldos and Hertel 2015). For enhanced urban food systems, climate stresses like heat island effects or increased water scarcity in urban areas may reduce the viability of food production in certain urban systems (da Silva et al. 2012). Improved food processing and retailing and improved energy use in agriculture are not likely to be impacted by climate change.

Risk management options: Most risk management response options are not affected by climate impacts *per se*, although the increased risks that people may face will increase the need for funding and support to deploy these options. For example, disaster risk management will likely increase in importance in helping people adapt to longer-term climate changes (Begum et al. 2014); it is also likely to cost more as increased impacts of climate change, such as intensification or frequency of storm events may increase. Management of urban sprawl may also be challenged by increased migration driven by climate change, as people displaced by climate change may move to unregulated urban areas (Adamo 2010). Livelihood diversification can assist in adapting to climate changes and is not likely to be constrained as a response option, as climate-sensitive livelihoods may be replaced by others less so. Use of local seeds as an effective response options may depend on the specific types of seeds and crops used, as some may not be good choices under increased heat and water stress (Gross et al. 2017). Risk sharing instruments are unlikely to be affected by climate change, with the exception of index and crop insurance, which may become unaffordable if too many climate shocks result in insurance claims decreasing the ability of the industry to provide this tool (Mills 2005).

Cross-Chapter Box 8: Ecosystem services and Nature's Contributions to People, and their relation to the land-climate system

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This Cross-Chapter Box describes the concepts of *ecosystem services (ES)* and *nature's contributions to people (NCP)*, and their importance to climate-land interactions. ES have become a useful concept to describe the benefits that humans obtain from ecosystems and have strong relevance to sustainable land management (SLM) decisions and their outcomes, while NCP is a new approach championed by the Intergovernmental Panel on Biodiversity and Ecosystem Services (IPBES) (explained below). It is timely that this SRCCL report includes attention to ES/NCP, as the previous Special Report on Land-Use, Land-Use Change and Forestry (LULUCF) did not make use of these concepts and focused mostly on carbon fluxes in land-climate interactions (IPCC 2000). The broader mandate of SRCCL is to address not just climate but land degradation, desertification and food security issues, all of which are closely linked to the provisioning of various ES/NCP, and the Decision and Outline for SRCCL explicitly requests an examination of how desertification and degradation "impacts on ecosystem services (e.g. water, soil and soil carbon and biodiversity that underpins them)". Attention to ES/NCP is particularly important in discussing co-benefits, trade-offs and adverse side effects of potential climate change mitigation, land management, or food security response options, as many actions may have positive impacts on climate mitigation or food production but may also come with a decline in ES provisioning, or adversely impact biodiversity {see 6.4.3}. This box considers the importance of the ES/NCP concepts, how definitions have changed over time, continuing debates over operationalisation and use of these ideas, and finally concludes with how ES/NCP are treated in various chapters in this report.

While the first uses of the term "ecosystem services" appeared in the 1980s (Lele et al. 2013; Mooney and Ehrlich 1997), the roots of interest in ES extends back to the late 1960s and the extinction crisis, with concern that species decline might cause loss of valuable benefits to humankind (King 1966; Helliwell 1969; Westman 1977). While concern over extinction was explicitly linked to biodiversity loss, later ideas beyond biodiversity have animated interest in ES, including the multi-functional nature of ecosystems. A seminal paper by Costanza et al. (1997) attempted to put an economic value on the stocks of global ES and natural capital on which humanity relied. Attention to ES expanded rapidly after the Millennium Ecosystem Assessment (Millenium Ecosystem Assessment (MA) 2005), and the linkages between ES and economic valuation of these functions were addressed by the Economics of Ecosystems and Biodiversity study (TEEB 2009). The ES approach has increasingly been used in global and national environmental assessments, including the United Kingdom National Ecosystem Assessment (Watson et al. 2011), and recent and ongoing regional and global assessments organised by the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) (Díaz et al. 2015). IPBES has recently completed an assessment on land degradation and restoration that addresses a range of ES issues of relevance to the SRCCL report (IPBES 2018).

The MA defined ES as "the benefits that ecosystems provide to people," and identified four broad groupings of ES: *provisioning services* such as food, water, or timber; *regulating services* that have impacts on climate, diseases or water quality, among others; *cultural services* that provide recreational, aesthetic, and spiritual benefits; and *supporting services* such as soil formation, photosynthesis, and nutrient cycling (Millenium Ecosystem Assessment (MA) 2005). The MA emphasised that people are components of ecosystems engaged in dynamic interactions, and particularly assessed how changes in ES might impact human well-being, such as access to basic materials for living (shelter, clothing, energy); health (clean air and water); social relations (including community cohesion); security (freedom from natural disasters); and freedom of choice (the opportunity to achieve) (Millenium Ecosystem Assessment (MA) 2005). Upon publication of the MA, incorporation of ES into land use change assessments increased dramatically, including studies on how to maximise provisioning of ES alongside human well-being (Carpenter et al. 2009); how intensive food production to feed growing populations required trading off a number of important ES (Foley et al. 2005); and how including ES in GCMs indicated increasing vulnerability to ES change or loss in future climate scenarios (Schröter et al. 2005).

Starting in 2015, IPBES has introduced a new related concept to ES, that of *nature's contributions to people (NCP)*, which are defined as "all the contributions, both positive and negative, of living nature (i.e., diversity of organisms, ecosystems and their associated ecological and evolutionary processes) to the quality of life of people" (Díaz et al. 2018). NCP are divided into regulating NCP, non-material NCP, and material NCP, a different approach than used by the MA (see figure 1). However, IPBES has stressed

NCP are a particular *way to think* of ES, rather than a replacement for ES. Rather, the concept of NCP is proposed to be broader umbrella to engage a wider range of scholarship, particularly from the social sciences and humanities, and a wider range of values, from intrinsic to instrumental to relational, particularly those held by indigenous and other peoples (Redford and Adams 2009; Schröter et al. 2014; Pascual et al. 2017; Díaz et al. 2018). The differences between the MA and IPBES approaches can be seen in Table 1.

Table 1. Comparison of MA and IPBES categories and types of ES and NCP

MA category	**MA: Ecosystem Services**	**IPBES category**	**IPBES: Nature's Contributions to People**
Supporting services	Soil formation		
	Nutrient cycling		
	Primary production		
Regulating services		**Regulating Contributions**	Habitat creation and maintenance
	Pollination		Pollination and dispersal of seeds and other propagules
	Air quality regulation		Regulation of air quality
	Climate regulation		Regulation of climate
	Water regulation		Regulation of ocean acidification
	See above		Regulation of freshwater quantity, flow and timing
	Water purification and waste treatment		Regulation of freshwater and coastal water quality
	Erosion regulation		Formation, protection and decontamination of soils and sediments
	Natural hazard regulation		Regulation of hazards and extreme events
	Pest regulation and disease regulation		Regulation of organisms detrimental to humans
Provisioning Services	Fresh water	**Material Contributions**	Energy
	Food		Food and feed
	Fibre		Materials and assistance

	Medicinal and biochemical and genetic		Medicinal, biochemical and genetic resources
Cultural Services	Aesthetic values	**Nonmaterial Contributions**	Learning and inspiration
	Recreation and ecotourism		Physical and psychological experiences
	Spiritual and religious values		Supporting identities
			Maintenance of options

Sources: Millenium Ecosystem Assessment (MA) 2005; Díaz et al. 2018

While there are many similarities between ES and NCP as seen above, the IPBES decision to use the NCP concept has been controversial, with some people arguing that an additional term is superfluous, that it incorrectly associates ES with economic valuation, and that the NCP concept is not useful for policy uptake (Braat 2018; Peterson et al. 2018). Others have argued that the MA approach is outdated, did not explicitly address biodiversity, and confused different concepts, like economic goods, ecosystem functions, and general benefits (Boyd and Banzhaf 2007). Moreover, for both ES and NCP approaches, it has been difficult to make complex ecological processes and functions amenable to assessments that can be used and compared across wider landscapes, different policy actors, and multiple stakeholders (de Groot et al. 2002; Naeem et al. 2015; Seppelt et al. 2011). There remain competing categorisation schemes for ES, as well as competing metrics on how most ES might be measured (Wallace 2007; Potschin and Haines-Young 2011; Danley and Widmark 2016; Nahlik et al. 2012). The implications of these discussions for this SRCCL report is that there remain many areas of uncertainty with regard to much ES/NCP measurement and valuation, which will have ramifications for choosing response options and policies.

This report addresses ES/NCP in multiple ways. Individual chapters have used the term ES in most cases, especially since the preponderance of existing literature uses the ES terminology. For example, Chapter 2 discusses CO_2 fluxes, nutrients, and water budgets as important ES deriving from land-climate interactions. Chapters 3 and 4 discuss issues such as biomass production, soil erosion, biodiversity loss, and other ES affected by land use change. Chapter 5 discusses both ES and NCP issues surrounding food system provisioning and trade-offs.

In chapter 6, the concept of NCP is used. For example, in chapter 6 Tables 6.70 to 6.72, possible response options to respond to climate change, to address land degradation or desertification, and to ensure food security are cross-referenced against the 18 NCP identified by Díaz et al. (2018) to see where there are co-benefits and adverse side-effects. For instance, while BECCS may deliver on climate mitigation, it results in a number of adverse side-effects that are significant with regard to water provisioning, food and feed availability, and loss of supporting identities if BECCS competes against local land uses of cultural importance. Chapter 7 has an explicit section 7.2.2.2 that covers risks due to loss of biodiversity and ES and Table 7.1 that includes policy responses to various land-climate-society hazards, some of which are likely to enhance risk of loss of biodiversity and ES. A case-study on the impact of renewable energy on biodiversity and ES is also included. Chapter 7 also notes that because there is no SDG covering freshwater biodiversity and aquatic ecosystems; this policy gap may have adverse consequences for the future of rivers and associated ES.

6.4.3 Impacts of integrated response options on Nature's Contributions to People and the UN Sustainable Development Goals

In addition to evaluating the importance of our response options for climate mitigation, adaptation, land degradation, desertification and food security, it is also necessary to pay attention to other co-benefits and trade-offs that may be associated with these responses. How the different options impact progress toward the SDG can be a useful shorthand for looking at the social impacts of these response options. Similarly, looking at how these response options increase or decrease the supply of ecosystem services/NCP (see Cross-Chapter Box 8 on Ecosystem Services in this chapter) can be a useful shorthand for a more comprehensive environmental impact beyond climate and land. Such evaluations are important as response option may lead to unexpected trade-offs with social goals (or potential co-benefits) and impacts on important environmental indicators like water or biodiversity. Similarly, there may be important synergies and co-benefits associated with some response options that may increase their cost-effectiveness or attractiveness. As we note in section 6.4.4, many of these synergies are not automatic, and are dependent on well-implemented and coordinated activities in appropriate environmental contexts (6.4.4.1), often requiring institutional and enabling conditions for success and participation of multiple stakeholders (6.4.4.3).

In the following sections and tables, we evaluate each response option against 17 SDG and 18 NCP. Some of the SDG categories appear similar to each other, such as SDG 13 on "climate action" and an NCP titled "climate regulation". However, SDG 13 includes targets for both mitigation and adaptation, so options were weighed by whether they were useful for one or both. On the other hand, the NCP "regulation of climate" does not include an adaptation component, and refers to specifically to "positive or negative effects on emissions of greenhouse gases and positive or negative effects on biophysical feedbacks from vegetation cover to atmosphere, such as those involving albedo, surface roughness, long-wave radiation, evapotranspiration (including moisture-recycling) and cloud formation or direct and indirect processes involving biogenic volatile organic compounds (BVOC), and regulation of aerosols and aerosol precursors by terrestrial plants and phytoplankton" (Díaz et al. 2018).

In all tables, colours represent the direction of impact: positive (blue) or negative (brown), and the scale of the impact (dark colours for large impact and/or strong evidence to light colours for small impact and/or less certain evidence). Supplementary tables show the values and references used to define the colour coding used in all tables. In cases where there is no evidence of an interaction or at least no literature on such interactions, the cell is left blank. In cases where there are both positive and negative interactions and the literature is uncertain about the overall impact, a note appears in the box. In all cases, many of these interactions are contextual, or the literature only refers to certain co-benefits in specific regions or ecosystems, so readers are urged to consult the supplementary tables for the specific caveats that may apply.

6.4.3.1 Impacts of integrated response options on Nature's Contributions to People

Tables 6.70–6.72 summarise the impacts of the response options on NCP supply. Examples of synergies between response options and NCP include positive impacts on habitat maintenance (NCP 1) from activities like invasive species management and agricultural diversification. For the evaluation process, we considered that NCP are about ecosystems, therefore options which may have overall positive effects, but which are *not* ecosystem-based are not included; for example, improved food transport and distribution could reduce ground-level ozone and thus improve air quality, but this is not an ecosystem-based NCP. Similarly, energy efficiency measures would increase energy availability, but the 'energy' NCP refers specifically to biomass-based fuel provisioning. This necessarily means that the land management options have more direct NCP effects than the value chain or governance options, which are less ecosystem-focused.

In evaluating NCP, we have also tried to avoid 'indirect' effects – that is a response option might increase household income which then could be invested in habitat-saving actions, or dietary change would lead to conservation of natural areas, which would then led to increased water quality. Similarly, material substitution would increase wood demand, which in turn might lead to deforestation which might have water regulation effects. These can all be considered *indirect* impacts on NCP, which were not evaluated[7]. Instead, the assessment focuses as much as possible on *direct* effects only: for example, local seeds policies preserve local landraces, which *directly* contribute to 'maintenance of genetic options' for the future. Therefore, this NCP table is a conservative estimation of NCP effects; there are likely many more secondary effects, but they are too difficult to assess, or the literature is not yet complete or conclusive.

Further, many NCP trade-off with one another (Rodriguez et al 2006), so supply of one might lead to less availability of another – for example, use of ecosystems to produce bioenergy will likely lead to decreases in water availability if mono-cropped high intensity plantations are used (Gasparaos et al 2011).

Overall, several response options stand out as having co-benefits across 10 or more NCP with no adverse impacts: improved cropland management, agroforestry, forest management and forest restoration, increased soil organic content, fire management, restoration and avoided conversion of coastal wetlands, and use of local seeds. Other response options may have strengths in some NCP but require trade-offs with others. For example, reforestation and afforestation bring many positive benefits for climate and water quality but may trade-off with food production (Table 6.70). Several response options, including increased food productivity, bioenergy and BECCS, and some risk sharing instruments like crop insurance, have significant negative consequences across multiple NCP.

[7] FOOTNOTE: The exception is NCP 6, regulation of ocean acidification, which is by itself an indirect impact. Any option that sequesters CO_2 would lower the atmospheric CO_2 concentration, which then indirectly increases the seawater pH. Therefore, any action that directly increases the amount of sequestered carbon is noted in this column, but not any action that avoids land use change and therefore indirectly avoids CO_2 emissions.

Table 6.70 Impacts on Nature's Contributions to People of integrated response options based on land management

Integrated response options based on land management	Habitat creation and maintenance	Pollination and dispersal of seeds and other propagules	Regulation of air quality	Regulation of climate	Regulation of ocean acidification	Regulation of freshwater quantity, flow and timing	Regulation of freshwater and coastal water quality	Formation, protection and decontamination of soils and sediments	Regulation of hazards and extreme events	Regulation of organisms detrimental to humans	Energy	Food and feed	Materials and assistance	Medicinal, biochemical and genetic resources	Learning and inspiration	Physical and psychological experiences	Supporting identities	Maintenance of options
Increased food productivity																		
Improved cropland management																		
Improved grazing land management																		
Improved livestock management																		
Agroforestry																		
Agricultural diversification																		
Avoidance of conversion of grassland to cropland																		
Integrated water management													+ or -					
Forest management and forest restoration									+ or -				+ or -					

Reduced deforestation and degradation																		
Reforestation									+ or -									
Afforestation								+ or -	+ or -									

Increased soil organic carbon content																		
Reduced soil erosion																		
Reduced soil salinisation																		
Reduced soil compaction																		
Biochar addition to soil																		

Fire management																		
Reduced landslides and natural hazards																		
Reduced pollution including acidification																		
Management of invasive species / encroachment																		
Restoration and avoided conversion of coastal wetlands												+ or -						
Restoration and avoided conversion of peatlands																		
Biodiversity conservation												+ or -						

Enhanced weathering of minerals																		
Bioenergy and BECCS[8]																		

LEGEND:	Large positive impacts, strong evidence	Medium positive impacts, some evidence	Small positive impacts or low evidence	Low negative impacts or low evidence	Medium negative impacts, medium evidence	Large negative impacts, high evidence

Table 6.71 Impacts on Nature's Contributions to People of integrated response options based on value chain management

<u>Integrated response options based on value chain management</u>	Habitat creation and maintenance	Pollination and dispersal of seeds and other propagules	Regulation of air quality	Regulation of climate	Regulation of ocean acidification	Regulation of freshwater quantity, flow and timing	Regulation of freshwater and coastal water quality	Formation, protection and decontamination of soils and	Regulation of hazards and extreme events	Regulation of organisms detrimental to humans	Energy	Food and feed	Materials and assistance	Medicinal, biochemical and genetic resources	Learning and inspiration	Physical and psychological experiences	Supporting identities	Maintenance of options
Dietary change																		
Reduced post-harvest losses																		

[8] FOOTNOTE: Note that this refers to large areas of bioenergy crops capable of producing large mitigation benefits (> 3 GtCO2 yr^{-1}). The effect of bioenergy and BECCS on NCPs is scale and context dependent (see Cross-Chapter Box 7 in this chapter; Section 6.2).

Reduced food waste (consumer or retailer)																		
Material substitution																		
Sustainable sourcing																		
Management of supply chains																		
Enhanced urban food systems																		
Improved food processing and retail																		
Improved energy use in food systems																		

LEGEND: Large positive impacts, strong evidence; Medium positive impacts, some evidence; Small positive impacts or low evidence; Low negative impacts or low evidence; Medium negative impacts, medium evidence; Large negative impacts, high evidence

Table 6.72 Impacts on Nature's Contributions to People of integrated response options based on risk management

Integrated response options based on risk management	Habitat creation and maintenance	Pollination and dispersal of seeds and other propagules	Regulation of air quality	Regulation of climate	Regulation of ocean acidification	Regulation of freshwater quantity, flow and timing	Regulation of freshwater and coastal water quality	Formation, protection and decontamination of soils and sediments	Regulation of hazards and extreme events	Regulation of organisms detrimental to humans	Energy	Food and feed	Materials and assistance	Medicinal, biochemical and genetic resources	Learning and inspiration	Physical and psychological experiences	Supporting identities	Maintenance of options
Management of urban sprawl	Medium positive	Small positive	Large positive	Large positive		Large positive	Large positive	Large positive	Medium positive			Medium positive						
Livelihood diversification												Small positive	Small positive					
Use of local seeds	Small positive	Medium positive						Medium positive		Large positive		Large positive	Large positive	Large positive	Large positive		Large positive	Large positive
Disaster risk management									Large positive			Medium positive						
Risk sharing instruments	Low negative	Low negative		Low negative			Low negative	Medium negative		Low negative		Large positive						Low negative

LEGEND:

Large positive impacts, strong evidence	Medium positive impacts, some evidence	Small positive impacts or low evidence	Low negative impacts or low evidence	Medium negative impacts, medium evidence	Large negative impacts, high evidence

6.4.3.2 Impacts of integrated response options on the UN Sustainable Development Goals

Tables 6.73–6.75 summarise the impact of the integrated response options on the UN SDG. Some of the synergies between response options and SDG in the literature include positive poverty reduction impacts (SDG 1) from activities like improved water management or improved management of supply chains, or positive gender impacts (SDG 5) from livelihood diversification or use of local seeds. Because many land management options only produce indirect or unclear effects on SDG, we did not include these where there was no literature. Therefore, the value chain and governance options appear to offer more direct benefits for SDG.

However, it is noted that some SDG are internally difficult to assess because they contain many targets, not all of which could be evaluated (e.g., SDG 17 is about partnerships, but has targets ranging from foreign aid to debt restructuring to technology transfer to trade openness). Additionally, it is noted that some SDG contradict one another – for example, SDG 9 to increase industrialisation and infrastructure and SDG 15 to improve life on land. More industrialisation is likely to lead to increased resource demands with negative effects on habitats. Therefore, a positive association on one SDG measure might be directly correlated with a negative measure on another, and the table needs to be read with caution for that reason. The specific caveats on each of these interactions can be found in the supplementary material tables in the Chapter 6 appendix.

Overall, several response options have co-benefits across 10 or more SDG with no adverse side effects on any SDG: increased food production, improved grazing land management, agroforestry, integrated water management, reduced post-harvest losses, sustainable sourcing, livelihood diversification and disaster risk management. Other response options may have strengths in some SDG but require trade-offs with others. For example, use of local seeds bring many positive benefits for poverty and hunger reduction, but may reduce international trade (SDG 17). Other response options like enhanced urban food systems, management of urban sprawl, or management of supply chains are generally positive for many SDG but may trade-off with one, like clean water (SDG 6) or decent work (SDG 8), as they may increase water use or slow economic growth. Several response options, including avoidance of grassland conversion, reduced deforestation and degradation, reforestation and afforestation, biochar, restoration and avoided conversion of peatlands and coastlands, have trade-offs across multiple SDG, primarily as they prioritise land health over food production and poverty reduction. Several response options such as bioenergy and BECCS and some risk sharing instruments, such as crop insurance, trade-off over multiple SDG with potentially significant adverse consequences.

Overall, across both categories of both SDG and NCP, 17 of 40 options deliver co-benefits or no adverse side-effects for the full range of NCP and SDG. This include most agriculture- and soil-based land management options, many ecosystem-based land management options, improved forest management, reduced post-harvest losses, sustainable sourcing, improved energy use in food systems, and livelihood diversification. Only three options (afforestation, bioenergy and BECCS and some types of risk sharing instruments, such as crop insurance) have potentially adverse side-effects for five or more NCP or SDG.

Table 6.73 Impacts on the UN SDG of integrated response options based on land management

Integrated response options based on land management	GOAL 1: No Poverty	GOAL 2: Zero Hunger	GOAL 3: Good Health and Well-being	GOAL 4: Quality Education	GOAL 5: Gender Equality	GOAL 6: Clean Water and Sanitation	GOAL 7: Affordable and Clean Energy	GOAL 8: Decent Work and Economic Growth	GOAL 9: Industry, Innovation and Infrastructure	GOAL 10: Reduced Inequality	GOAL 11: Sustainable Cities and Communities	GOAL 12: Responsible Consumption and Production	GOAL 13: Climate Action	GOAL 14: Life Below Water	GOAL 15: Life on Land	GOAL 16: Peace and Justice Strong Institutions	GOAL 17: Partnerships to achieve the Goal
Increased food productivity																	
Improved cropland management																	
Improved grazing land management																	
Improved livestock management																	
Agroforestry																	
Agricultural diversification										+ or -							
Avoidance of conversion of grassland to cropland																	
Integrated water management																	
Forest management and forest restoration																	
Reduced deforestation and degradation	+ or -																

Reforestation	+ or -																
Afforestation																	

Increased soil organic carbon content																	
Reduced soil erosion																	
Reduced soil salinisation																	
Reduced soil compaction																	
Biochar addition to soil																	

Fire management																	
Reduced landslides and natural hazards																	
Reduced pollution including acidification																	
Management of invasive species / encroachment																	
Restoration and avoided conversion of coastal wetlands	+ or -	+ or -															
Restoration and avoided conversion of peatlands																	
Biodiversity conservation																	

Enhanced weathering of minerals																	

Bioenergy and BECCS[9]	+ or -		+ or -														

LEGEND:	Large positive impacts, strong evidence	Medium positive impacts, some evidence	Small positive impacts or low evidence	Low negative impacts or low evidence	Medium negative impacts, medium evidence	Large negative impacts, high evidence

Table 6.74 Impacts on the UN SDG of integrated response options based on value chain interventions

Integrated response options based on value chain management	GOAL 1: No Poverty	GOAL 2: Zero Hunger	GOAL 3: Good Health and Well-being	GOAL 4: Quality Education	GOAL 5: Gender Equality	GOAL 6: Clean Water and Sanitation	GOAL 7: Affordable and Clean Energy	GOAL 8: Decent Work and Economic Growth	GOAL 9: Industry, Innovation and Infrastructure	GOAL 10: Reduced Inequality	GOAL 11: Sustainable Cities and Communities	GOAL 12: Responsible Consumption and Production	GOAL 13: Climate Action	GOAL 14: Life Below Water	GOAL 15: Life on Land	GOAL 16: Peace and Justice Strong Institutions	GOAL 17: Partnerships to achieve the Goal
Dietary change																	
Reduced post-harvest losses																	
Reduced food waste (consumer or retailer)																	
Material substitution																	

[9] FOOTNOTE: Note that this refers to large areas of bioenergy crops capable of producing large mitigation benefits (> 3 $GtCO_2$ yr^{-1}). The effect of bioenergy and BECCS on SDG is scale and context dependent (see Cross-Chapter Box 7 in this chapter; Section 6.2).

Sustainable sourcing																	
Management of supply chains																	
Enhanced urban food systems																	
Improved food processing & retail																	
Improved energy use in food systems																	

LEGEND:	Large positive impacts, strong evidence	Medium positive impacts, some evidence	Small positive impacts or low evidence	Small negative impacts or low evidence	Medium negative impacts, medium evidence	Large negative impacts, high evidence

Table 6.75 Impacts on the UN SDG of integrated response options based on risk management

<u>Integrated response options based on risk management</u>	GOAL 1: No Poverty	GOAL 2: Zero Hunger	GOAL 3: Good Health and Well-being	GOAL 4: Quality Education	GOAL 5: Gender Equality	GOAL 6: Clean Water and Sanitation	GOAL 7: Affordable and Clean Energy	GOAL 8: Decent Work and Economic Growth	GOAL 9: Industry, Innovation and Infrastructure	GOAL 10: Reduced Inequality	GOAL 11: Sustainable Cities and Communities	GOAL 12: Responsible Consumption and Production	GOAL 13: Climate Action	GOAL 14: Life Below Water	GOAL 15: Life on Land	GOAL 16: Peace and Justice Strong Institutions	GOAL 17: Partnerships to achieve the Goal
Management of urban sprawl																	
Livelihood diversification																	
Use of local seeds																	
Disaster risk management																	

Risk sharing instruments												+ or -					

LEGEND:	Large positive impacts, strong evidence	Medium positive impacts, some evidence	Small positive impacts or low evidence	Small negative impacts or low evidence	Medium negative impacts, medium evidence	Large negative impacts, high evidence

6.4.4 Opportunities for implementation of Integrated Response Options

6.4.4.1 Where can the response options be applied?

As shown in Section 6.1.3, a large part of the land area is exposed to overlapping land challenges, especially in villages, croplands and rangelands. The deployment of land management responses may vary with local exposure to land challenges. For instance, with croplands exposed to a combination of land degradation, food insecurity and climate change adaptation challenges, maximising the co-benefits of land management responses would require selecting responses having only co-benefits for these 3 overlapping challenges, as well as for climate change mitigation which is a global challenge. Based on these criteria, Figure 6.6 shows the potential deployment area of land management responses across land use types (or anthromes).

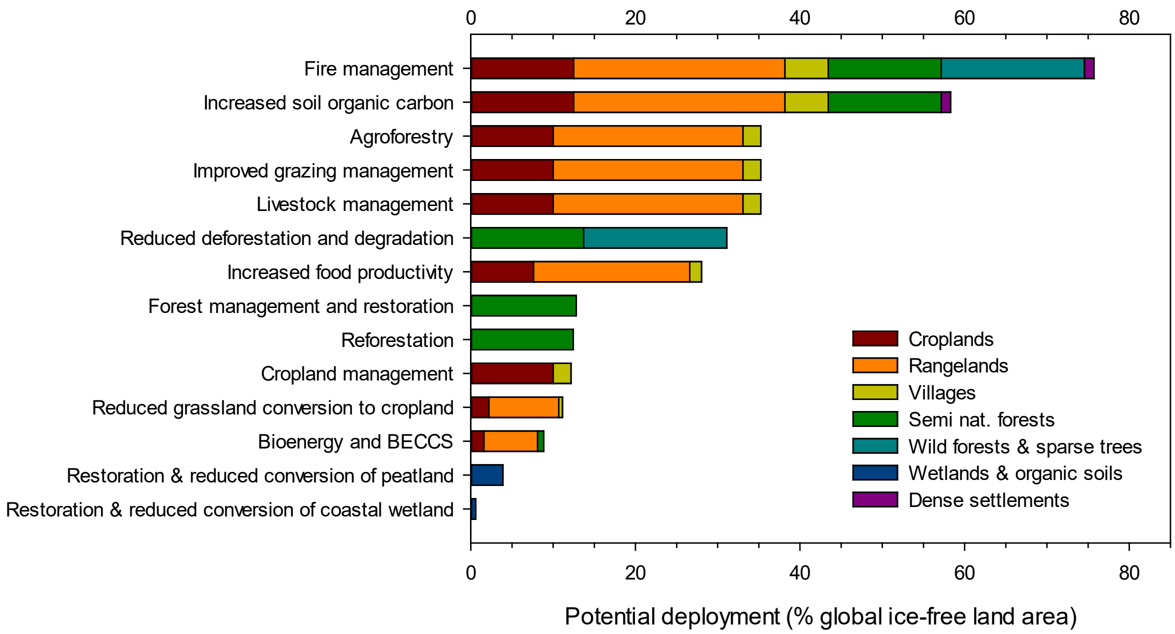

Figure 6.6 Potential deployment area of land management responses (see Table 6.1) across land use types (or anthromes, see section 6.3), when selecting responses having only co-benefits for local challenges and for climate change mitigation and no large adverse side-effect on global food security. See Figure 6.2 for the criteria used to map challenges (desertification, land degradation, climate change adaptation, chronic undernourishment, biodiversity, groundwater stress and water quality) considered. No response option was identified for barren lands.

Land management responses having co-benefits across the range of challenges, including climate change mitigation, could be deployed between one (coastal wetlands, peatlands, forest management and restoration, reforestation) and 5 (increased soil organic carbon) or 6 (fire management) land use types (Figure 6.6). Fire management and increased soil organic carbon have a large potential since they could be deployed with mostly co-benefits and few adverse effects over 76 and 58% of the ice-free land area. In contrast, other responses have a limited area-based potential due to biophysical constraints (e.g., limited extent of organic soils and of coastal wetlands for conservation and restoration responses), or due to the occurrence of adverse effects. Despite strong co-benefits for climate change mitigation, the deployment of bioenergy and BECCS would have co-benefits on only 9% of the ice-free land area (Figure 6.6), given adverse effects of this response option for food security, land degradation, climate change adaptation and desertification (see Tables 6.62-6.69).

Without including the global climate change mitigation challenge, there are up to 5 overlapping challenges on lands which are not barren (Fig. 6.7A, calculated from the overlay of individual challenges shown in Fig. 6.2) and up to 9 land management response options having only co-benefits for these challenges and for climate change mitigation (Fig. 6.7B). Across countries, the mean number of land management response options with mostly co-benefits declines ($p<0.001$, Spearman rank order correlation) with the mean number of land challenges. Hence, the higher the number of land challenges per country, the fewer the land management response options having only co-benefits for the challenges encountered.

Enabling conditions (see Section 6.1.2.2) for the implementation of land management responses partly depend upon human development (economics, health and education) as estimated by a country scale composite index, the Human Development index (HDI, United Nations Development Program, 2018) (Figure 6.7C). Across countries, HDI is negatively correlated ($p<0.001$, Spearman rank order correlation) with the mean number of land challenges. Therefore, on a global average, the higher the number of local challenges faced, the fewer the land management responses having only co-benefits and the lower the human development (Figure 6.7) that could favour the implementation of these responses.

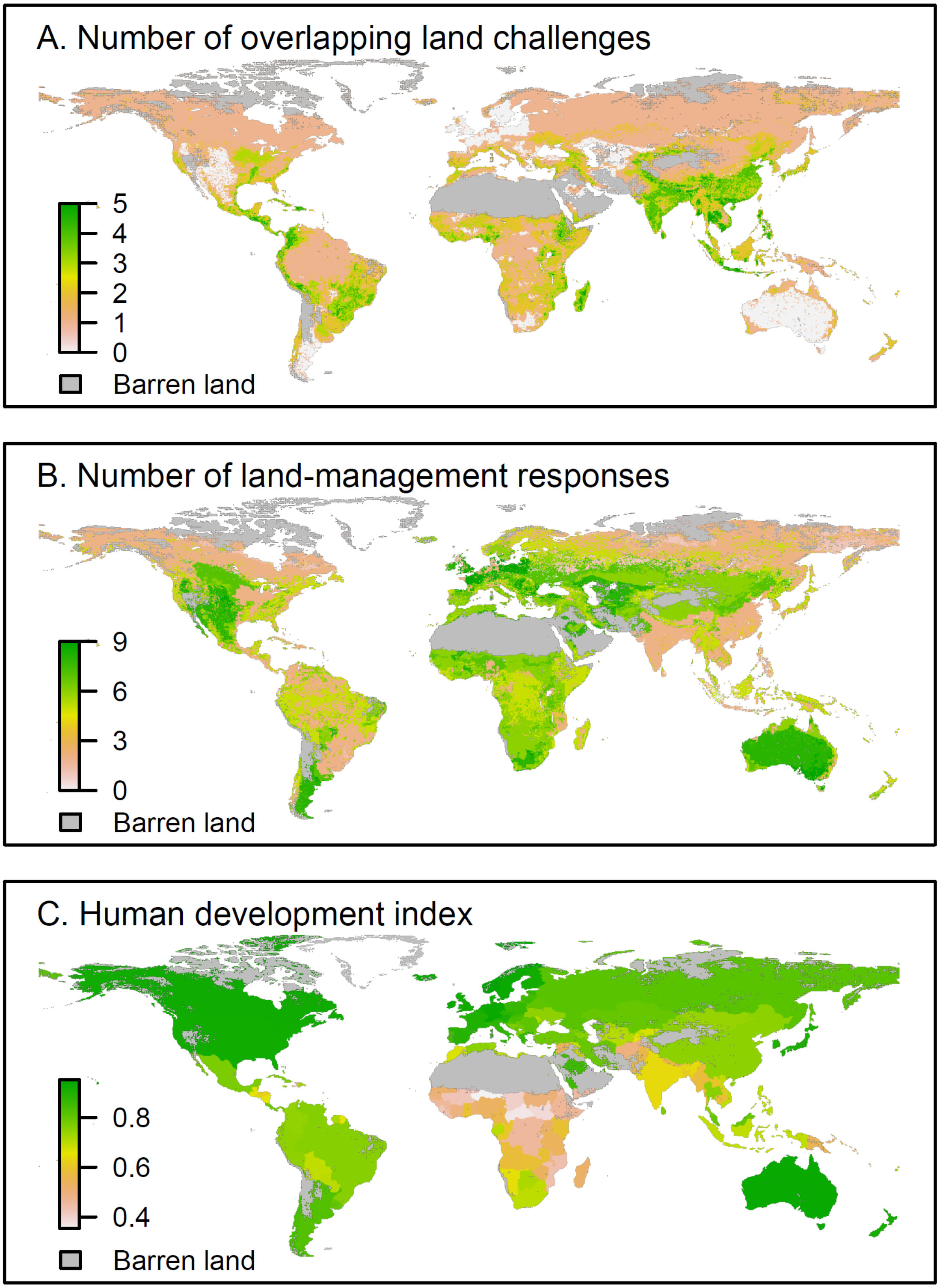

Figure 6.7 Global distributions of (A) number of overlapping land challenges (desertification, land degradation, climate change adaptation, chronic undernourishment, biodiversity, groundwater stress and water quality, see Fig. 6.2); (B) number of land management responses providing medium to large co-benefits and no adverse side-effects (see Fig. 6.6) across challenges; (C) Human Development Index (HDI) by country. The Human Development Index (United Nations Development Programme, 2018) is a country based composite statistical index measuring average achievement in three basic dimensions of human development

a long and healthy life (estimated from life expectancy at birth), knowledge (estimated from years of schooling) and a decent standard of living (estimated from gross national income per capita)

6.4.4.2 Interlinkages and response options in future scenarios

This section assesses more than eighty articles quantifying the effect of various response options in the future, covering a variety of response options and land-based challenges. These studies cover spatial scales ranging from global (Popp et al. 2017; Fujimori et al. 2018a) to regional (Calvin et al. 2016a; Frank et al. 2015) to country-level (Gao and Bryan 2017; Pedercini et al. 2018). This section focuses on models that can quantify interlinkages between response options, including agricultural economic models, land system models, and integrated assessment models. The IAM and non-IAM literature, however, is also categorised separately to elucidate what is and is not included in global mitigation scenarios, like those included in the SR15. Results from bottom-up studies and models (e.g., Griscom et al. 2017a) are assessed in Section 6.2-6.3.

Response options in future scenarios:

More than half of the 40 land-based response options discussed in this chapter are represented in global IAMs models used to develop and analyse future scenarios, either implicitly or explicitly (Table 6.76). For example, all IAMs include improved cropland management, either explicitly through technologies that improve N use efficiency (Humpenöder et al. 2018a) or implicitly through marginal abatement cost curves that link reductions in N_2O emissions from crop production to carbon prices (most other models).

However, the literature discussing the effect of these response options on land-based challenges is more limited (Table 6.76). Fifty-seven studies (forty-three IAM studies) articulate the effect of response options on mitigation, with most including bioenergy and BECCS or a combination of reduced deforestation, reforestation, and afforestation. Thirty-seven studies (twenty-one IAM studies) discuss the implications of response options on food security, usually using food price as a metric. While a small number of non-IAM studies examine the effects of response options on desertification (three studies) and land degradation (five studies), no IAM studies were identified. However, some studies quantify these challenges indirectly using IAMs either via climate outputs from the RCPs (Huang et al. 2016) or by linking IAMs to other land and ecosystem models (Brink et al. 2018; UNCCD 2017).

For many of the scenarios in the literature, land-based response options are included as part of a suite of mitigation options (Popp et al. 2017; van Vuuren et al. 2015a). As a result, it is difficult to isolate the effect of an individual option on land-related challenges. A few studies focus on specific response options (Calvin et al. 2014a; Popp et al. 2014b; Kreidenweis et al. 2016b; Humpenöder et al. 2018a), quantifying the effect of including an individual option on a variety of sustainability targets.

Table 6.76 Number of IAM and non-IAM Studies Including Specific Response Options (rows) and Quantifying Particular Land Challenges (columns). The third column shows how many IAM models include the individual response option; red indicates all models include the option, orange indicates more than half of all models, yellow indicates less than half, and white indicates no models. The remaining columns show challenges related to climate change (C), mitigation (M), adaptation (A), desertification (D), land degradation (L), food security (F), and biodiversity/ecosystem services/sustainable development (B). The colour indicates the number of total studies, with 0 (white), 1-5 (green), 6-10 (light blue), 11-15 (dark blue), and 16 or more (purple). Additionally, counts of total (left value) and IAM-only (right value) studies are included. Some IAMs include agricultural economic models which can also be run separately; these models are not counted as IAM literature when used on their own. Studies using a combination of IAMs and non-IAMs are included in the total only. A complete list of studies is included in the supplementary material.

Category	Response Option	IAMs[a]	Studies [Total/IAM]						
			C	M	A	D	L[b]	F[c]	B
Land Management	Increased food productivity		1/1	18/14	5/1	2/0	3/0	18/9	12/6
	Improved cropland management		0/0	15/11	7/2	0/0	0/0	13/6	7/4
	Improved grazing land management		0/0	1/0	1/0	0/0	0/0	1/0	0/0
	Improved livestock management		0/0	10/6	1/0	2/0	2/0	7/3	5/2
	Agroforestry		0/0	0/0	0/0	0/0	0/0	0/0	0/0
	Agricultural diversification		0/0	0/0	0/0	0/0	0/0	0/0	0/0
	Reduced grassland conversion to cropland		0/0	2/2	0/0	0/0	0/0	1/1	1/1
	Integrated water management		1/0	17/12	5/2	0/0	2/0	13/7	20/13
	Improved forest management		0/0	2/0	0/0	1/0	1/0	2/0	2/0
	Reduced deforestation and degradation		2/2	24/20	1/0	1/0	1/0	14/9	14/8
	Reforestation and forest restoration		3/3	19/18	1/1	1/0	2/0	9/8	9/6
	Afforestation		3/3	24/21	2/1	0/0	0/0	10/9	8/7
	Increased soil organic carbon content		0/0	3/1	0/0	0/0	0/0	1/1	0/0
	Reduced soil erosion		0/0	0/0	0/0	0/0	0/0	0/0	0/0
	Reduced soil salinisation		0/0	0/0	0/0	0/0	0/0	0/0	0/0
	Reduced soil compaction		0/0	0/0	0/0	0/0	0/0	0/0	0/0
	Biochar addition to soil		0/0	0/0	0/0	0/0	0/0	0/0	0/0
	Fire management		0/0	1/1	0/0	0/0	0/0	0/0	0/0
	Reduced landslides and natural hazards		0/0	0/0	0/0	0/0	0/0	0/0	0/0
	Reduced pollution including acidification		2/2	18/16	2/1	0/0	0/0	10/7	6/6
	Management of invasive species / encroachment		0/0	0/0	0/0	0/0	0/0	0/0	0/0
	Restoration and reduced conversion of coastal wetlands		0/0	0/0	0/0	1/0	1/0	0/0	1/0
	Restoration and reduced conversion of peatlands		0/0	0/0	0/0	0/0	0/0	0/0	0/0
	Biodiversity conservation		1/0	7/3	0/0	1/0	3/0	4/2	8/1
	Enhanced weathering of minerals		0/0	0/0	0/0	0/0	0/0	0/0	0/0
	Bioenergy and BECCS		5/4	50/40	7/4	0/0	2/0	25/18	21/13
Value Chain Management	Dietary change		0/0	15/12	1/0	2/0	2/0	13/9	10/7
	Reduced post-harvest losses		0/0	5/4	0/0	0/0	0/0	2/2	2/1
	Reduced food waste (consumer or retailer)		0/0	6/4	0/0	0/0	0/0	4/2	3/1
	Material substitution		0/0	0/0	0/0	0/0	0/0	0/0	0/0
	Sustainable sourcing		0/0	0/0	0/0	0/0	0/0	0/0	0/0
	Management of supply chains		1/1	11/9	8/1	2/0	3/0	17/9	7/3
	Enhanced urban food systems		0/0	0/0	0/0	0/0	0/0	0/0	0/0
	Improved food processing and retailing		0/0	0/0	0/0	0/0	0/0	0/0	0/0
	Improved energy use in food systems		0/0	0/0	0/0	0/0	0/0	0/0	0/0
Risk Management	Management of urban sprawl		0/0	0/0	0/0	1/0	1/0	0/0	1/0
	Livelihood diversification		0/0	0/0	0/0	0/0	0/0	0/0	0/0
	Use of local seeds		0/0	0/0	0/0	0/0	0/0	0/0	0/0
	Disaster risk management		0/0	0/0	0/0	0/0	0/0	0/0	0/0

	Risk sharing instruments		0/0	0/0	0/0	0/0	0/0	0/0	0/0

[a] Only IAMs that are used in the papers assessed are included in this column.
[b] There are many indicators for land degradation (see Chapter 4). In this table, studies are categorised as quantifying land degradation if they explicitly discuss land degradation.
[c] Studies are categorised is quantifying food security if they report food prices or the population at risk of hunger.

Interactions and Interlinkages between Response Options:

The effect of response options on desertification, land degradation, food security, biodiversity, and other sustainable development goals depends strongly on which options are included and the extent to which they are deployed. For example, sections 2.6, 6.3.6, and the Cross-Chapter Box 7 on Bioenergy note that bioenergy and BECCS has a large mitigation potential but could potentially have adverse side effects for land degradation, food security, and other sustainable development goals. Global modelling studies demonstrate that these effects are dependent on scale. Increased use of bioenergy can result in increased mitigation (Figure 6.8, Panel A) and reduced climate change, but can also lead to increased energy cropland expansion (Figure 6.8, Panel B), and increased competition for land resulting in increased food prices (Figure 6.8, Panel C). However, the exact relationship between bioenergy deployment and each sustainability target depends a number of other factors, including the feedstock used, the underlying socioeconomic scenario, assumptions about technology and resource base, the inclusion of other response options, and the specific model used (Calvin et al. 2014a; Clarke and Jiang 2014b; Popp et al. 2014b, 2017; Kriegler et al. 2014).

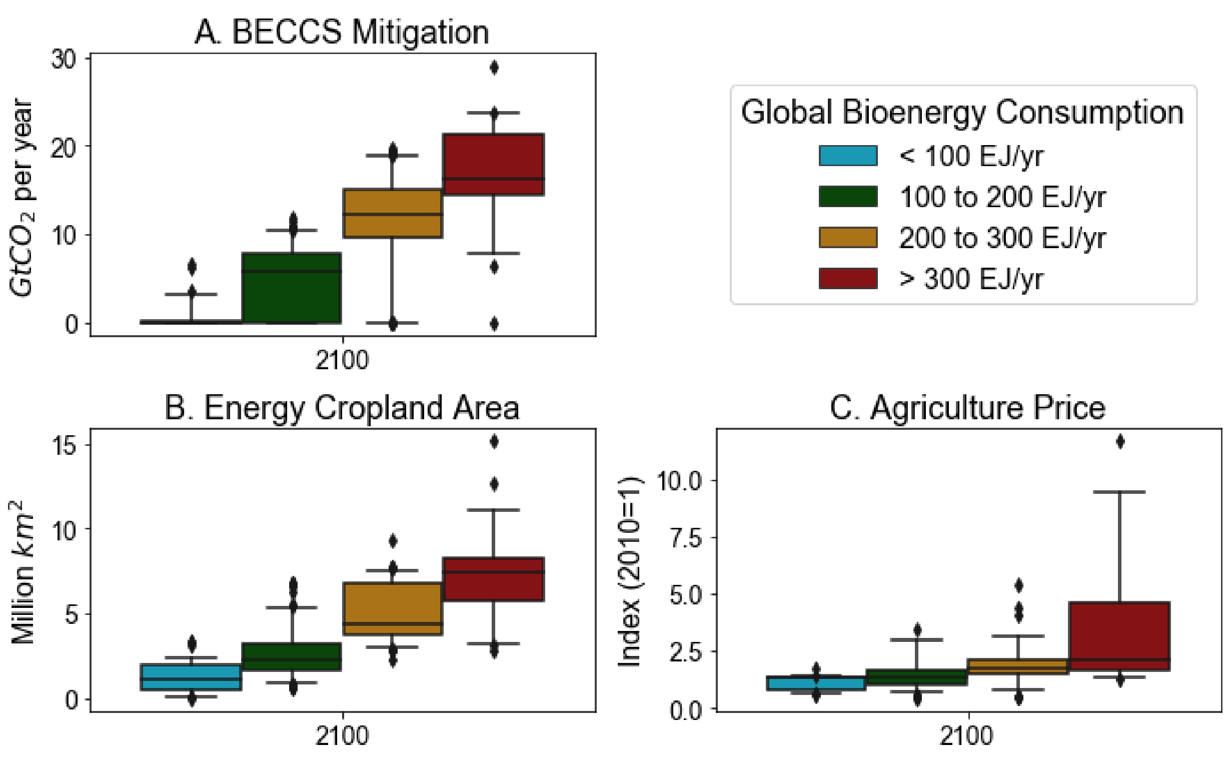

Figure 6.8 Correlation between Bioenergy Use and Other Indicators. Panel A shows global CO_2 sequestration by BECCS in 2100. Panel B shows global energy cropland area in 2100. Panel C shows agricultural prices in 2100 indexed to 2010. Data are binned based on the amount of bioenergy used globally in 2100. All scenario data that include both bioenergy consumption and the variable of interest are included in the figure; the resulting number of scenarios varies per panel with 352 in panel A, 262 in panel B, and 172 in panel C. The

boxes represent the interquartile range (i.e., the middle 50% of all scenarios), the line in the middle of the box represents the median, and the whiskers represent the 5 to 95% range of scenarios. Data is from an update of the IAMC Scenario Explorer developed for the SR15 (Huppmann et al. 2018; Rogelj et al. 2018b).

The previous sections have examined the effects of individual land-response options on multiple challenges. A number of studies using global modelling and analyses have examined interlinkages and interaction effects among land response options by incrementally adding or isolating the effects of individual options. Most of these studies focus on interactions with bioenergy and BECCS (Table 6.77). Adding response options that require land (e.g., reforestation, afforestation, reduced deforestation, avoided grassland conversion, or biodiversity conservation), results in increased food prices (Calvin et al. 2014a; Humpenöder et al. 2014a; Obersteiner et al. 2016a; Reilly et al. 2012a) and potentially increased temperature through biophysical climate effects (Jones et al. 2013). However, this combination can result in reduced water consumption (Hejazi et al. 2014c), reduced cropland expansion (Calvin et al. 2014a; Humpenöder et al. 2018a), increased forest cover (Calvin et al. 2014a; Humpenöder et al. 2018a; Wise et al. 2009a) and reduced biodiversity loss (Pereira et al. 2010), compared to scenarios with bioenergy and BECCS alone. While these options increase total mitigation, they reduce mitigation from bioenergy and BECCS as they compete for the same land (Wu et al. 2019; Baker et al. 2019a; Calvin et al. 2014a; Humpenöder et al. 2014a).

The inclusion of land-sparing options (e.g., dietary change, increased food productivity, reduced food waste, management of supply chains) in addition to bioenergy and BECCS results in reduced food prices, reduced agricultural land expansion, reduced deforestation, reduced mitigation costs, reduced water use, and reduced biodiversity loss (Bertram et al. 2018; Wu et al. 2019; Obersteiner et al. 2016a; Stehfest et al. 2009; van Vuuren et al. 2018a). These options can increase bioenergy potential, resulting in increased mitigation than from bioenergy and BECCS alone (Wu et al. 2019; Stehfest et al. 2009; Favero and Massetti 2014).

Other combinations of land response options create synergies, alleviating land pressures. The inclusion of increased food productivity and dietary change can increase mitigation, reduce cropland use, reduce water consumption, reduce fertiliser application, and reduce biodiversity loss (Springmann et al. 2018c; Obersteiner et al. 2016a). Similarly, improved livestock management combined with increased food productivity can reduce agricultural land expansion (Weindl et al. 2017). Reducing disturbances (e.g., fire management) in combination with afforestation can increase the terrestrial carbon sink, resulting in increased mitigation potential and reduced mitigation cost (Le Page et al. 2013a).

Studies including multiple land response options often find that the combined mitigation potential is not equal to the sum of individual mitigation potential as these options often share the same land. For example, including both afforestation and bioenergy and BECCS results in a cumulative reduction in GHG emissions of 1200 $GtCO_2$ between 2005 and 2100, which is much lower than the sum of the contributions of bioenergy (800 $GtCO_2$) and afforestation (900 $GtCO_2$) individually (Humpenöder et al. 2014a). More specifically, Baker et al. (2019a) find that woody bioenergy and afforestation are complementary in the near-term, but become substitutes in the long-term, as they begin to compete for the same land. Similarly, the combined effect of increased food productivity, dietary change, and reduced waste on GHG emissions is less than the sum of the individual effects (Springmann et al. 2018c).

Table 6.77 Interlinkages between bioenergy and BECCS and other response options. Table indicates the combined effects of multiple land-response options on climate change (C), mitigation (M), adaptation (A), desertification (D), land degradation (L), food security (F), and biodiversity/ecosystem services/sustainable development (O). Each cell indicates the implications of adding the option specified in the row in addition to bioenergy and BECCS. Blue colours indicate positive interactions (e.g., including the option in the second column increases mitigation, reduces cropland area, or reduces food prices relative to bioenergy and BECCS alone). Red colours indicate negative interactions; yellow indicates mixed interactions (some positive, some negative). Note that only response option combinations found in the assessed literature are included in the interest of space.

	C[a]	M[b]	A	D	L[c]	F	O[d]	Context and Sources
Increased food productivity								Sources: Humpenöder et al. 2018a; Obersteiner et al. 2016a
Increased food productivity; improved livestock management								Sources: van Vuuren et al. 2018a
Improved cropland management								Sources: Humpenöder et al. 2018a
Integrated water management								O: Reduces water use, but increases fertiliser use. Sources: Humpenöder et al. 2018a
Reduced deforestation								Sources: Calvin et al. 2014a; Humpenöder et al. 2018a
Reduced deforestation, Avoided grassland conversion								O: Reduces biodiversity loss and fertiliser, but increases water use. Sources: Calvin et al. 2014a; Obersteiner et al. 2016a
Reforestation								Sources: Reilly et al. 2012a
Reforestation, Afforestation, Avoided grassland conversion								Sources: Calvin et al. 2014a; Hejazi et al. 2014a; Jones et al. 2013
Afforestation								Sources: Humpenöder et al. 2014a
Biodiversity conservation								M: Reduces emissions but also reduces bioenergy potential. O: Reduces biodiversity loss but increases water use. Sources: Obersteiner et al. 2016a; Wu et al. 2019
Reduced pollution								Sources: van Vuuren et al. 2018a
Dietary change								Sources: Bertram et al. 2018; Stehfest et al. 2009; Wu et al. 2019
Reduced food waste; dietary change								Sources: van Vuuren et al. 2018a
Management of supply chains								Sources: Favero and Massetti 2014
Management of supply chains; increased productivity								Sources: Wu et al. 2019
Reduced deforestation; Improved cropland management; Improved food productivity; Integrated water management								Sources: Humpenöder et al. 2018a
Reduced deforestation; Management of Supply Chains; Integrated Water Management; Improved cropland management; Increased food productivity								Sources: Bertram et al. 2018
Reduced deforestation; Management of Supply Chains; Integrated Water Management; Improved cropland management;								Sources: Bertram et al. 2018

Increased food productivity; dietary change								

[a] Includes changes in biophysical effects on climate (e.g., albedo)
[b] Either through reduced emissions, increased mitigation, reduced mitigation cost, or increased bioenergy potential. For increased mitigation, a positive indicator in this column only indicates that total mitigation increases and not that the total is greater than the sum of the individual options.
[c] Uses changes in cropland or forest as an indicator (reduced cropland expansion or reduced deforestation are considered positive)
[d] Includes changes in water use or scarcity, fertiliser use, or biodiversity

Land-related response options can also interact with response options in other sectors. For example, limiting deployment of a mitigation response option will either result in increased climate change or additional mitigation in other sectors. A number of studies have examined limiting bioenergy and BECCS. Some such studies show increased emissions (Reilly et al. 2012a). Other studies meet the same climate goal, but reduce emissions elsewhere *via* reduced energy demand (Grubler et al. 2018; van Vuuren et al. 2018a), increased fossil CCS, nuclear energy, energy efficiency and/or renewable energy (van Vuuren et al. 2018a; Rose et al. 2014b; Calvin et al. 2014a; van Vuuren et al. 2017b), dietary change (van Vuuren et al. 2018a), reduced non-CO_2 emissions (van Vuuren et al. 2018a), or lower population (van Vuuren et al. 2018a). The co-benefits and adverse side-effects of non-land mitigation options are discussed in SR15, Chapter 5. Limitations on bioenergy and BECCS can result in increases in the cost of mitigation (Kriegler et al. 2014; Edmonds et al. 2013a). Studies have also examined limiting CDR, including reforestation, afforestation, and bioenergy and BECCS (Kriegler et al. 2018a,b). These studies find that limiting CDR can increase mitigation costs, increase food prices, and even preclude limiting warming to less than 1.5ºC above pre-industrial levels (Kriegler et al. 2018a,b; Muratori et al. 2016).

In some cases, the land challenges themselves may interact with land-response options. For example, climate change could affect the production of bioenergy and BECCS. A few studies examine these effects, quantifying differences in bioenergy production (Calvin et al. 2013a; Kyle et al. 2014) or carbon price (Calvin et al. 2013a) as a result of climate change. Kyle et al. (2014) finds increase in bioenergy production due to increases in bioenergy yields, while Calvin et al. (2013a) finds declines in bioenergy production and increases in carbon price due to the negative effects of climate on crop yield.

Gaps in the Literature:

Not all of the response options discussed in this chapter are included in the assessed literature, and many response options are excluded from the IAM models. The included options (e.g. bioenergy and BECCS; reforestation) are some of the largest in terms of mitigation potential (see Section 6.3). However, some of the options excluded also have large mitigation potential. For example, biochar, agroforestry, restoration/avoided conversion of coastal wetlands, and restoration/avoided conversion of peatland all have mitigation potential of ~1 $GtCO_2$ yr^{-1} (Griscom et al. 2017). Additionally, quantifications of and response options targeting land degradation and desertification are largely excluded from the modelled studies, with a few notable exceptions (Wolff et al. 2018; Gao and Bryan 2017; Brink et al. 2018; UNCCD 2017). Finally, while a large number of papers have examined interactions between bioenergy and BECCS and other response options, the literature examining other combinations of response options is more limited.

6.4.4.3 Resolving challenges in response option implementation

The 40 response options assessed in this chapter face a variety of barriers to implementation that require action across multiple actors to overcome (section 6.4.1). Studies have noted that while adoption of response options by individuals may depend on individual assets and motivation, larger structural and institutional factors are almost always equally important if not more so (Adimassu et al. 2016; Djenontin et al. 2018), though harder to capture in research variables (Schwilch et al. 2014). These institutional and governance factors can create an enabling environment for SLM practices, or challenges to their adoption (Adimassu et al. 2013). Governance factors include the institutions that manage rules and policies, the social norms and collective actions of participants (including civil society actors and the private sector), and the interactions between them (Ostrom 1990; Huntjens et al. 2012; Davies 2016). Many of Ostrom's design principles for successful governance can be applied to response options for SLM; these principles are: (1) clearly defined boundaries; (2) understanding of both benefits and costs; (3) collective choice arrangements; (4) monitoring; (5) graduated sanctions; (6) conflict-resolution mechanisms; (7) recognition of rights; and (8) nested (multi-scale) approaches. Unfortunately, studies of many natural resources and land management policy systems in developing countries in particular often show the opposite: a lack of flexibility, strong hierarchical tendencies, and a lack of local participation in institutional frameworks (Ampaire et al. 2017). Analysis of government effectiveness (GE)– defined as quality of public services, policy formulation and implementation, civil service and the degree of its independence from political pressures as well as credibility of the government's commitment to its policies (Kaufman et al. 2010) – has been shown to play a key role in land management. GE mediates land user actions on land management and investment, and government policies and laws can help land users adopt sustainable land management practices (Nkonya et al. 2016) (Figure 6.9).

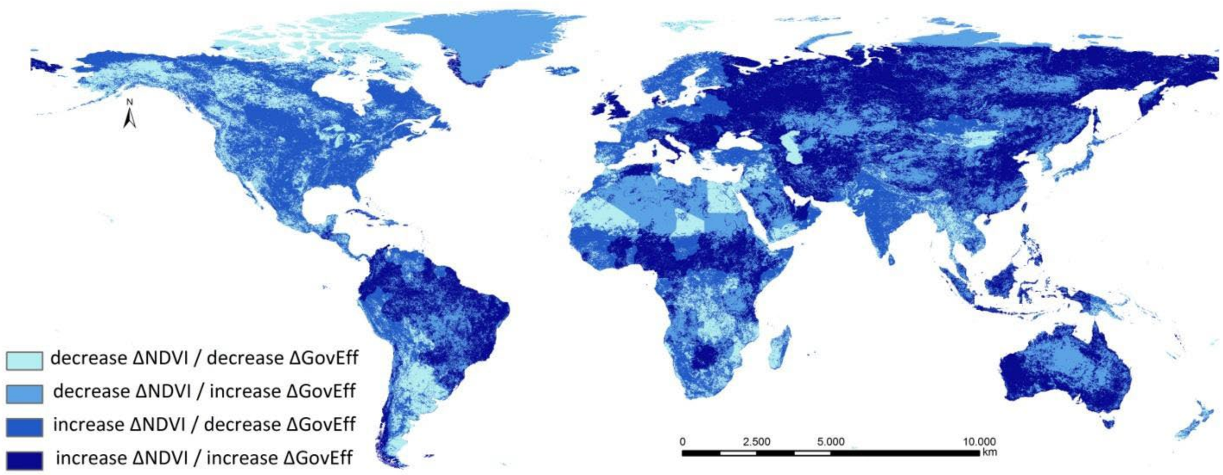

Figure 6.9 Relationship between changes in government effectiveness and changes in land management

Notes: ΔNDVI = Change in Normalized Difference Vegetation Index (baseline year 2001, Endline year 2010). Source of NDVI data: MODIS ΔGovEff = Change in Government effectiveness (baseline year 2001, Endline year 2010). Source of Government effectiveness: World Bank. Source: Nkonya et al 2016.

It is simply not a matter of putting the 'right' institutions or policies in place, however, as governance can be undermined by inattention to power dynamics (Fabinyi et al. 2014). Power shapes how actors gain access and control over resources, and negotiate, transform and adopt certain response options or not. These variable dynamics of power between different levels and stakeholders have an impact on the ability to implement different response options. The inability of many national governments to address social exclusion in general will have an effect on the implementation of many response options. Further, response options themselves can become avenues for actors to exert power claims over others

(Nightingale 2017). For example, there have been many concerns that reduced deforestation and degradation projects run the risk of reversing trends towards decentralisation in forest management and create new power disparities between the state and local actors (Phelps et al. 2010). Below we assess how two important factors, the involvement of stakeholders and the coordination of action across scales, will help in moving from response options to policy implementation, a theme chapter 7 takes up in further detail.

Involvement of stakeholders

There are a wide range of stakeholders that are necessary for successful land, agricultural and environmental policy, and implementing response options requires that a range of actors, including businesses, consumers, land managers, indigenous and local communities, scientists, and policymakers work together for success. Diverse stakeholders have a particularly important role to play in defining problems, assessing knowledge and proposing solutions (Phillipson et al. 2012; Stokes et al. 2006). Lack of connection between science knowledge and on-the-ground practice has hampered adoption of many response options in the past; simply presenting 'scientifically' derived response options is not enough (Marques et al. 2016). For example, the importance of recognising and incorporating local knowledge (LK) and indigenous knowledge (IK) is increasingly emphasised in successful policy implementation (see Cross-Chapter Box 13 on Indigenous Knowledge, Chapter 7), as local practices of water management, soil fertility management, improved grazing, restoration and sustainable management of forests are often well-aligned with response options assessed by scientists (Marques et al. 2016).

Stakeholder engagement is an important approach for successful environmental and climate policy and planning. Tools such as stakeholder mapping, in which affected and interested parties are identified and described in terms of their interrelationships and current or future objectives and aspirations, and scenario-based stakeholder engagement, which combines stakeholder analysis with climate scenarios, are increasingly being applied to facilitate better planning outcomes (Tompkins et al. 2008; Pomeroy & Douvere 2008; Star et al. 2016). Facilitated dialogues early in design processes have shown good success in bringing multiple and sometimes conflicting stakeholders to the table to discuss synergies and trade-offs around policy implementation (Gopnik et al 2012). Knowledge exchange, social learning, and other concepts are also increasingly being incorporated into understandings of how to facilitate sustainable land management (Djenontin et al. 2018), as evidence suggests that negotiating the complexity of SESs requires flexible learning arrangements in particular for multiple stakeholders (Gerlak and Heikkila 2011; Armitage et al. 2018; Heikkila and Gerlak 2018). Social learning has been defined as "a change in understanding and skills that becomes situated in groups of actors/communities of practice through social interactions," (Albert et al. 2012), and social learning is often linked with attempts to increase levels of participation in decision making, from consultation to more serious community control (Collins and Ison 2009; McCrum et al. 2009). Learning also facilitates responses to emerging problems and helps actors in SESs grapple with complexity. One outcome of learning can be adaptive risk management (ARM), in which "one takes action based on available information, monitors what happens, learns from the experience and adjusts future actions based on what has been learnt" (Bidwell et al. 2013). Suggestions to facilitate social learning, ARM, and decision-making include extending science-policy networks and using local bridging organisations, such as extension services, for knowledge co-production (Bidwell et al. 2013; Böcher and Krott 2014; Howarth and Monasterolo 2017) see further discussion in Chapter 7, section 7.5 on Decision-making for Climate and Land).

Insuring that women are included as key stakeholders in response option implementation is also important, as gender norms and roles affect vulnerability and access to resources, and gender inequality limits the possible range of responses for adoption by women (Lambrou and Piana 2006). For example, environmental change may increase women's workload as their access to natural resources may decline, or they may have to take up low-wage labour if agriculture becomes unsuitable in their local areas under climate change (Nelson et al. 2002). Every response option considered in this chapter potentially has a gender dimension to it that needs to be taken into consideration (Tables 6.73–6.75 note how response options intersect with SDG 5 Gender Equity); for example, to address food security through sustainable intensification will clearly have to address women farmers in Africa (Kondylis et al. 2016; Garcia and Wanner 2017) (For further information, see Cross-Chapter Box 11: Gender, in Chapter 7).

Challenges of coordination

Coordinated action to implement the response options will be required across a range of actors, including business, consumers, land managers, indigenous and local communities and policymakers to create enabling conditions. Conjoining response options to maximise social, climatic and environmental benefits will require framings of such actions as strong pathways to sustainable development (Ayers and Dodman 2010). As the chapter has pointed out, there are many potentials for synergies, especially among several response options that might be applied together and in coordination with one another (such as dietary change and improved land management measures). This coordination will help ensure that synergies are met and trade-offs minimised, but this will require deliberate coordination across multiple scales, actors and sectors. For example, there are a variety of response options available at different scales that could form portfolios of measures applied by different stakeholders from farm to international scales. Agricultural diversification and use of local seeds by smallholders can be particularly useful poverty reduction and biodiversity conservation measures, but are only successful when higher scales, such as national and international markets and supply-chains, also value these goods in trade regimes, and consumers see the benefits of purchasing these goods. However, the land and food sectors face particular challenges of institutional fragmentation, and often suffer from a lack of engagement between stakeholders at different scales (Biermann et al. 2009; Deininger et al 2014) (see section 7.6.2, Chapter 7).

Many of the response options listed in this chapter could be potentially implemented as 'community-based' actions, including community-based reforestation, community-based insurance, or community-based disaster risk management. Grounding response options in community approaches aims to identify, assist and implement activities "that strengthen the capacity of local people to adapt to living in a riskier and less predictable climate" (Ayers and Forsyth 2009). Research that shows that people willingly come together to provide mutual aid and protection against risk, to manage natural resources, and to work cooperatively to find solutions to environmental provisioning problems. Some activities that fall under this type of collective action can include the creation of institutions or rules; working cooperatively to manage a resource by restricting some activities and encouraging others; sharing information to improve public goods; or mobilising resources, such as capital, to fix a collective problem (Ostrom 2000; Poteete and Ostrom 2004); or engagement in participatory land use planning (Bourgoin 2012; Evers and Hofmeister 2011). These participatory processes "are likely to lead to more beneficial environmental outcomes through better informed, sustainable decisions, and win-win solutions regarding economic and conservation objectives" (Vente et al. 2016), and evaluations of community-based response options have been generally positive (Karim and Thiel 2017a; Tompkins & Adger 2004).

Agrawal (2001) has identified more than 30 different indicators that have been important in understanding who undertakes collective action for the environment, including the size of the group undertaking action; the type and distribution of the benefits from the action; the heterogeneity of the group; the dependence of the group on these benefits; the presence of leadership; presence of social capital and trust; and autonomy and independence to make and enforce rules. Alternatively, when households expect the government to undertake response actions, they have less incentive to join in collective action, as the state role has 'crowded out' local cooperation (Adger 2009). High levels of social trust and capital can increase willingness of farmers to engage in response options, such as improved soil management or carbon forestry (Stringer et al. 2012; Lee 2017), and social capital helps with connectivity across levels of SESs (Brondizio et al. 2009). (Dietz et al. 2013) lay out important policy directions for more successful facilitation of collective action across scales and stakeholders. These include: providing information; dealing with conflict; inducing rule compliance; providing physical, technical or institutional infrastructure; and being prepared for change. The adoption of participatory protocols and structured processes to select response options together with stakeholders will likely lead to greater success in coordination and participation (Bautista et al. 2017; Franks 2010; Schwilch et al. 2012a).

However, wider adoption of community-based approaches is potentially hampered by several factors: the fact that most are small scale (Forsyth 2013; Ensor et al. 2014) and it is often unclear how to assess criteria of success (Forsyth 2013). Others also caution that community-based approaches often are not able to adequately address the key drivers of vulnerability such as inequality and uneven power relations (Nagoda and Nightingale 2017).

Moving from response options to policies

Chapter 7 discusses in further depth the risks and challenges involved in formulating policy responses that meet the demands for sustainable land management and development outcomes, such as food security, community adaptation and poverty alleviation. Chapter 7 in Table 7.1 maps how specific response options might be turned into policies; for example, to implement a response option aimed at agricultural diversification, a range of policies from elimination of agricultural subsidies (which might favour single crops) to environmental farm programs and agro-environmental payments (to encourage alternative crops). Oftentimes, any particular response option might have a variety of potential policy pathways that might address different scales or stakeholders or take on different aspects of coordination and integration (section 7.6.1). Given the unique challenges of decision-making under uncertainty in future climate scenarios, Chapter 7 particularly discusses the need for flexible, iterative, and adaptive processes to turn response options into policy frameworks.

Cross-Chapter Box 9: Illustrative Climate and Land Pathways

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Future development of socioeconomic factors and policies influence the evolution of the land-climate system, among others in terms of the land used for agriculture and forestry. Climate mitigation policies can also have a major impact on land use, especially in scenarios consistent with the climate targets of the

Paris Agreement. This includes the use of bio-energy or Carbon Dioxide Removal (CDR), such as bioenergy with carbon dioxide capture and storage (BECCS) and afforestation. Land-based mitigation options have implications for GHG fluxes, desertification, land degradation, food insecurity, ecosystem services and other aspects of sustainable development.

Illustrative Futures

The three illustrative futures are based on the Shared Socioeconomic Pathways (SSPs; (O'Neill et al. 2014c; Riahi et al. 2017b; Popp et al. 2017; Rogelj et al. 2018b); Cross-Chapter Box 1 in Chapter 1). SSP1 is a scenario with a broad focus on sustainability including a focus on human development, technological development, nature conservation, globalised economy, economic convergence and early international cooperation including moderate levels of trade. The scenario assumes a low population growth, relatively high agricultural yields and a move towards less-meat intensive diets (van Vuuren et al. 2017b). Dietary change and reductions in food waste reduce agricultural demands and well-managed land systems enable reforestation and/or afforestation. SSP2 is a scenario in which societal as well as technological development follows historical patterns (Fricko et al. 2017). Land-based CDR is achieved through bioenergy and BECCS, and to a lesser degree by afforestation and reforestation. SSP3 is a scenario with limited technological progress and land-use regulation. Agricultural demands are high due to resource-intensive consumption and a regionalised world leads to reduced flows for agricultural goods. In SSP3, forest mitigation activities and abatement of agricultural GHG emissions are limited due to major implementation barriers such as low institutional capacities in developing countries and delayed as a consequence of low international cooperation (Fujimori et al. 2017a). Emissions reductions are achieved primarily through the energy sector, including the use of bioenergy and BECCS.

Policies in the Illustrative Futures

SSPs are complemented by a set of shared policy assumptions (Kriegler et al. 2014), indicating the types of policies that may be implemented in each future world. IAMs represent the effect of these policies on the economy, energy system, land use and climate with the caveat that they are assumed to be effective or in some cases the policy goals (e.g., dietary change) are imposed rather than explicitly modelled. In the real world, there are various barriers that can make policy implementation more difficult (see 7.4.9). These barriers will be generally higher in SSP3 than SSP1.

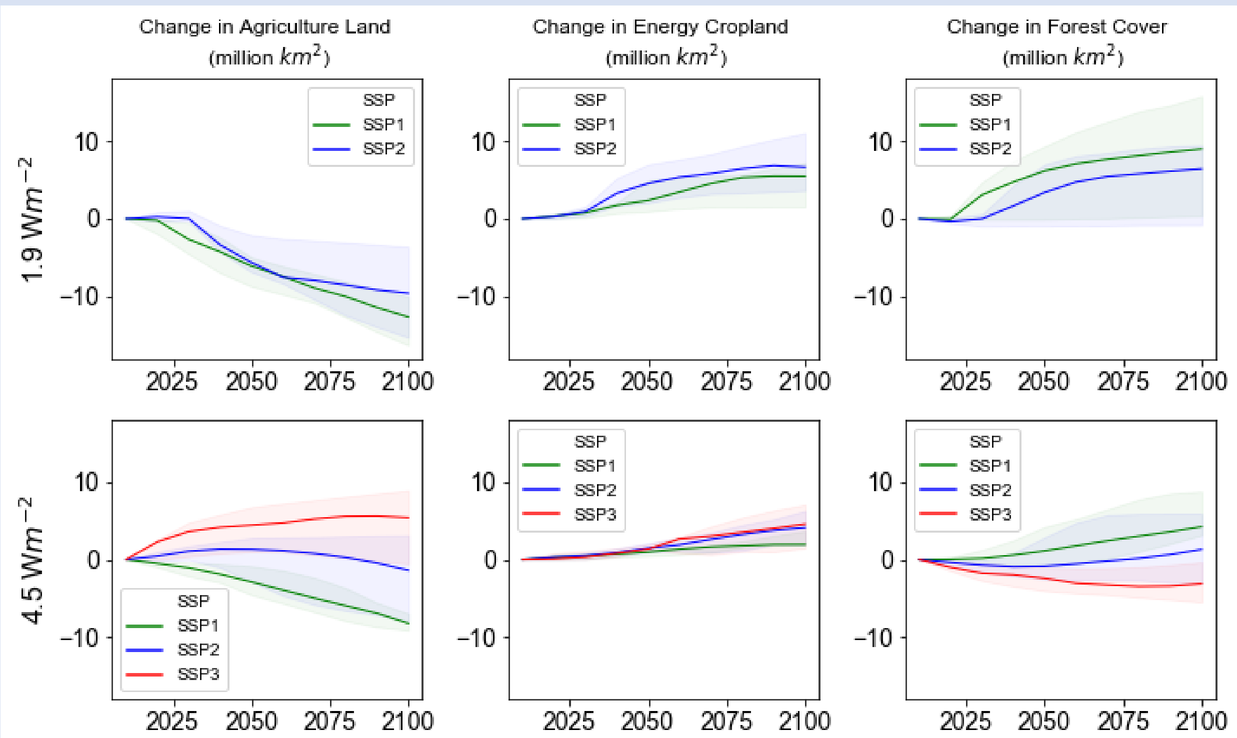
SSP1: A number of policies could support this SSP1 future including: effective carbon pricing, emission trading schemes (including net CO_2 emissions from agriculture), carbon taxes, regulations limiting GHG emissions and air pollution, forest conservation (mix of land-sharing and land sparing) through participation, incentives for ecosystem services and secure tenure, and protecting the environment, microfinance, crop and livelihood insurance, agriculture extension services, agricultural production subsidies, low export tax and import tariff rates on agricultural goods, dietary awareness campaigns, regulations to reduce and taxes on food waste, improved shelf life, sugar/fat taxes, and instruments supporting sustainable land management including payment for ecosystem services, land use zoning, REDD+, standards and certification for sustainable biomass production practices, legal reforms on land ownership and access, legal aid, legal education, including reframing these policies as entitlements for women and small agricultural producers (rather than sustainability) (O'Neill et al. 2017; van Vuuren et al. 2017b) (see 7.4).

SSP2: The same policies that support the SSP1 could support the SSP2 but may be less effective and only moderately successful. Policies may be challenged by adaptation limits (7.4.9), inconsistency in formal and informal institutions in decision making (7.5.1) or result in maladaptation (7.4.7). Moderately successful sustainable land management policies result in some land competition. Land degradation neutrality is moderately successful. Successful policies include those supporting bioenergy and BECCS (Rao et al. 2017; Riahi et al. 2017b; Fricko et al. 2017) (see 7.4.6).

SSP3: Policies that exist in SSP1 may or may not exist in SSP3, and are ineffective (O'Neill et al. 2014c). There are challenges to implementing these policies, as in SSP2. In addition, ineffective sustainable land management policies result in competition for land between agriculture and mitigation. Land degradation neutrality is not achieved (Riahi et al. 2017b). Successful policies include those supporting bioenergy and BECCS (see 7.4.6) (Kriegler et al. 2017; Fujimori et al. 2017a; Rao et al. 2017). Demand side food policies are absent and supply side policies predominate. There is no success in advancing land ownership and access policies for agricultural producer livelihood (7.6.5).

Land use and land cover change

Agricultural area in SSP1 declines as a result of the low population growth, agricultural intensification, low meat consumption, and low food waste. In contrast, SSP3 has high population and strongly declining rates of crop yield growth over time, resulting in increased agricultural land area. The SSP2 falls somewhere in between, with its modest growth in all factors. In the climate policy scenarios consistent with the Paris Agreement, bioenergy/BECCS and reforestation/afforestation play an important role in SSP1 and SSP2. The use of these options, and the impact on land, is larger in scenarios that limit radiative forcing in 2100 to 1.9 Wm^{-2} than in the 4.5 Wm^{-2} scenarios. In SSP3, the expansion of land for agricultural production implies that the use of land-related mitigation options is very limited, and the scenario is characterised by continued deforestation.



Cross-Chapter Box 9 Figure 1: Changes in agricultural land (left), energy cropland (middle) and forest cover (right) under three different SSPs (colours) and two different warming levels (rows). Agricultural land includes both pasture and non-energy cropland. Colours indicate SSPs, with SSP1 shown in green, SSP2 in blue, and SSP3 in red. Shaded area show the range across all IAMs; lines show the median across all models. Models are only included in a figure if they provided results for all SSPs in that panel. There is no SSP3 in the

top row, as 1.9 Wm^{-2} is infeasible in this world. Data is from an update of the IAMC Scenario Explorer developed for the SR15 (Huppmann et al. 2018; Rogelj et al. 2018a).

Implications for mitigation and other land challenges

The combination of baseline emissions development, technology options, and policy support makes it is much easier to reach the climate targets in the SSP1 scenario than in the SSP3 scenario. As a result, carbon prices are much higher in SSP3 than in SSP1. In fact, the 1.9 Wm^{-2} target was found to be infeasible in the SSP3 world (Cross-Chapter Box 9 Table 1). Energy system CO_2 emissions reductions are greater in the SSP3 than in the SSP1 to compensate for the higher land-based CO_2 emissions.

Accounting for mitigation and socioeconomics alone, food prices (an indicator of food insecurity) are higher in SSP3 than in the SSP1 and higher in the 1.9 Wm^{-2} than in the 4.5 Wm^{-2} (Cross-Chapter Box 9 Table 1). Forest cover is higher in the SSP1 than the SSP3 and higher in the 1.9 Wm^{-2} than in the 4.5 Wm^{-2}. Water withdrawals and water scarcity are in general higher in the SSP3 than the SSP1 (Hanasaki et al. 2013a; Graham et al. 2018b) and higher in scenarios with more bioenergy (Hejazi et al. 2014c); however, these indicators have not been quantified for the specific SSP-RCP combinations discussed here.

Climate change, results in higher impacts and risks in the 4.5 Wm^{-2} world than in the 1.9 Wm^{-2} world for a given SSP and these risks are exacerbated in SSP3 compared to SSP1 and SSP2 due to population's higher exposure and vulnerability. For example, the risk of fire is higher in warmer worlds; in the 4.5 Wm^{-2} world, the population living in fire prone regions is higher in the SSP3 (646 million) than in the SSP2 (560 million) (Knorr et al. 2016). Global exposure to multi-sector risk quadruples between the 1.5ºC[10] and 3ºC and is a factor of six higher in the SSP3-3ºC than in the SSP1-1.5ºC (Byers et al. 2018). Future risks resulting from desertification, land degradation and food insecurity are lower in the SSP1 compared to SSP3 at the same level of warming. For example, the transition moderate to high risk of food insecurity occurs between 1.3 and 1.7ºC for the SSP3, but not until 2.5 to 3.5ºC in the SSP1 (Section 7.2).

Table 1: Quantitative indicators for the illustrative pathways. Each cell shows the mean, minimum, and maximum value across IAM models for each indicator and each pathway in 2050 and 2100. All IAMs that provided results for a particular pathway are included here. Note that these indicators exclude the implications of climate change. Data is from an update of the IAMC Scenario Explorer developed for the SR15 (Huppmann et al. 2018; Rogelj et al. 2018b).

		SSP1		SSP2		SSP3	
		1.9 Wm^{-2} mean (min, max)	**4.5 Wm^{-2} mean (min, max)**	**1.9 Wm^{-2} mean (min, max)**	**4.5 Wm^{-2} mean (min, max)**	**1.9 Wm^{-2} mean (min, max)**	**4.5 Wm^{-2} mean (min, max)**
Population (billion)	2050	8.5 (8.5, 8.5)	8.5 (8.5, 8.5)	9.2 (9.2, 9.2)	9.2 (9.2, 9.2)	N/A	10.0 (10.0, 10.0)
	2100	6.9 (7.0, 6.9)	6.9 (7.0, 6.9)	9.0 (9.0, 9.0)	9.0 (9.1, 9.0)	N/A	12.7 (12.8, 12.6)
Change in GDP per capita (% rel to 2010)	2050	170.3 (380.1,	175.3 (386.2,	104.3 (223.4,	110.1 (233.8,	N/A	55.1 (116.1, 46.7)

[10] FOOTNOTE: Pathways that limit radiative forcing in 2100 to 1.9 Wm^{-2} result in median warming in 2100 to 1.5ºC in 2100 (Rogelj et al. 2018b). Pathways limiting radiative forcing in 2100 to 4.5 Wm^{-2} result in median warming in 2100 above 2.5ºC (IPCC 2014).

		130.9)	166.2)	98.7)	103.6)		
	2100	528.0 (1358.4, 408.2)	538.6 (1371.7, 504.7)	344.4 (827.4, 335.8)	356.6 (882.2, 323.3)	N/A	71.2 (159.7, 49.6)
Change in forest cover (Mkm^2)	2050	3.4 (9.4, -0.1)	0.6 (4.2, -0.7)	3.4 (7.0, -0.9)	-0.9 (2.9, -2.5)	N/A	-2.4 (-1.0, -4.0)
	2100	7.5 (15.8, 0.4)	3.9 (8.8, 0.2)	6.4 (9.5, -0.8)	-0.5 (5.9, -3.1)	N/A	-3.1 (-0.3, -5.5)
Change in cropland (Mkm^2)	2050	-1.2 (-0.3, -4.6)	0.1 (1.5, -3.2)	-1.2 (0.3, -2.0)	1.2 (2.7, -0.9)	N/A	2.3 (3.0, 1.2)
	2100	-5.2 (-1.8, -7.6)	-2.3 (-1.6, -6.4)	-2.9 (0.1, -4.0)	0.7 (3.1, -2.6)	N/A	3.4 (4.5, 1.9)
Change in energy cropland (Mkm^2)	2050	2.1 (5.0, 0.9)	0.8 (1.3, 0.5)	4.5 (7.0, 2.1)	1.5 (2.1, 0.1)	N/A	1.3 (2.0, 1.3)
	2100	4.3 (7.2, 1.5)	1.9 (3.7, 1.4)	6.6 (11.0, 3.6)	4.1 (6.3, 0.4)	N/A	4.6 (7.1, 1.5)
Change in pasture (Mkm^2)	2050	-4.1 (-2.5, -5.6)	-2.4 (-0.9, -3.3)	-4.8 (-0.4, -6.2)	-0.1 (1.6, -2.5)	N/A	2.1 (3.8, -0.1)
	2100	-6.5 (-4.8, -12.2)	-4.6 (-2.7, -7.3)	-7.6 (-1.3, -11.7)	-2.8 (1.9, -5.3)	N/A	2.0 (4.4, -2.5)
Change in other natural land (Mkm^2)	2050	0.5 (1.0, -4.9)	0.5 (1.7, -1.0)	-2.2 (0.6, -7.0)	-2.2 (0.7, -2.2)	N/A	-3.4 (-2.0, -4.4)
	2100	0.0 (7.1, -7.3)	1.8 (6.0, -1.7)	-2.3 (2.7, -9.6)	-3.4 (1.5, -4.7)	N/A	-6.2 (-5.4, -6.8)
Carbon price (2010 USD per tCO_2)[a]	2050	510.4 (4304.0, 150.9)	9.1 (35.2, 1.2)	756.4 (1079.9, 279.9)	37.5 (73.4, 13.6)	N/A	67.2 (75.1, 60.6)
	2100	2164.0 (35037.7, 262.7)	64.9 (286.7, 42.9)	4353.6 (10149.7, 2993.4)	172.3 (597.9, 112.1)	N/A	589.6 (727.2, 320.4)
Food price (Index 2010=1)	2050	1.2 (1.8, 0.8)	0.9 (1.1, 0.7)	1.6 (2.0, 1.4)	1.1 (1.2, 1.0)	N/A	1.2 (1.7, 1.1)
	2100	1.9 (7.0, 0.4)	0.8 (1.2, 0.4)	6.5 (13.1, 1.8)	1.1 (2.5, 0.9)	N/A	1.7 (3.4, 1.3)
Increase in Warming above pre-industrial (ºC)	2050	1.5 (1.7, 1.5)	1.9 (2.1, 1.8)	1.6 (1.7, 1.5)	2.0 (2.0, 1.9)	N/A	2.0 (2.1, 2.0)
	2100	1.3 (1.3, 1.3)	2.6 (2.7, 2.4)	1.3 (1.3, 1.3)	2.6 (2.7, 2.4)	N/A	2.6 (2.6, 2.6)
Change in per capita demand for food, crops (% rel to 2010)[b]	2050	6.0 (10.0, 4.5)	9.1 (12.4, 4.5)	4.6 (6.7, -0.9)	7.9 (8.0, 5.2)	N/A	2.4 (5.0, 2.3)
	2100	10.1 (19.9, 4.8)	15.1 (23.9, 4.8)	11.6 (19.2, -10.8)	11.7 (19.2, 4.1)	N/A	2.0 (3.4, -1.0)
Change in per capita demand for food, animal	2050	6.9 (45.0, -20.5)	17.9 (45.0, -20.1)	7.1 (36.0, 1.9)	10.3 (36.0, -4.2)	N/A	3.1 (5.9, 1.9)

products (% rel to 2010)[b,c]	2100	-3.0 (19.8, -27.3)	21.4 (44.1, -26.9)	17.0 (39.6, -24.1)	20.8 (39.6, -5.3)	N/A	-7.4 (-0.7, -7.9)
AFOLU CH_4 Emissions (% relative to 2010)	2050	-39.0 (-3.8, -68.9)	-2.9 (22.4, -23.9)	-11.7 (31.4, -59.4)	7.5 (43.0, -15.5)	N/A	15.0 (20.1, 3.1)
	2100	-60.5 (-41.7, -77.4)	-47.6 (-24.4, -54.1)	-40.3 (33.1, -58.4)	-13.0 (63.7, -45.0)	N/A	8.0 (37.6, -9.1)
AFOLU N_2O Emissions (% relative to 2010)	2050	-13.1 (-4.1, -26.3)	0.1 (34.6, -14.5)	8.8 (38.4, -14.5)	25.4 (37.4, 5.5)	N/A	34.0 (50.8, 29.3)
	2100	-42.0 (4.3, -49.4)	-25.6 (-3.4, -51.2)	-1.7 (46.8, -37.8)	19.5 (66.7, -21.4)	N/A	53.9 (65.8, 30.8)
Cumulative Energy CO_2 Emissions until 2100 ($GtCO_2$)		428.2 (1009.9, 307.6)	2787.6 (3213.3, 2594.0)	380.8 (552.8, -9.4)	2642.3 (2928.3, 2515.8)	N/A	2294.5 (2447.4, 2084.6)
Cumulative AFOLU CO_2 Emissions until 2100 ($GtCO_2$)		-127.3 (5.9, -683.0)	-54.9 (52.1, -545.2)	-126.8 (153.0, -400.7)	40.8 (277.0, -372.9)	N/A	188.8 (426.6, 77.9)

[a] The SSP2-19 is infeasible in two models. One of these models sets the maximum carbon price in the SSP1-19; the carbon price range is smaller for the SSP2-19 as this model is excluded there. Carbon prices are higher in the SSP2-19 than the SSP1-19 for every model that provided both simulations.

[a] Food demand estimates include waste.

[b] Animal product demand includes meat and dairy.

Summary

Future pathways for climate and land use include portfolios of response and policy options. Depending on the response options included, policy portfolios implemented, and other underlying socioeconomic drivers, these pathways result in different land-use consequences and their contribution to climate change mitigation. Agricultural area declines by more than 5 Mkm^2 in one SSP but increases by as much as 5 Mkm^2 in another. The amount of energy cropland ranges from nearly zero to 11 Mkm^2, depending on the SSP and the warming target. Forest area declines in the SSP3 but increases substantially in the SSP1. Subsequently, these pathways have different implications for risks related to desertification, land degradation, food insecurity, and terrestrial greenhouse gas fluxes, as well as ecosystem services, biodiversity, and other aspects of sustainable development.

6.4.5 Potential Consequences of Delayed Action

Delayed action, both in terms of overall GHG mitigation across both land and energy sectors, as well as delayed action in implementing the specific response options outlined in this chapter, will exacerbate the existing land challenges due to the continued impacts of climate change and socioeconomic and other pressures; can decrease the potential of response options and increase the costs of deployment; and will deprive communities of immediate co-benefits, among other pressures. The major consequences of delayed action are outlined below:

Delayed action exposes vulnerable people to continued and increasing climate impacts: Slower or delayed action in implementing response options exacerbates existing inequalities and impacts and will increase the number of people vulnerable to climate change, due to population increases and increasing climate impacts (SR15; AR 5). Future climate change will lead to exacerbation of the existing land challenges, increased pressure on agricultural livelihoods, potential for rapid land degradation, and millions more people exposed to food insecurity (Chapters 3, 4, 5; Schmidhuber & Tubiello 2007). Delay can also bring political risks and significant social impacts, including risks to human settlements (particularly in coastal areas), large-scale migration, and conflict (Barnett & Adger 2007; Hsaing et al. 2013). Early action reducing vulnerability and exposure can create an opportunity for a virtual circle of benefits: increased resilient livelihoods, reduced degradation of land, and improved food security (Bohle et al 1994).

Delayed action increases requirements for adaptation: Failure to mitigate climate change will increase requirements for adaptation. For example, it is likely that by 2100 with no mitigation or adaptation, 31–69 million people world-wide could be exposed to flooding (Rasmussen et al., 2018; SR15; Chapter 3); such outcomes could be prevented with investments in both mitigation and adaptation now. Some specific response options (e.g., reduced deforestation and degradation, reduced peatland and wetland conversion) prevent further detrimental effects to the land surface; delaying these options could lead to increased deforestation, conversion, or degradation, serving as increased sources of GHGs and having concomitant negative impacts on biodiversity and ecosystem services (section 6.2). Response options that aim at land restoration and rehabilitation can serve as adaptation mechanisms for communities facing climatic stresses like precipitation variability and changes in land quality, as well as providing benefits in terms of mitigation.

Delayed action increases response costs and reduces economic growth: Early action on reducing emissions through mitigation is estimated to result in both smaller temperature increases as well as lower mitigation costs than delayed action (Sanderson et al. 2016; Luderer et al. 2013; Fujimori et al., 2016; Rose et al., 2017; van Soest et al., 2017; Luderer et al., 2018). The cost of inaction to address mitigation, adaptation, and sustainable land use exceeds the cost of immediate action in most countries, depending on how damage functions and social cost of carbon are calculated (Dell et al. 2023; Moore & Diaz 2015). Costs of acting now would be one to two orders of magnitude lower than economic damages from delayed action, including both damage to assets from climate impacts, as well potentially reduced economic growth, particularly in developing countries (Moore and Diaz 2015; Luderer et al. 2013; 2016). Increased health costs and costs of energy (e.g. to run air-conditioners to combat increased heat waves) in the US by the end of the century alone are estimated to range from 10-58% of US GDP in 2010 (Deschênes and Greenstone 2011).

Delay also increases the costs of both mitigation and adaptation actions at later dates. In models of climate-economic interactions, deferral of emissions reductions now requires trade-offs leading to higher costs of several orders of magnitude and risks of higher temperatures in the longer term (Luderer et al 2013). Further, costs of action are likely to increase over time due to the increased severity of challenges in future scenarios.

Conversely, timely responses in implementing response options brings economic benefits. Carbon pricing is one component of economic responses to encourage adoption of response options (Jakob et al. 2016), but carbon pricing alone can induce higher risk in comparison to other scenarios and pathways that include additional targeted sustainability measures, such as promotion of less material- and energy-intensive lifestyles and healthier diets as noted in our response options (Bertram et al. 2018). While short term costs of deployment of actions may increase, better attainment of a broad set of sustainability targets can be achieved through these combined measures (Bertram et al. 2018).

There are also investments now that can lead to immediate savings in terms of avoided damages; for example, for each dollar spent on DMR, countries accrue avoided disaster-related economic losses of

USD4 or more (Mechler 2016). While they can require upfront investment, the economic benefits of actions to ensure sustainable land management, such as increased soil organic carbon, can more than double the economic value of rangelands and improve crop yields (Chapter 4; section 6.2).

Delayed action reduces future policy space and decreases efficacy of some response options: The potential for some response options decreases as climate change increases; for example, climate alters the sink capacity for soil and vegetation carbon sequestration, reducing the potential for increased soil organic carbon, afforestation, and reforestation (6.4.2). Additionally, climate change affects the productivity of bioenergy crops, influencing the potential mitigation of bioenergy and BECCS (Section 6.4.4).

For response options in the supply chain, demand side management, and risk management, while the consequences of delayed action are apparent in terms of continued GHG emissions from drivers, the tools for response options are not made more difficult by delay and could be deployed at any time. Additionally, given increasing pressures on land as a consequence of delay, some policy response options may become more cost effective while others become costlier. For example, over time, land-based mitigation measures like forest and ecosystem protection are likely to increase land scarcity leading to higher food prices, while demand side measures, like reduced impact diets and reducing waste, are less likely to raise food prices in economic models (Stevanović et al. 2017).

For risk management, some response options provide timely and rapidly-deployable solutions for preventing further problems, such as disaster risk management and risk sharing instruments. For example, early warning systems serve multiple roles in protecting lives and property and helping people adapt to longer-term climate changes and can be used immediately.

Delaying action can also result in problems of irreversibility of biophysical impacts and tipping points: Early action provides a potential way to avoid irreversibility - such as degradation of ecosystems that cannot be restored to their original baseline - and tipping points, whereby ecological or climate systems abruptly shift to a new state. Ecosystems, such as peatlands, are particularly vulnerable to irreversibility because of the difficulties of rewetting to original states (Section 6.2), and dryland grazing systems are vulnerable to tipping points when ground cover falls below 50%, after which productivity falls, infiltration declines, and erosion increases (Chapters 3 and 4). Further, tipping points can be especially challenging for human populations to adapt to, given lack of prior experience with such system shifts (Kates et al. 2012; Nuttall 2012).

Policy responses require lead time for implementation; delay makes this worse: For all the response options, particularly those that need to be deployed through policy implementation, there are unavoidable lags in this cycle. 'Policy lags', by which implementation is delayed by the slowness of the policy implementation cycle, are significant across many land-based, response options (Brown et al. 2019). Further, the behavioural change necessary to achieve some demand-side and risk management response options often takes a long time and delay only lengthens this process (Stern 1992; Steg & Vlek 2009). For example, actively promoting the need for healthier and more sustainable diets through individual dietary decisions is an important underpinning and enabling step for future changes, but is likely to be a slow-moving process, and delay in beginning will only exacerbate this.

Delay can lead to lock-in: Delay in implementation can cause 'lock-in' as decisions made today can constrain future development and pathways. For example, decisions made now on where to build infrastructure, make investments and deploy technologies, will have longer-term (decades-long) ramifications due to inertia of capital stocks (van Soest et al. 2017). In tandem, the vulnerability of the poor is likely to be exacerbated by climate change creating a vicious circle of "lock in" whereby an increasing share of the dwindling carbon budget may be needed to assist with improved energy use for the poorest (Lamb and Rao 2015).

Delay can increase the need for widespread deployment of land-based mitigation (afforestation, BECCS) (IPCC 2018; Streffler et al. 2018): Further delays in mitigation could result in an increased need for CDR options later; for example, delayed mitigation requires a 10% increase in cumulative CDR over the century (IPCC 2018).Similarly, strengthening near-term mitigation effort can reduce the CDR requirements in 2100 by a factor of 2-8 (Strefler et al., 2018). Conversely, scenarios with limited CDR require earlier emissions reductions (van Vuuren et al. 2017b) and may make more stringent mitigation scenarios, like the 1.5C, infeasible (Kriegler et al. 2018a,b).

Frequently Asked Questions

FAQ 6.1: What types of land-based options can help mitigate and adapt to climate change?

Land-based options that help mitigate climate change are various and differ greatly in their mitigation potential. The options with the moderate to large mitigation potential, and no adverse side-effects, include options that decrease pressure on land (e.g. by reducing the land needed for food production) and those that help to maintain or increase carbon stores both aboveground (e.g. forest measures, agroforestry, fire management) and belowground (e.g. increased soil organic matter or reduced losses, cropland and grazing land management, urban land management, reduced deforestation and forest degradation). These options also have co-benefits for adaptation by improving health, increasing yields, flood attenuation and reducing urban heat island effects. Another group of practices aim at reducing greenhouse emission sources, such as livestock management or nitrogen fertilisation management. Land-based options delivering climate change adaptation may be structural (e.g. irrigation and drainage systems, flood and landslide control), technological (e.g. new adapted crop varieties, changing planting zones and dates, using climate forecasts), or socio-economic and institutional (e.g. regulation of land use, associativity between farmers). Some adaptation options (e.g. new planting zones, irrigation) may have adverse-side effects for biodiversity and water. Adaptation options may be planned, such as those implemented at regional, national or municipal level (top-down approaches), or autonomous, such as many technological decisions taken by farmers and local inhabitants. In any case, their effectiveness depends greatly on the achievement of resilience against extreme events (e.g. floods, droughts, heat waves, etc.).

FAQ 6.2: Which land-based mitigation measures could affect desertification, land degradation or food security?

Some options for mitigating climate change are based on increasing carbon stores both above and below ground, so mitigation is usually related to increases in soil organic matter content and increased land cover by perennial vegetation. There is a direct relationship, with very few or no adverse side-effects for prevention or reversal of desertification and land degradation and the achievement of food security. This is so because both desertification and land degradation are closely associated with soil organic matter losses and the presence of bare ground surfaces. Food security depends on the achievement of healthy crops and high and stable yields over time, which is difficult to achieve in poor soils that are low in organic matter.

FAQ 6.3: What is the role of bioenergy in climate change mitigation and what are its challenges?

Plants absorb carbon as they grow. If plant-based material (biomass) is used for energy, the carbon it absorbed from the atmosphere is released back. Traditional use of bioenergy for cooking and heating is still widespread throughout the world. Modern conversion to electricity, heat, gas and liquid fuels can reduce the need to burn fossil fuels and this can reduce greenhouse gas emissions, helping to mitigate climate change. However, the total amount of emissions avoided depends on the type of biomass, where it is grown, how it is converted to energy, and what type of energy source it displaces. Some types of bioenergy require dedicated land (e.g., canola for biodiesel, perennial grasses, short rotation woody crops), while others can be co-produced or use agricultural or industrial residues (e.g., residues from sugar and starch crops for ethanol, manure for biogas). Depending on where, how, and the amount of bioenergy crops that are grown, the use of dedicated land for bioenergy could compete with food crops or other mitigation options. It could also result in land degradation, deforestation or biodiversity loss. In some circumstances, however, bioenergy can be beneficial for land, for example by increasing soil organic carbon. The use of co-products and residues for bioenergy limits the competition for land with food but could result in land degradation if carbon and nutrient-rich material is removed that would otherwise be left on the land. On the other hand, the by-products of some bioenergy conversion processes can be returned to the land as a fertiliser and may have other co-benefits (e.g. reducing pollution associated with manure slurry).

Supplementary Material for Chapter 6: Interlinkages between Desertification, Land Degradation, Food Security and GHG fluxes: synergies, trade-offs and Integrated Response Options

Supplementary Information for Section 6.4.1

Section 6.4.1 includes tables of feasibility dimensions for each of the 40 response options. This section includes the supporting material for those classifications.

Table SM6.1 Feasibility of land management response options in agriculture, considering cost, technological, institutional, socio-cultural and environmental and geophysical barriers and saturation and reversibility

Response option	Saturation	Reversibility	Cost	Technological	Institutional	Socio-cultural	Environmental and geophysical
Increased food productivity				Limited ability to define and measure indicators of sustainable intensification (Barnes and Thomson 2014b)	better access to credit, services, inputs and markets (Schut et al. 2016)	educational (e.g., educational needs of women; Pretty and Bharucha 2014, and cultural / behavioural (Martin et al. 2015)	since increasing food productivity can be limited by climatic and environmental factors (Olesen et al. 2002)
Improved cropland management			USD74 to USD226 ha^{-1}	e.g., need for further development of nitrification inhibitors (Singh and Verma 2007b)	can be institutional in some regions (e.g., poor sustainability frameworks, Madlener et al. 2006)	educational (e.g., lack of knowledge; Reichardt et al. 2009b)and cultural / behavioural (e.g., promotion of cover crops needs to account for farmers' needs; Roesch-McNally et al. 2017)	e.g., land access (Bryan et al. 2009b; Bustamante et al. 2014c)

Improved grazing land management			< USD1 kg of meat^{-1} (Rolfe et al., 2010)	e.g., need for further development of nitrification inhibitors (Singh and Verma 2007b)	can be institutional in some regions (e.g., need for extension services; Ndoro et al., 2014)	educational (e.g., poor knowledge of best animal husbandry practices among farmers; Ndoro et al., 2014), and cultural / behavioural (e.g. strong cultural importance of livestock and traditional practices in some communities (Herrero et al. 2016)	e.g., unless degraded, grazing lands are already closer to saturation than croplands; Smith et al. 2015
Improved livestock management			USD120 to USD621 ha^{-1} (Barnhart et al., 2000)	e.g., many dietary additives are still at low technology readiness level; Beauchemin et al., 2008	can be institutional in some regions (e.g., need for extension services; Ndoro et al., 2014),	educational (e.g., poor knowledge of best animal husbandry practices among farmers; Ndoro et al., 2014), and cultural / behavioural (e.g., strong cultural importance of livestock in some communities (Herrero et al. 2016)	e.g., climate suitability of different cattle breeds in a changing climate (Thornton et al. 2009b; Rojas-Downing et al. 2017b)
Agroforestry			< USD5 tCO_2e^{-1} (Torres et al. 2010) Note that lack of reliable financial support	There are likely to be relatively few technological barriers (Smith et al. 2007).	institutional in some regions (e.g., seed availability; (Lillesø et al. 2011)	educational (e.g., poor knowledge of how best to integrate trees into agro-ecosystems, (Meijer et al. 2015b); lack of	susceptibility to pests (Sileshi et al. 2008)

			(Hernandez-Morcillo et al. 2018) could be a barrier.			information, (Hernandez-Morcillo et al. 2018) and cultural / behavioural (e.g., farmers perceptions, Meijer et al. 2015b)	
Agricultural diversification			Minimal (Wimmer et al. 2016) Diversification results in cost-saving and risk reduction, thus expected cost is minimal. Note that not always economically viable (Barnes et al. 2015)	technological, biophysical, educational, and cultural barriers may emerge that limit the adoption of more diverse farming systems by farmers (Barnett and Palutikof 2015; Ahmed and Stepp 2016a); Roesch-McNally et al. 2016)		technological, biophysical, educational, and cultural barriers may emerge that limit the adoption of more diverse farming systems by farmers (Barnett and Palutikof 2015; Ahmed and Stepp 2016a); Roesch-McNally et al. 2016)	technological, biophysical, educational, and cultural barriers may emerge that limit the adoption of more diverse farming systems by farmers (Barnett and Palutikof 2015; Ahmed and Stepp 2016a); Roesch-McNally et al. 2016)
Reduced grassland conversion to cropland			Minimal (Garibaldi et al. 2017) With increased demand for livestock products, it is expected that livestock has higher returns than crops. Note that avoiding conversion is low cost, but there	Since the response option involves not cultivating a current grassland, there are likely to be few biophysical or technological barriers	There could be institutional barriers in some regions (e.g., poor governance to prevent conversion)	educational (e.g., poor knowledge of the impacts of ploughing grasslands, and cultural / behavioural (e.g., strong cultural importance of crop production in some communities	Since the response option involves not cultivating a current grassland, there are likely to be few biophysical or technological barriers

			may be significant opportunity costs associated with foregone production of crops.				
Integrated water management			Minimal (Lubell et al. 2011) Integrated water management expected to reduce production costs and increase economic efficiency				

Table SM6.2 Feasibility of land management response options in forests, considering cost, technological, institutional, socio-cultural and environmental and geophysical barriers and saturation and reversibility

Response option	Saturation	Reversibility	Cost	Technological	Institutional	Socio-cultural	Environmental and geophysical
Improved forest management			USD70 to USD160 ha^{-1} (Singer 2016)		e.g., better access to credit and markets, etc.	educational (e.g., limited knowledge of the most appropriate techniques)	Forest management affects the climate also through biophysical effects and the emissions of biogenic volatile organic compounds (BVOCs), which are both influenced by species composition.
Reduced deforestation and			USD500 to USD2600 ha^{-1}		e.g., land tenure, economic	educational (e.g., little information	e.g., susceptibility to climate and

degradation			Agricultural expansion is the major driver of deforestation in developing countries. Cost of reducing of deforestation is based on opportunity cost of not growing the most common crop in developing countries (Maize) for six years to reach tree maturity, with yield of 8 t ha^{-1} (high); 5 tons ha^{-1} (medium) & 1.5 t ha^{-1} & price of USD329 t^{-1}. Also, reduced deforestation practices have relatively moderate costs, but they requires transaction and administration costs (Overmars et al. ; Kindermann et al. 2008).		disincentives and transaction costs (Kindermann et al. 2008)	available in some regions) and cultural (different realities, e.g., small holder versus industrial production)	other unpredicted events (Ellison et al. 2017a)
Reforestation and forest restoration			USD10 to USD100 $tCO2e^{-1}$ (McLaren 2012b)			educational (e.g., low genetic diversity of planted forests)	e.g. availability of native species seedlings for planting

						and cultural (e.g., care of forest cultures)	
Afforestation			USD10 to USD100 $tCO2e^{-1}$ (McLaren 2012b)		e.g., policy makers commitment (Idris Medugu et al. 2010b)		

Table SM6.3 Feasibility of land management response options for soils, considering cost, technological, institutional, socio-cultural and environmental and geophysical barriers and saturation and reversibility

Response option	Saturation	Reversibility	Cost	Technological	Institutional	Socio-cultural	Environmental and geophysical
Increased soil organic carbon content			USD50 to USD170 ha^{-1} (FAO 2014) Based on smallholder farming - which accounts for 72% farms in the world; India farmers (medium farmers) and largescale farmers in the US (FAO 2014). The cost indicated is only for manure application and ignores other costs done under business as usual (BAU). Assumes application of 10 t ha^{-1} of organic	e.g., difficult to measure and verify; (Smith 2006)	Can be institutional in some regions (e.g., lack of institutional capacity; Bustamante et al. 2014c)	educational (e.g., poor knowledge of best practices among farmers; (Reichardt et al. 2009b) though cultural / behavioural barriers are likely to be small compared to other barriers (Smith et al. 2007; Wollenberg et al. 2016b)	e.g., soil type; (Baveye et al. 2018b)

			manure after every three years and minimum tillage.				
Reduced soil erosion			USD50 to USD240 ha^{-1} (Morokong et al. 2019) Based on prevention of soil erosion using terraces using rocks. Costs reported is only for avoided loss of carbon sequestration.	Limited technology choices and technical support (Haregeweyn et al. 2015)	For instance, in Ethiopia farmers have shown an increased understanding of the soil erosion problem, but soil conservation programs face a host of barriers related to limited access to capital, limited benefits, land tenure insecurity (Haregeweyn et al. 2015)	Poor community participation (Haregeweyn et al. 2015)	
Reduced soil salinisation			USD50 to USD250 ha^{-1} (ICARDA 2012) For NENA region, salinity control recommended practice is deep ploughing, done once every 4 to 5 years to breakdown the hardpan subsoil. Deep ploughing costs USD200 ha^{-1} for the four-year cycle or USD50 ha^{-1} for each cropping season.	e.g., lack of appropriate irrigation technology; (Machado and Serralheiro 2017b; CGIAR 2016; Bhattacharyya et al. 2015)	Lack of alternative irrigation infrastructure; (Evans and Sadler 2008; CGIAR 2016)	educational (poor knowledge of the causes and salinisation and how to address it; (Greene et al. 2016; Dagar et al. 2016b), and cultural / behavioural (persistence of traditional practices; (Greene et al. 2016; Dagar et al. 2016b)	e.g., lack of alternative water sources; (Bhattacharyya et al. 2015; Dagar et al. 2016b)

Reduced soil compaction			Negative cost (McLaren 2012b)	Both compaction process and remediation technologies are well-known (Antille et al. 2016b) but technological barriers exist (e.g., few decision support systems for implementation of precision management of traffic compaction)		educational (knowledge gaps; Antille et al. 2016b)	Some soils are prone to compaction (Antille et al. 2016b)
Biochar addition to soil			USD100 to USD800 $tCO2e^{-1}$ (McLaren 2012b) A small amount of biochar potential could be available at negative cost, and some at low cost, depending on markets for the biochar as a soil amendment (Shackley et al. 2011b; Meyer et al. 2011; Dickinson et al. 2014)	e.g., feedstock and pyrolysis temperature have large impacts on biochar properties	Can be institutional in some regions (e.g., lack of quality standards; Guo et al. 2016)	educational (e.g., low awareness among end users; Guo et al. 2016) and cultural / behavioural (Guo et al. 2016)	e.g., land available for biomass production (Woolf et al. 2010)

Table SM6.4 Feasibility of land management response options in any/other ecosystems, considering cost, technological, institutional, socio-cultural and environmental and geophysical barriers and saturation and reversibility

Response option	Saturation	Reversibility	Cost	Technological	Institutional	Socio-cultural	Environmental and geophysical
Fire management			USD0.2 to USD6.5 billion per country per year (USA, Australia, Canada)	Technologies for fire management exist, but the cost of its implementation is relatively moderate, since it requires constant maintenance (North et al. 2015a) and can be excessive for some local communities.	e.g., lack of social or political acceptance (Freeman et al. 2017b)	educational (e.g., poor knowledge of best practices, liability issues, casualty risks and little tolerance for management errors; North et al. 2015a)	e.g., susceptibility to climate and other unpredicted events (Hurteau et al. 2014) or steep or remote areas to its application (North et al. 2015a)
Reduced landslides and natural hazards				The implementation of practices for management of landslides and natural hazards is based on engineering works and more resilient cropping systems (Noble et al. 2014; Gill and Malamud 2017b), which are is often limited by their high costs, as well as biophysical, technological and educational barriers.	In the tropics, the most cited barriers for implementing landslide risk reduction measures are scientific and political in nature, and the ratio of implemented versus recommended landslide risk reduction measures is low for most landslide risk reduction components (Maes et al. 2017b).	The implementation of practices for management of landslides and natural hazards is based on engineering works and more resilient cropping systems (Noble et al. 2014; Gill and Malamud 2017b), which are often limited by their high costs, as well as biophysical, technological and educational barriers.	The implementation of practices for management of landslides and natural hazards is based on engineering works and more resilient cropping systems (Noble et al. 2014; Gill and Malamud 2017b), which are is often limited by their high costs, as well as biophysical, technological and educational barriers.
Reduced pollution including			USD2 to USD13 per household	e.g., lack of technology to	e.g., poor regulation and		Since air pollution is transboundary,

acidification			(Houtven et al. 2017)	inject fertilisers below ground to prevent ammonia emissions; Shah et al., 2018	enforcement of environmental regulations; Yamineva and Romppanen, 2017		sources are often far distant from the site of impact; Begum et al., 2011
Management of invasive species / encroachment			USD500 to USD6632 per ha (Jardine et al. 2017) High cost is for California invasive alien species control; low cost from control in Massachusetts	In the case of natural enemies can be technological (Dresner et al. 2015)	Where agricultural extension and advice services are poorly developed	Education can be a barrier, where populations are unaware of the damage caused by the invasive species. Cultural / behavioural barriers are likely to be small.	Restoration programmes can take a long time (Dresner et al. 2015)
Restoration and reduced conversion of coastal wetlands			Costs for coastal wetland restoration projects vary, but they can be cost-effective at scale (Erwin 2009)		Can be institutional in some regions (e.g., poor governance of wetland use in some regions; (Lotze et al. 2006)	educational (e.g., lack of knowledge of impact of wetland conversion), though technological and cultural / behavioural barriers are likely to be small compared to other barriers.	e.g., loss of large predators, herbivores, spawning and nursery habitat; (Lotze et al. 2006)
Restoration and reduced conversion of peatlands			USD4 to USD20 $tCO2e^{-1}$ (McLaren 2012b)		An be institutional in some regions (e.g., lack of inputs; Bonn et al. 2014)	educational (e.g., lack of skilled labour; Bonn et al. 2014), though technological and cultural / behavioural barriers are likely to be small	e.g., site inaccessibility; Bonn et al. 2014)

						compared to other barriers.	
Biodiversity conservation			USD10 to USD50 $tCO2e^{-1}$ (Minx et al. 2018)				

Table SM6.5 Feasibility of land management response options specifically for CDR, considering cost, technological, institutional, socio-cultural and environmental and geophysical barriers and saturation and reversibility

Response option	Saturation	Reversibility	Cost	Technological	Institutional	Socio-cultural	Environmental and geophysical
Enhanced weathering of minerals			USD10 to USD40 $tCO2e^{-1}$ (McLaren 2012b) The main cost (and large energy input) is in the mining and comminution of the minerals (Renforth et al. 2012) with higher total costs compared to other low cost land management options (Smith et al. 2016a).	High energy costs of comminution; Smith et al. 2016a	In some regions (e.g., lack of infrastructure for this new technology; Taylor et al. 2016c)	Educational (e.g., lack of knowledge of how to use these new materials in agriculture). Cultural barriers could occur in some regions, for example, due to minerals lying under undisturbed natural areas where mining might generate public acceptance issues (e.g., Renforth et al. 2012)	e.g., limited and inaccessible mineral formations (Renforth et al. 2012)
Bioenergy and BECCS		BECCS "is one of the NET options that is less vulnerable to reversal" (Fuss et al. 2018)	USD50 to USD250 $tCO2e^{-1}$ (McLaren 2012b)	While there are a few small BECCS demonstration facilities, BECCS has not been implemented at	Institutional barriers include governance issues (Gough 2016)	Cultural barriers include social acceptance (Sanchez and Kammen 2016b) with CCS facing	Competition for land and water

				scale (Kemper 2015b)		concerns of safety and environmental issues and bioenergy facing additional scrutiny because of competition for land and water.	

Table SM6.6 Feasibility of demand management response options, considering cost, technological, institutional, socio-cultural and environmental and geophysical barriers and saturation and reversibility

Response option	Saturation	Reversibility	Cost	Technological	Institutional	Socio-cultural	Environmental and geophysical
Dietary change				Inadequate storage options for e.g. fresh fruit and vegetables	Barriers might also be institutional in some regions (e.g., poorly developed dietary health advice, Wardle et al. 2000b)	cultural / behavioural (e.g., diets are deeply culturally embedded and behaviour change is extremely difficult to effect, even when health benefits are well known; Macdiarmid et al., 2016); educational (e.g., poor knowledge of what constitutes a healthy diet; Wardle et al. 2000b)	poor accessibility of healthy foods such and fruit and vegetables (e.g., Hearn et al. 1998b; Lock et al. 2005)
Reduced post-harvest losses				Lack of low-cost storage and preservation technologies	Barriers are largely institutional, since solutions may require	There are few biophysical, educational or cultural barriers, since preventing	There are few biophysical, educational or cultural barriers, since preventing

					dismantling and redesigning current food value chains	food loss is a priority in many developing countries.	food loss is a priority in many developing countries.
Reduced food waste (consumer or retailer)				Barriers in developing countries include reliability of transportation networks, market reliability, education, technology, capacity, and infrastructure (Kummu et al. 2012).	Specific barriers to reducing consumption waste in industrialised countries include inconvenience, lack of financial incentives, lack of public awareness, low cost of food, quality standards and regulations, consumer's ability to buy food products at any time, generalised oversupply in the distribution, and low prioritisation, among others (Kummu et al.); (Graham-Rowe et al. 2014); Diaz-Ruiz et al., 2018). Barriers in developing countries include reliability of transportation networks, market reliability, education, technology, capacity, and	Specific barriers to reducing consumption waste in industrialised countries include inconvenience, lack of financial incentives, lack of public awareness, and low prioritisation (Kummu et al.); (Graham-Rowe et al. 2014). Barriers in developing countries include reliability of transportation networks, market reliability, education, technology, capacity, and infrastructure (Kummu et al.)	

					infrastructure (Kummu et al.)		
Material substitution			Negligible (McLaren 2012b)	Improved treatments to prevent against fire and moisture needed (Ramage et al. 2017b)	Construction companies hesitant to take risks associated with wooden buildings and insurance companies rate wooden buildings as higher risk (Gustavson et al., 2006)	People perceive adverse effects of wood products on forests and increased risk of fire (Gustavson et al. 2006)	

Table SM6.7 Feasibility of supply management response options, considering cost, technological, institutional, socio-cultural and environmental and geophysical barriers and saturation and reversibility

Response option	Saturation	Reversibility	Cost	Technological	Institutional	Socio-cultural	Environmental and geophysical
Sustainable sourcing	Reversibility could be an issue and while there are low cost options, the implementations can be expensive.	Reversibility could be an issue and while there are low cost options, the implementations can be expensive.			There are institutional barriers in some contexts (e.g., in low income African, Asian and Latin American countries where challenges associated with food insecurity and climate change vulnerability are more acute) (Ingram et al.	No obvious biophysical or cultural barriers	No obvious biophysical or cultural barriers

					2016a)		
Management of supply chains					political will within trade regimes, economic laissez-faire policies that discourage interventions in markets, and the difficulties of coordination across economic sectors (Poulton et al. 2006; Cohen et al. 2009; Gilbert 2012b)		
Enhanced urban food systems				There are likely to be few biophysical, technological or cultural / behavioural barriers to implementing improved urban food systems, though institutional and education barriers could play a role.	There are likely to be few biophysical, technological or cultural / behavioural barriers to implementing improved urban food systems, though institutional and education barriers could play a role.	There are likely to be few biophysical, technological or cultural / behavioural barriers to implementing improved urban food systems, though institutional and education barriers could play a role.	There are likely to be few biophysical, technological or cultural / behavioural barriers to implementing improved urban food systems, though institutional and education barriers could play a role.
Improved food processing and retailing			The implementation of strategies to improve the efficiency and sustainability of retail and agri-food industries can be expensive	Adoption of specific sustainability instruments and eco-innovation practices	Successful implementation is dependent on organisational capacity, the agility and flexibility of business strategies, the	No obvious cultural/behavioural barriers, but educational barriers exist	No obvious biophysical and cultural/behavioural barriers

					strengthening of public-private policies and effectiveness of supply-chain governance.		
Improved energy use in food systems				e.g., low levels of farm mechanisation	e.g., energy efficiency in agriculture depends strongly on the technology level (Vlontzos et al. 2014)	educational (e.g., poor knowledge of alternative energy sources), and behavioural / cultural (e.g., high levels of repetitive labour, making farming unattractive to the youth, and disproportionally affecting women; (Baudron et al. 2015b)	

Table SM6.8 Feasibility of risk management response options, considering cost, technological, institutional, socio-cultural and environmental and geophysical barriers and saturation and reversibility

Response option	Saturation	Reversibility	Cost	Technological	Institutional	Socio-cultural	Environmental and geophysical
Management of urban sprawl			USD0.5 to USD3 trillion yr^{-1} globally (New Climate Economy 2018) Global cost of prevention of urban sprawl done		Barriers to policies against urban sprawl include institutional barriers to integrated land use planning and the costs to national		

			by: densification; provision of sustainable and affordable housing; and investment in shared, electric, and low-carbon transport.		governments of restricting or buying back development rights (Tan et al. 2009)		
Livelihood diversification			Barriers to diversification include the fact that poorer households and female headed households may lack assets to invest in new income streams or have a lack of education about new income sources (Berman et al. 2012b; Ahmed and Stepp 2016a; Ngigi et al. 2017)			Barriers to diversification include the fact that poorer households and female headed households may lack assets to invest in new income streams or have a lack of education about new income sources (Berman et al. 2012b; Ahmed and Stepp 2016a; Ngigi et al. 2017)	
Use of local seeds						Barriers to seed sovereignty include concerns about equitability in access to seed networks and the difficulty of sustaining such projects when development donors leave (Reisman 2017b),	

						and disputes over the intellectual property rights associated with seeds (Timmermann and Robaey 2016)	
Disaster risk management			Barriers to EWS include cost; an early warning system for the 80 most climate vulnerable countries in the world is estimated to cost USD 2 billion over five years to develop (Hallegatte 2012).		Institutional and governance barriers such as coordination and synchronisation among levels also effect some EWS (Birkmann et al. 2015b).		
Risk sharing instruments			USD10 to USD90 ha^{-1} (Schnitkey 2017) Insurance cost depends on value of crop. We use maize as an example in US (high) and Sub-Saharan Africa (low).				

1 # Supplementary Information for Section 6.4.3

2 Section 6.4.3 includes tables regarding interactions for each of the 40 response options with Nature's Contributions to People (NCP) and Sustainable
3 Development Goals (SDG). This section includes the supporting material for those classifications.

4 **Table SM6.9 Impacts on Nature's Contributions to People of integrated response options based on land management**

Integrated response options based on land management		**Habitat creation and maintenance**	**Pollination and dispersal of seeds and other propagules**	**Regulation of air quality**	**Regulation of climate**	**Regulation of ocean acidification**	**Regulation of freshwater quantity, flow and timing**	**Regulation of freshwater and coastal water quality**	**Formation, protection and decontamination of soils and sediments**	**Regulation of hazards and extreme events**	**Regulation of organisms detrimental to humans**	**Energy**	**Food and feed**	**Materials and assistance**	**Medicinal, biochemical and genetic resources**	**Learning and inspiration**	**Physical and psychological experiences**	**Supporting identities**	**Maintenance of options**
	Increased food productivity	Higher productivity spares land (e.g. Balmford et al. 2018) especially if intensification is done sustainably.	Likely may reduce native pollinators if reliant on increased chemical inputs (Potts et al. 2010) but not if through sustainable intensification.	N/A	N/A	Increased food productivity might be achieved through increased pesticide or fertiliser use, which causes runoff and dead zones in oceans (Beusen et al. 2016).	Food productivity increases could impact water quality if increases in chemicals used, but evidence is mixed on sustainable intensification (Rockström et al. 2009; Mueller et al. 2012).	Food productivity increases could impact water flow due to demand for irrigation (Rockström et al. 2009; Mueller et al. 2012).	Intensification through additional input of nitrogen fertiliser can result in negative impacts on climate, soil, water and air pollution (Tilman et al. 2002).	N/A	Increasing food production through agro-chemicals may increase pest resistance over time (Tilman et al. 2002).	N/A	Sustainable intensification has potential to close yield gaps (Tilman et al. 2011).	N/A	N/A	N/A	N/A	N/A	N/A
	Improved cropland management	Improved cropland management can contribute to diverse agroecosystems (Tscharntke et al. 2005) and promotes soil biodiversity (Oehl et al. 2017)	Better crop management can contribute to maintaining native pollinators (Gardiner et al. 2009).	N/A	See main text for mitigation potentials	Mitigation potential (see main text) will reduce ocean acidification.	Cropland conversion has major impacts on water quantity (Scanlon et al. 2007). Cropland management practices such as conservation tillage improve downstream water quality (Fawcett et al. 1994).	Cropland conversion leads to poorer water quality due to runoff (Scanlon et al. 2007).	Improved cropland management has positive impacts on soils (see main text) (Kern et al. 2003).	N/A	Some forms of improved cropland management can decrease pathogens and pests (Tscharntke et al. 2016).	N.A	Conservation agriculture contributes to food productivity and reduces food insecurity (Rosegrant and Cline 2003 ; Dar & Gowda 2011; Godfrey & Garnett 2014)	N/A	N/A	N/A	N/A	Many cropping systems have cultural components (Tenberg et al 2012).	N/A
Agriculture	**Improved grazing land management**	Can contribute to improved habitat (Pons et al. 2003; Plantureux et al 2005).	N/A	N/A	See main text for mitigation potentials	Mitigation potential (see main text) will reduce ocean acidification.	Likely will improve water quality (Hibbert 1983).	Likely will improve water flow (Hibbert 1983)..	Improved grassland management increases soil carbon and quality (Conant et al. 2001).	N/A	N/A	N/A	Improved grassland management could contribute to food security (O'Mara 2012)	Grassland management can provide other materials (e.g. biofuel materials)	N/A	N/A	N/A	Many pastoralists have close cultural connections to livestock (Ainslie 2013)	N/A

														(Prochnow et al. 2009)					
	Improved livestock management	Can contribute to improved habitat if more efficient animals used, leading to less feed required (Strassburg et al. 2014)	N/A	N/A	See main text for mitigation potentials	Mitigation potential (see main text) will reduce ocean acidification.	N/A	Improved industrial livestock production can reduce water contamination (e.g. reduced effluents) (Hooda et al 2000). Improved livestock management can contribute to better water quality such as through manure management (Herrero & Thornton 2013)	N/A	N/A	N/A	N/A	Improved livestock management can contribute to reduced food insecurity among smallholder pastoralists (van't Hooft et al. 2012).	Livestock production also produces materials for use (leather, etc) (Hesse 2006)	N/A	N/A	N/A	Many pastoralists have close cultural connections to livestock (Ainslie 2013)	N/A
	Agro-forestry	Agroforestry mimics natural diversity and can improve habitat (Jose 2009).	Even intensive agroforestry can be beneficial for pollinators (Klein et al 2002).	Trees in the landscape can remove air pollutants (Sutton et al., 2007)	See main text for mitigation potentials	Mitigation potential (see main text) will reduce ocean acidification.	Planting trees on farms can increase soil water infiltration capacity (Ilstedt et al. 2007). Agroforestry can be used to increase ecosystem services benefits, such as water quantity and quality (Jose 2009)	N/A	Likely to improve soil (Rao et al. 1997)	Agroforestry can reduce vulnerability to hazards like wind and drought (Thorlakson & Neufeldt 2012).	Landscape diversity generally improves opportunities for biological pest control (Gardiner et al. 2009); reduces pests/pathogens on smallholder farms (Vignola et al., 2015)	Agroforestry can be used to produce biomass for energy (Mbow et al., 2014).	Agroforestry contributes to food productivity and reduces food insecurity (Mbow et al. 2014).	Produces timber, firewood and animal fodder (Mbow et al., 2014)	Can provide medicinal and other resources (Rao et al., 2004).	N/A	N/A	Many cropping systems have cultural components (Rao et al., 2014)	Can contribute to maintaining diversity through native plantings (Rao et al., 2014).
	Agricultural diversification	Crop diversification improves resilience through enhanced diversity to mimic more natural systems and provide in-field habitat for natural pest defences (Lin 2011)	Diversification can enhance pollinator diversity (Altieri & Letrouneau 1982; Sardinas & Kremen 2015)	N/A	N/A	N.A	N/A	N/A	Diversification can introduce some crops that may have positive soil qualities (eg nitrogen fixation) and crop rotation with multiple crops can improve soil carbon (McDaniel et al. 2014).	N/A	Diverse agroecosystems tend to have less detrimental impacts from pests (Gardiner et al 2009; Altieri & Letourneau 1982)	N/A	Diversification is associated with increased access to income and additional food sources for the farming household (Pretty et al. 2003; Ebert 2014)	Diversification could provide additional materials and farm benefits (Van Huylenbroeck et al. 007)	Some agricultural diversification can produce medicinal plants (Chauhan 2010).	N/A	N/A	Many cropping systems have cultural components (Rao et al., 2014)	Can contribute to maintaining diversity through native plantings (Sardiñas et al. 2015)

	Avoidance of conversion of grassland to cropland	Can preserve natural habitat (Peeters, 2009)	N/A	N/A	See main text for mitigation potentials	Mitigation potential (see main text) will reduce ocean acidification.	Will likely improve water quality (inferred from improved soil quality in Saziozzi et al., 2001)	Will likely improve water flow (inferred from improved soil quality in Saziozzi et al., 2001)	Will improve soil quality (Saziozzi et al., 2001)	N/A	Diverse agroecosystems tend to have less detrimental impacts from pests (Gardiner et al 2009; Altieri & Letourneau 1982)	N/A	Reducing cropland conversion can reduce food production (West et al. 2010).	N/A	N/A	N/A	N/A	N/A	Retaining natural ecosystems can preserve genetic diversity (Ekins et al., 2003).
	Integrated water management	Ecosystem health and services can be enhanced by improving water management (Boelee E and E 2011). Securing ecosystem (Lloyd et al. 2013), integrated ecosystem-based management into water resources planning and management, linking ecosystem services and water security (Nicole Bernex 2016), improving correlation between amount of water resources and supply ecosystem services, combining water resources management and supply of ecosystem services (Liu et al. 2016).	Some integrated water management strategies generate synergies between multiple ecosystem services, such as pollination, yield and farm profitability (Hipólito et. al, 2018).	IWM practices exert strong influence on ecosystem structure and function, with potentially large implications for regulating air quality (Xia et al., 2017; Hardiman et al, 2019).	IWM supports favourable forests conditions thereby influencing the storage and flow of water in watersheds (Eisenbies et al. 2007) which are important for regulating microclimates (Pierzynski et al., 2017).	N/A	Improving regulations for water sharing, trading and pricing (ADB 2016), water smart appliance, water smart landscapes (Dawadi and Ahmad 2013), common and unconventional water sources in use (Rengasamy 2006) will increase water quantity.	Improving regulation to prevent aquifer and surface water depletion, controlling over water extraction, improvement of water management and management of landslides and natural hazards. Watering shifting sand dunes (sprinkler), water resources conservation (Nejad 2013; Pereira 2002a), enhancing rainwater management, reducing recharge and increasing water use in discharge areas (DERM 2011).	IWM provide co-benefits such as healthier soils, more resilient and productive ecosystems (Grey and Sadoff 2007; Liu et al. 2017; Scott et al. 2011)	Change in water availability through improving co-managing floods and groundwater depletion at the river basin such as Managed Aquifer Recharge (MAR), Underground Taming of Floods for Irrigation (UTFI), restore over-allocated or brackish aquifers, groundwater dependent ecosystems protection, reducing evaporation losses are significantly contributed to response climate change and reduced impacts of extreme weather event in desertification areas (Dillon and Arshad 2016b).		IWM can support the production of biomass for energy and firewood (Mbow et al., 2014).	Increasing demand for food, fiber and feed will put great strains on land, water, energy and other resources (WBCSD, 2014). Water conservation and balance in the use of natural resources enforcement (based water resources, water conservation measures, water allocations) (Ward et al. 2008) are good options to response climate change and nature's prevention.	IWM supports favourable forests conditions thereby providing wood and fodder and other materials (Locatelli et al. 2015a). However, conservation restrictions on the storage and flow of water in watersheds (Eisenbies et al. 2007) can restrict the access to resources (e.g. firewood).					

Forests	**Forest management and forest restoration**	Forest landscape restoration specifically aims to regain ecological integrity and enhance human well-being in deforested or degraded forest landscape (Maginnis and Jackson 2007; Stanturf et al. 2014). For example, facilitating tree species mixture means storing at least as much carbon as monocultures while enhancing biodiversity (Hulvey et al. 2013). Selective logging techniques are “middle way” between deforestation and total protection, allowing to retain substantial levels of biodiversity, carbon, and timber stocks (Putz et al. 2012),	Likely contributes to native pollinators (Kremen et al. 2007)	Trees remove air pollution by the interception of particulate matter on plant surfaces and the absorption of gaseous pollutants through the leaf stomata. Computer simulations with local environmental data reveal that trees and forests in the conterminous United States removed 17.4 million tonnes (t) of air pollution in 2010 (range: 9.0–23.2 million t), with human health effects valued at 6.8 billion U.S. dollars (range: USD1.5–13.0 billion) (Novak et al., 2014)	See main text for mitigation potentials	Mitigation potential (see main text) will reduce ocean acidification.	Forest cover can stabilise intense run-off during storms and flood events (Locatelli et al. 2015a) .Mangroves can protect coastal zones from extreme events (hurricanes) or sea level rise. However, forests also can have adverse side-effects for reduction of water yield and water availability for human consumption (Bryan and Crossman 2013).	Forests tend to maintain water quality by reducing runoff and trapping sediments and nutrients (Idris Medugu et al. 2010a; Salvati et al. 2014). Precipitation filtered through forested catchments delivers purified ground and surface water (co-benefits) (Calder 2005; Ellison et al. 2017; Neary et al. 2009).	Forests counteract wind-driven degradation of soils, and contribute to soil erosion protection and soil fertility enhancement for agricultural resilience (Locatelli et al. 2015a).	Forest cover can stabilise land against catastrophic movements associated with wave action and intense run-off during storms and flood events (Locatelli et al. 2015a). Reducing harvesting rates and prolonging rotation periods may induce an increased vulnerability of stands to external disturbances and catastrophic events (Yousefpour et al. 2018). Forest management strategies may decrease stand-level structural complexity and may make forest ecosystems more susceptive to natural disasters like wind throws, fires, and diseases (Seidl et al. 2014).	Forests can contribute to weed and pest control and landscape diversity generally improves opportunities for biological pest control (Gardiner et al. 2009)	SFM may increase availability of biomass for energy (Kraxner et al 2003; Sikkema et al 2014)	The proximity of forest to cropland constitutes a threat to livelihoods in terms of crop raiding by wild animals and in constraints in availability of land for farming (Few et al. 2017),. The competition for land between afforestation/reforestation and agricultural production is a potentially large adverse side-effect (Boysen et al. 2017a,b; Kreidenweis et al. 2016; Smith et al. 2013). An increase in global forest area can lead to increases in food prices through increasing land competition (Calvin et al. 2014; Kreidenweis et al. 2016; Reilly et al. 2012; Smith et al. 2013; Wise et al. 2009).	Forests provide wood and fodder and other materials (Locatelli et al. 2015a). However, conservation restrictions to preserve ecosystem integrity can restrict the access to resources (e.g. firewood).	Can provide medicinal and other resources.	Natural ecosystems often inspire learning (Turtle et al., 2015)	Forest landscape restoration specifically aims to enhance human well-being (Maginnis and Jackson 2007; Stanturf et al. 2014). Afforestation/reforestation and avoided deforestation benefit biodiversity and species richness, and generally improve the cultural and recreational value of ecosystems (co-benefits) (Knoke et al. 2014).	Many forest landscapes have cultural ecosystems services components (Plieninger et al. 2015)	Retaining natural ecosystems can preserve genetic diversity (Ekins et al., 2003).

	Reduced deforestation and degradation	Reduced deforestation can enhance connectivity between forest areas and conserve biodiversity hotspots (Ellison et al. 2017; Locatelli et al. 2011,a 2015a)	Likely contributes to native pollinators (Kremen et al. 2007)	Trees can improve air pollution problems (Novak et al., 2014)	See main text for mitigation potentials	Mitigation potential (see main text) will reduce ocean acidification.	Forests tend to maintain water quality by reducing runoff and trapping sediments and nutrients (Idris Medugu et al. 2010a; Salvati et al. 2014).	Due to evapotranspiration, trees recharge atmospheric moisture, contributing to rainfall locally and in distant location, and trees' microbial flora and biogenic volatile organic compounds can directly promote rainfall (Arneth et al. 2010). Trees enhance soil infiltration and, under suitable conditions, improve groundwater recharge (Calder 2005; Ellison et al. 2017; Neary et al. 2009).	Forests counteract wind-driven degradation of soils, and contribute to soil erosion protection and soil fertility enhancement for agricultural resilience (Locatelli et al. 2015a).	Forest cover can stabilise land against catastrophic movements associated with wave action and intense run-off during storms and flood events (Locatelli et al. 2015a)	Landscape diversity generally improves opportunities for biological pest control (Gardiner et al. 2009)	Reduced deforestation may increase availability of some wood for energy and industry	The proximity of forest to cropland constitutes a threat to livelihoods in terms of crop raiding by wild animals (Few et al. 2017),. The competition for land between afforestation/reforestation and agricultural production is a potentially large adverse side-effect (Boysen et al. 2017a,b; Kreidenweis et al. 2016; Smith et al. 2013) that can lead to increases in food prices (Calvin et al. 2014; Kreidenweis et al. 2016; Reilly et al. 2012; Smith et al. 2013; Wise et al. 2009).	Could increase availability of biomass (Griscom et al., 2017)	Reduced deforestation can protect forest medicinal plants (Arnold & Perez 2001)	Natural ecosystems often inspire learning (Turtle et al., 2015)	Forest ecosystems often support recreational opportunities (Liddle 1997)	Many forest landscapes have cultural ecosystems services components (Plieninger et al. 2015)	Retaining natural ecosystems can preserve genetic diversity (Ekins et al., 2003).
	Reforestation	Forest landscape restoration specifically aims to regain ecological integrity and enhance human well-being in deforested or degraded forest landscape (Maginnis and Jackson 2007; Stanturf et al. 2014). Adverse side-effects potentially associated to forests include establishment of non-native species, especially with the risks related to the spread of exotic fast growing tree species (Brundu and Richardson	Likely contributes to native pollinators if native forest species used (Kremen et al. 2007)	Trees can improve air pollution problems (Novak et al., 2014)	See main text for mitigation potentials	Mitigation potential (see main text) will reduce ocean acidification.	Forests tend to maintain water quality by reducing runoff and trapping sediments and nutrients (Idris Medugu et al. 2010a; Salvati et al. 2014).	Particular activities associated with forest landscape restoration, such as mixed planting, assisted natural regeneration, and reducing impact of disturbances (e.g. prescribed burning) have positive implications for fresh water supply (Ciccarese et al. 2012; Suding et al. 2015).	Forests contribute to soil erosion protection and soil fertility enhancement (Locatelli et al. 2015a).	Forest cover can stabilise land against catastrophic movements associated with wave action and intense run-off during storms and flood events (Locatelli et al. 2015a) Some forest ecosystems can be susceptive to natural disasters like wind throws, fires, and diseases (Seidl et al. 2014).	N/A	Reforestation can increase availability of biomass for energy (Swisher 1994).	The proximity of forest to cropland constitutes a threat to livelihoods in terms of crop raiding by wild animals and in constraints in availability of land for farming (Few et al. 2017),. The competition for land between afforestation/reforestation and agricultural production is a potentially large adverse side-effect (Boysen et al. 2017a,b; Kreidenweis et al. 2016;	Forests provide wood and fodder and other materials (Locatelli et al. 2015a). However, conservation restrictions to preserve ecosystem integrity can restrict the access to resources (e.g. firewoo	Source of medicines (UNEP, 2016)	Natural ecosystems often inspire learning (Turtle et al., 2015)	Afforestation/reforestation can increase areas available for recreation and tourism opportunities (Knoke et al. 2014).	Many forest landscapes have cultural ecosystems services components (Plieninger et al. 2015)	

		2016; Ellison et al. 2017).											Smith et al. 2013). An increase in global forest area can lead to increases in food prices through increasing land competition (Calvin et al. 2014; Kreidenweis et al. 2016; Reilly et al. 2012; Smith et al. 2013; Wise et al. 2009).	d).					
	Afforestation	Forest landscape restoration specifically aims to regain ecological integrity and enhance human well-being in deforested or degraded forest landscape (Maginnis and Jackson 2007; Stanturf et al. 2014). In the case of afforestation, simply changing the use of land to planted forests is not sufficient to increase abundance of indigenous species, as they depend on type of vegetation, scale of the land transition, and time required for a population to establish (Barry et al. 2014).	N/a	N/A	See main text for mitigation potentials	Mitigation potential (see main text) will reduce ocean acidification.	Depends on where reforesting and with what species (Scott et al. 2005). Trees enhance soil infiltration and, under suitable conditions, improve groundwater recharge (Calder 2005; Ellison et al. 2017; Neary et al. 2009).	Afforestation using some exotic species can upset the balance of evapotranspiration regimes, with negative impacts on water availability particularly in arid regions (Ellison et al. 2017; Locatelli et al. 2015a; Trabucco et al. 2008). Afforestation in arid and semiarid regions using species that have evapotranspiration rates exceeding the regional precipitation may aggravate the groundwater decline (Locatelli et al. 2015a; Lu et al. 2016). Changes in runoff affect water supply but can also contribute to changes in flood risks, and irrigation of forest plantations can increase water consumption (Sterling et al. 2013).	Afforestation and reforestation options are frequently used to counteract land degradation problems (Yirdaw et al. 2017). whereas when they are established on degraded lands they are instrumental to preserve natural forests (co-benefit) (Buongiorno and Zhu 2014). Afforestation runs the risk of decreasing soil nutrients, especially in intensively managed plantations; in one study, afforestation sites had lower soil P and N content (Berthrong et al 2009).	Some afforestation may make forest ecosystems more susceptive to natural disasters like wind throws, fires, and diseases (Seidl et al. 2014).	N/A	Afforestation may increase availability of biomass for energy use (Obersteiner et al 2006)	Future needs for food production are a constraint for large-scale afforestation plans (Locatelli et al. 2015a). Global food crop demand is expected by 50%–97% between 2005 and 2050 (Valin et al. 2014). Future carbon prices will facilitate deployment of afforestation projects at expenses of food availability (adverse side-effect), but more liberalised trade in agricultural commodities could buffer food price increases following afforestation in tropical regions (Kreidenweis et al. 2016).	Could increase availability of biomass (Griscom et al., 2017)	N/A	N/A	Green spaces support psychological wellbeing (Coldwell & Evans, 2018)	Afforestation/ reforestation can increase areas available for recreation and tourism opportunities (Knoke et al. 2014).	N/A
Soils	**Increased soil organic carbon content**	Improving soil carbon can increase overall resilience of	N/A	N/A	See main text for mitigation potentials	Rivers transport dissolved organic	Soil organic matter is known to increase water	Soil organic matter is known to increase water filtration and	Increasing SOM contributes to healthy soils (Lehmann &	N/A	Increased SOM decreases pathogens in soil (Lehmann	N/A	Lal 2006 notes that "Food-grain production in	In terms of raw material	In terms of raw materials, numerous	N/A	N/A	N/A	N/A

		landscapes (Tscharntke et al. 2005)				matter to oceans (Hedges et al 1997), but unclear if improved SOM will decrease this and by how much.	filtration and can regulate downstream flows (Keesstra et al., 2016)	protects water quality (Lehmann & Kleber 2015)	Kleber 2015)		& Kleber 2015)		developing countries can be increased by 24–39 (32+-11) million Mgy-1 through improving soil quality by increasing the SOC pool and reversing degradation processes".	s, numerous products (e.g. pharmaceuticals, clay for bricks and ceramics, silicon from sand used in electronics, and other minerals; SSSA, 2015) are provided by soils.	products (e.g. pharmaceuticals, clay for bricks and ceramics, silicon from sand used in electronics, and other minerals; SSSA, 2015) are provided by soils.				
	Reduced soil erosion	Managing soil erosion decreases need for expanded cropland into habitats (Pimental et al 1995)	N/A	Particulate matter pollution, a main consequence of wind erosion, imposes severe adverse impacts on materials, structures and climate which directly affect the sustainability of urban cities (Al-Thani et al. 2018)	N/A	N//A	Managing soil erosion improves water quality (Pimental et al 1995)	Managing soil erosion improves water flow (Pimental et al 1995)	Will improve soil quality (Keesstra et al., 2016)	Reducing soil erosion reduces vulnerability to hazards like wind storms in dryland areas and landslides in mountainous areas (El-Swify 1997)	N/A	N/A	Managing erosion can lead to increased food production on croplands; however, other forms of management (revegetation, zero tillage) might reduce land available for food.	N/A	N/A	N/A/	N/A	N/A	N/A
	Reduced soil salinisation	Salinisation decreases soil microbial diversity (Nie et al. 2009)	N/A	N/A	N/A	N/A	N/A	Management of soil salinity improves water quality (Kotb et al. 2000; Zalidis et al 2002; Soane & Ouwerkerk 1995)	Will improve soil quality (Keesstra et al., 2016)	N/A	N/A	N/A	Reversing degradation contributes to food productivity and reduces food insecurity (Pimiental et al. 1995; Shiferaw & Holden 1999).	N/A	N/A	N/A/	N/A	N/A	N/A

	Reduced soil compaction	Preventing compaction can reduce need to expand croplands (Lal, 2001).	N/A	N/A	N/A	N/A	Compaction can increase water runoff (Soane & Ouwerkerk 1995). Management of soil compaction improves water quality and quantity (Soane & van Ouwerkerk 1995; Zalidis et al 2002)	Management of soil compaction improves water quality and quantity (Soane & van Ouwerkerk 1995; Zalidis et al 2002)	Will improve soil quality (Keesstra et al., 2016)	Compaction in soils increases rates of runoff and can contribute to floods (Hümann et al 2011)	N/A	N/A	Compactions reduces agricultural productivity and thus contributes to food insecurity (Nawaz et al 2013)	N/A	N/A	N/A	N/A	N/A	N/A
	Biochar addition to soil	N/A	N/A	N/A	See main text for mitigation potentials	Mitigation potential (see main text) will reduce ocean acidification.	Biochar improves soil water filtration and retention (Spokas et al 2011; Beck et al. 2011)	Biochar improves soil water filtration and retention (Spokas et al 2011; Beck et al. 2011)	Can improve soil quality (Sohi, 2012)	N/A	N/A	N/A	Contributes to increased food production (Smith 2016; Jefferry et al., 2017)	N/A	N/A	N/A	N/A	N/A	N/A
	Fire management	Proactive fire management can improve natural habitat (Burrows 2008).	Reducing fire risk can improve habitat for pollinators (Brown et al. 2017)	Fire management improves air quality particularly in the periurban interface (Bowman et al. 2005)	See main text for mitigation potentials	Mitigation potential (see main text) will reduce ocean acidification.	Fires affect water quality and flow due to erosion exposure (Townsend & Douglas 2000).	Fires affect water quality and flow due to erosion exposure (Townsend & Douglas 2000).	Fire cause damage to soils, therefore fire management can improve them (Certini 2005)	Will reduce risk of wildfires as a hazard (McCaffrey 2002)	Landscape diversity generally improves opportunities for biological pest control (Gardiner et al. 2009)	Will increase availability of biomass, as fuel removal is a key management strategy (Becker et al. 2009)	N/A	N/A	N/A	N/A	Reduced wildlife risk will increase recreation opportunities in landscapes (Venn & Calkin 2011).	N/A	Retaining natural ecosystems can preserve genetic diversity (Ekins et al., 2003).
	Reduced landslides and natural hazards	Can preserve natural habitat (Dolidon et al. 2009)	N/A	N/A	N/A	N/A	Likely will improve water quality (Dolidon et al. 2009)	Likely will improve water flow (Dolidon et al. 2009)	Will improve soil quality (Keesstra et al., 2016)	Will reduce risk of disasters (Dolidon et al. 2009; Kausky 2010)	N/A	N/A	Landslides are one of the natural disasters that have impacts on food security (de Haen & Hemrich 2007)	N/A	N/A	N/A	N/A	N/A	N/A
Other ecosystems	**Reduced pollution including acidification**	Air pollution like acid rain has major impacts on habitats like lakes (Schindler et al 1989)	Pollution interferes with scents, which impact pollinators ability to detect resources (McFredrick et al 2008)	Will improve air quality with public health benefits (Nemet et al. 2010)	See main text for mitigation potentials	N/A	N/A	Pollution increases acidity of surface water, with likely ecological effects (Larssen et al 1999)	Soil acidification due to air pollution in a serious problem in many countries (Zhou et al. 2013)	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A

	Management of invasive species / encroachment	Improved management of IAS can lead to improved habitat and ecosystems (Richardson & van Wilgen 2004).	Invasive species can disrupt native plant-pollinator relations (Ghazoul 2006)	N/A	N.A	N/A	Many invasives can reduce water flow (Richardson & Van Wilgen 2004).	Invasive species can reduce water quality (Burnett et al. 2007; Chamier et al. 2012)	Likely to improve soil as invasive species generally have negative effects (Ehrenfeld & Scott 2001).	N/A	Many IAS are harmful pests (Charles & Dukes 2008).	N/A	IAS can compete with crops and reduce crop yields by billions of dollars annually (Pejchar & Mooney 2009)	Many invasives are important suppliers of materials (Pejchar & Mooney 2009).	N/A	N/A	N/A	N/A	Reducing invasives can increase biological diversity of native organisms (Simberloff 2005)
	Restoration and avoided conversion of coastal wetlands	Will preserve natural habitat (Griscom et al., 2017)	Will promote natural pollinators (Seddon et al., 2016)	N/A	See main text for mitigation potentials	Mitigation potential (see main text) will reduce ocean acidification.	The creation or restoration of wetlands, tidal marshes, or mangroves provide water retention and protect coastal cities from storm surge flooding and shoreline erosion during storms. Wetlands store freshwater and enhance water quality (Bobbink et al 2006)	Wetlands store freshwater and enhance water quality (Bobbink et al 2006)	Will improve soil quality (Griscom et al., 2017)	The creation or restoration of wetlands, tidal marshes, or mangroves provide water retention and protect coastal cities from storm surge flooding and shoreline erosion during storms (Haddad et al., 2015; Gittman et al. 2014; Kaplan et al. 2009).	Landscape diversity generally improves opportunities for biological pest control (Gardiner et al. 2009)	N/A	Mixed evidence: can affect agriculture/fisheries production when competition for land occurs, or could increase food production when ecosystems are restored (Crooks et al 2011)	Could increase availability of biomass (Griscom et al., 2017)	Wetlands can be sources of medicines (UNEP, 2016)	Natural ecosystems often inspire learning (Turtle et al., 2015)	Natural environments support psychological wellbeing (Coldwell & Evans, 2018)	Natural environments support psychological wellbeing (Coldwell & Evans, 2018)	Retaining natural ecosystems can preserve genetic diversity (Ekins et al., 2003).
	Restoration and avoided conversion of peatlands	Will preserve natural habitat (Griscom et al., 2017)	Could promote natural pollinators (Seddon et al., 2016)	N/A	See main text for mitigation potentials	Mitigation potential (see main text) will reduce ocean acidification.	Peatland restoration will improve water quality as they play important roles in water retention and drainage (Johnston 1991).	Peatland restoration will improve water quality as they play important roles in water retention and drainage (Johnston 1991).	Will improve soil quality (Griscom et al., 2017)	N/A	Landscape diversity generally improves opportunities for biological pest control (Gardiner et al. 2009)	Will reduce supply of any biomass or energy sourced from peatlands (Pin Koh 2007)	May reduce land available for smallholders in tropical peatlands (Jewitt et al 2014)	Will reduce supply of some materials sourced from peatlands (e.g palm oil, timber) (Murdiyarso et al. 2010)	Natural ecosystems are often source of medicines (UNEP, 2016)	Natural ecosystems often inspire learning (Turtle et al., 2015)	Natural environments support psychological wellbeing (Coldwell & Evans, 2018)	Natural environments support psychological wellbeing (Coldwell & Evans, 2018)	Retaining natural ecosystems can preserve genetic diversity (Ekins et al., 2003).

	Biodiversity conservation	Biodiversity conservation includes measures aiming to promote species richness and natural habitats, and to mantain them through protected areas (Cromsigt et al., 2018).	Reduced or absent populations of seed-dispersing animals result in poor to no dispersal,especially of large-seeded trees that depend on large animals such as elephants (Anzures-Dadda et al.2011; Brodie and Aslan2012; Beaune et al.2013; Brockerhoff et al. 2017). Animal pollination, which is fundamental to the reproduction and persistence of most flowering plants, is an important ecosystem service (Millennium Ecosystem Assessment 2005). As biodiversity contributes to various ecosystem processes, functions and services, the declining diversity and abundance of pollinators (mainly insects and birds) has raised concerns about the effects on both wild and crop plants (Potts et al. 2010).	Trees in the landscape ensured by protected areas can remove air pollutants (Sutton et al., 2007)	See main text for mitigation potentials		Many actions taken to increase biodiversity (eg protected areas) can also have incidental effects of improving water quantity (Egoh et al. 2009)	Many actions taken to increase biodiversity (eg protected areas) can also have incidental effects of improving water quality (Egoh et al. 2009)	Management of wild animals and protected habitats can influence soil conditions via changes in fire frequency (as grazers lower grass and vegetation densities as potential fuels) and nutrient cycling and transport (by adding nutrients to soils). Conserving and restoring megafauna in northern regions also prevents thawing of permafrost. Management of wild animals can influence land degradation processes by grazing, trampling and compacting soil surfaces, thereby altering surface temperatures and chemical reactions affecting sediment and carbon retention. (Cromsigt et al., 2018; Schmitz et al., 2018)	Management of wild animals can influence fire frequency as grazers lower grass and vegetation densities as potential fuels (Schmitz et al 2014).			Regulation of wild animals affects food for hunting and availability of potential feed for livestock (Cromsigt et al., 2018).		Source of medicines (UNEP, 2016)	Natural ecosystems often inspire learning (Turtle et al., 2015)	indigenous peoples commonly link forest landscapes and biodiversity to tribal identities, association with place, kinship ties, customs and protocols, stories, and songs (Gould et al. 2014; Lyver et al. 2017a, b).		Retaining natural ecosystems can preserve genetic diversity (Ekins et al., 2003).

Carbon dioxide removal	**Enhanced weathering of minerals**	N/A	N/A	N/A	See main text for mitigation potentials	Addition of basic minerals counteracts ocean acidification (Taylor et al., 2016)	N/A	May have negative effects on water quality (Atekwane et al. 2005)	Could improve soil quality (Rau & Caldiera 1999; Kantola et al 2017)	N/A	N/A	N/A	Can contribute to increase food production by replenishing plant available silicon, potassium and other plant nutrients (Beerling et al., 2018)	N/A	N/A	N/A	N/A	N/a	N/A
	Bioenergy and BECCS	Likely will reduce natural habitat with negative effects on biodiversity (Hof et al. 2018)	Would reduce natural pollinators due to decreased natural habitat if in competition (Keitt 2009).	The use of BECCS could reduce air pollution (SR15)	See main text for mitigation potentials	Mitigation potential (see main text) will reduce ocean acidification.	Will likely require water for plantations of fast growing trees and models show high risk of water scarcity if BECCS is deployed on widespread scale (Popp et al 2011; Smith et al. 2016; Hejazi et al., 2014) through both increases in water withdrawals (Hejazi et al., 2014; Bonsch et al., 2015) and changes in surface runoff (Cibin et al., 2015)	Bioenergy can affect freshwater quality via changes in nitrogen runoff from fertiliser application. However, the sign of the effect depends on what would have happened absent any bioenergy production, with some studies indicating improvements in water quality (Ng et al., 2010) and others showing declines (Sinha et al., 2019)	Will likely decrease soil quality if exotic fast growing trees used (Stoy et al. 2018)	N/A	N/A	BECCS and biofuels can contribute up to 300 EJ of primary energy by 2100 (Clarke et al., 2014).	BECCS will likely lead to significant trade-offs with food production (Smith et al 2016; Popp et al., 2017; Fujimori et al., in review)	N/A	N/A	N/A	BECCS would drive land use conversion and reduce opportunities for recreation/tourism.	BECCS would drive land use conversion and reduce culturally significant landscapes.	BECCS would drive land use conversion and reduce genetic diversity.

Table SM6.10 Impacts on Nature's Contributions to People of integrated response options based on value chain management

Integrated response options based on value chain management		**Habitat creation and maintenance**	**Pollination and dispersal of seeds and other propagules**	**Regulation of air quality**	**Regulation of climate**	**Regulation of ocean acidification**	**Regulation of freshwater quantity, flow and timing**	**Regulation of freshwater and coastal water quality**	**Formation, protection and decontamination of soils and sediments**	**Regulation of hazards and extreme events**	**Regulation of organisms detrimental to humans**	**Energy**	**Food and feed**	**Materials and assistance**	**Medicinal, biochemical and genetic resources**	**Learning and inspiration**	**Physical and psychological experiences**	**Supporting identities**	**Maintenance of options**
Demand management	**Dietary change**	Will lead to reduced expansion of ag lands, which can increase natural habitat (Tilman et al. 2001)	N/A	N/A	See main text on climate mitigation impacts	N/A	Will reduce water consumption if less water-intensive food/livestock needs to be produced (Tilman et al. 2001)	Reduced meat consumption will improve water quality (Stoll-Kleeman & O'Riordan 2015)	N/A	N/A	N/A	N/A	Will help increase global food supplies (Kastner et al. 2012)	N/A	N/A	N/A	N/A	N/A	N/A

	Reduced post-harvest losses	Will lead to reduced expansion of ag lands, which can increase natural habitat (Tilman et al. 2001)	N/A	N/A	See main text on climate mitigation impacts	N/A	Will reduce water consumption if less water-intensive food/livestock needs to be produced (Tilman et al. 2001)	N/A	N/A	N/A	Reducing postharvest losses will include measures to deal with pests, some of which could be biological (Wilson & Pusey 1985)	N/A	Will help increase global food supplies (Kastner et al. 2012)	N/A	N/A	N/A	N/A	N/A	N/A
	Reduced food waste (consumer or retailer)	Improved storage and distribution reduces food waste and the need for compensatory intensification of agricultural areas thereby creating co-benefits for reduced land degradation (Stathers et al. 2013).			See main text on climate mitigation impacts		Will reduce water consumption if less water-intensive food/livestock needs to be produced (Tilman et al. 2001)	Reduced food production will reduce N fertiliser use, improving water quality (Kibler et al. 2018)	N/A	N/A	N/A	N/A	Will help increase global food supplies (Kastner et al. 2012)	N/A	N/A	N/A	N/A	N/A	N/A
	Material substitution	Material substitution increases demand for wood, which can lead to loss of habitat (Sathre & Gustavsson 2006).			See main text on climate mitigation impacts	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	Material substitution supplies building materials to replace concrete and other nonrewewables (Gustavsson & Sathre 2011)	N/A	N/A	N/A	N/A	N/A
Supply management	**Sustainable sourcing**	Forest certification and other sustainable sourcing schemes can reduce habitat fragmentation as compared to conventional supply chains (Brown et al. 2001; Rueda et al. 2015))	N/A	Forest certification improved air quality in Indonesia by 5% due to reduced incidence of fire (Miteva et al. 2015)	N/A	N/A	Forest certification has led to improved water flow due to decreased road construction for logging (Miteva et al. 2015)	Forest certificaiton has improved riparian waterways and reduced chemical inputs in some schemes (Rueda et al 2015)	N/A	N/A	N/A	Sustainable sourcing can supply energy like biomass (Sikkema et al. 2014)	Sustainable sourcing can supply food and other goods (G. Smith 2007)	Sustainable sourcing is increasingly important in timber imports (Irland 2008)	Sustainable sourcing can supply medicinals (Pierce & Laird 2003).	N/A	N/A	N/A	N/A
	Management of supply chains	N/A	N/A	Better management of supply chains may reduce energy use and air pollution in transport (Zhu et al.	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	Improved supply chains will help increase global food supplies (Hamprecht 2005).	Improved supply chains will help increase material supplies due to efficiency gains (Burritt & Schaltegger 2014).	N/A	N/A	N/A	N/A	N/A

				2018)															
	Enhanced urban food systems	Urban gardening can improve habitat and biodiversity in cities (Orsini et al. 2014; Lin et al. 2015)	Urban beekeeping has been important in keeping pollinators alive (Gunnarsson & Federsel 2014)	Urban agriculture can increase vegetation cover and improve air quality in urban areas (Cameron et al. 2012; Lin et al. 2015).	See main text on climate mitigation impacts	N/A	Water access often a constraint on urban agriculture and can increase demands (De Bon et al 2010; Badami & Ramankutty 2015).	Urban agriculture can exacerbate urban water pollution problems (pesticide runoff, etc) (Pothukuchi & Kaufmann 1999)	N/A	N/A	N/A	N/A	Local urban food production is often more accessible to local populations and can increase food security (Eigenbrod & Gruda 2015)	N/A	N/A	Urban agriculture can be used for teaching and learning (Travaline & Hunold 2010).	N/A	Urban agriculture can promote cultural identities (Baker 2004)	Urban food can contribute to preserving local genetic diversity
	Improved food processing and retail	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
	Improved energy use in food systems	N/A	N/A	N/A	See main text on climate mitigation impacts	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A

Table SM6.11 Impacts on Nature's Contributions to People of integrated response options based on risk management

<u>Integrated response options based on risk management</u>	Habitat creation and maintenance	Pollination and dispersal of seeds and other propagules	Regulation of air quality	Regulation of climate	Regulation of ocean acidification	Regulation of freshwater quantity, flow and timing	Regulation of freshwater and coastal water quality	Formation, protection and decontamination of soils and sediments	Regulation of hazards and extreme events	Regulation of organisms detrimental to humans	Energy	Food and feed	Materials and assistance	Medicinal, biochemical and genetic resources	Learning and inspiration	Physical and psychological experiences	Supporting identities	Maintenance of options
Management of urban sprawl	Reducing urban sprawl can help preserve natural habitat in periurban areas (Pataki et al 2011)	Reducing urban sprawl will help reduce loss of natural pollinators from habitat conversion (Cane 2005)	Urban sprawl is a major contributor to air pollution (Frumkin 2002)	See main text on climate mitigation impacts		Managing urban sprawl can increase water availability (Pataki et al 2011)	Urban sprawl is associated with higher levels of water pollution due to loss of filtering vegetation and increasing impervious surfaces (Romero & Ordenes 2004; Tu et al 2007; Pataki et al 2011)	Likely to be beneficial for soils as soil sealing is major problem in urban areas (Scalenghe & Marsan 2009)	N/A	N/A		Urban sprawl often competes with land for food production and can reduce overall yields (Chen 2007, Barbero-Sierra et al., 2013)	N/A	N/A	N/A	N/A	N/A	N/A

	Livelihood diversification	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	Diversification is associated with increased access to income and additional food sources for the household (Pretty et al. 2003)	Diversification can increase access to materials (Smith et al. 2017)	N/A	N/A	N/A	N/A	N/A
	Use of local seeds	Use of commercial seeds can contribute to habitat loss (Upreti & Upreti 2002)	Use of open pollinated seeds is beneficial for pollinators and creates political will to conserve them (Helicke 2015)	N/A	N/A	N/A	Local seeds often have lower water demands, as well as less use of pesticides that can contaminate water (Adhikari 2014)	Likely to contribute to less pollution as local seeds are usually grown organically (Adhikari 2014)	Likely to contribute to better soils as local seeds are usually grown organically (Adhikari 2014)	N/A	Local seeds often need less pesticides thereby reducing pest resistance (Adhikari 2014)	N/A	Local seeds can lead to more diverse and healthy food in areas with strong food sovereignty networks (Coomes et al. 2015; Bisht et al. 2018). However local seeds often are less productive than improved varieties.		Many local seeds can have multiple functions, including medicinals (Hammer & Teklu 2008)	Passing on seed information is important cultural learning process (Coomes et al. 2015)		Seeds associated with specific cultural identities for many (Coomes et al. 2015)	Food sovereignty movements have promoted saving of genetic diversity of crops through on-farm maintenance (Isakson 2009)
	<u>Disaster risk management</u>	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	DRM helps people avoid extreme events and adapt to climate change (Mechler et al. 2014)	N/A	N/A	Famine early warning systems have been successful in Sahelian Africa to alert authorities to impending food shortages so that food acquisition and transportation from outside the region can begin, potentially helping millions of people (Genesio et al. 2011; Hillbruner and Moloney 2012)		N/A	N/A	N/A	N/A	N/A

	Risk sharing instruments	Commercial crop insurance often encourages habitat conversion; Wright and Wimberly (2013) found a 531,000 ha decline in grasslands in the Upper Midwest of the US 2006-2010 due to crop conversion driven by higher prices and access to insurance.	Crop insurance is likely to impact natural pollinators due to incentives for production (Horowitz & Lichtenberg 1993)	N/A	N/A	N/A	N/A	Likely to have negative effect as crop insurance encourages more pesticide use (Horowitz & Lichtenberg 1993).	One study found a 1% increase in farm receipts generated from subsidised farm programs (including crop insurance and others) increased soil erosion by 0.135 tons per acre (Goodwin and Smith 2003).	N/A	Crop insurance increasess nitrogen use and leads to treating more acreage with both herbicides and insecticides (Horowitz & Lichtenberg 1993)	N/A	Crop insurance has generally lead to (modest) expansions in cultivated land area and increased food production (Claassen et al. 2011; Goodwin et al. 2004)		Insurance encourages monocropping leading to loss of genetic diversity for future (Glauber 2004)	N/A	N/A	N/A	Insurance encourages monocropping leading to loss of genetic diversity for future (Glauber 2004)

Table SM6.12 Impacts on the UN SDG of integrated response options based on land management

Integrated response options based on land management		**GOAL 1: No Poverty**	**GOAL 2: Zero Hunger**	**GOAL 3: Good Health and Well-being**	**GOAL 4: Quality Education**	**GOAL 5: Gender Equality**	**GOAL 6: Clean Water and Sanitation**	**GOAL 7: Affordable and Clean Energy**	**GOAL 8: Decent Work and Economic Growth**	**GOAL 9: Industry, Innovation and Infrastructure**	**GOAL 10: Reduced Inequality**	**GOAL 11: Sustainable Cities and Communities**	**GOAL 12: Responsible Consumption and Production**	**GOAL 13: Climate Action**	**GOAL 14: Life Below Water**	**GOAL 15: Life on Land**	**GOAL 16: Peace and Justice Strong Institutions**	**GOAL 17: Partnerships to achieve the Goal**
Agriculture	**Increased food productivity**	Increasing farm yields for smallholders contributes to poverty reduction (Irz et al 2001; Pretty et al 2003)	Increasing farm yields for smallholders reduces food insecurity (Irz et al 2001; Pretty et al 2003).	Increased food productivity leads to better health status (Rosegrant & Cline 2003; Dar & Gowda 2011)	N/A	Increased productivity can benefit female farmers, who make up 50% of agricultural labor in sub-Saharan Africa (Ross et al 2015)	Food productivity increases could impact water quality if increases in chemicals used, but evidence is mixed on sustainable intensification (Rockstro ̈m et al 2009; Mueller et al 2012).	N/A	Increased agricultural production generally (Lal 2006) contributes to increased economic growth.	N/A	Increased agricultural production can contribute to reducing inequality among smallholders (Datt & Ravallion 1998).	Increased food production can increase urban food security (Ellis & Sumberg 1998).	N/A	See main text on climate mitigation and adaptation	Increased food productivity might be achieved through increased pesticide or fertiliser use, which causes runoff and dead zones in oceans (Beusen et al 2016)	See main text on desertification and degradation	N/A	Improved agricultural productivity generally correlates with increases in trade in agricultural goods (Fader et al. 2013)

	Improved cropland management	Improved cropland management increases yields for smallholders and contributes to poverty reduction (Irz et al 2001; Pretty et al 2003; Schneider & Gugerty 2011).	Conservation agriculture contributes to food productivity and reduces food insecurity (Rosegrant & Cline 2003; Dar & Gowda 2011; Godfray & Garnett 2014).Land consolidation has played an active role in China to in increase cultivated land area,promoting agricultural production scale, improving rural production conditions and living environment, alle-viating ecological risk and supporting for rural development (Zhou et al. 2019).	Conservation agriculture contributes to improved health through several pathways, including reduced fertiliser/pesticide use which cause health impacts (Erisman et al 2011) as well as improved food security.	N/A	N/A	Cropland management practices such as conservation tillage improve downstream and groundwater water quality (Fawcett et al 1994, Foster 2018). Good management practices can substantially decrease P losses from existing land use, to achieve ‘good’ water quality in catchment in New Zealand, United Kingdom and United States (	N/A	Increased agricultural production generally (Lal 2006) contributes to increased economic growth, mainly in smallholder agriculture (Abrahan and Pingali 2017).	N/A	Increased agricultural production can contribute to reducing inequality among smallholders (Datt & Ravallion 1998, Abrahan and Pingali 2017)).	N/A	Improved conservation agriculture contributes to sustainable production goals (Hobbs et al. 2008).	See main text on climate mitigation and adaptation	N/A	See main text on desertification and degradation	N/A	Improved agricultural productivity generally correlates with increases in trade in agricultural goods (Fader et al. 2013)
	Improved grazing land management	Increases yields for smallholders and contributes to poverty reduction (Boval & Dixon 2012)	Improved grassland management could contribute to food security (O'Mara 2012)	Improved livestock and grazing management could contribute to better health among smallholder pastoralists (van't Hooft et al. 2012) but pathways are not entirely clear.	N/A	N/A	Grassland management practices can improve downstream and groundwater water quality (Foster 2018).	N/A	Improved land management for livestock can increase economic productivity, especially in global South (Pender et al 2006)	N/A	Improved pastoral management strategies can contribute to reducing inequality but are context specific (Lesorogol 2003)	N/A	Improved grassland management contributes to sustainable production goals (O'Mara 2012).	See main text on climate mitigation and adaptation	N/A	See main text on desertification and degradation	Grazing land management requires collective action and therefore can increase social capital and build institutions (Mearns 1996)	N/A
	Improved livestock management	Improved livestock management (e.g. better breeding) can contribute to poverty reduction for smallholder pastoralists (van't Hooft	Improved livestock management can contribute to reduced food insecurity among smallholder pastoralists (van't Hooft et	N/A	N/A	N/A	Improved industrial livestock production can reduce water contamination (e.g. reduced effluents) (Hooda et al 2000).	N/A	Improved livestock management can increase economic productivity and employment opportunities in	N/A	N/A	N/A	Sustainable livestock management contributes to sustainable production goals	See main text on climate mitigation and adaptation	N/A	See main text on desertification and degradation	N/A	Improved livestock productivity would likely correlate with increases in trade (Herrero et al. 2009)

		et al. 2012)	al. 2012).				Improved livestock management can contribute to better water quality such as through manure management (Herrero & Thornton 2013)		global South (Mack 1990)				(de Wit et al 1995).					
	Agro-forestry	Agroforestry can be usefully used for poverty reduction (Leakey& Simons 1997).	Agroforestry contributes to food productivity and reduces food insecurity (Mbow et al. 2014).	Agroforestry positively contributes to food productivity and nutritious diets (Haddad 2000)	N/A	Increased use of agroforestry can benefit female farmers as it requires low overhead, but land tenure issues must be paid attention to (Kiptot & Franzel 2012).	Agroforestry can be used to increase ecosystem services benefits, such as water quantity and quality (Jose 2009)	Agroforestry could increase biomass for energy (Mbow et al. 2014)	Agroforestry and other forms of employment in forest management make major contributions to global GDP (Pimental et al 1997).	N/A	Agroforestry promotion can contribute to reducing inequality among smallholders (Leßmeister et al 2018).	N/A	Agroforestry contributes to sustainable production goals (Mbow et al 2014).	See main text on climate mitigation and adaptation	N/A	See main text on desertification and degradation	N/A	N/A
	Agricultural diversification	Agricultural diversification is associated with increased welfare and incomes and decreased levels of poverty in several country studies (Arslan et al. 2018; Asfaw et al. 2018; Weinberger & Lumpkin 2007).	Diversification is associated with increased access to income and additional food sources for the farming household (Pretty et al. 2003; Ebert 2014).Diversification can also reduce the risk of crop pathogens spreading across landscapes (Lin 2011).	More diversified agriculture leads to diversified diets which have better health outcomes (Block & Webb 2001; Ebert 2014; Kadiyala et al 2014) particularly for women and children (Pretty et al. 2003)	N/A	N/A	N/A	N/A	Agricultural diversification can lead to economic growth (Rahman 2009; Pingali & Rosegrant 1995). It allows farmers to choose a strategy that both increases resilience and provides economic benefits, including functional biodiversity at multiple spatial and/or temporal scales, through practices developed via	N/A	Increased agricultural diversification can contribute to reducing inequality among smallholders (Makate et al 2016), although there is mixed evidence of inequality also increasing in commercialised systems (Pingali & Rosegrant 1995; Weinberger & Lumpkin 2007)	N/A	N/A		N/A	See main text on desertification and degradation	N/A	N/A

									traditional and/or agroecological scientific knowledge (Lin 2011 ; Kremen et al. 2012).									
	Avoidance of conversion of grassland to cropland	May reduce land available for cropping or livestock for poorer farmers ; some grassland restoration programs in China have been detrimental to poor pastoralists (Foggin 2008)	Can affect food security when competition for land occurs (O'Mara 2012)	N/A	N/A	N/A	Retaining grasslands contributes to better water retention and improved quality (Scanlon et al 2007).	N/A	Reduced cropland expansion may decrease GDP (Lewandrowski et al 1999)	N/A	N/A	N/A	N/A	See main text on climate mitigation and adaptation	N/A	See main text on desertification and degradation	N/A	N/A
	Integrated water management	Green water harvesting contributes to alleviate poverty in Sub-Saharan Africa (Rockström and Falkenmark 2015), Improving water irrigation (Rengasamy 2006), improving rainfed agriculture (integrating soil and water management, rainfall infiltration and water harvesting, provides a large co-benefit to delivery of food security and poverty reduction (UNCTAD 2011)	Integrated, efficient, equitable and sustainable water resource management (as water for agroecosystem) plays importance for food production and benefits to people (Lloyd et al. 2013).	Water is a finite and irreplaceable resource that is fundamental to human well-being. It is only renewable if well managed. Integrated water management is vital option for reducing the global burden of disease and improving the health, welfare and productivity of populations. Today, more than 1.7 billion people live in river basins where depletion through use exceeds natural recharge, a trend that	N/A	Involving both women and men in integrated water resources initiatives can increase project effectiveness and efficiency (Green & Baden 1995)	Water resource management is intended to solve watershed problems on a sustainable basis, and these problems can be categorised into lack of water (quantity), deterioration in water quality, ecological effects, poor public participation, and low output economic value for investment in watershed-related activities (Lee et al. 2018). Integrated water management, increase	N/A	Water is at the core of sustainable development and is critical for socio-economic development, healthy ecosystems and for human survival itself. Integrated water management can play a key enabling role in strengthening the resilience of social, economic and environmental systems in the light of rapid and unpredictable changes	N/A	IWM can increase access of industry to water for economic growth (Rahman & Varis 2005)	Water is a limiting factor in urban growth and IWM can help improve access to urban water supplies (Bao &Fang 2012)	Poor sectoral coordination and institutional fragmentation have triggered an unsustainable use of resources and threatened the long-term sustainability of food, water, and energy security (Rassul 2016).	See main text on climate mitigation and adaptation	IWM on land is likely to improve water quality runoff into oceans (Agboola & Braimoh 2009)	See main text on desertification and degradation	Integrated water management, increase water-use efficiency across all sectors and ensure sustainable withdrawals and supply of freshwater to address water scarcity, and substantially reduce the number of people suffering from water scarcity (UN Water, 2015).	

				will see two-thirds of the world's population living in water-stressed countries by 2025 (UNWater 2015)			water-use efficiency across all sectors and ensure sustainable withdrawals and supply of freshwater to address water scarcity, and substantially reduce the number of people suffering from water scarcity (UNWater 2015).		(UN Water, 2015).									
Forestry	**Forest management and forest restoration**	May contribute to poverty reduction if conditions are right (Blomley & Ramadhani 2006; Donovan et al 2006), but conflicting data, as it may also favor large landowners who are less poor (Rametsteiner and Simula 2003).	Forest expansion can affect crop production when competition for land occurs (Angelsen 2010). An increase in global forest area can lead to increases in food prices through increasing land competition (Calvin et al. 2014b; Kreidenweis et al. 2016c; Reilly et al. 2012b; Smith et al. 2013a; Wise et al. 2009b)	N/A	N/A	Women face challenges in sustainable forest management (Mwangi et al 2011), but N/A how SFM affects gender equity.	Forests tend to maintain water quality by reducing runoff and trapping sediments and nutrients (Idris Medugu et al. 2010c; Salvati et al. 2014a). Due to evapotranspiration, trees recharge atmospheric moisture, contributing to rainfall locally and in distant location, and trees' microbial flora and biogenic volatile organic compounds can directly promote rainfall (Arneth et al. 2010). Trees enhance soil infiltration and, under	SFM may increase availability of biomass for energy (Kraxner et al. 2013; Sikkema et al. 2013)	Forest management often require employment for active replanting, etc. (Ros-Tonen et al 2008)	Forestry supplies wood for industrial use (Gustavsson & Sathre 2011)	N/A	Community forest management can contribute to stronger communities (Padgee et al 2006)	Improved forest management contributes to sustainable production goals, e.g. thru certification of timber (Rametsteiner and Simula 2003).	See main text on climate mitigation and adaptation	N/A	See main text on desertification and degradation	Sustainable forest management often requires collective action institutions (Ros-Tonen et al 2008).	Sustainable forest management can contribute to increases in demand for wood products (e.g. certification) (McDonald & Lane 2004)

							suitable conditions, improve groundwater recharge (Calder 2005; Ellison et al. 2017a; Neary et al. 2009b). Particular activities associated with forest landscape restoration, such as mixed planting, assisted natural regeneration, and reducing impact of disturbances (e.g. prescribed burning) have positive implications for fresh water supply (Ciccarese et al. 2012; Suding et al. 2015).											
	Reduced deforestation and degradation	May contribute to poverty reduction but conflicting data. Although poverty is a focus of many REDD+ projects (Arhin 2014), evidence is thin that poverty reduction has actually happened (Corbera et al. 2017; Porkorny et al 2013; Scheba 2018) and in some cases benefits have been captured by wealthier	Avoided deforestation can affect crop production when competition for land occurs (Angelsen 2010).	Reduced deforestation can enhance human well-being by microclimatic regulation for protecting people from heat stresses (Locatelli et al. 2015c) and generally improve the cultural and recreational value of ecosystems (Knoke et al. 2014).	N/A	Unclear how avoided deforestation might enhance gender equity, but REDD+ projects need to pay attention to gender issues to be successful (Westholm & Arora-Jonsson 2015)	Forests tend to maintain water quality by reducing runoff and trapping sediments and nutrients (Idris Medugu et al. 2010c; Salvati et al. 2014b). Due to evapotranspiration, trees recharge atmospheric moisture, contributing to rainfall locally and in distant location, and trees'	Avoiding deforestation can take biofuel land out of production as they both tend to compete for land (Dixon et al. 2016)	Reduced forest exploitation may decrease GDP and thus needs to be compensated for (e.g. REDD+) (Motel et al 2009)	N/A	REDD+ has been shown to have no impact on inequality (Shresta et al 2017) or to increase inequality in some project areas (Andersson et al 2018; Pelletier et al 2018)	N/A	N/A	See main text on climate mitigation and adaptation	N/A	See main text on desertification and degradation	N/A	Likely to contribute to decline in trade in forest products, but increases in partnerships between donors and countries with REDD+ (Motel et al 2009).

		participants					microbial flora and biogenic volatile organic compounds can directly promote rainfall (Arneth et al. 2010). Trees enhance soil infiltration and, under suitable conditions, improve groundwater recharge (Calder 2005; Ellison et al. 2017a; Neary et al. 2009b).											
	Reforestation	May contribute to poverty reduction but conflicting data (Tschakert 2007). Many projects for reforestation may have some small impacts on poor households, while others actually increased poverty due to land losses or lack of economic impacts (Jindal et al 2008).	Forest expansion can affect crop production when competition for land occurs (Angelsen 2010). An increase in global forest area can lead to increases in food prices through increasing land competition (Calvin et al. 2014b; Kreidenweis et al. 2016c; Reilly et al. 2012b; Smith et al. 2013a; Wise et al. 2009b)	Reforestation can enhance human well-being by microclimatic regulation for protecting people from heat stresses (Locatelli et al. 2015c) and generally improve the cultural and recreational value of ecosystems (Knoke et al. 2014). Trends of forest resources of nations are found to positively correlate with UNDP Human Development Index (Kauppi et al. 2018).	N/A	N/A	Particular activities associated with forest landscape restoration, such as mixed planting, assisted natural regeneration, and reducing impact of disturbances (e.g. prescribed burning) have positive implications for fresh water supply (Ciccarese et al. 2012; Suding et al. 2015).	Reforestation can increase availability of biomass for energy (Swischer 1994).	Reforestation often require employment for active replanting, etc. (Jindal et al 2008)	N/A	N/A	N/A	N/A	See main text on climate mitigation and adaptation	N/A	See main text on desertification and degradation	N/A	N/A
	Afforestation	Although some have argued that afforestation can be a tool for poverty	Future needs for food production are a constraint for large-scale afforestation	Afforestation can enhance human well-being by microclimat	N/A	N/A	Afforestation using some exotic species can upset the	Afforestation may increase availability of biomass for energy use (Obersteiner	Afforestation often requires employment for active	N/A	N/A	N/A	N/A	See main text on climate mitigation and	N/A	See main text on desertification and degradation	N/A	N/A

		reduction (Holden et al 2003), afforestation can compete with land available for cropping and poor farmers often do not benefit from afforestation projects (McElwee 2009)	plans (Locatelli et al. 2015c). Global food crop demand is expected by 50%-97% between 2005 and 2050 (Valin et al. 2014). Future carbon prices will facilitate deployment of afforestation projects at expenses of food availability (adverse side-effect), but more liberalised trade in agricultural commodities could buffer food price increases following afforestation in tropical regions (Kreidenweis et al. 2016c)	ic regulation for protecting people from heat stresses (Locatelli et al. 2015c) and generally improve the cultural and recreational value of ecosystems (Knoke et al. 2014). Trends of forest resources of nations are found to positively correlate with UNDP Human Development Index (Kauppi et al. 2018)			balance of evapotranspiration regimes, with negative impacts on water availability particularly in arid regions (Ellison et al. 2017a; Locatelli et al. 2015c; Trabucco et al. 2008). Afforestation in arid and semiarid regions using species that have evapotranspiration rates exceeding the regional precipitation may aggravate the groundwater decline (Locatelli et al. 2015a; Lu et al. 2016). Changes in runoff affect water supply but can also contribute to changes in flood risks, and irrigation of forest plantations can increase water consumption (Sterling et al. 2013)	et al 2006)	replanting, etc. (Mather & Murray 1987).					adaptation				
Soil management	**Increased soil organic carbon content**	Can increase yields for smallholders, which can contribute to poverty	Lal (2006b) notes that "Food-grain production in developing countries can	There is evidence that increasing soil organic carbon	N/A	Gender impacts use of soil organic matter practices	Soil organic matter is known to increase water	N/A	Increased agricultural production generally	N/A	Increased agricultural production can contribute	N/A	Improved conservation agriculture	See main text on climate mitigation and	Rivers transport dissolved organic matter to oceans	See main text on desertification and degradation	N/A	N/A

		reduction, but because adoption often depends on exogenous factors these need to be taken into consideration (Wollni et al 2010; Kassie et al 2013).	be increased by 24-39 (32+-11) million Mgy-1 through improving soil quality by increasing the SOC pool and reversing degradation processes".	could be effective in reducing the prevalence of disease-causing helminths (Lal 2016; Wall et al. 2015). Also indirectly contributes to food productivity which may have impact on diets.		(Quansah et al 2001) but N/A how the relationship works in reverse.	filtration and protects water quality (Lehmann & Kleber 2015)		(Lal 2006c) contributes to increased economic growth.		to reducing inequality among smallholders (Datt & Ravallion 1998).		contributes to sustainable production goals (Hobbs et al. 2008).	adaptation	(Hedges et al 1997), but unclear if improved SOM will decrease this and by how much.			
	Reduced soil erosion	Can increases yields for smallholders and contributes to poverty reduction (Ananda & Herath 2003)	Contributes to agricultural productivity and reduces food insecurity (Pimentel et al. 1995; Shiferaw & Holden 1999).	Contributes to food productivity and improves farmer health (Pimentel et al. 1995; Shiferaw & Holden 1999).	N/A	N/A	Various researchers showed a relationship between impact of soil erosion and degradation on water quality indicating the source of pollutant as anthropogenic and industrial activities. in China (Issaka & Asheraf 2017). Managing soil erosion improves water quality (Pimentel et al 1995)	N/A	N/A	N/A	N/A	Particulate matter pollution, a main consequence of wind erosion, imposes severe adverse impacts on materials, structures and climate which directly affect the sustainability of urban cities (Al-Thani et al. 2018)	N/A	See main text on climate mitigation and adaptation	N/A	See main text on desertification and degradation	N/A	N/A
	Reduced soil salinisation	Salinisation can impoverish farmers (Duraiappah 1998) therefore preventing or reversing can increases yields for smallholders and contributes to poverty reduction.	Reversing degradation contributes to food productivity and reduces food insecurity (Pimiental et al. 1995; Shiferaw & Holden 1999).	Salinisation is known to have human health impacts: wind-borne dust and respiratory health; altered ecology of mosquito-borne diseases; and mental health consequences (Jardine	N/A	N/A	Management of soil salinity improves water quality and quantity (Kotb et al. 2000; Zalidis et al 2002)	N/A	N/A	N/A	N/A	N/A	N/A	See main text on climate mitigation and adaptation	N/A	See main text on desertification and degradation	N/A	N/A

				et al 2007)														
	Reduced soil compaction	Soil compaction and other forms of degradation can impoverish farmers (Scherr 2000); prevention of compaction thus contributes to poverty reduction.	Compactions reduces agricultural productivity and thus contributes to food insecurity (Nawaz et al 2013)	Soil compaction has human health consequences as it contributes to runoff of water and pollutants into surface and groundwaters (Soane and van Ouwerkerk 1994)	N/A	N/A	Management of soil compaction improves water quality and quantity (Soane and van Ouwerkerk 1994; Zalidis et al 2002)	N/A	N/A	N/A	N/A	N/A	N/A	See main text on climate mitigation and adaptation	N/A	See main text on desertification and degradation	N/A	N/A
	Biochar addition to soil	Land to produce biochar may reduce land available for smallholders, and it tends to be unaffordable for poor farmers; as of yet, few biochar projects have shown poverty reduction benefits (Leach et al 2012)	Could potentially affect crop production if competition for land occurs (Ennis et al 2012)	N/A	N/A	N/A	Biochar improves soil water filtration and retention (Spokas et al 2011)	N/A	N/A	N/A	N/A	N/A	N/A	See main text on climate mitigation and adaptation	N/A	See main text on desertification and degradation	N/A	N/A
	Fire management	N/A	N/A	Fire management reduces health risks from particulates (Bowman & Johnston 2005).	N/A	N/A	Fires affect water quality and flow due to erosion exposure (Townsend & Douglas 2000).	N/A	N/A	N/A	N/A	Wildfires can threaten property and human health in urban areas, with unique vulnerabilities (Gill & Stevens 2009; Winter & Fried 2010), therefore management will reduce risk to urban areas.	N/A	See main text on climate mitigation and adaptation	N/A	See main text on desertification and degradation	N/A	N/A
Other ecosystem management	**Reduced landslides and natural hazards**	Landslides can increase vulnerability to poverty (Msilimba 2010), therefore management will reduce risks to the poor	Landslides are one of the natural disasters that have impacts on food security (de Haen & Hemrich 2007)	Managing landslides reduces health risks (Haines et al 2006)	N/A	N/A	N/A	N/A	N/A	N/A	N/A	Landslide hazards are a major risk to urban areas (Smyth & Royle 2000).	N/A	See main text on climate mitigation and adaptation	N/A	See main text on desertification and degradation	N/A	N/A
	Reduced pollution	N/A	N/A	Reducing acid	N/A	N/A	Pollution increases	N/A	N/A	Management of	N/A	Management of pollution can	N/A	See main	Reduction in pollution	See main text on	N/A	N/A

	including acidification			deposition reduces health risks, including respiratory illnesses and increased morbidity (Lübkert-Alcamo & Krzyzanowski 1995; Larssen et al 1999)			acidity of surface water, with likely ecological effects (Larssen et al 1999)			pollution can increase demand for new technologies (Popp 2006).		reduce exposure to health risks in urban areas (Bartone 1991)		text on climate mitigation and adaptation	can improve water quality running to oceans (Doney et al 2007).	desertification and degradation		
	Management of invasive species / encroachment	Invasive species removal policies have been beneficial to the poor (van Wilgen & Wannenburgh 2016)	IAS can compete with crops and reduce crop yields by billions of dollars annually (Pejchar & Mooney 2009)	IAS have strong negative effects on human well-being (Pejchar & Mooney 2009)	N/A	N/A	IAS like the golden apple snail/zebra mussel have damaged aquatic ecosystems (Pejchar & Mooney 2009)	N/A	IAS removal policies can increased employment due to need for labor (van Wilgen & Wannenburgh 2016)	N/A	N/A	N/A	N/A	See main text on climate mitigation and adaptation	N/A	See main text on desertification and degradation	N/A	N/A
	Restoration and avoided conversion of coastal wetlands	Impacts on poverty are mixed (Kumar et al 2011). May reduce land available for cropping, and poor design can impoverish people (Ingram et al 2006; Mangora 2011). Can also decrease vulnerability to coastal storms, however (Jones et al. 2012; Feagin et al 2010)	Mixed evidence: can affect agriculture/fisheries production when competition for land occurs, or could increase food production when ecosystems are restored (Crooks et al 2011)	Wetlands contribute to local well-being (Crooks et al 2011), and restoration generally improve the cultural and recreational value of ecosystems (Knoke et al. 2014).	N/A	N/A	Wetlands store freshwater and enhance water quality (Bobbink et al 2006)	N/A	Restoration projects often require employment for active replanting, etc. (Crooks et al. 2011).	Protecting coastal wetlands may reduce infrastructure projects in coastal areas (e.g. sea dikes, etc.) (Jones et al. 2012)	N/A	N/A	N/A	See main text on climate mitigation and adaptation	Restoration of coastal wetlands can play a large role in providing habitat for marine fish species (Bobbink et al 2006; Hale et al 2009)	See main text on desertification and degradation	N/A	N/A
	Restoration and avoided conversion of peatlands	May reduce land available for smallholders in tropical peatlands (Jewitt et al 2014)	Can affect crop production when competition for land occurs, although much use of peatlands in tropics is for palm oil, not food (Sellamuttu et al 2011)	N/A	N/A	N/A	Peatland restoration will improve water quality as they play important roles in water retention and drainage (Johnston 1991).	Peatlands in tropics are often used for biofuels and palm oil, so may reduce the availability of these (Danielsen et al 2008).	Reduced peatland exploitation may decrease GDP in Southeast Asia (Koh et al 2011)	N/A	N/A	N/A	N/A	See main text on climate mitigation and adaptation	N/A	See main text on desertification and degradation	N/A	N/A

	Biodiversity conservation	There is mixed evidence on the impacts of biodiversity conservation measures on poverty	Biodiversity, and its management, is crucial for improving sustainable and diversified diets (Global Panel on Agriculture and Food Systems for Nutrition 2016). Indirectly, the loss of pollinators (due to combined causes, including the loss of habitats and flowering species) would contribute to 1.42 million additional deaths per year from non-communicable and malnutrition-related diseases, and 27.0 million lost disability-adjusted life-years (DALYs) per year (Smith et al. 2015). However, at the same time, some options to preserve biodiversity, like protected areas, may potentially conflict with food production by local communities (Molotoks et al. 2017)	Biodiversity, and its management, is crucial for improving sustainable and diversified diets (Global Panel on Agriculture and Food Systems for Nutrition 2016).	N/A	N/A	33 out of 105 of the largest urban areas worldwide rely on biodiversity conservation measures such as protected areas for some, or all, of their drinking water (Secretariat of the Convention on Biological Diversity 2008)	Some biodiversity conservation measures might increase access to biomass supplies (Erb et al. 2012)							Biodiversity conservation measures like protected areas can increase ocean biodiversity (Selig et al 2014)	Indigenous peoples' roles in biodiversity conservation can increase institutions and conflict resolution (Garnett et al. 2018)	Indigenous peoples commonly link forest landscapes and biodiversity to tribal identities, association with place, kinship ties, customs and protocols, stories, and songs (Gould 2014; Lyver et al. 2017a, b).	
	Enhanced weathering of minerals	N/A	N/A	N/A	N/A	N/A	Mineral weathering can affect the chemical composition of soil and surface waters (Katz	N/A	N/A	Will require development of new technologies (Schuiling and Krijgsman	N/A	N/A	N/A	See main text on climate mitigation and adaptation	N/A	See main text on desertification and degradation	N/A	N/A

							1989)			2006)								
CDR	Bioenergy and BECCS	Bioenergy production could create jobs in agriculture, but could also compete for land with alternative uses. Therefore, bioenergy could have positive or negative effects on poverty rates among smallholders, among other social effects (IPCC 2018).	Biofuel plantations may lead to decreased food security through competition for land (Locatelli et al. 2015c). BECCS will likely lead to significant trade-offs with food production (Popp et al. 2011c; Smith et al. 2016b).	BECCS could have positive effects through improvements in air and water quality (IPCC 2018), but BECCS could have negative effects on health and wellbeing through impacts on food systems (Burns and Nicholson 2017). Additionally, there is a non-negligible risk of leakage of sequestered CO2 (IPCC 2018).	No direct interaction (IPCC 2018).	No direct interaction (IPCC 2018).	Will likely require water for plantations of fast growing trees and models show high risk of water scarcity if BECCS is deployed on widespread scale (IPCC 2018).	BECCS and biofuels can contribute up to 300 EJ of primary energy by 2100 (cross-chapter box 7 on bioenergy); bioenergy can provide clean, affordable energy (IPCC 2018).	Access to clean, affordable energy will help economic growth (IPCC 2018).	BECCS will require development of new technologies (Smith et al. 2016c).	No direct interaction (IPCC 2018).	No direct interaction (IPCC 2018).	Switching to bioenergy reduces depletion of natural resources (IPCC 2018).	See main text on climate mitigation and adaptation	Reductions in carbon emissions will reduce ocean acidification. See main text on climate mitigation.	See main text on desertification and degradation	No direct interaction (IPCC 2018).	No direct interaction (IPCC 2018).

Table SM6.13 Impacts on the UN SDG of integrated response options based on value chain interventions

<u>Integrated response options based on value chain management</u>		GOAL 1: No Poverty	GOAL 2: Zero Hunger	GOAL 3: Good Health and Well-being	GOAL 4: Quality Education	GOAL 5: Gender Equality	GOAL 6: Clean Water and Sanitation	GOAL 7: Affordable and Clean Energy	GOAL 8: Decent Work and Economic Growth	GOAL 9: Industry, Innovation and Infrastructure	GOAL 10: Reduced Inequality	GOAL 11: Sustainable Cities and Communities	GOAL 12: Responsible Consumption and Production	GOAL 13: Climate Action	GOAL 14: Life Below Water	GOAL 15: Life on Land	GOAL 16: Peace and Justice Strong Institutions	GOAL 17: Partnerships to achieve the Goal
Demand management	Dietary change	Reduced meat consumption can free up land for other activities to reduce poverty (Röös et al. 2017; Stoll-Kleemann and O'Riordan 2015). However, reduced demand for livestock will have negative effect on pastoralists and could suppress	High-meat diets in developed countries may limit improvement in food security in developing countries (Rosegrant et al. 1999); dietary change can contribute to	Overnutrition contributes to worse health outcomes, including diabetes and obesity (Tilman and Clark 2014a; McMichael et al. 2007). Dietary change away from meat consumption	No direct interaction (IPCC 2018)	No direct interaction (IPCC 2018)	Reduced meat consumption will reduce water consumption. (Muller et al. 2017b) found that lower impact agriculture could be practiced if dietary change and	Dietary shifts away from meat to fish/fruits/vegetables increases energy use in the US by over 30% (Tom et al. 2016)	Health costs of meat-heavy diets add to health care costs and reduce GDP (Popkin 2008)	N/A	There are currently large discrepancies in diets between developed and developing nations (Sans & Combris 2015). Dietary change will reduce food inequality by reducing meat overconsumption in Western countries and	Dietary change is most needed in urbanised, industrialised countries and can help contribute to demand for locally grown fruits and vegetables (Tom et al. 2016)	A dietary shift away from meat can contribute to sustainable consumption by reducing greenhouse gas emissions and reducing cropland and pasture requirements (Stehfest et al. 2009; Bajželj	See main text on climate mitigation and adaptation	Dietary change away from meat might put increased pressure on fish stocks (Vranken et al. 2014; Mathijs 2015). Overall reduced emissions	See main text on desertification and degradation	N/A	N/A

		demand for other inputs (grains) that would affect poor farmers (Garnett 2011; IPCC SR15)	food security goals (Godfray et al. 2010a; Bajželj et al. 2014)	has major health benefits, including reduced heart disease and mortality (Popkin 2008; Friel et al. 2008). Dietary change could contribute to 5.1 million avoided deaths per year (Springmann et al. 2016)			waste reduction were implemented, leading to lower GHG emissions, lower rates of deforestation, and decreases in use of fertiliser (nitrogen and phosphorus), pesticides, water and energy. However, Tom et al. (2016) found water footprints of fruit/veg dietary shift in the US to increase by 16%				free up some cereals for consumption in poorer diets (Rosegrant et al. 1999)		et al. 2014).		would decrease rate of ocean acidification (Doney et al. 2009)			
	Reduced post-harvest losses	Reducing food losses from storage and distribution operation can increase economic well-being without additional investment in production activities (Bradford et al. 2018; Temba et al. 2016)	Reducing food losses increases food availability, nutrition, and lower prices (Sheahan and Barrett 2017b; Abass et al. 2014; Affognon et al. 2015)	Improved storage enhances food quality and can reduce mycotoxin intake (Bradford et al. 2018; Temba et al. 2016; Stathers et al. 2013; Tirado et al. 2010) especially in humid climates (Bradford et al. 2018). The perishability and safety of fresh foods are highly susceptible to temperature increase (Bisbis et al. 2018; Ingram et al. 2016a).	Reduced losses can increase income that could be spent on education, but no data available	Postharvest losses do have a gender dimension (Kaminski and Christiaensen 2014), but unclear if reducing losses will contribute to gender equality (Rugumamu 2009)	Kummu et al. (2012a) reported that 24% of global freshwater use and 23% of global fertiliser use is attributed to food losses. Reduced post harvest losses can decrease need for additional agricultural production and irrigation.	Reduced losses would reduce energy demands in production; 2030 +-160 trillion BTU of energy were embedded in wasted food in 2007 in the US (Cuéllar and Webber 2010)	In East and Southern Africa, postharvest loss for six major cereals was USD1.6 billion or 15% of total production value; reducing losses would thus boost GDP substantially in developing countries with PHL (Hodges et al. 2011)	Reducing PHL can involve improving infrastructure for farmers and marketers (Parfitt et al. 2010)	Poorer households tend to experience more PHL, and thus reducing PHL can contribute to reducing inequality among farmers (Hodges et al. 2011).	N/A	Reducing PHL contributes to sustainable production goals (Parfitt et al. 2010)	See main text on climate mitigati on and adaptati on	N/A	See main text on desertification and degradation	N/A	Post harvest losses contribute to higher food prices and constraints on trade (Tefera 2012)
	Reduced food waste (consumer or retailer)	Food waste tends to rise as incomes rise (Parfitt et al. 2010; Liu et al. 2013), so it is not clear what the relationship to poverty is. Could be potentially beneficial as it would free up	People who are already food insecure tend not to waste food (Nahman et al. 2012). Reduced food waste would	Food waste can increase with healthier diets (Parizeau et al. 2015). Health and safety standards can restrict some approaches to reducing food	N/A	Reducing food waste within households often falls to women (Stefan et al. 2013) and can increase their labor workload	Kummu et al. (2012a) reported that 24% of global freshwater and 23% of global fertiliser is used in the production of	Reduced losses would reduce energy demands in production; 2030 +-160 trillion BTU of energy were embedded in wasted food in 2007 in the US (Cuéllar and Webber 2010). Food waste can be a	Waste generation has grown faster than GDP in recent years (Thogerson 1996). Households in the UK throw out USD745	Food waste could be an important source of needed chemicals for industrial development in resource constrained countries (Lin et	Wealthier households tend to waste more food (Parfitt et al. 2010), but unclear how reducing waste may contribute to reducing inequality.	There have been large increases in the throughput of materials such as the food-waste stream, import and solid-waste	Post-consumer food waste in industrialised countries (222 million ton) is almost as high as the total net food production in sub- Saharan Africa (230	See main text on climate mitigati on and adaptati on	Reducing food waste may be related to food packaging, which is a major source of ocean pollution,	See main text on desertification and degradation	N/A	Food waste can contribute to higher food prices and constraints on trade (Tefera 2012)

		money to spend on other activities (Dorward 2012). Redistribution of food surplus to the poor could also have impacts on poverty (Papargyropoulou et al. 2014)	increase the supply of food (FAO 2011; Smith 2013), but it is unclear if this would benefit those who are food insecure in developing countries (Hertel and Baldos 2016).	waste (Halloran et al. 2014). Changes in packaging to reduce waste might have negative health impacts (e.g. increased contamination) (Claudio 2012)		(Hebrok and Boks 2017). Women also generate more food waste and could be a site for intervention (Thyberg and Tonjes 2016)	food losses, so reduction in food waste could provide significant co-benefits for freshwater provision and on nutrient cycling (Kummu et al. 2012). Muller et al. (2017b) found that lower impact agriculture could be practiced if dietary change and waste reduction were implemented, leading to lower GHG emissions, lower rates of deforestation, and decreases in use of fertiliser (nitrogen and phosphorus), pesticides, water and energy.	sustainable source of biofuel (Uçkun Kiran et al. 2014)	of food and drink each year as food waste; South Africans throw out USD 7billion worth of food per year (Nahman and de Lange 2013). Reductions of postconsumer waste would increase household income (Hodges et al. 2011)	al. 2013)		accumulation in urban areas (Grimm et al. 2008). Reducing compostable food waste reduces need for landfills (Smit and Nasr 1992; Zaman and Lehmann 2011)	million ton). (FAO 2011), thereby reducing waste contributes to sustainable consumption.		but relationship is not known (Hornweg et al 2013)			
	Material substitution	N/A	Could increase demand for wood and compete with land for agriculture, but no evidence of this yet.	N/A	N/A	N/A	If water is used efficiently in production of wood, likely to be positive impact over cement production (Gustavsson and Sathre 2011)	Concrete frames require 60-80% more energy than wood (Börjesson and Gustavsson 2000). Material substitution can reduce embodied energy of buildings construction by up to 20% (Thormark 2006; Upton et al. 2008)	The relationship between material substitution and GDP growth is unclear (Moore et al. 1996)	Material substitution may reduce need for industrial production of cement etc. (Petersen and Solberg 2005)	N/A	Changing materials for urban construction can reduce cities' ecological footprint (Zaman and Lehmann 2013)	Material substitution is a form of sustainable production/consumption which replaces cement and other energy-intensive materials with wood (Fiksel 2006)	See main text on climate mitigation and adaptation	Overall reduced emissions would decrease rate of ocean acidification (Doney et al. 2009)	See main text on desertification and degradation	N/A	N/A
Supply management	**Sustainable sourcing**	Value adding has been promoted as a successful poverty reduction strategy in many countries (Lundy et al. 2002; Whitfield 2012; Swanson 2006). Volatility of food supply and food price spikes in 2007 increased the number of people	Poor farmers can benefit from value-adding and new markets (Bamman 2007) and may help to improve food security by increasing its economic performance	Value-chains can help increase the nutritional status of food reaching consumers (Fan et al. 2012)	Value-adding can increase income that could be spent on education, but no data available	Women are highly employed in value-added agriculture in many developing countries, but do not always gain substantive benefits (Dolan and Sorby 2003).	Value-added products might require additional water use (Guan and Hubacek 2007), but depends on context.	N/A	Value-adding and export diversification generates additional employment and expands GDP in developing countries in particular (Newfarmer et al. 2009)	Value adding can create incentives to improve infrastructure in processing (Delgado 2010). Expanding value chains can incorporate new sources of food producers into industrial systems of	Value-adding can be an important component of additional employment for poorer areas, and can contribute to reductions in overall inequality. However, data shows high-value agriculture	Value-adding can increase incentives to keep peri-urban agriculture, but faces threats from rising land prices in urban areas (Midmore and Jansen	Value-adding in agriculture (.e.g. fair trade, organic) can be an important source of sustainable consumption and production (de Haen and Réquillart 2014)	See main text on climate mitigation and adaptation	N/A	See main text on desertification and degradation	N/A	Value-adding has a strong relationship to expanding trade in developing countries in particular (Newfarmer et al. 2009)

		under the poverty line by between 100 million people (Ivanic and Martin 2008) to 450 million people (Brinkman et al. 2009), and caused welfare losses of 3% or more for poor households in many countries (Zezza et al. 2009).	and revenues to local farmers (Reidsma et al. 2010). However, much value-adding is captured upstream, not by poor producers (McMichael and Schneider 2011b). Food prices strongly affect food security (Lewis and Witham 2012; Regmi and Meade 2013; Fujimori et al. 2018a), and policies to decrease volatility will likely have strong impacts on food security (Timmer 2009; Torlesse et al. 2003b; Raleigh et al. 2015b).			Value-chains that target women could increase gender equity, but data is scare (Gengenbach et al. 2018)				distribution (Bloom and Hinrichs 2011)	is not always a pathway toward enhanced welfare (Dolan and Sorby 2003), and much value-adding is captured not by smallholders but higher up the chain (Neilson 2007)	2003)						
	Management of supply chains	Reducing food transport costs generally helps poor farmers (Altman et al. 2009). More than USD200 million is generated in fresh fruit and veg trade between Kenya and the UK; much has contributed to poverty reduction and better transport could increase the amount generated (MacGregor and Vorley 2006; Muriithi and Matz 2015). Volatility of food supply and food price spikes in 2007 increased the number of people under the poverty line by between 100 million people (Ivanic and Martin	Improving storage efficiency can reduce food waste and health risks associated with poor storage management practices (James and James 2010a; Bradford et al. 2018; Temba et al. 2016; Stathers et al. 2013; Tirado et al. 2010). There is some limited evidence that improved transport on-farm increases	Access to quality food is a major contributor to whether a diet is healthy or not (Neff et al. 2009). Increased distribution and access of packaged foods however can decrease health outcomes (Galal et al. 2010; Monteiro et al. 2011)	Reduction in staple food price costs to consumers in Bangladesh from food stability policies saved rural households USD887 million total (Torlesse et al. 2003b), but N/A if this increased spending on education in households	Women and girls are often the most effected ones in households when there are food shortages (Kerr 2005; Hadley et al. 2008)	Food imports can contributed to water scarcity through "embodied" or "virtual" water accounting (Yang and Zehnder 2002; Guan and Hubacek 2007; Hanjra and Qureshi 2010; Jiang 2009)	Food supply chains and flows have adverse effects due to reliance on non-renewable energy (Kurian 2017; Scott 2017). Shifts to biofuels can destabilise food supplies (Tirado et al. 2010; Chakauya et al. 2009)	Food supply instability is often driven by price volatility, which can be driven by rapid economic growth and which can contribute to consumer price inflation and higher import costs as a percentage of GDP leading to account deficits (Gilbert and Morgan 2010)	Excessive disruptions in food supply can place strains on infrastructure (e.g. needing additional storage facilities) (Yang and Zehnder 2002). Improved food transport can create demands for improved infrastructure (Akkerman et al. 2010; Shively and Thapa 2016). For example, weatherproofing transport systems and improving the efficiency of food trade (Ingram et al. 2016a; Stathers et al. 2013)	Food volatility makes it more challenging to supply food to vulnerable regions, and likely increases inequality (Baldos and Hertel 2015; Frank et al. 2017; Porter et al. 2014; Wheeler and von Braun 2013). Improved food distribution could reduce inequality in access to high quality nutritious foods. Food insecure consumers benefit from better access and distribution (e.g. elimination of food deserts) (Ingram 2011;	Improved food distribution can contribute to better food access and stronger urban communities (Kantor 2001; Hendrickson et al. 2006). Food price spikes often hit urban consumers the hardest in food importing countries, and increasing stability can reduce risk of food riots (Cohen and Garrett 2010)	Improved storage and distribution are likely to contribute to sustainable production by impacting biomass of paper/card and aluminum and iron-ore mining used for food packaging (Ingram et al. 2016a).	See main text on climate mitigation and adaptation	N/A	See main text on desertification and degradation	N/A	Better transport improves chances for expanding trade in developing countries (Newfarmer et al. 2009), Well-planned trade systems may act as a buffer to supply food to vulnerable regions (Baldos and Hertel 2015; Frank et al. 2017; Porter et al. 2014; Wheeler and von Braun 2013).

		2008) to 450 million people (Brinkman et al. 2009), and caused welfare losses of 3% or more for poor households	food security in developing countries (Hine 1993).							especially in countries with inadequate infrastructure and weak food distribution systems (Vermeulen et al. 2012a), can strengthen climate resilience against future climate-related shocks (Ingram et al. 2016a; Stathers et al. 2013).	Coveney and O'Dwyer 2009)							
	Enhanced urban food systems	Regional food systems present opportunities for interconnectedness of the food system ' s component resilient food supply systems and city-regions have an important role (Brinkley et al. 2016; Rocha 2016). However, mixed evidence on if urban agriculture contributes to poverty reduction (Ellis and Sumberg 1998)	Food insecurity in urban areas is often invisible (Crush and Frayne 2011). Improved urban food systems manage flows of food into, within, and out of the cities and have large role to play in reducing urban food security (Smit 2016; Benis and Ferrão 2017a; Brinkley et al. 2016; Rocha 2016; Maxwell and Wiebe 1999), particularly in fostering regional food self-reliance (Aldababseh et al. 2018; Bustamante et al. 2014b).	Since urban poor spend a great deal of their budget on food and urban diets are exposed to more unhealthy 'fast foods' (Dixon et al. 2007), local urban food systems can contribute to enhanced nutrition in urban areas (Tao et al. 2015; Maxwell 1999; Neff et al. 2009). However, local urban agriculture also may introduce pollution into food system through toxins in soil and water (Binns et al. 2003)	School feeding programs in urban areas can increase educational attendance and outcomes (Ashe and Sonnino 2013)	Urban and Peri-urban Agriculture and Forestry (UPAF) addresses gender-based differences in accessing food since women play an important role in the provisioning of urban food (Tao et al. 2015; Binns and Lynch 1998). Women also dominate informal urban food provisioning (wet markets, street food) (Smith 1998)	Water access often a constraint on urban agriculture (de Bon et al. 2010; Badami and Ramankutty 2015). Urban agriculture can exacerbate urban water pollution problems (pesticide runoff, etc) (Pothukuchi and Kaufman 1999)	Local food production and use can reduce energy use, due to lower demand of resources for production, transport and infrastructure (Lee-Smith 2010), but depends on context (Mariola 2008; Coley et al. 2009)	Urban food systems have as one aim to stimulate local economic development and increase employment in urban agriculture and food processing (Smith 1998). As many as 50% of some cities' retail jobs are in food-related sector (Pothukuchi and Kaufman 1999)	Urban food provisioning creates demands for expanded infrastructure in processing, refrigeration, and transportation (Pothukuchi and Kaufman 1999)	Many UFS in global South (e.g. Belo Horizonte, Brazil) have goals to reduce inequality in access to food. (Dixon et al. 2007; Allen 2010)	UFS aim at improving the health status of urban dwellers, reducing their exposure to pollution levels, and stimulating economic development (Tao et al. 2015)	UFS aim to combine sustainable production and consumption with local foodsheds (Tao et al. 2015; Allen 2010)	See main text on climate mitigati on and adaptati on	Overall reduced emissions would decrease rate of ocean acidification (Doney et al. 2009)	See main text on desertification and degradation	Building a resilient regional food system requires adjusting to the social and cultural environment and locally-specific natural resource base and building local institutions (Akhtar et al. 2016). Production of food within cities can potentially lead to less likelihood of urban food shortages and conflicts (Cohen & Garrett 2010).	N/A
	Improved food processing and retailing	Food processing has been a useful strategy for poverty reduction in some countries (Weinberger and Lumpkin 2007; Haggblade et al. 2010)	Efficiency in food processing and supply chains can contribute to more food reaching consumers and	Improved processing and distribution &storage systems can provide safer and healthier food to consumers (Vermeulen et	N/A	Improved food processing can displace street venders and informal food sellers, who are predominantly women	Food processing and packaging activities such as washing, heating, cooling are heavily	Food processing and packaging activities such as heating and cooling are heavily dependent on energy so improved efficiency could reduce energy demand (Garcia and	Phytosanitary barriers currently prevent much food export from developing countries, and improvements in processing	Improvements in processing, refrigeration, and transportation will require investments in improved infrastructure	N/A	Improved food transport can reduce cities' ecological footprints and reduce overall emissions (Du et al.	Improved food processing and agro-retailing contributes to sustainable production (Ingram 2011)	See main text on climate mitigati on and adaptati on	Overall reduced emissions would decrease rate of ocean acidification (Doney et	See main text on desertification and degradation	N/A	Improved processing increases chances for expanding trade in developing countries (Newfarmer

			improved nutrition (Vermeulen et al. 2012a; Keding et al. 2013)	al. 2012a) and reduce food waste and health risks associated with poor storage management practices (James and James 2010a), although overpackaged prepared foods that are less healthy are also on rise (Monteiro 2009; Monteiro et al. 2011).		(Smith 1998; Dixon et al. 2007)	dependent on freshwater so improved postharvest storage and distribution could reduce water demand via more efficiently performing systems (Garcia and You 2016).	You 2016).	would increase exports and GDP (Henson and Loader 2001; Jongwanich 2009).	(Ingram 2011)		2006)			al. 2009)			et al. 2009)
	Improved energy use in food systems	Might possibly have impact on poverty by reducing farmer costs, but no data.	Utilising energy-saving strategies can support reduced food waste (Ingram et al. 2016a) and increased production efficiencies (Smith and Gregory 2013).	Organic agriculture is associated with increased energy efficiency, which have can have co-benefits by reduced exposure to agrochemicals by farm workers (Gomiero et al. 2008)	N/A	Increased efficiency might reduce women's labor workloads on farms (Rahman 2010) but data is scarce.	Increased energy efficiency (e.g. in irrigation) can lead to more efficient water use (Rothausen and Conway 2011; Ringler and Lawford 2013)	Increased energy efficiency will reduce demands for energy but can have rebound effect in expanded acreage (Swanton et al. 1996)	There is no clear association between higher energy use in agriculture and economic growth; these have become decoupled in many countries (Bonny 1993). Data is unclear though on economic impacts of potential cost savings.	N/A	N/A	N/A	Reducing energy use in agriculture contributes to sustainable production goals (Ingram et al. 2016a).	See main text on climate mitigation and adaptation	Overall reduced emissions would decrease rate of ocean acidification (Doney et al. 2009).	See main text on desertification and degradation	N/A	N/A

Table SM6.14 Impacts on the UN SDG of integrated response options based on risk management

<u>**Integrated response options based on risk management**</u>		**GOAL 1: No Poverty**	**GOAL 2: Zero Hunger**	**GOAL 3: Good Health and Well-being**	**GOAL 4: Quality Education**	**GOAL 5: Gender Equality**	**GOAL 6: Clean Water and Sanitation**	**GOAL 7: Affordable and Clean Energy**	**GOAL 8: Decent Work and Economic Growth**	**GOAL 9: Industry, Innovation and Infrastructure**	**GOAL 10: Reduced Inequality**	**GOAL 11: Sustainable Cities and Communities**	**GOAL 12: Responsible Consumption and Production**	**GOAL 13: Climate Action**	**GOAL 14: Life Below Water**	**GOAL 15: Life on Land**	**GOAL 16: Peace and Justice Strong Institutions**	**GOAL 17: Partnerships to achieve the Goal**
	Management of urban sprawl	Inner city poverty closely associated with urban sprawl in US context (Frumkin 2002; Powell 1999; Jargowsky	There are likely to be some benefits for food security since it is often agricultural land that is	Strong association between urban sprawl and poorer health outcomes	N/A	N/A	Urban sprawl is associated with higher levels of water pollution due to loss of filtering vegetation and increasing impervious	Sprawling or informal settlements often do not have access to electricity or other services, increasing	Sprawl is associated with rapid economic growth in some areas (Brueckner 2000).	Urban sprawl often increases public infrastructure costs (Brueckner 2000), and densification and	Urban sprawl is associated with inequality (Jargowsky 2002)	Urban sprawl is associated with unsustainability, including increased transport and CO_2 emissions, lack of access to services,	Reducing urban sprawl and promoting community gardens and periurban agriculture can contribute to	See main text on climate mitigation and adaptation	N/A	See main text on desertification and degradation	There are debates over the role of urban sprawl in reducing social capital and weakening	N/A

		2002; Deng and Huang 2004)	sealed by the urban expansion (Barbero-Sierra et al. 2013a). Some evidence for sprawl reducing food production, particularly in China (Chen 2007b)	(air pollution, obesity, traffic accidents) (Frumkin 2002; Lopez 2004; Freudenberg et al. 2005)			surfaces (Romero and Ordenes 2004; Tu et al. 2007)	chances HH rely on dirty fuels (Dhingra et al. 2008)	Reducing urban sprawl is part of many managed "smart growth" plans, which may reduce overall economic growth in return for sustainability benefits (Godschalk 2003)	redevelopment can improve equality of access to infrastructure (Jenks and Burgess 2000).		and loss of civic life (Kombe 2005; Andersson 2006). Sustainable cities include compactness, sustainable transport, density, mixed land uses, diversity, passive solar design, and greening (Chen et al. 2008; Jabareen 2006; Andersson 2006)	more sustainable production in cities (Turner 2011)				participatory governance in cities (Frumkin 2002; Nguyen 2010)	
	Livelihood diversification	Diversification is associated with increased welfare and incomes and decreased levels of poverty in several country studies (Arslan et al. 2018b; Asfaw et al. 2018).	Diversification is associated with increased access to income and additional food sources for the household (Pretty 2003); likely some food security benefits but diversification can also lead to more purchased (unhealthy) foods (Niehof 2004; Barrett et al. 2001)	More diversified livelihoods have diversified diets which have better health outcomes (Block and Webb 2001; Kadiyala et al. 2014) particularly for women and children (Pretty 2003)	More diversified households tend to be more affluent, & have more disposal income for education (Ellis 1998; Estudillo and Otsuka 1999; Steward 2007), but diversification through migration may reduce educational outcomes for children (Gioli et al. 2014)	Women are participants in and benefit from livelihood diversification, such as having increased control over sources of HH income (Smith 2015), although it can increase their labor requirements (Angeles and Hill 2009)	Lack of access to affordable water may inhibit livelihood diversification (Calow et al. 2010)	Access to clean energy can provide additional opportunities for livelihood diversification (Brew-Hammond 2010; Suckall et al. 2015)	Livelihood diversification by definition contributes to employment by providing additional work opportunities (Ellis 1998; Niehof 2004)	N/A	The relationship between livelihood diversification and inequality is inconclusive (Ellis 1998). In some cases diversification reduced inequality (Adams 1994) while in others cases it increases it (Reardon et al 2000)	One part of urban livelihoods in developing countries are linkages between rural and urban areas through migration and remittances (Rakodi 1999; Rakodi & Lloyd 2002); this livelihood diversification can strengthen urban income (Ricci 2012)	Livelihood diversification does not always lead to sustainable production and consumption choices, but it can strengthen autonomy potentially leading to better choices (Elmqvist and Olsson 2007; Schneider and Niederle 2010)	See main text on climate mitigation and adaptation	N/A	See main text on desertification and degradation	N/A	N/A
	Use of local seeds	Many hundreds of millions of smallholders still rely on local seeds; without them they would have to find money to buy commercial seeds (Altieri et al. 2012b; McGuire and Sperling 2016; Howard 2015)	Local seeds revive and strengthen local food systems (McMichael and Schneider 2011b) and lead to more diverse and healthy food in areas with strong food sovereignty networks (Coomes et al. 2015a; Bisht et al. 2018). However local seeds often are less productive than improved varieties.	Local seed use is associated with fewer pesticides (Altieri et al. 2012b); loss of local seeds and substitution by commercial seeds is perceived by farmers to increase health risks (Mazzeo and Brenton 2013), although overall literature on links between food sovereignty and health is weak (Jones	N/A	Women play important roles in preserving and using local seeds (Ngcoya and Kumarakulasingam 2017; Bezner Kerr 2013) and sovereignty movements paying more attention to gender needs (Park et al. 2015)	Local seeds often have lower water demands, as well as less use of pesticides that can contaminate water (Adhikari 2014)	N/A	Food sovereignty supporters believe protecting smallholder agriculture provides more employment than commercial agriculture (Kloppenberg 2010)	N/A	Seed sovereignty advocates believe it will contribute to reduced inequality (Wittman 2011; Park et al. 2015) but there is inconclusive empirical evidence.	Seed sovereignty can help sustainable urban gardening (Demailly and Darly 2017) which can be part of a sustainable city by providing fresh, local food (Leitgeb et al. 2016).	Locally developed seeds can both help protect local agrobiodiversity and can often be more climate resilient than generic commercial varieties, leading to more sustainable production (Coomes et al. 2015a; van Niekerk and Wynberg 2017a).	See main text on climate mitigation and adaptation	N/A	See main text on desertification and degradation	Seed sovereignty is positively associated with strong local food movements, which contribute to social capital (McMichael and Schneider 2011b; Coomes et al. 2015a; Grey and Patel 2015).	Seed sovereignty could be seen as threat to free trade and imports of genetically modified seeds (Kloppenberg 2010; Howard 2015; Kloppenburg 2014)

				et al. 2015)														
	Disaster risk management	DRM can help prevent impoverishment as disasters are a major factor in poverty (Basher 2006; Fothergill and Peek 2004)	Famine early warning systems have been successful to prevent impending food shortages (Genesio et al. 2011; Hillbruner and Moloney 2012)	EWS very important for public health to ensure people can get shelter and medical care during disasters (Greenough et al. 2001; Ebi and Schmier 2005)	N/A	Women often disproportionately affected by disasters; gender-sensitive EWS can reduce their vulnerability (Enarson and Meyreles 2004; Mustafa et al. 2015)	Many EWS include water monitoring components that contribute to access to clean water (Wilhite 2005; Iglesias et al. 2007). Some urban areas use water EWS successfully to monitor levels of contaminants (Hasan et al. 2009; Hou et al. 2013)	N/A	DRM can help minimise damage from disasters, which impacts economic growth (Basher 2006)	DRM can help protect infrastructures from damage during disaster (Rogers and Tsirkunov 2011)	EWS can ensure inequality is taken into account when making predictions of impacts (Khan et al. 1992)	EWS can be very effective in urban settings such as heat wave EWS and flooding EWS to minimise vulnerability (Parnell et al. 2007; Bambrick et al. 2011; Djordjević et al. 2011)	DRM can make sustainable production more possible by providing farmers with advance notice of environmental needs (Stigter et al. 2000; Parr et al. 2003)	See main text on climate mitigation and adaptation	EWS can play important role in marine management, e.g. warnings of red tide, tsunami warnings for coastal communities (Lee et al. 2005; Lauterjung et al. 2010)	See main text on desertification and degradation	DRM can reduce risk of conflict (Meier et al. 2007), increase resilience of communities (Mathbor 2007) and strengthen trust in institutions (Altieri et al. 2012b)	N/A
	Risk sharing instruments	Crop insurance reduces risks which can improve poverty outcomes by avoiding catastrophic losses, but is often not used by poorest people (Platteau et al. 2017)	Availability of crop insurance has generally lead to (modest) expansions in cultivated land area and increased food production (Claassen et al. 2011; Goodwin et al. 2004)	General forms of social protection lead to better health outcomes; unclear how much crop insurance contributes (Tirivayi et al. 2016)	Households lacking insurance may withdraw children from school after crop shocks (Jacoby and Skoufias 1997; Bandara et al. 2015)	Women farmers vulnerable to crop shocks, but tend to be more risk-averse and skeptical of commercial insurance (Akter et al. 2016; Fletschner and Kenney 2014)	Crop insurance can be indexed to weather and water access and thereby increase adapation to water stress (Hoff and Bouwer 2003). Subsidised insurance can also be linked to reductions in pesticide use to reduce non-point source pollution, which has shown success in the US and China (Luo et al. 2014)	N/A	Subsidised crop insurance contributes to economic growth in the US (Atwood et al. 1996) but at considerable cost to the governance (Glauber 2004).	N/A	N/A	N/A	Crop insurance has been implicated as a driver of unsustainable production and disincentive to diversification (Bowman and Zilberman 2013), although community risk sharing might increase diversification and production	See main text on climate mitigation and adaptation	There is mixed evidence that crop insurance may encourage excess fertiliser use (Kramer et al. 1983; Wu 1999; Smith and Goodwin 1996), which contributes to ocean pollution; however, some governments re requiring reductions in nonpoint source pollution from farms otherwise farmers lose crop insurance (Iho et al. 2015)	See main text on desertification and degradation	Community risk sharing instruments can help strenthen resilience and institutions (Agrawal 2001)	Subsidised crop insurance can be seen as a subsidy and barrier to trade (Young and Westcott 2000)

Supplementary Information for Section 6.4.4

	IAM Study	C	M	A	D	L	F	O
Alexander et al. 2018	No			Yes				Yes
Baker et al. 2019a	No		Yes					
Baldos and Hertel 2014	No						Yes	
Bauer et al. 2018	Yes		Yes					
Bertram et al. 2018	Yes		Yes				Yes	Yes
Brink et al. 2018	Mixed				Yes	Yes	Yes	Yes
Calvin et al. 2013b	Yes		Yes	Yes				
Calvin et al. 2014b	Yes		Yes				Yes	Yes
Calvin et al. 2016a	Yes		Yes					
Calvin et al. 2016b	Yes		Yes					
Calvin et al. 2017c	Yes		Yes				Yes	
Calvin et al. 2019	Yes		Yes					Yes
Chaturvedi et al. 2013	Yes		Yes					Yes
Clarke et al. 2014a	Yes	Yes	Yes					Yes
Collins et al. 2013	No	Yes						
Daioglou et al. 2019	Yes		Yes					
Doelman et al. 2018	Yes		Yes				Yes	
Edmonds et al. 2013b	Yes		Yes					
Favero and Massetti 2014	Yes	Yes	Yes					
Frank et al. 2015	IAM-land		Yes					
Frank et al. 2017	Yes		Yes				Yes	
Fricko et al. 2017	Yes		Yes					
Fujimori et al. 2017b	Yes		Yes					
Fujimori et al. 2018a	Yes		Yes				Yes	
Fujimori et al. 2019	Mixed		Yes				Yes	
Gao and Bryan 2017b	No		Yes			Yes	Yes	Yes
Graham et al. 2018b	Yes							Yes
Grubler et al. 2018	Yes		Yes				Yes	Yes
Hanasaki et al. 2013b	Yes							Yes
Harrison et al. 2016	Yes							Yes
Hasegawa et al. 2015a	Yes						Yes	
Hasegawa et al 2015b	Yes						Yes	
Hasegawa et al. 2018	Mixed			Yes			Yes	
Heck et al. 2018	Mixed	Yes	Yes					Yes
Hejazi et al. 2014c	Yes		Yes					Yes
Hejazi et al. 2015d	Yes		Yes					Yes
Humpenöder et al. 2014	Yes		Yes					
Humpenöder et al. 2018b	IAM-land		Yes				Yes	Yes
Iyer et al. 2018	Yes		Yes				Yes	Yes
Jones et al., 2013	Yes	Yes						
Jones et al. 2015	Yes		Yes					
Kim et al. 2016a	Yes			Yes			Yes	Yes
Kraxner et al. 2013	No		Yes					Yes
Kreidenweis et al. 2016a	Yes		Yes				Yes	
Kriegler et al. 2017	Yes		Yes				Yes	
Kriegler et al. 2018a	Mixed		Yes					
Kriegler et al. 2018b	Yes		Yes					
Kyle et al. 2014	Yes		Yes	Yes				
Lamontagne et al. 2018	Yes		Yes					
Le Page et al. 2013b	Yes		Yes					

Liu et al. 2017	No			Yes			Yes	
Lotze-Campen et al. 2013	Mixed			Yes			Yes	
Monier et al. 2018	Yes	Yes	Yes	Yes				Yes
Mouratiadou et al. 2016	Yes		Yes					Yes
Muratori et al. 2016	Yes		Yes				Yes	
Nelson et al. 2014	Mixed			Yes			Yes	
Newbold et al. 2015	Mixed							Yes
Obersteiner et al. 2016b	IAM-land						Yes	Yes
Parkinson et al. 2019	Yes		Yes					Yes
Patrizio et al. 2018	No		Yes					Yes
Pedercini et al. 2018	No						Yes	Yes
Pikaar et al. 2018	IAM-land		Yes					Yes
Popp et al. 2014a	Yes		Yes					
Popp et al. 2017	Yes		Yes				Yes	
Powers and Jetz 2019	No							Yes
Riahi et al. 2017c	Yes		Yes				Yes	
Ringler et al. 2016	Yes			Yes			Yes	Yes
Rogelj et al. 2018b	Yes		Yes					
Springmann et al. 2018a	No		Yes					Yes
Stehfest et al. 2019	Mixed							
Stevanovic et al. 2016	IAM-land			Yes				
Stevanović et al. 2017	IAM-land		Yes				Yes	
Tai et al. 2014	No						Yes	
Thornton et al. 2017	Yes	Yes	Yes	Yes			Yes	
UNCCD 2017	Mixed				Yes	Yes	Yes	Yes
van Meijl et al. 2018	Mixed		Yes	Yes			Yes	
van Vuuren et al. 2015b	Yes		Yes				Yes	Yes
van Vuuren et al. 2017	Yes		Yes					
van Vuuren et al. 2018b	Yes		Yes					
Weindl et al. 2015	IAM-land			Yes			Yes	
Weindl et al. 2017	IAM-land		Yes					
Wiebe et al. 2015	Mixed			Yes			Yes	
Wolff et al. 2018	No				Yes	Yes		Yes
Wu et al. 2019	Yes							
Yamagata et al. 2018	No					Yes		Yes

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Chapter 7: Risk management and decision making in relation to sustainable development

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Executive summary

Increases in global mean surface temperature are projected to result in continued permafrost degradation and coastal degradation (*high confidence*), increased wildfire, decreased crop yields in low latitudes, decreased food stability, decreased water availability, vegetation loss (*medium confidence)*, decreased access to food and increased soil erosion (*low confidence)*.There is *high agreement and high evidence* that increases in global mean temperature will result in continued increase in global vegetation loss, coastal degradation, as well as decreased crop yields in low latitudes, decreased food stability, decreased access to food and nutrition, and *medium confidence* in continued permafrost degradation and water scarcity in drylands. Impacts are already observed across all components (*high confidence*). Some processes may experience irreversible impacts at lower levels of warming than others. There are high risks from permafrost degradation, and wildfire, coastal degradation, stability of food systems at 1.5°C while high risks from soil erosion, vegetation loss and changes in nutrition only occur at higher temperature thresholds due to increased possibility for adaptation (*medium confidence).* {7.2.2.1, 7.2.2.2, 7.2.2.3; 7.2.2.4; 7.2.2.5; 7.2.2.6; 7.2.2.7; Figure 7.1}

These changes result in compound risks to food systems, human and ecosystem health, livelihoods, the viability of infrastructure, and the value of land *(high confidence*). The experience and dynamics of risk change over time as a result of both human and natural processes (*high confidence*). There is *high confidence* that climate and land changes pose increased risks at certain periods of life (i.e. to the very young and ageing populations) as well as sustained risk to those living in poverty. Response options may also increase risks. For example, domestic efforts to insulate populations from food price spikes associated with climatic stressors in the mid-2000s inadequately prevented food insecurity and poverty, and worsened poverty globally. {7.2.1, 7.2.2, 7.3, Table 7.1}

There is significant regional heterogeneity in risks: tropical regions, including Sub-Saharan Africa, Southeast Asia and Central and South America are particularly vulnerable to decreases in crop yield (*high confidence*). Yield of crops in higher latitudes may initially benefit from warming as well as from higher CO_2 concentrations. But temperate zones, including the Mediterranean, North Africa, the Gobi desert, Korea and western United States are susceptible to disruptions from increased drought frequency and intensity, dust storms and fires (*high confidence*). {7.2.2}

Risks related to land degradation, desertification and food security increase with temperature and can reverse development gains in some socio-economic development pathways (*high confidence)* . SSP1 reduces the vulnerability and exposure of human and natural systems and thus limits risks resulting from desertification, land degradation and food insecurity compared to SSP3 (*high confidence*). SSP1 is characterized by low population growth, reduced inequalities, land use regulation, low meat consumption, increased trade and few barriers to adaptation or mitigation. SSP3 has the opposite characteristics. Under SSP1, only a small fraction of the dryland population (around 3% at 3°C for the year 2050) will be exposed and vulnerable to water stress. However under SSP3, around 20% of dryland populations (for the year 2050) will be exposed and vulnerable to water stress by 1.5°C and 24% by 3°C. Similarly under SSP1, at 1.5°C, 2 million people are expected to be exposed and vulnerable to crop yield change. Over 20 million are exposed and vulnerable to crop yield change in SSP3, increasing to 854 million people at 3°C (*low confidence*). Livelihoods deteriorate as a result of these impacts, livelihood migration is accelerated, and strife and conflict is worsened (*medium confidence*). {Cross-Chapter Box 9 in Chapter 6, 7.2.2, 7.3.2, Table 7.1, Figure 7.2}

Land-based adaptation and mitigation responses pose risks associated with the effectiveness and potential adverse side-effects of measures chosen (*high confidence*). Adverse side-effects on food security, ecosystem services and water security increase with the scale of bioenergy and bioenergy with carbon capture and storage (BECCS) deployment. In a SSP1 future, bioenergy and BECCS deployment up to 6 Mkm^2 is compatible with sustainability constraints, whereas risks are already high in a SSP3 future for this scale of deployment. {7.2.3}

There is *high confidence* that policies addressing vicious cycles of poverty, land degradation and greenhouse gas emissions implemented in a holistic manner can achieve climate resilient sustainable development. Choice and implementation of policy instruments determine future climate and land pathways (*medium confidence)*. Sustainable development pathways (described in SSP1) supported by effective regulation of land use to reduce environmental trade-offs, reduced reliance on traditional biomass, low growth in consumption and limited meat diets, moderate international trade with connected regional markets, and effective GHG mitigation instruments) can result in lower food prices, fewer people affected by floods and other climatic disruptions, and increases in forested land (*high agreement, limited evidence*) (SSP1). A policy pathway with limited regulation of land use, low technology development, resource intensive consumption, constrained trade, and ineffective GHG mitigation instruments can result in food price increases, and significant loss of forest (*high agreement, limited evidence*) (SSP3). {3.7.5, 7.2.2, 7.3.4, 7.5.5, 7.5.6, Table 7.1, Cross-Chapter Box 12: Traditional Biomass, in this chapter}

Delaying deep mitigation in other sectors and shifting the burden to the land sector, increases the risk associated with adverse effects on food security and ecosystem services*(high confidence*). The consequences are an increased pressure on land with higher risk of mitigation failure and of temperature overshoot and a transfer of the burden of mitigation and unabated climate change to future generations. Prioritising early decarbonisation with minimal reliance on carbon dioxide removal (CDR) decreases the risk of mitigation failure (*high confidence*). {2.5, 6.2, 6.4, 7.2.1, 7.2.2,, 7.2.3, 7.5.6, 7.5.7, Cross-Chapter Box 9 in Chapter 6, 7.5.6}

Trade-offs can occur between using land for climate mitigation or sustainable development goal (SDG) 7 (affordable clean energy) with biodiversity, food, ground-water and riverine ecosystem services (*medium confidence*). There is *medium confidence* that trade-offs currently do not figure into climate policies and decision making. Small hydro power installations (especially in clusters) can impact downstream river ecological connectivity for fish (*high agreement, medium evidence*). Large scale solar farms and wind turbine installations can impact endangered species and disrupt habitat connectivity (*medium agreement, medium evidence*). Conversion of rivers for transportation can disrupt fisheries and endangered species (through dredging and traffic) (*medium agreement, low evidence*). {7.5.6}

The full mitigation potential assessed in this report will only be realised if agricultural emissions are included in mainstream climate policy (*high agreement, high evidence*) . Carbon markets are theoretically more cost-effective than taxation but challenging to implement in the land-sector (*high confidence)* Carbon pricing (through carbon markets or carbon taxes) has the potential to be an effective mechanism to reduce GHG emissions, although it remains relatively untested in agriculture and food systems. Equity considerations can be balanced by a mix of both market and non-market mechanisms (*medium evidence, medium agreement*). Emissions leakage could be reduced by multi-lateral action (*high agreement, medium evidence*). {7.4.6, 7.5.5, 7.5.6, Cross Chapter Box 9 in Chapter 6}

A suite of coherent climate and land policies advances the goal of the Paris Agreement and the land-related SDG targets on poverty, hunger, health, sustainable cities and communities, responsible consumption and production, and life on land. There is *high confidence* that acting early will avert or minimise risks, reduce losses and generate returns on investment . The economic costs of action on sustainable land management, mitigation, and adaptation are less than the consequences of inaction for humans and ecosystems (*medium confidence*). Policy portfolios that make ecological restoration more attractive, people more resilient - expanding financial inclusion, flexible carbon credits, disaster risk and health insurance, social protection and adaptive safety nets, contingent finance and reserve funds, and universal access to early warning systems – could save USD 100 billion a year, if implemented globally. {7.3.1, 7.4.7, 7.4.8, 7.5.6, Cross-chapter box 10: Economic Dimensions, in this chapter}

Coordination of policy instruments across scales, levels, and sectors advances co-benefits, manages land and climate risks, advances food security, and addresses equity concerns (*medium confidence*). Flood resilience policies are mutually reinforcing and include flood zone mapping, financial incentives to move, and building restrictions, and insurance. Sustainability certification, technology transfer, land use standards and secure land tenure schemes, integrated with early action and preparedness, advance response options. Sustainable land management improves with investment in agricultural research, environmental farm practices, agri-environmental payments, financial support for sustainable agricultural water infrastructure (including dugouts), agriculture emission trading, and elimination of agricultural subsidies (*medium confidence*). Drought resilience policies (including drought preparedness planning, early warning and monitoring, improving water use efficiency), synergistically improve agricultural producer livelihoods and foster sustainable land management. {3.7.5, Cross-Chapter Box 5 in Chapter 3, 7.4.3, 7.4.6, 7.5.6, 7.4.8, , 7.5.6, 7.6.3}

Technology transfer in land use sectors offers new opportunities for adaptation, mitigation, international cooperation, R&D collaboration, and local engagement (*medium confidence*). International cooperation to modernise the traditional biomass sector will free up both land and labour for more productive uses. Technology transfer can assist the measurement and accounting of emission reductions by developing countries. {7.4.4, 7.4.6}

Measuring progress towards goals is important in decision-making and adaptive governance to create common understanding and advance policy effectiveness (*high agreement, medium evidence*). Measurable indicators, selected with the participation of people and supporting data collection, are useful for climate policy development and decision-making. Indicators include the SDGs, nationally determined contributions (NDCs), land degradation neutrality (LDN) core indicators, carbon stock measurement, measurement and monitoring for REDD+, metrics for measuring biodiversity and ecosystem services, and governance capacity. {7.5.5, 7.5.7, 7.6.4, 7.6.6}

The complex spatial, cultural and temporal dynamics of risk and uncertainty in relation to land and climate interactions and food security, require a flexible, adaptive, iterative approach to assessing risks, revising decisions and policy instruments (*high confidence*). Adaptive, iterative decision making moves beyond standard economic appraisal techniques to new methods such as dynamic adaptation pathways with risks identified by trigger points through indicators. Scenarios can provide valuable information at all planning stages in relation to land, climate and food; adaptive management addresses uncertainty in scenario planning with pathway choices made and reassessed to respond to new information and data as it becomes available. {3.7.5, 7.4.4, 7.5.2, 7.5.3, 7.5.4, 7.5.7, 7.6.1, 7.6.3}

Indigenous and local knowledge (ILK) can play a key role in understanding climate processes and impacts, adaptation to climate change, sustainable land management across different ecosystems, and enhancement of food security (*high confidence*). ILK is context-specific, collective, informally transmitted, and multi-functional, and can encompass factual information about the environment and guidance on management of resources and related rights and social behaviour. ILK can be used in decision-making at various scales and levels, and exchange of experiences with adaptation and mitigation that include ILK is both a requirement and an entry strategy for participatory climate communication and action. Opportunities exist for integration of ILK with scientific knowledge. {7.4.1, 7.4.5, 7.4.6, 7.6.4, Cross-Chapter Box 13: in this chapter}

Participation of people in land and climate decision making and policy formation allows for transparent effective solutions and the implementation of response options that advance synergies, reduce trade-offs in sustainable land management (*high confidence*), and overcomes barriers to adaptation and mitigation (*high confidence*). Improvements to sustainable land management are achieved by: (1) engaging people in citizen science by mediating and facilitating landscape conservation planning, policy choice, and early warning systems (*medium confidence*); (2) involving people in identifying problems (including species decline, habitat loss, land use change in agriculture, food production and forestry), selection of indicators, collection of climate data, land modelling, agricultural innovation opportunities. When social learning is combined with collective action, transformative change can occur addressing tenure issues and changing land use practices (*medium confidence*). Meaningful participation overcomes barriers by opening up policy and science surrounding climate and land decisions to inclusive discussion that promotes alternatives. {3.7.5, 7.4.1, 7.4.9; 7.5.1, 7.5.4, 7.5.5, 7.5.7, 7.6.4, 7.6.6}

Empowering women can bolster synergies among household food security and sustainable land management (*high confidence*). This can be achieved with policy instruments that account for gender differences. The overwhelming presence of women in many land based activities including agriculture provides opportunities to mainstream gender policies, overcome gender barriers, enhance gender equality, and increase sustainable land management and food security *(high confidence*). Policies that address barriers include gender qualifying criteria and gender appropriate delivery, including access to financing, information, technology, government transfers, training, and extension may be built into existing women's programs, structures (civil society groups) including collective micro enterprise (*medium confidence*) . {Cross-Chapter Box 11 in this chapter}

The significant social and political changes required for sustainable land use, reductions in demand and land-based mitigation efforts associated with climate stabilisation require a wide range of governance mechanisms. The expansion and diversification of land use and biomass systems and markets requires hybrid governance: public-private partnerships, transnational, polycentric, and state governance to insure opportunities are maximised, trade-offs are managed equitably and negative impacts are minimised *(medium confidence).* {7.4.6, 7.6.2, 7.6.3, Cross-Chapter Box 7 in Chapter 6}

Land tenure systems have implications for both adaptation and mitigation, which need to be understood within specific socio-economic and legal contexts, and may themselves be impacted by climate change and climate action (*limited evidence, high agreement*). Land policy (in a diversity of forms beyond focus on freehold title) can provide routes to land security and facilitate or constrain climate action, across cropping, rangeland, forest, fresh-water ecosystems and other systems. Large-scale land acquisitions are an important context for the relations between tenure security and climate change, but their scale, nature and implications are imperfectly understood. There is *medium confidence* that land titling and recognition programs, particularly those that authorize and respect indigenous and communal tenure, can lead to improved management of forests, including for carbon storage. Strong public coordination (government and public administration) can integrate land policy with national policies on adaptation and reduce sensitivities to climate change. {7.6.2; 7.6.3; 7.6.4, 7.6.5}

Significant gaps in knowledge exist when it comes to understanding the effectiveness of policy instruments and institutions related to land use management, forestry, agriculture and bioenergy. Interdisciplinary research is needed on the impacts of policies and measures in land sectors. Knowledge gaps are due in part to the highly contextual and local nature of land and climate measures and the long time periods needed to evaluate land use change in its socio-economic frame, as compared to technological investments in energy or industry that are somewhat more comparable. Significant investment is needed in monitoring, evaluation and assessment of policy impacts across different sectors and levels. {7.7}

7.1. Introduction and Relation to Other Chapters

Land is integral to human habitation and livelihoods, providing food and resources, and also serves as a source of identity and cultural meaning. However, the combined impacts of climate change, desertification, land degradation and food insecurity pose obstacles to resilient development and the achievement of the Sustainable Development Goals (SDGs). This chapter reviews and assesses literature on risk and uncertainty surrounding land and climate change, policy instruments and decision-making addressing those risks and uncertainty, and governance practices that advance response options with co-benefits identified in Chapter 6, lessen the socio-economic impacts of climate change and reduce trade-offs, and advance sustainable land management.

7.1.1. Findings of Previous IPCC Assessments and Reports

This chapter builds on earlier assessments contained in several chapters of the IPCC Fifth Assessment Report (the contributions of both Working Groups II and III), the IPCC Special Report on Managing the Risks of Extreme Events and Disasters to Advance Climate Change Adaptation (SREX)(IPCC 2012), and the IPCC Special Report on Global Warming of 1.5°C (SR15). (IPCC 2018a) The findings most relevant to decision-making on and governance of responses to land-climate challenges are set out in Box 7.1.

Box 7.1 Relevant Findings of Recent IPCC Reports

Climate change and sustainable development pathways

"Climate change poses a moderate threat to current sustainable development and a severe threat to future sustainable development" (Denton et al. 2014; Fleurbaey et al. 2014).

Significant transformations may be required for climate-resilient pathways (Denton et al. 2014; Jones et al. 2014).

The design of climate policy is influenced by: (1) differing ways that individuals and organisations perceive risks and uncertainties; (2) the consideration of a diverse array of risks and uncertainties as well as human and social responses which may be difficult to measure, are of low probability but which would have a significant impact if they occurred (Kunreuther et al. 2014; Fleurbaey et al. 2014; Kolstad et al. 2014).

Building climate resilient pathways requires iterative, continually evolving and complementary processes at all levels of government (Denton et al. 2014; Kunreuther et al. 2014; Kolstad et al. 2014; Somanthan et al. 2014; Lavell et al. 2012).

Important aspects of climate resilient policies include local level institutions, decentralisation, participatory governance, iterative learning, integration of local knowledge, and reduction of inequality (Dasgupta et al. 2014; Lavell et al. 2012; Cutter et al. 2012b; O' Brien et al. 2012; Roy et al. 2018).

Climate action and sustainable development are linked: adaptation has co-benefts for sustainable development while "sustainable development supports, and often enables, the fundamental societal and systems transitions and transformations that help limit global warming" (IPCC 2018b). Redistributive policies that shield the poor and vulnerable can resolve trade-offs between mitigation objectives and the hunger, poverty and energy access SDGs.

Land and rural livelihoods

Policies and institutions relating to land, including land tenure, can contribute to the vulnerability of rural people, and constrain adaptation. Climate policies, such as encouraging cultivation of biofuels, or payments under REDD+, will have significant secondary impacts, both positive and negative, in some rural areas (Dasgupta et al. 2014).

"Sustainable land management is an effective disaster risk reduction tool"(Cutter et al. 2012a).

Risk and risk management

A variety of emergent risks not previously assessed or recognised, can be identified by taking into account: a) the "interactions of climate change impacts on one sector with changes in exposure and vulnerability, as well as adaptation and mitigation actions", and; b) "indirect, trans-boundary, and long-distance impacts of climate change" including price spikes, migration, conflict and the unforeseen impacts of mitigation measures (Oppenheimer et al. 2014)

"Under any plausible scenario for mitigation and adaptation, some degree of risk from residual damages is unavoidable" (Oppenheimer et al. 2014).

Decision-making

"Risk management provides a useful framework for most climate change decision-making. Iterative risk management is most suitable in situations characterised by large uncertainties, long time frames, the potential for learning over time, and the influence of both climate as well as other socioeconomic and biophysical changes" (Jones et al. 2014).

"Decision support is situated at the intersection of data provision, expert knowledge, and human decision making at a range of scales from the individual to the organisation and institution" (Jones et al. 2014).

"Scenarios are a key tool for addressing uncertainty", either through problem exploration or solution exploration (Jones et al. 2014).

Governance

There is no single approach to adaptation planning and both top-down and bottom-up approaches are widely recognised. "Institutional dimensions in adaptation governance play a key role in promoting the transition from planning to implementation of adaptation" (Mimura et al. 2014). Adaptation is also essential at all scales, including adaptation by local governments, businesses, communities, and individuals (Denton et al. 2014).

"Strengthened multi-level governance, institutional capacity, policy instruments, technological innovation and transfer and mobilisation of finance, and changes in human behaviour and lifestyles are enabling conditions that enhance the feasibility of mitigation and adaptation options for 1.5°C – consistent systems transitions" (IPCC 2018b).

Governance is key for vulnerability and exposure represented by institutionalised rule systems and habitualised behaviour and norms that govern society and guide actors and , "it is essential to improve knowledge on how to promote adaptive governance within the framework of risk assessment and risk management" (Cardona 2012).

7.1.2. Treatment of Key Terms in the Chapter

While the term risk continues to be subject to a growing number of definitions in different disciplines and sectors, this chapter takes as a starting point the definition used in the IPCC Special Report on Global Warming of 1.5°C (SR15) (IPCC 2018a), which reflects definitions used by both Working Group II and Working Group III in the Fifth Assessment Report (AR5): "The potential for adverse consequences where something of value is at stake and where the occurrence and degree of an outcome is uncertain" (Allwood et al. 2014; Oppenheimer et al. 2014). The SR15 definition further specifies: "In the context of the assessment of climate impacts, the term risk is often used to refer to the potential for adverse consequences of a climate-related hazard, or of adaptation or mitigation responses to such a hazard, on lives, livelihoods, health and wellbeing, ecosystems and species, economic, social and cultural assets, services (including ecosystem services), and infrastructure". In SR15, as in the IPCC SREX and AR5 WGII, risk is conceptualised as resulting from the interaction of vulnerability (of the affected system), its exposure over time (to a hazard), as well as the (climate-related) impact and the likelihood of its occurrence (AR5 2014; IPCC 2018a, 2012). In the context of SRCCL, risk must also be seen as including risks to the implementation of responses to land-climate challenges from economic, political and governance factors. Climate and land risks must be seen in relation to human values and objectives (Denton et al. 2014). Risk is closely associated with concepts of vulnerability and resilience, which are themselves subject to differing definitions across different knowledge communities.

Risks examined in this chapter arise from more than one of the major land-climate-society challenges (desertification, land degradation, and food insecurity), or partly stem from mitigation or adaptation actions, or cascade across different sectors or geographical locations. They could thus be seen as examples of *emergent risks* (Oppenheimer et al. 2014, p. 1052): "aris[ing] from the interaction of phenomena in a complex system". Stranded assets in the coal sector due to proliferation of renewable energy and government response could be examples of emergent risks (Saluja, N and Singh 2018; Marcacci 2018). Additionally, the absence of an explicit goal for conserving fresh-water ecosystems and ecosystem services in SDGs (in contrast to a goal (Life Under Water) that is exclusively for marine biodiversity) is related to its trade-offs with energy and irrigation goals thus posing a substantive risk (Nilsson et al. 2016b; Vörösmarty et al. 2010).

Governance is not previously well defined in IPCC reports, but is used here to include all of the processes, structures, rules and traditions that govern, which may be undertaken by actors including governments, markets, organisations, or families (Bevir 2011), with particular reference to the multitude of actors operating in respect of land and climate interactions. Such definitions of governance allow for it to be decoupled from the more familiar concept of government and studied in the context of complex human-environment relations and environmental and resource regimes (Young 2017a). Governance involves the interactions among formal and informal institutions through which people articulate their interests, exercise their legal rights, meet their legal obligations, and mediate their differences (UNDP 1997).

7.1.3. Roadmap to the chapter

This chapter firstly discusses risks and their drivers, at various scales, in relation to land-climate challenges, including risks associated with responses to climate change (Section 7.2). The consequences of the principal risks in economic and human terms, and associated concepts such as tipping points and windows of opportunity for response are then described (Section 7.3). Policy responses at different scales to different land-climate risks, and barriers to implementation, are described in Section 7.4, followed by assessment of approaches to decision-making on land-climate challenges (Section 7.5), and questions of the governance of the land-climate interface (Section 7.6). Key uncertainties and knowledge gaps are identified (Section 7.7).

7.2. Climate-related risks for land-based human systems and ecosystems

This section examines risks that climate change pose to selected land-based human systems and ecosystems, and then further explores how social and economic choices, as well as responses to climate change, will exacerbate or lessen risks. Risk is the potential for adverse consequences for human or ecological systems, recognising the diversity of values and objectives associated with such systems. The interacting processes of climate change, land change, and unprecedented social and technological change, pose significant risk to climate resilient sustainable development. The pace, intensity, and scale of these sizeable risks affect the central issues in sustainable development: access to ecosystem services and resources essential to sustain people in given locations, how and where people live and work, and the means to safeguard human wellbeing against disruptions (Warner et al. 2019). In the context of climate change, adverse consequences can arise from the potential *impacts of* climate change as well as human *responses to* climate change. Relevant adverse consequences include those on lives, livelihoods, health and wellbeing, economic, social and cultural assets and investments, infrastructure, services (including ecosystem services), ecosystems and species (see Glossary). Risks result from dynamic interactions between climate-related hazards with the exposure and vulnerability of the affected human or ecological system to the hazards. Hazards, exposure and vulnerability may change over time and space as a result of socio-economic changes and human decision-making (*risk management*). Numerous uncertainties exist in the scientific understanding of risk (See Chapter 1.2.2).

7.2.1. Assessing Risk

This chapter applies and further improves methods used in previous IPCC reports including AR5 and the Special Report on Global Warming of 1.5° (SR15) to assess risks. Evidence is drawn from published studies, which include observations of impacts from human-induced climate change and model projections for future climate change. Such projections are based on IAMs, ESMs, regional climate models and global or regional impact models examining the impact of climate change on various indicators (see Cross-Chapter Box 1: Scenarios, in Chapter 1). Results of laboratory and field experiments that examine impacts of specific changes were also included in the review. Risks under differed future socio-economic conditions were assessed using recent publications based on Shared Socio-economic pathways (SSPs). SSPs provide storylines about future socio-economic development and can be combined with RCPs (Riahi et al. 2017)(see Cross-Chapter Box 9: Illustrative climate and land pathways, in Chapter 6). Risk arising from land-based mitigation and adaptation choices is assessed using studies examining the adverse side-effects of such responses (7.2.3).

Burning embers figures introduced in the IPCC Third Assessment Report through to the Fifth Assessment Report, and the SR15, were developed for this report to illustrate risks at different temperature thresholds. Key components involved in desertification, land degradation and food security were identified based on discussions with authors in Chapter 3 –5. The final list of burning embers in Figure 7.1 is not intended to be fully comprehensive, but represents processes for which sufficient literature exists to make expert judgements. Literature used in the burning embers assessment is summarised in table(s) in supplementary material. Following an approach articulated in O'Neill et al. (2017), expert judgements were made to assess thresholds of risk (O'Neill et al. 2017a). To further strengthen replicability of the method, a predefined protocol based on a modified Delphi process was followed (Mukherjee et al. 2015). This included two separate anonymous rating rounds, feedback in between rounds and a group discussion to achieve consensus.

Burning embers provide ranges of a given variable (typically global mean near-surface air temperature) for which risks transitions from one risk category to the next. Four categories are considered: undetectable, moderate, high and very high. Moderate risk indicates that impacts are detectable and attributable to climate-related factors. High risk indicates widespread impacts on larger number or proportion of population/ area but with the potential to adapt or recover. Very high risk indicates severe and possibly irreversible impacts with limited ability of societies and ecosystems to adapt to them. Transitions between risk categories were assigned confidence levels based on the amount, and quality, of academic literature supporting judgements: L= Low, M = Medium, and H = High. Further details of the procedure is provided in supplementary material.

7.2.2. Risks to land systems arising from climate change

At current levels of global mean surface temperature (GMST) increase, impacts are already detectable across numerous land-related systems (*high confidence*) (see chapters 2, 3, 4, 6). There is *high confidence* that unabated future climate change will result in continued changes to processes involved in desertification, land degradation and food security, including: water scarcity in drylands, soil erosion, coastal degradation, vegetation loss, fire, permafrost thaw as well as access, stability, utilisation and physical availability of food (Figure 7.1). These changes will increase risks to food systems, the health of humans and ecosystems, livelihoods, the value of land, infrastructure and communities (7.3). Details of the risks, and their transitions, are described in the following subsections.

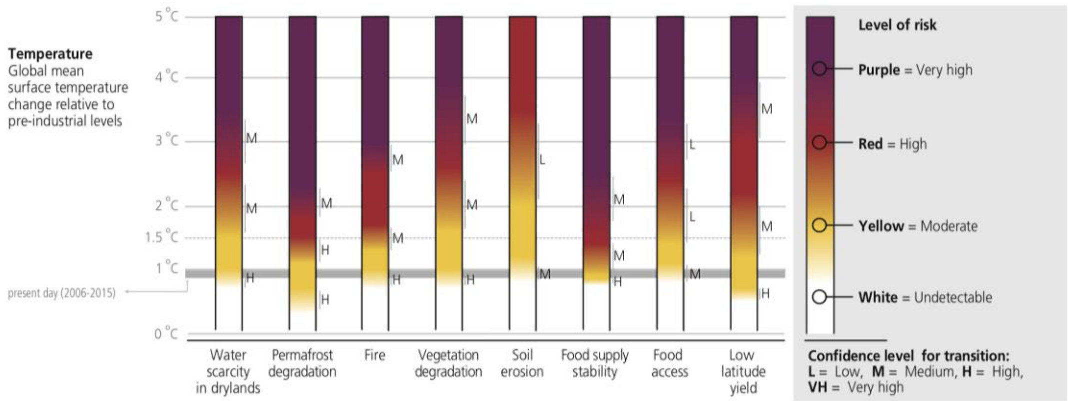

Figure 7.1: Risks to selected land system elements as a function of global mean surface temperature increase since pre-industrial times. Impacts on human and ecological systems include: 1) economic loss and declines in livelihoods and ecosystem services from water scarcity in drylands, 2) damage to natural and built environment from permafrost thaw related ground instability, 3) damage to infrastructure, altered land cover, accelerated erosion and increased air pollution from fires, 4) vegetation loss and shifts in vegetation structure, 5) economic loss and declines in livelihoods and ecosystem services from reduced land productivity due to soil erosion, 6) increased disruption of food supply (stability), 7) increased disruption of food access and 8) changes to crop yield and food availability in low-latitude regions. Risks are global (3,4,5,6,7) and specific to certain regions (1,2,8). Selected components are illustrative and not intended to be fully comprehensive of factors influencing food security, land degradation and desertification. The supporting literature is provided in Supplementary Material.

7.2.2.1. Crop yield in low latitudes

There is *high confidence* that climate change has resulted in decreases in yield (of wheat, rice, maize, soy) and reduced food availability in low-latitude regions (IPCC, 2018, 5.2.2). Countries in low-latitude regions are particularly vulnerable because the livelihoods of high proportions of the population are dependent on agricultural production. Even moderate temperature increases (1°C to 2 °C) have negative yield impacts for major cereals, because the climate of many tropical agricultural regions is already quite close to the high temperature thresholds for suitable production of these cereal (Rosenzweig et al. 2014). Thus, by 1.5°C GMT, or between approximately 1.6°C and approximately 2.6°C of local warming, risks to yields may already transition to high in West Africa, Southeast Asia and Central and South America (Faye et al. 2018) (*medium confidence*). For further information see 5.3.2.1. By contrast, higher latitudes may initially benefit from warming as well as well higher CO_2 concentrations (IPCC 2018a). Wheat yield losses are expected to be lower for the United States (−5.5 ± 4.4% per degree Celsius) and France (−6.0 ± 4.2% per degree Celsius) compared to India (−9.1 ± 5.4% per degree Celsius) (Zhao et al. 2017). Very high risks to low latitude yields may occur between 3°C and 4°C (*medium confidence).* At these temperatures, catastrophic reductions in crop yields may occur, of up to 60% in low latitudes (Rosenzweig et al. 2014)(5.2.2, 5.2.3). Some studies report significant population displacement from the tropics related to systemic livelihood disruption in agriculture systems (Tittonell 2014; Montaña et al. 2016; Huber-Sannwald et al. 2012; Wise et al. 2016; Tanner et al. 2015; Mohapatra 2013). However, at higher temperatures of warming, all regions of the world face risks of declining yields as a result of extreme weather events and reduced heat tolerance of maize, rice, wheat and soy (Zhao et al. 2017; IPCC 2018a).

7.2.2.2. Stability of and access to food supplies

Stability of food supply is expected to decrease as the magnitude and frequency of extreme events increase, disrupting food chains in all areas of the world (Wheeler and Von Braun 2013; Coates 2013; Puma et al. 2015; Deryng et al. 2014; Harvey et al. 2014b; Iizumi et al. 2013; Seaman et al. 2014)*(medium evidence, high agreement*)(5.3.2, 5.3.3, 5.6.2, 5.7.1). While international trade in food is assumed to be a key response for alleviating hunger, historical data and economic models suggest that international trade does not adequately redistribute food globally to offset yield declines or other food shortages when weather extremes reduce crop yields (Schmitz et al. 2012; Chatzopoulos et al. 2019; Marchand et al. 2016; Gilbert 2010; Wellesley et al. 2017) (*medium confidence*). When droughts, heat waves, floods or other extremes destroy crops, evidence has shown key producing countries have constrained exports contributing to price spikes and social tension in importing countries which reduces access to food (von Uexkull et al. 2016; Gleick 2014; Maystadt and Ecker 2014; Kelley et al. 2015; Church et al. 2017; Götz et al. 2013; Puma et al. 2015; Willenbockel 2012; Headey 2011; Distefano et al. 2018; Brooks 2014)(*medium evidence*, *medium agreement*). There is little understanding of how food system shocks cascade through a modern interconnected economy. Reliance on global markets may reduce some risks, but the on-going globalisation of food trade networks exposes the world food system to new impacts that have not been seen in the past (5.1.2, 5.2.1, 5.5.2.5, 5.6.5. 5.7.1). The global food system is vulnerable to systemic disruptions and increasingly interconnected inter-country food dependencies and changes in frequency and severity of extreme weather events may complicate future responses(Puma et al. 2015; Jones and Hiller 2017).

Impacts of climate change are already detectable on food supply and access as price and trade reactions have occurred in response to heat waves, droughts and other extreme events (Noble et al. 2014; O'Neill et al. 2017b)(*high evidence, high agreement*). The impact of climate change on food stability is underexplored (Schleussner et al. 2016; James et al. 2017). However, some literature assesses that by about 2035, daily maximum temperatures will exceed the 90th percentile of historical (1961–1990) temperatures on 25–30% of days (O'Neill et al. 2017b)(ref 35, Figs 11–17) with negative shocks to food stability and world food prices. O'Neill et al. (2017b) remark that in the future, return periods for precipitation events globally (land only) will reduce from one-in-20-year (historical) to about once-in-14-year or less by 2046–2065 in many areas of the world. Domestic efforts to insulate populations from food price spikes associated with climatic stressors in the mid-2000s have been shown to inadequately shield from poverty, and worsen poverty globally (Diffenbaugh et al. 2012; Meyfroidt et al. 2013; Hertel et al. 2010). The transition to high risk is estimated to occur around 1.4°C, possibly by 2035, due to changes in temperature and heavy precipitation events (*medium confidence*) (O'Neill et al. 2017b; Fritsche et al. 2017a; Harvey et al. 2014b). Very high risk may occur by 2.4°C (*medium confidence*) and 4°C of warming is considered catastrophic (IPCC 2018c; Noble et al. 2014) for food stability and access because a combination of extreme events, compounding political and social factors, and shocks to crop yields can heavily constrain options to ensure food security in import-reliant countries.

7.2.2.3. Soil Erosion

Soil erosion increases risks of economic loss and declines in livelihoods due to reduced land productivity. In the EU, on-site costs of soil erosion by wind has been reported at an average of 55 USD per hectare annually, but up to USD 450 per hectare for sugar beet and oilseed rape (Middleton et al. 2017)). Farmers in the Dapo watershed in Ethiopia lose about USD 220 per hectare of maize due to loss of nitrogen through soil erosion (Erkossa et al. 2015). Soil erosion not only increases crop loss but has been shown to have negative household feeding, with older farmers most vulnerable to losses from erosion (Ighodaro et al. 2016). Erosion also results in increased risks to human health, through air pollution from aerosols (Middleton et al. 2017), and brings risks of reduced ecosystem services including supporting services related to soil formation.

At current levels of warming, changes in erosion are already detected in many regions. Attribution to climate change is challenging as there are other powerful drivers of erosion (e.g., land use), limited global-scale studies (Li and Fang 2016a; Vanmaercke et al. 2016a) and the absence of formal detection and attribution studies (4.2.3). However, studies have found an increase in short-duration and intensity precipitation, due to anthropogenic climate change, which is a causative factor for soil erosion (Lenderink and van Meijgaard 2008; Li and Fang 2016b). High risks of erosion may occur between 2° and 3.5° (*low confidence*) as continued increases in intense precipitation is projected at these temperature thresholds (Fischer and Knutti 2015) in many regions. Warming also reduces soil organic matter, diminishing resistance against erosion. There is *low confidence* concerning the temperature threshold at which risks become very high due to large regional differences and limited global-scale studies (Li and Fang 2016b; Vanmaercke et al. 2016b) (4.4).

7.2.2.4. Dryland water scarcity

Water scarcity in drylands contributes to changes in desertification and hazards such as dust storms, increasing risks of economic loss, declines in livelihoods of communities and negative health effects (*high confidence*) (3.1.3). Further information specific to costs and impacts of water scarcity and droughts is detailed in Cross-Chapter Box 5: Case study on policy response to drought, in Chapter 3.

The IPCC AR5 report and the SR15 concluded that there is *low confidence* in the direction of drought trends since 1950 at the global scale. While these reports did not assess water scarcity with a specific focus on drylands, they indicated that there is *high confidence* in observed drought increases in some regions of the world, including in the Mediterranean and West Africa (IPCC AR5) and that there is *medium confidence* that anthropogenic climate change has contributed to increased drying in the Mediterranean region (including southern Europe, northern Africa and the Near East) and that this tendency will continue to increase under higher levels of global warming (IPCC 2018d). Some parts of the drylands have experienced decreasing precipitation over recent decades (IPCC AR5; Chapter 3, 3.2), consistent with the fact that climate change is implicated in desertification trends in some regions (3.2.2). Dust storms, linked to changes in precipitation and vegetation, appear to be occurring with greater frequency in some deserts and their margins (Goudie 2014) (3.3.1). There is therefore *high confidence* that the transition from undetectable to moderate risk associated with water scarcity in drylands occurred in recent decades in the range 0.7°C to 1°C (Fig. 7.1).

Between 1.5°C and 2.5°C, the risk level is expected to increase from moderate to high (*medium confidence*). Globally, at 2°C an additional 8% of the world population (of population in 2000) will be exposed to new of aggravated water scarcity (IPCC 2018d). However, at 2°C, the annual warming over drylands will reach 3.2°C –4.0°C, implying about 44% more warming over drylands than humid lands (Huang et al. 2017), thus potentially aggravating water scarcity issues through increased evaporative demand. (Byers et al. 2018a) estimate that 3–22% of the drylands population (range depending on socio-economic conditions) will be exposed and vulnerable to water stress. The Mediterranean, North Africa and the Levant will be particularly vulnerable to water shortages and expansion of desert terrain and vegetation is predicted to occur in the Mediterranean biome, an unparalleled change in the last 10,000 years (*medium confidence*) (IPCC 2018d). At 2.5°C–3.5°C risks are expected to become very high with migration from some drylands resulting as the only adaptation option (*medium confidence*). Scarcity of water for irrigation is expected to increase, in particular in Mediterranean regions, with limited possibilities for adaptation (Haddeland et al. 2014).

7.2.2.5. Vegetation degradation

There are clear links between climate change and vegetation cover changes, tree mortality, forest diseases, insect outbreaks, forest fires, forest productivity and net ecosystem biome production (Allen et al. 2010; Bentz et al. 2010; Anderegg et al. 2013; Hember et al. 2017; Song et al. 2018; Sturrock et al. 2011). Forest dieback, often a result of drought and temperature changes, not only produces risks to forest ecosystems but also to people with livelihoods dependent on forests. A 50 year study of temperate forest, dominated by beech (*Fagus sylvatica* L.), documented a 33% decline in basel area and 70% decline in juvenile tree species, possibly as a result of interacting pressures of drought, overgrazing and pathogens (Martin et al. 2015). There is *high confidence* that such dieback impacts ecosystem properties and services including soil microbial community structure (Gazol et al. 2018). Forest managers and users have reported negative emotional impacts from forest dieback such as pessimism about losses, hopelessness, and fear (Oakes et al. 2016). Practices and policies such as forest classification systems, projection of growth, yield and models for timber supply are already being affected by climate change (Sturrock et al. 2011).

While risks to ecosystems and livelihoods from vegetation degradation are already detectable at current levels of GMT increase, risks are expected to reach high levels between 1.6°C and 2.6°C (*medium confidence*). Significant uncertainty exists due to countervailing factors: CO_2 fertilisation encourages forest expansion but increased drought, insect outbreaks, and fires result in dieback (Bonan 2008; Lindner et al. 2010). The combined effects of temperature and precipitation change, with CO_2 fertilisation, make future risks to forests very location specific. It is challenging therefore to make global estimates. However, even locally specific studies make clear that very high risks occur between 2.6°C and 4°C (*medium confidence*). Australian tropical rainforests experience significant loss of biodiversity with 3.5°C increase. There are no areas with greater than 30 species and all endemics disappear from low and mid-elevation regions (Williams et al. 2003). Mountain ecosystems are particularly vulnerable (Loarie et al. 2009).

7.2.2.6. Fire damage

Increasing fires result in heightened risks to infrastructure, accelerated erosion, altered hydrology, increased air pollution, and negative mental health impacts. Fire not only destroys property but induces changes in underlying site conditions (ground cover, soil water repellency, aggregate stability and surface roughness) which amplifies runoff and erosion, increasing future risks to property and human lives during extreme rainfall events (Pierson and Williams 2016). Dust and ash from fires can impact air quality in a wide area. For example, a dust plume from a fire in Idaho, USA, in September 2010 was visible in MODIS satellite imagery and extended at least 100 km downwind of the source area (Wagenbrenner et al. 2013). Individuals can suffer from property damage or direct injury, psychological trauma, depression, post traumatic stress disorder and have reported negative impacts to well being from loss of connection to landscape (Paveglio et al. 2016; Sharples et al. 2016a). Costs of large wildfires in the United States can exceed USD 20 million a day (Pierson et al. 2011) and has been estimated at USD8.5 billion per year in Australia (Sharples et al. 2016b). Globally, human exposure to fire will increase due to projected population growth in fire-prone regions (Knorr et al. 2016a).

It is not clear how quickly, or even if, systems can recover from fires. Longevity of effects may differ depending on cover recruitment rate and soil conditions, recovering in one to two seasons or over ten growing seasons (Pierson et al. 2011). In Russia, one third of forest area affected by fires turned into unproductive areas where natural reforestation is not possible within 2–3 life cycles of major forest forming species (i.e., 300–600 years) (Shvidenko et al. 2012).

Risks under current warming levels are already moderate as anthropogenic climate change has caused significant increases in fire area (*high confidence)* due to availability of detection and attribution studies) (Cross-Chapter Box 3: Fire and climate change, in Chapter 2). This has been detected and attributed regionally, notably in Western US (Abatzoglou and Williams 2016; Westerling et al. 2006; Dennison et al. 2014), Indonesia (Fernandes et al. 2017) and other regions (Jolly et al. 2015). Regional increases have been observed despite a global-average declining trend induced by human fire suppression strategies especially in savannas (Yang et al. 2014a; Andela et al. 2017).

High risks of fire may occur between 1.3°C and 1.7°C (*medium confidence*). Studies note heightened risks as “fire weather” and land prone to fire increase above 1.5°C (Abatzoglou et al. 2019a), with *medium confidence* in this transition, due to complex interplay between (i) global warming (ii) CO_2-fertilisation, and (iii) human/economic factors affecting fire risk. Canada, the USA and Mediterranean may be particularly vulnerable as the combination of increased fuel due to CO_2 fertilisation, and weather conditions conducive to fire increase risks to people and property. Some studies show substantial effects at 3°C (Knorr et al. 2016b; Abatzoglou et al. 2019b), indicating a transition to very high risks (*medium confidence*). At high warming levels, climate change may become the primary driver of fire risk in the extratropics (Knorr et al. 2016b; Abatzoglou et al. 2019b; Yang et al. 2014b). Pyroconvection activity may increase, in areas such as southeast Australia (Dowdy and Pepler 2018), posing major challenges to adaptation.

7.2.2.7. Permafrost

There is a risk of damage to natural and built environment from permafrost thaw related ground instability. Residential, transportation, and industrial infrastructure in the pan-Arctic permafrost area are particularly at risk (Hjort et al., 2018). High risks already exist at low temperatures (*high confidence*). Approximately, 21–37% of Arctic permafrost is projected to thaw under 1.5°C of warming (Hoegh-Guldberg et al., 2018). This increases to very high risk around 2°C (between 1.8 and 2.3°C) of temperature increase since pre-industrial times (*medium confidence*) with 35–47% of the Arctic permafrost thawing (Hoegh-Guldberg et al., 2018). If climate stabilised at 2°C, still approximately 40% of permafrost area would be lost (Chadburn et al., 2017), leading to nearly four million people and 70% of current infrastructure in the pan-Arctic permafrost area exposed to permafrost thaw and high hazard (Hjort et al., 2018). Indeed between 2°C and 3°C a collapse of permafrost may occur with a drastic biome shift from tundra to boreal forest (Drijfhout et al. 2015; SR15). There is mixed evidence of a tipping point in permafrost collapse, leading to enhanced greenhouse gas emission and particularly methane, between 2°C and 3°C (Hoegh-Guldberg et al., 2018).

7.2.2.8. Risks of desertification, land degradation and food insecurity under different Future Development Pathways

Socio-economic developments and policy choices that govern land-climate interactions are an important driver of risk along with climate change (*very high confidence*). Risks under two different Shared Socio-economic Pathways (SSPs) were assessed using emerging literature. SSP1 is characterised by low population growth, reduced inequalities, land-use regulation, low meat consumption, and moderate trade (Riahi et al. 2017; Popp et al. 2017a). SSP3 is characterised by high population growth, higher inequalities, limited land-use regulation, resource-intensive consumption including meat-intensive diets, and constrained trade (for further details see Chapter 1 and Cross-Chapter Box 9: Illustrative climate and land pathways in Chapter 6). These two SSPs, among the set of five SSPs, were selected because they illustrate contrasting futures, ranging from low (SSP1) to high (SSP3) challenges to mitigation and adaptation. Figure 7.2 shows that for a given global mean temperature change, risks are different under SSP1 compared to SSP3. In SSP1, global temperature change does not increase above 3°C even in the baseline case (i.e., with no additional mitigation measures) because in this pathway the combination of low population and autonomous improvements, for example, in terms of carbon intensity and/or energy intensity, effectively act as mitigation measures (Riahi et al., 2017). Thus Figure 7.2 does not indicate risks beyond this point in either SSP1 and SSP3. Literature based on such socio-economic and climate models is still emerging and there is a need for greater research on impacts of different pathways. There are few SSP studies exploring aspects of desertification and land degradation, but a greater number of SSP studies on food security (see supplementary material). SSP1 reduces the vulnerability and exposure of human and natural systems and thus limits risks resulting from desertification, land degradation and food insecurity compared to SSP3 (*high confidence*).

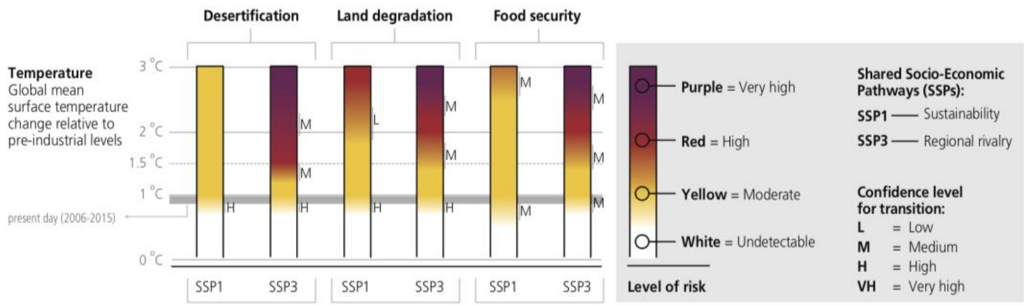

Figure 7.2: Risks associated with desertification, land degradation and food security as a function of climate change and level of socio-economic development. Increasing risks associated to desertification include a growing fraction of population exposed and vulnerable to water scarcity and changes in irrigation supply and demand. Risks related to land degradation include increased vegetation loss, population exposed to fire and floods, costs of floods, extent of deforestation, and ecosystem services including the ability of land to sequester carbon. Risks to food security include population at risk of hunger, food price increases, increases in disability adjusted life years. The risks are assessed for two contrasted socio-economic futures (SSP1 and SSP3) under unmitigated climate change {3.5; 4.2.1.2; 5.2.2; 5.2.3; 5.2.4; 5.2.5; 6.1.4; 7.2}. **The supporting literature is provided in Chapter 7 Supplementary Material.**

Changes to the water cycle due to global warming is an essential driver of desertification and of the risks to livelihood, food production and vegetation in dryland regions. Changes in water scarcity due to climate change have already been detected in some dryland regions (section 7.2.2.4) and therefore the transition to moderate risk occurred in recent decades (*high confidence*). (IPCC 2018d) noted that in the case of risks to water resources, socio-economic drivers are expected to have a greater influence than the changes in climate (*medium confidence*). Indeed, in SSP1 there is only moderate risk even at 3°C of warming, due to the lower exposure and vulnerability of human population (Hanasaki et al. 2013a; Arnell and Lloyd-Hughes 2014; Byers et al. 2018b). Considering drylands only, (Byers et al. 2018b) estimate, using a time sampling approach for climate change and the 2050 population, that at 1.5°C, 2°C and 3°C, the dryland population exposed and vulnerable to water stress in SSP1 will be 2%, 3% and 3% respectively, thus indicating relatively stable moderate risks. In SSP3, the transition from moderate to high risk occurs in the range 1.2°C to 1.5 °C (*medium confidence*) and the transition from high to very high risk is in the range 1.5°C to 2.8 °C (*medium confidence*). (Hanasaki et al. 2013b) found a consistent increase in water stress at higher warming levels due in large part related to growth in population and demand for energy and agricultural commodities and to a lesser extent due to hydrological changes induced by global warming. In SSP3, (Byers et al. 2018b) estimate that at 1.5°C, 2°C and 3°C, the population exposed and vulnerable to water stress in drylands will steadily increase from 20% to 22% and 24%, respectively, thus indicating overall much higher risks compared to SSP1 for the same global warming levels.

SSP studies relevant to land degradation assess risks such as: number of people exposed to fire, the costs of floods and coastal flooding, and loss of ecosystem services including the ability of land to sequester carbon. The risks related to permafrost melting (section 7.2.2.7) are not considered here due to the lack of SSP studies addressing this topic. Climate change impacts on various components of land degradation have already been detected (sections 7.2.2.3; 7.2.2.5; 7.2.2.6) and therefore the transition from undetectable to moderate risk is in the range 0.7 °C –1°C (*high confidence*). Less than 100 million people are exposed to habitat degradation at 1.5°C under SSP1 in non-dryland regions, increasing to 257 million at 2°C (Byers et al. 2018). This suggests a gradual transition to high risk in the range 1.8°C to 2.8°C, but a *low confidence* is attributed due to the very limited evidence to constrain this transition.

By contrast in SSP3, there are already 107 million people exposed to habitat degradation at 1.5°C, increasing to 1156 million people at 3°C (Byers et al. 2018b). Furthermore, (Knorr et al. 2016b) estimate that 646 million people will be exposed to fire at 2°C warming, the main risk driver being the high population growth in SSP3 rather than increased burned area due the climate change. Exposure to extreme rainfall, a causative factor for soil erosion and flooding, also differs under SSPs. Under SSP1 up to 14% of the land and population experience five day extreme precipitation events. Similar levels of exposure occur at lower temperatures in SSP3 (Zhang et al. 2018b). Population exposed to coastal flooding is lowest under SSP1 and higher under SSP3 with a limited effect of enhanced protection in SSP3 already after 2°C warming (Hinkel et al. 2014). The transition from high to very high risk will occur at 2.2°C –2.8°C in SSP3 (*medium confidence*), whereas this level of risk is not expected to be reached in SSP1.

The greatest number of SSP studies explore climate change impacts relevant to food security, including population at risk of hunger, food price increases, increases in disability adjusted life years (Hasegawa et al. 2018a; Wiebe et al. 2015a; van Meijl et al. 2018a; Byers et al. 2018b). Changes in crop yields and food supply stability have already been attributed to climate change (sections 7.2.2.1; 7.2.2.2) and the transition from undetectable to moderate risk is placed at 0.5°C – 1°C (*medium confidence*). At 1.5°C, about 2 million people are exposed and vulnerable to crop yield change in SSP1 (Hasegawa et al. 2018b; Byers et al. 2018b), implying moderate risk. A transition from moderate to high risk is expected above 2.5°C (*medium confidence*) with population at risk of hunger of the order of 100 million (Byers et al. 2018b). Under SSP3, high risks already exist at 1.5°C (*medium confidence*), with 20 million people exposed and vulnerable to crop yield change. By 2°C, 178 million are vulnerable and 854 million people are vulnerable at 3°C (Byers et al. 2018b). This is supported by the higher food prices increase of up to 20% in 2050 in a RCP6.0 scenario (i.e., slightly below 2°C) in SSP3 compared to up to 5% in SSP1 (van Meijl et al. 2018). Furthermore in SSP3, restricted trade increase this price effect (Wiebe et al. 2015). In SSP3, the transition from high to very high risk is in the range 2°C –2.7°C (*medium confidence*) while this transition is never reached in SSP1. This overall confirms that socio-economic development, by affecting exposure and vulnerability, has an even larger effect than climate change for future trends in the population at risk of hunger O'Neill et al. (2017) (p32). Changes can also threaten development gains (*medium confidence*). Disability adjusted life years due to childhood underweight decline in both SSP1 and SSP3 by 2030 (by 36.4 million disability adjusted life years in SSP1 and 16.2 million in SSP3). However by 2050, disability adjusted life years increase by 43.7 million in SSP3 (Ishida et al. 2014).

7.2.3. Risks arising from responses to climate change

7.2.3.1. Risk associated with land-based adaptation

Land-based adaptation relates to a particular category of adaptation measures relying on land management (Sanz et al. 2017). While most land-based adaptation options provide co-benefits for climate mitigation and other land challenges (Chapter 6, 6.4.1), in some contexts adaptation measures can have adverse side-effects, thus implying a risk to socio-ecological systems.

One example of risk is the possible decrease in farmer income when applying adaptive cropland management measures. For instance, conservation agriculture including the principle of no-till farming contribute to soil erosion management (Chap 6, 6.2. Yet, no-till management can reduce crop yields in some regions, and although this effect is minimised when no-till farming is complemented by the other two principle of conservation agriculture, this could induce a risk to livelihood in vulnerable smallholder farming systems (Pittelkow et al. 2015).

Another example is the use of irrigation against water scarcity and drought. During the long lasting drought from 2007–2009 in California, US, farmers adapted by relying on groundwater withdrawal and caused groundwater depletion at unsustainable levels (Christian-Smith et al. 2015). The long term effects of irrigation from groundwater may cause groundwater depletion, land subsidence, aquifer overdraft, and saltwater intrusion (Tularam and Krishna 2009). Therefore, it is expected to increase the vulnerability of coastal aquifers to climate change due to groundwater usage (Ferguson and Gleeson 2012). The long term irrigation practice from groundwater may cause severe combination of potential side effects and consequently irreversible results.

7.2.3.2. Risk associated with land-based mitigation

While historically land use activities have been a net source of GHG emissions, in future decades the land sector will not only need to reduce its emissions, but also to deliver negative emissions through Carbon Dioxide Removal (CDR) to reach the objective of limiting global warming at 2°C or below (Chapter 2 Section 2.5). Although land-based mitigation in itself is a risk-reduction strategy aiming at abating climate change, it also entails risks to humans and ecosystems depending on the type of measures and the scale of deployment. These risks fall broadly into two categories: risk of mitigation failure - due to uncertainties about mitigation potential, potential for sink reversal and moral hazard - and risks arising from adverse side-effects - due to increased competition for land and water resources. This section focuses specifically on bioenergy and BECCS since it is one of the most prominent land-based mitigation strategies in future mitigation scenarios (along with large-scale forest expansion discussed in Cross-Chapter Box 1: Scenarios, in Chapter 1) and it is assessed in Chapter 6 as being, at large scales, the only response option with adverse side-effects across all dimensions (adaptation, food security, land degradation and desertification; see 6.4.1).

Risk of mitigation failure. The mitigation potential from bioenergy and BECCS is highly uncertain with estimates ranging from 0.4 to 11.3 $GtCO_2e\ yr^{-1}$ for the technical potential while consideration of sustainability constraints suggest an upper end around 5 $GtCO_2e\ yr^{-1}$ (Chapter 2, section 2.6). In comparison, IAM-based mitigation pathways compatible with limiting global warming at 1.5°C project bioenergy and BECCS deployment exceeding this range (Chapter 2, Fig. 2.24). There is *medium confidence* that IAMs currently do not reflect the lower end and exceed the upper end of bioenergy and BECCS mitigation potential estimates (Anderson and Peters 2016; Krause et al. 2018; IPCC 2018c), with implications for the risk associated with reliance on bioenergy and BECCS deployment for climate mitigation.

In addition, land-based CDR strategies are subject to a risk of carbon sink reversal. This implies a fundamental asymmetry between mitigation achieved through fossil fuel emissions reduction compared to CDR. While carbon in fossil fuel reserves - in the case of avoided fossil fuel emissions - is locked permanently (at least over time scale of several thousand years), carbon sequestered into the terrestrial biosphere – to compensate fossil fuel emissions – is subject to various disturbances in particular from climate change and associated extreme events (Fuss et al. 2018; Dooley and Kartha 2018). The probability of sink reversal therefore increases with climate change, implying that the effectiveness of land-based mitigation depends on emission reductions in other sectors and can be sensitive to temperature overshoot (*high confidence*). In the case of bioenergy associated with CCS (BECCS), the issue of the long-term stability of the carbon storage is linked to technical and geological constraints, independent of climate change but presenting risks due to limited knowledge and experience (Chapter 6; Cross-Chapter Box 7: Bioenergy, in Chapter 6).

Another factor in the risk of mitigation failure, is the moral hazard associated with CDR technologies. There is *medium evidence and medium agreement* that the promise of future CDR deployment, bioenergy and BECCS in particular, can deter or delay ambitious emission reductions in other sectors (Anderson and Peters 2016; Markusson et al. 2018a; Shue 2018a). The consequences are an increased pressure on land with higher risk of mitigation failure and of temperature overshoot and a transfer of the burden of mitigation and unabated climate change to future generations. Overall, there is therefore *medium evidence and high agreement* that prioritising early decarbonisation with minimal reliance on CDR decreases the risk of mitigation failure and increases intergenerational equity (Geden et al. 2019; Larkin et al. 2018; Markusson et al. 2018b; Shue 2018b).

Risk from adverse side-effects. At large scales, bioenergy (with or without CCS) is expected to increase competition for land, water resources and nutrients, thus exacerbating the risks of food insecurity, loss of ecosystem services and water scarcity (Chapter 6; Cross-Chapter Box 7: Bioenergy in Chapter 6). Figure 7.3 shows the risk level (from undetectable to very high, aggregating risks of food insecurity, loss of ecosystem services and water scarcity) as a function of the global amount of land (million km^2) used for bioenergy, considering second generation bioenergy. Two illustrative future socio-economic pathways (SSP1 and SSP3; see section 7.2.2 for more details) are depicted, in SSP3 the competition for land is exacerbated compared to SSP1 due to higher food demand resulting from larger population growth and higher consumption of meat-based products. The literature used in this assessment is based on IAM and non-IAM-based studies examining the impact of bioenergy crop deployment on various indicators, including food security (food prices or population at risk of hunger with explicit consideration of exposure and vulnerability), SDGs, ecosystem losses, transgression of various planetary boundaries and water consumption (see supplementary material). Since most of the assessed literature is centered around 2050 prevailing demographic and economic conditions for this year are used for the risk estimate. An aggregated risk metric including risks of food insecurity, loss of ecosystem services and water scarcity is used because there is no unique relationship between bioenergy deployment and the risk outcome for a single system. For instance, bioenergy deployment can be implemented in such a way that food security is prioritised at the expense of natural ecosystems, while the same scale of bioenergy deployment implemented with ecosystem safeguards would lead to a fundamentally different outcome in terms of food security (Boysen et al. 2017a). Considered as a combined risk, however, the possibility of a negative outcome on either food security, ecosystems or both can be assessed with less ambiguity and independently of possible implementation choices.

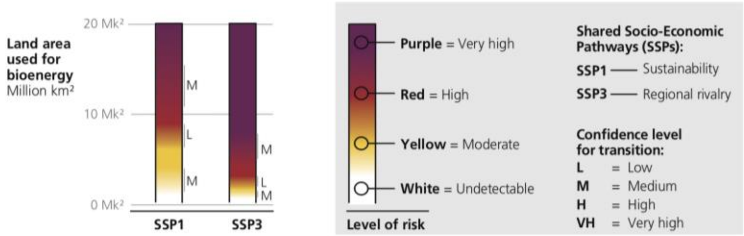

Figure 7.3: Risks associated with bioenergy crop deployment as a land-based mitigation strategy under two SSPs (SSP1 and SSP3). The assessement is based on literature investigating the consequences of bioenergy expansion for food security, ecosystem loss and water scarcity. These risk indicators were aggregated as a single risk metric in the figure. In this context, very high risk indicates that important adverse consequences are expected for all these indicators (more than 100 million people at risk of hunger, major ecosystem losses and severe water scarcity issues). The climate scenario considered is a mitigation scenario consistent with limiting global warming at 2°C (RCP2.6), however some studies considering other scenarios (e.g., no climate change) were considered in the expert judgement as well as results from other SSPs (e.g., SSP2). The literature supporting the assessment is provided in Table SM7.3.

In SSP1, there is *medium confidence* that 1 to 4 million km^2 can be dedicated to bioenergy production without significant risks to food security, ecosystem services and water scarcity. At these scales of deployment, bioenergy and BECCS could have co-benefits for instance by contributing to restoration of degraded land and soils (Cross-Chapter Box 7: Bioenergy and BECCS in Chapter 6). Although currently degraded soils (up to 20 million km^2) represent a large amount of potentially available land (Boysen et al. 2017a), trade-offs would occur already at smaller scale due to fertiliser and water use (Hejazi et al. 2014; Humpenöder et al. 2017; Heck et al. 2018a; Boysen et al. 2017b). There is *low confidence* that the transition from moderate to high risk is in the range 6-8.7 million km^2. In SSP1, (Humpenöder et al. 2017) found no important impacts on sustainability indicators at a level of 6.7 million km^2, while (Heck et al. 2018b) note that several planetary boundaries (biosphere integrity; land-system change; biogeochemical flows; freshwater use) would be exceeded above 8.7 million km^2. There is *very high confidence* that all the risk transitions occur at lower bioenergy levels in SSP3, implying higher risks associated with bioenergy deployment, due to the higher competition for land in this pathway. In SSP3, land-based mitigation is therefore strongly limited by sustainability constraints such that moderate risk occur already between 0.5 and 1.5 million km^2 (*medium* *confidence*). There is *medium confidence* that a bioenergy footprint beyond 4 to 8 million km^2 would entail very high risk with transgression of most planetary boundaries (Heck et al. 2018b), strong decline in sustainability indicators (Humpenöder et al. 2017) and increase in the population at risk of hunger well above 100 million (Fujimori et al. 2018a; Hasegawa et al. 2018b).

7.2.4. Risks arising from Hazard, Exposure, and Vulnerability

Table 7.1 shows hazards from land-climate-society interactions identified in previous chapters, or in other IPCC reports (with supplementary hazards appearing in the Appendix); the regions that are exposed or will be exposed to these hazards; components of the land-climate systems and societies that are vulnerable to the hazard; the risk associated with these impacts and the available indicative policy responses. The last column shows representative supporting literature.

Included are forest dieback, extreme events in multiple economic and agricultural regimes (also see 7.2.2.1, 7.2.2.2), disruption in flow regimes in river systems, climate change mitigation impacts (also see 7.2.3.2), competition for land (plastic substitution by cellulose, charcoal production), land degradation and desertification (also see 7.2.2.8), loss of carbon sinks, permafrost destabilisation (also see 7.2.2.7), and stranded assets (also see 7.3.4). Other hazards such as from failure of carbon storage, renewable energy impacts on land use, wild-fire in forest-urban transition context, extreme events effects on cultural heritage and urban air pollution from surrounding land-use are covered in Table 7.1 extension in the appendix as well in 7.5.6.

Table 7.1 Characterising land-climate risk and indicative policy responses. Table shows hazards from land-climate-society interactions identified in previous chapters or in *other* IPCC reports; the regions that are exposed or will be exposed to these hazards; components of the land-climate systems and societies that are vulnerable to the hazard; the risk associated with these impacts and the available policy responses and response options from Chapter 6. The last column shows representative supporting literature

Land-Climate-Society interaction Hazard	Exposure	Vulnerability	Risk	Policy Response (Indicative)	References
Forest dieback	Widespread across biomes and regions	Marginalised Population with insecure land tenure	• Loss of forest-based livelihoods • Loss of identity	• Land rights • Community based conservation • Enhanced political enfranchisement • Manager-scientist partnerships for adaptation silviculture	(Allen et al. 2010; McDowell and Allen 2015; Sunderlin et al. 2017; Belcher et al. 2005; Soizic et al 2013)(Nagel et al. 2017)
		Endangered species and ecosystems	• Extinction • Loss of ecosystem services • Cultural loss	• Effective enforcement of protected areas and curbs on illegal trade • Ecosystem Restoration • Protection of indigenous people	(Bailis et al. 2015; Cameron et al. 2016)
Extreme events in multiple economic and agricultural regimes	Global	• Food importing countries • Low income indebtedness • Net food buyer	• Conflict • Migration • Food inflation • Loss of life • Disease, malnutrition • Farmer suicides	• Insurance • Social Protection encouraging diversity of sources • Climate smart agriculture • Land rights and tenure • Adaptive Public Distribution Systems	(Fraser et al. 2005; Schmidhuber and Tubiello 2007; Lipper et al. 2014a; Lunt et al. 2016; Tigchelaar et al. 2018; Casellas Connors and Janetos 2016)

Land-Climate-Society interaction Hazard	Exposure	Vulnerability	Risk	Policy Response (Indicative)	References
Disruption of flow regimes in river systems	1.5 billion people, Regional (e.g., South Asia, Australia) Aral sea and others	• Water intensive agriculture • Fresh-water, estuarine and near coastal ecosystems • Fishers • Endangered species and ecosystems	• Loss of livelihoods and identity • Migration • Indebtedness	• Build alternative scenarios for economies and livelihoods based on non-consumptive use (e.g., wild capture fisheries) • Define and maintain ecological flows in rivers for target species and ecosystem services • Experiment with alternative less water consuming crops and water management strategies • Redefine SDGs to include fresh-water ecosystems or adopt alternative metrics of sustainability Based on Nature Contributions to People (NCP)	(Craig 2010; Di Baldassarre et al. 2013; Verma et al. 2009; Ghosh et al. 2016; Higgins et al. 2018;) (Hall et al. 2013; Youn et al. 2014)
Depletion/ exhaustion of ground-water	Wide-spread across semi-arid and humid biomes India, China and the United States Small Islands	• Farmers, drinking water supply • Irrigation • See forest note above • Agricultural production • Urban sustainability (Phoenix, US)	• Food insecurity • Water insecurity • Distress migration • Conflict • Disease • Inundation of coastal regions, estuaries and deltas	• Monitoring of emerging ground-water-climate linkages • Adaptation strategies that reduce dependence on deep ground water • Regulation of ground-water use • Shift to less water-	(Wada et al. 2010; Rodell et al. 2009; Taylor et al. 2013; Aeschbach-Hertig and Gleeson 2012)

Land-Climate-Society interaction Hazard	Exposure	Vulnerability	Risk	Policy Response (Indicative)	References
		• Reduction in dry-season river flows • Sea level rise		intensive rain fed crops and pasture • Conjunctive use of surface and ground-water	
Climate change Mitigation impacts	Across various biomes especially semi-arid and aquatic where renewable energy projects (solar, biomass, wind and small hydro) are sited	• Fishers and pastoralists • Farmers • Endangered range restricted species and ecosystems	• Extinction of species • Downstream loss of ecosystem services Loss of livelihoods and identity of fisher/pastoralist communities • Loss of regional food security	• Avoidance and informed siting in priority basins • Mitigation of impacts • Certification	(Zomer et al. 2008; Nyong et al. 2007; Pielke et al. 2002; Schmidhuber and Tubiello 2007; Jumani et al. 2017; Eldridge et al. 2011; Bryan et al. 2010; Scarlat and Dallemand 2011)
Competition for land e.g., Plastic substitution by cellulose, Charcoal production	Peri-urban and rural areas in developing countries	• Rural landscapes; farmers; charcoal suppliers; small businesses	• Land degradation; loss of ecosystem services; GHG emissions; lower adaptive capacity	• Sustainability certification; producer permits; subsidies for efficient kilns	(Woollen et al. 2016; Kiruki et al. 2017a)
Land degradation and desertification	Arid, Semi-arid and sub-humid regions	• Farmers • Pastoralists • Biodiversity	• Food insecurity • Drought • Migration • Loss of agro and wild biodiversity	• Restoration of ecosystems and management of invasive species • Climate smart agriculture and	(Fleskens, Luuk, Stringer 2014; Lambin et al. 2001; Cowie et al. 2018a; Few and Tebboth 2018; Sandstrom and

Land-Climate-Society interaction Hazard	Exposure	Vulnerability	Risk	Policy Response (Indicative)	References
				livestock management • Managing economic impacts of global and local drivers • Changes in relief and rehabilitation policies • Land degradation neutrality	Juhola 2017)
Loss of carbon sinks	Wide-spread across biomes and regions	• Tropical forests • Boreal soils	• Feed-back to global and regional climate change	• Conservation prioritisation of tropical forests • Afforestation	(Barnett et al. 2005; Tribbia and Moser 2008)
Permafrost destabilisation	Arctic and Sub-Arctic regions	• Soils • Indigenous communities • Biodiversity	• Enhanced GHG emissions	• Enhanced carbon uptake from novel ecosystem after thaw • Adapt to emerging wetlands	(Schuur et al. 2015)
Stranded assets	Economies transitioning to low carbon pathways Oil economies Coastal regions facing inundation	Coal based power Oil refineries Plastic industry Large dams Coastal infrastructure	• Disruption of regional economies and conflict • Unemployment • Push-back against renewable energy • Migration	• Insurance and tax cuts • Long-term power purchase agreements • Economic and technical support for transitioning economies • transforming oil wealth into renewable energy leadership • Redevelopment using adaptation • OPEC investment in information sharing for transition	(Farfan and Breyer 2017; Ansar et al. 2013; Van de Graaf 2017; Trieb et al. 2011)

7.3. Consequences of climate – land change for human well-being and sustainable development

To further explore what is at stake for human systems, this section assesses literature about potential consequences of climate and land change for human well-being and ecosystems upon which humans depend. Risks described in 7.2 have significant social, spiritual, and economic ramifications for societies across the world and this section explores potential implications of the risks outlined above to food security, livelihood systems, migration, ecosystems, species, infectious disease, and communities and infrastructure. Because food and livelihood systems are deeply tied to one another, combinations of climate and land change could pose higher present risks to humans and ecosystems than examination of individual elements alone might suggest.

7.3.1. What is at stake for food security?

This section examines risks to food security when access to food is jeopardised by yield shortfall and instability related to climate stressors. Past assessments of climate change impacts have sometimes assumed that when grain and food yields in one area of the world are lower than expected, world trade can redistribute food adequately to ensure food security. There is *medium confidence* that severe and spatially extensive climatic stressors pose *high risk* to stability of and access to food for large numbers of people across the world.

The 2007–2008, and 2010–2011 droughts in several regions of the world resulted in crop yield decline that in turn led some governments to protect their domestic grain supplies rather than engaging in free trade to offset food shortfalls in other areas of the world. These responses cascaded and strongly affected regional and global food prices. Simultaneous crop yield impacts combined with trade impacts have proven to play a larger and more pervasive role in global food crises than previously thought (Sternberg 2012, 2017; Bellemare 2015) (Chatzopoulos et al. 2019). There is *high confidence* that regional climate extremes already have significant negative domestic and international economic impacts (Chatzopoulos et al. 2019).

7.3.2. Risks to where and how people live: Livelihood systems and migration

There is *high confidence* that climate- and land change interact with social, economic, political, and demographic factors that affect how well and where people live (Sudmeier-Rieux et al. 2017; Government Office for Science 2011; Laczko and Piguet 2014; Bohra-Mishra and Massey 2011; Raleigh et al. 2015; Warner and Afifi 2011; Hugo 2011; Warner et al. 2012). There is *high evidence and high agreement* that people move to manage risks and seek opportunities for their safety and livelihoods, recognising that people respond to climatic change and land-related factors in tandem with other variables (Hendrix and Salehyan 2012; Lashley and Warner 2015; van der Geest and Warner 2014; Roudier et al. 2014; Warner and Afifi 2014)(McLeman 2013; Kaenzig and Piguet 2014; Internal Displacement Monitoring Center 2017; Warner 2018; Cohen and Bradley 2010; Thomas and Benjamin 2017). People move towards areas offering safety and livelihoods such as in rapidly growing settlements in coastal zones (Black et al. 2013; Challinor et al. 2017; Adger et al. 2013); burgeoning urban areas also face changing exposure to combinations of storm surges and sea level rise, coastal erosion and soil and water salinisation, and land subsidence (Geisler and Currens 2017; Maldonado et al. 2014; Bronen and Chapin 2013).

There is *medium confidence* that livelihood-related migration can accelerate in the short to medium term when weather dependent livelihood systems deteriorate in relation to changes in precipitation, changes in ecosystems, and land degradation and desertification (Abid et al. 2016)(Scheffran et al. 2012; Fussell et al. 2014; Bettini and Gioli 2016; Reyer et al. 2017)(Warner and Afifi 2014)(Handmer et al. 2012; Nawrotzki and Bakhtsiyarava 2017; Nawrotzki et al. 2016; Steffen et al. 2015; Black et al.

2013). Slow onset climate impacts and risks can exacerbate or otherwise interact with social conflict corresponding with movement at larger scales (see Section 7.2.3.2) and long term deterioration in habitability of regions could trigger spatial population shifts (Denton et al. 2014).

There is *medium evidence* and *medium agreement* that climatic stressors can worsen the complex negative impacts of strife and conflict (Schleussner et al. 2016; Barnett and Palutikof 2014; Scheffran et al. 2012). Climate change and human mobility could be a factor that heightens tensions over scarce strategic resources, a further destabilising influence in fragile states experiencing socio-economic and political unrest (Carleton and Hsiang 2016a). Conflict and changes in weather patterns can worsen conditions for people working in rain fed agriculture or subsistence farming, interrupting production systems, degrading land and vegetation further (Papaioannou 2016; Adano and Daudi 2012). In recent decades, droughts and other climatic stressors have compounded livelihood pressures in areas already torn by strife (Tessler et al. 2015; Raleigh et al. 2015), such as in the Horn of Africa. Seizing of agricultural land by competing factions, preventing food distribution in times of shortage have in this region and others contributed to a triad of food insecurity, humanitarian need, and large movements of people (Theisen et al. 2011; Mohmmed et al. 2018; Ayeb-Karlsson et al. 2016; von Uexkull et al. 2016; Gleick 2014; Maystadt and Ecker 2014). People fleeing complex situations may return if peaceful conditions can be established. Climate change and climate change induced development responses in countries and regions are likely to exacerbate tensions over water and land its impact on agriculture, fisheries, livestock and drinking water downstream. Shared pastoral landscapes used by disadvantaged or otherwise vulnerable communities are particularly impacted by conflicts that are likely to become more severe under future climate change (Salehyan and Hendrix 2014; Hendrix and Salehyan 2012). Extreme events could considerably enhance these risks, in particular long-term drying trends (Kelley et al. 2015; Cutter et al. 2012a). There is *medium evidence* and *medium agreement* that governance is key in magnifying or moderating climate change impact and conflict (Bonatti et al. 2016).

There is *low evidence and medium agreement* that longer-term deterioration in the habitability of regions could trigger spatial population shifts (Seto 2011). Heat waves, rising sea levels that salinise and inundate coastal and low-lying aquifers and soils, desertification, loss of geologic sources of water such as glaciers and freshwater aquifers could affect many regions of the world and put life-sustaining ecosystems under pressure to support human populations (Flahaux and De Haas 2016; Chambwera et al. 2015; Tierney et al. 2015; Lilleør and Van den Broeck 2011).

7.3.3. Risks to humans from disrupted ecosystems and species

Risks of loss of biodiversity and ecosystem services

Climate change poses significant threat to species survival, and to maintaining biodiversity and ecosystem services. Climate change reduces the functionality, stability, and adaptability of ecosystems (Pecl et al. 2017). For example, drought affects cropland and forest productivity and reduces associated harvests (provisioning services). In additional, extreme changes in precipitation may reduce the capacity of forests to provide stability for groundwater (regulation and maintenance services). Prolonged periods of high temperature may cause widespread death of trees in tropical mountains, boreal and tundra forests, impacting diverse ecosystem services including impacting aesthetic and cultural services (Verbyla 2011; Chapin et al. 2010; Krishnaswamy et al. 2014). According to the Millennium Ecosystem Assessment (Millennium Ecosystem Assessment 2005), climate change is likely to become one of the most significant drivers of biodiversity loss by the end of the century.

There is *high confidence* that climate change already poses a moderate risk to biodiversity, and is projected to become a progressively widespread and high risk in the coming decades; loss of Arctic sea ice threatens biodiversity across an entire biome and beyond; the related pressure of ocean acidification, resulting from higher concentrations of carbon dioxide in the atmosphere, is also already being observed (UNEP 2009). There is ample evidence that climate change and land change negatively affects biodiversity across wide spatial scales. Although there is relatively *limited evidence* of current extinctions caused by climate change, studies suggest that climate change could surpass habitat destruction as the greatest global threat to biodiversity over the next several decades (Pereira et al. 2010). However, the multiplicity of approaches and the resulting variability in projections make it difficult to get a clear picture of the future of biodiversity under different scenarios of global climatic change (Pereira et al. 2010) . Biodiversity is also severely impacted by climate change induced land degradation and ecosystem transformation (Pecl et al. 2017). This may impact humans directly and indirectly through cascading impacts on ecosystem function and ecosystem services (Millennium Assessment 2005). Climate change related human migration is likely to impact biodiversity as people move into and contribute to land stress in biodiversity hotspots now and in the future; and as humans concurrently move into areas where biodiversity is also migrating to adapt to climate change (Oglethorpe et al. 2007).

Climate and land change increases risk to respiratory and infectious disease

In addition to risks related to nutrition articulated in Figure 7.1, human health can be affected by climate change through extreme heat and cold, changes in infectious diseases, extreme events, and land cover and land use (Hasegawa et al. 2016; Ryan et al. 2015; Terrazas et al. 2015; Kweka et al. 2016; Yamana et al. 2016). Evidence indicates that action to prevent the health impacts of climate change could provide substantial economic benefits (Martinez et al. 2015; Watts et al. 2015).

Climate change exacerbates air pollution with increasing UV and ozone concentration. It has negative impacts on human health and increases mortality rate especially in urban region (Silva et al. 2016, 2013; Lelieveld et al. 2013; Whitmee et al. 2015; Anenberg et al. 2010). In the Amazon, research shows that deforestation (both net loss and fragmentation) will increases malaria, where vectors are expected to increase their home range (Alimi et al. 2015; Ren et al. 2016), confounded with multiple factors, such as social-economic conditions and immunity (Tucker Lima et al. 2017; Barros and Honório 2015). Deforestation has been shown to enhance the survival and development of major malaria vectors (Wang et al. 2016). The WHO estimates 60,091 additional deaths for climate change induced malaria for the year 2030 and 32,695 for 2050 (World Health Organization 2014).

Human encroachment on animal habitat in combination with the bushmeat trade in Central African countries has contributed to the increased incidence of zoonotic (i.e., animal-derived) diseases in human populations, including Ebola virus epidemic (Alexander et al. 2015a; Nkengasong and Onyebujoh 2018). The composition and density of zoonotic reservoir populations, such as rodents, is also influenced by land-use and climate change (*high confidence*) (Young et al. 2017a). The bushmeat trade in many regions of central and west African forests (particularly in relation to chimpanzee and gorilla populations) elevates the risk of ebola by increasing human-animal contact (Harrod 2015).

7.3.4. Risks to Communities and Infrastructure

There is *high confidence* that policies and institutions which accentuate vicious cycles of poverty and ill-health, land degradation and greenhouse gas emissions undermine stability and are barriers to achieving climate resilient sustainable development. There is *high confidence* that change in climate and land pose high periodic and sustained risk to the very young, those living in poverty, and ageing populations. Older people are particularly exposed due to more restricted access to resources, changes in physiology, and decreased mobility resulting from age which may limit adaptive capacity of individuals and populations as a whole (Filiberto et al. 2010).

Combinations of food insecurity, livelihood loss related to degrading soils and ecosystem change, or other factors that diminish the habitability of where people live disrupt social fabric and are currently detected in most regions of the world (Carleton and Hsiang 2016b) There is *high confidence* that coastal flooding and degradation already poses widespread and rising future risk to infrastructure value and stranded infrastructure, as well as livelihoods made possible by urban infrastructure (Radhakrishnan et al. 2017; Pathirana, A., Radhakrishnan, M., Quan 2018; Pathirana, A., Radhakrishnan, M., Ashley 2018; Radhakrishnan, M., Nguyen, H., Gersonius 2018; EEA 2016; Pelling and Wisner 2012; Oke et al. 2017; Parnell and Walawege 2011; Uzun and Cete 2004; Melvin et al. 2017).

There is *high evidence and high agreement* that climate and land change pose high risk to communities and interdependent infrastructure systems including electric power, and transportation are highly vulnerable and interdependent (Below et al. 2012; Adger et al. 2013; Pathirana, A., Radhakrishnan, M., Quan 2018)(Conway and Schipper 2011; Caney 2014; Chung Tiam Fook 2017; Pathirana, A., Radhakrishnan, M., Quan 2018). These systems are exposed to disruption from severe climate events such as weather-related power interruptions lasting for hours to days (Panteli and Mancarella 2015). Increased magnitude and frequency of high winds, ice storms, hurricanes and heat waves have caused widespread damage to power infrastructure and have caused severe outages, affecting significant numbers of customers in urban and rural areas (Abi-Samra and Malcolm 2011).

Increasing populations, enhanced per capita water use, climate change, and allocations for water conservation are potential threats to adequate water availability. As climate change produces variations in rainfall, these challenges will intensify, evidenced by severe water shortages in recent years in Capetown, Los Angeles, Rio de Janeiro among others (Watts et al. 2018; Majumder 2015; Ashoori et al. 2015; Mini et al. 2015; Otto et al. 2015)(Cross-Chapter Box 5: Case study on policy responses to drought in Chapter 3)(Ranatunga et al. 2014)(Ray and Shaw 2016; Gopakumar 2014).

Cross-chapter Box 10: Economic dimensions of climate change and land

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Sustainable land management (SLM) makes strong social and economic sense. Early action in implementing SLM for climate change adaptation and mitigation provides distinct societal advantages. Understanding the full scope of what is at stake from climate change presents challenges because of inadequate accounting of the degree and scale at which climate change and land interactions impact society, and the importance society places on those impacts (Santos et al. 2016)(7.2.2, 5.3.1, 5.3.2, 4.1). The consequences of inaction and delay bring significant risks including irreversible change and loss in land ecosystem services, including food security, with potentially substantial economic damage to many countries in many regions of the world (*high confidence).*

This cross-chapter box brings together the salient economic concepts underpinning the assessments of sustainable land management and mitigation options presented in this report. Four critical concepts are required to help assess the social and economic implications of land-based climate action:

i. value to society;
ii. damages from climate and land-induced interventions on land ecosystems;
iii. costs of action and inaction;
iv. decision-making under uncertainty.

(i) Value to society

Healthy functioning land and ecosystems are essential for human health, food and livelihood security. Land derives its value to humans from being both a finite resource and vital for life, providing vital ecosystem services from water recycling, food, feed, fuel, biodiversity and carbon storage and sequestration.

Many of these ecosystem services may be difficult to estimate in monetary terms, including when they hold high symbolic value, linked to ancestral history, or traditional and indigenous knowledge systems (Boillat and Berkes 2013). Such incommensurable values of land are core to social cohesion— social norms and institutions, trust that enables all interactions, and sense of community.

(ii) Damages from climate and land-induced interventions on land ecosystems;

Values of many land-based ecosystem services and their potential loss under land-climate change interaction can be considerable: the global value of ecosystem services was valued in 2011 at USD

125 trillion per year and the annual loss due to land use change was between USD 4.3 to 20.2 trillion per year from 2007 (Costanza et al. 2014; Rockström et al. 2009). The annual costs of land degradation are estimated to be about USD 231 billion per year or about 0.41% of the global GDP of USD 56.49 trillion in 2007 (Nkonya et al. 2016) (4.4.1, 4.4.2).

Studies show increasingly negative effects on GDP from damage and loss to land-based values and service as global mean temperatures increase, although the impact varies across regions (Kompas et al. 2018).

(iii) Costs of action and inaction

Evidence suggests that the cost of inaction in mitigation and adaptation, and land use, exceeds the cost of interventions in both individual countries, regions, and worldwide (Nkonya et al. 2016). Continued inaction reduces the future policy option space, dampens economic growth and increases the challenges of mitigation as well as adaptation (Moore and Diaz 2015)(Luderer et al. 2013). The cost of reducing emissions is estimated to be considerably less than the costs of the damages at all levels (Kainuma et al. 2013; Moran 2011; Sánchez and Maseda 2016).

The costs of adapting to climate impacts are also projected to be substantial, although evidence is limited (summarised in Chambwera et al. 2014a). Estimates range from USD 9 to 166 billion per year at various scales and types of adaptation, from capacity building to specific projects (Fankhauser 2017). Inadequate literature exists on the costs of adaptation in the agriculture or land-based sectors (Wreford and Renwick 2012) due to lack of baselines, uncertainty around biological relationships and inherent uncertainty about anticipated avoided damage estimates, but economic appraisal of actions to maintain the functions of the natural environment and land sector generate positive net present values (Adaptation Sub-committee 2013).

Preventing land degradation from occurring is considered more cost-effective in the long term compared to the magnitude of resources required to restore already degraded land (Cowie et al. 2018a) (3.6.1). Evidence from drylands shows that each US dollar invested in land restoration provides between 3 and 6 in social returns over a 30 year period, using a discount rate between 2.5 and 10% (Nkonya et al. 2016). SLM practices reverse or minimise economic losses of land degradation, estimated at between USD 6.3 and 10.6 trillion annually, (ELD Initiative 2015) more than five times the entire value of agriculture in the market economy (Costanza et al. 2014; Fischer et al. 2017; Sandifer et al. 2015; Dasgupta et al. 2013) (3.7.5).

Across other areas such as food security, disaster mitigation and risk reduction, humanitarian response, and healthy diet (malnutrition as well as disease), early action generates economic benefits greater than costs (*high evidence, high agreement*) (Fankhauser 2017; Wilkinson et al. 2018; Venton 2018; Venton et al. 2012) (Clarvis et al. 2015)(Nugent et al. 2018) (Watts et al. 2018) (Bertram et al. 2018)(6.3, 6.4).

(iv) Decision-making under uncertainty

Given that significant uncertainty exists regarding the future impacts of climate change, effective decisions must be made under unavoidable uncertainty (Jones et al., 2014).
Approaches that allow for decision-making under uncertainty are continually evolving (see 7.5). An emerging trend is towards new frameworks that will enable multiple decision makers with multiple objectives to explore the trade-offs between potentially conflicting preferences to identify strategies that are robust to deep uncertainties (Singh et al. 2015; Driscoll et al. 2016; Araujo Enciso et al. 2016; Herman et al. 2014; Pérez et al. 2016; Girard et al. 2015; Haasnoot et al. 2018; Roelich and Giesekam 2019).

Valuation of benefits and damages and costing interventions: Measurement issues

Cost appraisal tools for climate adaptation are many and their suitability depends on the context (7.5.2.2). Cost-benefit analysis (CBA) and cost-effectiveness analysis (CEA) are commonly applied, especially for current climate variability situations. However, these tools are not without criticism and their limitations have been observed in the literature (see Rogelj et al. 2018). In general measuring costs and providing valuation are influenced by four conditions: measurement and valuation; the time dimension; externalities; and aggregate versus marginal costs:

- **Measurement and value issues**

Ecosystem services that are not traded in the market fall outside the formal or market-based valuation and their value is thus either not accounted for or underestimated in both private and public decisions (Atkinson et al. 2018). Environmental valuation literature uses a range of techniques to assign monetary values to environmental outcomes where no market exists (Atkinson et al. 2018) (Dallimer et al. 2018), but some values remain inestimable. For some indigenous cultures and peoples, land is not considered something that can be sold and bought, so economic valuations are not meaningful even as proxy approaches (Boillat and Berkes 2013)(Kumpula et al. 2011; Pert et al. 2015; Xu et al. 2005).

While a rigorous CBA is broader than a purely financial tool and can capture non-market values where they exist, it can prioritise certain values over others (such as profit maximisation for owners, efficiency from the perspective of supply chain processes, and judgements about which parties bear the costs). Careful consideration of whose perspectives are considered when undertaking a CBA and the limitations of these methods for policy interventions.

- **Time dimension (short vs long term) and the issue of discount rates**

Economics uses a mechanism to convert future values to present day values known as discounting, or the pure rate of time preference. Discount rates are increasingly being chosen to reflect concerns about intergenerational equity, and some countries (e.g., the UK and France) apply a declining discount rate for long term public projects. The choice of discount rate has important implications for policy evaluation (Anthoff, Tol, & Yohe, 2010; Arrow et al., 2014; Baral, Keenan, Sharma, Stork, & Kasel, 2014; Dasgupta et al., 2013; Lontzek, Cai, Judd, & Lenton, 2015; Sorokin et al., 2015; van den Bergh & Botzen, 2014)(*high evidence, high agreement*). Stern (Stern 2007), for example, used a much lower discount rate (giving almost equal weight to future generations) than the mainstream authors (e.g., Nordhaus) and obtained much higher estimates of the damage of climate change.

- **Positive and negative externalities (consequences and impacts not accounted for in market economy),**

All land use generates externalities (unaccounted for side-effects of an activity). Examples include loss of ecosystem services (e.g., reduced pollinators; soil erosion, increased water pollution, nitrification etc.). Positive externalities include sequestration of CO_2 and improved soil water filtration from afforestation. Externalities can also be social (e.g., displacement and migration) and economic (e.g., loss of productive land). In the context of climate change and land, the major externality is the AFOLU sourced emissions of GHGs. Examples of mechanisms to internalise externalities are discussed in 7.5.

- **Aggregate versus marginal costs**

Costs of climate change are often referred to through the marginal measure of the Social Cost of Carbon (SCC), which measures the total net damages of an extra metric ton of CO_2 emissions due to the associated climate change (Nordhaus 2014). The SCC can be used to determine a carbon price, but SCC depends on discount rate assumptions and may neglect processes including large losses of biodiversity, political instability, violent conflicts, large-scale migration flows, and the effects of climate change on the development of economies (Stern 2013; Pezzey 2019).

At the sectoral level, marginal abatement cost (MAC) curves are widely used for the assessment of costs related to CO_2 or GHG emissions reduction. MAC measures the cost of reducing one more GHG unit and MAC curves are either expert-based or model-derived and offer a range of approaches and assumptions on discount rates or available abatement technologies (Moran 2011).

7.3.4.1. Windows of Opportunity

Windows of opportunity are important learning moments wherein an event or disturbance in relation to land, climate, and food security triggers responsive social, political, policy change (*medium agreement*). Policies play an important role in windows of opportunity and are important in relation to managing risks of desertification, soil degradation, food insecurity, and supporting response options for sustainable land management (Chapter 6) (*high agreement*) (Kivimaa and Kern 2016; Gupta et al. 2013b; Cosens et al. 2017b; Darnhofer 2014; Duru et al. 2015).

A wide range of events or disturbances may initiate windows of opportunity ranging from climatic events and disasters, recognition of a state of land degradation, an ecological social or political crisis, and a triggered regulatory burden or opportunity. Recognition of a degraded system such as land degradation and desertification (Chapters 3 and 4) and associated ecosystem feedbacks, allows for strategies, response options and policies to address the degraded state (Nyström et al. 2012). Climate related disasters (flood, droughts etc.) and crisis may trigger latent local adaptive capacities leading to systemic equitable improvement (McSweeney and Coomes 2011), or novel and innovative recombining of sources of experience and knowledge, allowing navigation to transformative social ecological transitions (Folke et al. 2010). The occurrence of a series of punctuated crisis such as floods or droughts, qualify as windows of opportunity when they enhance society's capacity to adapt over the long term (Pahl-Wostl et al. 2013). A disturbance from an ecological, social, or political crisis may be sufficient to trigger the emergence of new approaches to governance wherein there is a change in the rules of the social world such as informal agreements surrounding human activities or formal rules of public policies (Olsson et al. 2006; Biggs et al. 2017) (See 7.6). A combination of socio-ecological changes may provide windows of opportunity for a socio-technical niche to be adopted on a greater scale transforming practices towards sustainable land management such as biodiversity based agriculture (Darnhofer 2014; Duru et al. 2015).

Policy may also create windows of opportunity. A disturbance may cause inconvenience, including high costs of compliance with environmental regulations, thereby initiating a change of behaviour (Cosens et al. 2017a). In a similar vein, multiple regulatory requirements existing at the time of a disturbance may result in emergent processes and novel solutions in order to correct for piecemeal regulatory compliance (Cosens et al. 2017a). Lastly, windows of opportunity can be created by policy mixes or portfolio that provide for creative destruction of old social processes and thereby encourage new innovative solutions (Kivimaa et al. 2017b) (See 7.4.8).

7.4. Policy Instruments for Land and Climate

This section outlines policy responses to risk. It describes multi-level policy instruments (7.4.1), policy instruments for social protection (7.4.2), policies responding to hazard (7.4.3), GHG fluxes (7.4.4), desertification (7.4.5), land degradation (7.4.6), economic instruments (7.4.7), enabling effective policy instruments through policy mixes (7.4.8), and barriers to sustainable land management and overcoming these barriers (7.4.9).

Policy instruments are used to influence behaviour and affect a response to do, not do, or continue to do certain things (Anderson 2010) and can be invoked at multiple levels (international, national, regional, and local) by multiple actors (See Table 7.2). For efficiency, equity and effectiveness considerations, the appropriate choice of instrument for the context is critical, and across the topics addressed in this report the instruments will vary considerably. A key consideration is whether the benefits of the action will generate private or public social net benefits. Pannell (2008) provides a widely-used framework for identifying the appropriate type of instrument depending on whether the actions encouraged by the instrument are private or public, and positive or negative. Positive incentives (such as financial or regulatory instruments) are appropriate where the public net benefits are highly positive and the private net benefits are close to zero. This is likely to be the case for GHG mitigation measures such as carbon pricing. Many other GHG mitigation measures (more effective water or fertiliser use, better agricultural practices, less food waste, agroforestry systems, better forest management) discussed in previous chapters may have substantial private as well as public benefit. Extension (knowledge provision) is recommended for when public net benefits are highly positive and private net benefits slightly positive, again for some GHG mitigation measures, and many adaptations, food security and sustainable land management measures. Where the private net benefits are slightly positive but the public net benefits highly negative, negative incentives (such as regulations and prohibitions) are appropriate, for example over-application of fertiliser.

While Pannell (2008)'s framework is useful, it does not address considerations relating to the time-scale of actions and their consequences particularly in the long time-horizons involved under climate change: private benefits may accrue in the short term but become negative over time (Outka 2012) and some of the changes necessary will require transformation of existing systems (Park et al. 2012; Hadarits et al. 2017) for which a more comprehensive suite of instruments would be necessary. Furthermore, the framework applies to private land ownership, so where land is in different ownership structures, different mechanisms will be required. Indeed, land tenure is recognised as a factor in barriers to Sustainable Land Management and an important Governance consideration (see 7.4.9, 7.6.4). A thorough analysis of the implications of policy instruments temporally, spatially and across other sectors and goals (e.g., climate v. development) is essential before implementation to avoid unintended consequences and achieve policy coherence (7.4.8).

7.4.1. Multi-level Policy Instruments

Policy responses and planning in relation to land and climate interactions occur at and across multiple levels, involve multiple actors, and utilise multiple planning mechanisms (Urwin and Jordan 2008). Climate change is occurring on a global scale while the impacts of climate change vary from region to region and even within a region. Therefore, in addressing local climate impacts, local governments and communities are key players. Advancing governance of *climate change* across all *levels* of government and relevant stakeholders is crucial to avoid policy gaps between local action *plans* and national/sub-national policy frameworks (Corfee-Morlot et al. 2009).

This section of the chapter identifies policies by level that respond to land and climate problems and risks. As risk management in relation to land and climate occurs at multiple levels by multiple actors, and across multiple sectors in relation to hazards (as listed on Table 7.2), risk governance, or the consideration of the landscapes of risk arising from Chapters 2 through 6 is addressed in Sections 7.5 and 7.6. Categories of instruments include regulatory instruments (command and control measures), economic and market instruments (creating a market, sending price signals, or employing a market strategy), voluntary of persuasive instruments (persuading people to internalise behaviour), and managerial (arrangements including multiple actors in cooperatively administering a resource or overseeing an issue) (Gupta et al. 2013a; Hurlbert 2018b).

Given the complex spatial and temporal dynamics of risk, a comprehensive, portfolio of instruments and responses is required to comprehensively manage risk. Operationalising a portfolio response can mean layering, sequencing or integrating approaches. Layering means that within a geographical area, households are able to benefit from multiple interventions simultaneously (e.g., those for family planning and those for livelihoods development). A sequencing approach starts with those interventions, which address the initial binding constraints, and then further interventions are later added (e.g., the poorest households first receive grant-based support before then gaining access to appropriate microfinance or market-oriented initiatives). Integrated approaches involve cross-sectoral support within the framework of one program (Scott et al. 2016; Tengberg and Valencia 2018) (see 7.4.8, 7.5.6, and 7.6.3).

Climate related risk could be categorised by climate impacts such as flood, drought, cyclone etc. (Christenson et al. 2014). Table 7.2 outlines instruments relating to impacts responding to the risk of climate change, food insecurity, land degradation and desertification, and hazards (flood, drought, forest fire), and GHG fluxes (climate mitigation).

Table 7.2 Policies/Instruments that address multiple land-climate risks at different jurisdictional levels

Scale	Policy/Instrument	Food Security	Land degradation & desertification	Sustainable land management	Climate related Extremes	GHG flux climate change mitigation
Global/ Cross Border	Finance mechanisms (also National)	X	X	X	X	X
	Certification (also National)		X	X		X
	Standards (including Risk Standards)(also National)		X	X	X	X
	Market based systems (also National)			X		X
	Payments for Ecosystem Services (also National)		X	X	X	X
	Disaster assistance (also National)				X	
National	Taxes	X		X		X
	Subsidies	X	X	X		X
	Direct Income Payments (with Cross-Compliance)	X	X	X		X
	Border adjustments (e.g., tariffs)	X				X
	Grants	X	X	X	X	X
	Bonds	X	X	X		X
	Forecast-based finance, targeted microfinance	X	X	X		X
	Insurance (various forms)	X			X	
	Hazard information and communication (also sub-national and local)	X			X	
	Drought preparedness plans (also sub-national and local)	X			X	
	Fire policy (suppression or prescribed fire management)			X	X	X
	Regulations	X	X	X	X	X
	Land ownership laws (reform of, if necessary, for secure land title, or access/control)	X	X	X		
	Protected Area Designation and management		X	X		
	Extension – including skill and community development for livelihood diversification (also sub-national and local)	X	X	X	X	X
Sub-national	Spatial and landuse planning	X	X		X	
	Watershed management	X	X			
Local	Landuse zoning, spatial planning and integrated landuse planning	X		X	X	
	Community-based awareness programmes	X	X	X	X	X

This table highlights policy and instruments addressing key themes identified in this chapter;

an X indicates the relevance of the policy or instrument to the corresponding theme.

7.4.2. Policies for Food Security and Social Protection

There is *medium evidence* and *high agreement* that a combination of structural and non-structural policies is required in averting and minimising as well as responding to land and climate change risk, including food and livelihood security. If disruptions to elements of food security are long-lasting, policies are needed to change practices

If disruptions to food and livelihood systems are temporary, then policies aimed at stemming worsening human wellbeing and stabilising short-term income fluctuations in communities (such as increasing rural credit or providing social safety net programs) may be appropriate (Ward 2016).

7.4.2.1. Policies to ensure availability, access, utilisation, and stability of food

Food security is affected by interactions between climatic factors (rising temperatures, changes in weather variability and extremes), changes in land-use and land degradation, and socio-economic pathways and policy choices related to food systems (see Figure 7.1 and Figure 7.2). As outlined in Chapter 5, key aspects of food security are food availability, access to food, utilisation of food, and stability of food systems.

While comprehensive reviews of policy are rare and additional data is needed (Adu et al. 2018), evidence indicates the result of food security interventions vary widely due to differing values underlying the design of instruments. A large portfolio of measures is available to shape outcomes in these areas from the use of tariffs or subsidies to payments for production practices (OECD 2018). In the past, efforts to increase food production through significant investment in agricultural research including crop improvement have benefited farmers by increasing yields and reducing losses, and have helped consumers by lowering food prices (Pingali 2012, 2015; Alston and Pardey 2014; Popp et al. 2013). Public spending on agriculture research and development has been more effective at raising sustainable agriculture productivity than irrigation or fertiliser subsidies (OECD 2018).Yet, on average between 2015 and 2017, governments spent only around 14% of total agricultural support on services which includes physical and knowledge infrastructure, transport and ICT.

In terms of increasing food availability and supply, producer support, including policies mandating subsidies or payments, have been used to boost production of certain commodities or protect ecosystem services. Incentives can distort markets and farm business decisions in both negative and positive ways. For example, the European Union promotes meat and dairy production through voluntary coupled direct payments. These do not yet internalise external damage to climate, health, and groundwater (Velthof et al. 2014; Bryngelsson et al. 2016). In most countries, producer support has been declining since the mid-1990s (OECD 2018). Yet new evidence indicates that a government policy supporting producer subsidy could encourage farmers to adopt new technologies and reduce GHG reductions in agriculture (*medium evidence, high agreement).* However, this will require large capital (Henderson 2018). Since a 1995 reform in its Forest Law, Costa Rica has effectively used a combination of fuel tax, water tax, loans and agreements with companies, to pay landowners for agroforestry, reforestation and sustainable forest management (Porras and Asquith 2018).

Inland capture fisheries and aquaculture are an integral part of nutrition security and livelihoods for large numbers of people globally (Welcomme et al. 2010; Hall et al. 2013; Tidwell and Allan 2001; Youn et al. 2014) and are increasingly vulnerable to climate change and competing land and water use (Allison et al. 2009; Youn et al. 2014). Future production may increase in some high-latitude regions (*low confidence*) but production is likely to decline in low latitude regions under future warming (*high confidence*)(Brander and Keith 2015; Brander 2007). However over-exploitation and degradation of rivers has resulted in a decreasing trend in contribution of capture fisheries to protein security in comparison to managed aquaculture (Welcomme et al. 2010). Aquaculture however competes for land and water resources with many negative ecological and environmental impacts (Verdegem and Bosma 2009; Tidwell and Allan 2001). Inland capture fisheries are undervalued in national and regional food security, ecosystem services and economy, are data deficient and are neglected in terms of supportive policies at national levels and absent in Sustainable Development Goals (Cooke et al. 2016; Hall et al. 2013; Lynch et al. 2016). Revival of sustainable capture fisheries and converting aquaculture to environmentally less damaging management regimes is likely to succeed by investment in recognition of their importance, improved valuation and assessment, secure tenure and adoption of social, ecological and technological guidelines besides upstream-downstream river basin cooperation and maintenance of ecological flow regimes in rivers (Youn et al. 2014; Mostert et al. 2007; Ziv et al. 2012; Hurlbert and Gupta 2016; Poff et al. 2003; Thomas 1996; FAO 2015a).

Extension services, and policies supporting agricultural extension systems, are also critical. Smallholder farmer-dominated agriculture is currently the backbone of global food security in the developing world. Without education and incentives to manage land and forest resources in a manner that allows regeneration of both the soils and wood stocks, smallholder farmers tend to generate income through inappropriate land management practices, engage in agricultural production on unsuitable land and use fertile soils, timber and firewood for brick production and construction and secondly engage in charcoal production (deforestation) as a coping mechanism (increasing income) against food deficiency (Munthali and Murayama 2013). Through extension services, governments can play a proactive role in providing information on climate and market risks, animal and plant health. Farmers with greater access to extension training retain more crop residues for mulch on their fields (Jaleta et al. 2015, 2013; Baudron et al. 2014).

Food security cannot be achieved by increasing food availability alone. Policy instruments, which increase access to food at the household level, include safety net programming and universal basic income. The graduation approach, developed and tested over the past decade using randomised control trials in six countries, has lasting positive impacts on income, as well as food and nutrition security (Banerjee et al. 2015; Raza and Poel 2016) (*robust evidence*, *high agreement*). The graduation approach layers and integrates a series of interventions designed to help the poorest: consumption support in the form of cash or food assistance, transfer of an income generating asset (such as a livestock) and training on how to maintain the asset, assistance with savings and coaching or mentoring over a period of time to reinforce learning and provide support. Due to its success, the graduation approach is now being scaled up, now used in over 38 countries and included by an increasing number of governments in social safety-net programs (Hashemi, S.M. and de Montesquiou 2011).

At the national and global level, food price and trade policies impact access to food. Fiscal policies, such as taxation, subsidies, or tariffs, can be used to regulate production and consumption of certain foods and can affect environmental outcomes. In Denmark, tax on saturated fat content of food adopted to encourage healthy eating habits accounted for 0.14% of total tax revenues between 2011 and 2012 (Sassi et al. 2018). A global tax on GHG emissions for example has large mitigation potential and will generate tax revenues, but may also result in large reductions in agricultural production (Henderson 2018). Consumer-level taxes on GHG intensive food may be applied to address competitiveness issues between different countries, if some countries use taxes while others do not. However, increases in prices might impose disproportionate financial burdens on low-income households, and may not be publicly acceptable. A study examining the relationship between food prices and social unrest found that between 1990 and 2011, food price increases have led to increases in social unrest, whereas food price volatility has not been associated with increases in social unrest (Bellemare 2015).

Interventions that allow people to maximise their productive potential while protecting the ecosystem services may not ensure food security in all contexts. Some household land holdings are so small that self-sufficiency is not possible (Venton 2018). Value chain development has in the past increased farm income but delivered fewer benefits to vulnerable consumers (Bodnár et al. 2011). Ultimately, a mix of production activities and consumption support is needed. Consumption support can be used to help achieve the second important element of food security – access to food.

Agricultural technology transfer can help optimise food and nutrition security (see 7.4.4.3). Policies that affect agricultural innovation span sectors and include “macro-economic policy-settings; institutional governance; environmental standards; investment, land, labor and education policies; and incentives for investment, such as a predictable regulatory environment and robust intellectual property rights”.

The scientific community can partner across sectors and industries for better data sharing, integration, and improved modelling and analytical capacities (Janetos et al. 2017; Lunt et al. 2016). To better predict, respond to and prepare for concurrent agricultural failures, and gain a more systematic assessment of exposure to agricultural climate risk, large data gaps need to be filled, as well as gaps in empirical foundation and analytical capabilities (Janetos et al. 2017; Lunt et al. 2016). Data required include global historical datasets, many of which are unreliable, inaccessible, or not available (Maynard 2015; Lunt et al. 2016). Participation in co-design for scenario planning can build social and human capital while improving understanding of food system risks and creating innovative ways for collectively planning for a more equitable and resilient food system (Himanen et al. 2016; Meijer et al. 2015; Van Rijn et al. 2012).

Demand management for food, including promoting healthy diets, reducing food loss and waste, is covered in Chapter 5. There is a gap in knowledge regarding what policies and instruments support demand management. There is *robust evidence and robust agreement* that changes in household wealth and parents' education can drive changes in diet and improvements in nutrition (Headey et al. 2017). Bangladesh has managed to sustain a rapid reduction in the rate of child undernutrition for at least two decades. Rapid wealth accumulation and large gains in parental education are the two largest drivers of change (Headey et al. 2017). Educating consumers, and providing affordable alternatives, will be critical to changing unsustainable food use habits relevant to climate change.

7.4.2.2. Policies to secure social protection

There is *medium evidence and high agreement* from all regions of the world that safety nets and social protection schemes can provide stability which prevents and reduces abject poverty (Barrientos 2011; Hossain 2018) (Cook and Pincus 2015; Huang and Yang 2017; Slater 2011; Sparrow et al. 2013; Rodriguez-Takeuchi and Imai 2013; Bamberg et al. 2018) in the face of climatic stressors and land change (Davies et al. 2013; Cutter et al. 2012b; Pelling 2011; Ensor 2011).

The World Bank estimates that globally social safety net transfers have reduced the absolute poverty gap by 45% and the relative poverty gap by 16% (World Bank 2018). Adaptive social protection builds household capacity to deal with shocks as well as the capacity of social safety nets to respond to shocks. For low-income communities reliant on land and climate for their livelihoods and wellbeing, social protection provides a way for vulnerable groups to manage weather and climatic variability and deteriorating land conditions to household income and assets *(robust evidence, high agreement)*(Baulch et al. 2006; Barrientos 2011; Harris 2013; Fiszbein et al. 2014; Kiendrebeogo et al. 2017; Kabeer et al. 2010; FAO 2015b; Warner et al. 2018)(World Bank 2018).

Life cycle approaches to social protection are one approach, which some countries (such as Bangladesh) are using when developing national social protection policies. These policies acknowledge that households face risks across the life cycle from which they need to be protected. If shocks are persistent, or occur numerous times, then policies can address concerns of a more structural nature (Glauben et al. 2012). Barrett (2005), for example, distinguishes between the role of safety nets (which include programs such as emergency feeding programs, crop or unemployment insurance, disaster assistance, etc.) and cargo nets (which include land reforms, targeted microfinance, targeted school feeding program, etc.). While the former prevents non-poor and transient poor from becoming chronically poor, the latter is meant to lift people out of poverty by changing societal or institutional structures. The graduation approach has adopted such systematic thinking with successful results (Banerjee et al. 2015).

Social protection systems can provide buffers against shocks through vertical or horizontal expansion, piggybacking on pre-established programmes, aligning social protection and humanitarian systems or refocusing existing resources (Wilkinson et al. 2018; O'Brien, C.O., Scott, Z., Smith, G., Barca, V., Kardan, A., Holmes, R. Watson 2018); (Jones and Presler-Marshall 2015). There is increasing evidence that forecast-based financing, linked to a social protection, can be used to enable anticipatory actions based on forecast triggers and guaranteed funding ahead of a shock (Jjemba et al. 2018). Accordingly scaling up social protection based on an early warning could enhance timeliness, predictability and adequacy of social protection benefits (Kuriakose et al. 2012; Costella et al. 2017a;

Wilkinson et al. 2018; O'Brien, C.O., Scott, Z., Smith, G., Barca, V., Kardan, A., Holmes, R. Watson 2018).

Countries at high-risk of natural disasters often have lower safety net coverage percent (World Bank 2018), and there is *medium evidence and medium agreement* that those countries with few financial and other buffers have lower economic and social performance (Cutter et al. 2012b; Outreville 2011a). Social protection systems have also been seen as an unaffordable commitment of public budget in many developing and low-income countries (Harris 2013). National systems may be disjointed and piecemeal, and subject to cultural acceptance and competing political ideologies (Niño-Zarazúa et al. 2012). For example, Liberia and Madagascar each have five different public works programs, each with different donor organisations and different implementing agencies (Monchuk 2014). These implementation shortcomings mean that positive effects of social protection systems might not be robust enough to shield recipients completely against the impacts of severe shocks or from long-term losses and damages from climate change (*limited evidence*,*high agreement*) (Davies et al. 2009; Umukoro 2013; Béné et al. 2012; Ellis et al. 2009).

There is increasing support for establishment of public-private safety nets to address climate related shocks which are augmented by proactive preventative (adaptation) measures and related risk transfer instruments that are affordable to the poor (Kousky et al. 2018b). Studies suggest that adaptive capacity of communities have improved with regard to climate variability like drought when ex-ante tools including insurance have been employed holistically; providing insurance in combination with early warning and institutional and policy approaches that aim to reduce livelihood and food insecurity as well as strengthen social structures(Shiferaw et al. 2014; Lotze-Campen and Popp 2012). Bundling insurance with early warning and seasonal forecasting can reduce the cost of insurance premiums (Daron and Stainforth 2014). The regional risk insurance scheme Africa Risk Capacity has the potential to significantly reduce the cost of insurance premiums (Siebert 2016) while bolstering contingency planning against food insecurity.

Work-for-insurance programs applied in the context of social protection have been shown to improve livelihood and food security in Ethiopia (Berhane 2014; Mohmmed et al. 2018) and Pakistan . The R4 Rural Resilience Program in Ethiopia is a widely cited example of a program that serves the most vulnerable and includes aspects of resource management, and access by the poor to financial services including insurance and savings (Linnerooth-bayer et al. 2018). Weather index insurance (such as index based crop insurance) is being presented to low-income farmers and pastoralists in developing countries (e.g., Ethiopia, India, Kazakhstan, China, South Asia) to complement informal risk sharing, reducing the risk of lost revenue associated with variations in crop yield, and provide an alternative to classic insurance (Bogale 2015a; Conradt et al. 2015; Dercon et al. 2014; Greatrex et al. 2015; Mcintosh et al. 2013). The ability of insurance to contribute to adaptive capacity depends on the overall risk management and livelihood context of households — studies find that rain fed agriculturalists and foresters with more years of education and credit but limited off-farm income are more willing to pay for insurance than households who have access to remittances (such as from family members who have migrated)(Bogale 2015a; Gan et al. 2014; Hewitt et al. 2017; Nischalke 2015). In Europe, modelling suggests that insurance incentives such as vouchers would be less expensive than total incentivised damage reduction and may reduce residential flood risk by 12% in Germany and 24% by 2040 (Hudson et al. 2016).

7.4.3. Policies Responding to Climate Related Extremes

7.4.3.1. Risk Management Instruments

Risk management addressing climate change has broadened to include mitigation, adaptation and disaster preparedness in a process of risk management through instruments facilitating contingency and cross sectoral planning (Hurlimann and March 2012; Oels 2013), social community planning, and strategic, long term planning (Serrao-Neumann et al. 2015a). A comprehensive consideration integrates principles from informal support mechanisms to enhance formal social protection programming (Mobarak and Rosenzweig 2013; Stavropoulou et al. 2017) such that the social safety net, disaster risk management, and climate change adaptation are all considered to enhance livelihoods of the chronic poor (see char dwellers and recurrent floods in Jamuna and Brahmaputra basins of Bangladesh (Awal 2013) (see also 7.4.7). Iterative risk management is an on-going process of assessment, action, reassessment and response (Mochizuki et al. 2015) (see 7.5.2 and 7.4.7.2).

Important elements of risk planning include education, creation of hazard and risk maps; important elements of predicting include hydrological and meteorological monitoring to forecast weather, seasonal climate forecasts, aridity, flood and extreme weather; effective responding requires robust communication systems that pass on information to enable response (Cools et al. 2016).

Gauging effectiveness of policy instruments is challenging. Timescale may influence outcomes. To evaluate effectiveness researchers, program managers and communities strive to develop consistency, comparability, comprehensiveness and coherence in their tracking. In other words, practitioners utilise a consistent and operational conceptualisation of adaptation; focus on comparable units of analysis; develop comprehensive datasets on adaptation action; and be coherent with our understanding of what constitutes real adaptation (Ford and Berrang-Ford 2016). Increasing the use of systematic reviews or randomised evaluations may also be helpful (Alverson and Zommers 2018).

Many risk management policy instruments are referred to by the International Organization of Standardization which lists risk management principles, guidelines, and frameworks for explaining the elements of an effective risk management program (ISO 2009). The standard provides practical risk management instruments and makes a business case for risk management investments (McClean et al. 2010). Insurance addresses impacts associated with extreme weather events (storms, floods, droughts, temperature extremes), but it can provide disincentives for reducing disaster risk at the local level through the transfer of risk spatially to other places or temporally to the future (Cutter et al. 2012b) and uptake is unequally distributed across regions and hazards (Lal et al. 2012). Insurance instruments (see 7.4.2 and 7.4.6) can take many forms (traditional indemnity based, market based crop insurance, property insurance), and some are linked to livelihoods sensitive to weather as well as food security (linked to social safety net programs) and ecosystems (coral reefs and mangroves). Insurance instruments can also provide a framework for risk signals to adaptation planning and implementation and facilitate financial buffering when climate impacts exceed current capabilities delivered through both public and private finance (Bogale 2015b; Greatrex et al. 2015; Surminski et al. 2016). A holistic consideration of all instruments responding to extreme impacts of climate change (drought, flood etc.) is required when assessing if policy instruments are promoting livelihood capitals and contributing to the resilience of people and communities (Hurlbert 2018b). This holistic consideration of policy instruments leads to a consideration of risk governance (see 7.6).

Early warning systems are critical policy instruments for protecting lives and property, adapting to climate change, and effecting adaptive climate risk management (*high confidence*) (Selvaraju 2011; Cools et al. 2016; Travis 2013; Henriksen et al. 2018; Seng 2013; Kanta Kafle 2017; Garcia and Fearnley 2012). Early warning systems exist at different levels and for different purposes including the FAO global Information and Early Warning System (GIEWS) on food and agriculture, USAID Famine, national and local extreme weather, species extinction, community based flood and landslide, and informal pastoral drought early warning systems (Kanta Kafle 2017). Medium term warning systems can identify areas of concern, hotspots of vulnerabilities and sensitivities, or critical zones of land degradation (areas of concern)(see chapter 6) critical to reduce risks over five to ten years (Selvaraju 2012). Early warning systems for dangerous climate shifts are emerging with considerations of rate of onset, intensity, spatial distribution and predictability. Growing research in the area is considering positive and negative lessons learned from existing hazard early warning systems including lead time and warning response (Travis 2013).

For effectiveness, communication methods are best adapted to local circumstances, religious and cultural based structures and norms, information technology, and local institutional capacity (Cools et al. 2016; Seng 2013). Considerations of governance or the actors and architecture within the socio-ecological system, is an important feature of successful early warning system development (Seng 2013). Effective early warning systems consider the critical links between hazard monitoring, risk assessment, forecasting tools, warning and dissemination (Garcia and Fearnley 2012). These effective systems incorporate local context by defining accountability, responsibility, acknowledging the importance of risk perceptions and trust for an effective response to warnings. Although increasing levels and standardisation nationally and globally is important, revising these systems through participatory approaches cognizant of the tension with technocratic approaches improves success (Cools et al. 2016; Henriksen et al. 2018; Garcia and Fearnley 2012).

7.4.3.2. Drought related risk minimising instruments

A more detailed review of drought instruments, and three broad policy approaches for responding to drought, is provided in Cross-chapter Box 5: Case study on policy drought in Chapter 3. Three broad approaches include: (1) early warning systems and response to the disaster of drought (through instruments such as disaster assistance or crop insurance); (2) disaster response ex-ante preparation (through drought preparedness plans); and (3) drought risk mitigation (proactive polices to improve water use efficiency, make adjustments to water allocation, funds or loans to build technology such as dugouts or improved soil management practices).

Drought plans are still predominantly reactive crisis management plans rather than proactive risk management and reduction plans. Reactive crisis management plans treat only the symptoms and are inefficient drought management practices. More efficient drought preparedness instruments are those that address the underlying vulnerability associated with the impacts of drought thereby building agricultural producer adaptive capacity and resilience (*high confidence*)(Cross-chapter Box 5: Case study on policy drought, chapter 3).

7.4.3.3. Fire related risk minimising instruments

There is *robust evidence and high agreement* that fire strategies need to be tailored to site specific conditions in an adaptive application that is assessed and reassessed over time (Dellasala et al. 2004; Rocca et al. 2014). Strategies for fire management include fire suppression, prescribed fire and mechanical treatments (such as thinning the canopy), and allowing wildfire with little or no active management (Rocca et al. 2014). Fire supression can degrade the effectiveness of forest fire management in the long run (Collins et al. 2013).

Different forest types have different fire regimes and require different fire management policies (Dellasala et al. 2004). For instance, Cerrado, a fire dependent savannah, utilises a fire management policy different than the fire suppression policy (Durigan and Ratter 2016). The choice of strategy depends on local considerations including land ownership patterns, dynamics of local meteorology, budgets, logistics, federal and local policies, tolerance for risk and landscape contexts. In addition there are trade-offs among the management alternatives and often no single management strategy will simultaneously optimise ecosystem services including water quality and quantity, carbon sequestration, or run off erosion prevention (Rocca et al. 2014).

7.4.3.4. Flood related risk minimising instruments

Flood risk management consists of command and control measures including spatial planning and engineered flood defences (Filatova 2014), financial incentive instruments issued by regional or national governments to facilitate cooperative approaches through local planning, enhancing community understanding and political support for safe development patterns and building standards, and regulations requiring local government participation and support for local flood planning (Burby and May 2009). However, Filatova (2014) found that if autonomous adaptation is downplayed, people are more likely to make land use choices that collectively lead to increased flood risks and leave costs to governments. Taxes and subsidies that do not encourage (and even counter) perverse behaviour (such as rebuilding in flood zones) are important instruments mitigating this cost to government. Flood insurance has been found to be maladaptive as it encourages rebuilding in flood zones (O'Hare et al. 2016)) and government flood disaster assistance negatively impacts average insurance coverage the following year (Kousky et al. 2018a). Modifications to flood insurance can counter perverse behaviour. One example is the provision of discounts on flood insurance for localities that undertake one of 18 flood mitigation activities including structural mitigation (constructing dykes, dames, flood control reservoirs), and non-structural initiaves such as point source control and watershed management efforts, education and maintenance of flood-related databases (Zahran et al. 2010). Flood insurance that provides incentives for flood mitigation, marketable permits and transferable development rights (see case study of Flood and Food Security in Section 7.6) instruments can provide price signals to stimulate autonomous adaptation, countering barriers of path dependency, and the time lag between private investment decisions and consequences (Filatova 2014). To build adaptive capacity, consideration needs to be made of policy instruments responding to flood including flood zone mapping, land use planning, flood zone building restrictions, business and crop insurance, disaster assistance payments, preventative instruments including environmental farm planning (including soil and water management (see Chapter 6)) and farm infrastructure projects, and recovery from debilitating flood losses ultimately through bankruptcy (Hurlbert 2018a). Non-structural measures have been found to advance sustainable development as they are more reversible, commonly acceptable and environmentally friendly (Kundzewicz 2002).

7.4.4. Policies Responding to GHG fluxes

7.4.4.1. GHG fluxes and climate change mitigation

Pathways reflecting current nationally stated mitigation ambitions as submitted under the Paris Agreement would not limit global warming to 1.5°C with no or limited overshoot, but instead result in a global warming of about 3°C by 2100 with warming continuing afterwords (IPCC 2018d). Reversing warming after an overshoot of .2°C or larger during this century would require deployment of CDR at rates and volumes that might not be achievable given considerable implementation challenges (IPCC 2018d). This significant gap (Höhne et al. 2017; Rogelj et al. 2016) creates a significant risk of global warming impacting land degradation, desertification, and food security (see 7.2;(IPCC 2018d). Action can be taken by 2030 adopting already known cost effective technology (United Nations Environment Programme 2017), improving the finance, capacity building, and technology transfer mechanisms of the UNFCCC, improving food security (listed by 73 nations in their NDCs) and nutritional security (listed by 25 nations) (Richards, M., Bruun, T.B., Campbell, B.M., Gregersen, L.E., Huyer 2015). UNFCCC Decision 1.CP21 reaffirmed the UNFCCC target that 'developed country parties provide USD 100 billion annually by 2020 for climate action in developing countries' (Rajamani 2011) and a new collective quantified goal above this floor is to be set taking into account the needs and priorities of developing countries (Fridahl and Linnér 2016).

Mitigation policy instruments to address this shortfall include financing mechanisms, carbon pricing, cap and trade or emissions trading, and technology transfer. While climate change is a global commons problem containing free-riding problems, cost effective international policies that insure countries get the most environmental benefit out of mitigation investments promote an international climate policy regime (Nordhaus 1999; Aldy and Stavins 2012). Carbon pricing instruments may provide an entry point for inclusion of agricultural appropriate carbon instruments. Models of cost efficient distribution of mitigation across regions and sectors typically employ a global uniform carbon price, but such treatment in the agricultural sector may impact food security (see 7.4.4.4).

One policy initiative to advance climate mitigation policy coherence (see 7.4.8) in this section is the phase out of subsidies for fossil fuel production. The G20 agreed in 2009, and the G7 agreed in 2016, to phase out these subsidies by 2025. Subsidies include lower tax rates or exemptions and rebates of taxes on fuels used by particular consumers (diesel fuel used by farming, fishing etc.), types of fuel, or how fuels are used. The OECD estimates the overall value of these subsides to be between USD 160–200 billion annually between 2010 and 2014 (OECD 2015). The phase out of fossil fuel subsidies has important economic, environmental and social benefits. Coady et al. (2017) estimate the economic and environmental benefits of reforming fossil fuel subsidies could be valued worldwide at USD 4.9 trillion in 2013, and USD 5.3 trillion in 2015. Eliminating subsidies could have reduced emissions by 21% and raised 4% of global GDP as revenue (in 2013) and improved social welfare (Coady et al. 2017).

Legal instruments addressing perceived deficiencies in climate change mitigation include human rights and liability. Developments in attribution science are improving the ability to detect human influence on extreme weather and Marjanac et al. (2017) argue this broadens the legal duty of government, business and others to manage foreseeable harms and may lead to more climate change litigation (Marjanac et al. 2017). Peel and Osofsky (2017) argue that courts are becoming increasingly receptive to employ human rights claims in climate change lawsuits (Peel and Osofsky 2017); citizen suits in domestic courts are not a universal phenomenon and even if unsuccessful, Estrin (2016) concludes they are important in underlining the high level of public concern.

7.4.4.2. Mitigation instruments

Similar instruments for mitigation could be applied to the land sector as in other sectors, including market-based measures such as taxes and cap and trade systems; as well as standards and regulations; subsidies and tax credits; information instruments and management tools; R&D investment; and voluntary compliance programmes, but few regions have implemented agricultural mitigation instruments(Cooper et al. 2013). Existing regimes focus on subsidies, grants and incentives, and voluntary offset programmes.

Market-based instruments

Although carbon pricing is recognised to be an important cost-effective instrument in a portfolio of climate policies (Aldy et al. 2010) (*high evidence, high agreement*), as yet no country is exposing their agricultural sector emissions to carbon pricing in any comprehensive way. A carbon tax, fuel tax, and carbon markets (cap and trade system or Emissions Trading Scheme (ETS), or baseline and credit schemes, and voluntary markets) are predominant policy instruments that implement carbon pricing. The advantage of carbon pricing is environmental effectiveness at relatively low cost (Baranzini et al. 2017; Fawcett et al. 2014) *(high evidence, high agreement)*. Furthermore, carbon pricing could be used to raise revenue to reinvest in public spending, either to help certain sectors transition to lower carbon systems, or to invest in public spending unrelated to climate change. Both of these options may make climate policies more attractive and enhance overall welfare (Siegmeier et al. 2018), but there is as yet no evidence of the effectiveness of emissions pricing in agriculture (Grosjean et al. 2018). There is however, a clear need for progress in this area as without effective carbon pricing, the mitigation potential identified in chapters 5 and 6 of this report will not be realised (Boyce 2018)(*high evidence, high agreement*).

The price may be set at the Social Cost of Carbon (the incremental impact of emitting an additional tonne of CO_2, or the benefit of slightly reducing emissions), but estimates of the SCC vary widely and are contested (Pezzey 2019) (*high evidence, high agreement*). An alternative to the SCC includes a pathways approaches that sets an emissions target and estimates the Carbon prices required to achieve this at the lowest possible cost (Pezzey 2019). Theoretically, higher costs throughout the entire economy result in reduction of carbon intensity as consumers and producers adjust their decisions in relation to prices corrected to reflect the climate externality (Baranzini et al. 2017).

Both carbon taxes and cap and trade systems can reduce emissions, but cap and trade systems are generally more cost effective (*medium evidence, high agreement*) (Haites 2018a). In both cases, the design of the system is critical to its effectiveness at reducing emissions (Bruvoll and Larsen 2004; (Lin and Li 2011)) (*high evidence*, *high agreement*). The trading system allows the achievement of emission reductions in the most cost-effective manner possible and results in a market and price on emissions that create incentives for the reduction of carbon pollution. The way allowances are allocated in a cap and trade system is critical to its effectiveness and equity. Free allocations can be provided to trade-exposed sectors such as agriculture either through historic allocations or output based; the choice of which has important implications (Quirion 2009). Output based allocations may be most suitable for agriculture also minimising leakage risk (see below) (Grosjean et al. 2018) (Quirion 2009). There is *medium evidence* and *high agreement* that properly designed, a cap and trade system can be a powerful policy instrument (Wagner 2013) and may collect more rents than a variable carbon tax (Siegmeier et al. 2018; Schmalensee and Stavins 2017).

In the land sector carbon markets are challenging to implement. Although several countries and regions have ETSs in place (for example the EU, Switzerland, the Republic of Korea, Quebec in Canada, California in the USA (Narassimhan et al. 2018)), none have included non-CO_2 (methane and nitrous oxide) emissions from agriculture. New Zealand is the only country currently considering ways to incorporate agriculture into its ETS (see Case Study on the New Zealand Emissions Trading Scheme).

Three main reasons explain the lack of implementation to date:

1. The large number of heterogeneous buyers and sellers, combined with the difficulties of monitoring, reporting and verification (MRV) of emissions from biological systems introduce potentially high levels of complexity (and transaction costs). Effective policies therefore depend on advanced MRV systems which are lacking in many (particularly developing) countries (Wilkes et al. 2017). This is discussed in more detail in the Case Study on the New Zealand Emissions Trading Scheme.

2. Adverse distributional consequences (Grosjean et al. 2018) (*medium evidence, high agreement*). Distributional issues depend, in part, on the extent that policy costs can be passed on to consumers, and there is *medium evidence* and *medium agreement* that social equity can be increased through a combination of non-market and market-based instruments (Haites 2018b).

3. Regulation, market-based or otherwise, adopted in only one jurisdiction and not elsewhere may result in ‘leakage’ or reduced effectiveness – where production relocates to weaker regulated regions, potentially reducing the overall environmental benefit. Although modelling studies indicate the possibility of leakage following unilateral agricultural mitigation policy implementation (e.g. Fellmann et al. 2018), there is no empirical evidence from the agricultural sector yet available. Analysis from other sectors shows an overestimation of the extent of carbon leakage in modelling studies conducted before policy implementation compared to evidence after the policy was implemented (Branger and Quirion 2014). Options to avoid leakage include border adjustments (emissions in non-regulated imports are taxed at the border, and payments made on products exported to non-regulated countries are rebated); differential pricing for trade-exposed products and; output based allocation (which effectively works as a subsidy for trade-exposed products). Modelling shows that border adjustments are the most effective at reducing leakage, but may exacerbate regional inequality (Böhringer et al. 2012) and through their trade-distorting nature may contravene WTO rules. The opportunity for leakage would be significantly reduced ideally through multi-lateral commitments (Fellmann et al. 2018) (*medium evidence, high agreement)* but could also be reduced through regional or bi-lateral commitments within trade agreements.

Case study: Including agriculture in the Emissions Trading Scheme in New Zealand

New Zealand has a high proportion of agricultural emissions at 49% (Ministry of the Environment 2018) - the next highest developed country agricultural emitter is Ireland at around 32% (EPA 2018) - and is considering to incorporate agricultural non-CO_2 gases into the existing national ETS. In the original design of the ETS in 2008, agriculture was intended to be included from 2013, but successive Governments deferred the inclusion (Kerr and Sweet 2008) due to concerns about competitiveness, lack of mitigation options and the level of opposition from those potentially affected (Cooper and Rosin 2014). Now though, as the country's agricultural emissions are 12% above 1990 levels, and the country's total gross emissions have increased 19.6% above 1990 levels (New Zealand Ministry for the Environment 2018), there is a recognition that without any targeted policy for agriculture, only 52% of the country's emissions face any substantive incentive to mitigate (Narassimhan et al. 2018). Including agriculture in the ETS is one option to provide incentives for emissions reductions in that sector. Other options are discussed in Section 7.4.4. Although some producer groups raise concern that including agriculture will place New Zealand producers at a disadvantage compared with their international competitors who do not face similar mechanisms (New Zealand Productivity Commission 2018), there is generally greater acceptance of the need for climate policies for agriculture.

The inclusion of non-CO_2 emissions from agriculture within an ETS is potentially complex however, due to the large number of buyers and sellers if obligations are placed at farm level, and different choices of how to estimate emissions from biological systems in cost-effective ways. New Zealand is currently investigating practical and equitable approaches to include agriculture through advice being provided by the Interim Climate Change Committee (ICCC 2018). Main questions centre around the point of obligation for buying and selling credits, where trade-offs have to be made between providing incentives for behaviour change at farm level and the cost and complexity of administering the scheme (Agriculture Technical Advisory Group 2009; Kerr and Sweet 2008). The two potential points of obligation are at the processor level or at the individual farm level. Setting the point of obligation at the processor level means that farmers would face limited incentive to change their management practices, unless the processors themselves rewarded farmers for lowered emissions. Setting it at the individual farm level would provide a direct incentive for farmers to adopt mitigation practices, however the reality of having thousands of individual points of obligation would be administratively complex and could result in high transaction costs (Beca Ltd 2018).

Monitoring, reporting and verification (MRV) of agricultural emissions presents another challenge especially if emissions have to be estimated at farm level. Again, trade-offs have to be made between accuracy and detail of estimation method and the complexity, cost and audit of verification (Agriculture Technical Advisory Group 2009).

The ICCC is also exploring alternatives to an ETS to provide efficient abatement incentives (ICCC 2018).

Some discussion in New Zealand also focuses on a differential treatment of methane compared to nitrous oxide, Methane is a short-lived gas with a perturbation lifetime of twelve years in the atmosphere; nitrous oxide on the other hand is a long-lived gas and remains in the atmosphere for 114 years (Allen et al. 2016). Long-lived gases have a cumulative and essentially irreversible effect on the climate (IPCC 2014b) so their emissions need to reduce to net-zero in order to avoid climate change. Short-lived gases however could potentially be reduced to a certain level and then stabilised and would not contribute further to warming, leading to suggestions of treating these two gases separately in the ETS or alternative policy instruments, possibly setting different budgets and targets for each (New Zealand Productivity Commission 2018). Reisinger et al. (2013) demonstrate that different metrics can have important implications globally and potentially at national and regional scales on the costs and levels of abatement.

While the details are still being agreed on in New Zealand, almost 80% of NDCs committed to action on mitigation in agriculture (FAO 2016), so countries will be looking for successful examples.

Australia's Emissions Reduction Fund, and the preceding Carbon Farming Initiative, are an example of a baseline-and-credit scheme, which set an emissions intensity baseline and creates credits for activities that generate emissions below the baseline, effectively a subsidy (Freebairn 2016). It is a voluntary scheme, and has potential to create real and additional emission reductions through projects reducing emissions and sequestering carbon (Verschuuren 2017) *(low evidence, low agreement*). Key success factors in the design of such an instrument are policy-certainty for at least ten to twenty years, regulation that focuses on projects and not uniform rules, automated systems for all phases of the projects, and a wider focus of the carbon farming initiative on adaptation, food security, sustainable farm business, and creating jobs (Verschuuren 2017). A recent review highlighted the issue of permanence and reversal, and recommended that projects detail how they will maintain carbon in their projects and deal with the risk of fire.

7.4.4.3. Technology transfer and land use sectors

Technology transfer has been part of the UNFCCC process since its inception and is a key element of international climate mitigation and adaptation efforts under the Paris Agreement. The IPCC definition of Technology transfer includes transfer of knowledge and technological cooperation (see Glossary) and can include modifications to suit local conditions and/or integration with indigenous technologies (Metz et al. 2000). This definition suggests greater heterogeneity in the applications for climate mitigation and adaptation, especially in land use sectors where indigenous knowledge may be important for long-term climate resilience Nyong et al. (2007). For land use sectors, the typical reliance on trade and patent data for empirical analyses is generally not feasible as the "technology" in question is often related to resource management and is neither patentable nor tradable (Glachant and Dechezleprêtre 2017) and ill-suited to provide socially beneficially innovation for poorer farmers in developing countries (Lybbert and Sumner 2012; Baker, Dean; Jayadev, Arjun; Stiglitz 2017).

Technology transfer has contributed to emissions reductions (*medium confidence*). A detailed study for nearly 4000 Clean Development Mechanism (CDM) projects showed that 39% of projects had a stated and actual technology transfer component, accounting for 59% of emissions reductions; however, the more land-intensive projects (e.g., afforestation, bioenergy) showed lower percentages (Murphy et al. 2015). Bioenergy projects that rely on agricultural residues offer substantially more development benefits than those based on industrial residues from forests (Lee and Lazarus 2013). Energy projects tended to have a greater degree of technology transfer under the CDM compared to non-energy projects (Gandenberger et al. 2016). However, longer-term cooperation and collaborative R&D approaches to technology transfer will be more important in land use sectors (compared to energy or industry) due to the time needed for improved resource management and interaction between researchers, practitioners and policy-makers. These approaches offer longer-term technology transfer that is more difficult to measure compared to specific cooperation projects; empirical research on the effects of R&D collaboration could help to avoid the "one-policy-fits-all" approach (Ockwell et al. 2015).

There is increasing recognition of the role of technology transfer in climate adaptation, but in the land use sector there are inherent adoption challenges specific to adaptation, due to uncertainties arising from changing climatic conditions, agricultural prices, and suitability under future conditions (Biagini et al. 2014). Engaging the private sector is important, as adoption of new technologies can only be replicated with significant private sector involvement (Biagini and Miller 2013).

7.4.4.4. International Cooperation under the Paris Agreement

New cooperative mechanisms under the Paris Agreement illustrate the shift away from the Kyoto Protocol's emphasis on obligations of developed country Parties to pursue investments and technology transfer, to a more pragmatic, decentralised and collaborative approach (Savaresi 2016; Jiang et al. 2017). These approaches can effectively include any combination of measures or instruments related to adaptation, mitigation, finance, technology transfer and capacity-building, which could be of particular interest in land use sectors where such aspects are more intertwined than in energy or industry sectors. Article 6 sets out several options for international cooperation (Gupta and Dube 2018).

The close relationship between emission reductions, adaptive capacity, food security and other sustainability and governance objectives in the land sectors means that Article 6 could bring co-benefits that increase its attractiveness and the availability of finance, while also bringing risks that need to be monitored and mitigated against, such as uncertainties in measurements and the risk of non-permanence (Thamo and Pannell 2016; Olsson et al. 2016; Schwartz et al. 2017). There has been progress in accounting for land-based emissions, mainly forestry and agriculture (*medium evidence, low agreement*), but various challenges remain (Macintosh 2012; Pistorius et al. 2017; Krug 2018).

Like the Clean Development Mechanism (CDM) and other existing carbon trading mechanisms, participation in Article 6.2 and 6.4 of the Paris Agreement requires certain institutional and data management capacities in the land sector to effectively benefit from the cooperation opportunities (Totin et al. 2018). While the rules for the implementation of the new mechanisms are still under development, lessons from REDD+ may be useful, which is perceived as more democratic and participative than the CDM (Maraseni and Cadman 2015). Experience with REDD+ programs emphasise the necessity to invest in "readiness" programs that assist countries to engage in strategic planning and build management and data collection systems to develop the capacity and infrastructure to participate in REDD+ (Minang et al. 2014). The overwhelming majority of countries (93%) cite weak forest sector governance and institutions in their applications for REDD+ readiness funding (Kissinger et al. 2012). Technology transfer for advanced remote sensing technologies that help to reduce uncertainty in monitoring forests helps to achieve REDD+ "readiness" (Goetz et al. 2015).

As well as new opportunities for finance and support, the Paris cooperation mechanisms and the associated roles for technology transfer bring new challenges, particularly in reporting, verifying and accounting in land use sectors. Since developing countries must now achieve, measure and communicate emission reductions, they now have value for both developing and developed countries in achieving their NDCs, but reductions cannot be double-counted (i.e., towards multiple NDCs). All countries have to prepare and communicate NDCs, and many countries have included in their NDCs either economy-wide targets that include the land use sectors, or specific targets for the land use sectors. The Katowice climate package clarifies that all Parties have to submit 'Biennial Transparency Reports' from 2024 onwards using common reporting formats, following most recent IPCC Guidelines (use of the 2013 Supplement on Wetlands is encouraged), identifying key categories of emissions, ensuring time-series consistency, and providing completeness and uncertainty assessments as well as quality control (UNFCCC 2018a; Schneider and La Hoz Theuer 2019). In total, the ambiguity in how countries incorporate land use sectors into their NDC is estimated to lead to an uncertainty of more than 2 $GtCO_2$ in 2030 (Fyson and Jeffery 2018). Uncertainty is lower if the analysis is limited to countries that have provided separate land use sector targets in their NDCs (Benveniste et al. 2018).

7.4.5. Policies Responding to Desertification and Degradation – Land Degradation Neutrality (LDN)

Land degradation neutrality (LDN) (SDG Target 15.3), evolved from the concept of Net Zero Land Degradation, which was introduced by the UNCCD to promote sustainable land management (Kust et al. 2017; Stavi and Lal 2015; Chasek et al. 2015). Neutrality here implies no net loss of the land-based natural resource and ecosystem services relative to a baseline or a reference state (UNCCD 2015; Kust et al. 2017; Easdale 2016; Cowie et al. 2018a; Stavi and Lal 2015; Grainger 2015; Chasek et al. 2015). Land degradation neutrality can be achieved by reducing the rate of land degradation (and concomitant loss of ecosystem services) and increasing the rate of restoration and rehabilitation of degraded or desertified land. Therefore, the rate of global land degradation is not to exceed that of land restoration in order to achieve land degradation neutrality goals (adopted as national platform for actions by > 100 countries)(Stavi and Lal 2015; Grainger 2015; Chasek et al. 2015; Cowie et al. 2018a; Montanarella 2015). Achieving land degradation neutrality would decrease the environmental footprint of agriculture, while supporting food security and sustaining human wellbeing (UNCCD 2015; Safriel 2017; Stavi and Lal 2015; Kust et al. 2017).

Response hierarchy - avoiding, reducing and reversing land degradation - is the main policy response (Chasek et al. 2019, Wonder and Bodle 2019, Cowie et al. 2018, Orr et al. 2017). The LDN response hierarchy encourages through regulation, planning and mangagement instruments, the adoption of diverse measures to avoid, reduce and reverse land degradation in order to achieve LDN (Cowie et al. 2018b; Orr et al. 2017).

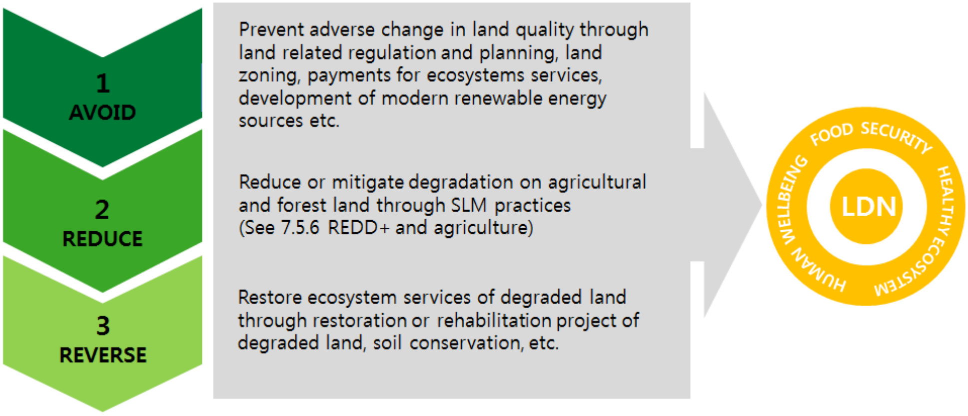

Figure 7.4 LDN response hierarchy
Source: Adapted from (Liniger et al. 2019; UNCCD/Science-Policy-Interface 2016)

Chapter 3 categorised policy responses into two categories; (1) avoiding, reducing and reversing it through sustainable land management; and (2) providing alterative livelihoods with economic diversification. Land degradation neutrality could be achieved through planned effective actions, particularly by motivated stakeholders those who play an essential role in a land-based climate change adaptation (Easdale 2016; Qasim et al. 2011; Cowie et al. 2018a; Salvati and Carlucci 2014). Human activities impacting the sustainability of drylands is a key consideration in adequately reversing degradation through restoration or rehabilitation of degraded land (Easdale 2016; Qasim et al. 2011; Cowie et al. 2018a; Salvati and Carlucci 2014).

LDN actions and activities play an essential role for a land-based approach to climate change adaptation (UNCCD 2015). Policies responding to degradation and desertification include improving market access, gender empowerment, expanding access to rural advisory services, strengthening land tenure security, payments for ecosystem services, decentralised natural resource management, investing into research and development, investing into monitoring of desertification and desert storms, developing modern renewable energy sources, investing into modern renewable energy sources, and developing and strengthening climate services. Policy supporting economic diversification include investing in irrigation, expanding agricultural commercialisation, and facilitating structural transformations in rural economies. (Chapter 3). Policies and actions also include promoting local and indigenous knowledge, soil conservation, agroforestry, crop-livestock interactions as an approach to manage land degradation, and forest based activities such as afforestation, reforestation, and changing forest management (Chapter 4). Measures identified for achievement of LDN include; effective financial mechanisms (for implementation of land restoration measures and the long-term monitoring of progress), parameters for assessing land degradation, detailed plans with quantified objectives and timelines (Kust et al. 2017; Sietz et al. 2017; Cowie et al. 2018a; Montanarella 2015; Stavi and Lal 2015).

Implementing the international LDN target into national policies has been a challenge (Cowie et al. 2018a; Grainger 2015) as baseline land degradation or desertification information is not always available (Grainger 2015) and challenges exist in monitoring LDN as it is a dynamic process (Sietz et al. 2017; Grainger 2015; Cowie et al. 2018a). Wunder and Bodle (2019) propose that LDN be implemented and monitored through indicators at the national level. Effective implementation of global LDN will be supported by integrating lessons learned from existing programs designed for other environmental objectives and closely coordinate LDN activities with actions for climate change adaptation and mitigation at both global and national levels (*high confidence*) (Stavi and Lal 2015; Grainger 2015).

7.4.6. Policies Responding to Land Degradation

7.4.6.1. Land Use Zoning

Land use zoning divides a territory (including local, sub-regional or national) into zones with different rules and regulations for land use (mining, agriculture, urban development etc.), management practices and land cover change (Metternicht 2018). While the policy instrument is zoning ordinances, the process of determining these regulations is covered in integrated land use planning (See 7.6.2). Urban zoning can guide new growth in urban communities outside current and forecasted hazard areas, assist relocating existing dwellings to safer sites and manage postevent redevelopment in ways to reduce future vulnerability (Berke and Stevens 2016). Holistic integration of climate mitigation and adaptation are interdependent and can be implemented by restoring urban forests, improving parks (Brown 2010; Berke and Stevens 2016). Zoning ordinances can contribute to sustainable land management through protection of natural capital by preventing or limiting vegetation clearing, avoiding degradation of planning for rehabilitation of degraded land or contaminated sites, promoting conservation and enhancement of ecosystems and ecological corridors (Metternicht 2018; Jepson and Haines 2014). Zoning ordinaces can also encourage higher density development, mixed use, local food production, encourage transportation alternatives (bike paths and transit oriented development), preserve a sense of place, and increase housing diversity and affordability (Jepson and Haines 2014). Conservation planning varies by context and may include one or several adaptation approaches including protecting current patterns of biodiversity, large intact natural landscapes, and geophysical settings. Conservation planning may also maintain and restore ecological connectivity, identify and manage areas that provide future climate space for species expected to be displaced by climate change, and identify and protect climate refugia (Stevanovic et al. 2016; Schmitz et al. 2015).

Anguelovski et al. (2016) studied land use interventions in eight cities in the global north and south and concluded that historic trends of socioeconomic vulnerability can be reinforced which could be avoided with a consideration of the distribution of adaptation benefits and prioritising beneficial outcomes for disadvantaged and vulnerable groups when making future adaptation plans. Concentration of adaptation resources within wealthy business districts creating ecological enclaves exacerbated climate risks elsewhere and building of climate adaptive infrastructure such as sea walls or temporary flood barriers occurred at the expense of underserved neighbourhoods (Anguelovski et al. 2016a).

7.4.6.2. Conserving biodiversity and ecosystem services

There is *limited evidence but high agreement* that ecosystem-based adaptation (biodiversity, ecosystem services, and nature’s contribution to people (see chapter 6)) and incentives for ecosystem services (including PES) play a critical part of an overall strategy to help people adapt to the adverse effects of climate change on land (UNEP 2009) (Bonan 2008; Millar et al. 2007; Thompson et al. 2009).

Ecosystem based adaptation can promote socio-ecological resilience by enabling people to adapt to the impacts of climate change on land and reduce their vulnerability (Ojea 2015). Ecosystem based adaptation can promote nature conservation while alleviating poverty and even provide co-benefits by removing greenhouse gas (Scarano 2017) and protecting livelihoods (Munang et al. 2013). For example, mangroves provide diverse ecosystem services such as carbon storage, fisheries, non-timber forest products, erosion protection, water purification, shore-line stabilisation and also regulate storm surge and flooding damages, thus enhancing resilience and reducing climate risk from extreme events such as cyclones (Rahman, M.M., Khan, M.N.I., Hoque, A.K.F., Ahmed 2014; Donato et al. 2011; Das and Vincent 2009; Ghosh et al. 2015; Ewel et al. 1998).

There has been considerable increase in the last decade of payments for ecosystem services (PES), or programmes that exchange value for land management practices intended to ensure ecosystem services (Salzman et al. 2018; Yang and Lu 2018; Barbier 2011). However, there is a deficiency in comprehensive and reliable data concerning PES' impact on ecosystems, human well-being, their efficiency, and effectiveness (Pynegar et al. 2018; Reed et al. 2014; Salzman et al. 2018; Barbier 2011; Yang and Lu 2018). While some studies assess ecological effectiveness and social equity, fewer assess economic efficiency (Yang and Lu 2018). Part of the challenge surrounds the fact that the majority of ecosystem services are not marketed, so determining how changes in ecosystems structures, functions and processes influence the quantity and quality of ecosystem service flows to people is challenging (Barbier 2011). PES include agri-environmental targeted outcome based payments, but challenges exist in relation to scientific uncertainty, pricing, timing of payments, increasing risk to land managers, World Trade Organization compliance, and barriers of land management and scale (Reed et al. 2014).

PES is contested (Wang and Fu 2013; Czembrowski and Kronenberg 2016) (Perry 2015) for four reasons: (1) understanding and resolving trade-offs between conflicting groups of stakeholders (Wam et al. 2016) (Matthies et al. 2015); (2) knowledge and technology capacity (Menz et al. 2013); (3) challenges integrating PES with economic and other policy instruments (Ring and Schröter-Schlaack 2011; Tallis et al. 2008)(Elmqvist et al. 2003; Albert et al. 2014); and (4) top down climate change mitigation initiatives which are still largely carbon centric with limited opportunities for decentralised ecological restoration at local and regional scales (Vijge and Gupta 2014).

These challenges and contestations can be resolved with the participation of people in establishing PES thereby addressing trust issues, negative attitudes, and resolving trade-offs between issues (such as retaining forests that consume water versus the provision of run off, or balancing payments to providers versus cost to society) (Sorice et al. 2018; Matthies et al. 2015). Similarly, a 'co-constructive' approach is used involving a diversity of stakeholders generating policy relevant knowledge for sustainable management of biodiversity and ecosystem services at all relevant spatial scales, by the current Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) initiative (Díaz et al. 2015). Invasive species are also best identified and managed with the participation of people through collective decisions, coordinated programs, and extensive research and outreach to address their complex social-ecological impacts (Wittmann et al. 2016; Epanchin-Niell et al. 2010).

Ecosystem restoration with co-benefits for diverse ecosystem services can be achieved through passive restoration, passive restoration with protection and active restoration with planting (Birch et al. 2010; Cantarello et al. 2010). Taking into account costs of restoration and co-benefits from bundles of ecosystem services (carbon, tourism, timber), the benefit cost ratio of active restoration and passive restoration with protection was always less than 1, suggesting that financial incentives would be required. Passive restoration was the most cost-effective with BCR was generally between 1 and 100 for forest, grassland and shrubland restoration (TEEB 2009; Cantarello et al. 2010). Passive restoration is generally more cost-effective but there is a danger that it could be confused with abandoned land in the absence of secure tenure and long time period (Zahawi et al. 2014). Net Social Benefits of degraded land restoration in dry regions range from about 200–700 USD per hectare (Cantarello et al., 2010). Investments in active restoration could benefit from analyses of past land use, the natural resilience of the ecosystem, and the specific objectives of each project (Meli et al. 2017). One successful example is the Working for Water initiative in South Africa that linked restoration through removal of invasie species and enhancing water security(Milton et al. 2003).

Forest, water and energy cycle interactions and teleconnections such as contribution to rainfall potentially (2.5.4)(Aragão 2012; Ellison et al. 2017; Paul et al. 2018; Spracklen et al. 2012) provide a foundation for achieving forest-based adaptation and mitigation goals. They are however poorly integrated in policy and decision making including PES.

7.4.6.3. Standards and certification for sustainability of biomass and land use sectors

During the past two decades, standards and certification have emerged as important sustainability and conservation instruments for agriculture, forestry, bioenergy, land use management and bio-based products (Lambin et al. 2014; Englund and Berndes 2015; Milder et al. 2015; Giessen et al. 2016a; Endres et al. 2015; Byerlee et al. 2015; van Dam et al. 2010). Standards are normally voluntary but can also become obligatory through legislation. A standard provides specifications or guidelines to ensure that materials, products, processes and services are fit for their purpose, whereas certification is the procedure through which an accredited party confirms that a product, process or service is in conformity with certain standards. Standards and certification are normally carried out by separate organisations for legitimacy and accountability (see 7.6.6). The International Organization for Standardization (ISO) is a key source for global environmental standards. Those with special relevance for land and climate include a recent standard on combating land degradation and desertification (ISO 2017) and an earlier standard on sustainable bioenergy and biomass use (ISO 2015; Walter et al. 2018). Both aim to support the long-term transition to a climate-resilient bioeconomy; there is *medium evidence* on the sustainability implications of different bioeconomy pathways, but *low agreement* as to which pathways are socially and environmentally desirable (Priefer et al. 2017; Johnson 2017; Bennich et al. 2017a).

Table 7.3 provides a summary of selected standards and certification schemes with a focus on land use and climate: the tickmark shows inclusion of different sustainability elements, with all recognising the inherent linkages between the biophysical and social aspects of land use. Some certification schemes and best practice guidelines are specific to a particular agriculture crop (e.g., soya, sugarcane) or a tree (oil palm) while others are general. International organisations promote sustainable land and biomass use through good practice guidelines, voluntary standards and jurisdictional approaches (Scarlat and Dallemand 2011; Stattman et al. 2018a; ISEAL Alliance). Other frameworks, such as the Global Bioenergy Partnership (GBEP) focus on monitoring land and biomass use through a set of indicators that are applied across partner countries, thereby also promoting technology (knowledge) transfer (GBEP 2017). The Economics of Land Degradation Initiative (ELD) provides common guidelines for economic assessments of land degradation (Nkonya et al. 2013).

Whereas current standards and certification focus primarily on land, climate and biomass impacts where they occur, more recent analysis considers trade-related land use change by tracing supply chain impacts from producer to consumer, leading to the notion of “imported deforestation” that occurs from increasing demand and trade in unsustainable forest and agriculture products, which is estimated to account for 26% of all tropical deforestation (Pendrill et al. 2019). Research and implementation efforts aim to improve supply chain transparency and promote commitments to “zero deforestation” (Gardner et al. 2018a; Garrett et al. 2019; Newton et al. 2018; Godar and Gardner 2019; Godar et al. 2015, 2016). France has developed specific policies on imported deforestation that are expected to eventually include a zero deforestation label (Government of France 2019).

Table 7.3 Selected standards and certification schemes and their components or coverage

				Environmental						Socio-economic		
Acronym	**Scheme, programme or standard**	**Commodity/process, relation to others**	**Type of mechanism**	**GHG emissions**	**Biodiversity**	**Carbon stock**	**Soil**	**Air**	**Water**	**Land use management[a]**	**Land rights**	**Food security[b]**
ISCC	International Sustainability & Carbon Certification	All feedstocks, all supply chains	Certification	√	√	√	√	√	√	√	√	√
Bonsucro	BonsucroEU	Sugar cane and derived products	Certification	√	√	√	√	√	√	√	√	
RTRS	Roundtable on Responsible Soy EU	Soy based products	Certification	√	√	√	√	√	√	√	√	
RSB	Roundtable on Sustainable Biomaterials EU	Biomass for biofuels and biomaterials	Certification	√	√	√	√	√	√	√	√	√
SAN	Sustainable Agriculture	Various agricultural crops and commodities; Linked to Rain Forest Alliance	Technical Network		√	√	√	√	√	√		
RSPO RED	Roundtable on Sustainable Palm Oil RED	Palm oil products	Certification	√	√	√	√	√	√	√	√	√
PEFC	Programme for Endorsement of Forest Certification	Forest management	Certification		√	√	√	√	√	√	√	c
FSC	Forest Stewardship Council	Forest Management	Certification		√	√	√	√	√	√	√	
SBP	Sustainable Biomass Programme	woody biomass (e.g., wood pellets, wood chips); Linked to PEFC and FSC	Certification	√	√	√	√	√	√	√	√	
WOCAT	World Overview of Conservation Approaches and Technologies	Global network on sustainable land management	Best Practice Network			√	√	√	√	√		
ISO 13065: 2015	Bioenergy	biomass and bioenergy, including conversion processes	Standard	√	√	√	√	√	√	√	√	√d
ISO 14055-1: 2017	Land Degradation and Desertification	land use management, including restoration of degradaed land	Standard	√				√	√	√	√	

Source: Modified from (European Commission 2012; DIAZ-CHAVEZ 2015).
√ indicates that the issue is addressed in the standard or scheme
[a] includes restoration of degraded land in some cases (especially ISO 14055-1)
[b] where specifically indicated
[c] reference to the RSB certification/standard
[d] where specifically noted

The sustainability of biofuels and bioenergy has been in particular focus during the past decade or so due to biofuel mandates and renewable energy policies in the U.S., EU and elsewhere (van Dam et al. 2010; Scarlat and Dallemand 2011). The European Union Renewable Energy Directive (EU-RED) established sustainability criteria in relation to EU renewable energy targets in the transport sector (European Commission 2012), which subsequently had impacts on land use and trade with third-party countries (Johnson et al. 2012). In particular, the EU-RED marked a departure in the context of Kyoto/UNFCCC guidelines by extending responsibility for emissions beyond the borders of final use, and requiring developing countries wishing to sell into the EU market to meet the sustainability criteria (Johnson 2011b). The recently revised EU-RED provides sustainability criteria that include management of land and forestry as well as socio-economic aspects (European Union 2018; Faaij 2018; Stattman et al. 2018b). Standards and certification aim to address potential conflicts between different uses of biomass and most schemes also consider co-benefits and synergies (see Cross-chapter Box 7: Bioenergy and BECCS in mitigation scenarios, in Chapter 6). Bioenergy may offer additional income and livelihoods to farmers as well as improvements in technical productivity and multi-functional landscapes (Rosillo Callé and Johnson 2010a; Kline et al. 2017; Araujo Enciso et al. 2016). Results depend on the commodities involved, and also differ between rural and urban areas.

Analyses on the implementation of standards and certification for land and biomass use have focused on their stringency, effectiveness and geographical scope as well as socio-economic impacts such as land tenure, gender and land rights (Diaz-Chavez 2011; German and Schoneveld 2012; Meyer and Priess 2014). The level of stringency and enforcement varies with local environmental conditions, governance approaches and the nature of the feedstock produced (Endres et al. 2015; Lambin et al. 2014; Giessen et al. 2016b; Stattman et al. 2018b). There is *low evidence and low agreement* on how the application and use of standards and certification has actually improved sustainability beyond the local farm, factory or plantation level; the lack of harmonisation and consistency across countries that has been observed, even within a common market or economic region such as the EU, presents a barrier to wider market impacts (Endres et al. 2015; Stattman et al. 2018b; ISEAL Alliance). In the forest sector, there is evidence that certification programmes such as FSC have reduced deforestation in the aggregate as well as reducing air pollution (Miteva et al. 2015; Mcdermott et al. 2015). Certification and standards cannot address global systemic concerns such as impacts on food prices or other market-wide effects but rather are aimed primarily at insuring best practices in the local context. More general approaches to certification such as the Gold Standard are designed to accelerate progress toward the SDGs as well as the Paris Climate Agreement by certifying investment projects while also emphasising support to governments (Gold Standard).

7.4.6.4. Energy access and biomass use

Access to modern energy services is a key component of SDG 7, with an estimated 1.1 billion persons lacking access to electricity while nearly three billion people relying on traditional biomass (fuelwood, agriculture residues, animal dung, charcoal) for household energy needs (IEA 2017). Lack of access to modern energy services is significant in the context of land-climate systems because heavy reliance on traditional biomass can contribute to land degradation, household air pollution and GHG emissions (see Cross Chapter box 12: Traditional Biomass use, in this Chapter). A variety of policy instruments and programmes have been aimed at improving energy access and thereby reducing the heavy reliance on traditional biomass (see Table 7.2); there is *high evidence and high agreement* that programmes and policies that reduce dependence on traditional biomass will have benefits for health and household productivity as well as reducing land degradation (see section 4.5.4) and GHG emissions (Bailis et al. 2015; Cutz et al. 2017a; Masera et al. 2015; Goldemberg et al. 2018a; Sola et al. 2016a; Rao and Pachauri 2017; Denton et al. 2014). There can be trade-offs across different options, especially between health and climate benefits since more efficient wood stoves might have only limited effect, whereas gaseous and liquid fuels (e.g., biogas, LPG, bioethanol) will have highly positive health benefits and climate benefits that vary depending on specific circumstances of the substitution (Cameron et al. 2016; Goldemberg et al. 2018b). Unlike traditional biomass, modern bioenergy offers high quality energy services, although for household cookstoves, even the cleanest options using wood may not perform as well in terms of health and/or climate benefits (Fuso Nerini et al. 2017; Goldemberg et al. 2018b).

Case Study: Forest conservation instruments: REDD+ in the Amazon and India

Over 50 countries have developed national REDD+ strategies, which have key conditions for addressing deforestation and forest degradation (improved monitoring capacities, understanding of drivers, increased stakeholder involvement, and provided a platform to secure indigenous and community land rights), however to achieve its original objectives and to be effective under current conditions, forest-based mitigation actions need to be incorporated in national development plans and official climate strategies, and mainstreamed across sectors and levels of government (Angelsen et al. 2018a).

The Amazon region can illustrate the complexity of the implementation of REDD+, in the most biodiverse place of the planet, with millions of inhabitants and hundreds of ethnic groups, under the jurisdiction of eight countries. While different experiences can be drawn at different spatial scales, at the regional-level, for example, Amazon Fund (van der Hoff et al. 2018), at the subnational level (Furtado 2018), and at the local level (Alvarez et al. 2016; Simonet et al. 2019), there is *medium evidence and high agreement* that REDD+ has stimulated sustainable land-use investments but also is competing with other land uses (e.g., agroindustry) and scarce international funding (both public and private) (Bastos Lima et al. 2017b; Angelsen et al. 2018b)

In the Amazon, at the local level, a critical issue has been the incorporation of indigenous people in the planning and distribution of benefits of REDD+ projects. While REDD+, in some cases, has enhanced participation of community members in the policy-planning process, fund management, and carbon baseline establishment increased project reliability and equity (West 2016), it is clear that, in this region, insecure and overlapping land rights, as well as unclear and contradictory institutional responsibilities, are probably the major problems for REDD+ implementation (Loaiza et al. 2017). Despite legal and rhetoric recognition of indigenous land rights, effective recognition is still lacking (Aguilar-Støen 2017). The key to the success of REDD+ in the Amazon, has been the application of both, incentives and disincentives on key safeguard indicators, including land security, participation, and well-being (Duchelle et al. 2017).

On the other hand, at the subnational level, REDD+ has been unable to shape land-use dynamics or landscape governance, in areas suffering strong exogenous factors, such as extractive industries, and in the absence of effective regional regulation for sustainable land use (Rodriguez-Ward et al. 2018; Bastos Lima et al. 2017b). Moreover, projects with weak financial incentives, engage households with high off-farm income, which already are better off than the poorest families (Loaiza et al. 2015). Beyond, operational issues, clashing interpretations of results might bring clashes between implementing countries or organisations and donor countries, which have revealed concerns over the performance of projects (van der Hoff et al. 2018)

REDD+ Amazonian projects often face methodological issues, including how to assess the opportunity cost among landholders, and informing REDD+ implementation (Kweka et al. 2016). REDD+ based projects depend on consistent environmental monitoring methodologies for measuring, reporting and verification and, in the Amazon, land cover estimates are crucial for environmental monitoring efforts (Chávez Michaelsen et al. 2017).

In India forests and wildlife concerns are on the concurrent list of the Constitution since an amendment in 1976 thus giving the central or federal government a strong role in matters related to governance of forests. High rates of deforestation due to development projects led to the Forest (Conservation) Act (1980) which requires central government approval for diversion of forest land in any state or union territory.

Before 2006 forest diversion for development projects leading to deforestation needed the forest clearance from the Central Government under the provisions of the Forest (Conservation Act) 1980. In order to regulate forest diversion and as payment for ecosystem services a Net Present Value (NPV) frame-work was introduced by the Supreme Court of India informed by the Kanchan Chopra committee (Chopra 2017). The Supreme Court established the Compensatory Afforestation Management and Planning Authority (CAMPA) under which the fund collected for compensatory afforestation and on account of NPV from project developers is deposited. The Forest (Conservation) Act of 1980 does require compensatory afforestation in lieu of forest diversion and in addition after CAMPA the payment of NPV to get the forest clearance for diversion has been added.

As of February 2018, USD 6,825 million had accumulated in CAMPA funds in lieu of NPV paid by developers diverting forest land throughout India for non-forest use. Funds are released by the central government to state governments out of this fund for afforestation and conservation related activities to “compensate” for diversion of forests. This is now governed by legislation called CAMPA Act passed by the Parliament of India in July 2016. The CAMPA mechanism has however invited criticism on various counts in terms of undervaluation of forest, inequality, lack of participation and environmental justice (Temper and Martinez-Alier 2013).

The other significant development related to forest land was the landmark legislation called the Scheduled Tribes and Other Traditional Forest Dwellers (Recognition of Forest Rights) Act, 2006 or Forest Rights Act passed by the Parliament of India in 2007. This is the largest forest tenure legal instrument in the world and attempted to undo a historical injustice to forest dwellers and forest dependent communities whose traditional rights and access were legally denied under forest and wildlife conservation laws. The FRA recognises the right to individual land titles on land already cleared as well as community forest rights such as collection of forest produce. Till November 2018, a total of 64,328 community forest rights and a total of 17,040,343 individual land titles had been approved and granted up to the end of 2017. Current concerns on policy and implementation gaps are about strengths and pitfalls of decentralisation, identifying genuine right holders, verification of land rights using technology and best practices, and curbing illegal claims (Sarap et al. 2013; Reddy et al. 2011; Aggarwal 2011; Ramnath 2008; Ministry of Environment and Forests and Ministry and Tribal Affairs, Government of India 2010).

As per the FRA, the forest rights shall be conferred free of all encumbrances and procedural requirements. Furthermore, without implementation of the provision of FRA on getting the informed consent of local communities for both diversion of community forest land as well as for reforestation, it poses legal and administrative hurdles in using existing forest land for implementation of India's ambitious Green India Mission that aims to respond to climate change by a combination of adaptation and mitigation measures in the forestry sector. It aims to increase forest/tree cover to the extent of 5 million hectares (Mha) and improve quality of forest/tree cover on another 5 Mha of forest/non-forest lands and support forest based livelihoods of 3 million families and generate co-benefits through ecosystem services (Government of India).

Thus, the community forest land recognised under FRA can be used for the purpose of Compensatory Afforestation or restoration under REDD+ only with informed consent of the communities and a decentralised mechanism for using CAMPA funds. India's forest and forest restoration can potentially move away from a top-down carbon centric model with the effective participation of local communities (Vijge and Gupta 2014; Murthy et al. 2018a).

India has also experimented with the world's first national inter-governmental ecological fiscal transfer (EFT) from central to local and state government to reward them for retaining forest cover. In 2014, India's 14th Finance Commission added forest cover to the formula that determines the amount of tax revenue the central government distributes annually to each of India's 29 states. It is estimated that in four years it would have distributed USD 6.9–12 billion per year to states in proportion to their 2013 forest cover, amounting to around USD 174– 303 per hectare of forest per year (Busch and Mukherjee 2017). State governments in India now have a sizeable fiscal incentive based on extent of forest cover at the time of policy implementation contributing to the achievement of India's climate mitigation and forest conservation goals. India's tax revenue distribution reform has created the world's first EFTs for forest conservation, and a potential model for other countries. However, it is to be noted that EFT is calculated based on a one-time estimate of forest cover prior to policy implementation, hence does not incentivise ongoing protection and this is a policy gap. It's still too early but its impact on trends in forest cover in the future and its ability to conserve forests without other investments and policy instruments is promising but untested (Busch and Mukherjee 2017; Busch 2018).

In order to build on the new promising policy developments on forest rights and fiscal incentives for forest conservation in India, incentivising ongoing protection, further investments in monitoring (Busch 2018), decentralisation (Somanathan et al. 2009) and promotion of diverse non-agricultural forest and range land based livelihoods (e.g., sustainable non-timber forest product extraction, regulated pastures, carbon credits for forest regeneration on marginal agriculture land and ecotourism revenues) as part of individual and community forest tenure and rights are ongoing concerns. Decentralised sharing of CAMPA funds between government and local communities for forest restoration as originally suggested and filling in implementation gaps could help reconcile climate change mitigation through forest conservation, REDD+ and environmental justice (Vijge and Gupta 2014; Temper and Martinez-Alier 2013; Badola et al. 2013; Sun and Chaturvedi 2016; Murthy et al. 2018b; Chopra 2017; Ministry of Environment and Forests and Ministry and Tribal Affairs, Government of India 2010).

7.4.7. Economic and financial instruments for adaptation, mitigation, and land

There is an urgent need to increase the volume of climate financing and bridge the gap between global adaptation needs and available funds (*medium confidence*) (Valérie Masson-Delmotte et al. 2018; Kissinger et al. 2019; Chambwera and Heal 2014), especially in relation to agriculture (FAO 2010). The land sector offers the potential to balance the synergies between mitigation and adaptation

(Locatelli et al. 2016) (although context and unavailability of data sets makes cost comparisons between mitigation and adaptation difficult (UNFCCC 2018b)). Estimates of adaptation costs range from USD 140 to 300 billion by 2030, and between USD 280 and 500 billion by 2050; (UNEP 2016). These figures vary according to methodologies and approaches (de Bruin et al. 2009; IPCC 2014 2014; Organization for Economic Cooperation and Development 2008; Nordhaus 1999; UNFCCC 2007; Plambeck et al. 1997).

7.4.7.1. Financing mechanisms for land mitigation and adaptation

A startling array of diverse and fragmented climate finance sources exist: more than 50 international public funds, 60 carbon markets, 6000 private equity funds, 99 multilateral and bilateral climate funds (Samuwai and Hills 2018). Most public finance for developing countries flows through bilateral and multilateral institutions such as the World Bank, the International Monetary Fund, International Finance Corporation, regional development banks, as well as specialised multilateral institutions such as the Global Environmental Fund, and the EU Solidarity Fund. Some governments have established state investment banks (SIBs) to close the financing gap, including the UK (Green Investment Bank), Australia (Clean Energy Finance Corporation) and in Germany (Kreditanstalt für Wiederaufbau) the Development Bank has been involved in supporting low-carbon finance (Geddes et al. 2018). The Green Climate Fund (GCF) now offers additional finance, but is still a new institution with policy gaps, a lengthy and cumbersome process related to approval (Brechin and Espinoza 2017; Khan and Roberts 2013; Mathy and Blanchard 2016), and challenges with adequate and sustained funding (Schalatek and Nakhooda 2013). Private adaptation finance exists, but is difficult to define, track, and coordinate (Nakhooda et al. 2016).

The amount of funding dedicated to agriculture, land degradation or desertification is very small compared to total climate finance (FAO 2010). Funding for agriculture is accessed through the smaller adaptation funds (rather than mitigation) (Lobell et al. 2013). Focusing on synergies, between mitigation, adaptation, and increased productivity, such as through Climate Smart Agriculture (CSA)(see 7.5.6), (Lipper et al. 2014b), may leverage greater financial resources (Suckall et al. 2015; Locatelli et al. 2016). Payments for Ecosystem Services (see 7.4.6) are another emerging area to encourage environmentally desirable practices, although they need to be carefully designed to be effective (Engel and Muller 2016).

The UNCCD established the Land Degradation Neutrality Fund (LDN Fund) to mobilise finance and scale up land restortion and sustainable business models on restored land to achieve the target of a land degradation neutral world (SDG target 15.3) by 2030. The LDN Fund generates revenues from sustainable use of natural resources, creating green job opportunities, sequestering CO_2, and increasing food and water security (Cowie et al. 2018a; Akhtar-Schuster et al. 2017). The fund leverages public money to raise private capital for sustainable land management and land restoration projects (Quatrini and Crossman 2018; Stavi and Lal 2015). Many small-scale projects are demonstrating that sustainable landscape management (see 7.6.3) is key to achieving LDN, and it is also more financially viable in the long term than the unsustainable alternative (Tóth et al. 2018; Kust et al. 2017).

7.4.7.2. Instruments to manage the financial impacts of climate and land change disruption

Comprehensive risk management (see 7.4.3.1) designs a portfolio of instruments which are used across a continuum of preemptive, planning and assessment, and contingency measures in order to bolster resilience (Cummins and Weiss 2016) and address limitations of any one instrument (Surminski 2016; Surminski et al. 2016; Linnerooth-bayer et al. 2019). Instruments designed and applied in isolation have shown short-term rather than sustained intended impacts (Vincent et al. 2018). Risk assessments limited to events and impacts on particular asset classes or sectors can misinform policy and drive misallocation of funding (Gallina et al. 2016; Jongman et al. 2014).

Comprehensive risk assessment combined with risk layering approaches that assign different instruments to different magnitude and frequency of events, have better potential to provide stability to societies facing disruption (Mechler et al. 2014; Surminski et al. 2016). Governments and citizens define limits of what they consider acceptable risks, risks for which market or other solutions can be developed and catastrophic risks that require additional public protection and intervention. Different financial tools may be used for these different categories of risk or phases of the risk cycle (preparedness, relief, recovery, reconstruction).

In order to protect lives and livelihoods early action is critical, including a coordinated plan for action agreed in advance, a fast, evidence-based decision-making process, and contingency financing to ensure that the plan can be implemented (Clarke and Dercon 2016a). Forecast-based finance mechanisms incorporate these principles, using climate or other indicators to trigger funding and action prior to a shock (Wilkinson 2018). Forecast-based mechanisms can be linked with social protection systems by providing contingent scaled-up finance quickly to vulnerable populations following disasters, enhancing scalability, timeliness, predictability and adequacy of social protection benefits (Wilkinson 2018; Costella et al. 2017b; World Food Programme 2018).

Measures in advance of risks set aside resources before negative impacts related to adverse weather, climatic stressors, and land changes occur. These tools are frequently applied in extreme event, rapid onset contexts. These measures are the main instruments for reducing fatalities and limiting damage from extreme climate and land change events (Surminski et al. 2016). Finance tools in advance of risk include insurance (macro, meso, micro), green bonds, and forecast based finance (Hunzai et al. 2018).

There is *high confidence* that insurance approaches which are designed to effectively reduce and communicate risks to the public and beneficiaries, designed to reduce risk and foster appropriate adaptive responses, and provide value in risk transfer, improve economic stability and social outcomes in both higher and lower income contexts (Kunreuther and Lyster 2016; Outreville 2011b)(Surminski et al. 2016; Kousky et al. 2018b), bolster food security, helping keep children in school, and helping safeguard the ability of low income households to pay for essentials like medicines (Shiferaw et al. 2014; Hallegatte et al. 2017).

Low income households show demand for affordable risk transfer tools, but demand is constrained by liquidity, lack of assets, financial and insurance literacy, or proof of identity required by institutions in the formal sector (Eling et al. 2014; Cole 2015; Cole et al. 2013; Ismail et al. 2017). Microinsurance participation takes many forms including through mobile banking (Eastern Africa, Bangladesh), linked with social protection or other social stabilisation programs (Ethiopia, Pakistan, India), through flood or drought protection schemes (Indonesia, the Philippines, the Caribbean, and Latin America), often in the form of weather index insurance. Insurance faces challenges around low public awareness of how insurance works, risk, low capacity in financial systems to administer insurance, data deficits, and market imperfections (Mechler et al. 2014; Feyen et al. 2011; Gallagher 2014; Kleindorfer et al. 2012; Lazo et al.; Meyer and Priess 2014; Millo 2016).

Countries also request grant assistance, and contingency debt finance that includes dedicated funds, set aside for unpredictable climate-related disasters, household savings, loans with “catastrophe risk deferred drawdown option” (CATDDO) (which allows countries to divert loans from development objectives such as health, education, and infrastructure to make immediate disbursement of funds in the event of a disaster) (Kousky and Cooke 2012; Clarke and Dercon 2016b). Contingency finance is suited to manage frequently occurring, low-impact events (Campillo et al. 2017; Mahul and Ghesquiere 2010; Roberts 2017) and may be linked with social protection systems. These instruments are limited by uncertainty surrounding the size of contingency fund reserves, given unpredictable climate disasters (Roberts 2017) and lack of borrowing capacity of a country (such as small island states) (Mahul and Ghesquiere 2010).

In part because of its link with debt burden, contingency, or post event finance can disrupt development and is not suitable for higher consequence events and processes such as weather extremes or structural changes associated with climate and land change. Post event finance of negative impacts such as sea level rise, soil salinisation, depletion of groundwater, and widespread land degradation is likely to become infeasible for multiple, high cost events and processes. There is *high confidence* post-extreme event assistance may face more severe limitations given impacts of climate change (Linnerooth-bayer et al. 2019; Surminski et al. 2016; Deryugina 2013; Dillon et al. 2014; Clarke 2016; Shreve and Kelman 2014; Von Peter et al. 2012).

In a catastrophe risk pool, multiple countries in a region pool risks in a diversified portfolio. Examples include Africa Risk Capacity (ARC), the Caribbean Catastrophe Risk Insurance Facility (CCRIF), and the Pacific Catastrophe Risk Assessment and Financing Initiative (PCRAFI) (Bresch et al. 2017; Iyahen and Syroka 2018). ARC payouts have been used to assist over 2.1 million food insecure people and provide over 900,000 cattle with subsidised feed in the affected countries (Iyahen and Syroka 2018). ARC has also developed the Extreme Climate Facility, which is designed to complement existing bilateral, multilateral and private sources of finance to enable proactive adaptation (Vincent et al. 2018). It provides beneficiaries the opportunity to increase their benefit by reducing exposure to risk through adaptation and risk reduction measures, thus side-stepping "moral hazard" problems sometimes associated with traditional insurance.

Governments pay coupon interest when purchasing catastrophe (CAT) bonds from private or corporate investors. In the case of the pre-defined catastrophe, the requirement to pay the coupon interest or repay the principal may be deferred or forgiven (Nguyen and Lindenmeier 2014). CAT bonds are typically short-term instruments (3–5 years) and the payout is triggered once a particular threshold of disaster/damage is passed (Härdle and Cabrera 2010; Campillo et al. 2017; Estrin and Tan 2016; Hermann, A., Koferl, P., Mairhofer 2016; Michel-Kerjan 2011; Roberts 2017). The primary advantage of CAT bonds is their ability to quickly disburse money in the event of a catastrophe (Estrin and Tan 2016). Green bonds, social impact bonds, and resilience bonds are other instruments that can be used to fund land based interventions. However, there are significant barriers for developing country governments to enter into the bond market: lack of familiarity with the instruments; lack of capacity and resources to deal with complex legal arrangements; limited or non-existent data and modelling of disaster exposure; and other political disincentives linked to insurance. For these reasons the utility and application of bonds is currently largely limited to higher-income developing countries (Campillo et al. 2017; Le Quesne 2017).

7.4.7.3. Innovative financing approaches for transition to low carbon economies

Traditional financing mechanisms have not been sufficient and thereby leave a gap in facilitating a rapid transition to a low carbon economy or building resilience (Geddes et al. 2018). More recently there have been developments in more innovative mechanisms including crowdfunding (Lam and Law 2016), often supported by national governments (in the U.K. through regulatory and tax support)(Owen et al. 2018). Crowdfunding has no financial intermediaries and thus low transaction costs, and the projects have a greater degree of independence than bank or institution funding (Miller et al. 2018). Other examples of innovative mechanisms are community shares for local projects, such as renewable energy (Holstenkamp and Kahla 2016), or Corporate Power Purchase Agreements (PPAs) used by companies such as Google and Apple to purchase renewable energy directly or virtually from developers (Miller et al. 2018). Investing companies benefit from avoiding unpredictable price fluctuations as well as increasing their environmental credentials. A second example is auctioned price floors, or subsidies that offer a guaranteed price for future emission reductions, currently being trialled in developing countries, by the World Bank Group, known as the Pilot Auction Facility (PAF) (Bodnar et al. 2018). Price floors can maximise the climate impact per public dollar while incentivising private investment in low-carbon technologies, and ideally would be implemented in conjunction with complementary policies such as carbon pricing.

In order for climate finance to be as effective and efficient as possible, cooperation between private, public and third sectors (e.g., NGOs, cooperatives, community groups) is more likely to create an enabling environment for innovation (Owen et al. 2018). While innovative private sector approaches are making significant progress, the existence of a stable policy environment that provides certainty and incentives for long term private investment is critical.

7.4.8. Enabling effective policy instruments – Policy Portfolio Coherence

An enabling environment for policy effectiveness includes: 1) the development of comprehensive policies, strategies and programs (section 7.4); 2) human and financial resources that ensure policies, programs and legislation are translated into action; 3) decision making that draws on evidence generated from functional information systems that make it possible to monitor trends; track and map actions; and assess impact in a manner that is timely and comprehensive (see 7.5); 4) governance coordination mechanisms and partnerships; and 5) a long term perspective in terms of response options, monitoring, and maintenance (see 7.6) (FAO 2017a).

A comprehensive consideration of policy portfolios achieves sustainable land and climate management (*medium confidence*) (Mobarak and Rosenzweig 2013; Stavropoulou et al. 2017) (Jeffrey et al. 2017) (Howlett and Rayner 2013) (Aalto et al. 2017; Brander and Keith 2015; Williams and Abatzoglou 2016) (Linnerooth-Bayer and Hochrainer-Stigler 2015) (FAO 2017b; Bierbaum and Cowie 2018). Supporting the study of enabling environments, the study of policy mixes has emerged in the last decade in regards to the mix or set of instruments that interact together and are aimed at achieving policy objectives in a dynamic setting (Reichardt et al. 2015). The study of policy mixes includes studying the ultimate objectives of a policy mix (such as biodiversity (Ring and Schröter-Schlaack 2011)), the interaction of policy instruments within the mix (including climate change mitigation and energy (del Río and Cerdá 2017)) (see Trade-offs and Synergies, 7.5.6), and the dynamic nature of the policy mix (Kern and Howlett 2009)).

Studying policy mixes allows for a consideration of policy coherence which is broader than the study of discrete policy instruments in rigidly defined sectors, but entails studying policy in relation to the links and dependencies among problems and issues (FAO 2017b). Consideration of policy coherence is a new approach rejecting simplistic solutions, but acknowledging inherently complex processes involving collective consideration of public and private actors in relation to policy analysis (FAO 2017b). A coherent, consistent mix of policy instruments can solve complex policy problems (Howlett and Rayner 2013) as it involves lateral, integrative, and holistic thinking in defining and solving problems (FAO 2017b). Such a consideration of policy coherence is required to achieve sustainable development (FAO 2017b; Bierbaum and Cowie 2018). Considerations of policy coherence potentially addresses three sets of challenges: challenges that exist with assessing multiple hazards and sectors (Aalto et al. 2017; Brander and Keith 2015; Williams and Abatzoglou 2016); challenges in mainstreaming adaptation and risk management into on-going development planning and decision making (Linnerooth-Bayer and Hochrainer-Stigler 2015); challenges in scaling up community and ecosystem based initiatives in countries overly focused on sectors, instead of sustainable use of biodiversity and ecosystem services (Reid 2016). There is a gap in integrated consideration of adaptation, mitigation, climate change policy and development. A study in Indonesia found while internal policy coherence between mitigation and adaptation is increasing, external policy coherence between climate change policy and development objectives is still required (Di Gregorio et al. 2017).

There is *medium evidence and high agreement* that a suite of agricultural business risk programs (which would include crop insurance and income stability programs) increase farm financial performance, reduce risk, and also reinforce incentives to adopt stewardship practices (beneficial management practices) improving the environment (Jeffrey et al. 2017). Consideration of the portfolio of instruments responding to climate change and its associated risks, and the interaction of policy instruments, improve agricultural producer livelihoods (Hurlbert 2018b). In relation to hazards, or climate related extremes (7.4.3), the policy mix has been found to be a key determinant of the adaptive capacity of agricultural producers. In relation to drought, the mix of policy instruments including crop insurance, sustainable land management practices, bankruptcy and insolvency, co-management of community in water and disaster planning, and water infrastructure programmes are effective at responding to drought (Hurlbert 2018b; Hurlbert and Mussetta 2016; Hurlbert and Pittman 2014; Hurlbert and Montana 2015; Hurlbert 2015a) (Hurlbert and Gupta 2018). Similarly in relation to flood, the mix of policy instruments including flood zone mapping, land use planning, flood zone building restrictions, business and crop insurance, disaster assistance payments, preventative instruments including environmental farm planning (including soil and water management (see Chapter 6)) and farm infrastructure projects, and recovery from debilitating flood losses ultimately through bankruptcy are effective at responding to flood (Hurlbert 2018a)(see 7.6.3 Case Study Flood and Flood Security).

In respect of land conservation and management goals, consideration of differing strengths and weakness of instruments is necessary. While direct regulation may secure effective minimum standards of biodiversity conservation and critical ecosystem service provision, economic instruments may achieve reduced compliance costs as costs are borne by policy addressees (Rogge and Reichardt 2016). In relation to GHG emissions and climate mitigation a comprehensive mix of instruments targeted at emissions reductions, learning, and research and development is effective (*high confidence*) (Fischer and Newell 2008). The policy coherence between climate policy and public finance is critical in ensuring the efficiency, effectiveness and equity of mitigation policy, and ultimately to make stringent mitigation policy more feasible (Siegmeier et al. 2018). Recycling carbon tax revenue to support clean energy technologies can decrease losses from unilateral carbon mitigation targets with complementary technology polices (Corradini et al. 2018).

When evaluating a new policy instrument, its design in relation to achieving an environmental goal or solving a land and climate change issue, includes consideration of how the new instrument will interact with existing instruments operating at multiple levels (international, regional, national, sub-national, and local) (Ring and Schröter-Schlaack 2011)(see 7.4.1).

7.4.9. Barriers to Implementing Policy Responses

There are barriers to implementing the policy instruments that arise in response to the risks from climate-land interactions. Such barriers to climate action help determine the degree to which society can achieve its sustainable development objectives (Dow et al. 2013; Langholtz et al. 2014; Klein et al. 2015). However, some policies can also be seen as being designed specifically to overcome barriers, while in some cases policies may actually create or strengthen barriers to climate action (Foudi and Erdlenbruch 2012; Linnerooth-Bayer and Hochrainer-Stigler 2015). The concept of barriers to climate action is used here in a sense close to that of “soft limits” to adaptation (Klein, et al. 2014). “Hard limits” by contrast are seen as primarily biophysical. Predicted changes in the key factors of crop growth and productivity—temperature, water, and soil quality— are expected to pose limits to adaptation in ways that affect the world’s population to get enough food in the future (Altieri et al. 2015; Altieri and Nicholls 2017).

This section assesses research on barriers specific to policy implementation in adaptation and mitigation respectively, then addresses the cross-cutting issue of inequality as a barrier to climate action, including the particular cases of elite capture and corruption, before assessing how policies on climate and land can be used to overcome barriers.

7.4.9.1. Barriers to Adaptation

There are human, social, economic, and institutional barriers to adaptation to land-climate challenges as described in Tabel 7.4 (*medium evidence, high agreement*). Considerable literature exists around changing behaviours through response options targeting social and cultural barriers (Rosin 2013; Eakin; Marshall et al. 2012) (See Chapter 6 Value chain interventions).

Table 7.4 Soft Barriers and Limits to Adaptation

Category	Description	References
Human	Cognitive and behavioural obstacles. Lack of knowledge and information.	(Hornsey et al. 2016; Prokopy et al. 2015) (Wreford et al. 2017)
Social	Undermined participation in decision making and social equity	(Burton et al. 2008) (Laube et al. 2012)
Economic	Market failures and missing markets, transaction costs and political economy, ethical and distributional issues. Perverse incentives. Lack of domestic funds, inability to access international funds	(Chambwera et al. 2014b) (Wreford et al. 2017) (Rochecouste et al. 2015; Baumgart-Getz et al. 2012)
Institutional	Mal-coordination of policies and response options, unclear responsibility of actors and leadership, misuse of power, all reducing social learning. Government failures. Path dependent institutions.	(Oberlack 2017) (Sánchez et al. 2016; Greiner and Gregg 2011)
Technological	Systems of mixed crop and livestock. Polycultures.	(Nalau and Handmer 2015)

Since AR5 research examining the role of governance, institutions and in particular policy instruments, in creating or overcoming barriers to adaptation to land and climate change in the land use sector is emerging (Foudi and Erdlenbruch 2012; Linnerooth-Bayer and Hochrainer-Stigler 2015). Evidence shows that understanding the local context and targeted approaches are generally most successful (Rauken et al. 2014). Understanding the nature of constraints to adaptation is critical in determining how barriers may be overcome. Formal institutions (rules, laws, policies) and informal institutions (social and cultural norms and shared understandings) can be barriers and enablers of climate adaptation (Jantarasami et al. 2010). Governments play a key role in intervening and confronting existing barriers by changing legislation, adopting policy instruments, providing additional resources, and building institutions and knowledge exchange (Ford and Pearce 2010; Measham et al. 2011; Mozumder et al. 2011; Storbjörk 2010). Understanding institutional barriers is important in addressing barriers (*high confidence*). Institutional barriers may exist due to the path-dependent nature of institutions governing natural resources and public good, bureaucratic structures that undermine horizontal and vertical integration (see 7.6.2), and lack of policy coherence (see 7.4.8).

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7.4.9.2. Barriers to land based climate mitigation

Barriers to land based mitigation relate to full understanding of the permanence of carbon sequestration in soils or terrestrial biomass, the additionality of this storage, its impact on production and production shifts to other regions, measurement and monitoring systems and costs (Smith et al. 2007). Agricultural producers are more willing to expand mitigation measures already employed (including efficient and effective management of fertiliser including manure and slurry) and less favourable to those not employed such as using dietary additives, adopting genetically improved animals, or covering slurry tanks and lagoons (Feliciano et al. 2014). Barriers identified in land based mitigation include physical environmental constraints including lack of information, education, and suitability for size and location of farm. For instance precision agriculture is not viewed as efficient in small scale farming (Feliciano et al. 2014).

Property rights may be a barrier when there is no clear single party land ownership to implement and manage changes (Smith et al. 2007). In forestry, tenure arrangements may not distribute obligations and incentives for carbon sequestration effectively between public management agencies and private agents with forest licenses. Including carbon in tenure and expanding the duration of tenure may provide stronger incentive for tenure holders to manage carbon as well as timber values (Williamson and Nelson 2017). Effective policy will require answers as to the current status of agriculture in regard to GHG emissions, the degree that emissions are to change, the best pathway to achieve the change, and an ability to know when the target level of change is achieved (Smith et al. 2007). Forest governance may not have the structure to advance mitigation and adaptation. Currently top down traditional modes do not have the flexibility or responsiveness to deal with the complex, dynamic, spatially diverse, and uncertain features of climate change (Timberlake and Schultz 2017; Williamson and Nelson 2017).

In respect of forest mitigation, two main institutional barriers have been found to predominate. First forest management institutions do not consider climate change to the degree necessary for enabling effective climate response and do not link adaptation and mitigation; Second, institutional barriers exist if institutions are not forward looking, do not enable collaborative adaptive management, promote flexible approaches that are reversible as new information becomes available, promote learning and allow for diversity of approaches that can be tailored to different local circumstances (Williamson and Nelson 2017).

Land-based climate mitigation through expansions and enhancements in agriculture, forestry and bioenergy has great potential but also poses great risks and its success will therefore require improved land use planning, strong governance frameworks and coherent and consistent policies. "Progressive developments in governance of land and modernisation of agriculture and livestock and effective sustainability frameworks can help realise large parts of the technical bioenergy potential with low associated GHG emissions"(Smith et al. 2014b, p. 97).

7.4.9.3. Inequality

There is *medium evidence and high agreement* that one of the greatest challenges for land based adaptation and sustainable land management is posed by inequalities that influence vulnerability and coping and adaptive capacity - including age, gender, wealth, knowledge, access to resources and power (Kunreuther et al. 2014; IPCC 2012; Olsson et al. 2014). Gender is the dimension of inequality that has been the focus of most research while research demonstrating differential impacts, vulnerability and adaptive capacity based on age, ethnicity and indigeneity is less well developed (Olsson et al. 2015a). Cross-Chapter Box 11 sets out both the contribution of gender relations to differential vulnerability and available policy instruments for greater gender inclusivity.

One response to the vulnerability of poor people and other categories differentially affected is effective and reliable social safety nets (Jones and Hiller 2017). Social protection coverage is low across the world and informal support systems continue to be the key means of protection for a majority of the rural poor and vulnerable (Stavropoulou et al. 2017)(See 7.4.2). However, there is a gap in knowledge in understanding both positive and negative synergies between formal and informal systems of social protection and how local support institutions might be used to implement more formal forms of social protection (Stavropoulou et al. 2017).

7.4.9.4. Corruption and elite capture

Inequalities of wealth and power can allow processes of corruption and elite capture which can affect both adaptation and mitigation actions, at levels from the local to the global, that in turn risk creating inequitable or unjust outcomes (Sovacool 2018) (*limited evidence, medium agreement*). This includes risks of corruption in REDD+ processes (Sheng et al. 2016; Williams and Dupuy 2018) and of corruption or elite capture in broader forest governance (Sundström 2016; Persha and Andersson 2014), as well as elite capture of benefits from planned adaptation at a local level (Sovacool 2018).

Peer-reviewed empirical studies that focus on corruption in climate finance and climate interventions, particularly at a local level, are rare, due in part to the obvious difficulties of researching illegal and clandestine activity (Fadairo et al. 2017). At the country level, historical levels of corruption are shown to affect current climate polices and global cooperation (Fredriksson and Neumayer 2016). Brown (2010) sees three likely inlets of corruption into REDD: in the setting of forest baselines, the reconciliation of project and natural credits, and the implementation of control of illegal logging. The transnational and north-south dimensions of corruption are highlighted by debates on which US legislative instruments (e.g., the Lacey Act, the Foreign Corrupt Practices Act) could be used to prosecute the northern corporations that are involved in illegal logging (Gordon 2016; Waite 2011).

Fadairo et al. (2017) carried out a structured survey of perceptions of households in forest-edge communities served by REDD+, as well as those of local officials, in south eastern Nigeria. They report high rates of agreement that allocation of carbon rights is opaque and uncertain, distribution of benefits is untimely, uncertain and unpredictable, and the REDD+ decision-making process is vulnerable to political interference that benefits powerful individuals. Only 35% of respondents had an overall perception of transparency in REDD+ process as "good". Of eight institutional processes or facilities previously identified by the Government of Nigeria and international agencies as indicators of commitment to transparent and equitable governance, only three were evident in the local REDD+ office as "very functional" or "fairly functional".

At the local level, the risks of corruption and elite capture of the benefits of climate action are high in decentralised regimes (Persha and Andersson 2014). (Rahman 2018) discusses elicitation of bribes (by local-level government staff) and extortion (by criminals) to allow poor rural people to gather forest products. The results are a general undermining of households' adaptive capacity and perverse incentives to over-exploit forests once bribes have been paid, leading to over-extraction and biodiversity loss. Where there are pre-existing inequalities and conflict, participation processes need careful management and firm external agency to achieve genuine transformation and avoid elite capture (Rigon 2014). An illustration of the range of types of elite capture is given by Sovacool (2018) for adaptation initiatives including coastal afforestation, combining document review and key informant interviews in Bangladesh, with an analytical approach from political ecology. Four processes are discussed: enclosure, including land grabbing and preventing the poor establishing new land rights; exclusion of the poor from decision-making over adaptation; encroachment on the resources of the poor by new adaptation infrastructure; and entrenchment of community disempowerment through patronage. The article notes that observing these processes does not imply they are always present, nor that adaptation efforts should be abandoned.

7.4.9.5. Overcoming Barriers

Policy instruments that strengthen agricultural producer assets or capitals reduce vulnerability and overcome barriers to adaptation (Hurlbert 2018b, 2015b). Additional factors like formal education and knowledge of traditional farming systems, secure tenure rights, access to electricity and social institutions in rice-farming areas of Bangladesh have played a positive role in reducing adaptation barriers (Alam 2015). A review of over 168 publications over 15 years about adaptation of water resources for irrigation in Europe found the highest potential for action is in improving adaptive capacity and responding to changes in water demands, in conjunction with alterations in current water policy, farm extension training, and viable financial instruments (Iglesias and Garrote 2015). Research on the Great Barrier Reef, the Olifants River in Southern Africa, and fisheries in Europe, North America, and the Antarctic Ocean, suggests the leading factors in harnessing the adaptive capacity of ecosystems is to reduce human stressors by enabling actors to collaborate across diverse interests, institutional settings, and sectors (Biggs et al. 2017; Schultz et al. 2015; Johnson and Becker 2015). Fostering equity and participation are correlated with the efficacy of local adaptation to secure food and livelihood security (Laube et al. 2012). In this chapter, the literature surrounding appropriate policy instruments, decision making, and governance practices to overcome limits and barriers to adaptation is proposed.

Incremental adaptation consists of actions where the central aim is to maintain the essence and integrity of a system or process at a given site whereas transformational adaptation is adaptation that changes the fundamental attributes of a system in response to climate and its effects; the former is characterised as doing different things and the latter, doing things differently (Noble et al. 2014). Transformational adaptation is necessary in situations where there are hard limits to adaptation or it is desirable to address deficiencies in sustainability, adaptation, inclusive development and social equity (Kates et al. 2012; Mapfumo et al. 2016). In other situations, incremental changes may be sufficient (Hadarits et al. 2017).

Cross-chapter Box 11: Gender in inclusive approaches to climate change, land, and sustainable development

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Gender is a key axis of social inequality that intersects with other systems of power and marginalisation—including “race”, culture, class/socioeconomic status, location, sexuality, and age—to cause unequal experiences of climate change vulnerability and adaptive capacity. However, “policy frameworks and strong institutions that align development, equity objectives, and climate have the potential to deliver ‘triple-wins’” (Roy et al. 2018), including enhanced gender equality. Gender in relation to this report is introduced in Chapter 1, referred to as a leverage point in women’s participation in decisions relating to land desertification (3.6.3), land degradation (4.1.6), food security (5.2.5.1), and enabling land and climate response options (6.1.2.2).

Focusing on ‘gender’ as a relational and contextual construct can help avoid homogenising “women” as a uniformly and consistently vulnerable category (Arora-Jonsson 2011; Mersha and Van Laerhoven 2016; Ravera et al. 2016). There is *high agreement* that using a framework of intersectionality to integrate gender into climate change research helps to recognise overlapping and interconnected systems of power (Djoudi et al. 2016; Fletcher 2018; Kaijser and Kronsell 2014; Moosa and Tuana 2014; Thompson-Hall et al. 2016), which create particular inequitable experiences of climate change vulnerability and adaptation. Through this framework, both commonalities and differences may be found between the experiences of rural and urban women, or between women in high-income and low-income countries, for example.

In rural areas, women generally experience greater vulnerability than men, albeit through different pathways (Djoudi et al., 2016; Goh, 2012; Jost et al., 2016; Kakota, Nyariki, Mkwambisi, & Kogi-

Makau, 2011). In masculinised agricultural settings of Australia and Canada, for example, climate adaptation can increase women's work on- and off-farm, but without increasing recognition for women's undervalued contributions (Alston et al. 2018a; Fletcher and Knuttila 2016). A study in rural Ethiopia found that male-headed households had access to a wider set of adaptation measures than female-headed households (Mersha and Van Laerhoven 2016).

Due to engrained patriarchal social structures and gendered ideologies, women may face multiple barriers to participation and decision-making in land-based adaptation and mitigation actions in response to climate change (*high confidence*) (Alkire et al. 2013a; Quisumbing et al. 2014). These barriers include: (i) disproportionate responsibility for unpaid domestic work, including care-giving activities (Beuchelt and Badstue 2013) and provision of water and firewood (UNEP, 2016); (ii) risk of violence in both public and private spheres, which restricts women's mobility for capacity-building activities and productive work outside the home (Day et al., 2005; Jost et al., 2016; UNEP, 2016); (iii) less access to credit and financing (Jost et al. 2016); (iv) lack of organisational social capital, which may help in accessing credit (Carroll et al. 2012); (v) lack of ownership of productive assets and resources (Kristjanson et al., 2014; Meinzen-Dick et al., 2010), including land. Constraints to land access include not only state policies, but also customary laws (Bayisenge 2018) based on customary norms and religion that determine women's rights (Namubiru-Mwaura 2014a).

Differential vulnerability to climate change is related to inequality in rights-based resource access, established through formal and informal tenure systems. In only 37% of 161 developing and developed countries do men and women have equal rights to use and control land, and in 59% customary, traditional, and religious practices discriminate against women (OECD 2014), even if the law formally grants equal rights. Women play a significant role in agriculture, food security and rural economies globally, forming 43% of the agricultural labour force in developing countries (FAO, IFAD, UNICEF, & WHO, 2018, p. 102), ranging from 25 % in Latin America (FAO, 2017, pp. 89) to nearly 50% in Eastern Asia and Central and South Europe (FAO, 2017, p. 88) and 47% in sub-Saharan Africa (FAO, 2017, pp. 88). Further, the share of women in agricultural employment has been growing in all developing regions except East Asia and Southeast Asia (FAO, 2017, p. 88). At the same time, women constitute less than 5% of landholders (with legal rights and/or use-rights (Doss et al. 2018a) in North Africa and West Asia, about 15% in sub-Saharan Africa, 12% in Southern and Southeastern Asia, 18% in Latin America and Caribbean (FAO 2011b, p. 25), 10% in Bangladesh, 4% in Nigeria (FAO 2015c). Patriarchal structures and gender roles can also affect women's control over land in developed countries (Carter 2017; Alston et al. 2018b). Thus, longstanding gender inequality in land rights, security of tenure, and decision-making may constrict women's adaptation options (Smucker and Wangui 2016).

Adaptation options related to land and climate (see Chapter 6) may produce environment and development trade-offs as well as social conflicts (Hunsberger et al. 2017) and changes with gendered implications. Women's strong presence in agriculture provides opportunity to bring gender dimensions into climate change adaptation, particularly regarding food security (Glemarec 2017; Jost et al. 2016; Doss et al. 2018b). Some studies point to a potentially emancipatory role played by adaptation interventions and strategies, albeit with some limitations depending on context. For example, in developing contexts, male out-migration may cause women in socially disadvantaged groups to engage in new livelihood activities, thus challenging gendered roles (Djoudi and Brockhaus 2011; Alston 2006). Collective action and agency of women in farming households, including widows, have led to prevention of crop failure, reduced workload, increased nutritional intake, increased sustainable water management, diversified and increased income and improved strategic planning (Andersson and Gabrielsson 2012). Women's waged labour can help stabilise income from more land- and climate-dependent activities such as agriculture, hunting, or fishing (Alston et al., 2018; Ford & Goldhar, 2012). However, in developed contexts like Australia, women's participation in off-farm employment may exacerbate existing masculinisation of agriculture (Clarke and Alston 2017).

Literature suggests that land-based mitigation measures may lead to land alienation either through market or appropriation (acquisition) by the government, interfere with traditional livelihoods in rural areas, and lead to decline in women's livelihoods (Hunsberger et al. 2017). If land alienation is not prevented, existing inequities and social exclusions may be reinforced (*medium agreement*) (Mustalahti and Rakotonarivo 2014; Chomba et al. 2016; Poudyal et al. 2016). These activities also can lead to land grabs, which remain a focal point for research and local activism (Borras Jr. et al. 2011; White et al. 2012; Lahiff 2015). Cumulative effects of land-based mitigation measures may put families at risk of poverty. In certain contexts, they lead to increased conflicts. In conflict situations, women are at risk of personal violence, including sexual violence (UNEP, 2016).

Policy instruments for gender inclusive approaches to climate change, land, and sustainable development

Integrating, or mainstreaming, gender into land and climate change policy requires assessments of gender-differentiated needs and priorities, selection of appropriate policy instruments to address barriers to women's sustainable land management, and selection of gender indicators for monitoring and assessment of policy (*medium confidence*) (Huyer et al. 2015a; Alston 2014). Important sex-disaggregated data can be obtained at multiple levels, including the intra-household level (Seager 2014; Doss et al. 2018b), village- and plot-level information (Theriault et al. 2017a), and through national surveys (Agarwal 2018a; Doss et al. 2015a). Gender-disaggregated data provides a basis for selecting, monitoring and reassessing policy instruments that account for gender differentiated land and climate change needs (*medium confidence*) (Rao 2017a; Arora-Jonsson 2014; Theriault et al. 2017b) (Doss et al. 2018b). While macro-level data can reveal ongoing gender trends in SLM, contextual data are important for revealing intersectional aspects, such as the difference made by family relations, socioeconomic status, or cultural practices about land use and control (Rao 2017a; Arora-Jonsson 2014; Theriault et al. 2017b), as well as on security of land holding (Doss et al. 2018b). Indices such as the Women's Empowerment in Agriculture Index (Alkire et al. 2013b) may provide useful guidelines for quantitative data collection on gender and SLM, while qualitative studies can reveal the nature of agency and whether policies are likely to be accepted, or not, in the context of local structures, meanings, and social relations (Rao 2017b).

Women's economic empowerment, decision-making power and voice is a necessity in SLM decisions (Mello and Schmink 2017a; Theriault et al. 2017b). Policies that address barriers include: gender considerations as qualifying criteria for funding programs or access to financing for initiatives; government transfers to women under the auspices of anti-poverty programs; spending on health and education; and subsidised credit for women (*medium confidence*) (Jagger and Pender 2006; Van Koppen et al. 2013a; Theriault et al. 2017b; Agarwal 2018b). Training and extension for women to facilitate sustainable practices is also important (Mello and Schmink 2017b; Theriault et al. 2017b). Such training could be built into existing programs or structures, such as collective microenterprise (Mello and Schmink 2017b). Huyer et al. (2015) suggest that information provision (e.g., information about SLM) could be effectively dispersed through women's community-based organisations, although not in such a way that it overwhelms these organisations or supersedes their existing missions. SLM programs could also benefit from intentionally engaging men in gender-equality training and efforts (Fletcher 2017), thus recognising the relationality of gender. Recognition of the household level, including men's roles and power relations, can help avoid the de-contextualised and individualistic portrayal of women as purely instrumental actors (Rao 2017b).

Technology, policy, and programs that exacerbate women's workloads or reinforce gender stereotypes (MacGregor 2010; Huyer et al. 2015b), or which fail to recognise and value the contributions women already make (Doss et al. 2018b), may further marginalise women. Accordingly, some studies have described technological and labour interventions that can enhance sustainability while also decreasing women's workloads; for example, Vent et al. (2017) described the system of rice intensification as one such intervention. REDD+ initiatives need to be aligned with the SDGs to achieve complementary synergies with gender dimensions.

Secure land title and/or land access/control for women increases sustainable land management by increasing women's conservation efforts, increasing their productive and environmentally-beneficial agricultural investments, such as willingness to engage in tree planting and sustainable soil management (*high confidence*) as well as improving cash incomes (Higgins et al. 2018; Agarwal 2010; Namubiru-Mwaura 2014b; Doss et al. 2015b; Van Koppen et al. 2013b; Theriault et al. 2017b; Jagger and Pender 2006). According FAO (2011b, p. 5), if women had the same access to productive resources as men, the number of hungry people in the world could be reduced by 12-17%. Policies promoting secure land title include legal reforms at multiple levels, including national laws on land ownership, legal education, and legal aid for women on land ownership and access (Argawal 2018). Policies to increase women's access to land could occur through three main avenues of land acquisition: inheritance/family (Theriault et al. 2017b), state policy, and the market (Agarwal 2018). Rao (2017) recommends framing land rights as entitlements rather than as instrumental means to sustainability. This reframing may address persistent, pervasive gender inequalities (FAO 2015d).

7.5. Decision-making for Climate Change and Land

The risks posed by climate change generate considerable uncertainty and complexity for decision-makers responsible for land use decisions (*robust evidence, high agreement*). Decision-makers balance climate ambitions, encapsulated in the NDCs, with other SDGs, which will differ considerably across different regions, sociocultural conditions and economic levels (Griggs et al. 2014). The interactions across SDGs also factor into decision-making processes (Nilsson et al. 2016b). The challenge is particularly acute in Least Developed Countries where a large share of the population is vulnerable to climate change. Matching the structure of decision-making processes to local needs while connecting to national strategies and international regimes is challenging (Nilsson and Persson 2012). This section explores methods of decision-making to address the risks and inter-linkages outlined in previous sections. As a result, this section outlines policy inter-linkages with SDGs and NDCs, trade-offs and synergies in specific measures, possible challenges as well as opportunities going forward.

Even in cases where uncertainty exists, there is *medium evidence and high agreement* in the literature that it need not present a barrier to taking action, and there are growing methodological developments and empirical applications to support decision-making. Progress has been made in identifying key source of uncertainty and addressing them (Farber 2015; Lawrence et al. 2018; Bloemen et al. 2018). Many of these approaches involve principles of robustness, diversity, flexibility, learning, or choice editing (see 7.5.2).

Since the Fifth Assessment Report Chapter on Decision-making (Jones et al. 2014) considerable advances have been made in decision making under uncertainty, both conceptually and in economics (see 7.5.2), and in the social/qualitative research areas (see 7.5.3 and 7.5.4). In the land sector, the degree of uncertainty varies and is particularly challenging for climate change adaptation decisions (Hallegatte 2009; Wilby and Dessai 2010). Some types of agricultural production decisions can be made in short time-frames as changes are observed, and will provide benefits in the current time period (Dittrich et al. 2017).

7.5.1. Formal and Informal decision-making

Informal decision making facilitated by open platforms can solve problems in land and resource management by allowing evolution and adaptation, and incorporation of local knowledge (*medium confidence* (Malogdos and Yujuico 2015a; Vandersypen et al. 2007). Formal centers of decision making are those that follow fixed procedures (written down in statutes or moulded in an organisation backed by the legal system) and structures (Onibon et al. 1999). Informal centers of decision making are those following customary norms and habits based on conventions (Onibon et al. 1999) where problems are ill-structured, complex problems (Waddock 2013).

7.5.1.1. Formal Decision Making

Formal decision making processes can occur at all levels including the global, regional, national and sub-national levels (see 7.4.1). Formal decision support tools can be used, for example, by farmers, to answer "what-if" questions as to how to respond to the effects of changing climate on soils, rainfall and other conditions (Wenkel et al. 2013).

Optimal formal decision-making is based on realistic behaviour of actors, important in land-climate systems, assessed through participatory approaches, stakeholder consultations and by incorporating results from empirical analyses. Mathematical simulations and games (Lamarque et al. 2013), behavioural models in land-based sectors (Brown et al. 2017), agent-based models (ABMs) and micro-simulations are examples useful to decision-makers (Bishop et al. 2013). These decision making tools are expanded on in 7.5.2.

There are different ways to incorporate local knowledge, informal institutions and other contextual characteristics that capture non-deterministic elements, as well as social and cultural beliefs and systems more generally, into formal decision making (see 7.6.4) (*medium evidence, medium agreement*). Classic scientific methodologies now include participatory and interdisciplinary methods and approaches (Jones et al. 2014). Consequently, this broader range of approaches may very well capture informal and indigenous knowledge improving the participation of indigenous peoples in decision-making processes and thereby promote their rights to self-determination (Malogdos and Yujuico 2015b) (see Cross-Chapter Box 13: Indigenous and Local Knowledge in this chapter).

7.5.1.2. Informal Decision Making

Informal institutions have contributed to sustainable resources management (common pool resources) through creating a suitable environment for decision-making. The role of informal institutions and decision making can be particularly relevant for land use decisions and practices in rural areas in the global south and north (Huisheng 2015). Understanding informal institutions is crucial for adapting to climate change, advancing technological adaptation measures achieving comprehensive disaster management and advancing collective decision making (Karim and Thiel 2017). Informal institutions have been found to be a crucial entry point in dealing with vulnerability of communities and exclusionary tendencies impacting marginalised and vulnerable people (Mubaya and Mafongoya 2017).

Many studies underline the role of local/informal traditional institutions in the management of natural resources in different parts of the world (Yami et al. 2009; Zoogah et al. 2015; Bratton 2007; Mowo et al. 2013; Grzymala-Busse 2010). Traditional systems include: traditional silvo-pastoral management (Iran), management of rangeland resources (South Africa), natural resource management (Ethiopia, Tanzania, Bangladesh) communal grazing land management (Ethiopia) and management of conflict over natural resources (Siddig et al. 2007; Yami et al. 2011; Valipour et al. 2014; Bennett 2013; Mowo et al. 2013).

Formal-informal institutional interaction could take different shapes such as: complementary, accommodating, competing, and substitutive. There are many examples when formal institutions might obstruct, change, and hinder informal institutions (Rahman et al. 2014; Helmke and Levitsky 2004; Bennett 2013) (Osei-Tutu et al. 2014). Similarly, informal institutions can replace, undermine, and reinforce formal institutions (Grzymala-Busse 2010). In the absence of formal institutions, informal institutions gain importance requiring focus in relation to natural resources management and rights protection (Estrin and Prevezer 2011; Helmke and Levitsky 2004; Kangalawe.R.Y.M, Noe.C, Tungaraza.F.S.K 2014; Sauerwald and Peng 2013; Zoogah et al. 2015).

Community forestry comprises 22% of forests in tropical countries in contrast to large-scale industrial forestry (Hajjar et al. 2013) and is managed with informal institutions ensuring a sustainable flow of forest products and income utilising traditional ecological knowledge to determine access to resources (Singh et al. 2018). Policies that create an open platform for local debates and allow actors their own active formulation of rules strengthen informal institutions. Case studies in Zambia, Mali, Indonesia and Bolivia confirm that enabling factors for advancing the local ownership of resources and crafting durability of informal rules require recognition in laws, regulations and policies of the state (Haller et al. 2016).

7.5.2. Decision Making, Timing, Risk, and Uncertainty

This section assesses decision making literature concluding advances in methods have been made in the face of conceptual risk literature and together with a synthesis of empirical evidence, near term decisions have significant impact on costs.

7.5.2.1. Problem Structuring

Structured decision making occurs when there is scientific knowledge about cause and effect, little uncertainty, and agreement exists on values and norms relating to an issue (Hurlbert and Gupta 2016). This decision space is situated within the "known" space where cause and effect is understood and predictable (although uncertainty is not quite zero) (French 2015). Figure 7.5 displays the structured problem area in the bottom left corner corresponding with the 'known' decision making space. Decision making surrounding quantified risk assessment and risk management (7.4.3.1) occurs within this decision making space. Examples in the land and climate area include cost benefit analysis surrounding implementation of irrigation projects (Batie 2008) or adopting soil erosion practices by agricultural producers baced on anticipated profit (Hurlbert 2018b). Comprehensive risk management also occupies this decision space (Papathoma-Köhle et al. 2016), encompassing risk assessment, reduction, transfer, retention, emergency preparedness and response, and disaster recovery by combining quantified proactive and reactive approaches (Fra.Paleo 2015) (see 7.4.3).

A moderately structured decision space is characterised as one where there is either some disagreement on norms, principles, ends and goals in defining a future state or there is some uncertainty surrounding land and climate including land use, observations of land use changes, early warning and decision support systems, model structures, parameterisations, inputs, or from unknown futures informing integrated assessment models and scenarios (see Chapter 1, 1.2.2 and Cross chapter Box 1 on Scenarios). Environmental decision making often takes place in this space where there is limited information and ability to process it, and individual stakeholders make different decisions on the best future course of action (Waas et al. 2014) (*medium confidence)* (Hurlbert and Gupta 2016, 2015; Hurlbert 2018b). Figure 7.5 displays the moderately structured problem space characterised by disagreement surrounding norms on the top left hand side. This corresponds with the complex decision making space, the realm of social sciences and qualitative knowledge, where cause and effect is difficult to relate with any confidence (French 2013).

The moderately structured decision space characterised by uncertainty surrounding land and climate on the bottom right hand side of Figure 7.5 as well and corresponds to the knowable decision making space, where the realm of scientific inquiry investigates cause and effects. Here there is sufficient understanding to build models, but not enough understanding to define all parameters (French 2015).

The top right hand corner of Figure 7.5 corresponds to the 'unstructured' problem or chaotic space where patterns and relationships are difficult to discern and unknown unknowns reside (French 2013). It is in the complex but knowable space, the structured and moderately structured space, that decision making under uncertainty occurs.

7.5.2.2. Decision Making Tools

Decisions can still be made despite uncertainty *(medium confidence)*, and a wide range of possible approaches are emerging to support decision-making under uncertainty (Jones et al. 2014), applied both to adaptation and mitigation decisions.

Traditional approaches for economic appraisal, including cost benefit analysis and cost effectiveness analysis referred to in 7.5.2.1 do not handle or address uncertainty well (Hallegatte 2009) (Farber 2015) and favour decisions with short term benefits (see Cross-Chapter Box 10: Economic Dimensions in this chapter). Alternative economic decision making approaches aim to better incorporate uncertainty while still delivering adaptation goals, by selecting projects that meet their purpose across a variety of plausible futures (Hallegatte et al. 2012); so-called 'robust' decision-making approaches. These are designed to be less sensitive to uncertainty about the future (Lempert and Schlesinger 2000).

Much of the research for adaptation to climate change has focused around three main economic approaches: Real Options Analysis, Portfolio Analysis, and Robust Decision-Making. Real Options Analysis develops flexible strategies that can be adjusted when additional climate information becomes available. It is most appropriate for large irreversible investment decisions. Applications to climate adaptation are growing quickly, with most studies addressing flood risk and sea-level rise (Gersonius et al. 2013; Woodward et al. 2014; Dan 2016), but studies in land use decisions are also emerging, including identifying the optimal time to switch land use in a changing climate (Sanderson et al. 2016) and water storage (Sturm et al. 2017; Kim et al. 2017). Portfolio analysis aims to reduce risk by diversification, by planting multiple species rather than only one, in forestry (Knoke et al. 2017) or crops (Ben-Ari and Makowski 2016), for example, or in multiple locations. There may be a trade-off between robustness to variability and optimality (Yousefpour and Hanewinkel 2016; Ben-Ari and Makowski 2016); but this type of analysis can help identify and quantify trade-offs. Robust Decision Making identifies how different strategies perform under many climate outcomes, also potentially trading off optimality for resilience (Lempert 2013).

Multi-criteria decision making continues to be an important tool in the land-use sector, with the capacity to simulataneously consider multiple goals across different domains (e.g., economic, environmental, social) (Bausch et al. 2014; Alrø et al. 2016), and is thus useful as a mitigation as well as an adaptation tool. Life-cycle assessment (LCA) can also be used to evaluate emissions across a system (for example in livestock production (McClelland et al. 2018)) and identify areas to prioritise for reductions. Bottom-up Marginal Abatement Cost Curves calculate the most cost-effective cumulative potential for mitigation across different options (Eory et al. 2018).

In the climate adaptation literature, these tools may be used in adaptive management (see 7.5.4), using a monitoring, research, evaluation and learning process (cycle) to improve future management strategies (Tompkins and Adger 2004). More recently these techniques have been advanced with iterative risk management (IPCC 2014a) (see 7.4.1, 7.4.7), adaptation pathways (Downing 2012), and dynamic adaptation pathways (Haasnoot et al. 2013) (see 7.6.3). Decision making tools can be selected and adapted to fit the specific land and climate problem and decision making space. For instance, dynamic adaptation pathways processes (Haasnoot et al. 2013; Wise et al. 2014) identify and sequence potential actions based on alternative potential futures and are situated within the complex, unstructured space (see Figure 7.5). Decisions are made based on trigger points, linked to indicators and scenarios, or changing performance over time (Kwakkel et al. 2016). A key characteristic of these pathways is rather than making irreversible decisions now, decisions evolve over time, accounting for learning (see 7.6.4), knowledge, and values. Combining Dynamic Adaptive Pathways and a form of Real Options Analysis with Multi Criteria Decision Analysis has enabled changing risk over time to be included in assessment of adaptation options through a participatory learning process in New Zealand (Lawrence et al. 2019).

Scenario analysis is also situated within the complex, unstructured space (although unlike adaptation pathways, it does not allow for changes in pathway over time) and is important for identifying technology and policy instruments to ensure spatial-temporal coherence of land use allocation simulations with scenario storylines (Brown and Castellazzi 2014) and identifying technology and policy instruments for mitigation of land degradation (Fleskens et al. 2014).

While economics is usually based on the idea of a self-interested, rational agent, more recently insights from psychology are being used to understand and explain human behaviour in the field of behavioural economics (Shogren and Taylor 2008; Kesternich et al. 2017), illustrating how a range of cognitive factors and biases can affect choices (Valatin et al. 2016). These insights can be critical in supporting decision-making that will lead to more desirable outcomes relating to land and climate change. Once example of this is ‘policy nudges’ (Thaler and Sunstein 2008) which can ‘shift choices in socially desirable directions’ (Valatin et al. 2016). Tools can include framing tools, binding pre-commitments, default settings, channel factors, or broad choice bracketing (Wilson et al. 2016). Although relatively few empirical examples exist in the land sector, there is evidence that nudges could be applied successfully, for example in woodland creation (Valatin et al. 2016) and agri-environmental schemes (Kuhfuss et al. 2016) (*Medium certainty, low evidence*). Consumers can be ‘nudged’ to consume less meat (Rozin et al. 2011) or to waste food less (Kallbekken and Sælen 2013).

Programmes supporting and facilitating desired practices can have success at changing behaviour, particularly if they are co-designed by the end-users (farmers, foresters, land-users) (*medium evidence*, *high agreement*). Programmes that focus on demonstration or trials of different adaptation and mitigation measures, and facilitate interaction between farmers, industry specialists are perceived as being successful (Wreford et al. 2017; Hurlbert 2015b) but systematic evaluations of their success at changing behaviour are limited (Knook et al. 2018).

Different approaches to decision making are appropriate in different contexts. Dittrich et al. (2017) provide a guide to the appropriate application in different contexts for adaptation in the livestock sector in developed countries. While considerable advances have been made in the theoretical approaches, a number of challenges arise when applying these in practice, and partly relate to the necessity of assigning probabilities to climate projects, and the complexity of the approaches being a prohibitive factor beyond academic exercises. Formalised expert judgement can improve how uncertainty is characterised (Kunreuther et al. 2014) and these methods have been improved utilising Bayesian belief networks to synthesise expert judgements and include fault trees and reliability block diagrams to overcome standard reliability techniques (Sigurdsson et al. 2001) as well as mechanisms incorporating transparency (Ashcroft et al. 2016).

It may also be beneficial to combine decision making approaches with the precautionary principle, or the idea that lack of scientific certainty is not to postpone action when faced with serious threats or irreversible damage to the environment (Farber 2015). The precautionary principle requires cost effective measures to address serious but uncertain risks (Farber 2015). It supports a rights based policy instrument choice as consideration is whether actions or inactions harm others moving beyond traditional risk management policy considerations that surround net benefits (Etkin et al. 2012). Farber, (2015)concludes the principle has been successfully applied in relation to endangered species and situations where climate change is a serious enough problem to justify some response. There is *medium confidence* that combining the precautionary principle with integrated assessment models, risk management, and cost benefit analysis in an integrated, holistic manner, together would be a good combination of decision making tools supporting sustainable development (Farber 2015; Etkin et al. 2012).

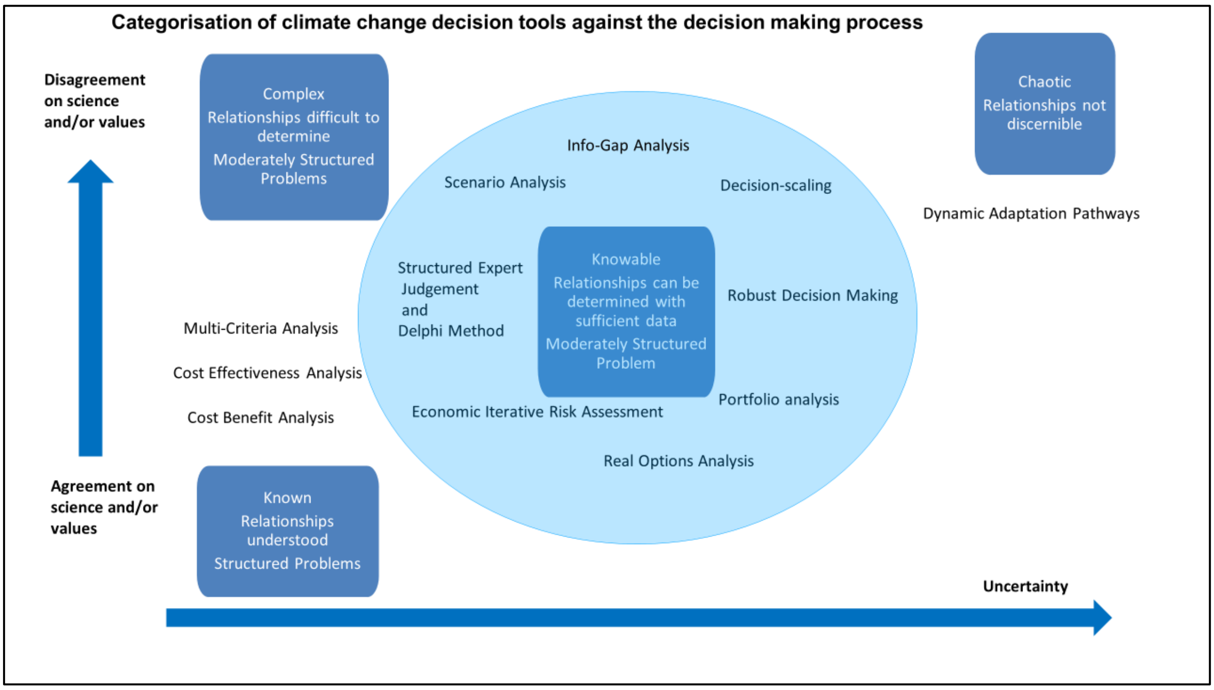


Figure 7.5 Structural and Uncertain Decision Making

7.5.2.3. Cost and timing of action

The Cross-Chapter Box 10 on Economics Dimensions deals with the costs and timing of action. In terms of policies, not only is timing important, but the type of intervention itself can influence returns (*high evidence, high agreement*). Policy packages that make people more resilient - expanding financial inclusion, disaster risk and health insurance, social protection and adaptive safety nets, contingent finance and reserve funds, and universal access to early warning systems (see 7.4.1, 7.6.3) – could save USD 100 billion a year, if implemented globally (Hallegatte et al. 2017). In Ethiopia, Kenya and Somalia, every 1 USD spent on safety net/resilience programming results in net benefits of between USD 2.3 and 3.3 (Venton 2018). Investing in resilience building activities, which increase household income by USD 365 to 450 per year in these countries, is more cost effective than providing ongoing humanitarian assistance.

There is a need to further examine returns on investment for land-based adaptation measures, both in the short and long term. Other outstanding questions include identifying specific triggers for early response. Food insecurity, for example, can occur due to a mixture of market and environmental factors (changes in food prices, animal or crop prices, rainfall patterns) (Venton 2018). The efficacy of different triggers, intervention times and modes of funding are currently being evaluated (see for example forecast based finance study (Alverson and Zommers 2018)). To reduce losses and maximise returns on investments, this information can be used to develop: 1) coordinated, agreed plans for action; 2) a clear, evidence-based decision-making process, and; 3) financing models to ensure that the plans for early action can be implemented (Clarke and Dercon 2016a).

7.5.3. Best practices of decision making toward sustainable land management

Sustainable land management is a strategy and also an outcome (Waas et al. 2014) and decision making practices are fundamental in achieving it as an outcome (*medium evidence, medium agreement*). Sustainable land management decision making is improved (*medium evidence and high* *agreement*) with ecological service mapping with three characteristics: robustness (robust modelling, measurement, and stakeholder-based methods for quantification of ecosystem service supply, demand and/or flow, as well as measures of uncertainty and heterogeneity across spatial and temporal scales and resolution); transparency (to contribute to clear information-sharing and the creation of linkages with decision support processes); and relevancy to stakeholders (people-central in which stakeholders are engaged at different stages) (Willemen et al. 2015; Ashcroft et al. 2016). Practices that advance sustainable land management include remediation practices as well as critical interventions that are reshaping norms and standards, joint implementation, experimentation, and integration of rural actors' agency in analysis and approaches in decision-making (Hou and Al-Tabbaa 2014). Best practices are identified in the literature after their implementation demonstrates effectiveness at improving water quality, the environmnent, or reducing pollution (Rudolph et al. 2015; Lam et al. 2011).

There is *medium evidence and medium agreement* about what factors consistently determine the adoption of agricultural best management practices (Herendeen and Glazier 2009) and these positively correlate to education levels, income, farm size, capital, diversity, access to information, and social networks. Attending workshops for information and trust in crop consultants are also important factors in adoption of best management practices (Ulrich-Schad, J.D., Garcia de Jalon, S., Babin, N., Paper, A. 2017; Baumgart-Getz et al. 2012). More research is needed on the sustained adoption of these factors over time (Prokopy et al. 2008).

There is *medium evidence and high agreement* that sustainable land management practices and incentives require mainstreaming into relevant policy; appropriate market based approaches, including payment for ecosystem services and public private partnerships, need better integration into payment schemes (Tengberg et al. 2016). There is *medium evidence and high agreement* that many of the best sustainable land management decisions are made with the participation of stakeholders and social learning (Section 7.6.4) (Stringer and Dougill 2013). As stakeholders may not be in agreement, either practices of mediating agreement, or modelling that depicts and mediates the effects of stakeholder perceptions in decision making may be applicable (Hou 2016; Wiggering and Steinhardt 2015).

7.5.4. Adaptive management

Adaptive management is an evolving approach to natural resource management founded on decision making approaches in other fields (such as business, experimental science, and industrial ecology) (Allen et al. 2011; Williams 2011) and decision making that overcomes management paralysis and mediates multiple stakeholder interests through use of simple steps. (Adaptive governance considers a broader socio-ecological system that includes the social context that facilitates adaptive management (Chaffin et al. 2014)). Adaptive management steps include evaluating a problem and integrating planning, analysis and management into a transparent process to build a road map focused on achieving fundamental objectives. Requirements of success are clearly articulated objectives, the explicit acknowledgment of uncertainty, and a transparent response to all stakeholder interests in the decision making process (Allen et al. 2011). Adaptive management builds on this foundation by incorporating a formal iterative process acknowledging uncertainty and achieving management objectives through a structured feedback process that includes stakeholder participation (see 7.6.4) (Foxon et al. 2009). In the adaptive management process the problem and desired goals are identified, evaluation criteria formulated, the system boundaries and context are ascertained, tradeoffs evaluated, decisions are made regarding responses and policy instruments, which are implemented, and monitored, evaluated and adjusted (Allen et al. 2011). The implementation of policy strategies and monitoring of results occurs in a continuous management cycle of monitoring, assessment and revision (Hurlbert 2015b; Newig et al. 2010; Pahl-Wostl et al. 2007) as illustrated in Figure 7.6.

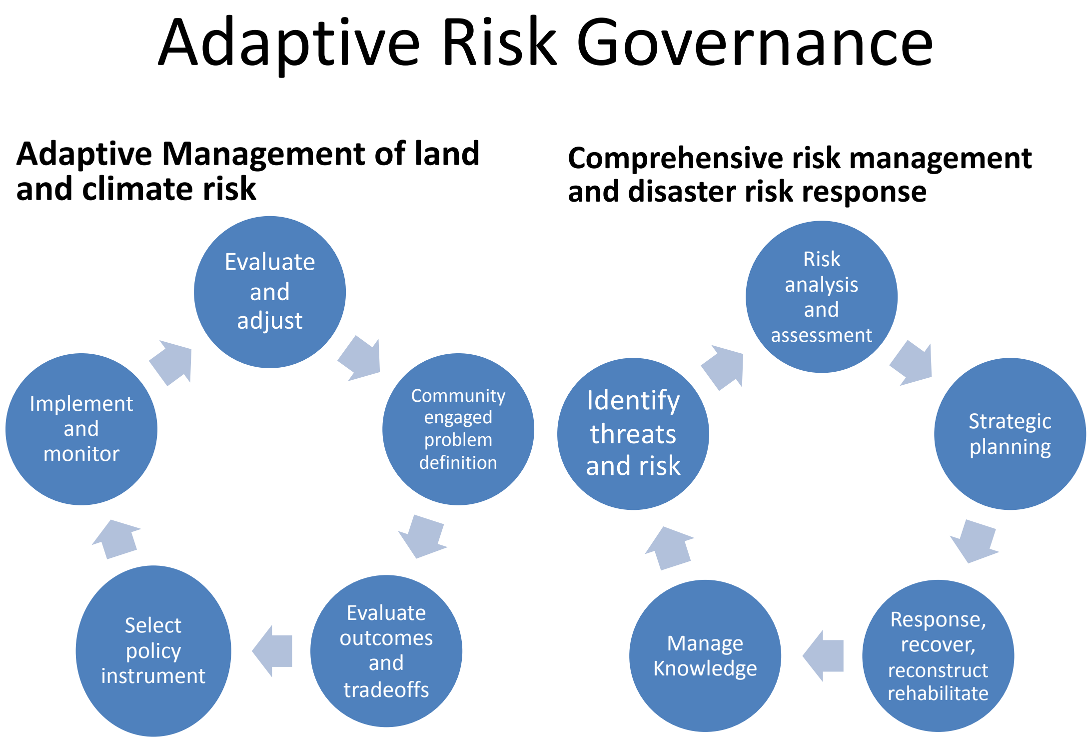


Figure 7.6 Adaptive Governance, Management, and Comprehensive Iterative Risk Management. Source: Adapted from (Ammann 2013; Allen et al. 2011)

A key focus on adaptive management is the identification and reduction of uncertainty (as described in Chapter 1, 1.2.2 and Cross-Chapter Box 1 on Scenarios) and partial controllability whereby policies used to implement an action are only indirectly responsible (for example setting a harvest rate) (Williams 2011). There is *medium evidence and high agreement* that adaptive management is an ideal method to resolve uncertainty when uncertainty and controllability (resources will respond to management) are both high (Allen et al. 2011). Where uncertainty is high, but controllability is low, developing and analysing scenarios may be more appropriate (Allen et al. 2011). Anticipatory governance has developed combining scenarios and forecasting in order to creatively design strategy to address complex, fuzzy and wicked challenges (Ramos 2014; Quay 2010) (see 7.5). Even where there is low controllability, such as in the case of climate change, adaptive management can help mitigate impacts including changes in water availability and shifting distributions of plants and animals (Allen et al. 2011).

There is *medium evidence and high agreement* that adaptive management can help reduce anthropogenic impacts of changes of land and climate including: species decline and habitat loss (participative identification, monitoring, and review of species at risk as well as decision making surrounding protective measures) (Fontaine 2011; Smith 2011) including quantity and timing of harvest of animals (Johnson 2011a), human participation in natural resource-based recreational activities including selection fish harvest quotas and fishing seasons from year to year (Martin and Pope 2011), managing competing interests of land use planners and conservationists in public lands (Moore et al. 2011), managing endangered species and minimising fire risk through land cover management (Breininger et al. 2014), land use change in hardwood forestry through mediation of hardwood plantation forestry companies and other stakeholders including those interested in water, environmnent or farming (Leys and Vanclay 2011), and sustainable land management protecting biodiversity, increasing carbon storage, and improving livelihoods (Cowie et al. 2011). There is *medium evidence and medium agreement* that despite abundant literature and theoretical explanation, there has remained imperfect realisation of adaptive management because of several challenges: lack of clarity in definition and approach, few success stories on which to build an experiential base practitioner knowledge of adaptive management, paradigms surrounding management, policy and funding that favour reactive approaches instead of the proactive adaptive management approach, shifting objectives that do not allow for the application of the approach, and failure to acknowledge social uncertainty (Allen et al. 2011). Adaptive management includes participation (7.6.4), the use of indicators (7.5.5), in order to avoid maladaptation and trade-offs while maximising synergies (7.5.6).

7.5.5. Performance indicators

Measuring performance is important in adaptive management decision-making, policy instrument implementation, and governance and can help evaluate policy effectiveness (*medium evidence, high agreement*) (Wheaton and Kulshreshtha 2017; Bennett and Dearden 2014; Oliveira Júnior et al. 2016; Kaufmann 2009). Indicators can relate to specific policy problems (climate mitigation, land degradation), sectors (agriculture, transportation etc.), and policy goals (SDGs, food security).

It is necessary to monitor and evaluate the effectiveness and efficiency of performing climate actions to ensure the long-term success of *climate* initiatives or plans. Measurable indicators are useful for climate policy development and decision-making process since they can provide quantifiable information regarding the progress of climate actions. The Paris Agreement (UNFCCC 2015) focused on reporting the progress of implementing countries' pledges, i.e., NDCs and national adaptation needs in order to examine the aggregated results of mitigation actions that have already been implemented. For the case of measuring progress toward achieving land degradation neutrality, it was suggested to use land-based indicators, i.e., trend in land cover, trends in land productivity or functioning of the land, and trends in carbon stock above and below ground (Cowie et al. 2018a). There is *medium evidence and high agreement* that indicators for measuring biodiversity and ecosystem services in response to governance at local to international scale meet the criteria of parsimony and scale specificity, are linked to some broad social, scientific and political consensus on desirable states of ecosystems and biodiversity, and include normative aspects such as environmental justice or socially just conservation (Layke 2009) (Van Oudenhoven et al. 2012) (Turnhout et al. 2014)(Häyhä and Franzese 2014), (Guerry et al. 2015)(Díaz et al. 2015).

Important in making choices of metrics and indicators is understanding that the science, linkages and dynamics in systems are complex, not amenable to be addressed by simple economic instruments, and are often unrelated to short-term management or governance scales (Naeem et al. 2015) (Muradian and Rival 2012). Thus, ideally stakeholders participate in the selection and use of indicators for biodiversity and ecosystem services and monitoring impacts of governance and management regimes on land-climate interfaces. The adoption of non-economic approaches that are part of the emerging concept of Nature's Contributions to People (NCP) could potentially elicit support for conservation from diverse sections of civil society (Pascual et al. 2017).

Recent studies increasingly incorporate the role of stakeholders and decision makers in selection of indicators for land systems (Verburg et al. 2015) including sustainable agriculture (Kanter et al. 2016), bioenergy sustainability (Dale et al. 2015), desertification (Liniger et al. 2019), and vulnerability (Debortoli et al. 2018). Kanter et al. (2016) propose a four-step cradle-to-grave approach for agriculture trade-off analysis, which involves co-evaluation of indicators and trade-offs with both stakeholders and decision-makers.

7.5.6. Maximising Synergies and Minimising Trade-offs

Synergies and trade-offs to address land and climate related measures are identified and discussed in Chapter 6. Here we outline policies supporting Chapter 6 response options (see Table 7.5), and discuss synergies and trade-offs in policy choices and interactions among policies. Trade-offs will exist between broad policy approaches. For example, while legislative and regulatory approaches may be effective at achieving environmental goals, they may be costly and ideologically unattractive in some countries. Market-driven approaches such as carbon pricing are cost effective ways to reduce emissions, but may not be favoured politically and economically (see 7.4.4). Information provision involves little political risk or ideological constraints, but behavioural barriers may limit their effectiveness (Henstra 2016). This level of trade-off is often determined by the prevailing political system.

Synergies and trade-offs also result from interaction between policies (policy interplay (Urwin and Jordan 2008)) at different levels of policy (vertical) and across different policies (horizontal) (see also section on policy coherence, 7.4.8)). If policy mixes are designed appropriately, acknowledging and incorporating trade-offs and synergies, they are better placed to deliver an outcome such as transitioning to sustainability (Howlett and Rayner 2013; Huttunen et al. 2014) (*medium evidence and medium agreement).* However, there is *limited evidence and medium agreement* that evaluating policies for coherence in responding to climate change and its impacts is not occurring, and policies are instead reviewed in a fragmented manner (Hurlbert and Gupta 2016).

1 **Table 7.5 Selection of Policies/Programmes/Instruments that support response options**

Category	Integrated Response Option	Policy instrument supporting response option
Land management in agriculture	Increased food productivity	Investment in agricultural research for crop and livestock improvement, agricultural technology transfer, inland capture fisheries and aquaculture {7.4.7} agricultural policy reform and trade liberalisation
	Improved cropland, grazing, and livestock management	Environmental farm programs/agri-environment schemes, water efficiency requirements and water transfer {3.7.5}, extension services
	Agroforestry	Payment for ecosystem services {7.4.6}
	Agricultural diversification	Elimination of agriculture subsidies {5.7.1}, environmental farm programs, agri-environmental payments {7.4.6}, rural development programmes
	Reduced grassland conversion to cropland	Elimination of agriculture subsidies, remove insurance incentives, ecological restoration {7.4.6}
	Integrated water management	Integrated governance {7.6.2}, multi-level instruments {7.4.1}
Land management in forests	Forest management, Reduced deforestation and degradation, Reforestation and forest restoration, Afforestation	REDD+, forest conservation regulations, payments for ecosystem services, recognition of forest rights and land tenure {7.4.6}, adaptive management of forests {7.5.4}, land use moratoriums, reforestation programs and investment {4.9.1}
Land management of soils	Increased soil organic carbon content, Reduced soil erosion, Reduced soil salinisation, Reduced soil compaction, Biochar addition to soil	Land degradation neutrality {7.4.5}, drought plans, flood plans, flood zone mapping{7.4.3}, technology transfer (7.4.4), land use zoning {7.4.6}, ecological service mapping and stakeholder based quantification {7.5.3}, environmental farm programs/agri-environment schemes, water efficiency requirements and water transfer {3.7.5}
Land management in all other ecosystems	Fire management	Fire suppression, prescribed fire management, mechanical treatments {7.4.3}
	Reduced landslides and natural hazards	Land use zoning {7.4.6}
	Reduced pollution - acidification	Environmental regulations, Climate mitigation (carbon pricing) {7.4.4}
	Management of invasive species / encroachment	Invasive species regulations, trade regulations {5.7.2, 7.4.6}
	Restoration and reduced conversion of coastal wetlands	Flood zone mapping {7.4.3}, land use zoning {7.4.6}
	Restoration and reduced conversion of peatlands	Payment for ecosystem services {7.4.6; 7.5.3}, standards and certification programs {7.4.6}, land use moratoriums
	Biodiversity conservation	Conservation regulations, protected areas policies
CDR Land management	Enhanced weathering of minerals	No data
	Bioenergy and BECCS	Standards and certification for sustainability of biomass and land use {7.4.6}
Demand management	Dietary change	Awareness campaigns/education, changing food choices through nudges, synergies with health insurance and policy {5.7.2}
	Reduced post-harvest losses Reduced food waste (consumer or retailer), Material substitution	Agricultural business risk programs {7.4.8}; regulations to reduce and taxes on food waste, Improved shelf life, circularising the economy to produce substitute goods, carbon pricing, sugar/fat taxes {5.7.2}
Supply management	Sustainable sourcing	Food labelling, innovation to switch to food with lower environmental footprint, public procurement policies {5.7.2}, standards and certification programs {7.4.6}
	Management of supply chains	Liberalised international trade {5.7.2}, food purchasing and storage policies of governments, standards and certification programs {7.4.6}, regulations on speculation in food systems
	Enhanced urban food systems	Buy local policies; land use zoning to encourage urban agriculture, nature-based solutions and green infrastructure in cities; incentives for technologies like vertical farming
	Improved food processing and retailing, Improved energy use in food systems	Agriculture emission trading {7.4.4}; investment in research and development for new technologies; certification
Risk management	Management of urban sprawl	Land use zoning {7.4.6}
	Livelihood diversification	Climate-smart agriculture policies, adaptation policies, extension services {7.5.6}
	Disaster risk management Risk sharing instruments	Disaster risk reduction {7.5.4; 7.4.3}, adaptation planning Insurance, iterative risk management, Cat bonds, risk layering, contingency funds {7.4.3}, agriculture business risk portfolios {7.4.8}

2

Cross-Chapter Box 9 on Illustrative Climate and Land Pathways

Katherine Calvin (The United States of America), Edouard Davin (France/Switzerland), Margot Hurlbert (Canada), Jagdish Krishnaswamy (India), Alexander Popp (Germany), Prajal Pradhan (Nepal/Germany)

Future development of socioeconomic factors and policies influence the evolution of the land-climate system, among others in terms of the land used for agriculture and forestry. Climate mitigation policies can also have a major impact on land use, especially in scenarios consistent with the climate targets of the Paris Agreement. This includes the use of bio-energy or Carbon Dioxide Removal (CDR), such as bioenergy with carbon dioxide capture and storage (BECCS) and afforestation. Land-based mitigation options have implications for GHG fluxes, desertification, land degradation, food insecurity, ecosystem services and other aspects of sustainable development.

Illustrative Futures

The three illustrative futures are based on the Shared Socioeconomic Pathways (SSPs; (O'Neill et al. 2014c; Riahi et al. 2017b; Popp et al. 2017; Rogelj et al. 2018b); Cross-Chapter Box 1 in Chapter 1). SSP1 is a scenario with a broad focus on sustainability including a focus on human development, technological development, nature conservation, globalised economy, economic convergence and early international cooperation including moderate levels of trade. The scenario assumes a low population growth, relatively high agricultural yields and a move towards less-meat intensive diets (van Vuuren et al. 2017b). Dietary change and reductions in food waste reduce agricultural demands and well-managed land systems enable reforestation and/or afforestation. SSP2 is a scenario in which societal as well as technological development follows historical patterns (Fricko et al. 2017). Land-based CDR is achieved through bioenergy and BECCS, and to a lesser degree by afforestation and reforestation. SSP3 is a scenario with limited technological progress and land-use regulation. Agricultural demands are high due to resource-intensive consumption and a regionalised world leads to reduced flows for agricultural goods. In SSP3, forest mitigation activities and abatement of agricultural GHG emissions are limited due to major implementation barriers such as low institutional capacities in developing countries and delayed as a consequence of low international cooperation (Fujimori et al. 2017a). Emissions reductions are achieved primarily through the energy sector, including the use of bioenergy and BECCS.

Policies in the Illustrative Futures

SSPs are complemented by a set of shared policy assumptions (Kriegler et al. 2014), indicating the types of policies that may be implemented in each future world. IAMs represent the effect of these policies on the economy, energy system, land use and climate with the caveat that they are assumed to be effective or in some cases the policy goals (e.g., dietary change) are imposed rather than explicitly modelled. In the real world, there are various barriers that can make policy implementation more difficult (see 7.4.9). These barriers will be generally higher in SSP3 than SSP1.

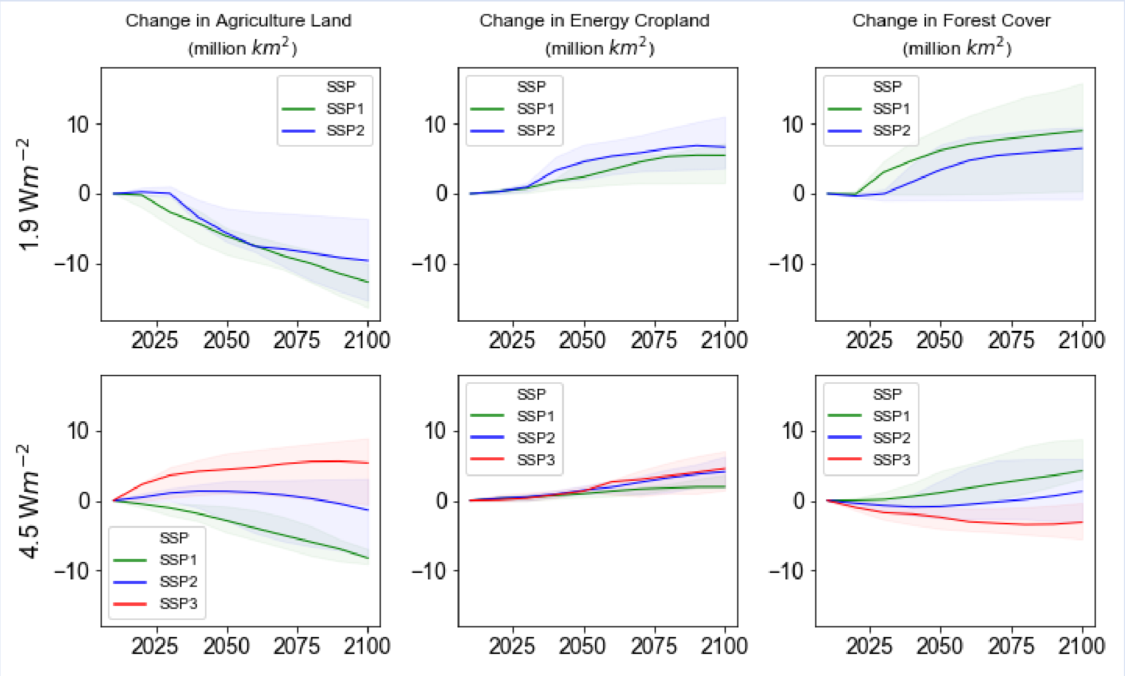
SSP1: A number of policies could support this SSP1 future including: effective carbon pricing, emission trading schemes (including net CO_2 emissions from agriculture), carbon taxes, regulations limiting GHG emissions and air pollution, forest conservation (mix of land-sharing and land sparing) through participation, incentives for ecosystem services and secure tenure, and protecting the environment, microfinance, crop and livelihood insurance, agriculture extension services, agricultural production subsidies, low export tax and import tariff rates on agricultural goods, dietary awareness campaigns, regulations to reduce and taxes on food waste, improved shelf life, sugar/fat taxes, and instruments supporting sustainable land management including payment for ecosystem services, land use zoning, REDD+, standards and certification for sustainable biomass production practices, legal reforms on land ownership and access, legal aid, legal education, including reframing these policies as entitlements for women and small agricultural producers (rather than sustainability) (O'Neill et al. 2017; van Vuuren et al. 2017b) (see 7.4).

SSP2: The same policies that support the SSP1 could support the SSP2 but may be less effective and only moderately successful. Policies may be challenged by adaptation limits (7.4.9), inconsistency in formal and informal institutions in decision making (7.5.1) or result in maladaptation (7.4.7). Moderately successful sustainable land management policies result in some land competition. Land degradation neutrality is moderately successful. Successful policies include those supporting bioenergy and BECCS (Rao et al. 2017; Riahi et al. 2017b; Fricko et al. 2017) (see 7.4.6).

SSP3: Policies that exist in SSP1 may or may not exist in SSP3, and are ineffective (O'Neill et al. 2014c). There are challenges to implementing these policies, as in SSP2. In addition, ineffective sustainable land management policies result in competition for land between agriculture and mitigation. Land degradation neutrality is not achieved (Riahi et al. 2017b). Successful policies include those supporting bioenergy and BECCS (see 7.4.6) (Kriegler et al. 2017; Fujimori et al. 2017a; Rao et al. 2017). Demand side food policies are absent and supply side policies predominate. There is no success in advancing land ownership and access policies for agricultural producer livelihood (7.6.5).

Land use and land cover change

Agricultural area in SSP1 declines as a result of the low population growth, agricultural intensification, low meat consumption, and low food waste. In contrast, SSP3 has high population and strongly declining rates of crop yield growth over time, resulting in increased agricultural land area. The SSP2 falls somewhere in between, with its modest growth in all factors. In the climate policy scenarios consistent with the Paris Agreement, bioenergy/BECCS and reforestation/afforestation play an important role in SSP1 and SSP2. The use of these options, and the impact on land, is larger in scenarios that limit radiative forcing in 2100 to 1.9 Wm^{-2} than in the 4.5 Wm^{-2} scenarios. In SSP3, the expansion of land for agricultural production implies that the use of land-related mitigation options is very limited, and the scenario is characterised by continued deforestation.



Cross-Chapter Box 9 Figure 1: Changes in agricultural land (left), energy cropland (middle) and forest cover (right) under three different SSPs (colours) and two different warming levels (rows). Agricultural land includes both pasture and non-energy cropland. Colours indicate SSPs, with SSP1 shown in green,

SSP2 in blue, and SSP3 in red. Shaded area show the range across all IAMs; lines show the median across all models. Models are only included in a figure if they provided results for all SSPs in that panel. There is no SSP3 in the top row, as 1.9 Wm^{-2} is infeasible in this world. Data is from an update of the IAMC Scenario Explorer developed for the SR15 (Huppmann et al. 2018; Rogelj et al. 2018a).

Implications for mitigation and other land challenges

The combination of baseline emissions development, technology options, and policy support makes it is much easier to reach the climate targets in the SSP1 scenario than in the SSP3 scenario. As a result, carbon prices are much higher in SSP3 than in SSP1. In fact, the 1.9 Wm^{-2} target was found to be infeasible in the SSP3 world (Cross-Chapter Box 9 Table 1). Energy system CO_2 emissions reductions are greater in the SSP3 than in the SSP1 to compensate for the higher land-based CO_2 emissions.

Accounting for mitigation and socioeconomics alone, food prices (an indicator of food insecurity) are higher in SSP3 than in the SSP1 and higher in the 1.9 Wm^{-2} than in the 4.5 Wm^{-2} (Cross-Chapter Box 9 Table 1). Forest cover is higher in the SSP1 than the SSP3 and higher in the 1.9 Wm^{-2} than in the 4.5 Wm^{-2}. Water withdrawals and water scarcity are in general higher in the SSP3 than the SSP1 (Hanasaki et al. 2013a; Graham et al. 2018b) and higher in scenarios with more bioenergy (Hejazi et al. 2014c); however, these indicators have not been quantified for the specific SSP-RCP combinations discussed here.

Climate change, results in higher impacts and risks in the 4.5 Wm^{-2} world than in the 1.9 Wm^{-2} world for a given SSP and these risks are exacerbated in SSP3 compared to SSP1 and SSP2 due to population's higher exposure and vulnerability. For example, the risk of fire is higher in warmer worlds; in the 4.5 Wm^{-2} world, the population living in fire prone regions is higher in the SSP3 (646 million) than in the SSP2 (560 million) (Knorr et al. 2016). Global exposure to multi-sector risk quadruples between the 1.5ºC[1] and 3ºC and is a factor of six higher in the SSP3-3ºC than in the SSP1-1.5ºC (Byers et al. 2018). Future risks resulting from desertification, land degradation and food insecurity are lower in the SSP1 compared to SSP3 at the same level of warming. For example, the transition moderate to high risk of food insecurity occurs between 1.3 and 1.7ºC for the SSP3, but not until 2.5 to 3.5ºC in the SSP1 (Section 7.2).

Table 1: Quantitative indicators for the illustrative pathways. Each cell shows the mean, minimum, and maximum value across IAM models for each indicator and each pathway in 2050 and 2100. All IAMs that provided results for a particular pathway are included here. Note that these indicators exclude the implications of climate change. Data is from an update of the IAMC Scenario Explorer developed for the SR15 (Huppmann et al. 2018; Rogelj et al. 2018b).

		SSP1		SSP2		SSP3	
		1.9 Wm^{-2} mean (min, max)	**4.5 Wm^{-2} mean (min, max)**	**1.9 Wm^{-2} mean (min, max)**	**4.5 Wm^{-2} mean (min, max)**	**1.9 Wm^{-2} mean (min, max)**	**4.5 Wm^{-2} mean (min, max)**
Population (billion)	2050	8.5 (8.5, 8.5)	8.5 (8.5, 8.5)	9.2 (9.2, 9.2)	9.2 (9.2, 9.2)	N/A	10.0 (10.0, 10.0)
	2100	6.9 (7.0, 6.9)	6.9 (7.0, 6.9)	9.0 (9.0, 9.0)	9.0 (9.1, 9.0)	N/A	12.7 (12.8, 12.6)
Change in GDP per capita (% rel to	2050	170.3 (380.1,	175.3 (386.2,	104.3 (223.4,	110.1 (233.8,	N/A	55.1 (116.1, 46.7)

[1] FOOTNOTE: Pathways that limit radiative forcing in 2100 to 1.9 Wm^{-2} result in median warming in 2100 to 1.5ºC in 2100 (Rogelj et al. 2018b). Pathways limiting radiative forcing in 2100 to 4.5 Wm^{-2} result in median warming in 2100 above 2.5ºC (IPCC 2014).

2010)		130.9)	166.2)	98.7)	103.6)		
	2100	528.0 (1358.4, 408.2)	538.6 (1371.7, 504.7)	344.4 (827.4, 335.8)	356.6 (882.2, 323.3)	N/A	71.2 (159.7, 49.6)
Change in forest cover (Mkm^2)	2050	3.4 (9.4, -0.1)	0.6 (4.2, -0.7)	3.4 (7.0, -0.9)	-0.9 (2.9, -2.5)	N/A	-2.4 (-1.0, -4.0)
	2100	7.5 (15.8, 0.4)	3.9 (8.8, 0.2)	6.4 (9.5, -0.8)	-0.5 (5.9, -3.1)	N/A	-3.1 (-0.3, -5.5)
Change in cropland (Mkm^2)	2050	-1.2 (-0.3, -4.6)	0.1 (1.5, -3.2)	-1.2 (0.3, -2.0)	1.2 (2.7, -0.9)	N/A	2.3 (3.0, 1.2)
	2100	-5.2 (-1.8, -7.6)	-2.3 (-1.6, -6.4)	-2.9 (0.1, -4.0)	0.7 (3.1, -2.6)	N/A	3.4 (4.5, 1.9)
Change in energy cropland (Mkm^2)	2050	2.1 (5.0, 0.9)	0.8 (1.3, 0.5)	4.5 (7.0, 2.1)	1.5 (2.1, 0.1)	N/A	1.3 (2.0, 1.3)
	2100	4.3 (7.2, 1.5)	1.9 (3.7, 1.4)	6.6 (11.0, 3.6)	4.1 (6.3, 0.4)	N/A	4.6 (7.1, 1.5)
Change in pasture (Mkm^2)	2050	-4.1 (-2.5, -5.6)	-2.4 (-0.9, -3.3)	-4.8 (-0.4, -6.2)	-0.1 (1.6, -2.5)	N/A	2.1 (3.8, -0.1)
	2100	-6.5 (-4.8, -12.2)	-4.6 (-2.7, -7.3)	-7.6 (-1.3, -11.7)	-2.8 (1.9, -5.3)	N/A	2.0 (4.4, -2.5)
Change in other natural land (Mkm^2)	2050	0.5 (1.0, -4.9)	0.5 (1.7, -1.0)	-2.2 (0.6, -7.0)	-2.2 (0.7, -2.2)	N/A	-3.4 (-2.0, -4.4)
	2100	0.0 (7.1, -7.3)	1.8 (6.0, -1.7)	-2.3 (2.7, -9.6)	-3.4 (1.5, -4.7)	N/A	-6.2 (-5.4, -6.8)
Carbon price (2010 US$ per tCO_2)[a]	2050	510.4 (4304.0, 150.9)	9.1 (35.2, 1.2)	756.4 (1079.9, 279.9)	37.5 (73.4, 13.6)	N/A	67.2 (75.1, 60.6)
	2100	2164.0 (35037.7, 262.7)	64.9 (286.7, 42.9)	4353.6 (10149.7, 2993.4)	172.3 (597.9, 112.1)	N/A	589.6 (727.2, 320.4)
Food price (Index 2010=1)	2050	1.2 (1.8, 0.8)	0.9 (1.1, 0.7)	1.6 (2.0, 1.4)	1.1 (1.2, 1.0)	N/A	1.2 (1.7, 1.1)
	2100	1.9 (7.0, 0.4)	0.8 (1.2, 0.4)	6.5 (13.1, 1.8)	1.1 (2.5, 0.9)	N/A	1.7 (3.4, 1.3)
Increase in Warming above pre-industrial (ºC)	2050	1.5 (1.7, 1.5)	1.9 (2.1, 1.8)	1.6 (1.7, 1.5)	2.0 (2.0, 1.9)	N/A	2.0 (2.1, 2.0)
	2100	1.3 (1.3, 1.3)	2.6 (2.7, 2.4)	1.3 (1.3, 1.3)	2.6 (2.7, 2.4)	N/A	2.6 (2.6, 2.6)
Change in per capita demand for food, crops (% rel to 2010)[b]	2050	6.0 (10.0, 4.5)	9.1 (12.4, 4.5)	4.6 (6.7, -0.9)	7.9 (8.0, 5.2)	N/A	2.4 (5.0, 2.3)
	2100	10.1 (19.9, 4.8)	15.1 (23.9, 4.8)	11.6 (19.2, -10.8)	11.7 (19.2, 4.1)	N/A	2.0 (3.4, -1.0)
Change in per capita demand for food, animal products (% rel to 2010)[b,c]	2050	6.9 (45.0, -20.5)	17.9 (45.0, -20.1)	7.1 (36.0, 1.9)	10.3 (36.0, -4.2)	N/A	3.1 (5.9, 1.9)
	2100	-3.0 (19.8, -27.3)	21.4 (44.1, -26.9)	17.0 (39.6, -24.1)	20.8 (39.6, -5.3)	N/A	-7.4 (-0.7, -7.9)
AFOLU CH_4 Emissions (%	2050	-39.0 (-3.8, -68.9)	-2.9 (22.4, -23.9)	-11.7 (31.4, -59.4)	7.5 (43.0, -15.5)	N/A	15.0 (20.1, 3.1)

relative to 2010)	2100	-60.5 (-41.7, -77.4)	-47.6 (-24.4, -54.1)	-40.3 (33.1, -58.4)	-13.0 (63.7, -45.0)	N/A	8.0 (37.6, -9.1)
AFOLU N_2O Emissions (% relative to 2010)	2050	-13.1 (-4.1, -26.3)	0.1 (34.6, -14.5)	8.8 (38.4, -14.5)	25.4 (37.4, 5.5)	N/A	34.0 (50.8, 29.3)
	2100	-42.0 (4.3, -49.4)	-25.6 (-3.4, -51.2)	-1.7 (46.8, -37.8)	19.5 (66.7, -21.4)	N/A	53.9 (65.8, 30.8)
Cumulative Energy CO_2 Emissions until 2100 ($GtCO_2$)		428.2 (1009.9, 307.6)	2787.6 (3213.3, 2594.0)	380.8 (552.8, -9.4)	2642.3 (2928.3, 2515.8)	N/A	2294.5 (2447.4, 2084.6)
Cumulative AFOLU CO_2 Emissions until 2100 ($GtCO_2$)		-127.3 (5.9, -683.0)	-54.9 (52.1, -545.2)	-126.8 (153.0, -400.7)	40.8 (277.0, -372.9)	N/A	188.8 (426.6, 77.9)

[a] The SSP2-19 is infeasible in two models. One of these models sets the maximum carbon price in the SSP1-19; the carbon price range is smaller for the SSP2-19 as this model is excluded there. Carbon prices are higher in the SSP2-19 than the SSP1-19 for every model that provided both simulations.

[a] Food demand estimates include waste.

[b] Animal product demand includes meat and dairy.

Summary

Future pathways for climate and land use include portfolios of response and policy options. Depending on the response options included, policy portfolios implemented, and other underlying socioeconomic drivers, these pathways result in different land-use consequences and their contribution to climate change mitigation. Agricultural area declines by more than 5 Mkm^2 in one SSP but increases by as much as 5 Mkm^2 in another. The amount of energy cropland ranges from nearly zero to 11 Mkm^2, depending on the SSP and the warming target. Forest area declines in the SSP3 but increases substantially in the SSP1. Subsequently, these pathways have different implications for risks related to desertification, land degradation, food insecurity, and terrestrial greenhouse gas fluxes, as well as ecosystem services, biodiversity, and other aspects of sustainable development.

7.5.6.1. *Trade-offs and Synergies between ES*

Unplanned or unintentional trade-offs and synergies between policy driven response options related to ecosystem service (ES) can happen over space (e.g., upstream-downstream, IWM 3.7.5.2) or intensify over time (reduced water in future dry-season due to growing tree plantations, 6.4.1). Trade-offs can occur between two or more ecosystem services (land for climate mitigation vs food 6.2, 6.3, 6.4, Cross-Chapter Box 8: Ecosystem services, Chapter 6; Cross-Chapter Box 9: Ilustrative climate and land pathways, Chapter 6), and between scales such as forest biomass based livelihoods versus global ES carbon storage (Chhatre and Agrawal 2009)(*medium evidence*, *medium agreement*). Tradeoffs can be reversible or irreversible (Rodríguez et al. 2006; Elmqvist et al. 2013)(for example a soil carbon sink is reversible (6.4.1.1)

Although there is *robust evidence* and *high agreement* that ES are important for human well-being, the relationship between poverty alleviation and ES can be surprisingly complex, understudied and dependent on the political economic context; current evidence is largely about provisioning services and often ignores multiple dimensions of poverty (Suich et al. 2015; Vira et al. 2012). Spatially explicit mapping and quantification of stake-holder choices vis-à-vis distribution of various ES can help enhance synergies and reduce trade-offs (Turkelboom et al. 2018; Locatelli et al. 2014)(see 7.5.5).

7.5.6.2. *Sustainable Development Goals (SDGs): Synergies and Trade-offs*

The SDGs, an international persuasive policy instrument, apply to all countries, and measure sustainable and socially just development of human societies at all scales of governance (Griggs et al. 2013). The UN SDGs rest on the premise that the goals are mutually reinforcing and there exist inherent linkages, synergies and trade-offs (to a greater or lesser extent) between and within the sub-goals (Fuso Nerini et al. 2018; Nilsson et al. 2016b)(Le Blanc 2015). There is *high confidence* that opportunities, trade-offs and co-benefits are context and region specific and depend on a variety of political, national and socio-economic factors (Nilsson et al. 2016b) depending on perceived importance by decision and policy makers (Figure 7.7, Table 7.6 below). Aggregation of targets and indicators at the national level can mask severe biophysical and socio-economic trade-offs at local and regional scales (Wada et al. 2016).

There is *medium evidence and high agreement* that SDGs must not be pursued independently, but in a manner that recognises trade-offs and synergies with each other, consistent with a goal of 'policy coherence.' Policy coherence also refers to spatial trade-offs and geo-political implications within and between regions and countries implementing SDGs. For instance, supply side food security initiatives of land-based agriculture are impacting marine fisheries globally through creation of dead-zones due to agricultural run-off (Diaz and Rosenberg 2008).

SDG 7 (Affordable and clean energy) and efficient and less carbon intensive transportation (SDG 7 and 9) are important SDGs related to mitigation with adaptation co-benefits, but have local trade-offs with biodiversity and competing uses of land and rivers (see Case Study: Green Energy: Biodiversity Conservation vs Global Environment Targets) (*medium evidence, high agreement*) (Bogardi et al. 2012) (Nilsson and Berggren 2000; Hoeinghaus et al. 2009) (Winemiller et al. 2016). This has occurred despite emerging knowledge about the role that rivers and riverine ecosystems play in human development and in generating global, regional and local ecosystem services (Nilsson and Berggren 2000; Hoeinghaus et al. 2009). The transformation of river ecosystems for irrigation, hydropower and water requirements of societies worldwide is the biggest threat to fresh-water and estuarine biodiversity and ecosystems services (Nilsson and Berggren 2000; Vörösmarty et al. 2010). These projects address important energy and water-related demands, but their economic benefits are often overestimated in relation to trade-offs with respect to food (river capture fisheries), biodiversity and downstream ecosystem services (Winemiller et al. 2016). Some trade-offs and synergies related to SDG7 impact aspirations of greater welfare and well-being, as well as physical and social infrastructure for sustainable development (Fuso Nerini et al. 2018)(see 7.5.6.1 where tradeoffs exist between climate mitigation and food).

There are also spatial trade-offs related to large river diversion projects and export of "virtual water" through water intensive crops produced in one region exported to another, with implications for food-security, water security and downstream ecosystem services of the exporting region (Hanasaki et al. 2010; Verma et al. 2009). Synergies include cropping adaptation that increase food system production and eliminate hunger (SDG2) (Rockström et al. 2017; Lipper et al. 2014a; Neufeldt et al. 2013). Well-adapted agricultural systems have shown to have synergies - positive returns on investment and contribution to safe drinking water, health, biodiversity and equity goals (DeClerck 2016). Assessing the water footprint of different sectors at the river basin scale can provide insights for interventions and decision making(Zeng et al. 2012)

Sometimes the trade-offs in SDGs can arise in the articulation and nested hierarchy of seventeen goals and targets under them. In terms of aquatic life and ecosystems, there is an explicit SDG for sustainable management of marine life (SDG 14, Life below Water). There is no equivalent goal exclusively for fresh-water ecosystems, but hidden under SDG 6 (Clean Water and Sanitation) out of 6 listed targets, the sixth target is about protecting and restoring water-related ecosystems, which suggests a lower order of global priority compared to being listed as a goal in itself (e.g., SDG 14).

There is *limited evidence and limited agreement* that binary evaluations of individual SDGs and synergies and trade-offs that categorise interactions as either 'beneficial' or 'adverse' may be subjective and challenged further by the fact that feedbacks can often not be assigned as unambiguously positive or negative (Blanc et al. 2017). The Special Report on Global Warming of 1.5°C notes, "A reductive focus on specific SDGs in isolation may undermine the long-term achievement of sustainable climate change mitigation" (Holden et al. 2017). Greater work is needed to tease out these relationships; studies that include quantitative modelling (see Karnib 2017) and nuanced scoring scales (ICSU 2017) of these relationships have started.

A nexus approach is increasingly being adopted to explore synergies and trade-offs between a select subset of goals and targets (such as the interaction between water, energy, and food (see, e.g., Yumkella and Yillia 2015; Conway et al. 2015; Ringler et al. 2015)). However, even this approach ignores systemic properties and interactions across the system as a whole (Weitz et al. 2017a). Pursuit of certain targets in one area can generate rippling effects across the system, and these effects in turn can have secondary impacts on yet other targets. (Weitz et al. 2017a) found that SDG target 13.2 (climate change policy/ planning) is influenced by actions in six other targets. SDG 13.1 (climate change adaption) and also 2.4 (food production) receive the most positive influence from progression in other targets.

There is *medium evidence and high agreement* that to be effective, truly sustainable, and to reduce or mitigate emerging risks, SDGs need knowledge dissemination and policy initiatives that recognise and assimilate concepts of co-production of ecosystem services in socio-ecological systems, cross-scale linkages, uncertainty, spatial and temporal trade-offs between SDGs and ecosystem services that recognise biophysical, social and political constraints and an understanding of how social change occurs at various scales (Rodríguez et al. 2006; Norström et al. 2014; Palomo et al. 2016). Several methods and tools are proposed in literature to address and understand SDG interactions. Nilsson et al. (2016a) suggest going beyond a simplistic synergies-trade-offs framing to understanding various relationship dimensions proposing a seven-point scale to understand these interactions.

This approach, and the identification of clusters of synergy, can help indicate that government ministries work together or establish collaborations to reach their specific goals. Finally, context specific analysis is needed. Synergies and trade-offs will depend on the natural resource base (such as land or water availability), governance arrangements, available technologies, and political ideas in a given location (Nilsson et al. 2016b). Figure 7.7 below shows that at the global scale there is less uncertainty in the evidence surrounding SDGs, but also less agreement on norms, priorities and values for SDG implementation. Although there is some agreement on the regional and local scale surrounding SDGs, there is higher certainty on the science surrounding ESs.

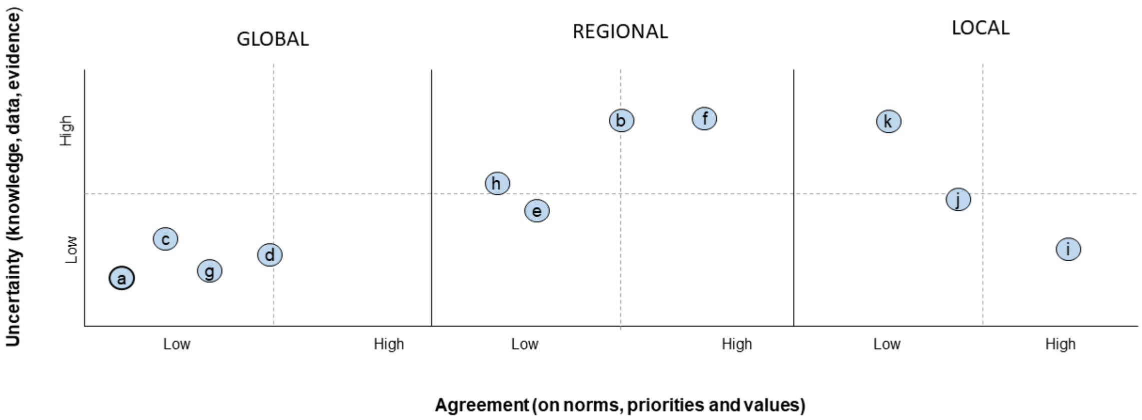


Figure 7.7 and Table 7.6: Risks at various scales, levels of uncertainty and agreement in relation to trade-offs among SDGs and other goals

	Land-climate-society Hazard	SDGs impacted or involved in mutual trade-offs	Selected Literature
a	Decline of fresh-water and riverine ecosystems	2,3,6,7,8,12,16,18	(Falkenmark 2001; Zarfl et al. 2014; Canonico et al. 2005)
b	Forest browning	3, 8,13,15,	(Verbyla 2011; Krishnaswamy et al. 2014; McDowell and Allen 2015b; Anderegg et al. 2013; Samanta et al. 2010)
c	Exhaustion of ground water	1,3,6,8,11,12,13,18	(Barnett and O’Neill 2010; Wada et al. 2010; Harootunian 2018; Dalin et al. 2017; Rockström, Johan Steffen et al. 2009; Falkenmark 2001)
d	Loss of biodiversity	6,7,12,15,18	(Pereira et al. 2010; Pascual et al. 2017; Pecl et al. 2017; Jumani et al. 2017, 2018)
e	Extreme events in cities and towns	3,6,11,13	(Douglas et al. 2008; Stone et al. 2010; Chang et al. 2007; Hanson et al. 2011);

f	Stranded assets	8, 9,11,12,13	(Ansar et al. 2013; Chasek et al. 2015; Melvin et al. 2017; Surminski 2013; Hallegatte et al. 2013; Larsen et al. 2008; Nicholls and Cazenave 2010)
g	Expansion of the agricultural frontier into tropical forests	15, 13	(Celentano et al. 2017; Nepstad et al. 2008; Bogaerts et al. 2017; Fearnside 2015; Beuchle et al. 2015; Grecchi et al. 2014)
h	Food and nutrition security	2,1,3,10, 11	(Hasegawa et al. 2018a; Frank et al. 2017; Fujimori et al. 2018b; Zhao et al. 2017)
i	Emergence of Infectious Diseases	3,1,6, 10, 11, 12, 13	(Wu et al. 2016; Patz et al. 2004; McMichael et al. 2006; Young et al. 2017b; Smith et al. 2014a; Tjaden et al. 2017; Naicker 2011)
j	Decrease in Agricultural Productivity	2,1,3,10, 11, 13	(Porter et al. 2014; Müller et al. 2013; Rosenzweig et al. 2014)
k	Expansion of farm and fish ponds	1, 2, 3, 6, 8, 10, 13, 14	(Kale 2017; Boonstra and Hanh 2015)

Sustainable Development Goals
1: No Poverty
2: Zero Hunger
3: Good Health and Well-being
4: Quality Education
5: Gender Equality
6: Clean Water and Sanitation
7: Affordable and Clean Energy
8: Decent Work and Economic Growth
9: Industry, Innovation and Infrastructure
10: Reduced Inequality
11: Sustainable Cities and Communities
12: Responsible Consumption and Production
13: Climate Action
14: Life Below Water
15: Life on Land
16: Peace and Justice Strong Institutions
17: Partnerships to achieve the goals

7.5.6.3. **Forests and agriculture**

Retaining existing forests, restoring degraded forest and afforestation are response options for climate change mitigation with adaptation benefits (6.4.1). Policies at various levels of governance that foster ownership, autonomy, and provide incentives for forest cover can reduce trade-off between carbon sinks in forests and local livelihoods (especially when the size of forest commons is sufficiently large) (Chhatre and Agrawal 2009; Locatelli et al. 2014) (see Table 7.6 this section, Case Study: Forest conservation instruments: REDD+ in the Amazon and India, 7.4.6).

Forest restoration for mitigation through carbon sequestration and other ecosystem services or co-benefits (e.g., hydrologic, NTFP, timber and tourism) can be passive or active (although both types largely exclude livestock). Passive restoration is more economically viable in relation to restoration costs as well as co-benefits in other ESs, calculated on a NPV basis, especially under flexible carbon credits (Cantarello et al. 2010). Restoration can be more cost effective with positive socioeconomic and biodiversity conservation outcomes, if costly and simplistic planting schemes are avoided (Menz et al. 2013). Passive restoration takes longer to demonstrate co-benefits and net economic gains, can be confused with land abandonment in some regions and countries, and therefore secure land-tenure at individual or community scales is important for its success (Zahawi et al. 2014). Potential approaches include improved markets and payment schemes for ecosystem services (Tengberg et al. 2016)(see

7.4.6). Proper targeting of incentive schemes and reducing poverty through access to ecosystem services requires knowledge regarding the distribution of beneficiaries and about those whose livelihoods are likely to be impacted in what manner (Nayak et al. 2014; Loaiza et al. 2015; Vira et al. 2012). Institutional arrangements to govern ecosystems are believed to synergistically influence maintenance of carbon storage and forest based livelihoods, especially when they incorporate local knowledge and decentralised decision making (Chhatre and Agrawal 2009). Earning carbon credits from reforestation with native trees involves a higher cost of the certification and validation processes, increasing the temptation to choose fast-growing (perhaps non-native) species with consequences for native biodiversity. Strategies and policies that aggregate landowners or forest dwellers are needed to reduce the cost to individuals and payment for ecosystem services (PES) schemes can generate synergies (Bommarco et al. 2013; Chhatre and Agrawal 2009). Bundling several PES schemes that address more than one ES can increase income generated by forest restoration (Brancalion et al. 2012). In the forestry sector, there is evidence that adaptation and mitigation can be fostered in concert. A recent assessment of the California forest offset program shows that such programs, by compensating individuals and industries for forest conservation, can deliver mitigation and sustainability co-benefits (Anderson et al. 2017). Adaptive forest management focussing on re-introducing native tree species can provide both mitigation and adaptation benefit by reducing fire risk and increasing carbon storage (Astrup et al. 2018).

In the agricultural sector, there has been little published empirical work on interactions between adaptation and mitigation policies. Smith and Oleson (2010) describe potential relationships, focussing particularly on the arable sector and predominantly on mitigation efforts and more on measures than policies. The considerable potential of the agro-forestry sector for synergies and contributing to increasing resilience of tropical farming systems is discussed in (Verchot et al. 2007) with examples from Africa.

'Climate Smart Agriculture' has emerged in recent years as an approach to integrate food security and climate challenges. The three pillars of CSA are to: (1) adapt and build resilience to climate change; (2) reduce GHG emissions, and; (3) sustainably increase agricultural productivity, ultimately delivering 'triple-wins' (Lipper et al. 2014c). While the concept is conceptually appealing, a range of criticisms, contradictions and challenges exist in using CSA as the route to resilience in global agriculture, notably around the political economy (Newell and Taylor 2017), the vagueness of the definition, and consequent assimilation by the mainstream agricultural sector, as well as issues around monitoring, reporting and evaluation (Arakelyan et al. 2017).

Land-based mitigation is facing important trade-offs with food production, biodiversity and local bio geophysical effects (Humpenöder et al. 2017; Krause et al. 2017; Robledo-Abad et al. 2017; Boysen et al. 2016, 2017a,b). Synergies between bio energy and food security could be achieved by investing in a combination of instruments including technology and innovations, infrastructure, pricing, flex crops, and improved communication and stakeholder engagement (Kline et al. 2017). Managing these trade-offs might also require demand side interventions including dietary change incentives (see 5.7.1).

Synergies and trade-offs also result from interaction between policies (policy interplay (Urwin and Jordan 2008)) at different levels of policy (vertical) and across different policies (horizontal) – see also section on policy coherence. If policy mixes are designed appropriately, acknowledging and incorporating trade-offs and synergies, they are more apt to deliver an outcome such as transitioning to sustainability (Howlett and Rayner 2013; Huttunen et al. 2014) (*medium evidence and medium agreement).* However, there is *medium evidence and medium agreement* that evaluating policies for coherence in responding to climate change and its impacts is not occurring, and policies are instead reviewed in a fragmented manner (Hurlbert and Gupta 2016).

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7.5.6.4. Water, food and aquatic ES

Trade-offs between some types of water use (eg irrigation for food security) and other ecosystem services are expected to intensify under climate change (Hanjra and Ejaz Qureshi 2010). There is an urgency to develop approaches to understand and communicate this to policy and decision makers (Zheng et al. 2016). Reducing water use in agriculture (Mekonnen and Hoekstra 2016) through policies on both supply and demand side such as shift to less-water intensive crops (Richter et al. 2017; Fishman et al. 2015), and shift in diets (Springmann et al. 2016) has potential to reduce trade-offs between food security and fresh-water aquatic ecosystem services (*medium evidence, high agreement*). There is strong evidence that improved efficiency in irrigation can actually increase overall water use in agriculture and therefore its contribution to improved flows in rivers is questionable (Ward and Pulido-Velazquez 2008).

There are now powerful new analytical approaches, high-resolution data and decision making tools that help to predict cumulative impacts of dams, assess trade-offs between engineering and environmental goals, and can help funders and decision makers compare alternative sites or designs for dam building as well as manage flows in regulated rivers based on experimental releases and adaptive learning. This could minimise ecological costs and maximise synergies with other development goals under climate change (Poff et al. 2003; Winemiller et al. 2016). Furthermore the adoption of metrics based on the emerging concept of Nature's Contributions to People (NCP) under the IPBES framework brings in non-economic instruments and values that in combination with conventional valuation of ecosystem services approaches could elicit greater support for non-consumptive water use of rivers for achieving SDG goals (De Groot et al. 2010; Pascual et al. 2017).

7.5.6.5. Considering Synergies and Tradeoffs to Avoid Maladaptation

Coherent policies that consider synergies and tradeoffs can also reduce the likelihood of maladaptation, which is the opposite of sustainable adaptation (Magnan et al. 2016). Sustainable adaptation is adaptation that "contributes to socially and environmentally sustainable development pathways including both social justice and environmental integrity" (Eriksen et al. 2011). In AR5 there was *medium evidence* and *high agreement* that maladaptation is 'a cause of increasing concern to adaptation planners, where intervention in one location or sector could increase the vulnerability of another location or sector, or increase the vulnerability of a group to future climate change' (Noble et al. 2014). AR5 recognised that maladaptation arises not only from inadvertent, badly planned adaptation actions, but also from deliberate decisions where wider considerations place greater emphasis on short-term outcomes ahead of longer-term threats, or that discount, or fail to consider, the full range of interactions arising from planned actions (Noble et al. 2014).

Some maladaptations are only beginning to be recognised as we become aware of unintended consequences of decisions. An example prevalent across many countries is irrigation as an adaptation to water scarcity. During a drought from 2007–2009 in California, farmers adapted by using more groundwater thereby depleting groundwater elevation by 15 metres. This volume of groundwater depletion is unsustainable environmentally and also emits GHG emissions during the pumping (Christian-Smith et al. 2015). Despite the three years of drought, the agricultural sector performed financially well, due to the groundwater use and crop insurance payments. Drought compensation programmes through crop insurance policies may reduce the incentive to shift to lower water-use crops, thereby perpetuating the maladaptive situation. Another example of maladaptation that may appear as adaptation to drought is pumping out groundwater and storing in surface farm ponds with consequences for water justice, inequity and sustainability (Kale 2017). These examples highlights both the potential for maladaptation from farmers' adaptation decisions as well as the unintended consequences of policy choices and illustrates the findings of Barnett and O'Neill (2010) that maladaptation can include high opportunity costs (including economic, environmental, and social); reduced incentives to adapt (adaptation measures that reduce incentives to adapt by not addressing underlying causes); and path dependency or trajectories that are difficult to change.

In practice, maladaptation is a specific instance of policy incoherence, and it may be useful to develop a framework in designing policy to avoid this type of trade-off. This would specify the type, aim and target audience of an adaptation action, decision, project, plan, or policy designed initially for adaptation, but actually at high risk of inducing adverse effects either on the system in which it was developed, or another connected system, or both. The assessment requires identifying system boundaries including temporal and geographical scales at which the outcome are assessed (Magnan 2014; Juhola et al. 2016). National level institutions that cover the spectrum of sectors affected, or enhanced collaboration between relevant institutions is expected to increase the effectiveness of policy instruments, as are joint programmes and funds (Morita and Matsumoto 2018).

As new knowledge about trade-offs and synergies amongst land-climate processes emerges regionally and globally, concerns over emerging risks and the need for planning policy responses grow. There is *medium evidence and medium agreement* that trade-offs currently do not figure into existing climate policies including NDCs and SDGs being vigorously pursued by some countries (Woolf et al. 2018). For instance, the biogeophysical co-benefits of reduced deforestation and re/afforestation measures (Chapter 6) are usually not accounted for in current climate policies or in the NDCs, but there is increasing scientific evidence to include them as part of the policy design (Findell et al. 2017; Hirsch et al. 2018; Bright et al. 2017).

Case Study: Green Energy: Biodiversity Conservation vs Global Environment Targets?

Green and renewable energy and transportation are emerging as an important part of climate change mitigation globally (*medium evidence, high agreement*) (McKinnon 2010; Zarfl et al. 2015; Creutzig et al. 2017). Evidence is however emerging across many biomes (from coastal to semi-arid and humid) how green energy may have significant trade-offs with biodiversity and ecosystem services thus demonstrating the need for closer environmental scrutiny and safeguards (Gibson et al. 2017)(Hernandez et al. 2015). In most cases, the accumulated impact of pressures from decades of land-use and habitat loss set the context within which the potential impacts of renewable energy generation need to be considered.

Small hydropower or SHPs were until recently considered as environmentally benign compared to large dams and are poorly understood, especially since the impacts of clusters of small dams are just becoming evident (Mantel et al. 2010; Fencl et al. 2015; Kibler and Tullos 2013). SHPs (<25/30 MW) and being labelled "green" are often exempt from environmental scrutiny (Abbasi and Abbasi 2011;

Pinho et al. 2007; Premalatha et al. 2014b; Era Consultancy 2006). Being promoted in mountainous global biodiversity hotspots, SHPs have changed the hydrology, water quality and ecology of head-water streams and neighbouring forests significantly. SHPs have created dewatered stretches of stream immediately downstream and introduced sub-daily to sub-weekly hydro-pulses that have transformed the natural dry-season flow regime. Hydrologic and ecological connectivity have been impacted, especially for endemic fish communities and fragmented forests in the Himalayas and Western Ghats biodiversity hotspots in India, and regions in China, and Central America (*medium evidence, medium agreement*) (Jumani et al. 2017, 2018; Chhatre and Lakhanpal 2018; Anderson et al. 2006; Grumbine and Pandit 2013). Some regions have opposed SHPs over concerns about impacts on local culture and livelihoods (Jumani et al. 2017, 2018; Chhatre and Lakhanpal 2018).

Large scale solar farms that involve large land resources are being installed at a rapid rate. In India, semi-arid and arid regions are targeted for wind and solar farms. India's renewable energy targets are often sited in semi-arid areas which includes the last remaining habitats of the highly endangered Great Indian Bustard (*Ardeotis nigriceps*). Installing solar and wind farms linked to lethal power transmission lines cause mortality of a species whose global population is now reduced to about 150 (Collar et al. 2015). The loss of habitat over the decades has been largely due to agricultural intensification driven by irrigation and bad management in designated reserves (Collar et al. 2015; Ledec, George C.; Rapp, Kennan W.; Aiello 2011) but intrusion of power lines in its last remaining refuges is a major worry for its future persistence (Government of India 2012). In many regions around the world, wind-turbines and solar farms pose a threat to many other species especially predatory birds and insectivorous bats (*medium evidence, medium agreement*) (Thaker, M, Zambre, A. Bhosale 2018) and disrupt habitat connectivity (Northrup and Wittemyer 2013).

Additionally, conversion of rivers into waterways has been touted as a fuel-efficient (low carbon emitting) and environment-friendly alternative to surface land transport (IWAI 2016; Dharmadhikary, S., and Sandbhor 2017). India's National Waterways (funded partly by a USD 375 million loan from the World Bank) seeks to cut transportation time and costs and reduce carbon emissions from road transport (Admin 2017). However given the low water levels in India's rivers in the dry-season (due to upstream demands and abstraction) the programme relies on large scale dredging to maintain deep channels. Evidence from elsewhere suggests that dredging could severely impact the water quality, human health and habitat of fish species (Junior et al. 2012; Martins et al. 2012), disrupt artisanal fisheries and potentially cause severe threat to the endangered Ganges River Dolphin (*Platanista gangetica*), India's National Aquatic Animal (Kelkar 2016). The most severe impact of dredging and vessel traffic on this unique species is the disruption through under-water noise of the acoustic signals that the endangered and naturally blind animal relies on for navigation, foraging and communication (*low evidence, medium agreement)* (Dey Mayukh 2018). Off-shore renewable energy projects in coastal zones have been known to have similar impacts on marine fauna (Gill 2005).

Policy response to mitigate and reduce the negative impacts of small dams include changes in SHP operations and policies to enable the conservation of river fish diversity. These include mandatory environmental impact assessments, conserving remaining undammed headwater streams in regulated basins, maintaining adequate environmental flows, and implementing other adaptation measures based on experiments with active management of fish communities in impacted zones (Jumani et al. 2018). Location of large solar farms needs to be carefully scrutinised (Sindhu et al. 2017). For mitigating negative impacts of power lines associated with solar and wind-farms in bustard habitat, suggested measures include diversion structures to prevent collision, underground cables and avoidance in core wildlife habitat as well as incentives for maintaining low intensity rain-fed agriculture and pasture around existing reserves, and curtailing harmful infrastructure in priority areas (Collar et al. 2015). Mitigation for minimising the ecological impact of Inland Waterways on biodiversity and fisheries is more complicated but may involve improved boat technology to reduce under-water noise, maintaining ecological flows and thus reduced dredging, and avoidance in key habitats (Dey Mayukh 2018).

The management of ecological trade-offs of green energy and green infrastructure and transportation projects may be crucial for long-term sustainability and acceptance of emerging low-carbon economies.

7.6. Governance: Governing the land-climate interface

Building on the definition of governance in section 7.1.2, governance situates decision making and selection or calibration of policy instruments within the reality of the multitude of actors operating in respect of land and climate interactions. Governance includes all of the processes, structures, rules and traditions that govern; governance processes may be undertaken by actors including a government, market, organisation, or family (Bevir 2011). Governance processes determine how people in societies make decisions (Patterson et al. 2017) and involve the interactions among formal and informal institutions (see 7.4.1) through which people articulate their interests, exercise their legal rights, meet their legal obligations, and mediate their differences (Plummer and Baird 2013).

The act of governance "is a social function centred on steering collective behaviour toward desired outcomes and away from undesirable outcomes" (Young 2017a), here sustainable climate resilient development. This definition of governance allows for it to be decoupled from the more familiar concept of government and studied in the context of complex human-environment relations and environmental and resource regimes (Young 2017a) and used to address the interconnected challenges facing food and agriculture (FAO 2017b). These challenges include assessing, combining, and implementing policy instruments at different governance levels in a mutually reinforcing way, managing trade-offs while capitalising on synergies (see 7.5.6), and employing experimentalist approaches for improved and effective governance (FAO 2017b), here adaptive climate governance (7.6.3). Emphasising governance also represents a shift of traditional resource management (focused on hierarchical state control) towards recognition that political and decision making authority can be exercised through interlinked groups of diverse actors (Kuzdas et al. 2015).

This section will start with describing institutions and institutional arrangements (the core of a governance system (Young 2017)) that build adaptive and mitigative capacity, outlining modes, levels and scales of governance for sustainable climate resilient development, describing adaptive climate governance that responds to uncertainty, exploring institutional dimensions of adaptive governance that create an enabling environment for strong institutional capital, discussing land tenure (an important institutional context for effective and appropriate selection of policy instruments), and end with the participation of people in decision making through inclusive governance.

7.6.1. Institutions Building Adaptive and Mitigative Capacity

Institutions are rules and norms held in common by social actors that guide, constrain, and shape human interaction. Institutions can be formal, such as laws, policies, and structured decision making processes (see 7.5.1.1) or informal, such as norms, conventions, and decision making following customary norms and habits (see 7.5.1.2). Organisations – such as parliaments, regulatory agencies, private firms, and community bodies – as well as people, develop and act in response to institutional frameworks and the incentives they frame. "Institutions can guide, constrain, and shape human interaction through direct control, through incentives, and through processes of socialization" (AR5, 2014 at p. 1768). Nations with "well developed institutional systems are considered to have greater adaptive capacity," and better institutional capacity to help deal with risks associated with future climate change (IPCC, 2001 at p. 896). Institutions may also prevent the development of adaptive capacity when they are 'sticky' or characterised by strong path dependence (Mahoney 2000) (North 1991) and prevent changes that are important to address climate change (see 7.4.9).

Formal and informal governance structures are composed of these institutionalised rule systems that determine vulnerability as they influence power relations, risk perceptions and establish the context wherein risk reduction, adaptation and vulnerability are managed (Cardona 2012). Governance institutions determine the management of a community's assets, the community members' interrelationship, and their relationships with natural resources (Hurlbert and Diaz 2013). Traditional or locally-evolved institutions, backed by cultural norms, can contribute to resilience and adaptive capacity. Anderson et al. suggest these are particularly a feature of dry land societies that are highly prone to environmental risk and uncertainty (Anderson et al. 2010). Concepts of resilience, and specifically the resilience of socio-ecological systems have advanced analysis of adaptive institutions and adaptive governance in relation to climate change and land (Boyd and Folke 2011a). In their characterisation, "resilience is the ability to reorganise following crisis, continuing to learn, evolving with the same identity and function, and also innovating and sowing the seeds for transformation. It is a central concept of adaptive governance" (Boyd and Folke 2012). In the context of complex and multi-scale socio-ecological systems, important features of adaptive institutions that contribute to resilience include the chartacteristics of an adaptive governance system (see 7.6.6).

There is *high confidence* that adaptive institutions include a strong learning dimension and include:

(1) Institutions advancing the capacity to learn through availability, access to, accumulation of, and interpretation of information (such as drought projections, costing of alternatives land, food, and water strategies). Government supported networks, learning platforms, and facilitated interchange between actors with boundary and bridging organisations, creates the necessary self-organisation to prepare for the unknown. Through transparent, flexible networks,whole sets of complex problems of land, food, and climate can be tackled to develop shared visions and critique land and food management systems assessing gaps and generating solutions;
(2) Institutions advancing learning by experimentation (in interpretation of information, new ways of governing, and treating policy as an ongoing experiment) through many interrelated decisions, but especially those that connect the social to the ecological and entail anticipatory planning by considering a longer term time frame. Mechanisms to do so include ecological stewardship and rituals and beliefs of indigenous societies that sustain ecosystem services;
(3) Institutions that decide on pathways to realise system change through cultural, inter and intra organisational collaboration, with a flexible regulatory framework allowing for new cognitive frames of 'sustainble' land management and 'safe' water supply that open alternative pathways (Karpouzoglou et al. 2016; Bettini et al. 2015; Boyd et al. 2015; Boyd and Folke 2011b)) (Boyd and Folke 2012).

Shortcomings of resilience theory include limits in relation to its conceptualisation of social change (Cote and Nightingale 2012), its potential to be used as a normative concept implying politically prescriptive policy solutions (Thorén and Olsson 2017; Weichselgartner and Kelman 2015; Milkoreit et al. 2015), its applicability to local needs and experiences (Forsyth 2018), and its potential to hinder evaluation of policy effectiveness (Newton 2016; Olsson et al. 2015b). Regardless, concepts of adaptive institutions building adaptive capacity in complex socio-ecological systems governance have progressed (Karpouzoglou et al. 2016; Dwyer and Hodge 2016) in relation to adaptive governance (Koontz et al. 2015).

The study of institutions of governance, levels, modes, and scale of governance, in a multi-level and polycentric fashion is important because of the multi-scale nature of the challenges to resilience, dissemination of ideas, networking and learning.

7.6.2. Integration - Levels, Modes, and Scale of Governance for Sustainable Development

Different types of governance can be distinguished according to intended levels (e.g., local, regional, global), domains (national, international, transnational), modes (market, network, hierarchy), and scales (global regimes to local community groups) (Jordan et al. 2015b). Implementation of climate change adaptation and mitigation has been impeded by institutional barriers including multi-level governance and policy integration issues (Biesbroek et al. 2010). To overcome these barriers, climate governance has evolved significantly beyond the national and multilateral domains that tended to dominate climate efforts and initiatives during the early years of the UNFCCC. The climate challenge has been placed in an "earth system" context, showing the existence of complex interactions and governance requirements across different levels and calling for a radical transformation in governance, rather than minor adjustments (Biermann et al. 2012). Climate governance literature has expanded since AR5 in relation to the sub-national and transnational levels, but all levels and their interconnection is important. Expert thinking has evolved from implementing good governance at high levels of governance (with governments) to a decentred problem solving approach consistent with adaptive governance. This approach involves iterative bottom up and experimental mechanisms that might entail addressing tenure of land or forest management through a territorial approach to development, thereby supporting multi-sectoral governance in local, municipal, and regional contexts (FAO 2017b).

Local action in relation to mitigation and adaptation continues to be important by complementing and advancing global climate policy (Ostrom 2012). Sub-national governance efforts for climate policy, especially at the level of cities and communities, have become significant during the past decades (*medium evidence, medium agreement*) (Castán Broto 2017; Floater et al. 2014; Albers et al. 2015; Archer et al. 2014). A transformation of sorts has been underway through deepening engagement from the private sector and NGOs as well as Government involvement at multiple levels. It is now recognised that business organisations, civil society groups, citizens, and formal governance all have important roles in governance for sustainable development (Kemp et al. 2005).

Transnational governance efforts have increased in number, with application across different economic sectors, geographical regions, civil society groups and non-governmental organisations. When it comes to climate mitigation, transnational mechanisms generally focus on networking and may not necessarily be effective in terms of promoting real emissions reductions (Michaelowa and Michaelowa 2017). However, acceleration in national mitigation measures has been determined to coincide with landmark international events such as the build up to the Copenhagen Climate Conference (Iacobuta et al. 2018). There is a tendency for transnational governance mechanisms to lack monitoring and evaluation procedures (Jordan et al. 2015a).

To address shortcomings of transnational governance, polycentric governance considers the interaction between actors at different levels of governance (local, regional, national, and global) for a more nuanced understanding of the variation in diverse governance outcomes in the management of common-pool resources (such as forests) based on the needs and interests of citizens (Nagendra and Ostrom 2012). A more "polycentric climate governance" system has emerged that incorporates bottom-up initiatives that can support and synergise with national efforts and international regimes (Ostrom 2010). Although it is clear that many more actors and networks are involved, the effectiveness of a more polycentric system remains unclear (Jordan et al. 2015a).

There is *high confidence* that a hybrid form of governance combining the advantages of centralised governance (with coordination, stability, compliance) with those of more horizontal structures (that allow flexibility, autonomy for local decision making, multi-stakeholder engagement, co-management) is required for effective mainstreaming of mitigation and adaptation in sustainable land

and forest management (Keenan 2015; Gupta 2014; Williamson and Nelson 2017; Liniger et al. 2019). Polycentric institutions self-organise developing collective solutions to local problems as they arise (Koontz et al. 2015). The public sector (governments and administrative systems) are still important in climate change initiatives as these actors retain the political will to implement and make intiatives work (Biesbroek et al. 2018).

Sustainable development hinges on the holistic integration of interconnected land and climate issues, sectors, levels of government, and policy instruments (see Policy Coherence 7.4.8), that address the increasing volatility in oscillating systems and weather patterns (Young 2017b; Kemp et al. 2005). Climate adaptation and mitigation goals must be integrated or mainstreamed into existing governance mechanisms around key land use sectors such as forestry and agriculture. In the EU, mitigation has generally been well-mainstreamed in regional policies but not adaptation (Hanger et al. 2015). Climate change adaptation has been impeded by institutional barriers including the inherent challenges of multi-level governance and policy integration (Biesbroek et al. 2010).

Integrative polycentric approaches to land use and climate interactions take different forms and operate with different institutions and governance mechanisms. Integrative approaches can provide coordination and linkages to improve effectiveness and efficiency and minimise conflicts (*high confidence*). Different types of integration with special relevance for the land-climate interface can be characterised as follows:

1. Cross-level integration: local and national level efforts must be coordinated with national and regional policies and also be capable of drawing direction and financing from global regimes, thus requiring multi-level governance. Integration of sustainable land management to prevent, reduce, and restore degraded land is advanced with national and subnational policy includes passing the necessary laws establishing frameworks and providing financial incentives. Examples include: integrated territorial planning addressing specific land use decisions; local landscape participatory planning with farmer associations, microenterprises, and local institutions identifying hot spot areas, identifying land use pressures and scaling out sustainable land management response options (Liniger et al. 2019).
2. Cross-sectoral integration: rather than approach each application or sector (e.g., energy, agriculture, forestry) separately, there is a conscious effort at co-management and coordination in policies and institutions, such as with the energy-water-food nexus (Biggs et al. 2015).
3. End-use/market integration: often involves exploiting economies of scope across products, supply chains, and infrastructure (Nuhoff-Isakhanyan et al. 2016; Ashkenazy et al. 2017). For instance land-use transport models consider land use, transportation, city planning , and climate mitigation (Ford et al. 2018).
4. Landscape integration: rather than physical separation of activities (e.g., agriculture, forestry, grazing), uses are spatially integrated by exploiting natural variations while incorporating local and regional economies (Harvey et al. 2014a). In an assessment of 166 initiatives in 16 countries, integrated landscape initiatives were found to address the drivers of agriculture, ecosystem conservation, livelihood preservation and institutional coordination. However, such initiatives struggled to move from planning to implementation due to lack of government and financial support and powerful stakeholders sidelining the agenda (Zanzanaini et al. 2017). Special care helps ensure initiatives don't exacerbate socio-spatial inequalities across diverse developmental and environmental conditions (Anguelovski et al. 2016b). Integrated land use planning coordinated through multiple government levels balances property rights, wildlife and forest conservation, encroachment of settlements and agricultural areas and can reduce conflict (*high confidence*) (Metternicht 2018). Land use planning can also enhance management of areas prone to natural disasters such as floods and resolve issues of competing land uses and land tenure conflicts (Metternicht 2018).

Another way to analyse or characterise governance approaches or mechanisms might be according to a temporal scale with respect to relevant events, for example those that may occur gradually vs. abruptly (Cash et al. 2006). Desertification and land degradation are drawn-out processes that occur over many years, whereas extreme events are abrupt and require immediate attention. Similarly, the frequency of events might be of special interest, for example events that occur periodically vs. those that occur infrequently and/or irregularly. In the case of food security abrupt and protracted events of food insecurity might occur. There is a distinction between "hunger months" and longer-term food insecurity. Some indigenous practices already incorporate hunger months whereas structural food deficits have to be addressed differently (Bacon et al. 2014). Governance mechanisms that facilitate rapid response to crises are quite different from those aimed at monitoring slower changes and responding with longer-term measures.

Governance Case Study: Biofuels and bioenergy

New policies and initiatives during the past decade or so have increased support for bioenergy as a non-intermittent (stored) renewable with wide geographic availability that is cost-effective in a range of applications. Significant upscaling of bioenergy requires dedicated (normally land-based) sources in addition to use of wastes and residues. As a result a disadvantage is high land use intensity compared to other renewables (Fritsche et al. 2017b) that in turn place greater demands on governance. Bioenergy, especially traditional fuels currently provides the largest share of renewable energy globallyand has a significant role in nearly all climate stabilisation scenarios, although estimates of its potential vary widely (see Cross-Chapter Box 7 on Bioenergy and BECCS in Chapter 6). Policies and governance for bioenergy systems and markets must address diverse applications and sectors across levels from local to global; here we briefly review the literature in relation to governance for **modern** bioenergy and biofuels with respect to land and climate impacts whereas **traditional biomass** use (see Glossary) (> 50% of energy used today with greater land use and GHG emissions impacts in low and medium-income countries (Bailis et al. 2015; Masera et al. 2015; Bailis et al. 2017a; Kiruki et al. 2017b)) is addressed elsewhere (see sections 4.5.4 and 7.4.6.4 and Cross-Chapter Box 12 on Traditional Biomass in this chapter). The bioenergy cycle is relevant in accounting for—and attributing—land impacts and GHG emissions (see section 2.5.1.5). Integrated responses across different sectors can help to reduce negative impacts and promote sustainable development opportunities (Table 6.9, Table 6.58). It is *very likely* that bioenergy expansion at a scale that contributes significantly to global climate mitigation efforts (see Cross-Chapter Box 7 on Bioenergy and BECCS in Chapter 6) will result in substantial land use change (Berndes et al. 2015; Popp et al. 2014a; Wilson et al. 2014; Behrman et al. 2015; Richards et al. 2017; Harris et al. 2015; Chen et al. 2017a). There is *medium evidence and high agreement* that land use change at such scale presents a variety of positive and negative socio-economic and environmental impacts that lead to risks and trade-offs that must be managed or governed across different levels (Pahl-Wostl et al. 2018a; Kurian 2017; Franz et al. 2017; Chang et al. 2016; Larcom and van Gevelt 2017; Lubis et al. 2018; Alexander et al. 2015b; Rasul 2014; Bonsch et al. 2016; Karabulut et al. 2018; Mayor et al. 2015). There is *medium evidence and high agreement* that impacts vary considerably with factors such as initial land use type, choice of crops, initial carbon stocks, climatic region, soil types and the management regime and technologies adopted (Qin et al. 2016; Del Grosso et al. 2014; Popp et al. 2017; Davis et al. 2013; Mello et al. 2014; Hudiburg et al. 2015; Carvalho et al. 2016; Silva-Olaya et al. 2017; Whitaker et al. 2018; Alexander et al. 2015b);

There is *medium evidence and high agreement* that significant socio-economic impacts requiring additional policy responses can occur when agricultural lands and/or food crops are used for bioenergy due to competition between food and fuel (Harvey and Pilgrim 2011; Rosillo Callé and Johnson 2010b), including impacts on food prices (Martin Persson 2015; Roberts and Schlenker 2013;

Borychowski and Czyżewski 2015; Koizumi 2014; Muratori et al. 2016; Popp et al. 2014b; Araujo Enciso et al. 2016) and impacts on food security (Popp et al. 2014b; Bailey 2013; Pahl-Wostl et al. 2018b; Rulli et al. 2016; Yamagata et al. 2018; Kline et al. 2017; Schröder et al. 2018; Franz et al. 2017; Mohr et al. 2016). Additionally crops such as sugar-cane which are water-intensive when used for ethanol production have a trade-off with water and downstream ecosystem services and other crops more important for food security (Rulli et al. 2016; Gheewala et al. 2011). Alongside negative impacts that might fall on urban consumers (who purchase both food and energy), there is *medium evidence and medium agreement* that rural producers or farmers can increase income or strengthen livelihoods by diversifying into biofuel crops that have an established market (Maltsoglou et al. 2014; Mudombi et al. 2018a; Gasparatos et al. 2018a,b; von Maltitz et al. 2018; Gasparatos et al. 2018c; Kline et al. 2017; Rodríguez Morales and Rodríguez López 2017; Dale et al. 2015; Lee and Lazarus 2013; Rodríguez-Morales 2018). A key governance mechanism that has emerged in response to such concerns, especially during the past decade are standards and certification systems that include food security and land rights in addition to general criteria or indicators related to sustainable use of land and biomass (see section 7.4.6.3 on Standards and Certification). There is *medium evidence and medium agreement* that policies promoting use of wastes and residues, the use of non-edible crops and/or reliance on degraded and marginal lands for bioenergy could reduce land competition and associated risk for food security (Manning et al. 2015; Maltsoglou et al. 2014; Zhang et al. 2018a; Gu and Wylie 2017; Kline et al. 2017; Schröder et al. 2018; Suckall et al. 2015; Popp et al. 2014a; Lal 2013).

There is *medium evidence and high agreement* that good governance, including policy coherence and coordination across the different sectors involved (agriculture, forestry, livestock, energy, transport) (see 7.6.2) can help to reduce the risks and increase the co-benefits of bioenergy expansion (Makkonen et al. 2015; Di Gregorio et al. 2017; Schut et al. 2013; Mukhtarov et al.; Torvanger 2019a; Müller et al. 2015; Nkonya et al. 2015; Johnson and Silveira 2014a; Lundmark et al. 2014; Schultz et al. 2015; Silveira and Johnson 2016; Giessen et al. 2016b; Stattman et al. 2018b; Bennich et al. 2017b). There is *medium evidence and high agreement* that the nexus approach can help to address interconnected biomass resource management challenges and entrenched economic interests, as well to leverage synergies in the systemic governance of risk. (Bizikova et al. 2013; Rouillard et al. 2017; Pahl-Wostl 2017a; Lele et al. 2013; Rodríguez Morales and Rodríguez López 2017; Larcom and van Gevelt 2017; Pahl-Wostl et al. 2018a; Rulli et al. 2016; Rasul and Sharma 2016; Weitz et al. 2017b; Karlberg et al. 2015).

A key issue for governance of biofuels and bioenergy, as well as land use governance more generally, during the past decade is the need for new governance mechanisms across different levels as land use policies and bioenergy investments are scaled up and result in wider impacts (see section 7.6). There is *low evidence and medium agreement* that hybrid governance mechanisms can promote sustainable bioenergy investments and land use pathways. This hybrid governance can include multi-level, transnational governance, and private-led or partnership-style (polycentric) governance complementing national-level, strong public coordination (government and public administration){7.6.2} (Pahl-Wostl 2017a; Pacheco et al. 2016; Winickoff and Mondou 2017; Nagendra and Ostrom 2012; Jordan et al. 2015a; Djalante et al. 2013; Purkus, Alexandra; Gawel, Erik; Thrän 2012; Purkus et al. 2018; Stattman et al.; Rietig 2018; Cavicchi et al. 2017; Stupak et al. 2016; Stupak and Raulund-Rasmussen 2016; Westberg and Johnson 2013; Giessen et al. 2016b; Johnson and Silveira 2014b; Stattman et al. 2018b; Mukhtarov et al.; Torvanger 2019b).

Cross-Chapter Box 12: Traditional biomass use: land, climate and development implications

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Introduction and significance

Most biomass used for energy today is in traditional forms (fuelwood, charcoal, agricultural residues) for cooking and heating by some 3 billion persons worldwide (IEA 2017). Traditional biomass has high land and climate impacts, with significant harvesting losses, GHG emissions, soil impacts and high conversion losses (Cutz et al. 2017b; Masera et al. 2015; Ghilardi et al. 2016a; Bailis et al. 2015; Fritsche et al. 2017b; Mudombi et al. 2018b). In addition to these impacts, indoor air pollution from household cooking is a leading cause of mortality in low and medium-income countries and affects especially women and children (Smith et al. 2014a; HEI/IHME 2018; Goldemberg et al. 2018b). In rural areas, the significant time needed for gathering fuelwood imposes further costs on women and children (Njenga and Mendum 2018; Gurung and Oh 2013a; Behera et al. 2015a).

Both agricultural and woody biomass can be upgraded and used sustainably through improved resource management and modern conversion technologies, providing much greater energy output per unit of biomass (Cutz et al. 2017b; Hoffmann et al. 2015a; Gurung and Oh 2013b). More relevant than technical efficiency is the improved quality of energy services: with increasing income levels and/or access to technologies, households transition over time from agricultural residues and fuelwood to charcoal and then to gaseous or liquid fuels and electricity (Leach 1992; Pachauri and Jiang 2008; Goldemberg and Teixeira Coelho 2004; Smeets et al. 2012a). However, most households use multiple stoves and/or fuels at the same time, known as "fuel stacking" for economic flexibility and also for sociocultural reasons (Ruiz-Mercado and Masera 2015a; Cheng and Urpelainen 2014; Takama et al. 2012).

Urban and rural use of traditional biomass

In rural areas, fuelwood is often gathered at no cost to the user and burned directly whereas in urban areas, traditional biomass use may often involve semi-processed fuels, particularly in sub-Saharan Africa where charcoal is the primary urban cooking fuel. Rapid urbanisation and/or commercialisation drives a shift from fuelwood to charcoal, which results in significantly higher wood use *(very high confidence)* due to losses in charcoal supply chains and the tendency to use whole trees for charcoal production (Santos et al. 2017; World Bank. 2009a; Hojas-Gascon et al. 2016a; Smeets et al. 2012b). One study in Myanmar found that charcoal required 23 times the land area of fuelwood (Win et al. 2018). In areas of woody biomass scarcity, animal dung and agricultural residues as well as lower quality wood are often used (Kumar Nath et al. 2013a; Go et al. 2019a; Jagger and Kittner 2017; Behera et al. 2015b). The fraction of woody biomass harvested that is not "demonstrably renewable" is the fraction of non-renewable biomass (fNRB) under UNFCCC accounting; default values for fNRB for least developed countries and small island developing states ranged from 40% to 100% (CDM Executive Board 2012). Uncertainties in woodfuel data, complexities in spatiotemporal woodfuel modelling and rapid forest regrowth in some tropical regions present sources of variation in such estimates, and some fNRB values are *likely* to have been over-estimated (McNicol et al. 2018a; Ghilardi et al. 2016b; Bailis et al. 2017b).

GHG emissions and traditional biomass

Due to overharvesting, incomplete combustion and the effects of short-lived climate pollutants, traditional woodfuels (fuelwood and charcoal) contribute 1.9-2.3% of global GHG emissions; non-renewable biomass is concentrated especially in "hotspot" regions of East Africa and South Asia

(Bailis et al. 2015). The estimate only includes woody biomass and does not account for possible losses in soil carbon or the effects of nutrient losses from use of animal dung, which can be significant in some cases (Duguma et al. 2014a; Achat et al. 2015a; Sánchez et al. 2016). Reducing emissions of black carbon alongside GHG reductions offers immediate health co-benefits (Shindell et al. 2012; Pandey et al. 2017; Weyant et al. 2019a; Sparrevik et al. 2015). Significant GHG emissions reductions, depending on baseline or reference use, can be obtained through fuel-switching to gaseous and liquid fuels, sustainable harvesting of woodfuels, upgrading to efficient stoves, and adopting high-quality processed fuels such as wood pellets (*medium evidence, high agreement*) (Wathore et al. 2017; Jagger and Das 2018; Quinn et al. 2018a; Cutz et al. 2017b; Carter et al. 2018; Bailis et al. 2015; Ghilardi et al. 2018; Weyant et al. 2019b; Hoffmann et al. 2015b).

Land and forest degradation

Land degradation is itself a significant source of GHG emissions and biodiversity loss, with overharvesting of woodfuel as a major cause in some regions and especially in sub-Saharan Africa (Pearson et al. 2017; Joana Specht et al. 2015a; Kiruki et al. 2017b; Ndegwa et al. 2016; McNicol et al. 2018b). Reliance on traditional biomass is quite land-intensive: supplying one household sustainably for a year can require more than half a hectare of land, which, in dryland countries such as Kenya, can result in substantial percentage of total tree cover (Fuso Nerini et al. 2017). In sub-Saharan Africa and in some other regions, land degradation is widely associated with charcoal production (*high confidence*), often in combination with timber harvesting or clearing land for agriculture (Kiruki et al. 2017a; Ndegwa et al. 2016; Hojas-Gascon et al. 2016b). Yet charcoal makes a significant contribution to livelihoods in many areas and thus in spite of the ecological damage, halting charcoal production is difficult due to the lack of alternative livelihoods and/or the affordability of other fuels (Smith et al. 2015; Zulu and Richardson 2013a; Jones et al. 2016a; World Bank. 2009b).

Use of agricultural residues and animal dung for bioenergy

Although agricultural wastes and residues from almost any crop can be used in many cases for bioenergy, excessive removal or reduction of forest (or agricultural) biomass can contribute to a loss of soil carbon, which can also in turn contribute to land degradation (James et al. 2016; Blanco-Canqui and Lal 2009a; Carvalho et al. 2016; Achat et al. 2015b; Stavi and Lal 2015). Removals are be limited to levels at which problems of soil erosion, depletion of soil organic matter, soil nutrient depletion and decline in crop yield are effectively mitigated (Ayamga et al. 2015a; Baudron et al. 2014; Blanco-Canqui and Lal 2009b). Application or recycling of residues may in some cases be more valuable for soil improvement (*medium confidence*). Tao et al (2017) used leftover oil palm fruit bunches and demonstrated that application of 30 to 90 t ha^{-1} empty fruit bunches maintains high palm oil yield with low temporal variability. A wide variety of wastes from palm oil harvesting can be used for bioenergy, including annual crop residues (Go et al. 2019b; Ayamga et al. 2015b; Gardner et al. 2018b).

Animal dung is a low-quality fuel used where woody biomass is scarce, such as in South Asia and some areas of eastern Africa (Duguma et al. 2014b; Behera et al. 2015b; Kumar Nath et al. 2013b). Carbon and nutrient losses can be significant when animal dung is dried and burned as cake, whereas using dung in a biodigester provides high-quality fuel and preserves nutrients in the by-product slurry (Clemens et al. 2018; Gurung and Oh 2013b; Quinn et al. 2018b).

Production and use of biochar

Converting agricultural residues into biochar can also help to reverse trends of soil degradation (see section 4.10.7). The positive effects of using biochar have been demonstrated in terms of soil aggregate improvement, increase of exchangeable cations, cation exchange capacity, available P, soil pH and carbon sequestration as well as increased crop yields (Huang et al. 2018; El-Naggar et al.

2018; Wang et al. 2018; Oladele et al. 2019; Blanco-Canqui and Lal 2009b). The level of biochar effectiveness varies depending on the kind of feedstock, soil properties and rate of application (Shaaban et al. 2018; Pokharel and Chang 2019). In addition to adding value to an energy product, the use of biochar offers a climate-smart approach to address agricultural productivity (Solomon and Lehmann 2017).

Relationship to food security and other SDGs

The population that is food insecure also intersects significantly with those relying heavily on traditional biomass such that poor and vulnerable populations often expend considerable time (gathering fuel) or use a significant share of household income for low quality energy services (Fuso Nerini et al. 2017; McCollum et al. 2018; Rao and Pachauri 2017; Pachauri et al. 2018; Muller and Yan 2018; Takama et al. 2012). Improvements in energy access and reduction or elimination of traditional biomass use thus have benefits across multiple SDGs *(medium evidence, high agreement)* (Masera et al. 2015; Rao and Pachauri 2017; Pachauri et al. 2018; Hoffmann et al. 2017; Jeuland et al. 2015; Takama et al. 2012; Gitau et al. 2019; Quinn et al. 2018b; Ruiz-Mercado and Masera 2015b; Duguma et al. 2014b; Sola et al. 2016b). Improved energy access contributes to adaptive capacity although charcoal production itself can also serve as a diversification or adaptation strategy (Perera et al. 2015; Ochieng et al. 2014; Sumiya 2016; Suckall et al. 2015; Jones et al. 2016b).

Socio-economic choices and shifts

When confronted with the limitations of higher-priced household energy alternatives, climate mitigation policies can result in trade-offs with health, energy access and other SDGs (Cameron et al. 2016; Fuso Nerini et al. 2018). The poorest households have no margin to pay for higher-cost efficient stoves; a focus on product-specific characteristics, user needs and/or making clean options more available would improve the market take-up *(medium confidence)* (Takama et al. 2012; Mudombi et al. 2018c; Khandelwal et al. 2017; Rosenthal et al. 2017; Cundale et al. 2017; Jürisoo et al. 2018). Subsidies for more efficient end-use technologies in combination with promotion of sustainable harvesting techniques would provide the highest emissions reductions while at the same time improving energy services (Cutz et al. 2017a).

Knowledge Gaps

Unlike analyses on modern energy sources, scientific assessments on traditional biomass use are complicated by its informal nature and the difficulty of tracing data and impacts; more systematic analytical efforts are needed to address this research gap (Cerutti et al. 2015). In general, traditional biomass use is associated with poverty. Therefore, efforts to reduce the dependence on fuelwood use are to be conducted in coherence with poverty alleviation (McCollum et al. 2018; Joana Specht et al. 2015b; Zulu and Richardson 2013b). The substantial potential co-benefits suggest that the traditional biomass sector remains under-researched and under-exploited in terms of cost-effective emissions reductions as well as for synergies between climate stabilisation goals and other SDGs.

7.6.3. Adaptive Climate Governance Responding to Uncertainty

In the 1990s, adaptive governance emerged from adaptive management (Holling 1978, 1986), combining resilience and complexity theory, and reflecting the trend of moving from government to governance (Hurlbert 2018b). Adaptive governance builds on multi-level and polycentric governance. Adaptive governance is “a process of resolving trade-offs and charting a course for sustainability” (Boyle, Michelle; Kay, James J.; Pond, 2001 at p. 28) through a range of “political, social, economic and administrative systems that develop, manage and distribute a resource in a manner promoting resilience through collaborative, flexible and learning based issue management across different scales” (Margot A. Hurlbert, 2018 at p. 25). There is *medium evidence and medium agreement* that few alternative governance theories handle processes of change characterised by nonlinear dynamics, threshold effects, cascades and limited predictability; however, the majority of literature relates to the United States or Canada (Karpouzoglou et al. 2016). Combining adaptive governance with other theories has allowed good evaluation of important governance features such as power and politics, inclusion and equity, short term and long term change, and the relationship between public policy and adaptive governance (Karpouzoglou et al. 2016).

There is *robust evidence and high agreement* that resource and disaster crises are crises of governance (Pahl-Wostl 2017b; Villagra and Quintana 2017; Gupta et al. 2013b). Adaptive governance of risk has emerged in response to these crises and involves four critical pillars including 1) sustainability as a response to environmental degradation, resource depletion and ecosystem service deterioration; 2) recognition that governance is required as government is unable to resolve key societal and environmental problems including climate change and complex problems; 3) mitigation is a means to reduce vulnerability and avoid exposure; and 4) adaptation responds to changes in environmental conditions (Fra.Paleo 2015).

Closely related to (and arguably components of) adaptive governance are adaptive management (see 7.5.4) (a regulatory environment that manages ecological system boundaries through hypothesis testing, monitoring, and re-evaluation (Mostert et al. 2007)), adaptive co-management (flexible community based resource management (Plummer and Baird 2013), and anticipatory governance (flexible decision making through the use of scenario planning and reiterative policy review (Boyd et al. 2015). Adaptive governance can be conceptualised as including multilevel governance with a balance between top-down and bottom-up decision making that is performed by many actors (including citizens) in both formal and informal networks, allowing policy measures and governance arrangements to be tailored to local context and matched at the appropriate scale of the problem, allowing for opportunities for experimentation and learning by individuals and social groups (Rouillard et al. 2013; Hurlbert 2018b).

There is *high confidence* that anticipation is a key component of adaptive climate governance wherein steering mechanisms in the present are developed to adapt to and/or shape uncertain futures (Vervoort and Gupta 2018; Wiebe et al. 2018; Fuerth 2009). Effecting this anticipatory governance involves silmultaneously making short term decisions in the context of longer term policy visioning, anticipating future climate change models and scenarios in order to realise a more sustainable future (Bates and Saint-Pierre 2018; Serrao-Neumann et al. 2013; Boyd et al. 2015). Utilising the decision making tools and practices in 7.5, policy makers operationalise anticipatory governance through a foresight system considering future scenarios and models, a networked system for integrating this knowledge into the policy process, a feedback system using indicators (see 7.5.5)to guage performance, an open-minded institutional culture allowing for hybrid and polycentric governance (Fuerth and Faber 2013; Fuerth 2009).

There is *high confidence* that in order to manage uncertainty, natural resource governance systems need to allow agencies and stakeholders to learn and change over time responding to ecosystem changes and new information with different management strategies and practices that involve experimentation (Camacho 2009; Young 2017b). There is an emerging literature on experimentation in governance surrounding climate change and land use (Kivimaa et al. 2017a) including policies such as REDD+ (Kaisa et al. 2017). Governance experiment literature could be in relation to scaling up policies from the local level for greater application, or downscaling policies addressing broad complex issues such as climate change, or addressing necessary change in social processes across sectors (such as water energy and food) (Laakso et al. 2017). Successful development of new policy instruments occurred in a governance experiment relating to coastal policy adapting to rising sea levels and extreme weather events through planned retreat (Rocle and Salles 2018). Experiments in emission trading between 1968 and 2000 in the United States of America helped to realise specific models of governance and material practices through mutually supportive lab experiments and field application that advanced collective knowledge (Voß and Simons 2018).

There is *high confidence* that a sustainable land management plan is dynamic and adaptive over time to (unforeseen) future conditions by monitoring indicators as early warnings or signals of tipping points initiating a process of change in policy pathway before a harmful threshold is reached (Stephens et al. 2018, 2017; Haasnoot et al. 2013; Bloemen et al. 2018)(see 7.5.2.2). This process has been applied in relation to coastal sea level rise starting with low risk, low cost measures and working up to measures requiring greater investment after review and reevaluation (Barnett et al. 2014). A first measure was stringent controls of new development, graduating to managed relocation of low lying critical infrastructure, and eventually movement of habitable dwellings to more elevated parts of town, as flooding and inundation triggers are experienced (Haasnoot et al. 2018; Lawrence et al. 2018; Barnett et al. 2014; Stephens et al. 2018). Nanda et al. (2018) apply the concept to a wetland in Australia to identify a mix of short and long-term decisions, and Prober et al. (2017) develop adaptation pathways for agricultural landscapes, also in Australia. Both studies identify that longer-term decisions may involve a considerable change to institutional arrangements at different scales. Viewing climate mitigation as a series of connected decisions over a long time period and not an isolated decision, reduces the fragmentation and uncertainty endemic of models and effectiveness of policy measures (Roelich and Giesekam 2019).

There is *medium evidence and high agreement* that participatory processes in adaptive governance within and across policy regimes overcome limitations of polycentric governance allowing priorities to be set in sustainable development through rural land management and integrated water resource management (Rouillard et al. 2013). Adaptive governance addresses large uncertainties and their social amplification through differing perceptions of risk (Kasperson 2012; Fra.Paleo 2015) offering an approach to co-evolve with risk by implementing policy mixes and assessing effectiveness in an ongoing process, making mid-point corrections when necessary (Fra.Paleo 2015). In respect of climate adaptation to coastal and riverine land erosion due to extreme weather events impacting communities, adaptive governance offers the capacity to monitor local socio-economic processes and implement dynamic locally informed institutional responses. In Alaska adaptive governance responded to the dynamic risk of extreme weather events and issue of climate migration by providing a continuum of policy from protection in place to community relocation, integrating across levels and actors in a more effective and less costly response option than other governance systems (Bronen and Chapin 2013). In comparison to other governance initiatives of ecosystem management aimed at conservation and sustainable use of natural capital, adaptive governance has visible effects on natural capital by monitoring, communicating and responding to ecosystem-wide changes at the landscape level (Schultz et al. 2015). Adaptive governance can be applied to manage drought assistance as a common property resource managing complex, interacting goals to create innovative policy options, facilitated through nested and polycentric systems of governance effected by areas of natural resource management including landscape care and watershed or catchment management groups (Nelson et al. 2008).

There is *medium evidence and high agreement* that transformational change is a necessary societal response option to manage climate risks which is uniquely characterised by the depth of change needed to reframe problems and change dominant mindsets, the scope of change needed (that is larger than just a few people) and the speed of change required to reduce emissions (O' Brien et al. 2012; Termeer et al. 2017). Transformation of governance occurs with changes in values to reflect an understanding that the environmental crisis occurs in the context of our relation with the earth (Hordijk et al. 2014; Pelling 2010). Transformation can happen by intervention strategies that enable small in-depth wins, amplify these small wins through integration into existing practices, and unblock stagnations (locked in structures) preventing transformation by confronting social and cognitive fixations with counterintuitive interventions (Termeer et al. 2017). Iterative consideration of issues and reformulation of policy instruments and response options facilitates transformation by allowing experimentation (Monkelbaan 2019).

Box 7.2 Adaptive Governance and interlinkages of food, fiber, water, energy and land

Emerging literature and case studies recognise the connectedness of the environment and human activities and the interrelationships of multiple resource-use practices in an attempt to understand synergies and trade-offs (Albrecht et al. 2018). Sustainable adaptation - or actions contributing to environmentally and socially sustainable development pathways (Eriksen et al. 2011) - requires consideration of the interlinkage of different sectors (Rasul and Sharma 2016). Integrating considerations can address sustainability (Hoff 2011) showing promise (Allan et al. 2015) for effective adaptation to climate impacts in many drylands (Rasul and Sharma 2016).

Case studies of integrated water resources management (IWRM), landscape and ecosystem based approaches illustrate important dimensions of institutions, institutional coordination, resource coupling and local and global connections (Scott et al. 2011). Integrated governance, policy coherence, and use of multi-functional systems are required to advance synergies across land, water, energy and food sectors (Liu et al. 2017).

Case Study: Flood and Food Security

Between 2003–2013 floods were the most impacting natural disaster on crop production (FAO 2015b) (albeit in certain contexts such riverine ecosystems and flood plain communities floods can be beneficial).

In developing countries flood jeopardises primary access to food and impacts livelihoods. In Bangladesh the 2007 flood reduced average consumption by 103Kcal/cap/day (worsening the existing 19.4% calories deficit) and in Pakistan the 2010 flood resulted in a loss of 205 Kcal/cap/day (or 8.5% of the Pakistan average food supply). The Pakistan 2010 flood affected over 4.5 million workers, two thirds employed in agriculture; 79% of farms lost greater than one half of their expected income (Pacetti et al. 2017).

Policy instruments and response respond to the sequential and cascading impacts of flood. In a Malawi study, flood impacts cascaded through labour, trade and transfer systems. First a harvest failure occurred, followed by the decline of employment opportunities and reduction in real wages, followed by a market failure or decline in trade, ultimately followed by a failure in informal safety nets (Devereux 2007). Planned policy responses include those that address the sequential nature of the cascading impacts starting with 'productivity-enhancing safety nets" addressing harvest failure, then public works programmes addressing the decline in employment opportunities, followed by food price subsidies to address the market failure, and finally food aid to address the failure of informal safety nets (Devereux 2007). In another example in East Africa range lands, flood halted livestock sales, food prices fell, and grain production ceased. Local food shortages couldn't be supplemented with imports due to destruction of transport links, and pastoral incomes were inadequate to purchase food. Livestock diseases became rampant and eventually food shortages led to escalating prices. Due to the contextual nature and timing of events, policy response initially addressed mobility and resource access, and eventually longer term issues such as livestock disease (Little et al. 2001).

In North America floods are often described in terms of costs. For instance, the 1997 Red River Basin flood cost Manitoba, Canada $1 billion US and the United States of America, $4 billion US in terms of impact on agriculture and food production (Adaptation to Climate Change Team 2013). In Canada floods accounted for 82% of disaster financial assistance spent from 2005–2014 (Public

Safety Canada 2017) and this cost may increase in the future. Future climate change may result in a six foot rise in sea level by 2100 costing from USD 507 to 882 billion, affecting 300 American cities (losing one half of their homes) and the wholesale loss of 36 cities (Lemann 2018).

Policy measures are important as an increasingly warming world may make post disaster assistance and insurance increasingly unaffordable (Surminski et al. 2016). Historic legal mechanisms for retreating from low lying and coastal areas have failed to encourage relocation of people out of flood plains and areas of high risk (Stoa 2015). In some places cheap flood insurance and massive aid programs have encouraged the populating of low-lying flood prone and coastal areas (Lemann 2018). Although the state makes disaster assistance payments, it is local governments that determine vulnerability through flood zone mapping, restrictions from building in flood zones, building requirements (Stoa 2015), and integrated planning for flood. A comprehensive policy mix (see 7.4.8) (implemented through adaptive management as illustrated on Figure 7.6) reduces vulnerability (Hurlbert 2018b) (Hurlbert 2018a). Policy mixes that allow people to respond to disasters include bankruptcy, insolvency rules, house protection from creditors, income minimums, and basic agricultural implement protection laws. The portfolio of policies allows people to recover, and if necessary migrate to other areas and occupations (Hurlbert 2018b).

At the international level, reactionary disaster response has evolved to proactive risk management that combines adaptation and mitigation responses to ensure effective risk response, build resilient systems and solve issues of structural social inequality (Innocenti and Albrito 2011). Advance measures of preparedness are the main instruments to reduce fatalities and limit damage, as illustrated on the figure below. The Sendai Declaration and Framework for Disaster Risk Reduction 2015-2030, is an action plan to reduce mortality, the numbers of affected people and economic losses with four priorities - understanding disaster risk, strengthening its governance to enhance the ability to manage disaster risk, investing in resilience, and enhancing disaster preparedness. There is *medium evidence and high agreement* that the Sendai Framework significantly refers to adaptive governance and could be a window of opportunity to transform disaster risk reduction to address the causes of vulnerability (Munene et al. 2018). Addressing disasters increasingly requires individual, household, community and national planning and commitment to a new path of resilience and shared responsibility through whole community engagement and linking private and public infrastructure interests (Rouillard et al. 2013). It is recommended that a vision and overarching framework of governance be adopted to allow participation and coordination by government, nongovernmental organisations, researchers and the private sector, individuals in the neighbourhood community. Disaster risk response is enhanced with complementary structural and non-structural measures implemented together with measurable scorecard indicators (Chen 2011).

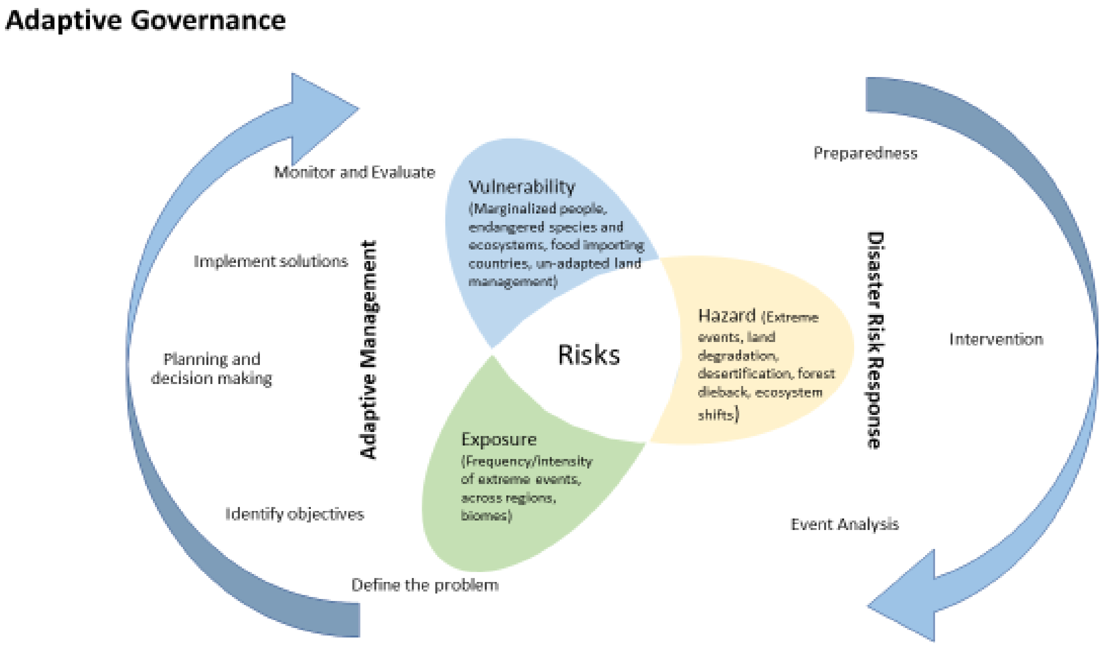

Figure 7.8 Adaptive Governance

Adaptive management identifies and responds to exposure and vulnerability to land and climate change impacts by identifying problems and objectives, making decisions in relation to response options and instruments advancing response options in the context of uncertainty. These decisions are continuously monitored, evaluated and adjusted to changing conditions. Similarly disaster risk management responds to hazards through preparation, prevention, response, analysis, and reconstruction in an iterative process.

7.6.4. Participation

It is recognised that more benefits are derived when citizens actively participate in land and climate decision making, conservation, and policy formation (*high confidence*) (Jansujwicz et al. 2013) (Coenen and Coenen 2009; Hurlbert and Gupta 2015). Local leaders supported by strong laws, institutions, collaborative platforms, are able to draw on local knowledge, challenge external scientists, and find transparent and effective solutions for climate and land conflicts (Couvet and Prevot 2015; Johnson et al. 2017). Meaningful participation is more than providing technical/scientific information to citizens in order to accept decisions already made, but allows citizens to deliberate about climate change impacts to determine shared responsibilities creating genuine opportunity to construct, discuss, and promote alternatives (*high confidence*)(Lee et al. 2013; Armeni 2016; Pieraccini 2015)(Serrao-Neumann et al. 2015b; Armeni 2016). Participation is an emerging quality of collective-action and social-learning processes (see below) (Castella et al. 2014) when barriers for meaningful participation are surpassed (Clemens et al. 2015). The absence of systematic leadership, the lack of consensus on the place of direct citizen participation, and the limited scope and powers of participatory innovations limits the utility of participation (Fung 2015).

Multiple methods of participation exist, including multi-stakeholder forums, participatory scenario analyses, public forums and citizen juries (Coenen and Coenen 2009). No one method is superior, but each method must be tailored for local context (*high confidence*)(Blue and Medlock 2014; Voß and

Amelung 2016). Strategic innovation in developing policy initiatives requires a strategic adaptation framework involving pluralistic and adaptive processes and use of boundary organisations (Head 2014).

The framing of a land and climate issue can influence the manner of public engagement (Hurlbert and Gupta 2015) and studies have found local frames of climate change are particularly important (Hornsey et al. 2016; Spence et al. 2012), emphasising diversity of perceptions to adaptation and mitigation options (Capstick et al. 2015) (although Singh and Swanson (2017) found little evidence framing impacted the perceived importance of climate change).

Recognition and use of indigenous and local knowledge (ILK) is an important element of participatory approaches of various kinds. ILK can be used in decision-making on climate change adaptation, Sustainable Land Management and food security at various scales and levels and is important for long-term sustainability *(high confidence).* Cross-Chapter Box 13 discusses definitional issues associated with ILK, evidence of its usefulness in responses to land-climate challenges, constraints on its use, and possibilities for its incorporation in decision-making.

Cross-Chapter Box 13: Indigenous and Local Knowledge

John Morton (United Kingdom), Fatima Denton (The Gambia), James Ford (United Kingdom), Joyce Kimutai (Kenya), Pamela McElwee (The United States of America), Marta Rivera Ferre (Spain), Lindsay Stringer (United Kingdom)

Indigenous and local knowledge (ILK) can play a key role in climate change adaptation (*high confidence*) (Mapfumo et al. 2017; Nyong et al. 2007b; Green and Raygorodetsky 2010; Speranza et al. 2010; Alexander et al. 2011a; Leonard et al. 2013; Nakashima et al. 2013; Tschakert 2007). The Summary for Policy-Makers of the Working Group II Contribution to the IPCC's Fifth Assessment Report (IPCC 2014b, p. 26) states that "Indigenous, local, and traditional knowledge systems and practices, including indigenous peoples' holistic view of community and environment, are a major resource for adapting to climate change, but these have not been used consistently in existing adaptation efforts. Integrating such forms of knowledge with existing practices increases the effectiveness of adaptation" (see also Ford et al. 2016). The Special Report on Global Warming of 1.5 °C (IPCC 2018e; de Coninck et al. 2018) confirms the effectiveness and potential feasibility of adaptation options based on ILK but also raises concerns that such knowledge systems are being threatened by multiple socio-economic and environmental drivers (*high confidence*). The Intergovernmental Platform on Biodiversity and Ecosystem Services (IPBES) Land Degradation and Restoration Assessment (IPBES 2018) finds the same– that ILK can support adaptation to land degradation but is threatened.

A variety of terminology has been used to describe indigenous and local knowledge: "Indigenous knowledge", "local knowledge", "traditional knowledge", "traditional ecological knowledge" and other terms are used in overlapping and often inconsistent ways (Naess 2013). The Special Report on Global Warming of 1.5°C (IPCC 2018a) reserves "indigenous knowledge" for culturally distinctive ways of knowing associated with "societies with long histories of interaction with their natural surroundings", while using "local knowledge" for "understandings and skills developed by individuals and populations, specific to the places where they live", but not all research studies observe this distinction. This Special Report generally uses "indigenous and local knowledge" (ILK) as a combined term for these forms of knowledge, but in some sections the terminology used follows that from the research literature assessed.

In contrast to scientific knowledge, ILK is context-specific, collective, transmitted informally, and is multi-functional (Mistry and Berardi 2016; Naess 2013; Janif et al. 2016). Persson et al. (2018) characterise ILK as "practical experience", as locally-held knowledges are acquired through processes of experience and interaction with the surrounding physical world. ILK is embedded in local institutions (Naess 2013) and in cultural aspects of landscape and food systems (Fuller and Qingwen 2013; Koohafkan and Altieri 2011). ILK can encompass such diverse content as factual information about the environment; guidance on rights and management; value statements about interactions with others; and cosmologies and worldviews that influence how information is perceived and acted upon, among other topics (Spoon 2014; Usher 2000).

This Cross-Chapter Box assesses evidence for the positive role of ILK in understanding climate change and other environmental processes, and in managing land sustainably in the face of climate change, desertification, land degradation and food insecurity. It also assesses constraints on and threats to the use of ILK in these challenges, and processes by which ILK can be incorporated in decision-making and governance processes.

ILK in understanding and responding to climate change impacts

ILK can play a role in understanding climate change and other environmental processes, particularly where formal data collection is sparse (Alexander et al. 2011a; Schick et al. 2018), and can contribute to accurate predictions of impending environmental change (Green and Raygorodetsky 2010; Orlove et al. 2010) (*medium confidence*). Both at global level (Alexander et al. 2011a; Green and Raygorodetsky 2010), and local level (Speranza et al. 2010; Ayanlade et al. 2017), strong correlations between local perceptions of climate change and meteorological data have been shown, as calendars, almanacs, and other seasonal and interannual systems knowledge embedded in ILK hold information about environmental baselines (Orlove et al. 2010; Cochran et al. 2016).

ILK is strongly associated with sustainable management of natural resources, including land, and with autonomous adaptation to climate variability and change, while also serving as a resource for externally-facilitated adaptation (Stringer et al. 2009). For example, women’s traditional knowledge adds value to a society’s knowledge base and supports climate change adaptation practices (Lane and McNaught 2009). In dryland environments, populations have historically demonstrated remarkable resilience and innovation to cope with high climatic variability, manage dynamic interactions between local communities and ecosystems, and sustain livelihoods (Safriel and Adeel 2008; Davies 2017). There is *high confidence* that pastoralists have created formal and informal institutions based on ILK for regulating grazing, collection and cutting of herbs and wood, and use of forests across the Middle East and North Africa (Louhaichi and Tastad 2010; Domínguez 2014; Auclair et al. 2011), Mongolia (Fernandez-Gimenez 2000), The Horn of Africa (Oba 2013) and the Sahel (Krätli and Schareika 2010). Herders in both the Horn of Africa and the Sahel have developed complex livestock breeding and selection systems for their dryland environment (Krätli 2008; Fre 2018). Numerous traditional water harvesting techniques are used across the drylands to adapt to climate variability: planting pits (“zai”, “ngoro”) and micro-basins and contouring hill slopes and terracing (Biazin et al. 2012), alongside the traditional “ndiva” water harvesting system in Tanzania to capture runoff in community-managed micro-dams for small-scale irrigation (Enfors and Gordon 2008).

Across diverse agro-ecological systems, ILK is the basis for traditional practices to manage the landscape and sustain food production, while delivering co-benefits in the form of biodiversity and ecosystem resilience at a landscape scale (*high confidence*). Flexibility and adaptiveness are hallmarks of such systems (Richards 1985; Biggs et al. 2013), and documented examples include: traditional integrated watershed management in the Philippines (Camacho et al. 2016); widespread use of terracing with benefits in cases of both intensifying and decreasing rainfall (Arnáez et al. 2015; Chen et al. 2017b) and management of water harvesting and local irrigation systems in the Indo-Gangetic Plain (Rivera-Ferre et al. 2016). Rice cultivation in East Borneo is sustained by traditional forms of shifting cultivation, often involving intercropping of rice with bananas, cassava and other food crops (Siahaya et al. 2016), although the use of fire in land clearance implies trade-offs for climate change mitigation which have been sparsely assessed. Indigenous practices for enhanced soil fertility have been documented among South Asian farmers (Chandra et al. 2011; Dey and Sarkar 2011) and among Mayan farmers where management of carbon has positive impacts on mitigation (Falkowski et al. 2016). Korean traditional groves or “bibosoop” have been shown to reduce wind speed and evaporation in agricultural landscapes (Koh et al. 2010). Particularly in the context of changing climates, agriculture based on ILK that focuses on biodiversification, soil management, and sustainable water harvesting holds promise for long-term resilience (Altieri and Nicholls 2017) and rehabilitation of degraded land (Maikhuri et al. 1997). ILK is also important in other forms of ecosystem management, such as forests and wetlands, which may be conserved by efforts such as sacred sites (Ens et al. 2015; Pungetti et al. 2012) and ILK can play an important role in ecological restoration efforts, including for carbon sinks, through knowledge surrounding species selection and understanding of ecosystem processes, like fire (Kimmerer 2000).

Constraints on the use of ILK

Use of ILK as a resource in responding to climate change can be constrained in at least three ways (high confidence). Firstly the rate of climate change and the scale of its impacts may render incremental adaptation based on the ILK of smallholders and others, less relevant and less effective (Lane and McNaught 2009; Orlowsky and Seneviratne 2012; Huang et al. 2016; Morton 2017). Secondly, maintenance and transmission of ILK across generations may be disrupted by e.g.: formal education, missionary activity, livelihood diversification away from agriculture, and a general perception that ILK is outdated and unfavourably contrasted with scientific knowledge (Speranza et al. 2010), and by HIV-related mortality (White and Morton 2005). Urbanisation can erode ILK, although ILK is constantly evolving, and becoming integrated into urban environments (Júnior et al. 2016; Oteros-Rozas et al. 2013; van Andel and Carvalheiro 2013). Thirdly, ILK holders are experiencing difficulty in using ILK due to loss of access to resources, such as through large-scale land acquisition (Siahaya et al. 2016; Speranza et al. 2010; de Coninck et al. 2018) and the increasing globalisation of food systems and integration into global market economy also threatens to erode ILK (Gómez-Baggethun et al. 2010; Oteros-Rozas et al. 2013; McCarter et al. 2014). The potential role that ILK can play in adaptation at the local level depends on the configuration of a policy-institutions-knowledge nexus (Stringer et al. 2018), which includes power relations across levels and interactions with government strategies (Alexander et al. 2011b; Naess 2013).

Incorporation of ILK in decision-making

ILK can be used in decision-making on climate change adaptation, Sustainable Land Management and food security at various scales and levels and is important for long-term sustainability (high confidence). Respect for ILK is both a requirement and an entry strategy for participatory climate action planning and effective communication of climate action strategies (Nyong et al. 2007b). The nature, source, and mode of knowledge generation are critical to ensure that sustainable solutions are community-owned and fully integrated within the local context (Mistry and Berardi 2016). Integrating ILK with scientific information is a prerequisite for such community-owned solutions. Scientists can engage farmers as experts in processes of knowledge co-production (Oliver et al. 2012), helping to introduce, implement, adapt and promote locally appropriate responses (Schwilch et al. 2011). Specific approaches to decision-making that aim to integrate indigenous and local knowledge include some versions of decision support systems (Jones et al. 2014) as well as citizen science and participatory modelling (Tengö et al. 2014).

ILK can be deployed in the practice of climate governance especially at the local level where actions are informed by the principles of decentralisation and autonomy (Chanza and de Wit 2016; Harmsworth and Awatere 2013). International environmental agreements also are increasingly including attention to ILK and diverse cultural perspectives, for reasons of social justice and inclusive decision-making (Brondizio and Tourneau 2016). However, the context-specific, and dynamic nature of ILK and its embeddedness in local institutions and power relations needs consideration (Naess 2013). It is also important to take a gendered approach so as not to further marginalise certain knowledge, as men and women hold different knowledge, expertise and transmission patterns (Díaz-Reviriego et al. 2017).

Citizen Science

Citizen science is a democratic approach to science involving citizens in collecting, classifying, and interpreting data to influence policy and assist decision processes, including issues relevant to the environment (Kullenberg and Kasperowski 2016). It has flourished in recent years due to easily available technical tools for collecting and disseminating information (e.g., cell phone-based apps, cloud-based services, ground sensors, drone imagery, and others), recognition of its free source of labour, and requirements of funding agencies for project related outreach (Silvertown 2009). There is significant potential for combining citizen science and participatory modelling to obtain favourable outcomes and improve environmental decision making (*medium confidence*) (Gray et al. 2017). Citizen participation in land use simulation integrates stakeholders' preferences through the generation of parameters in analytical and discursive approaches (Hewitt et al. 2014), and thereby supports the translation of narrative scenarios to quantitative outputs (Mallampalli et al. 2016), supports the development of digital tools to be used in co-designing decision making participatory structures (Bommel et al. 2014), and supports the use of games to understand the preferences of local decision making when exploring various balanced policies about risks (Adam et al. 2016).

There is *medium confidence* that citizen science improves sustainable land management through mediating and facilitating landscape conservation decision making and planning, as well as boosting environmental awareness and advocacy (Lange and Hehl-Lange 2011; Bonsu et al. 2017; Graham et al. 2015) (Bonsu et al. 2017) (Lange and Hehl-Lange 2011) (Sayer, J. Margules, C., Boedhihartono 2015) (McKinley et al. 2017) (Johnson et al. 2017, 2014) (Gray et al. 2017). One study found limited evidence of direct conservation impact (Ballard et al. 2017) and most of the cases derive from rich industrialised countries (Loos et al. 2015). There are many practical challenges to the concept of citizen science at the local level, which include differing methods and the lack of universal implementation framework (Conrad and Hilchey 2011; Jalbert and Kinchy 2016; Stone et al. 2014). Uncertainty related to citizen science needs to be recognised and managed (Swanson et al. 2016; Bird et al. 2014; Lin et al. 2015) and citizen science projects around the world need better coordination to understand significant issues, such as climate change (Bonney et al. 2014).

Participation, Collective Action, and Social Learning

As land and climate issues cannot be solved by one individual, a diverse collective action issue exists for land use policies and planning practices (Moroni 2018) at local, national, and regional levels. Collective action involves individuals and communities in land planning processes in order to determine successful climate adaptation and mitigation (Nkoana et al. 2017) (Liu and Ravenscroft 2017) (Nieto-Romero et al. 2016; Nikolakis et al. 2016), or as Sarzynski (2015) finds, a community 'pulling together' to solve common adaptation and land planning issues.

Collective action offers solutions for emerging land and climate change risks, including strategies that target maintenance or change of land use practices, increase livelihood security, risk share through pooling, and sometimes also aim to promote social and economic goals such as reducing poverty (Samaddar et al. 2015)(Andersson and Gabrielsson 2012). Collective action has resulted in the successful implementation of national-level land transfer policies (Liu and Ravenscroft 2017), rural development and land sparing (Jelsma et al. 2017), and the development of tools to identify shared objectives, trade-offs and barriers to land management (Nieto-Romero et al. 2016; Nikolakis et al. 2016). Collective action can also produce mutually binding agreements, government regulation, privatisation, and incentive systems (IPCC 2014c).

Successful collective action requires understanding and implementation of factors that determine successful participation in climate adaptation and mitigation (Nkoana et al. 2017). These include ownership, empowerment or self-reliance, time effectiveness, economic and behavioural interests, livelihood security, and the requirement for plan implementation (Samaddar et al. 2015; Djurfeldt et al. 2018) (Sánchez and Maseda 2016). In a UK study, dynamic trust relations among members around specific issues, determined the potential of agri-environmental schemes to offer landscape-scale environmental protection (Riley et al. 2018). Collective action is context specific and rarely scaled up or replicated in other places (Samaddar et al. 2015).

Collective action in land use policy has been shown to be more effective when implemented as bundles of actions rather than as single-issue actions. For example, land tenure, food security, and market access can mutually reinforce each other when they are interconnected (Corsi et al. 2017). For example, (Liu and Ravenscroft 2017) found that financial incentives embedded in collective forest reforms in China have increased forest land and labour inputs in forestry.

A product of participation, equally important in practical terms, is social learning (*high confidence*) (Reed et al. 2010) (Dryzek and Pickering 2017) (Gupta 2014), which is learning in and with social groups through interaction (Argyris 1999) including collaboration and organisation which occurs in networks of interdependent stakeholders (Mostert et al. 2007). Social learning is defined as a change in understanding measured by a change in behaviour, and perhaps worldview, by individuals and wider social units, communities of practice and social networks (Reed et al. 2010) (Gupta 2014). Social learning is an important factor contributing to long-term climate adaptation whereby individuals and organisations engage in a multi-step social process, managing different framings of issues while raising awareness of climate and land risks and opportunities, exploring policy options and institutionalising new rights, responsibilities, feedback and learning processes (Tàbara et al. 2010). It is important for engaging with uncertainty (Newig et al. 2010) and addressing the increasing unequal geography of food security (Sonnino et al. 2014).

Social learning is achieved through reflexivity or the ability of a social structure, process, or set of ideas to reconfigure itself after reflection on performance though open-minded people interacting iteratively to produce reasonable and well-informed opinions (Dryzek and Pickering 2017). These processes develop through skilled facilitation attending to social difference and power resulting in a shared view of how change might happen (Harvey et al. 2012; Ensor and Harvey 2015). When combined with collective action, social learning can make transformative change measured by a change in worldviews (beliefs about the world and reality) and understanding of power dynamics (Gupta 2014) (Bamberg et al. 2015).

7.6.5. Land Tenure

Land tenure, defined as “the terms under which land and natural resources are held by individuals, households or social groups”, is a key dimension in any discussion of land-climate interactions, including the prospects for both adaptation and land-based mitigation, and possible impacts on tenure and thus land security of both climate change and climate action (Quan and Dyer 2008) (*medium evidence, high agreement*).

Discussion of land tenure in the context of land-climate interactions in developing countries needs to consider the prevalence of informal, customary and modified customary systems of land tenure: estimates range widely, but perhaps as much as 65% of the world’s total land area is managed under some form of these local, customary or communal tenure systems, and only a small fraction of this (around 15%) is formally recognised by governments (Rights and Resources Initiative 2015a). These customary land rights can extend across many categories of land, but are difficult to assess properly due to poor reporting, lack of legal recognition, and lack of access to reporting systems by indigenous and rural peoples (Rights and Resources Initiative 2018a). Around 521 million ha of forest land is estimated to be legally owned, recognised, or designated for use by indigenous and local communities as of 2017 (Rights and Resources Initiative 2018b), predominantly in Latin America, followed by Asia. However in India approximately 40 million ha of forest land is managed under customary rights not recognised by the government (Rights and Resources Initiative 2015b). In 2005 only 1% of land in Africa was legally registered (Easterly 2008a).

Much of the world's carbon is stored in the biomass and soil on the territories of customary landowners including indigenous peoples (Walker et al. 2014; Garnett et al. 2018), making securing of these land tenure regimes vital in land and climate protection. These lands are estimated to hold at least 293 GtC of carbon, of which around one-third (72 GtC) is located in areas where indigenous peoples and local communities lack formal recognition of their tenure rights (Frechette et al. 2018).

Understanding the interactions between land tenure and climate change has to be based on underlying understanding of land tenure and land policy and how they relate to sustainable development, especially in low- and middle-income countries: such understandings have changed considerably over the last three decades, and now show that informal or customary systems can provide secure tenure (Toulmin and Quan 2000). For smallholder systems, (Bruce and Migot-Adholla 1994) among other authors established that African customary tenure can provide the necessary security for long-term investments in farm fertility such as tree-planting. For pastoral systems, (Behnke 1994; Lane and Moorehead 1995) and other authors showed the rationality of communal tenure in situations of environmental variability and herd mobility. However, where customary systems are unrecognised or weakened by governments or the rights from them undocumented or unenforced, tenure insecurity may result (Lane 1998; Toulmin and Quan 2000). There is strong empirical evidence of the links between secure communal tenure and lower deforestation rates, particularly in intact forests (Nepstad et al., 2006; Persha, Agrawal, & Chhatre, 2011; Vergara-Asenjo & Potvin, 2014). Securing and recognising tenure for indigenous communities (such as through revisions to legal or policy frameworks) has been shown to be highly cost effective in reducing deforestation and improving land management in certain contexts, and is therefore also apt to help improve indigenous communities' ability to adapt to climate changes(Suzuki 2012; Balooni et al. 2008; Ceddia et al. 2015; Pacheco et al. 2012; Holland et al. 2017).

Rights to water for agriculture or livestock are linked to land tenure in complex ways still little understood and neglected by policy-makers and planners(Cotula 2006a). Provision of water infrastructure tends to increase land values, but irrigation schemes often entail reallocation of land rights (Cotula 2006b) and new inequalities based on water availability such as the creation of a category of tailenders in large-scale irrigation (Chambers 1988) and disruption of pastoral grazing patterns through use of riverine land (Behnke and Kerven 2013).

Understanding of land tenure under climate change also has to take account of the growth in large-scale land acquisitions (LSLAs), also referred to as land-grabbing, in developing countries. These LSLAs are defined by acquisition of more than 200 ha per deal (Messerli et al. 2014a). Klaus Deininger (2011) links the growth in demand for land to the 2007-2008 food price spike, and demonstrates that high levels of demand for land at the country level are statistically associated with weak recognition of land rights. Land grabs, where LSLAs occur despite local use of lands, are often driven by direct collaboration of politicians, government officials and land agencies (Koechlin et al. 2016), involving corruption of governmental land agencies, failures to register community land claims and illegal lands uses and lack of the rule of law and enforcement in resource extraction frontiers (Borras Jr et al. 2011). Though data is poor, overall, small and medium scale domestic investment has in fact been more important than foreign investment (Deininger 2011; Cotula et al. 2014). There are variations in estimates of the scale of large-scale land acquisitions:the Nolte et al. (2016) report concluded deals totalling 42.2 million ha worldwide. Cotula et al. (2014) using cross-checked data for completed lease agreements in Ethiopia, Ghana and Tanzania conclude they cover 1.9%, 1.9% and 1.1% respectively of each country's total land suitable for agriculture. The literature expresses different views on whether these acquisitions concern marginal lands or lands already in use thereby displacing existing users (Messerli et al. 2014b). Land-grabbing is associated with and may be motivated by the acquisition of rights to water, and erosion of those rights for other users such as those downstream (Mehta et al. 2012). Quantification of the acquisition of water rights resulting from LSLAs raises major issues of definition, data availability, and measurement. One estimate of the total acquisition of gross irrigation water associated with land-grabbing across the 24 countries most affected is 280 billion m^3 (Rulli et al. 2013).

While some authors see LSLAs as investments that can contribute to more efficient food production at larger scales (World Bank 2011; Deininger and Byerlee 2012), others have warned that local food security may be threatened by them(Daniel 2011; Golay and Biglino 2013; Lavers 2012). Reports suggest that recent land grabbing has affected 12 million people globally in terms of declines in welfare (Adnan 2013; Davis et al. 2014). De Schutter (2011) argues that large-scale land acquisitions will a) result in types of farming less liable to reduce poverty than smallholder systems, b) increase local vulnerability to food price shocks by favouring export agriculture and c) accelerate the development of a market for land with detrimental impacts on smallholders and those depending on common property resources. Land grabbing can threaten not only agricultural lands of farmers, but also protected ecosystems, like forests and wetlands (Hunsberger et al. 2017; Carter et al. 2017; Ehara et al. 2018).

The primary mechanisms for combatting LSLAs have included restrictions on the size of land sales (Fairbairn 2015); pressure on agribusiness companies to agree to the Voluntary Guidelines on the Responsible Governance of Tenure of Land, Fisheries and Forests in the Context of National Food Security, known as the VGGT, or similar principles (Collins 2014; Goetz 2013); attempts to repeal biofuels standards (Palmer 2014); strengthening of existing land law and land registration systems (Bebbington et al. 2018); use of community monitoring systems (Sheil et al. 2015); and direct protests against the land acquisitions (Hall et al. 2015; Fameree 2016).

Table 7.7 sets out, in highly summarised form, some key findings on the multi-directional inter-relations between land tenure and climate change, with particular reference to developing countries. The rows represent different categories of landscape or resource systems. For each system the second column summarises current understandings on land tenure and sustainable development, in many case predating concerns over climate change. The third column summarises the most important implications of land tenure systems, policy about land tenure, and the implementation of that policy, for vulnerability and adaptation to climate change, and the fourth gives a similar summary for mitigation of climate change. The fifth column summarises key findings on how climate change and climate action (both adaptation and mitigation) will impact land tenure, and the final column, findings on implications of climate change for evolving land policy.

1 **Table 7.7 Major Findings on the Interactions between Land Tenure and Climate Change**

Landscape or natural resource system	State of understanding of land tenure, land policy and sustainable development	Implications of land tenure for vulnerability and adaptation to climate change	Implications of land tenure for mitigation of climate change	Impacts of climate change and climate action on land tenure	Implications of climate change and climate action for land policy
Smallholder cropland	In South Asia and Latin America the poor suffer from limited access including insecure tenancies, though this has been partially alleviated by land reform.[1] In Africa informal/customary systems may provide considerable land tenure security and enable long-term investment in land management, but are increasingly weakened by demographic pressures on available land resources increase. However, creation of freehold rights through conventional land titling is not a necessary condition for tenure security and may be cost-ineffective or counter-productive.[2,3,4,5] Alternative approaches utilising low cost technologies and participatory methods are available.[6] Secure and defendable land tenure, including modified customary tenure, has been positively correlated with food production increases.[7,8,9]	Insecure land rights are one factor deterring adaptation and accentuating vulnerability.[10,11] Specific dimensions of inequity in customary systems may act as constraints on adaptation in different contexts.[12] LSLAs may be associated with monoculture and other unsustainable land use practices, have negative consequences for soil degradation[13] and disincentivise more sustainable forms of agriculture.[14]	Secure land rights, including through customary systems, can incentivise farmers to adopt long-term climate-smart practices,[15] e.g., planting trees in mixed cropland/forest systems.[16]	Increased frequency and intensity of extreme weather can lead to displacement and effective loss of land rights.[17] REDD+ programmes tend slightly to increase land tenure insecurity on agricultural forest frontier lands, - but not in forests.[18]	Landscape governance and resource tenure reforms at farm and community levels can facilitate and incentivise planning for landscape management and enable the integration of adaptation and mitigation strategies.[11]
Rangelands	Communal management of rangelands in pastoral systems is a rational and internally sustainable response to climate variability and the need for mobility. Policies favouring individual or small group land-tenure may have negative impacts on both ecosystems and livelihoods.[19,20,21]	Many pastoralists in lands at risk from desertification do not have secure land tenure, and erosion of traditional communal rangeland tenure has been identified as a determinant of increasing vulnerability to drought and climate change and as a driver of dryland	Where pastoralists' traditional land use does not have legal recognition, or where pastoralists are unable to exclude others from land use, this presents significant challenges for carbon sequestration initiatives.[27,28]	Increasing conflict on rangelands is a possible result of climate change and environmental pressures, but depends on local institutions.[29] Where land use rights for pastoralists are absent or unenforced, demonstrated potential for carbon sequestration may assist	Carbon sequestration initiatives on rangelands may require clarification and maintenance of land rights.[27,28]

		degradation.[22,23,24,25,26]		advocacy.[28]	
Forests	Poor management of state and open-access forests has been combatted in recent years by a move towards forest decentralisation and community co-management.[30,31,32,33,34,35] Land tenure systems have complex interactions with deforestation processes. Land tenure security is generally associated with less deforestation, regardless of whether the tenure form is private, customary or communal.[33,36,37,38] Historical injustices towards forest dwellers can be ameliorated with appropriate policy, e.g., 2006 Forest Rights Act in India.[39]	Land tenure security can lead to improved adaptation outcomes[40,41,42,43] but land tenure policy for forests that focuses narrowly on cultivation has limited ability to reduce ecological vulnerability or enhance adaptation.[39] Secure rights to land and forest resources can facilitate efforts to stabilise shifting cultivation and promote more sustainable resource use if appropriate technical and market support are available.[44]	Land tenure insecurity has been identified as a key driver of deforestation and land degradation leading to loss of sinks and creating sources of GHGs[45,46,47,48,49] . While land tenure systems interact with land-based mitigation actions in complex ways,[36] forest decentralisation and community co-management has shown considerable success in slowing forest loss and contributing to carbon mitigation.[30,31,32,33,34,35] Communal tenure systems may lower transaction costs for REDD+ schemes, though with risk of elite capture of payments.[16]	Findings on both direction of change in tenure security and extent to which this has been influenced by REDD+ are very diverse.[m] The implications of land-based mitigation (e.g., BECCS) on land tenure systems is currently understudied, but evidence from biofuels expansion shows negative impacts on local livelihoods and loss of forest sinks where LSLAs override local land tenure.[50,51]	Forest tenure policies under climate change need to accommodate and enable evolving and shifting boundaries linked to changing forest livelihoods.[10] REDD+ programmes need to be integrated with national-level forest tenure reform.[18]
Poor and informal urban settlements	Residents of poor and informal urban settlements enjoy varying degrees of tenure security from different forms of tenure. Security will be increased by building on de facto rights rather than through abrupt changes in tenure systems.[52]	Public land on the outskirts of urban areas can be used to adapt to increasing flood risks by protecting natural assets.[53] Secure land titles in hazardous locations may make occupants reluctant to move and raise the costs of compensation and resettlement.[17]	Urban land use strategies such as tree planting, establishing public parks, can save energy usage by moderating urban temperature and protect human settlement from natural disaster such as flooding or heatwaves.[54]	Without proper planning, climate hazards can undermine efforts to recognise and strengthen informal tenure rights without proper planning.[55,56]	Climate risks increase the requirements for land use planning and settlement that increases tenure security, with direct involvement of residents, improved use of public land, and innovative collaboration with private and traditional land owners.[56,57]
Riverscapes and riparian fringes	Well-defined but spatially flexible community tenure can support regulated and sustainable artisanal capture	Unequal land rights and absence of land management	Mitigation measures such as protection of riparian forests and grasslands can		Secured but spatially flexible tenure will enable climate change mitigation

	fisheries and biodiversity.[58,59,60,61,62,63,64]	arrangements in floodplains increases vulnerability and constrains adaptation.[65] Marginalised or landless fisherfolk will be empowered by tenurial rights and associated identity to respond more effectively to ecological changes in riverscapes including riparian zones.[66,67,68,69]	potentially play a major role, provided rights to land and trees are sufficiently clear.[70,71]		in riverscapes to be synergised with local livelihoods and ecological security.[67,72]

Sources: 1) Binswanger et al. 1995 2) Schlager and Ostrom 1992 3) Toulmin and Quan 2000 4) Bruce and Migot-Adholla 1994 5) Easterly 2008 6) McCall and Dunn 2012 7) Maxwell and Wiebe 1999 8) Holden and Ghebru 2016 9) Corsi et al. 2017 10) Quan et al. 2017 11) Harvey et al. 2014 12) Antwi-Agyei et al. 2015 13) Balehegn, 2015 14) Friis & Nielsen, 2016 15) Scherr et al. 2012 16) Barbier and Tesfaw 2012 17) Mitchell 2010 18) Sunderlin et al. 2018 19) Behnke 1994 20) Lane and Moorehead 1995 21) Davies et al. 2015 22) Morton 2007 23) López-i-Gelats et al. 2016 24) Oba 1994 25) Fraser et al. 2011 26) Dougill et al. 2011 27) Roncoli et al. 2007. 28) Tennigkeit and Wilkes 2008 29) Adano et al. 2012 30) Agrawal, Chhatre, & Hardin, 2008 31) Chhatre & Agrawal, 2009 32 Gabay & Alam, 2017 33) Holland et al., 2017 34) Larson & Pulhin, 2012 35) Pagdee, Kim, & Daugherty, 2006) 36) Robinson et al. 2014 37) Blackman et al. 2017 38) Nelson et al. 2001; 38) Ramnath 2008 40) Suzuki 2012 41) Balooni et al. 2008 42) Ceddia et al. 2015 43) Pacheco et al. 2012) 44) Garnett et al. 2013 45) Clover & Eriksen, 2009 46) Damnyag, Saastamoinen, Appiah, & Pappinen, 2012 47) Finley-Brook, 2007 48) Robinson, Holland, & Naughton-Treves, 2014 49) Stickler, Huntington, Haflett, Petrova, & Bouvier, 2017 50) Romijn, 2011 51) Aha & Ayitey, 2017 52) Payne 2001 53) Barbedo et al. 2015 54) Zhao et al. 2018 55) Satterthwaite et al. 2018 56) Mitchell et al. 2015 57) Satterthwaite 2007 58) Thomas 1996 59) Welcomme et al. 2010 60) Silvano and Valbo-Jørgensen 2008 61) Biermann et al. 2012 62) Abbott et al. 2007 63) Béné et al. 2011 64) McGrath et al. 1993 65) Barkat et al. 2001 66) FAO 2015 67) Hall et al. 2013 68) Berkes 2001 69) ISO 2017 70) Rocheleau and Edmunds 1997 71) Baird and Dearden 2003 72) Béné et al. 2010.

In drylands, weak land tenure security, either for households disadvantaged within a customary tenure system or more widely as such a system is eroded, can be associated with increased vulnerability and decreased adaptive capacity (*limited evidence, high agreement*). There is *medium evidence* and *medium agreement* that land titling and recognition programs, particularly those that authorise and respect indigenous and communal tenure, can lead to improved management of forests, including for carbon storage (Suzuki 2012; Balooni et al. 2008; Ceddia et al. 2015; Pacheco et al. 2012), primarily by providing legally secure mechanisms for exclusion of others (Nelson et al. 2001; Blackman et al. 2017). However, these titling programs are highly context-dependent and there is also evidence that titling can exclude community and common management, leading to more confusion over land rights, not less, where poorly implemented (Broegaard et al. 2017). For all the systems, an important finding is that land policies can provide both security and flexibility in the face of climate change, but through a diversity of forms and approaches (recognition of customary tenure, community mapping, redistribution, decentralisation, co-management, regulation of rental markets, strengthening the negotiating position of the poor) rather than sole focus on freehold title (Quan & Dyer, 2008; K Deininger & Feder, 2009; St. Martin, 2009) (*medium evidence, high agreement*). Land policy can be climate-proofed and integrated with national policies such as NAPAs (Quan and Dyer 2008). Land administration systems have a vital role in providing land tenure security, especially for the poor, especially when linked to an expanded range of information relevant to mitigation and adaptation (Quan and Dyer 2008; van der Molen and Mitchell 2016). Challenges to such a role include outdated and overlapping national land and forest tenure laws, which often fail to recognise community property rights and corruption in land administration (Monterrosso et al. 2017), as well as lack of political will and the costs of improving land administration programs (Deininger and Feder 2009).

7.6.6. Institutional dimensions of adaptive governance

Institutional systems that demonstrate the institutional dimensions, or indicators, in Table 7.8 enhance the adaptive capacity of the socio-ecological system to a greater degree than institutional systems that do not demonstrate these dimensions (*high confidence*) (Gupta et al. 2010; Mollenkamp and Kasten 2009). Governance processes and policy instruments supporting these characteristics are context specific (*medium evidence, high agreement*) (Biermann 2007; Gunderson and Holling 2001; Hurlbert and Gupta 2017; Bastos Lima et al. 2017a; Gupta et al. 2013a; Mollenkamp and Kasten 2009; Nelson et al. 2010; Olsson et al. 2006; Ostrom 2011; Pahl-Wostl 2009; Verweij et al. 2006; Weick and Sutcliffe 2001).

Consideration of these indicators is important when implementing climate change mitigation instruments. For example, a 'Variety,' redundancy, or duplication of climate mitigation policy instruments is an important consideration for meeting Paris Commitments. Given 58% of EU emissions are outside of the EU Emissions Trading system, implementation of a 'redundant' carbon tax may add co-benefits (Baranzini et al. 2017). Further, a carbon tax phased in over time through a schedule of increases allows for 'Learning.' The tax revenues could be earmarked to finance additional climate change mitigation and or redistributed to achieve the indicator of 'Fair Governance - Equity'. It is recommended that carbon pricing measures be implemented using information sharing and communication devices to enable public acceptance, openness, provide measurement and accountability (Baranzini et al. 2017; Siegmeier et al. 2018).

The impact of flood on a socio-ecoloigical system is reduced with the governance indicator of both leadership and resources (Emerson and Gerlak 2014). 'Leadership' pertains to a broad set of stakeholders that facilitate adaptation (and might include scientists and leaders in NGOs) and those that respond to flood in an open, inclusive, and fair manner identifying the most pressing issues and actions needed. Resources are required to support this leadership and includes upfront financial investement in human capital, technology, and infrastrucuture (Emerson and Gerlak 2014).

Policy instruments advancing the indicator of 'Participation' in community forest management include favourable loans, tax measures, and financial support to catalyse entrepreneurial leadership, and build in rewards for supportive and innovative elites to reduce elite capture and ensure more inclusive participation (Duguma et al. 2018) (see 7.6.4).

Table 7.8 Institutional Dimensions or Indicators of Adaptive Governance
This table represents a summation of characteristics, evaluative criteria, elements, indicators or institutional design principles that advance adaptive governance

Indicators/Institutional Dimensions	Description	References
Variety	Room for a variety of problem frames reflecting different opinions and problem definitions	(Biermann 2007; Gunderson and Holling 2001; Hurlbert and Gupta 2017; Bastos Lima et al. 2017a; Gupta, J., van der Grijp, N., Kuik 2013; Mollenkamp and Kasten 2009; Nelson et al. 2010; Olsson et al. 2006; Ostrom 2011; Pahl-Wostl 2009; Verweij et al. 2006; Weick and Sutcliffe 2001)
	Participation. Involving different actors at different levels, sectors, and dimensions	
	Availability of a wide range or diversity of policy options to address a particular problem	
	Redundancy or duplication of measures, back-up systems	
Learning	Trust	
	Single loop learning or ability to improve routines based on past experience	
	Double loop learning or changed underlying assumptions of institutional patterns	
	Discussion of doubts (openness to uncertainties, monitoring and evaluation of policy experiences)	
	Institutional memory (monitoring and evaluation of policy experiences over time)	
Room for autonomous change	Continuous access to information (data institutional memory and early warning systems)	
	Acting according to plan (especially in relation to disasters)	
	Capacity to improvise (in relation to self-organisation and fostering social capital)	
Leadership	Visionary (Long term and reformist)	
	Entrepreneurial which leads by example	
	Collaborative	
Resources	Authority resources or legitimate forms of power	
	Human resources of expertise, knowledge and labour	
	Financial resources	
Fair governance	Legitimacy or public support	
	Equity in relation to institutional fair rules	
	Responsiveness to society	
	Accountability in relation to procedures	

7.6.7. Inclusive Governance for Sustainable Development

Many sustainable development efforts fail because of lack of attention to societal issues including inequality, discrimination, social exclusion and marginalisation (see Cross-Chapter Box 11: Gender in this chapter) (Arts 2017a). However, the human rights based approach of the 2030 Agenda and Sustainable Development Goals commits to leaving no one behind (Arts 2017b). Inclusive governance focuses attention in issues of equity and the human rights based approach for development as it includes social, ecological and relational components used for assessing access to, as well as the allocations of rights, responsibilities and risks with respect to social and ecological resources (medium agreement) (Gupta and Pouw 2017).

Governance processes that are inclusive of all people in decision making and management of land, are better able to make decisions addressing trade offs of sustainable development (Gupta et al. 2015) and achieve SDGs focusing on social and ecological inclusiveness (Gupta and Vegelin 2016). Citizen engagement is important in enhancing natural resource service delivery by citizen inclusion in management and governance decisions (see 7.5.5). In governing natural resources, focus is now not only on rights of citizens in relation to natural resources, but also on citizen obligations, responsibilities (Karar and Jacobs-Mata 2016; Chaney and Fevre 2001), feedback and learning processes (Tàbara et al. 2010). In this respect, citizen engagement is also an imperative particularly for analysing and addressing aggregated informal coping strategies of local residents in developing countries, which are important drivers of natural resource depletions (but often overlooked in a conventional policy development processes in natural resource management) (Ehara et al. 2018).

Inclusive adaptive governance makes important contributions to the management of risk. Inclusive governance concerning risk integrates people's knowledge and values by involving them in decision making processes where they are able to contribute their respective knowledge and values to make effective, efficient, fair, and morally acceptable decisions (Renn and Schweizer 2009). Representation in decision making would include major actors - government, economic sectors, the scientific community and representatives of civil society (Renn and Schweizer 2009). Inclusive governance focuses attention on the well being and meaningful participation in decision making of the poorest (in income), vulnerable (in terms of age, gender, and location), and the most marginalised and is inclusive of all knowledges (Gupta et al. 2015).

7.7. Key uncertainties and knowledge gaps

Uncertainties in land, society and climate change processes are outlined in 7.2 and Chapter 1. This chapter has reviewed literature on risks arising from GHG Fluxes, climate change, land degradation, desertification and food security, policy instruments responding to these risks, as well as decision making and adaptive climate and land governance, in the face of uncertainty.

More research is required to understand the complex interconnections of land, climate, water, society, ecosystem services and food, including:

- New models that allow incorporation of considerations of justice, inequality and human agency in socio-environmental systems;
- Understanding how policy instruments and response options interact and augment or reduce risks in relation to acute shocks and slow-onset climate events;
- Understanding how response options, policy, and instrument portfolios can reduce or augment the cascading impacts of land, climate and food security and ecosystem service interactions through different domains such as health, livelihoods, and infrastructure, especially in relation to non-linear and tipping-point changes in natural and human systems.
- Consideration of trade-offs and synergies in climate, land, water, ecosystem services and food policies;
- The impacts of increasing use of land due to climate mitigation measures such as BECCS, carbon centric afforestation/REDD+ and their impacts on human conflict, livelihoods and displacement;
- Understanding how different land tenure systems, both formal and informal, and the land policies and administration systems that support them, can constrain or facilitate climate adaptation and mitigation: and on how forms of climate action can enhance or undermine land tenure security and land justice.
- Expanding understanding of barriers to implementation of land-based climate policies at all levels from the local to the global, including methods for monitoring and documenting corruption, misappropriation and elite capture in climate action;
- Identifying characteristics and attributes signalling impending socio-ecological tipping points and collapse;
- Understanding the full cost of climate change in the context of disagreement on accounting for climate change interactions and their impact on society, as well as issues of valuation, and attribution uncertainties across generations;
- New models and Earth observation to understand complex interactions described in this section.

- The impacts, monitoring, effectiveness, and appropriate seclection of certification and standards for sustainability (see 7.4.6.3) (ISEAL Alliance; Stattman et al. 2018) and the effectiveness of its implementation through the landscape governance approach (Pacheco et al. 2016) (see 7.6.3).

Actions to mitigate climate change are rarely evaluated in relation to impact on adaptation, SDGs, and trade-offs with food security. For instance, there is a gap in knowledge in the optimal carbon pricing or emission trading scheme together with monitoring, reporting and verification system for agricultural emissions that will advance GHG reductions, food security, and sustainable land management. Better understanding is needed of the triggers and leveraging actions that build sustainable development and sustainable land management, as well as the effective organisation of the science and society interaction jointly shaping policies in the future. What societal interaction in the future will form inclusive and equitable governance processes and achieve inclusive just governance institutions including. Land tenure?

As there is a significant gap in NDCs and achieving commitments to keep global warming well below 2°C (7.4.4.1), governments might consider evaluating national, regional, and local gaps in knowledge surrounding response options, policy instruments portfolios, and sustainable land management supporting the achievement of NDCs in the face of land and climate change.

Frequently Asked Questions

FAQ 7.1 How can indigenous knowledge and local knowledge inform land-based mitigation and adaptation options?

Indigenous knowledge (IK) refers to the understandings, skills and philosophies developed by societies with long histories of interaction with their natural surroundings. Local knowledge (LK) refers to the understandings and skills developed by individuals and populations, specific to the place where they live. These forms of knowledge are often highly context-specific and embedded in local institutions, providing biological and ecosystem knowledge with landscape information. This means they can contribute to effective land management, predictions of natural disasters and identification of longer-term climate changes, for example, and IK can be particularly useful where formal data collection on environmental conditions may be sparse. IK and LK are often dynamic, with knowledge holders often experimenting with mixes of local and scientific approaches. Water management, soil fertility practices, grazing systems, restoration and sustainable harvesting of forests, and ecosystem based-adaptation are many of the land management practices often informed by IK and LK. LK can also be used as an entry point for climate adaptation by balancing past experiences with new ways to cope. To be effective, initiatives need to take into account the differences in power between the holders of different types of knowledge. For example, including indigenous and/or local people in programmes related to environmental conservation, formal education, land management planning and security tenure rights is key to facilitate climate change adaptation. Formal education is necessary to enhance adaptive capacity of IK and LK since some researchers have suggested these knowledge systems may become less relevant in certain areas where the rate of environmental change is rapid and the transmission of IK and LK between generations is becoming weaker.

FAQ 7.2 What are the main barriers to and opportunities for land-based responses to climate change?

Land-based responses to climate change can be mitigation (e.g., renewable energy, vegetation or crops for biofuels, afforestation) or adaptation (e.g., change in cropping pattern, less water intensive crops in response to moisture stress), or adaptation with mitigation co-benefits (e.g., dietary shifts, new uses for invasive tree-species, siting solar farms on highly degraded land). Productive land is an increasingly scarce resource under climate change. In the absence of adequate deep mitigation in the less land intensive energy sector, competition for land and water for mitigation and for other sectors such as food security, ecosystem services and biodiversity conservation could become a source of conflict and a barrier to land-based responses.

Barriers to land-based mitigation include opposition due to real and perceived trade-offs between land for mitigation and food security and ecosystem services. These can arise due to absence of or uncertain land and water rights. Significant upscaling of mitigation requires dedicated (normally land-based) sources in addition to use of wastes and residues. This requires high land use intensity compared to other mitigation options that in turn place greater demands on governance. A key governance mechanism that has emerged in response to such concerns, especially during the past decade are standards and certification systems that include food security and land and water rights in addition to general criteria or indicators related to sustainable use of land and biomass with an emphasis on participatory approaches. Other governance responses include linking land based mitigation (e.g., forestry) to secure tenure and support for local livelihoods. A barrier to land-based mitigation is our choice of development pathway. Our window of opportunity/ whether or not we face barriers or opportunities to land based mitigation depends on socio-economic decisions or pathways. If we have high population growth and resource intensive consumption (i.e., SSP3) we will have more barriers. High population and low land use regulation results in less available space for land based mitigation. But if we have the opposite trends (SSP1) we can have more opportunities.

Other barriers can arise when in the short term adaptation to a climate stress (eg increased dependence on ground-water during droughts) can become unsustainable in the longer term and become a maladaptation. Policies and approaches that lead to land management that synergises multiple ecosystem services and reduce trade-offs could find greater acceptance and enjoy more success.

Opportunities to obtain benefits or synergies from land-based mitigation and adaptation arise especially from their relation to the land availability and the demand for such measures in rural areas that may otherwise lack incentives for investment in infrastructure, livelihoods and institutional capacity. After decades of urbanisation around the world facilitated by significant investment in urban infrastructure and centralised energy and agricultural systems, rural areas have been somewhat neglected even as farmers in these areas provide critical food and materials needed for urban areas. As land and biomass becomes more valuable, there will be benefits for farmers, forest owners and associated service providers as they diversify away from feed and feed into economic activities supporting bioenergy, value-added products, preservation of biodiversity and carbon sequestration (storage).

A related opportunity for benefits is the potentially positive transformation in rural and peri-urban landscapes that could be facilitated by investments that prioritise more effective management of ecosystem services and conservation of water, energy, nutrients and other resources that have been priced too low in relation to their environmental or ecological value. Multifunctional landscapes supplying food, feed, fiber and fuel to both local and urban communities in combination with reduced waste and healthier diets could restore the role of rural producers as stewards of resources rather than providing food at the lowest possible price. Some of these landscape transformations will function as both mitigation and adaptation responses by increasing resilience even as they provide value-added bio-based products.

Governments can introduce a variety of regulations and economic instruments (taxes, incentives) to encourage citizens, communities and societies to adopt sustainable land management practices with with further benefits in addition to mitigation. Windows of opportunity for redesigning and implementing mitigation and adaptation can arise in the aftermath of a major disaster or extreme climate event. They can also arise when collective action and citizen science motivate voluntary shifts in lifestyles supported by supportive top-down policies.

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Supplementary Material

Additional material on Section 7.2.2 in separate file.

Additional material from Section 7.2.4:

Table 7.1 Appendix

Land-Climate-Society interaction Hazard	Exposure	Vulnerability	Risk	Policy Response (Indicative)	References
Inefficient carbon capture and storage	Global	Developing countries	• Disincetivising low carbon pathways/renewables • Loss of water resources, biodiversity and ecosystem services • Dangerous climate change ie SSP2 and SSP3 pathways	• Certification • Transdisciplinary research on feasibility and pilot projects	(Smith et al. 2016; Fuss et al. 2014; Torvanger 2019b)
Increasing incidences of wildfires at the wildland-urban interface	USA, Canada, Australia	Peri-urban communities next to forests	• Loss of life and property • •	• Willingness to pay for prescribed fire • Local early warning and communication • Wildlife frequency and risk mapping • • •	(Abatzoglou and Williams 2016; Gan et al. 2014; Kaval et al. 2007; Mozumder et al. 2009; Brenkert–Smith et al. 2006)(Radeloff et al. 2018)

Land-Climate-Society interaction Hazard	Exposure	Vulnerability	Risk	Policy Response (Indicative)	References
Use of land for renewable energy	India, China, semi-arid regions	Pastoralists Farmers Biodiversity	• Loss of biodiversity and ecosystem services • •	• See 7.5.6 • •	See 7.5.6
Urban air pollution **from surrounding land-use**	Urban centres existing and emerging in developing countries	Marginalized communities, pedestrians, commuters, street vendors, children	• Health risk • allergic respiratory diseases	• Air pollution regulation • Fuel conversion to clean energy • Incentives to reduce crop stubble burning	(Sharma et al., 2013, D'Amato et al., 2010)
Severe weather hazards for cultural heritage (sensitive historic material)	Regions with increase precipitation Increase in the freeze-thaw cycle in northern regions Extreme heat and drought in dry area Landslide and	Buildings and sites in areas with increasing intensities of rain and humidity	• Loss of culture and identity	• Restoration and protection measures incorporated in regulations and management plan	(Sesana et al, 2018, Sabbioni et al., 2008)

Land-Climate-Society interaction	Hazard	Exposure	Vulnerability	Risk	Policy Response (Indicative)	References
		groundwater flooding				

Supplementary Material

Supplementary information to Section 7.2

The burning embers diagrams (Figure 7.1, 7.2 and 7.3) outline risks associated with climate change as a function of global warming, socio-economic development and mitigation choices. Diagrams indicate transitions between undetectable, moderate, high, and very high risks to humans and ecosystems. The method is based on a literature review of estimated impacts at different global mean surface temperature levels (O'Neill et al. 2017) on different components of desertification, land degradation and food security, including emerging literature on Shared Socio-economic Pathways (SSPs) as well as literature from IPCC AR5 and SR15.

Most studies focus on changes in hazards as a function of climate change (e.g. as represented by RCP scenarios or other climate change scenarios) or climate change superimposed on present-day exposure. Only a limited number of studies focus on changes in risk as a function of both RCPs and SSPs (climate and socio-economic change and adaptation decisions). This was addressed by splitting the embers into different figures. Figure 7.1 focuses on the impact of climate change on risk, under present-day exposure and vulnerability. Figure 7.2 examines the relationship between climate change and risks under two SSPs (SSP1 and SSP3). Figure 7.3 depicts risks to humans and ecosystems as a function of the land area employed for mitigation through bioenergy plantations.

Further, a formal expert elicitation protocol, based on the modified-Delphi technique (Mukherjee et al. 2015) and the Sheffield Elicitation Framework (Oakley and O'Hagan 2016; Gosling 2018), was followed to develop threshold judgments on risk transitions. Specifically, experts participated in a multi-round elicitation process, with feedback of group opinion provided after each round: the first two rounds involved independent anonymous threshold judgment, and the final round involved a group consensus discussion (von der Gracht 2012). To strengthen the rigor of developing expert consensus on risk transitions (Hasson and Keeney 2011), the protocol pre-specified the following prior to beginning the elicitation exercise (Grant et al. 2018): the research question, eligibility criteria and strategy to recruit experts, research materials, data collection procedure, and analysis plan. This systematic process of developing expert consensus on threshold judgments for risk transitions can better inform subsequent analytical approaches—an approach that may be further developed for use in future IPCC cycles (Bojke et al. 2010; Sperber et al. 2013). References for the current and past assessments are listed at the end of this document and by the relevant tables.

Table SM7.1: literature considered in the expert judgement of risk transitions for figure 7.1

Reference	Risk	variable (unit)	Direction of impact	climate scenario	Time frame	D/A of current impact	Impact at 1 degree	Impact at 2 degree	Impact at 3 degree	Impact at 4 degree	Impact at 4.5 degree	Adaptation potential	Region (Including Regional Differences)
AVAILABILITY													
Rosenzweig, Cynthia, Joshua Elliott, Delphine Deryng, Alex C. Ruane, Christoph Müller, Almut	Availability Yield	yield	Strong negative effect on yields,	NA		-	See Figure 1. Maize mid to	Maize - 20 to +5 % yeild	Maize about - 20 to +5%	Maize - +15 to minus	Maize is now all	Between 3 and 4 degrees	Use RCPs so could examine yield

Arneth, Kenneth J. Boote, et al. 2014. "Assessing Agricultural Risks of Climate Change in the 21st Century in a Global Gridded Crop Model Intercomparison." Proceedings of the National Academy of Sciences. https://doi.org/10.1073/pnas.1222463110.			especially at higher levels of warming and at lower latitudes,				hight latitude is -10 to +15 % yield change	change	yield change in mid latitude and ALL negative in low latiude	20% yield change in mid latitude. Catastrophic in low latitude with - 10 to - 60 Percent change!	negative in mid latitude	seems to me catstrophic in low latitudes for maize, wheat also significant declines around 4 degrees and same for rice	according to different pathways.
Zscheischler, Jakob, Seth Westra, Bart J.J.M. Van Den Hurk, Sonia I. Seneviratne, Philip J. Ward, Andy Pitman, Amir Aghakouchak, et al. 2018. "Future Climate Risk from Compound Events." Nature Climate Change. https://doi.org/10.1038/s41558-018-0156-3.	Availability (crop failure)	crop yield	" increases the likelihood of such events considerably, and may make events of the rarity of the Russian event foreseeable and to some extent predictable"	Review	2010	-	-	-	-	-	-	-	
IPCC Special Report on Global Warming of 1.5˚C, 2018	Availability (crop yields)	yield	Decrease to yields	NA		-	-	-	-	-	-	Limiting global warming to 1.5°C compared to 2°C would result in a lower global reduction in crop yields	
Medina, Angel, Asya Akbar, Alaa Baazeem, Alicia Rodriguez, and Naresh Magan. 2017. "Climate Change, Food Security and Mycotoxins: Do We Know Enough?" Fungal Biology Reviews. https://doi.org/10.1016/j.fbr.2017.04.002.	Availability (increased loss of crops and livestock; increased pest burden, increased disease burden; higher post-harvest losses due to mycotoxins)	infection of staple food commodities by fungal diseases pre-harvest and by spoilage fungi post-harvest	reduced availability of food	NA		-	-	-	-	-	-		low to moderate

Paterson, R. R.M., and N. Lima. 2011. "Further Mycotoxin Effects from Climate Change." Food Research International. https://doi.org/10.1016/j.foodres.2011.05.038.	Availability (increased loss of crops and livestock; increased pest burden, increased disease burden; higher post-harvest losses due to mycotoxins)	crops after harvest	reduced availability of food	NA	NA	-	-	-	-	-	-		unclear. "Crops introduced to exploit altered climate may be subject to fewer mycotoxin producing fungi (the "Parasites Lost" phenomenon). Increased mycotoxins and UV radiation may cause fungi to mutate on crops and produce different mycotoxins"
Magan, N., A. Medina, and D. Aldred. 2011. "Possible Climate-Change Effects on Mycotoxin Contamination of Food Crops Pre- and Postharvest." Plant Pathology. https://doi.org/10.1111/j.1365-3059.2010.02412.x.	Availability (increased loss of crops and livestock; increased pest burden, increased disease burden; higher post-harvest losses due to mycotoxins)	crops after harvest	reduced availability of food	NA	NA	-	-	-	-	-	-	from high risk to permanent between 3 and 5 degrees	low to moderate
Rivera-Ferre, M. G., M. Di Masso, I. Vara, M. Cuellar, A. Calle, M. Mailhos, F. López-i-Gelats, G. Bhatta, and D. Gallar. 2016. "Local Agriculture Traditional Knowledge to Ensure Food Availability in a Changing Climate: Revisiting Water Management Practices in the Indo-Gangetic Plains." Agroecology and Sustainable Food Systems. https://doi.org/10.1080/21683565.2016.1215368.	Availability (increased loss of crops and livestock; increased pest burden, increased disease burden; higher post-harvest losses due to mycotoxins)	crop yield	reduced availability of food	NA	NA	-	-	-	-	-	-	-	Local\ntraditional knowledge in agriculture (LTKA) is proposed in this article\nas valid knowledge to ensure food availability under climate change,\n given its long experience in dealing with climate variability

Zimmermann, Andrea, Heidi Webber, Gang Zhao, Frank Ewert, Johannes Kros, Joost Wolf, Wolfgang Britz, and Wim de Vries. 2017. "Climate Change Impacts on Crop Yields, Land Use and Environment in Response to Crop Sowing Dates and Thermal Time Requirements." Agricultural Systems. https://doi.org/10.1016/j.agsy.2017.07.007.	Availability (increased yields if management assumptions hold, thermal management)	crop yields in Europe	increased yields	three SRES climate change scenarios to 2050	three SRES climate change scenarios to 2050	-	-	-	-	-	-		high
Faye, Babacar, Heidi Webber, Jesse B. Naab, Dilys S. MacCarthy, Myriam Adam, Frank Ewert, John P.A. Lamers, et al. 2018. "Impacts of 1.5 versus 2.0 °c on Cereal Yields in the West African Sudan Savanna." Environmental Research Letters. https://doi.org/10.1088/1748-9326/aaab40.	Availability (modeled crop yield)	crop yield	negative	NA		-	-	-	-	-	-	between 1 and 2 with success of intensification the key factor making the difference between whether risk remains moderate or red to purple	low to moderate ("despite the larger losses, yields were always two to three times higher with intensification, irrespective of warming scenario")
Tesfaye, Kindie, P. H. Zaidi, Sika Gbegbelegbe, Christian Boeber, Dil Bahadur Rahut, Fite Getaneh, K. Seetharam, Olaf Erenstein, and Clare Stirling. 2017. "Climate Change Impacts and Potential Benefits of Heat-Tolerant Maize in South Asia." Theoretical and Applied Climatology. https://doi.org/10.1007/s00704-016-1931-6.	Availability (modeled crop yield)	crop yield	negative	NA		-	-	"at regional scale, they found maize yields declines in 2050 of up to 12% to 14% in rainfed and irrigated maize"	-	-	-	between 1.0 and 1.5	low
Scheelbeek, Pauline F. D., Frances A. Bird, Hanna L. Tuomisto, Rosemary Green, Francesca B. Harris, Edward J. M. Joy, Zaid Chalabi, Elizabeth Allen, Andy Haines, and Alan D. Dangour. 2018. "Effect of Environmental Changes on Vegetable and Legume Yields and Nutritional Quality." Proceedings of the National Academy of Sciences. https://doi.org/10.1073/pnas.1800442115.	Availability (modeled crop yield)	crop yield	negative	NA		-	-	-	-	mean yield declines of fruits -31.5%	-		
Rippke, Ulrike, Julian Ramirez-Villegas, Andy	Availability	crop yield	negative	NA	to end	-	-	"30-60% of	-	-	-	between 2.6	low

Jarvis, Sonja J. Vermeulen, Louis Parker, Flora Mer, Bernd Diekkrüger, Andrew J. Challinor, and Mark Howden. 2016. "Timescales of Transformational Climate Change Adaptation in Sub-Saharan African Agriculture." Nature Climate Change. https://doi.org/10.1038/nclimate2947.	(modeled crop yield)				of 21st century			common bean growing area and 20-40% of banana growing areas in Afria will lose viability in 2078-2098 with a global temperature increase of 2.6 and 4.0"				and 4.0 (""30-60% of common bean growing area and 20-40% of banana growing areas in Afria will lose viability in 2078-2098 with a global temperature increase of 2.6 and 4.0")	
Bisbis, M. B., N. Gruda, and M. Blanke, 2018: Potential impacts of climate change on vegetable production and product quality - A review. J. Clean. Prod., 170, 1602–1620, doi:10.1016/j.jclepro.2017.09.224.	Availability (modeled fruit crop yield), and utilization (reduced quality, more spoilage, reduced nutrition)	crop yield	negative	NA		-	-	-	-	-	-	between 1.0 and 1.5	medium
Tebaldi, Claudia, and David Lobell. 2018. "Estimated Impacts of Emission Reductions on Wheat and Maize Crops." Climatic Change. https://doi.org/10.1007/s10584-015-1537-5.	Availability (models relation between climate variables, CO_2 concentrations, and yields)	crop yield	negative	RCP4.5 and RCP8.5	short (2021–2040), medium (2041–2060) and long (2061–2080) time horizons	-	-	"critical or "lethal" heat extreme	-	-	-	modeling results in RCP8.5 (tripling of lethal heat extremes), modeling results in RCP4.5 (doubling of lethal heat extremes) towards end of 21st century	low
Schleussner, Carl Friedrich, Delphine Deryng, Christoph Müller, Joshua Elliott, Fahad Saeed, Christian Folberth, Wenfeng Liu, et al. 2018. "Crop Productivity Changes in 1.5 °c and 2 °c Worlds under Climate Sensitivity Uncertainty."	Availability (reduced yields and soil fertility and increased land degradation for	yield	negative for half a degree additional warming (1.5 to 2)	HAPPI		-	-	"half a degree warming will also lead to more extreme low yields, in	-	-	-		

Environmental Research Letters. https://doi.org/10.1088/1748-9326/aab63b.	some regions and crops)							particular over tropical regions"					
Ovalle-Rivera, Oriana, Peter Läderach, Christian Bunn, Michael Obersteiner, and Götz Schroth. 2015. "Projected Shifts in Coffea Arabica Suitability among Major Global Producing Regions Due to Climate Change." PLoS ONE. https://doi.org/10.1371/journal.pone.0124155.	Availability (reduced yields and soil fertility and increased land degradation for some regions and crops)	yield	Decrease in coffee yields	NA		-	-	-	-	-	-		
Bunn, Christian, Peter Läderach, Oriana Ovalle Rivera, and Dieter Kirschke. 2015. "A Bitter Cup: Climate Change Profile of Global Production of Arabica and Robusta Coffee." Climatic Change. https://doi.org/10.1007/s10584-014-1306-x.	Availability (reduced yields and soil fertility and increased land degradation for some regions and crops)	yield	Decrease in coffee yields by 50%	NA		-	-	-	-	-	-		
Roberts, Michael J., and Wolfram Schlenker. 2013. "Identifying Supply and Demand Elasticities of Agricultural Commodities: Implications for the US Ethanol Mandate." American Economic Review. https://doi.org/10.1257/aer.103.6.2265. 2009	Availability (reduced yields and soil fertility and increased land degradation for some regions and crops)	yield	productivity of major crops will decline as a result of climate change, particularly from increasing warming	NA		-	-	-	-	-	-		
Peng, S., J. Huang, J. E. Sheehy, R. C. Laza, R. M. Visperas, X. Zhong, G. S. Centeno, G. S. Khush, and K. G. Cassman. 2004. "Rice Yields Decline with Higher Night Temperature from Global Warming." Proceedings of the National Academy of Sciences. https://doi.org/10.1073/pnas.0403720101.	Availability (reduced yields and soil fertility and increased land degradation for some regions and crops)	grain yields	Grain yield of rice declined 10% for each 1°C increase in night-time temperature during the dry season	NA		-	-10%	-20%	-30%	-40%	-50%		
Asseng, S., F. Ewert, P. Martre, R. P. Rötter, D. B. Lobell, D. Cammarano, B. A. Kimball, et al. 2015. "Rising Temperatures Reduce Global Wheat Production." Nature Climate Change. https://doi.org/10.1038/nclimate2470. et al., 2015	Availability (reduced yields and soil fertility and increased land degradation for some	soy bean & maize yields	while maize and soy bean yields are expected to decline by 6% for each day	NA		-	-6%/day above 30°C	-12%/day above 30°C	-18%/day above 30°C	-24%/day above 30°C	-30%/day above 30°C		

	regions and crops)		above 30°C.										
Asseng, Senthold, Davide Cammarano, Bruno Basso, Uran Chung, Phillip D. Alderman, Kai Sonder, Matthew Reynolds, and David B. Lobell. 2017. “Hot Spots of Wheat Yield Decline with Rising Temperatures.” Global Change Biology. https://doi.org/10.1111/gcb.13530.	Availability (reduced yields and soil fertility and increased land degradation for some regions and crops)	wheat yields	wheat yields are expected to decline by 6% for each 1°C increase;	NA		warming is already slowing yield gains at a majority of wheat-growing locations.	-0.06	-0.12	-0.18	-0.24	-0.3	tiping point above 28 degrees C, no yield	medium
Porter, John R., Liyong Xie, Andrew J Challinor, Kevern Cochrane, S. Mark Howden, Muhammad Mohsin Iqbal, David B. Lobell, and Maria Isabel Travasso. 2014. “Food Security and Food Production Systems.” In Climate Change 2014: Impacts, Adaptation, and Vulnerability. Part A: Global and Sectoral Aspects. Contribution of Working Group II to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change, 485–533. https://doi.org/10.1111/j.1728-4457.2009.00312.x.	Availability (reduced yields and soil fertility and increased land degradation for some regions and crops)	crop yields all crops	If global temperature increases beyond 3°C it will have negative yield impacts on all crops	NA		-	-	-	negative yield impact	-	-		
Schleussner, Carl Friedrich, Tabea K. Lissner, Erich M. Fischer, Jan Wohland, Mahé Perrette, Antonius Golly, Joeri Rogelj, et al. 2016. “Differential Climate Impacts for Policy-Relevant Limits to Global Warming: The Case of 1.5 °c and 2 °c.” Earth System Dynamics. https://doi.org/10.5194/esd-7-327-2016.	Availability (reduced yields and soil fertility and increased land degradation for some regions and crops)	competition for land	increasing competition for land from the expansion of bioenergy	NA		-	-	-	-	-	-		
Fischer, Günther, Mahendra Shah, Francesco N. Tubiello, and Harrij Van Velhuizen. 2005. “Socio-Economic and Climate Change Impacts on Agriculture: An Integrated Assessment, 1990-2080.” In Philosophical Transactions of the Royal Society B: Biological Sciences. https://doi.org/10.1098/rstb.2005.1744.	Availability (reduced yields and soil fertility and increased land degradation for some regions and crops)		Decrease in yields	NA		-	10%	10-20%	10-20%	10-20%	-		on-farm and via market mechanisms
Smith, Pete, R. Stuart Haszeldine, and Stephen M. Smith. 2016. “Preliminary Assessment of the Potential for, and Limitations to, Terrestrial Negative	Availability (reduced yields and soil fertility and	soil	reduced yields	NA	NA	-	-	-	-	-	-	-	moderate

Emission Technologies in the UK." Environmental Science: Processes and Impacts. https://doi.org/10.1039/c6em00386a.	increased land degradation for some regions and crops)												
Challinor, A. J., J. Watson, D. B. Lobell, S. M. Howden, D. R. Smith, and N. Chhetri. 2014. "A Meta-Analysis of Crop Yield under Climate Change and Adaptation." Nature Climate Change. https://doi.org/10.1038/nclimate2153.	Availability (reduced yields and soil fertility and increased land degradation for some regions and crops)	crop yield	reduced yields	NA	2050 to end of century	-	-	-	-	-	-	likely between 1.5 and 2.0	low to moderate
FAO 2018a	Availability (reduced yields and soil fertility and increased land degradation for some regions and crops)	crop yield	reduced yields	NA		-	-	-	-	-	-	likely between 1.0 and 1.5	low to moderate
Roberts, Michael J., and Wolfram Schlenker. 2013. "Identifying Supply and Demand Elasticities of Agricultural Commodities: Implications for the US Ethanol Mandate." American Economic Review. https://doi.org/10.1257/aer.103.6.2265. 2009	Availability (reduced yields and soil fertility and increased land degradation for some regions and crops)(3 crops)		Decrease in yields	NA		-	30-46%	30-46%	63-80%	63-80%	-		
Richard A Betts, Lorenzo Alfieri, John Caesar, Luc Feyen, Laila Gohar, Aristeidis Koutroulis, et al. 2018. "Subject Areas : Author for Correspondence : Changes in Climate Extremes , Fresh Water Availability and Vulnerability to Food Insecurity Projected at 1 . 5 ° C and 2 ° C Global Warming with a Higher-Resolution Global Climate Model." et al, 2018	Availability (reduced yields and soil fertility and increased land degradation for some regions and crops)(food crops)	yield	decreae	NA		-	-	-	-	-	-		
Tigchelaar, M, D Battisti, R.L Naylor, and D.K Ray. 2018. "Probability of Globally Synchronized Maize Production Shocks." Proceedings of the National Academy of	Availability (reduced yields and soil fertility and increase		Decrease in yields	NA		-	-	7-10%	-	87%	-		

Sciences 115 (26): 6644–49.	d land degradation for some regions and crops)(Maize)												
Leng, Guoyong, and Jim Hall. 2019. “Crop Yield Sensitivity of Global Major Agricultural Countries to Droughts and the Projected Changes in the Future.” Science of the Total Environment. https://doi.org/10.1016/j.scitotenv.2018.10.434.	Availability (reduced yields and soil fertility and increased land degradation for some regions and crops)(six crops)		Declining yield (but varies between crops and regions)	NA		-	-	-	-	-	-		Study doesn't consider adaptations
Bocchiola, D., L. Brunetti, A. Soncini, F. Polinelli, and M. Gianinetto. 2019. “Impact of Climate Change on Agricultural Productivity and Food Security in the Himalayas: A Case Study in Nepal.” Agricultural Systems. https://doi.org/10.1016/j.agsy.2019.01.008.	Availability (reduced yields and soil fertility and increased land degradation for some regions and crops)(wheat, rice, maize)		Declining	NA		-	-	-	-	-	-		Increasing altitude - increases yield for maize and rice slightly
Rozenzweig et al. 2017	Availability (simulated wheat and maize yield changes)	crop yield	negative	AgMIP coordinated global and regional assessment (CGRA)		-	-	-	-	-	-	between 1.5 and 2.0	low
Parkes et al. 2017	Availability (simulated wheat and maize yield changes)	crop yield	negative	NA		-	-	-	-	-	-	between 1.0 and 1.5	low
Lombardozzi, Danica L., Nicholas G. Smith, Susan J. Cheng, Jeffrey S. Dukes, Thomas D. Sharkey, Alistair Rogers, Rosie Fisher, and Gordon B. Bonan. 2018. “Triose Phosphate Limitation in Photosynthesis Models Reduces Leaf Photosynthesis and Global Terrestrial Carbon Storage.” Environmental Research Letters. https://doi.org/10.1088/1748-9326/aacf68.	Availability (Yield)	yield	positive effect of CO2 on future crop yields muted by negative impacts of climate	CESM/CLM4.5 under RCP8.5	2006-2100	-	-	-	-	Corn: -10 to +20% Wheat +40 to +100 %; Soy -10 to +5 %; Rice +10 to +50%	-		
Chen, Y. et al. (2018) ESD Impacts of climate change and climate	Availability (Yield)	yield	decrease in organic	NA		-	-	-	-	-	-		

extremes on major crops productivity in China at a global warming of 1.5 and 2.0C			matter in soil, soil erosion										
Leng, G. (2018) SOTE Keeping global warming within 1.5C reduces future risk of yield loss in the United States: A probabilistic modeling approach	Availability (Yield)	yield		NA		-	-	-	-	-	-		
Byers, Edward, Matthew Gidden, David Leclère, Juraj Balkovic, Peter Burek, Kristie Ebi, Peter Greve, et al. 2018. "Global Exposure and Vulnerability to Multi-Sector Development and Climate Change Hotspots." Environmental Research Letters 13 (5): 055012. https://doi.org/10.1088/1748-9326/aabf45.	Availability (Yield)	yield		NA		-	-	-	-	-	-		
Xie, Wei, Wei Xiong, Jie Pan, Tariq Ali, Qi Cui, Dabo Guan, Jing Meng, Nathaniel D. Mueller, Erda Lin, and Steven J. Davis. 2018. "Decreases in Global Beer Supply Due to Extreme Drought and Heat." Nature Plants. https://doi.org/10.1038/s41477-018-0263-1.	Availability barley yields (beer)	yield	Decrease in barley yield, consumption (and hence global beer supply)	NA		-	-	-3%	-10%	-17%	-		
Leng, Guoyong, and Jim Hall. 2019. "Crop Yield Sensitivity of Global Major Agricultural Countries to Droughts and the Projected Changes in the Future." Science of the Total Environment. https://doi.org/10.1016/j.scitotenv.2018.10.434.	Availability Corn Yields	yield	Decrease to yields.	NA		2.5% decrease of corn yield for the historical period, which is reduced to 1.8% if accounting for the effects of corn growing pattern changes	Negative corn yield response to warmer growing season, largest yield reduction up to 20% by 1° increase of temperature	majority of impacts will be driven by trends in temperature rather than precipitation	-	-	-	Negative corn yield response to warmer growing season	Corn yield is predicted to decrease by 20~40% by 2050s
Leng, Guoyong. 2018. "Keeping Global Warming within 1.5 °C Reduces Future Risk of Yield Loss in the United States: A Probabilistic Modeling Approach." Science of the Total Environment. https://doi.org/10.1016/j.scitotenv.2018.06.344.	Availability crop yields	yield	Decrease in yields	NA		-	-	-	-	-	-		
Su, B. et al. (2018) Drought losses in China might double between the 1.5C and 2.0C warming, PNAS	Availability crop yields	yield	Decrease in yields	NA		-	-	-	-	-	-		
Zhao, Chuang, Bing Liu, Shilong Piao, Xuhui Wang, David B. Lobell, Yao Huang, Mengtian	Availability maize yields	yield, production/ per hectare	Decrease in yield	NA		-	-	-	-	-	-		

Huang, et al. 2017. “Temperature Increase Reduces Global Yields of Major Crops in Four Independent Estimates.” Proceedings of the National Academy of Sciences. https://doi.org/10.1073/pnas.1701762114.													
Brisson, Nadine, Philippe Gate, David Gouache, Gilles Charmet, François Xavier Oury, and Frédéric Huard. 2010. “Why Are Wheat Yields Stagnating in Europe? A Comprehensive Data Analysis for France.” Field Crops Research. https://doi.org/10.1016/j.fcr.2010.07.012.	Availability Yield	yield	yield losses/plateauing	NA		-	-	-	-	-	-		
Lin, M., and P. Huybers. 2012. “Reckoning Wheat Yield Trends.” Environmental Research Letters. https://doi.org/10.1088/1748-9326/7/2/024016.	Availability Yield	yield	yield losses/plateauing	NA		-	-	-	-	-	-		
Grassini, Patricio, Kent M. Eskridge, and Kenneth G. Cassman. 2013. “Distinguishing between Yield Advances and Yield Plateaus in Historical Crop Production Trends.” Nature Communications. https://doi.org/10.1038/ncomms3918.	Availability Yield	yield	yield losses/plateauing	NA		-	-	-	-	-	-		
Myers, S.S.; M. R. Smith, S. Guth, C. D. Golden, B. Vaitla, N. D. Mueller, A. D. Dangour, and P. Huybers, 2017: Climate Change and Global Food Systems: Potential Impacts on Food Security and Undernutrition. Annu. Rev. Public Health, 38, 259–277, doi:10.1146/annurev-publhealth-031816-044356. http://www.annualreviews.org/doi/10.1146/annurev-publhealth-031816-044356.	Availability yield declines	yield		NA		-	-	-	-	-	-		adaptation could lead to crop yields that are 7-15% higher. Gains will be highest in temperate areas but will be unlikely to help tropical maize and wheat production
Hasegawa, Tomoko, Shinichiro Fujimori, Petr Havlík, Hugo Valin, Benjamin Leon Bodirsky, Jonathan C. Doelman, Thomas Fellmann, et al. 2018. “Risk of Increased Food Insecurity under Stringent Global Climate Change Mitigation Policy.” Nature Climate Change 8 (8): 699–703. https://doi.org/10.1038/s41558-018-0230-x.	Mitigation policy combined with climate effect on yields	available land		NA		-	-	-	-	-	-		
ACCESS													
Schmidhuber, J., and F. N. Tubiello. 2007.	Access Price	Price	increase in price	NA		-	-	-	80%	170%	-	current period	

"Global Food Security under Climate Change." Proceedings of the National Academy of Sciences. https://doi.org/10.1073/pnas.0701976104., 2007	(cereal)											(timewise)	
IPCC AR4 (Easterling et al, 2007)	Access Price (cereal)	price	increase in price	NA		-	10-30%	10-30%	10-40%	10-40%	10-40%		
Parry, M. L., C. Rosenzweig, A. Iglesias, M. Livermore, and G. Fischer. 2004. "Effects of Climate Change on Global Food Production under SRES Emissions and Socio-Economic Scenarios." Global Environmental Change. https://doi.org/10.1016/j.gloenvcha.2003.10.008.	Access Price (food crops)	Price	increase in price	NA		-	-	5-35%	-	-	-		Increase fertiliser and pesticide application, irrigation
Fujimori, Shinichiro, Tomoko Hasegawa, Joeri Rogelj, Xuanming Su, Petr Havlik, Volker Krey, Kiyoshi Takahashi, and Keywan Riahi. 2018. "Inclusive Climate Change Mitigation and Food Security Policy under 1.5 °C Climate Goal." Environmental Research Letters 13 (7): 074033. https://doi.org/10.1088/1748-9326/aad0f7.	Access Price (food crops)	price	increase in price	NA		-	-	-	-	-	-		food policy scenarios (international aid, domestic reallocation, bioenergy tax)
Hertel, Thomas W., Marshall B. Burke, and David B. Lobell. 2010. "The Poverty Implications of Climate-Induced Crop Yield Changes by 2030." Global Environmental Change. https://doi.org/10.1016/j.gloenvcha.2010.07.001.	Access Price (major staples)	Price	increase in price	NA		3.60%	10-15%	-	-	-	-		new crop varieties, significant expansion of irrigation Infrastructure
UNCCD 2017	Access (disproportionate impact on low-income consumers, in particular women and girls, due to lack of resources to purchase food)	soil health	negative	NA		-	-	-	-	-	-		low (soil health provides key adaptation option, without which lit reviewed by UNCCD points towards low adaptation potential)
Vermeulen, Sonja J., Bruce Campbell, and John S. Ingram. 2012. "Climate Change and Food Systems." SSRN. https://doi.org/10.1146/annurev-environ-020411-130608.	Access (inability to invest in adaptation and diversification measures to endure price rises)	agricultural yields and earnings, food prices, reliability of delivery, food quality, and, notably,	reduced access to food	NA		-	-	-	-	-	-		low

		food safety											
Morris, George Paterson, Stefan Reis, Sheila Anne Beck, Lora Elderkin Fleming, William Neil Adger, Timothy Guy Benton, and Michael Harold Depledge. 2017. “Scoping the Proximal and Distal Dimensions of Climate Change on Health and Wellbeing.” Environmental Health: A Global Access Science Source. https://doi.org/10.1186/s12940-017-0329-y.	Access (indirect impacts due to spatial dislocation of consuption from production for many societies)	crop yield	reduced access to food	GGCMs		-	-	-	-	-	-	strong negative effects of climate change, especially at higher levels of warming and at low latitudes	
FAO 2016a	Access (loss of agricultural income due to reduced yields and higher costs of production inputs, such as water, limits ability to buy food)	crop yield	negative	NA		-	-	-	-	-	-	likely 1.0 and 1.5	low to moderate
Abid, Muhammad, Uwe A. Schneider, and Jürgen Scheffran. 2016. “Adaptation to Climate Change and Its Impacts on Food Productivity and Crop Income: Perspectives of Farmers in Rural Pakistan.” Journal of Rural Studies. https://doi.org/10.1016/j.jrurstud.2016.08.005.	Access (loss of agricultural income due to reduced yields and higher costs of production inputs, such as water, limits ability to buy food)	farm income	negative	NA		-	-	-	-	-	-	likely 1.0 and 1.5	low
Harvey, Celia A., Zo Lalaina Rakotobe, Nalini S. Rao, Radhika Dave, Hery Razafimahatratra, Rivo Hasinandrianina Rabarijohn, Haingo Rajaofara, and James L. MacKinnon. 2014. “Extreme Vulnerability of Smallholder Farmers to Agricultural Risks and Climate Change in Madagascar.” Philosophical Transactions of the Royal Society B: Biological Sciences. https://doi.org/10.1098/rstb.2013.0089.	Access (loss of agricultural income due to reduced yields and higher costs of production inputs, such as water, limits ability to buy food)	farm income	negative	NA		-	-	-	-	-	-	likely 1.0 and 1.5	low

Calvin, Katherine, Marshall Wise, Page Kyle, Pralit Patel, Leon Clarke, and Jae Edmonds. 2014. "Trade-Offs of Different Land and Bioenergy Policies on the Path to Achieving Climate Targets." Climatic Change 123 (3–4): 691–704. https://doi.org/10.1007/s10584-013-0897-y.	Access (Price)	Price	increase in price	NA		-	-	-	-	320%	-		
Kreidenweis, Ulrich, Florian Humpenöder, Miodrag Stevanović, Benjamin Leon Bodirsky, Elmar Kriegler, Hermann Lotze-Campen, and Alexander Popp. 2016. "Afforestation to Mitigate Climate Change: Impacts on Food Prices under Consideration of Albedo Effects." Environmental Research Letters 11 (8): 085001. https://doi.org/10.1088/1748-9326/11/8/085001.	Access (Price)	Price	increase in price	NA		-	-	60-80%	-	-	-		Increase investment in R&D, etc
Tilman, David, and Michael Clark. 2014. "Global Diets Link Environmental Sustainability and Human Health." Nature. https://doi.org/10.1038/nature13959.	Access demand	demand	doubling of demands by 2050	NA		-	-	-	-	-	-		
Chatzopoulos, Thomas, Ignacio Pérez Domínguez, Matteo Zampieri, and Andrea Toreti. 2019. "Climate Extremes and Agricultural Commodity Markets: A Global Economic Analysis of Regionally Simulated Events." Weather and Climate Extremes. https://doi.org/10.1016/j.wace.2019.100193. et al., 2019	Access	Economic impacts				negative. Large-scale events will 'very likely' occur more frequently, more intensely, and last longer	key wheat-growing regions display yield reductions from −28% (Australia) to −6% (US and Ukraine). ...consumer prices increase by up to one third, most notably in Asian countries	"Besides Australia, three more regions exceed a reduction of -20%: Canada, Russia, and Kazakhstan."	"persistent large-scale harvest failures may deplete grain stocks and thus render future prices even more responsive."	-	-	unspecified in the modeling approach based on extreme events, implied 1.5GMST	governments trapped in risk-averse or risk-taking behavior, difficult to achieve and sustain crop stocks to buffer
UTILIZATION													
Müller, Christoph, Joshua Elliott, and Anders Levermann. 2014. "Food Security: Fertilizing Hidden Hunger." Nature Climate Change. https://doi.org/10.1038/nclimate2290.	Utilization (decline in nutritional quality resulting from increasing	human migration	negative (heat stress induced long-term migration of people)	NA		-	-	-	-	-	-	likely between 1.0 and 1.5 due to heat stress peaks	low (unless long term migration is considered an acceptable form of migration)

	atmospheric CO_2)												
Myers, Samuel S., Antonella Zanobetti, Itai Kloog, Peter Huybers, Andrew D.B. Leakey, Arnold J. Bloom, Eli Carlisle, et al. 2014. "Increasing CO2 Threatens Human Nutrition." Nature. https://doi.org/10.1038/nature13179.	Utilization (decline in nutritional quality resulting from increasing atmospheric CO_2)	zinc and iron	reduced nutrition	NA	2050 or 550ppm	-	-	-	-	-	-	550ppm	Low/Moderate. Differences between cultivars of a single crop suggest that breeding for decreased sensitivity to atmospheric CO_2 concentration could partly address these new challenges to global health.
Smith, M. R., C. D. Golden, and S. S. Myers. 2017. "Potential Rise in Iron Deficiency Due to Future Anthropogenic Carbon Dioxide Emissions." GeoHealth. https://doi.org/10.1002/2016gh000018.	Utilization (decline in nutritional quality resulting from increasing atmospheric CO_2)	iron	negative (iron deficiency)	NA		-	-	550 ppm	-	-	-	likely between 1.0 and 1.5 due to heat stress peaks	low to moderate
Myers, Samuel S., K. Ryan Wessells, Itai Kloog, Antonella Zanobetti, and Joel Schwartz. 2015. "Effect of Increased Concentrations of Atmospheric Carbon Dioxide on the Global Threat of Zinc Deficiency: A Modelling Study." The Lancet Global Health. https://doi.org/10.1016/S2214-109X(15)00093-5.	Utilization (decline in nutritional quality resulting from increasing atmospheric CO_2)	zinc deficiency under different CO_2 concentrations	negative (zinc deficiency)	NA	2050	-	-	The total number of people estimated to be placed at new risk of zinc deficiency by 2050 was 138 million (95% CI 120-156).	-	-	-		moderate
Moretti, Antonio, Michelangelo Pascale, and Antonio F. Logrieco. 2019. "Mycotoxin Risks under a Climate Change Scenario in Europe." Trends in Food Science and Technology. https://doi.org/10.1016/j.tifs.2018.03.008.	Utilization (higher post-harvest losses due to mycotoxins)	crops after harvest	reduced availability of food	NA	current to 2050	-	-	-	-	-	-	possibly between 1.0 and 1.5	low to moderate
Fels-Klerx, H.J. Van der, C. Liu, and P. Battilani. 2016. "Modelling Climate Change Impacts on Mycotoxin	Utilization (negative impact on food	crops after harvest	reduced utilization of food	NA		-	-	-	-	-	-	likely between 1.0 and 1.5	not yet clear

Contamination." World Mycotoxin Journal. https://doi.org/10.3920/wmj2016.2066.	safety due to effect of increased temperatures on microorganisms, including increased mycotoxins in food and feed)												
Tirado, Maria Cristina, and Janice Meerman. 2012. "Climate Change and Food and Nutrition Security." In The Impact of Climate Change and Bioenergy on Nutrition. https://doi.org/10.1007/978-94-007-0110-6-4.	Utilization (negative impact on food safety due to effect of increased temperatures on microorganisms, including increased mycotoxins in food and feed)		reduced utilization of food	NA	to midcentury	-	-	-	-	-	-		moderate
Aberman, Noora Lisa, and Cristina Tirado. 2014. "Impacts of Climate Change on Food Utilization." In Global Environmental Change. https://doi.org/10.1007/978-94-007-5784-4_124.	Utilization (negative impact on nutrition resulting from reduced water quantity and quality used to prepare food)	food availability, utilization, access	negative	NA	2020-end of century	-	-	-	-	-	-	likely between 1.0 and 1.5	low (water availability)
Thompson, Brian, and Marc J. Cohen. 2012. The Impact of Climate Change and Bioenergy on Nutrition. The Impact of Climate Change and Bioenergy on Nutrition. https://doi.org/10.1007/978-94-007-0110-6.	Utilization (negative impact on nutrition resulting from reduced water quantity and quality used to prepare food)	nutrition, distribution of food	negative	NA		-	-	-	-	-	-		low
Special Report on Global Warming of 1.5˚C Summary for Policymakers, 2018	Utilization (nutrition)	nutrients	Decrease in nutritional content	NA		at 0.87, yellow - associated impacts are both detecta	associated impacts are both detectable and attributable to	indicates closer to severe and widespread	-	-	-	Limiting global warming to 1.5°C compared to 2°C	

						ble and attributable to climate change with at least medium confidence.	climate change with at least medium confidence.	impacts.				would result in a lower global reduction innutritional quality	
Bahrami, Helale, Luit J. De Kok, Roger Armstrong, Glenn J. Fitzgerald, Maryse Bourgault, Samuel Henty, Michael Tausz, and Sabine Tausz-Posch. 2017. "The Proportion of Nitrate in Leaf Nitrogen, but Not Changes in Root Growth, Are Associated with Decreased Grain Protein in Wheat under Elevated [CO2]." Journal of Plant Physiology. https://doi.org/10.1016/j.jplph.2017.05.011.	Utilization Nutrients	nutrients	above ground biomass production and yield will typically increase by 17–20% while concentrations of nutrients such as N will decrease by 9–15% in plant tissues. Here they found - The 12% loss in grain protein under e[CO2]	NA		-	-	-	-	-	-		Grain yield per plant was greater under e[CO2]. Irrigation treatment significantly enhanced grain yield by 128%. Grain protein concentration (%) decreased by 12% in e[CO2] grown wheat compared to a[CO2]. Grain protein concentration (%) was 15% higher in rain-fed than well-watered treatments but did not differ between the two wheat cultivars. Continuing favourable water supply conditions for photosynthesis during grain filling can prolong carbohydrate delivery to grains and thereby increase yield but depress grain protein, which is consistent with greater

													grain yield and lower grain protein concentrations in wellwater edcompared to rain-fed crops in our study
Medek, Danielle E., Joel Schwartz, and Samuel S. Myers. 2017. "Estimated Effects of Future Atmospheric Co2concentrations on Protein Intake and the Risk of Protein Deficiency by Country and Region." Environmental Health Perspectives. https://doi.org/10.1289/EHP41.	Utilization nutrition	protein content	Decrease Under eCO2, rice, wheat ,barley, and potato protein contents decreased by 7.6% ,7.8%, 14.1% ,and 6.4%,respectively .	NA		-	-	-	-	-	-		
Smith, M. R., C. D. Golden, and S. S. Myers. 2017. "Potential Rise in Iron Deficiency Due to Future Anthropogenic Carbon Dioxide Emissions." GeoHealth. https://doi.org/10.1002/2016gh000018.	Utilization nutrition	nutrients	CO2concentrations of 550 ppm can lead to 3–11% decreases of zinc and iron concentrations in cereal grains and legumes and 5–10% reductions in the concentration of phosphorus, potassium, calcium, sulfur, magnesium, iron, zinc, copper, and manganese across a wide range of crops under more extreme conditions of 690 ppmCO2	NA		-	-	-	-	-	-		
Puma, Michael J., Satyajit Bose, So Young Chon, and Benjamin I. Cook. 2015. "Assessing the Evolving Fragility of	Utlilization (disruptions to food	crops after harvest	reduced utilization of food	NA	1992-2009	moderate risk at present	increased connectivity and flows	-	-	-	-		low

the Global Food System." Environmental Research Letters. https://doi.org/10.1088/1748-9326/10/2/024007.	storage and transportation networks)						within global trade networks suggest that the global food system is vulnerable to systemic disruptions, especially considering tendency for exporting countries to switch to non-exporting states during times of food scarcity in the global markets.						
Wellesley, Laura, Felix Preston, Johanna Lehne, and Rob Bailey. 2017. "Chokepoints in Global Food Trade: Assessing the Risk." Research in Transportation Business and Management. https://doi.org/10.1016/j.rtbm.2017.07.007.	Utlilization (disruptions to food storage and transportation networks)	food prices	reduced utilization of food	NA		-	-	-	-	-	-	likely 1.0 and 1.5	moderate
STABILITY													
Schmidhuber, J., and F. N. Tubiello. 2007. "Global Food Security under Climate Change." Proceedings of the National Academy of Sciences. https://doi.org/10.1073/pnas.0701976104., 2007	Stability		High Fluctuation (price, supply, yields)	NA		negative. increased fluctuations in crop yields and local food supplies and higher risks of landslides and erosion damage, they can adversely affect the stability of food supplies and thus	In semiarid areas, droughts can dramatically reduce crop yields and livestock numbers and productivity (most in sub-Saharan Africa and parts of South Asia) poorest regions with the highest level of	-	-	-	-		Food import, freer trade, investment (storage, irrigation, transport, communication)

						food security.	chronic undernourishment will also be exposed to the highest degree of instability in food production						
Zheng, Jingyun, Lingbo Xiao, Xiuqi Fang, Zhixin Hao, Quansheng Ge, and Beibei Li. 2014. "How Climate Change Impacted the Collapse of the Ming Dynasty." Climatic Change. https://doi.org/10.1007/s10584-014-1244-7.	Stability (civil disturbance, social tension)	social tension	disruption food supply	NA		-	1. Extreme events will severely disrupt the food supply 2. Extreme events will escalate popular unrest, rebellions and wars 2. Extreme events will increase expenditure to 60 -70%	-	-	-	-		
Diffenbaugh, Noah S., Thomas W. Hertel, Martin Scherer, and Monika Verma. 2012. "Response of Corn Markets to Climate Volatility under Alternative Energy Futures." Nature Climate Change. https://doi.org/10.1038/nclimate1491.	Stability (impacts on world market export prices that carry through to domestic consumer prices due to climate shocks)	price of corn	negative	NA		-	-	-	-	-	-		low
Verma, Monika, Thomas Hertel, and Noah Diffenbaugh. 2014. "Market-Oriented Ethanol and Corn-Trade Policies Can Reduce Climate-Induced US Corn Price Volatility." Environmental Research Letters. https://doi.org/10.1088/1748-9326/9/6/064028.	Stability (impacts on world market export prices that carry through to domestic consumer prices due to climate shocks)	price of corn	likely negative	NA		-	-	-	-	-	-		low
Willenbockel, Dirk. 2012. Extreme Weather Events and Crop Price Spikes in a Changing	Stability (impacts on world market	food price	negative (potential food price	NA	2030	-	1. Extreme events, such as	-	-	-	-		moderate

Climate. Illustrative Global Simulation Scenarios. Oxfam Research Reports.	export prices that carry through to domestic consumer prices due to climate shocks)		impacts of a number of extreme weather event scenarios in 2030 for each of the main exporting regions for rice, maize and wheat)				flooding, can wipe out economic infrastructure; 2. Agricultural infrastructure will be affected 3. weather-related yield shocks occurred will occur 4. Global crop production will drop						
Salmon, J.Meghan, Mark A. Friedl, Steve Frolking, Dominik Wisser, and Ellen M. Douglas. 2015. “Global Rain-Fed, Irrigated, and Paddy Croplands: A New High Resolution Map Derived from Remote Sensing, Crop Inventories and Climate Data.” International Journal of Applied Earth Observation and Geoinformation. https://doi.org/10.1016/j.jag.2015.01.014.	stability (political and economic)	rainfall, temperature	disruption food supply, price fluctuation, decrease in production	NA		-	-	-	-	-	-		agricultural intensification, ghanges in land use practices
Medina-Elizalde, Martín, and Eelco J. Rohling. 2012. “Collapse of Classic Maya Civilization Related to Modest Reduction in Precipitation.” Science. https://doi.org/10.1126/science.1216629.	stability (political and economic)	rainfall	Low yields	NA		-	-	-	-	-	-		
Challinor, Andy J., W. Neil Adger, Tim G. Benton, Declan Conway, Manoj Joshi, and Dave Frame. 2018. “Transmission of Climate Risks across Sectors and Borders.” Philosophical Transactions of the Royal Society A: Mathematical, Physical and Engineering Sciences. https://doi.org/10.1098/rsta.2017.0301.	Stability (widespread crop failure contributing to migration and conflict)	crop failure	negative	NA		-	-	-	-	-	-		moderate
Hendrix, Cullen S. 2018. “Searching for Climate–conflict Links.” Nature Climate Change. https://doi.org/10.1038/s41558-018-0083-3.	Stability (widespread crop failure contributing to migration and conflict)	crop failure	negative	NA	current	-	-	-	-	-	-		moderate
Kelley, Colin, Shahrzad Mohtadi, Mark Cane,	Stability (widespr	crop failure	negative	NA	current	negative.severe	"Multiyear	-	-	-	-		low to medium.

Richard Seager, and Yochanan Kushnir. 2017. "Commentary on the Syria Case: Climate as a Contributing Factor." Political Geography. https://doi.org/10.1016/j.polgeo.2017.06.013.	ead crop failure contributing to migration and conflict)					drought 2006/2007 caused northeastern "breadbasket" region to collapse (zero or near-zero production, livestock herds lost).	drought episodes in the late 1950s, 1980s, and 1990s, the total population of Syria (Fig. 1D) grew from 4 million in the 1950s to 22 million in recent years; (ii) decline ground water supply (iii) drought occurred shortly after the 1990s drought						
Kelley, Colin P., Shahrzad Mohtadi, Mark A. Cane, Richard Seager, and Yochanan Kushnir. 2015. "Climate Change in the Fertile Crescent and Implications of the Recent Syrian Drought." Proceedings of the National Academy of Sciences 112 (11): 3241–46. https://doi.org/10.1073/pnas.1421533112.	Stability (widespread crop failure contributing to migration and conflict)	crop failure	negative, low yields and price increase	NA	current	-	1.Extreme events will lead to unprecedented rise in food prices 2. Extreme events will obiltrate livestock	-	-	-	-		low
Schmidhuber, J., and F. N. Tubiello. 2007. "Global Food Security under Climate Change." Proceedings of the National Academy of Sciences. https://doi.org/10.1073/pnas.0701976104.	Stability (production, supply chain, extreme events)	extreme events	Fluctuation (yield and supply), Reduction (labour, productivity), Increase (disease burden)	NA		-	1. droughts can dramatically reduce crop yields and livestock productivity 2. exposed to the highest degree of instability in food production	-	-	-	-		Food imports, Freer trade, Investment (storage, irrigation, transport, communication
Chatzopoulos, Thomas, Ignacio Pérez Domínguez, Matteo Zampieri, and Andrea Toreti. 2019. "Climate Extremes and	stability(variability in supply, price)	yield, market, price	Fluctuation (yield, market and price)	NA		negative. climate extremes collide	key wheat-growing regions display yield	Besides Australia, three more	The transmission of domestic		-	unspecified in the modeling approac	buffer stock schemes for stabilizing supply

Agricultural Commodity Markets: A Global Economic Analysis of Regionally Simulated Events." Weather and Climate Extremes. https://doi.org/10.1016/j.wace.2019.100193. et al., 2019						with major drivers (population growth, dietary shifts, environmental degradation, andtrade interdependence.	reductions −28% (Australia) to −6% (US and Ukraine).	regions exceed a reduction of -20%: Canada, Russia, and Kazakhstan. The highest absolute drops, corresponding to −0.9 t/ha and −0.7 t/ha, were found in Canada and Russia.	prices to global markets is visible in most scenarios with large shocks in key exporters and importers being responsible for the most pronounced effects.			h based on extreme events, implied 1.5GMST. "Economic simulation models typically operate under the assumption of 'normal' growing conditions, contain no explicit parameterization of climatic anomalies on the supply side, and confound multifarious sources of yield fluctuation in harvest-failure scenarios"	and prices of major staple commodities in food-insecure regions may mitigate some of the induced price volatility but are generally difficult to achieve and sustain in practice
Bellemare, Marc F. "Rising Food Prices, Food Price Volatility, and Social Unrest." American Journal of Agricultural Economics, 2015, doi:10.1093/ajae/aau038.	Stability (trade)	trade, supply, price	negative, trade in situations where global grain production is reduced does not distribute world food stocks / inadequate and counter to modeling results (in reality producing countries protect domestic grain reserves; prices	NA	2007-2010	negative.	990-2011 food price increases led to increases in social unrest, food price volatility has not been associated with increases in social unrest	-	-	-	-		medium in SSP1-like world

			spike upwards in times of reduced yields but do not fall as much in times of normal or increased yields)										
Zampieri, M., A. Ceglar, F. Dentener, and A. Toreti. 2017. "Wheat Yield Loss Attributable to Heat Waves, Drought and Water Excess at the Global, National and Subnational Scales." Environmental Research Letters. https://doi.org/10.1088/1748-9326/aa723b.	stability(variability in supply, price)	yield, market, price	Fluctuation (yield, market and price)	NA		negative.	-	-	-	-	-		
Donati, Michele, et al. "The Impact of Investors in Agricultural Commodity Derivative Markets." Outlook on Agriculture, 2016, doi:10.5367/oa.2016.0233.	Stability (trade)	trade, supply, price	negative, trade in situations where global grain production is reduced does not distribute world food stocks / inadequate and counter to modeling results (in reality producing countries protect domestic grain reserves; prices spike upwards in times of reduced yields but do not fall as much in times of normal or increased yields)		2007-2010	negative.	open trade helps improve access to food at lower prices, combined with observations in other articles about impact of market speculation (US) combined with export restraints (Russia, Ukraine, India, Vietnam) in 2007-2011 drought periods.	-	-	-	-		
Gilbert, C. L., and C. W. Morgan. "Food Price Volatility." Philosophical Transactions of the Royal Society B: Biological Sciences, 2010,	Stability (trade)	trade, supply, price	negative, trade in situations where global grain production is		2007-2010	negative. not yet clear if trend in food price volatilit	"World dollar prices of major agricultural food commod	-	-	-	-	moderate	global

doi:10.1098/rstb.2010.0139.			reduced does not distribute world food stocks / inadequate and counter to modeling results (in reality producing countries protect domestic grain reserves; prices spike upwards in times of reduced yields but do not fall as much in times of normal or increased yields)			y is permanent	ities rose dramatically from late 2006 through to mid-2008. Prices collapsed dramatically in the second half of 2008 with the onset of the financial crisis. periods of high volatility have been relatively short and interspaced with longer periods of market tranquillity. It would therefore be wrong simply to extrapolate recent and current high volatility levels into the future. However, it remains valid to ask whether part of the volatility rise may be permanent."						
Gilbert, Christopher L. "How to Understand High Food Prices." Journal of Agricultural Economics, 2010, doi:10.1111/j.1477-9552.2010.00248.x.	Stability (trade)	trade, supply, price	negative, trade in situations where global grain production is reduced does not		2007-2010	negative. not yet clear if trend in food price volatility is perman	index-based investment in agricultural futures markets is seen as the	-	-	-	-		moderate depending on exposure to market speculation

			distribute world food stocks / inadequate and counter to modeling results (in reality producing countries protect domestic grain reserves; prices spike upwards in times of reduced yields but do not fall as much in times of normal or increased yields)			ent	major channel through which macroeconomic and monetary factors generated the 2007–2008 food price rise						
Headey, Derek. “Rethinking the Global Food Crisis: The Role of Trade Shocks.” Food Policy, 2011, doi:10.1016/j.foodpol.2010.10.003.	Stability (trade)	trade, supply, price	negative, trade in situations where global grain production is reduced does not distribute world food stocks / inadequate and counter to modeling results (in reality producing countries protect domestic grain reserves; prices spike upwards in times of reduced yields but do not fall as much in times of normal or increased yields)			negative.	when food prices peaked in June of 2008, they soared well above the new equilibrium price. observations that international rice prices surged in response to export restrictions by India and Vietnam suggested that trade-related factors could be an important basis for overshooting, especially given the very tangible	"In all cases except soybeans, we find that large surges in export volumes preceded the price surges. The presence of these large demand surges, together with back-of-the-envelope estimates of their price impacts, suggests that trade events played a much larger and	-	-	-		monthly data from Thailand (the largest exporter of rice), and the United States (the largest exporter of wheat and maize and the third largest exporter of soybeans).

							link between export volumes and export prices	more pervasive role than previously thought."					
Marchand, Philippe, et al. "Reserves and Trade Jointly Determine Exposure to Food Supply Shocks." Environmental Research Letters, 2016, doi:10.1088/1748-9326/11/9/095009.	Stability (trade)	trade, supply, price	negative, trade in situations where global grain production is reduced does not distribute world food stocks / inadequate and counter to modeling results (in reality producing countries protect domestic grain reserves; prices spike upwards in times of reduced yields but do not fall as much in times of normal or increased yields)		2007-2010	negative. without coordinated and effective international and domestic risk management of food stocks.	supply shocks driven not only by the intensification of trade, but as importantly by changes in the distribution of reserves. trade dependency may accentuate the risk of food shortages from foreign production shocks	increased number and volume of trade links (relative to production), decrease and a more even distribution of global reserves (still relative to production). -->distribution of reserves matters more than their aggregate quantity in terms of conferring resilience to shocks.	Possibility of multiple supply side shocks across different regions of the world (multi-breadbasket failure)	Compounded risk: Tra greater reliance on imports increases the risk of critical food supply losses following a foreign shock, notably in the case of several Central American and Caribbean countries that import grains from the United States"	-		Medium. Trade dependency has substantially increased in the last few decades and more than doubled since the mid-1980s (Porkka et al 2013, D'Odorico et al 2014) likely as a result of liberalization and the associated removal of subsidies and trade protections in developing countries (e.g., Shafaeddin 2005)."
Sternberg, Troy. "Chinese Drought, Bread and the Arab Spring." Applied Geography, 2012, doi:10.1016/j.apgeog.2012.02.004.	Stability (trade, political)	trade, supply, price	negative, trade in situations where global grain production is reduced does not distribute world food stocks / inadequate and counter to modeling results (in		2007-2010	"Chinese drought contributed to a doubling of global wheat prices. The drought affected the price of bread in Egypt which influenced	-	-	-	-	-	-	Depends on food reserves, trade policy (risk management) and if multi-breadbasket failure is present

			reality producing countries protect domestic grain reserves; prices spike upwards in times of reduced yields but do not fall as much in times of normal or increased yields)			political protest. The process exemplifies the potential global consequences of climate hazards today.″							
Permafrost degradation													
Chadburn, S. E. et al., 2017 NCC	permafrost degradation	Permafrost area change (million km^2)	increased loss of permafrost, leading to radical changes in high-latitude hydrology and biogeochemical cycling. Estimated sensitivity of permafrost area loss to global mean warming at stabilization of 4.0 +/- 1.1 million km^2 °C^-1.	CMIP5, multiple RCPs	1850-2300	Indirectly	13	9	6	4	2	-	Global
Burke, E. J. et al., 2018 ERL	permafrost degradation	Increased land carbon emissions at stabilization Gt C /yr	Additional emissions between 225 and 345 GtC (10th to 90th percentile) from permafrost thaw under 2 °C stabilised warming. 60–100 GtC less in a 1.5	JULES-IMOGEN intermediate complexity climate model	1.5° and 2°C stabilization	-	1.5: 0.08 to 0.16 Gt C yr−1 (10th to 90th percentile)	0.09 to 0.19 GtC yr−1 (10th to 90th percentile)	-	-	-	-	Global

			°C world.										
Jorgenson & Osterkamp 2005	permafrost degradation	Water erosion	Increased water erosion	Review	-	-	-	-	-	-	-	-	Global
Gauthier et al., 2015	permafrost degradation	Tree mortality	Permafrost thawing in dry continental Siberia may trigger widespread drought-induced mortality in dark coniferous forests and larch forests that cover 20% of the global boreal forest	Review	-	-	-	-	-	-	-	-	Fennoscandia, Siberia and the northern reaches of North America
FAO 2012	permafrost degradation	Damage to forest hydrological regimes	Permafrost thawing will reinforce the greenhouse effect and induce irreversible damage to forest hydrological regimes, especially across regions receiving litle rainfall.	Review	2012-2030	-	-	-	-	Carbon release by 2100 could be several times that of current tropical deforestation	-	-	Siberia
Price et al., 2013	permafrost degradation	Permafrost thaw	Increases in nearsurface permafrost temperatures during 2007–2009 are up to 2 °C warmer compared to 2-3 decades, and there is a concurrent trend in its degradati	Review	1995-2100	-	Permafrost is now warming at almost all sites across the North American permafrost zones, except for site where the permafrost is already close to	-	Rapid degradation and disappearance over extensive areas within next 50–100 years (Camill 2005; Smith et al. 2005). Accelerated degrad	16%–35% of Canadian permafrost area in 2000 may be lost by 2100 (Zhang et al., 2008a; 2008b)	-	-	Canada

			on and disappearance. Overall transient responses of permafrost to warming are likely to be nonlinear.				0 °C and vertical ground temperature profiles are isothermal, indicating ongoing phase changes (Smith et al. 2010)		ation by 2050 likely in several regions.				
Hjort et al., 2018 NatComm	permafrost degradation	Proportion of all residential, transportation, and industrial infrastructure in areas of nearsurface permafrost thaw (a) and high hazard (b) in the pan-Arctic permafrost area (%)	Arctic infrastructure at risk from degrading permafrost by mid-century	Infrastructure hazard computations	2041 – 2060	-	-	4 million people, 70% of current infrastructure	-	-	-	-	Global
Fire													
Bajocco et al., 2010	fire	Area burned	Multidirectional relationships between climate, land degradation and fire may be amplified under future land use change and climate scenarios (Bajocco et al. 2010).	-	1990-2000	-	-	-	-	-	-	-	Mediterranean
Marlon et al., 2016	fire	Biomass burning	Increase in charcoal influx (i.e. biomass burning) during the undustrial period (probably not related to climate	Paleoclimate reconstruction	22ka-2000	-	-	-	-	-	-	-	Global

			but human activities)										
Giglio et al., 2016	fire	Area burned	Trends in land area burnt have varied regionally	Recent observations	1995-2011	Regionally varying trends	-	-	-	-	-	-	Northern Hemisphere Africa has experienced a fire decrease of 1.7 Mha yr-1 (–1.4% yr-1) since 2000, while Southern Hemisphere Africa saw an increase of 2.3 Mha yr-1 (+1.8% yr-1) during the same period. Southeast Asia witnessed a small increase of 0.2 Mha yr-1 (+2.5% yr-1) since 1997, while Australia experienced a sharp decrease of about 5.5 Mha yr-1 (-10.7% yr-1) during 2001-11, followed by an upsurge in 2011 that exceeded the annual area burned in the previous 14 years
Andela et al., 2017 Science	fire	Area burned	A recent analysis using the Global Fire Emissions Database v.4 that includes small fires concluded that the net reduction in land	Remote sensing	1998-2015	Global decline	-	-	-	-	-	high in the tropics	Global

			area burnt globally during 1998–2015 was -24.3± 8.8% (-1.35 ± 0.49% yr–1) . However, from the point of fire emissions it is important to consider the land cover types which have experienced changes in area burned; in this instance, most of the declines have come from grasslands, savannas and other non-forest land cover types (Andela et al. 2017).										
Abatzoglou and Williams, 2016	fire	Forest area burned	Significant recent increases in forest area burned (with higher fuel consumption per unit area) recorded in western and boreal North America.	Detection/attribution	1979-2015	plus100 % cumulative forest fire area, CC acounted for 55% of incease in fuel arididty	-	-	-	-	-	moderate (rise in forest fires despite increasing adaptation measures)	western and boreal north America
Ansmann et al., 2018	fire	Forest area burned	Clear link between the western Canadian fires and	Aerosoles, case study	2017-2017	-	-	-	-	-	-	-	western and boreal north America

			aerosol loading over Europe.										
Pechony and Shindell 2010	fire	Fire activity (% rel to pre-industrial)	Temperature increase and precipitation decline may become the major driver of fire regimes under future climates as evapotranspiration increases and soil moisture decreases.	Driving forces, A2, A1B, B1; singe GCM, AR4-era	800-2100	-	plus0-10%	plus0-10%	plus5-10%	plus10-35%	plus15%	low under high warming levels	"Although temperatures rise throughout the country, it becomes more humid and rainy in the East and drier in theWest (Fig. 4B). Consequently, in the eastern United States fire activity declines, while rising considerably in the western United States (Fig. 4A). In both cases increasing population densities and land-cover changes (Fig. 4C) generally reduce fire activity."
Aldersley et al., 2011	fire	Fire regimes	Temperature increase and precipitation decline may become the major driver of fire regimes under future climates as evapotranspiration increases and soil moisture decreases.	Random forest on data sets	2000-2000	-	-	-	-	-	-	-	Global
Fernandes et al., 2017	fire	Fire regimes	Temperature increase and precipita	Logistic regression	1995-2015	Yes, for Indonesia during moderat	-	-	-	-	-	-	Indonesia

			tion decline may become the major driver of fire regimes under future climates as evapotranspiration increases and soil moisture decreases.			e to wet years							
Liu et al., 2010	fire	Probability of fire	The risk of wildfires in future could be expected to change, increasing significantly in North America, South America, central Asia, southern Europe, southern Africa, and Australia	KBDI on GCM data	2070-2100	-	-	-	-	-	-	-	North America, South America, central Asia, southern Europe, southern Africa, and Australia
Jolly et al., 2015	fire	Fire weather season length	Fire weather season has already increased by 18.7% globally between 1979 and 2013, with statistically significant increases across 25.3% but decreases only across 10.7% of Earth's land surface covered with vegetation; even sharper changes have	Weather analysis	1979-2013	Yes, global	plus18.7 %	-	-	-	-	-	Global

			been observed during the second half of this period.										
Jolly et al., 2015	fire	Area experiencing long weather fire season	Global area experiencing long weather fire season has increased by 3.1% per annum or 108.1% during 1979–2013.	Weather analysis	1979-2013	Yes, global	plus108.1%	-	-	-	-	-	Global
Huang et al., 2014	fire	Fire frequencies	Fire frequencies by 2050 are projected to increase by ~27% globally, relative to the 2000 levels, with changes in future fire meteorology playing the most important role in enhancing the future global wildfires, followed by land cover changes, lightning activities and land use, while changes in population density exhibits the opposite effects.	A1B	2000-2050	-	-	-	19%	-	-	-	Global
Knorr et al., 2016a NCC	fire	Area burned	Climate is only one driver of a complex set of	SIMFIRE +LPJGUESS RCP4.5/8.5	1971-2100	-	no change	no change	no change	plus5%	plus10%	-	Global

			environmental, ecological and human factors in influencing fire (Bowman et al. 2011). Interplay leads to complex projections of future burnt area and fire emissions (Knorr et al. 2016b,a), yet human exposure to wildland fires is projected to increase because of population expansion into areas already under high risk of fires.										
Knorr et al., 2016a NCC	fire	Axposure (#people)	Climate is only one driver of a complex set of environmental, ecological and human factors in influencing fire (Bowman et al. 2011). Interplay leads to complex projections of future burnt area and fire emissions (Knorr et al. 2016b,a), yet human exposure	SIMFIRE +LPJGUESS RCP4.5/8.5	1971-2100	-	413	-	497-646	-	527-716	-	Global

			to wildland fires is projected to increase because of population expansion into areas already under high risk of fires.										
Knorr et al., 2016b BG	fire	Greenhouse gas emissions from fire	Climate is only one driver of a complex set of environmental, ecological and human factors in influencing fire (Bowman et al. 2011). Interplay leads to complex projections of future burnt area and fire emissions (Knorr et al. 2016b,a), yet human exposure to wildland fires is projected to increase because of population expansion into areas already under high risk of fires.	SIMFIRE +LPJGUESS RCP4.5/8.5	1971-2100	-	-15%	-	-	-	-	-	Global
Flannigan et al., 2009	fire	Area burned, fire season length	General increase in area burned and fire occurrence but a lot of spatial variability, with	Review	pres-2100	-	-	-	-	-	-	-	Review of regional studies

			some areas of no change or even decreases in area burned and occurrence. Fire seasons are lengthening for temperate and boreal regions and trend will continue in a warmer world. Future trends of fire severity and intensity are difficult to determine owing to the complex and non-linear interactions between weather, vegetation and people.										
Abatzoglou et al., 2019	fire	Multimodel median proportion of burnable terrestrial surfaces for which emergence occurs (%)	Anthropogenic increases in extreme Fire Weather Index days emerge for an increasingly large fraction of burnable land area under higher global temperatures.	Fire Weather Index on 17 CMIP5 climate models	1861-2099	Yes, on 22% of burnable land	0-3%	15-30%	30-50%	-	-	-	Global (pronounced effects in Mediterranean and Amazon)
Westerling et al., 2006 Science	fire	Wildfire frequency and duration	Higher large-wildfire frequency, longer wildfire durations, and longer wildfire	Fire reports	1970-2003	Yes, for Western US	-	-	-	-	-	-	Western US

			seasons.										
Yang et al., 2014 JGR	fire	Area burned	Global decline in recent burned area (1.28 × 104km2 yr1), driven signficant decline in tropics and extratropics causd by human factors. warming and droughts are expected to increase wildfire activity towards the future.	DLEM-Fire	1901-2007	-	-	-	-	-	-	-	Global
Turco, M. et al., 2018	fire	Area burned	Increase in burned area scales with warming levels. Substantial benefits from limiting warming to well below 2 °C.	SM and NSM under RCP2.6 and RCP8.5	1981-2100	-	-	plus50-75%	plus75-175%	-	-	-	Mediterranean
Flannigan et al., 2005	fire	Area burned	Increase burned area under enhanced CO2 scenarios	2xCo2, 3xCO2 (cfr SRES A2)	1975-1995; 2050; 2100	-	-	-	plus78 %	-	plus143%	-	Canada
Coastal degradation													
Mentaschi et al., 2018	coastal degradation	Coastal erosion area (km2)	Substantial global-scale increases in coastal erosion in recent decades.	Remote sensing	1984 – 2015	No	28,000 km2 eroded globally	-	-	-	-	-	Global
Neumann, B., et al., 2015 Plos One	coastal degradation	Number of people exposed to a 1-in-100 year flood event incoastal regions (million)	Increased population exposure to 1-in-100 year storm surge. Strongest chages	Population projections	2000-2060	No	625	879-949	1053-1388	-	-	-	Coastal regions are also characterised by high population density, particularly in Asia (Banglade

			in exposure in Egypt and sub-Saharan countries in Western and Eastern Africa.										sh, China, India, Indonesia, Vietnam) whereas the highest population increase of coastal regions is projected in Africa (East Africa, Egypt, and West Africa)
Nicholls et al. 2011	coastal degradation	Number of people displaced (million)	Increases in coastal erosion.	DIVA model framework	2000-2100	No	-	-	-	-	72-187 (0.9-2.4%)	high: most of the threatened population could be protected.	Global
Cazenave and Cozannet 2014	coastal degradation	-	Increases in coastal erosion.	Review, mostly qualitatively	2000-2100	No	-	-	-	-	-	-	Global (with Southeast Asia concentrating many locations highly vulnerable to relative sea level rise)
Rahmstorf 2010	coastal degradation	-	Increases in coastal erosion.	Commentary	2000-2100	Yes	-	-	-	-	-	-	Global
Meeder and Parkinson 2018	coastal degradation	Coastal erosion	Increases in coastal erosion.	Sedimentary record	1900-2000	-	-	-	-	-	-	-	Everglades, USA
Shearman et al. 2013	coastal degradation	Coastal erosion	Net contraction in mangrove area	Land cover classification	1980s-2000s	Indirectly	-0.28%	-	-	-	-	-	Asia-Pacific Region
McInnes et al. 2011	coastal degradation	Coastal erosion	CMIP3 wind speed exhibit low skill over land areas.	CMIP3 evaluation wind speed, SRES	1981-2100	-	-	-	-	-	-	-	Global
Mori et al. 2010	coastal degradation	Coastal erosion	Wave heights increase in future climates across mid-latitudes and the Antarctic Ocean.	GCM combined with a wave model under SRES	1979-2099	-	-	-	-	-	-	-	Global (rise in wave height in midlatitudes and southern ocean, decrease in tropics)
Savard et al., 2009	coastal degradation	Coastal erosion	Increases in coastal erosion	Stakeholder discussions	2005-2007	-	-	-	-	-	-	-	Canada
Tamarin-Brodsky and Kaspi 2017	coastal degradat	Tropical cyclones	Poleward shift in	Storm tracking	1980-2099	-	-	-	-	-	-	-	Midlatitudes

	ion		the genesis latitude and increased latitudinal displacement of tropical cyclones under global warming.	algorithm to CMIP5									
Ruggiero 2013	coastal degradation	Total water level	Increases in wave height (and period), increasing the probability of coastal flooding/erosion more than sea level rise alone.	Simple total water level model	1965-2010	-	-	-	-	-	-	-	U.S. Pacific Northwest
Elliott et al., 2014	coastal degradation	Nexus	Nexus of climate change and increasing concentration of people .	Review, mostly qualitatively	-	-	-	-	-	-	-	-	Global
Knutson et al., 2010	coastal degradation	Tropical cyclone intensity	Increased intensity and frequency of high-intensity hurricanes with higher warming levels.	Review	1950-2100	Yes globally, regionally difficult	-	-	-	-	-	-	Tropical cyclone regions
Bender et al., 2010	coastal degradation	Atlantic hurricane category 4 frequency	Increased intensity and frequency of high-intensity hurricanes with higher warming levels.	CMIP3 downscaling with hurricane model; SRES A1B	2001-2020; 2081-2100	-	-	-	plus75-81%	-	-	-	Atlantic (with the largest increase projected over the Western Atlantic, north of 20°N)
Vecchi et al., 2008	coastal degradation	Hurrican Power Dissipation Index Anomaly (10^11 m^3 s^-2)	Increased intensity and frequency of high-intensity hurricanes with higher warming	Statistical regression SST PDI applied to CMIP	1950-2100	-	plus1	-1 to +4	-1 to +6	-	-	-	Atlantic

			levels.										
Bhatia et al., 2018	coastal degradation	Tropical cyclone category 4 frequency (# TCs)	Frequency, intensity, and intensification distribution of TCs all shift to higher values during the twenty-first century.	RCP4.5, single GCM	2016-2035; 2081-2100	-	plus26-67%	plus27 -133%		-	-	-	Tropical cyclone regions
Bhatia et al., 2018	coastal degradation	Tropical cyclone category 5 frequency (# TCs)	Frequency, intensity, and intensification distribution of TCs all shift to higher values during the twenty-first century.	RCP4.5, single GCM	2016-2035; 2081-2100	-	plus46-50%	plus85 -200%		-	-	-	Tropical cyclone regions
Tu et al., 2018	coastal degradation	Tropical cyclones	Regime shift in the destructive potential of tropical cyclones around 1998, with regional regulation by the ElNiño/Southern Oscillation and the Pacific Decadal Oscillation.	PDI on observations	1979-2016	No	-	-	-	-	-	-	Western North Pacific
Sharmila and Walsh 2018	coastal degradation	Tropical cyclones paths	Tropical cyclones paths shift poleward	Reanalysis	1980-2014	Indirectly: hadley cell expansion has been linked to climate change	-	-	-	-	-	-	Tropical cyclone regions
Kossin 2018	coastal degradation	Tropical cyclones translation speed	Over the last seven decades, the speed at which tropical	Best-track data from IBTrACS	1949-2016	Indirectly: trend analysis	-	-	-	-	-	-	Tropical cyclone regions

			cyclones move has decreased significantly as expected from theory, exacerbating the damage on local communities from increasing rainfall amounts										
Luke et al., 2016	coastal degradation	Forest composition	The heterogeneity of land degradation on at coasts that are affected by tropical cyclones can be further enhanced by the interaction of its components (for example, rainfall, wind speed, and direction) with topographic and biological factors (for example, species susceptibility	Case studies of TC impacts on vegetation	2004-2007	-	-	-	-	-	-	-	West Indies
Emmanuel 2005 Nature	coastal degradation	Tropical cyclone Power Dissipation Index	Potential destructiveness of hurricanes has increased markedly since the mid-1970s due to both longer storm lifetimes and greater storm intensities.	'best track' tropical data sets	1930-2010	Indirectly: consistency with increase in SST	-	-	-	-	-	-	Global
Emmanuel 2017 PNAS	coastal degradat	Tropical cyclone	Increase in	downscaling of large	1981-2000;	-	x6 increase	-	x18 increas	-	-	-	Texas

	ion	precipitation	inetnse precipitation associated with tropical cyclones	numbers of tropical cyclones from three climate reanalyses and six climate models	2081-2100		in probability since late 20th century		e in probability since late 20th century				
Wehner, M. F. et al., 2018 ESD	coastal degradation	Tropical cyclone counts of category 4/5	Increase in frequency and intensity of most intense tropiccal cyclolnes under 1.5°C and 2°C warming levels.	single GCM, HAPPI protocol	HAPPI	-	at 1.5°C: plus2.1/plus1.2	plus1.4/plus1.2	-	-	-	-	Tropical cyclone regions
Hanson et al., 2011 CC	coastal degradation	People exposed to 1-in-100-year coastal floodig (# people)	Enhanced exposure to extreme coastal flooding, with total population exposure possibly increasing threefold by 2070.	Global rise of 0.5 m above current levels by 2070, +10% increase in extreme water levels	2005; 2070s	-	38.5 M people (0.6%)	150 M people	-	-	-	high! "This research shows the high potential benefits from risk-reduction planning and policies at the city scale to address the issues raised by the possible growth in exposure." (paper)	Global
Hanson et al., 2011 CC	coastal degradation	Assets exposed to 1-in-100-year coastal floodig (%global GDP of that period)	Enhanced exposure to extreme coastal flooding, with total population exposure possibly increasing threefold by 2070.	Global rise of 0.5 m above current levels by 2070, +10% increase in extreme water levels	2005; 2070s	-	5%	9%	-	-	-	high! "This research shows the high potential benefits from risk-reduction planning and policies at the city scale to address the issues raised by the possible	Global

												growth in exposure." (paper)	
Vousdoukas et al., 2016 CDD	coastal degradation	Extreme storm surge levels	The anticipated increase in relative sea level rise can be further enforced by an increase in extreme storm surge levels.	RCP4.5 + 8.5, 8 CMIP5 models	1970-2100	-	-	-	-	-	-	present and needed	Europe
Vousdoukas et al., 2017 EF	coastal degradation	Extreme sea level change compared to present-day	100-year extreme sea level along Europe’s coastlines is on average projected to increase by 57/81 cm for RCP4.5/ 8.5.	RCP4.5 + 8.5, 6 CMIP5 models	1980-2014; 2100	-	-	plus57 cm	-	plus81cm	-	-	Europe
Vousdoukas et al., 2017 EF	coastal degradation	Extreme sea level return period affecting 5 Million Europeans	100-year extreme sea level along Europe’s coastlines is on average projected to increase by 57(81) cm for RCP4.5(8.5).	RCP4.5 + 8.5, 6 CMIP5 models	1980-2014; 2100	-	100year	3 year	-	1 year	-	-	Europe
Vousdoukas et al., 2018 NComm	coastal degradation	Extreme sea level chang compared to present-day	By 2050, extreme sea level rise would annually expose a large part of the tropics to the present-day 100-year event. Unprecedented flood risk levels by the end of the century unless	RCP4.5 + 8.5, 6 CMIP5 models	1980-2014; 2100	-	-	plus34 –76 cm	-	plus58-172cm	-	-	Global

			timely adaptation measures are taken.										
Rasmussen, D. J. et al., 2018	coastal degradation	Human population exposure under 2150 local SLR projections (millions)	Increased permafrost melt, increased coastal erosion	1.5K, 2.0K, 2.5K stabilisation scenarios	2010; 2150	-	1.5: 56.05 (32.54–112.97)	61.84 (32.89 – 138.63)	2.5: 62.27 (34.08 – 126.95)	-	-	-	Global
Moftakhari et al., 2017 PNAS	coastal degradation	Coastal flooding	Compound flooding from river flow and coastal water level enhances risk derived from univariate assessments.	RCP4.5 + 8.5	2030; 2050	-	-	-	-	-	-	-	Global
van den Hurk et al., 2015 ERL	coastal degradation	Coastal flooding	Compound flooding from river flow and coastal water level enhances risk derived from univariate assessments.	800 sim years with an RCM	2012-2012	-	-	-	-	-	-	-	The Netherlands
Zscheischler et al., 2018 NCC	coastal degradation	Coastal flooding	Interaction between multiple climate drivers and/or hazards play a major role in coastal extremes.	Review	-	-	-	-	-	-	-	-	USA
Jevrejeva, S. et al., 2018 ERL	coastal degradation	Coastal flooding	Rising global annual flood costs with future warming.	1.5K, 2.0K, stabilisation scenarios + RCP8.5 in CMIP5	2100	-	1.5°C: 1	1.2	-	14-27	-	"Adaptation could potentially reduce sea level induced flood costs by a factor of 10" (paper)	Global, "Upper middle income countries are projected to experience the largest increase in annual flood

													costs (up to 8% GDP) with a large proportion attributed to China. High income countries have lower projected flood costs, in part due to their high present-day protection standards." (paper)
Brown, S. et al., 2018 REC	coastal degradation	Decadal average of land inundatd by flooding (km2)	Increased soil erosion, increased soil salinity, subsiding land with future warming.	1.5, 2.0 and 3.0 stabilisation scenarios from SRES A1B, with Delta Dynamic Integrated Emulator Model	1986-2005; 2050; 2100	-	1.5°C: 1000-1500	1500-1700	2000-2500	-	-	"With slow rates of sea-level rise, adaptation remains possible, but further support is required" (paper)	Ganges-Brahmaputra-Meghna and other vulnerable deltas
Nicholls, R. J. et al., 2018	coastal degradation	Expected people flooded (millions yr–1)	Increase in coastal inundation and number of people exposed under future warming levels.	1.5K, 2.0K, stabilisation scenarios + RCP8.5 in CMIP5; Warming Acidification and Sea Level Projector Earth systems model , large ensembles	1986-2300	-	1.5°C: 150 (100-230)	170 (120-270)	-	-	400 (220-700)	"adaptation remains essential in densely populated and economically important coastal areas under climate stabilization. Given the multiple adaptation steps that this will require, an adaptation pathways approach has merits for coastal areas."	Global

												(paper)	
Mentaschi et al., 2017 GRL	coastal degradation	Extreme wave energy flux in 100yr return level	More extreme wave activity in the southern hemisphere towards the end of the century.	Spectral wave model Wavewatch III forced by 6 CMIP5 models under RCP8.5	1980-2010; 2070-2100	-	-	-	-	up to plus30%	-	-	Southern hemisphere
Villarini et al., 2014 BAMS	coastal degradation	Coastal flooding	Flooding from tropical cyclones affects large areas of the United States.	Discharge measurements	1981-2011	-	-	-	-	-	-	-	Eastern US
Woodruff et al., 2013 Nature	coastal degradation	Coastal flooding	Increase in future extreme flood elevations.	Review of global and regional studies	1981-2100	-	-	-	-	-	-	-	Global
Brecht et al., 2012 JED	coastal degradation	Coastal flooding	Strong inequalities in the risk from future disasters.	Implications of tropical storm intensification for 31 developing countries and 393 of their coastal cities with populations greater than 100,000	2000-2100	-	-	-	-	-	-	-	Selected cities across the world
Hallegatte et al 2013	coastal degradation	Flood losses (Billion US$ yr-1)	Increasing global flood future warming.	Quantification of present and future flood losses in the 136 largest coastal cities.	2005; 2050 (20 and 40 cm sea level rise; assume 2°C but no info in paper)	-	6	1000 without adaptation, 60-63 with adaptation keeping constant flood probability	-	-	-	huge challenge: "To maintain present flood risk, adaptation will need to reduce flood probabilities below present values" (paper)	Global
Jongman et al., 2012 GEC	coastal degradation	People and value of assets in flood-prone regions (Trillion US$ in 1/00 coastal flood hazard areas)	Increased people and asset exposure in 1-in-100-year coastal flood hazard areas.	Population density and GDP per capita estimate; land-use estimate	2010; 2050	-	27-46	80-158	-	-	-	-	Global (largest population exposure increase in Asia (absolute) and Sub-Sahran+North Africa (relative))

Muis et al., 2018 EF	coastal degradation	Coastal flooding	Significant correlations across the Pacific between ENSO and extreme sea levels.	Tides and storm surge reanalysis	1979-2014	No	-	-	-	-	-	-	Global
Reed et al., PNAS	coastal degradation	Return period of 1/500yr pre-industrial flood height (yr)	Mean flood heights increased by ~1.24 m from ~A.D.850 to present.	Proxy sea level records and downscaled CMIP5	850-1800; 1970-2005	Yes	24 year	-	-	-	-	-	New York
Wahl et al., NCC	coastal degradation	Return period of 1/100yr pre-industrial flood height (yr)	Increase in the number of coastal compound events over the past century .	Statistical analyses	1900-2012	Yes	42 year	-	-	-	-	-	USA & New York
Vegetation degradaton													
Allen et al., 2010	vegtation changes	Tree mortality	Increases in tree mortality	Global assessment of recent tree mortality attributed to drought and heat stress. "Although episodic mortality occurs in the absence of climate change, studies compiled here suggest that at least some of the world's forested ecosystems already may be responding to climate change and raise concern that forests may become increasingly vulnerable to higher background tree mortality	1970-2008	Yes but not formally	-	-	-	-	-	-	quasi-Global

				rates and die-off in response to future warming and drought, even in environments that are not normally considered water-limited" (paper)									
Trumbore et al.,2015	vegtation changes	Forest health	Intensification of stresses on forests	Review	-	-	-	-	-	-	-	-	-
Hember et al., 2017	vegtation changes	Net ecosystem biomass production (NEBP)	A 90% increase in NEBP driven by environ mental changes.	Observations at 10,307 plots across southern ecozones of Canada	1501-2012	Yes but not formally	rise in wet climates, decline in dry climates	-	-	-	-	-	Canada
Midgley and Bond 2015	vegtation changes	Vegetation structure	Climate, atmospheric CO2 and disturbance changes are able to shift vegetation between states.	Review	-	-	-	-	-	-	-	-	Africa
Norby et al., 2010	vegtation changes	Net Primary Productivity (NPP, kg dry matter m-2 yr-1)	Increasing N limitation, expected from stand development and exacerbated by elevated CO2.	FACE: CO2 vs N	1998-2008	-	reduction in NPP difference between abient and elevated CO2 experiments	-	-	-	-	-	High latitudes
Gauthier et al., 2015	vegtation changes	Boreal forest shift to woodland /shrubland biome	Increase in drought-induced mortality, changes in climate and related disturbances may overwhelm the resilience of species and ecosystems, possibly leading to importan	Review	-	-	climate zones shift faster than adaptation capacity	-	-	-	-	-	Fennoscandia, Siberia and the northern reaches of North America

			t biome-level changes.										
FAO 2012	vegtation changes	Boreal forest productivity	Enhanced dieback and timber quality decrease despite increase in forest productivity.	Review	2012-2030	-	"Higher forest mortality is already being observed in practically all areas of the boreal belt."	-	-	-	mass destruction of forest stands.	"The state of knowledge regarding adaptive potential and the regional vulnerability of forests to climate change is insufficient" (paper)	Siberia (highest risks for Southern regions and forest steppe)
Price et al., 2013	vegtation changes	Boreal forest productivity	Where precipitation is generally nonlimiting, warming coupled with increasing atmospheric carbon dioxide may stimulate higher forest productivity. Increase in large wildfires. Risk of endemic forest insect pests population outbreaks in response to relatively small temperature increases.	Review	1995-2100	-	-	-	-	-	-	-	Canada
Girardin et al., 2016	vegtation changes	Boreal forest productivity	Tree growth dependence on soil moisture in boreal Canada since the mid-20th century. Projections of	Dendrochronology	1950-2015	drought and heat control boreal tree growth	no change	-	-	-	-	-	North America

			future drying pose risk to forests especially in moisture-limited regimes.										
Beck et al., 2011	vegtation changes	Boreal forest productivity	Growth increases at the boreal–tundra ecotones in contrast with drought-induced productivity declines throughout interior Alaska. Initiating biome shift.	Dendrochronology and remote sensing	1982-2010	drought-induced productivity declines	-	-	-	-	-	-	North America
Lewis et al., 2004	vegtation changes	Tropical forest health	Widespread changes observed in mature tropical forests.	Review	1900-2001	-	-	-	-	-	-	-	Global
Bonan et al., 2008	vegtation changes	Forest health	Forests under large pressure from global change.	Review	-	-	-	-	-	-	-	-	Global
Miles et al., 2004	vegtation changes	Species becoming non-viable (%)	Little change in the realized distributions of most species due to delays in population responses.	HADCM2 GSa1 1%CO2 (old ref)	1990-2095	-	-	-	-	-	43% by 2095	-	Amazonia (highest risks over lowland and montane forests of Western Amazonia)
Anderegg et al., 2012	vegtation changes	Tree mortality	Increased tree mortality	Review	-	-	-	-	-	-	-	-	Global
Sturrock et al., 2011	vegtation changes	Tree mortality	Increased tree mortality	Review "We Review knowledge of relationships between climate variables and several forest diseases, as well as current evidence	-	-	-	-	-	-	-	"Regardless of these uncertainties, impacts of climate change on forest health must be mitigated. This will require	Global

				of how climate, host and pathogen interactions are responding or might respond to climate change." (paper)								proactive thinking and a modified suite of forest management approaches, because status quo management strategies will not protect forest values in a changing climate. Climate change is already disrupting practices and policies for managing commercial and non-commercial forests, such as forest classification systems, projections of growth and yield and subsequent models of supply for timber and other forest products, plans and projections for managing habitat for differen	

												t species of animals , and cycling of carbon, nutrient s and water (Graha m et al., 1990)." (paper)	
Bentz et al., 2010	vegtatio n changes	Tree mortality	Increase d tree mortality	Population models forced with CRCM climate projections under A2	1961-2100	-	-	-	e.g. Spruce beetle: "In the period 2001–2030 and again from 2071 to 2100, we would expect substa ntial increas es in spruce forest area with high probab ility of spruce beetle offspri ng produc ed annual ly rather than semian nually (figure 1b, 1c, 1e, 1f). By the end of the centur y, the change in temper atures across the boreal forests of central Canad a may cause marke dly	-	-	-	North America

									higher probability of spruce beetle outbreak potential, based on developmental timing alone. A model for predicting the cold tolerance of this insect is not available" (paper)				
McDowell et al., 2011	vegtation changes	Tree mortality	Increased tree mortality	Synthetic theory	1850-2100	-	-	-	-	-	-	-	Global
Lindner et al., 2010	vegtation changes	Tree mortality	positive effects on forest growth and wood production from increasing atmospheric CO2 content and warmer temperatures especially in northern and western Europe. Increasing drought and disturbance (e.g. fire) risks will cause adverse effects, outweighing positive trends in southern and eastern Europe.	Review	2000-2100	Some changes already detected (e.g. in Pyrenees)	-	-	-	-	-	-	Europe
Mokria et al. 2015	vegtation changes	Tree mortality	Decreasing trend in tree	Dendrochronology	2006-2013	-	-	-	-	-	-	-	Northern Ethiopia, dry

			mortality with increasing elevation.										afromontane forest
Shanahan et al., 2016	vegtation changes	Abrupt woodland - grassland shifts	Interactions between climate, CO2 and fire can make tropical ecosystems more resilient to change, but systems are dynamically unstable and potentially susceptible to abrupt shifts between woodland and grassland dominated states in the future.	28,000-year integrated record of vegetation, climate and fire from West Africa	15-28Ka	-	-	-	-	-	-	-	West Africa
Ferry Slik et al., 2002	vegtation changes	Tree mortality	Reduction in number of trees and tree species per surface area directly after disturbance (fire).	Forest plot monitoring	1970-2002	-	-	-	-	-	-	-	Indonesia
Dale et al., 2001	vegtation changes	Tree mortality	Altered frequency, intensity, duration and timing of fires, droughts, introduced species and other disturbances can affect forests.	Review	-	-	-	-	-	-	-	-	Global
Schlesinger and Jasechko 2014	vegtation changes	ratio of transpiration over evapotranspiration (%)	Changes in transpiration due to rising CO2	Review	-	-	-	-	-	-	-	-	Global

			concentrations, land use changes, shifting ecozones and climate warming.										
Loucks et al., 2010 CC	coastal degradation	Number of breeding tiger species	Tiger habitat loss under future climate change. High agreement that the joint effect of climate change and land degradation will be very negative for the area.	Sea level rise scenarios of 0, 12, 28cm (assumed 1,2,3K)	2000-2090	-	115	105	5	-	-	-	Sundarban, Bangladesh
Payo et al., 2016 CC	coastal degradation	Mangrove area loss (km2)	Increasing mangrove area losses by 2100 relative to 2000 due to sea level rise.	Sea level rise scenarios of 0.46, 0.75 and 1.48m	2000; 2100	-	-	-	81-1391km² lost	-	-	-	Sundarban, Bangladesh
Song et al., 2018	vegtation changes	Land change	60% of all recent land changes are associated with direct human activities whereas 40% with indirect drivers such as climate change.	Remote Sensing	1982-2016	-	40% of land change from indirect drivers such as climate change	-	-	-	-	-	Global
Mc Kee et al. 2004 GEB	vegtation changes	Salt marsh dieback (ha)	Vegetation dieback and soil degradation.	Areal and ground surveys	2000-2001	-	More than 100,000 ha affected, with 43,000 ha everely damaged	-	-	-	-	-	USA
soil erosion													
Li and Fang, 2016	soil erosion	Soil erosion rates (t ha^-1 yr^-1)	more often than not studies project an	Review	1990-2100	Indirectly: close links demonstrated	0-73.04	-	-	-	-	-	Global

			increase in erosion rates (+1.2 to + 1600%, 49 out of 205 studies project more than 50% increase).			regionally, no formal D&A							
Serpa et al., 2015	soil erosion	Sediment export change in humid/dry catchment (%)	Decrease in streamflow (2071-2100)	SWAT + ECHAM SRES A1B and B1	1971-2000; 2071-2100	-	-	-22/+5%	-29/+222%	-	-	-	Mediterranean
Neupane and Kumar, 2015	soil erosion	Change in river flow	Dominant effect of LULCC	SWAT under SRES B1, A1B, A2	1987-2001; 2091-2100	-	-	-	-	-	-	-	Big Sioux River
Mullan et al., 2012	soil erosion	Change in soil erosion	Erosion rates without land management changes would decrease by 2020s, 2050s and 2100s, dominant effect of land management	WEPP under SRES	2020s; 2050s; 2080s	-	-	-	-	-	-	-	Northern Ireland
Bond-Lamberty et al., 2018	soil erosion	Soil organic matter (SOM)	Soil carbon decline	Global soil respiration data base	1990-2014	-	-	-	-	-	-	-	Global
Bellmay et al., 2005 Nature	soil erosion	Soil property changes	Soil carbon decline	National soil inventory of England and Wales	1978-2003	Indirectly: relation between rate of carbon loss and carbon content irrespective of land use, suggesting a link to climate change.	-0.6%/yr	-	-	-	-	-	UK
Ramankutty et al., 2002	soil erosion	Suitability for agriculture (%)	Increased suitability for agriculture in northern high latitutdes,	IS92a 'business as usual' "calibrating the satellite-based IGBP-DIS 1-km land-cover	1992; 2070-2099	-	-	-	-	plus16%	-	-	Global

			decrease in tropics	classification dataset (Loveland et al ., 2000) against a worldwide collection of agricultural census data." (paper)									
Zabel et al., 2014	soil erosion	Suitability for agriculture (million km²)	Increased suitability for agriculture in northern high latitutdes , decrease in tropics	ECHAM5 SRES A1B	1980-2010; 2071-2100	-	-	-	-	plus 5.6	-	-	Global
Burt et al., 2016b	soil erosion	Extreme precipitation indices	Soil erosion may increase in a warmer, wetter world, yet land management is first-order control.	Commentary	1900-2016	-	-	-	-	-	-	-	India
Capolongo et al., 2008	soil erosion	Climate erosivity	Influence on soil erosion in Mediterranean	Simplified rainfall erosivity model	1951-2000	-	-	-	-	-	-	-	Mediterranean
Barring et al. 2003 Catena	soil erosion	Wind erosion	No clear trend in wind erosion.	Review	1901-2000	-	-	-	-	-	-	-	Sweden
Munson et al., 2011 PNAS	soil erosion	Wind erosion	Enhanced wind erosion.	Wind erosion model	1989-2008	-	-	-	-	-	-	-	USA
Allen & Breshears 1998 - PNAS	soil erosion	Water erosion	Increased water erosion.	Observations	1950-1990	-	-	-	-	-	-	-	USA
Shakesby 2011 Earth Science Reviews	soil erosion	Water erosion	Water erosion after wildfire not notably distinct in Mediterranean, likely due to land use effects	Review	-	-	-	-	-	-	-	-	Mediterranean
Pruski and Nearing 2002	soil erosion	Water erosion	Complex interactions betweeen several factors that affect erosion.	HadCM3	1990-2099	-	-	-	-	-	-	-	USA

Knorr et al., 2005 Nature	soil erosion	Soil Organic Carbon (SOC) turnover time	Soil carbon decline	Three-pool model, theoretical study	-	-	-	-	-	-	-	-	Global
Bond-Lamberty & Thompson 2010 Nature	soil erosion	Soil respiration change (PgC yr^-1)	Enhanced soil respiration.	Database of worldwide soil respiration observations	1961-2008	-	plus0.1 Pg C/yr	-	-	-	-	-	Global
Jiang et al., 2014	soil erosion	Soil erosion rates (t ha^-1 yr^-1)	No significnat change in soil erosion during one decade	Revised Universal Soil Loss Equation (RUSLE)	2000; 2006; 2012	-	-	-	-	-	-	-	Mount Elgon
Vanmaercke et al. 2011 (Science of the Total Environment)	soil erosion	Sediment yield	High sediment yield indicates desertification.	Review	-	-	-	-	-	-	-	-	Europe
Vanmaercke et al. 2016 (Earth-Science Reviews)	soil erosion	Volumetric gully headcut retreat rate change (%)	Increase in headcut retreat rates	Gully headcut retreat sensitivity to climate	-	-	gully erosion already forms an important problem in many regions	-	plus27-300%	-	-	-	Global
de Vente et al. 2013 ESR	soil erosion	Soil erosion and sediment yield	Importance of spatial and temporal scales when considering erosion processes.	Review	-	-	-	-	-	-	-	-	Global
Broeckx et al., 2018 ESR	soil erosion	Landslide susceptibility	precipitation not a significant driver of landslide susceptibility, but is significant in non-arid climates	Review	-	-	-	-	-	-	-	-	Africa
Gariano and Guzetti 2016 ESR	soil erosion	Landslide susceptibility	Increase in the number of people exposed to landslide risk in regions with future enhanced frequenc	Review	-	-	-	-	-	-	-	-	Global

			y and intensity of severe rainfall events.										
Water scarcity in drylands													
IPCC AR5	water scarcity	drought		observations	historical	high confidence in observed trends in some regions of the world, including drought increases in the Mediterranean and West Africa and drought decreases in central North America and northwest Australia							
Hoegh-Guldberg et al., 2018	water scarcity	drought		observations	historical	medium confidence that greenhouse forcing has contributed to increased drying in the Mediterranean region (including southern Europe, northern Africa and the Near East)							
Greve et al., GRL, 2015	water scarcity	P-ET (mm)	generally a decrease in P-ET in dryland regions but not statistically significant	RCP8.5	2080–2099 compared to 1980–1999	-	-	-	-	-	-	-	global
Byers et al., ERL, 2018	water scarcity	water stress	increased water	time sampling	2050	-	-	391 (11%)	418 (12%)	-	-	-	Drylands particularl

		index (population exposed and vulnerable in drylands, in millions and in percentage of drylands population)	stress with temperature	approach using a combination of RCPs									y impacted, including southwestern North America, southeastern Brazil, northern Africa, the Mediterranean, the Middle East, and western, southern and eastern Asia
Hanasaki, N., et al, 2013, Hydrol. Earth Syst. Sci., 17, 2393–2413, doi:10.5194/hess-17-2393-2013.	water scarcity	percentage of population under severely water-stressed conditions based on Cumulative Abstraction to Demand ratio CAD≤0.5	increase with time and RCP	RCP2.6, 4.5, 8.5	(2071 – 2100 compared to 1971 – 2000	-		3.6% - 12%	6.2% - 16%	-	12.3% - 22.4%	-	global
Huang, J. et al. 2017 (NCC) Drylands face potential threat under 2C global warming target (CarbonBrief)	impact of temperature increase	temperature	higher temperature increase in drylands compared to rest of the world			-	-	44% more warming over drylands than humid lands	-	-	-	-	drylands/global
Zeng and Yoon, GRL, 2009	increase desert area	expansion of desert area (i.e. LAI less than 1)	increase in desert area	A1B	2099 compared to 1901	-	-	-	-	-	2.5 million km2 (10% increase)/ with vegetation - albedo feedback: +8.5 million km2 (34% increase)	-	drylands/global
Liu, W. et al. 2018 (ESD) Global drought and severe drought-Affected populations in 1.5 and 2C warmer worlds	water scarcity	increase in population exposed to severe drought	increase in exposed population globally	time sampling approach at 1.5 and 2 degree		-	-	194.5± 276.5 M	-	-	-	-	global
Naumann, G. et al. (2018) Global Changes in Drought Conditions	water scarcity	drought magnitude	increase in drought	time sampling approach		-	-	Doubling of drough	-	-	-	-	global

Under Different Levels of Warming			magnitude	at 1.5 and 2 degree				t magnitude for 30% of global landmass					
Schewe et al., 2014 PNAS	water scarcity	river runoff as a proxi for water resources	increase in population confronted to water scarcity	RCP8.5		-	-	severe reduction in water resources for about 8% of the global population	severe reduction in water resources for about 14% of the global population	-	-	-	global
Haddeland et al., 2014 PNAS	Irrigation water scarcity	percentage of population under worsened water-stressed conditions based on Cumulative Abstraction to Demand ratio	irrigation water scarcity increases with temperature in most regions			-	-	-	-	-	-	-	global

Table SM7.2: literature considered in the expert judgement of risk transitions for figure 7.2

Reference	**Risk**	**variable (unit)**	**climate scenario**	**timeframe**	**GMST level**	**Direction of impact**	**SSP 1**	**SSP2**	**SSP3**	**SSP 4**	**SSP5**	**Region (Including Regional Differences)**
Food security												
(Palazzo et al. 2017)	food availability	percent deviaiton from 2010 Kilocalorie	RCP 8.5	2050		increase	up to 30%		only up to 10%			West Africa
(Hasegawa et al. 2018)	change in crop yield combined with exposure and vulnerability based on prevalence of the undernourishment (PoU) concept	population at risk of hunger (million)	RCP2.6	2050		increasing population at risk of hunger	approx 2M	approx 5M	approx 24M	-	-	sub-Saharan Africa and South Asia have highest impacts
(Hasegawa et al. 2018)	change in crop yield combined with exposure and vulnerability based on prevalence of the undernourishment (PoU) concept	population at risk of hunger (million)	RCP6.0	2050		increasing population at risk of hunger	approx 5M (0-30M) (RCP to GMT conversion based on SM SR15 ch3)	24M (2-56M) (RCP to GMT conversion based on SM SR15 ch3)	approx 80M (2-190M)	-	-	sub-Saharan Africa and South Asia have highest impacts

(Byers et al. 2018)	crop yield change	crop yield change (Number of exposed and vulnerable people)	time sampling approach using a combination of RCPs	2050	1.5		2	8	20	-	-	
(Byers et al. 2018)	crop yield change	crop yield change (Number of exposed and vulnerable people)	time sampling approach using a combination of RCPs	2050	2		24	81	178	-	-	
(Byers et al. 2018)	crop yield change	crop yield change (Number of exposed and vulnerable people)	time sampling approach using a combination of RCPs	2050	3		118	406	854	-	-	
(Wiebe et al. 2015)	Economic access	% change in price	RCP4.5	2050		Increase in price	~3% to ~17% (interquartile range)					
(Wiebe et al. 2015)	Economic access	% change in price	RCP6.0	2050		Increase in price	-	0 to ~12% increase (interquartile range)	-	-	-	
(Wiebe et al. 2015)	Economic access	% change in price	RCP8.5	2050		Increase in price			~5% to 30% (interquartile range), median by crop varies from 10% to 30%; restricting trade increases effects			
(van Meijl et al. 2018)	Crop production	% change in production	RCP6.0	2050		Decrease in production	2-3% decline		1-4% decline			
(van Meijl et al. 2018)	Economic access	% change in price	RCP6.0	2050		Increase in price	up to 5%		up to 20%			
(Ishida et al. 2014)	undernourishment	DALYs attributable to childhood underweight (DAtU)	Used RCP 4.5 for BAU	2050 compared to 2005		generally decrease in undernourishment	Health burden decreases by 36.4 million DALYS by 2030 and to 11.6 DALYS by 2050	decrease by 30.4 DALYS by 2030 and 17.0 DAYS by 2050	decrease by 16.2 DALYS by 2030 but increase to 43.7 by 2050	-	-	These are global statistics but there are regional differences. E.g. sub-Saharan Africa has higher DALYS
(Ishida et al. 2014)	undernourishment	DALYs attributable to childhood underweight (DAtU)	Used RCP 2.6	2050 compared to 2005		generally decrease in undernoursihment, although there are some climate impacts	Difference in health burden of 0.2% compared to BAU	Difference of 0.5% in 2050 compared to BAU	Difference of 2.0% compared to BAU	-	-	These are global statistics but there are regional differences. E.g. sub-Saharan Africa has higher DALYS
(Fujimori et al. 2018)	Economic access	GDP loss	RCP8.5	2100		Decline in GDP	0%	0.04%	0.57% decrease			

									in "GDP change rate"			
(Springmann et al. 2016)	Deaths due to changes in dietary and weight-related risk facors	Climate-related deaths	RCP2.6 to RCP8.5	2050			more avoided deaths compared to SSP2 and 3	intermediate	fewer avoided deaths			
Land degradation												
(Byers et al. 2018)	habitat degradation	population (Million) exposed and vulnerable in relation to share of land area within a pixel being converted from natural land to agricultural land	time sampling approach using a combination of RCPs	2050	1.5		88	88	107	-	-	non-drylands only; data provided by authors
(Byers et al. 2018)	habitat degradation	population (Million) exposed and vulnerable in relation to share of land area within a pixel being converted from natural land to agricultural land	time sampling approach using a combination of RCPs	2050	2		257	551	564	-	-	non-drylands only; data provided by authors
(Byers et al. 2018)	habitat degradation	population (Million) exposed and vulnerable in relation to share of land area within a pixel being converted from natural land to agricultural land	time sampling approach using a combination of RCPs	2050	3		652	1068	1156	-	-	non-drylands only; data provided by authors
(Hinkel et al. 2014)	flooding and sea level rise, Coastal erosion	number of people exposed to annual flooding		2100			Lowest number of people flooded	-	highest number of people flooded	-	-	
(Hinkel et al. 2014)	Flood costs, Coastal erosion	cost of flooding (% GDP)		2100		The global costs of protecting the coast with dikes are	-	-	lowest costs under contstant protection	-	highest costs under constant protecti	

						significant with annual investment and maintenance costs of US$ 12–71 billion in 2100, but much smaller than the global cost of avoided damages even without accounting for indirect costs of damage to regional production supply.			but highest under enhanced protection !		on	
(Zhang et al. 2018)	Extreme preciptation	population exposed to precipitation extremes (RX5day events exceeding 20-year return values)	time sampling approach on RCP8.5 and RCP4.5	2100	2	exposed population steadily increases with temperature, with only marginal differences between SSPs						
(Knorr et al. 2016a)	fire	exposure (#people)	RCP4.5 transient	2071-2100 vs 1971-2000	2		-	560	646	-	508	globally
(Knorr et al. 2016a)	fire	exposure (#people)	RCP8.5 transient	2071-2100 vs 1971-2000	4		-	610	716	-	527	globally
(Knorr et al. 2016b)	fire	emissions (Pg C yr^-1)	RCP4.5 transient	2071-2100 vs 1971-2000	2		-	1.22	1.11	-	1.31	globally
(Knorr et al. 2016b)	fire	emissions (Pg C yr^-1)	RCP8.5 transient	2071-2100 vs 1971-2000	4		-	1.33	1.22	-	1.43	globally
Desertification												
(Zhang et al. 2018)	Extreme preciptation	population exposed to precipitation extremes (RX5day events exceeding 20-year return values)	time sampling approach on RCP8.5 and RCP4.5	2100	2	exposed population steadily increases with temperature, with only marginal differences between SSPs						
(Byers et al. 2018)	water scarcity	water stress index (2050); population exposed and vulnerable in drylands (Units: Million and percentage of drylands population)	time sampling approach using a combination of RCPs	2050	1.5		76 (2%)	349 (10%)	783 (20%)	-	-	Dryland only: data provided by authors

(Byers et al. 2018)	water scarcity	water stress index (2050); population exposed and vulnerable in drylands (Units: Million and percentage of drylands population)	time sampling approach using a combination of RCPs	2050	2		82 (3%)	391 (11%)	864 (22%)	-	-	Dryland only: data provided by authors
(Byers et al. 2018)	water scarcity	water stress index (2050); population exposed and vulnerable in drylands (Units: Million and percentage of drylands population)	time sampling approach using a combination of RCPs	2050	3		91 (3%)	418 (12%)	919 (24%)	-	-	Dryland only: data provided by authors
(Arnell and Lloyd-Hughes 2014)	water scarcity	Numbers of people (millions) exposed to increased water resources stress	RCP2.6	2050			379–2997	473–3434	626–4088	508–3481	418–3033	
(Arnell and Lloyd-Hughes 2014)	water scarcity	Numbers of people (millions) exposed to increased water resources stress	RCP4.5	2050			810–2845	881–3239	1037–3975	884–3444	854–2879	
(Arnell and Lloyd-Hughes 2014)	water scarcity	Numbers of people (millions) exposed to increased water resources stress	RCP6	2050			759–2668	807–3054	924–3564	809–3227	803–2682	
(Arnell and Lloyd-Hughes 2014)	water scarcity	Numbers of people (millions) exposed to increased water resources stress	RCP8.5	2050			802–2947	(919–3416	1006–4201	950–3519	854–2981	
(Hanasaki et al. 2013)	water scarcity	Population living in grid cells with CAD < 0.5	RCP8.5	2041-2070			-	-	4188 - 4434 (baseline is ~2000; all regions increase)	-	-	Global. Paper includes maps and graphs with regional

												information.
(Hanasaki et al. 2013)	water scarcity	Population living in grid cells with CAD < 0.5 (millions)	RCP6.0	2041-2070			2853 - 3043 (baseline is ~2000; all regions increase)					Global. Paper includes maps and graphs with regional information.
UNCCD, 2017	mean species abundance, aridity; biodiversity, land degradation, water scarcity	population living in drylands					-	43% increase	-	-	-	

Table SM7.3: literature considered in the expert judgement of risk transitions for figure 7.3

Reference	Risk	Variable	Climate scenario	SSP	Timeframe	Non-climatic hazard	Bioenergy area	Impacts	Notes
(Humpenöder et al. 2017)	trade-offs with SDGs	sustainability indicators: SDG 2; 7; 13; 14; 15	no climate change (consistent with strong mitigation)	SSP1	2100 compared to baseline without bioenergy	bioenergy deployment	636 Mha	only slight impact on sustainability indicators (i.e. no trade-offs due to lower food demand in SSP1) compared to baseline	
(Humpenöder et al. 2017)	trade-offs with SDGs	sustainability indicators: SDG 2; 7; 13; 14; 15	no climate change (consistent with strong mitigation)	SSP2	2100 compared to baseline without bioenergy	bioenergy deployment	636 Mha	pronounced decrease in all sustainability indicators (i.e. increase in adverse side-effects) compared to case without bioenergy	
(Humpenöder et al. 2017)	trade-offs with SDGs	sustainability indicators: SDG 2; 7; 13; 14; 15	no climate change (consistent with strong mitigation)	SSP5	2100 compared to baseline without bioenergy	bioenergy deployment	636 Mha	pronounced decrease in all sustainability indicators (i.e. increase in adverse side-effects) even more severe than in SSP2	
(Heck et al. 2018)	planetary boundaries transgression	Planetary Boundaries (PBs): biosphere integrity; land-system change; biogeochemical flows; freshwater use	RCP2.6	SSP1	2050 compared to baseline without bioenergy	bioenergy deployment	870Mha	upper limit of most PBs is transgressed implying high risk of irreversible shifts	
(Heck et al. 2018)	planetary boundaries transgression	Planetary Boundaries (PBs): biosphere integrity; land-system change; biogeochemical flows; freshwater use	RCP2.6	SSP2	2050 compared to baseline without bioenergy	bioenergy deployment	778Mha	upper limit of most PBs is transgressed implying high risk of irreversible shifts	
(Boysen et al. 2017)	food production	kcal cap−1 day−1 production loss (%); N application (Mt yr^-1)	4.5°C trajectory	NA	2100	bioenergy deployment	1078Mha	-43%; 96 Mt yr^-1	
(Boysen et al. 2017)	food production	kcal cap−1 day−1 production loss	4.5°C trajectory	NA	2100	bioenergy deployment	2176Mha	-73%; 151 Mt yr^-1	

		(%); N application (Mt yr^-1)							
(Boysen et al. 2017)	food production	kcal cap−1 day−1 production loss (%); N application (Mt yr^-1)	4.5°C trajectory	NA	2100	bioenergy deployment	4267Mha	-100%; 196 Mt yr^-1	
(Hasegawa et al. 2018)	population at risk of hunger	population at risk of hunger (million)	RCP2.6	SSP1	2050 compared to baseline	mitigation policies (including bioenergy)	262Mha (106-490) (provided by authors)	approx +25M	
(Hasegawa et al. 2018)	population at risk of hunger	population at risk of hunger (million)	RCP2.6	SSP2	2050 compared to baseline?	mitigation policies (including bioenergy)	752Mha (175-1904) (provided by authors)	approx +78M (0-170)	
(Hasegawa et al. 2018)	population at risk of hunger	population at risk of hunger (million)	RCP2.6	SSP3	2050 compared to baseline?	mitigation policies (including bioenergy)	813Mha (171-1983) (provided by authors)	approx +120M	
(Fujimori et al. 2018)	population at risk of hunger	population at risk of hunger (million)	RCP2.6	SSP1	2050 compared to baseline	mitigation policies (including bioenergy)	90Mha	approx +20M	
(Fujimori et al. 2018)	population at risk of hunger	population at risk of hunger (million)	RCP2.6	SSP2	2050 compared to baseline	mitigation policies (including bioenergy)	170Mha	approx +100M	
(Fujimori et al. 2018)	population at risk of hunger	population at risk of hunger (million)	RCP2.6	SSP3	2050 compared to baseline	mitigation policies (including bioenergy)	220Mha	approx +260M	
(Obersteiner et al. 2016)	agricultural water use	km3		SSP1	2030	bioenergy	210Mha	approx + 13 km3	
(Obersteiner et al. 2016)	agricultural water use	km3		SSP2	2030	bioenergy	210Mha	approx +12km3	
(Obersteiner et al. 2016)	agricultural water use	km3		SSP3	2030	bioenergy	210Mha	approx +11km3	
(Hejazi et al. 2014)	bioenergy water withdrawal	km3		SSP3	2050	bioenergy	150 Mha	approx +300km3	Paper uses a pre-cursor to the SSP3, with a similar population and storyline.
(Hasegawa et al. 2015)	population at risk of hunger	population	RCP2.6	SSP2	2050	bioenergy	280Mha	approx +2M	
Fujimori et al., NSust, accepted	population at risk of hunger	population	No climate; but assessed in SM as small effect	SSP2	2050	bioenergy	38 - 395 Mha	approx 25 - 160 M	Difference between 1.5C scenario and Baseline for both bioenergy and impact. Total population at risk of hunger is ~300 to >500 million; total increase in population at risk of hunger is 50 to 320 M. Authors state that roughly half is attributed to bioenergy; those numbers are included here.
Fujimori et al., NSust, accepted	population at risk of hunger	population	No climate; but assessed in SM as small effect	SSP2	2050	bioenergy	43 - 225 Mha	approx 20 - 145 M	Difference between 2C scenario and Baseline for both bioenergy and impact. Total population at risk of hunger is ~290 to ~500 million; total increase in

									population at risk of hunger is 40 to 290 M. Authors state that roughly half is attributed to bioenergy; those numbers are included here.

Table SM7.4. Risks thresholds for different components of desertification, land degradation and food security as a function of global mean surface temperature change relative to pre-industrial times. The confidence levels are defined according to the IPCC guidance note on consistent treatment of uncertainties (Mastrandrea et al., 2010). These data are used in Figure 7.1

Component	**Risk Transition**	**Global mean surface temperature change above pre-industrial levels °C**		**Confidence**
Low Latitude Crop Yield	Undetectable to Moderate	Min	0.5	high
		Max	0.7	
	Moderate to High	Min	1.2	medium
		Max	2.2	
	High to Very High	Min	3.0	medium
		Max	4.0	
Food Supply Stability	Undetectable to Moderate	Min	0.75	high
		Max	0.85	
	Moderate to High	Min	0.9	medium
		Max	1.4	
	High to Very High	Min	1.5	medium
		Max	2.5	
Permafrost Degradation	Undetectable to Moderate	Min	0.3	high
		Max	0.7	
	Moderate to High	Min	1.1	high
		Max	1.5	
	High to Very High	Min	1.8	medium
		Max	2.3	
Vegetation Loss	Undetectable to Moderate	Min	0.7	high
		Max	1.0	
	Moderate to High	Min	1.6	medium
		Max	2.6	
	High to Very High	Min	2.6	medium
		Max	4.0	
Coastal Degradation	Undetectable to Moderate	Min	0.8	high
		Max	1.05	
	Moderate to High	Min	1.1	high
		Max	1.6	
	High to Very High	Min	1.8	high

		Max	2.7	
Soil Erosion	Undetectable to Moderate	Min	0.8	medium
		Max	1.2	
	Moderate to High	Min	2.0	low
		Max	3.5	
	High to Very High	Min	4.0	low
		Max	6.0	
Fire	Undetectable to Moderate	Min	0.7	high
		Max	1.0	
	Moderate to High	Min	1.3	medium
		Max	1.7	
	High to Very High	Min	2.5	medium
		Max	3.0	
Water Scarcity in Drylands	Undetectable to Moderate	Min	0.7	high
		Max	1.0	
	Moderate to High	Min	1.5	medium
		Max	2.5	
	High to Very High	Min	2.5	medium
		Max	3.5	
Food Access	Undetectable to Moderate	Min	0.8	medium
		Max	1.1	
	Moderate to High	Min	1.4	low
		Max	2.4	
	High to Very High	Min	2.4	low
		Max	3.4	
Food Nutrition	Undetectable to Moderate	Min	1.1	low
		Max	1.7	
	Moderate to High	Min	1.9	low
		Max	2.2	
	High to Very High	Min	2.3	low
		Max	3.3	

7. SM. 1. Additional embers

Details of two embers (nutrition and coastal degradation) where not included in Chapter 7 due to space limitations. Changes in atmospheric CO2, will result in reduced nutritional value of crops (iron, protein, zinc, other micronutrients, and increases in mycotoxins), impacting food utilization, with potential risks to health of vulnerable groups such as children and pregnant women (*high confidence, high agreement*). This may create nutrition-related health risks for 600 million people (Zhou et al. 2018). Further details are provided in Chapter 5 of this Report.

Coastal flooding and degradation bring risk of damage to infrastructure and livelihoods. There are very few global studies investigating past changes in coastal degradation (erosion and flooding) and associated risk (Muis et al. 2018; Mentaschi et al. 2018), yet strong evidence exists that anthropogenic climate change is already affecting the main drivers of coastal degradation, including: mean and extreme sea level (IPCC, 2013), storm surges (Wahl et al. 2015) and tropical cyclones (Kossin 2018). It is also clear that land-based processes, such as groundwater extraction and land subsidence, may impact coastal degradation {See Chapter 4, including 4.8.5}.

At 1.5°C there is a high risk of destruction of coastal infrastructure and livelihoods (Hoegh-Guldberg et al. 2018) (*high confidence*). There is an associated strong increase in people and assets exposed to mean and extreme sea level rise and to coastal flooding above 1.5°C. Very high risks start to occur above 1.8 °C (*high confidence*) (Hanson et al. 2011; Vousdoukas et al. 2017; Jevrejeva et al. 2018; Hallegatte et al. 2013). Impacts of climate change on coasts is further explored in the IPCC Special Report on the Ocean and Cryosphere in a Changing Climate.

7. SM 2 SSP and Mitigation Burning Embers

Table SM7.5 Risks thresholds associated to desertification, land degradation and food security as a function of Global mean surface temperature change relative to pre-industrial levels and socio-economic development. Risks associated to desertification include, population exposed and vulnerable to water scarcity and changes in irrigation supply and demand. Risks related to land degradation include vegetation loss, population exposed to fire and floods, costs of floods, extent of deforestation, and ecosystem services including the ability of land to sequester carbon. Risks to food security include population at risk of hunger, food price increases, disability adjusted life years. The risks are assessed for two contrasted socio-economic futures (SSP1 and SSP3) under unmitigated climate change up to 3°C. These data are used in Figure 7.2.

Component	Risk Transition	Global mean surface temperature change above pre-industrial levels °C		Confidence
Land Degradation (SSP1)	Undetectable to Moderate	Min	0.7	High
		Max	1.0	
	Moderate to High	Min	1.8	low
		Max	2.8	
	High to Very High	Min		does not reach this threshold
		Max		
Land Degradation (SSP3)	Undetectable to Moderate	Min	0.7	High
		Max	1.0	
	Moderate to High	Min	1.4	Medium
		Max	2.0	
	High to Very High	Min	2.2	Medium
		Max	2.8	
Food Security (SSP1)	Undetectable to Moderate	Min	0.5	Medium
		Max	1.0	
	Moderate to High	Min	2.5	Medium
		Max	3.5	
	High to Very High	Min		does not reach this threshold
		Max		

Food Security (SSP3)	Undetectable to Moderate	Min	0.5	Medium
		Max	1.0	
	Moderate to High	Min	1.3	Medium
		Max	1.7	
	High to Very High	Min	2	Medium
		Max	2.7	
Desertification (SSP1)	Undetectable to Moderate	Min	0.7	High
		Max	1.0	
	Moderate to High	Min		Does not reach this threshold
		Max		
	High to Very High	Min		Does not reach this threshold
		Max		
Desertification (SSP3)	Undetectable to Moderate	Min	0.7	High
		Max	1.0	
	Moderate to High	Min	1.2	Medium
		Max	1.5	
	High to Very High	Min	1.5	Medium
		Max	2.8	

Table SM7.6 Risk thresholds associated with 2nd generation bioenergy crop deployment (in 2050) as a land-based mitigation strategy under two SSPs (SSP1 and SSP3). The assessment is based on literature investigating the consequences of bioenergy expansion for food security, ecosystem loss and water scarcity, these indicators being aggregated as a single risk metric. These data are used in Figure 7.3.

Component	Risk Transition	Land area used for bioenergy crop (Mkm2)		Confidence
Risk due to bioenergy deployment (SSP1)	Undetectable to Moderate	Min	1	Medium
		Max	4	
	Moderate to High	Min	6	Low
		Max	8.7	
	High to Very High	Min	8.8	Medium
		Max	20	
Risk due to bioenergy deployment (SSP3)	Undetectable to Moderate	Min	0.5	Medium
		Max	1.5	
	Moderate to High	Min	1.5	Low
		Max	3	
	High to Very High	Min	4	Medium
		Max	8	

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