Chapter 7: Agriculture, Forestry, and Other Land Uses (AFOLU)

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Executive summary

The expectation on land to deliver mitigation is very high, yet the pressures on land have grown with population, diet changes, impacts of climate change and conversion of natural land to agriculture and other land uses (*high confidence*).

Agriculture, Forestry and Other Land Use (AFOLU) is expected to play a vital dual role in the portfolio of mitigation options across all sectors because of its necessary contribution to reduced emissions, as well as being the only sector for which it is currently feasible to enhance removals at scales that can contribute to carbon neutrality. AFOLU is globally a source of 23% of important driver of GHG emissions, as well as being a large carbon store, a human induced sink and a natural sink. The latter for around 29% of anthropogenic CO_2 emissions. Around 14% of total global anthropogenic CO_2 emissions, 44% of CH_4 and 88% of N2O came from AFOLU during 2007-2016. Global tree cover has increased since 1981, while other sources show a decline (low confidence) with strong regional differences of generally losses in tropical regions and gains in temperate and boreal regions.

Country pledges (Nationally Determined Contributions under the Paris Agreement) expect 25% of pledged mitigation to come from AFOLU, primarily reduced deforestation, afforestation and some agricultural measures. The role of albedo, evapotranspiration, and VOCs in the total climate forcing of land use remains unclear.

Technologies and measures to reduce emissions or to enhance removals are well known and can be employed cost-effectively *(high confidence)*. The main challenges are the regionally diverse optimal measures across vegetation and management types, millions of landowners operating different sizes and types of holdings, forces that aim at opposing short-term economic gains and failing governance and institutional aspects.

Globally, the AFOLU sector has so far contributed modestly to net mitigation as emissions continue to rise. CO_2 emissions from AFOLU have remained more or less constant over the last 50 years with high uncertainty and no clear trend, while CH_4 and N_2O emissions have increased globally (*high confidence*). In some regions and countries CO_2 net emissions have gone down due to reduced rates of deforestation (e.g. Brazil, Indonesia) or sinks have increased due to afforestation and forest conservation (e.g. Europe, China, India, USA), or beneficial effects e.g. from bioenergy (although accounted in energy sectors) (*high confidence*). Only a small proportion of CDM (Clean Development Mechanism) projects under the Kyoto Protocol included the land sector. REDD+ (Reduced Emissions from Deforestation and Degradation) has been successful in some places, and additional projects that could provide emission reductions are underway, those benefits will emerge in the future. Trends in demand for food due to population and income growth, shifts towards greater meat consumption, high intensification with increased fertilizer use, and disturbances under climate change indicate that AFOLU will instead contribute to more emissions, while sinks in some regions show signs of saturation *[medium confidence*]. **What we learn from last 30 years is that there is no free ride in this sector to compensate for emissions in other sectors.**

Policies adapted to local circumstances have most chance of being successful, taking into account trade-offs and synergies with other services including food and fiber and climate adaptation (*high confidence*). More novel policy development and implementation is needed. Optimal land management that yields the largest sustained mitigation benefit in the long term will ideally maintain or increase carbon stocks, while producing an annual sustained yield of food, feed, timber, fiber and biomass feedstocks. In natural high carbon lands, protection of carbon stocks and biodiversity will be of most benefit (*high confidence*). While many mitigation activities require up-front investment, some can be low cost or even save money. Benefits may be immediate or may accrue over for many years.

Although past policies have encouraged 7.8 Gt CO_2 of AFOLU mitigation in total over nearly 30 years, substantially more resources and effort are needed to achieve 30% of the mitigation necessary to meet a 2 degree temperature threshold. Current funding for AFOLU mitigation programs is estimated to be less than $1 billion per year for direct efforts in tropical countries, with more funding for indirect measures in developed countries. This amounts to only a small share of the more than $400 billion per year that is estimated to be necessary to achieve up to 30% of global mitigation effort. Successful policies include establishing tenure rights and community forestry, payments for ecosystem services, forest certification, voluntary supply chain management efforts, and regulatory efforts. The success of different policies, however, is dependent on numerous factors in addition to funding, including governance, institutions, and the specific policy setting.

Integrated assessment model scenarios (IAMs) indicate the necessity of afforestation/reforestation and BECCS-based removals in meeting high ambition mitigation pathways *(high confidence)*. As a median value across SSPs and IAMs, required Carbon Dioxide Removal (CDR) reaches up to -14.9 $GtCO_2$ yr^{-1} for BECCS and -2.4 $GtCO_2$ yr^{-1} for afforestation in 2100. Across the different scenarios median change of global forest area throughout the 21st century reaches up to a required 7.2 Mkm^2 increases between 2010 and 2100, and agricultural land used for second generation bioenergy crop production may require up to 6.6 Mkm^2 in 2100, both enhancing competition for land and affecting potentially sustainable development.

The scientific literature indicates a range of mitigation options with a large emission reduction or sink enhancement in the AFOLU sector by 2050 (medium/high confidence). The largest potential exists for avoiding deforestation and peat/mangrove conversion with 3.7 $GtCO_2yr^{-1}$. Afforestation/reforestation is the second largest with 3.0 $GtCO_2yr^{-1}$. Agriculture and agricultural soils can achieve 1.7 $GtCO_2yr^{-1}$. Better forest management, peat restoration and harvested wood products can achieve 1.5 $GtCO_2yr^{-1}$; totaling 9.9 Pg CO_2yr^{-1}. Partly overlapping with the re-/afforestation results, bioenergy can substitute in the energy sector between 2.8 and 7 $GtCO_2yr^{-1}$. Reduction of food loss and waste and shift to a more plant-based diet will further strengthen the above mentioned achieved reductions. Land can be freed up through sustainable intensification of agriculture, reduced food wastes and diet change; in this way, a xx share of the 7.2 Mkm^2 required can be allocated and applied to improved agriculture with less emissions, forest restoration, climate-smart forestry and bioenergy plantations for enhanced sinks (medium confidence). As BECCS scales up, it may compete with other land based mitigation efforts. To ensure that BECCS and other forest mitigation efforts are complementary, efforts need to be undertaken to develop forest mitigation approaches that value or otherwise protect high carbon ecosystems, or optimise for different types of management where appropriate.

Different mixes of options are important in different regions (high confidence). Implementation has up to now remained limited due to alternative (economically more profitable) uses of land, short term economic gains and impacts of climate change. Most successful mitigation is there where synergies with other functions of land are found in an equitable manner, building on land tenure, and with food and fiber security and social justice (e.g. safety nets). Good governance and increased access to climate finance will be needed. **Mitigation options have to consider different development aspirations and pathways mediated by drivers and enabling conditions as well as new finance mechanisms.**

Transparency, credibility and accuracy in estimating and reporting GHG fluxes can contribute to incentivizing action through the Global Stock take (high confidence). 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.

7.1. Introduction

7.1.1. Key findings from previous reports

From all previous IPCC reports with land chapters, several overarching messages stand out (IPCC 2019):

1. While there is the potential for the AFOLU sector to contribute several $GtCO_2e$ of climate mitigation, the potential is constrained due to land availability, the need to continue to produce food and other land products, and the potential future impacts of climate change. Thus land can only be part of the solution alongside rapid emission reduction in other sectors;
2. There is a large uncertainty over AFOLU's mitigation potential, in part because current stocks and fluxes are uncertain and are subject to variability over time with weather and. climate change;
3. Technically many AFOLU mitigation measures are already well established, but for some it can take a long time for mitigation impact to be realised (e.g. as a forest grows).
4. Many AFOLU mitigation measures can be achieved at modest costs, although costs are very context specific;
5. Carbon stocks and greenhouse gas fluxes are under pressure from climate change;
6. Mitigation potential is not being realised, due to insufficient policies, insufficient incentives and drivers to stimulate implementation among the millions of land owners and other stakeholders in regionally, socially and economically diverse contexts;
7. Trade-offs with food and fibre provision and other ecosystem services are a major challenge, although there are also many potential synergies between options and with other Sustainable Development Goals.

AFOLU is expected to contribute roughly 30% to mitigation pledged in NDCs under the Paris Agreement (Grassi et al, 207) and mitigation scenarios also find a potentially large role for AFOLU (Chapter 3). AFOLU mitigation can contribute in several ways: (a) reduced emissions of GHG CO_2, CH_4 and N_2O, and (b) enhanced carbon removals from e.g. afforestation/reforestation, soil carbon management - this is the only sector where carbon removals are currently possible at scale; and (c) biomass products could provide a potentially low-carbon substitute for other sectors (e.g. biomass for energy generation/fuels, biochemicals, bioplastics and wood for buildings).

Several individual mitigation response options have a 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)*. Sustainable intensification, improved efficiency of agriculture production, reduced food loss and waste, and a switch to more plant based diets (where possible) can reduce emissions and free up land for further mitigation through afforestation/reforestation, agroforestry and bioenergy (SRCCL).

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)(SRCCL).* This will *likely* lead to altered trade patterns. The carbon stocks and current net carbon sink in natural ecosystems may be at risk from climate change (*SRCCL, AR6-WGII chapter Ecosystems and chapter Food & Fibre*).

While both AR5 and SRCCL both found that AFOLU contributes almost a quarter of GHG net emissions (23%, SRCCL), they found that uncertainty in both sources and sinks of CO_2 in particular are high, and this is exacerbated by difficulties in separating natural and anthropogenic fluxes. According to models, net fluxes of CO_2 due to anthropogenic activities was 5.2 ± 2.6 $GtCO_2$ yr^{-1}. At the same time, the natural response of the land to environmental change such as rising CO_2 and climate created a sink of 11.2 ± 2.6 $GtCO_2$ yr^{-1} during 2007-2016 Thus the overall net land-atmosphere flux due to both natural and anthropogenic processes on managed and unmanaged lands *very likely* provided a global net removal from 2007 to 2016 of (-6.0 ± 3.7 $GtCO_2$ yr^{-1}). However the anthropogenic emissions of CO_2 from AFOLU reported in countries' GHG inventories were 0.1 ± 32 1.0 $GtCO_2$ yr^{-1} globally during 2005 to 2014 as some of the sinks due to environmental change are considered anthropogenic if they occur on managed lands, and their definition of managed lands is broader than the models use. Here we update these numbers and assess further work towards understanding and reconciling them. 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.

7.1.2. Boundaries, scope and changing context of the current report

Land and its management impact on the global climate, in complex ways through biophysical and biogeochemical interactions that are described in detail in the SRCCL. The demands for food, fibre, wood, fresh water and fuel, affect the land and its ecosystem services in different spatial and temporal scales. This chapter assesses GHG fluxes between the land and atmosphere due to AFOLU and its drivers, and climate mitigation response options including policy incentives at time scales of 2030 and 2050. It builds on the Special Report on Climate Change and Land SRCCL (IPCC 2019), attempting to give a more detailed regionalised assessment.

Land has many interactions with other chapters in this report, including in fulfilling demand for food and fibre (Ch 5), providing biomass for bioenergy (Ch 6), providing woody material for buildings (Ch 9), raw materials to industry (Ch 11), and providing biofuels for transport (Ch 10). Namely mitigation options in those chapters are to more or less degree determined by (im) possibilities in the land sector.

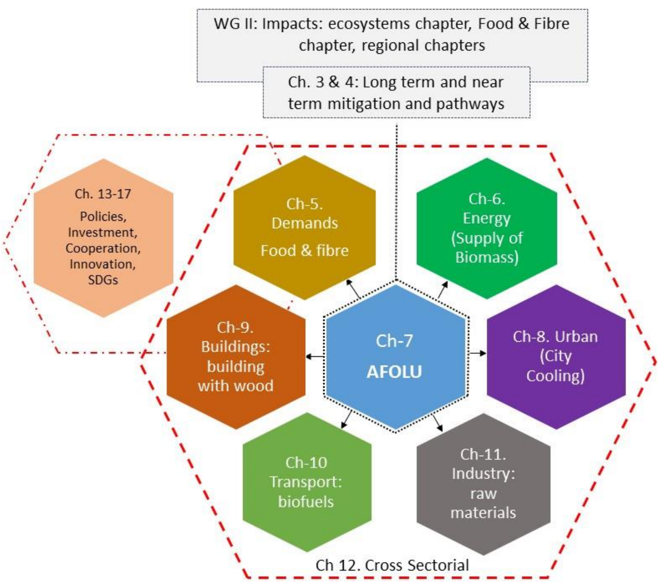


Figure 7.1 Interactions of chapter 7 to WGII and to other chapters in this report.

Our effort in this chapter concentrates on regionalizing the mitigation options with costs estimates and at the same time providing clear policy handles and incentives to change potentials into workable practice with impact at time scales of 2030 and 2050. In our analyses, not only carbon in the forest and agriculture systems is included (in biomass and soils), but also in its products and in substituted more energy intensive materials as well as bioenergy options. Furthermore, biophysical aspects of land management are touched upon as far as scientific evidence goes and we include the agriculture emissions as well with its GHG gases CH_4 and N_2O.

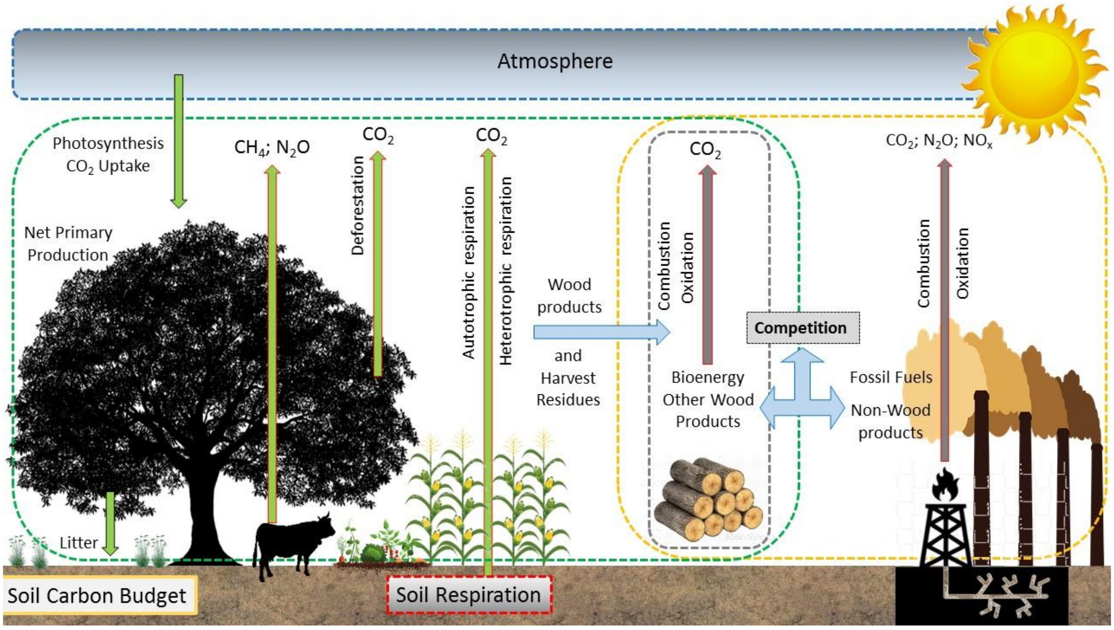


Fig 7.2 Conceptual figure on the systems approach.

In a world with soon 9 billion people, the human influence over terrestrial ecosystems will be determined by the need to provide goods and livelihoods. Therefore, a management strategy aimed at reducing emissions, maintaining or increasing carbon stocks while producing sustainable yields of timber, food, fibre or energy from land, will generate a most optimal and sustained mitigation benefit. Most mitigation activities require up-front investment with some co-benefits accruing for many years later.

Food, fibre and ecosystem services required from land are predetermined conditions that steer the regionalisation of mitigation actions as well as the policy handles. Land is owned or managed by millions of stakeholders each with their own motivations and needs. Land supports multiple services such as biodiversity, food, water, adaptation, etc. with synergies and trade-offs that may be context specific. Furthermore land use change can have biophysical effects such as through changing albedo and evapotranspiration in addition to GHG effects (section 7.3).

A relatively recent changing context is also determined by country's Nationally determined Commitments and the role of land in them. 105 countries have pledged to reduce agricultural emissions and xx countries have pledged to enhance sinks. Very few give details on how this will be achieved through policies and other incentives.

In this chapter, we aim to keep the technical description of measures short as that has already been covered in AR4, AR5 and the SRCCL. We try to deepen the assessment on regional contexts, realistic potentials and costs and policy measures and the steps towards the land representation in the upcoming Global Stocktakes. The concrete questions we want to answer in this chapter are:

- Given the global potentials as framed in SRCCL, what is the extent of realistic and feasible mitigation in various regions in the world
- How to achieve these locally and regionally, without compromising food, fibre, biodiversity
- Which policies and incentives are needed, and what are the costs.

7.2. Drivers

Drivers of land use changes

The demands that humanity places on the land systems have increased substantially over the last century, modifying and altering them with large consequences for the local and global environment, and for human well-being (Verburg et al 2013). Human decisions play a crucial role in driving changes in the land system and the dynamic interaction between socioeconomic and biophysical drivers of change (GLP 2005) (Figure 7.3 from van Vliet et al. 2015). The drivers of change are continuously developing due to the complexity of the coupled human-environmental system and the evolution or radical shifts in economic, social, cultural or environmental conditions (Friis and Reenberg, 2010).

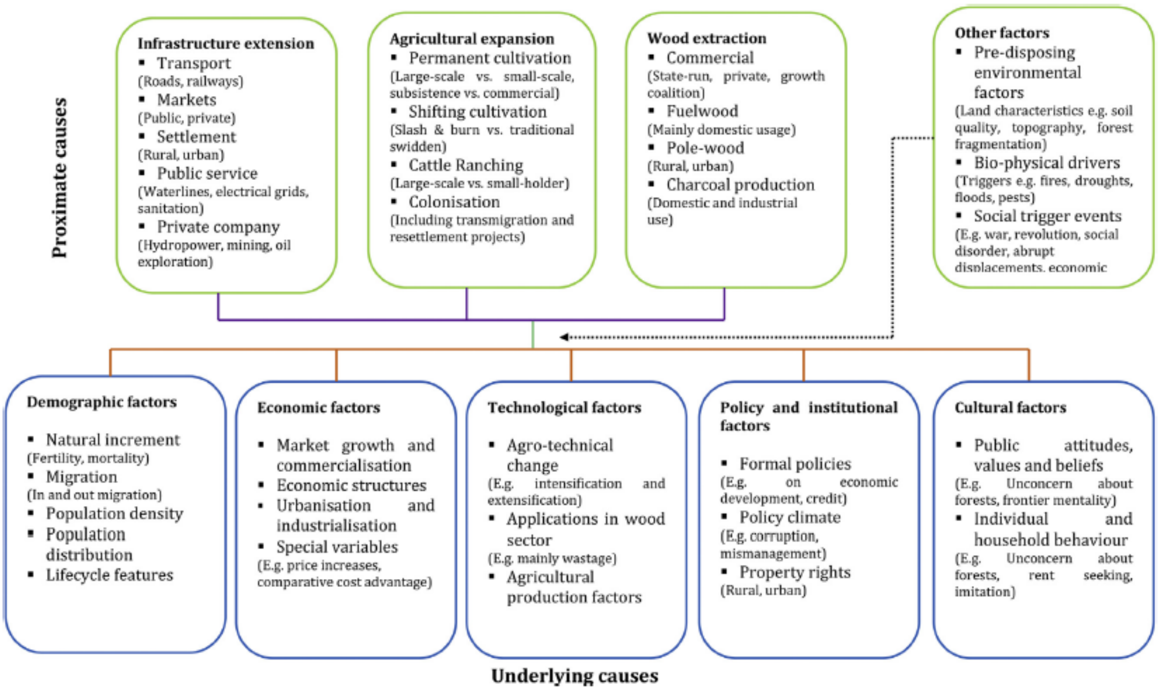


Figure 7.3 Drivers of land use change and cover change (redrawn from Geist and Lambin 2002)

A review of econometric studies of the drivers of deforestation that encompassed studies published 1996 and 2013 generated statistics on the consistency with which driver variables are associated with higher or lower rates of deforestation across many analyses and studies (Figure 7.4) (Busch and Ferretti-Gallon, 2017). Higher agriculture prices appear as the driver with higher association with deforestation while law enforcement, protected areas, and payments for ecosystem services were consistently less associated with deforestation.

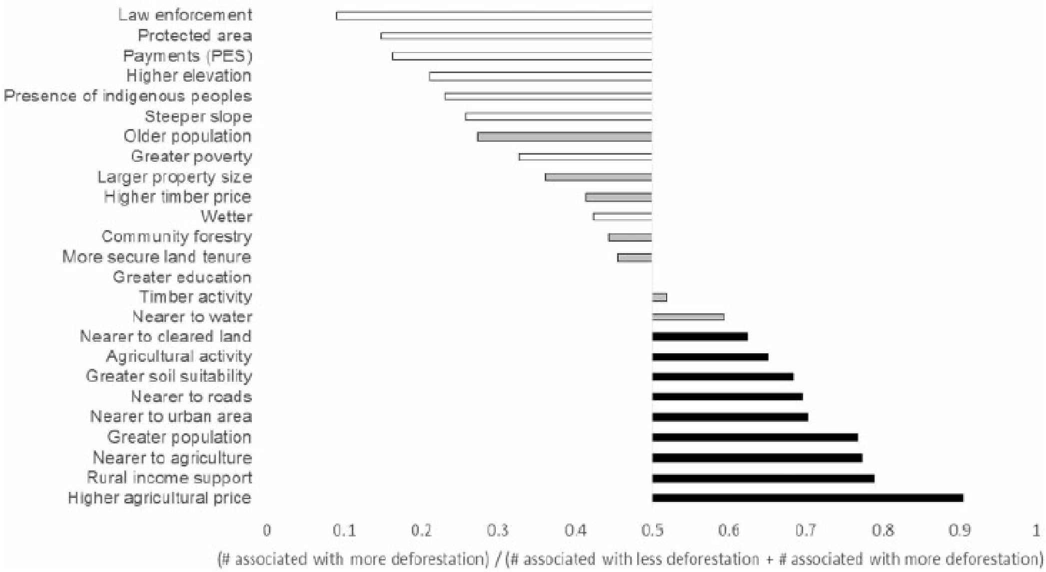


Figure 7.4 Consistency of association of driver variable with more or less deforestation (Busch and Ferretti-Gallon, 2017)

Global land-use changes

A comprehensive record of global land-change dynamics during the period 1982–2016 based on satellite imagery (Song et 2018) showed that globally tree cover has increased by 2.24 million km^2 (+7.1% relative to the 1982 level) with extratropical regions with a net gain while in the tropics there was a net loss. The same study indicated that global bare ground cover has decreased by 1.16 million km^2 (−3.1%), most notably in agricultural regions in Asia. Of all land changes, 60% are associated with direct human activities and 40% with indirect drivers such as climate change. Regional trends in terms of land-use change differ with deforestation and agricultural expansion in the tropics, reforestation or afforestation, cropland intensification and urbanization in the temperate zones. The mapped land changes and the driver attributions reflect the dominance of human activities.

Also using satellite imagery, a forest loss classification model to determine a spatial attribution of forest disturbance to the dominant drivers of land cover and land use change over the period 2001 to 2015 (Curtis et al 2018) indicated that 27% of global forest loss can be attributed to deforestation through permanent land use change for commodity production. The remaining areas maintained the same land use over 15 years; in those areas, loss was attributed to forestry (26%), shifting agriculture (24%), and wildfire (23%). The rate of deforestation remained steady across the 15-year period analyzed at approximately 5 Mha year–1 with a geographic shift away from Brazil toward tropical forests elsewhere in Latin America and Southeast Asia An additional 0.6 ±0.3% of forest loss was attributed to the intensification and expansion of urban centers.

Production of commercial agricultural commodities for domestic and foreign markets is increasingly driving land clearing in tropical regions, creating teleconnections. The quantification of tropical deforestation area and carbon emissions from LUC induced by the production and the export of four commodities (beef, soybeans, palm oil, and wood products) in seven countries with high deforestation rates (Argentina, Bolivia, Brazil, Paraguay, Indonesia, Malaysia, and Papua New Guinea) showed that between 2000–2011, the production of analyzed commodities was responsible for 40% of total tropical deforestation and resulting carbon losses (Henders et al. 2015). Also, the comparison of the impacts in 2000 and 2011 evidenced that the growing influence of global markets in deforestation dynamics.

Regional patterns of changes in forest cover

Drivers of forest loss varied regionally (Curtis et al. 2018). Forestry and wildfire were the dominant disturbance factors in temperate and boreal forests. In tropical regions, shifting agriculture and commodity-driven deforestation were more relevant. An analysis of national data from 46 tropical and sub-tropical countries on drivers of deforestation and forest degradation that have been provided as part of REDD+ readiness documents and activities (De Sy 2016) highlighted that commercial agriculture is the most prevalent deforestation driver, accounting for 40% of deforestation and most prominent in the early-transition phase. The other important land use is local/subsistence agriculture, which accounts for 33% of deforestation. Thus, agriculture alone causes 73% of all deforestation. An increasing role in the expansion of commercial agriculture into the forest is observed especially in the Amazon region and Southeast Asia, while deforestation in Africa is still largely driven by small-scale subsistence activities, but this might change in the coming years.

In the Southeast Asia (Indonesia and Malaysia), widespread deforestation was linked to oil palm plantations. While some Asian countries are experiencing rapid deforestation, some have experienced forest transition (FT) from net deforestation to net reforestation. Forest scarcity has been indicated as causal factor for forest transition. In Southeast Asia, the Philippines, Thailand and Viet Nam are experiencing FT since 1990 and have lower remaining forest area (30±8%) than the other countries (68±6%, Cambodia, Indonesia, Laos, Malaysia, and Myanmar) with prevalence of deforestation (Imai et al, 2018).
Across Central and South America, forests were converted to row crop agriculture and cattle grazing lands. Shifting agriculture was the dominant driver in sub-Saharan Africa.

Sourcing regions for the global forest products industry are concentrated in North America, Europe, Russia, China, southern Brazil, Chile, South Africa, and Australia. Most forestry activities in South America, the United States, Europe, China, South Africa, and Australia showed signs of planted or human assisted natural regeneration of, sometimes evidenced by distinct rows of planted trees, whereas forestry activity in Canada and Russia contained predominantly large clearcuts without visibly distinct plantation rows. In Southeast Asia, most forestry activity took the form of low-intensity selective logging, especially on the island of Borneo. All forms of forestry were characterized by a dominant forest regrowth signal in the years following loss.

Regarding forest degradation, the most prominent degradation driver for Latin America and Asia is unsustainable timber extraction and logging (> 70%), while fuelwood collection and charcoal is the main degradation driver in Africa (48%) (De Sy 2016). However, this last figure for Africa may be overstated, as a detailed GIS-based analysis of the impacts of fuelwood and charcoal harvesting on forest degradation and deforestation suggest that only 26% of total direct harvesting is conducted in a non-sustainable manner. Although there are woodfuels-related hotspots at national and subnational levels with values exceeding 50%, the continental average is lower as stated above (Masera et al. 2015)

Trade and commodities export

Main flows of embodied land use change in the export of commodities in Latin American are beef and soybean exports to markets in Europe, China, the former Soviet bloc, the Middle East and Northern Africa, whereas embodied emission flows are dominated by Southeast Asian exports of palm oil and wood products to consumers in China, India and the rest of Asia, as well as to the European Union (Henders et al. 2015) (Figure 7.5). China's macroeconomic growth boosted soybean production and exports from Brazil and the US. Brazil's strong soybean productivity growth over 2004-2011 contributed to cropland expansion in Brazil and allowed that country to become dominant in the global soybean market, displacing the US in the Chinese market (Yao et al. 2018).

In Latin America, palm oil output has doubled since 2001, and the majority of expansion seems to be occurring on non-forested lands. A survey of oil palm plantations across Latin America (Furumo and Aide 2017) indicated that 79% replaced previously intervened lands (e.g. pastures, croplands, bananas), primarily cattle pastures (56%) while remaining 21% came from areas that were classified as woody vegetation (e.g. forests), The expansion onto previously cleared lands seems to be guided by certifications programs. While the growth of the oil palm sector may be driven by global factors, the environmental and economic outcomes vary between regions (i.e. Asia and Latin America), within regions (i.e. Colombia and Peru), and within single countries (i.e. Guatemala), suggesting that local conditions are influential.

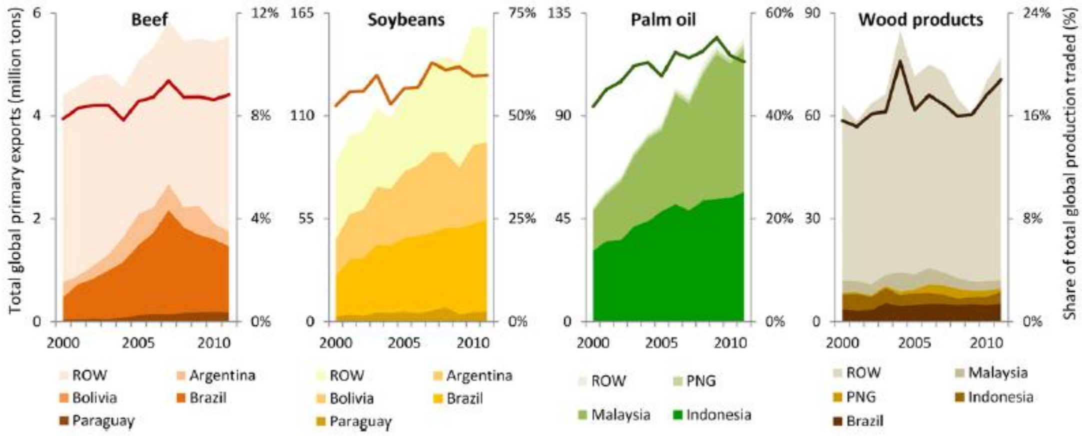


Figure 7.5 Total global primary exports (left vertical axes) of the four forest-risk commodities analyzed, for the period 2000-2011, highlighting the amount of exports coming from our case countries for each commodity. The share of global production that is traded on international markets is also displayed for each commodity (right vertical axes). All units are in million tons, except wood product values which are in million tons of carbon. Data: own calculations based on FAOSTAT (http://faostat3.fao.org)

Major drivers of mangroves deforestation

Most biodiverse mangrove forests are located in South Asia, Southeast Asia and Asia-Pacific that contain ~46% of the world's mangrove ecosystems (Giri et al. 2011; Rivera-Monroy et al. 2017; Miettinen et al. 2019) . The highest global rates of mangrove loss are also seen in these regions (Thomas et al. 2017). Globally declination in mangrove distribution is attributed primarily to anthropogenic drivers (anthropogenic activities) (Gandhi and Jones 2019; Fauzi et al. 2019). Urbanisation, industrialisation and increasing demand for commodities for population increase play crucial role for the loss of mangroves in form of mangrove deforestation (Richards and Friess 2016; Rivera-Monroy et al. 2017). Sea level rise, sedimentation reduction, nutrient enrichment (salt water intrusion and high salinity), increase in ocean temperatures, increasing frequency and intensity of tropical storms (hurricanes, cyclones), tsunamis, coastal erosion and pest infestation and disease outbreak have been found to adversely impact mangrove loss and dynamics also (Di Nitto et al. 2014; Alongi 2015; Godoy and De Lacerda 2015; Richards and Friess 2016; Osorio et al. 2017). Primary drivers include conversion of land to different land cover changes and land use practices viz. clearing of forest for agriculture (e.g. rice cultivation), expansion of aquaculture, plantations of oil palm, over-extraction for woody materials (timber, fuel wood), pollution, infrastructure, coastal development and other human activities (Bhattarai 2011; Webb et al. 2014; Giri et al. 2015; Fauzi et al. 2019). Therefore, through increased reforestation programs, sustainable mangrove management is essential for mangrove conservations.

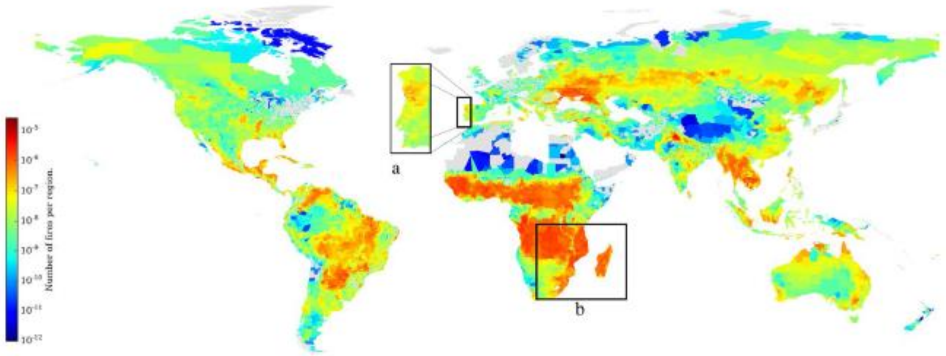
Wildfires

Wildfires are the largest contributor to global biomass burning and constitute a large global source of atmospheric traces gases and aerosols (Knorr et al. 2016a).

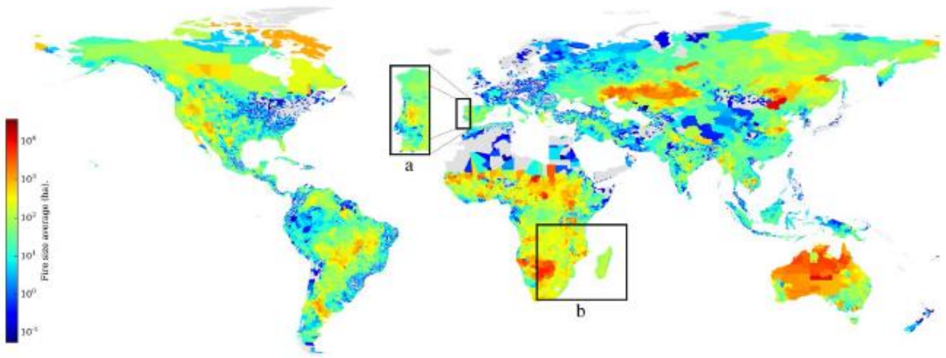
Natural and human-ignited fires affect all major biomes, altering ecosystem structure and functioning (Argañaraz et al. 2015, Engel et al. 2019, Mancini et al. 2018, Nunes et al. 2016, Remy et al. 2017) among others.

Wildfires have multiple causes with the primary driver in tropical region being land clearing for agriculture, for example, for industrial oil-palm and paper-pulp plantations in Indonesia (Chisholm et al. 2016), and for pastures in the Amazon (refs.). Other socioeconomic factors are also associated with wildfire regimes as issues land-use conflicts and socio-demographic attributes (Nunes et al. 2016, Mancini et al, 2018).

Wildfire regimes are changing by the influence of climate change with wildfire seasons are becoming longer, increases in wildfire average sizes in many areas of the world and wildfires are occurring in areas where they did not occur before (Artés et al. 2019). Climate change is driving some forests into a stressed state, reducing the vegetation water content and leading to high-severity wildfires (Brando et al. 2019).



(a) Number of fires per region surface from 2001 to 2017.



(b) Fire size average per region 2001 to 2017.

Figure 7.6 Number of fires per region surface (a) and fire size average per region (b) from 2001 to 2017 (Artés et al. 2019).

Forest management drivers in temperate and boreal zones

In the temperate and boreal zones of the world forests, the driver of land use change is mainly abandonment of agricultural lands, resulting in (semi)natural expansion of forests (Song et al 2018, Fuchs et al. 2012, Chen et al. 2019). For significant areas in temperate and boreal zones, a regular (sustainable) management is characteristic mostly driven by the wood market. Currently harvesting pressure is relatively low with ~ 15-20% of NEP being harvested globally, although this is concentrated in zones like SE-USA, West Canada, Eastern

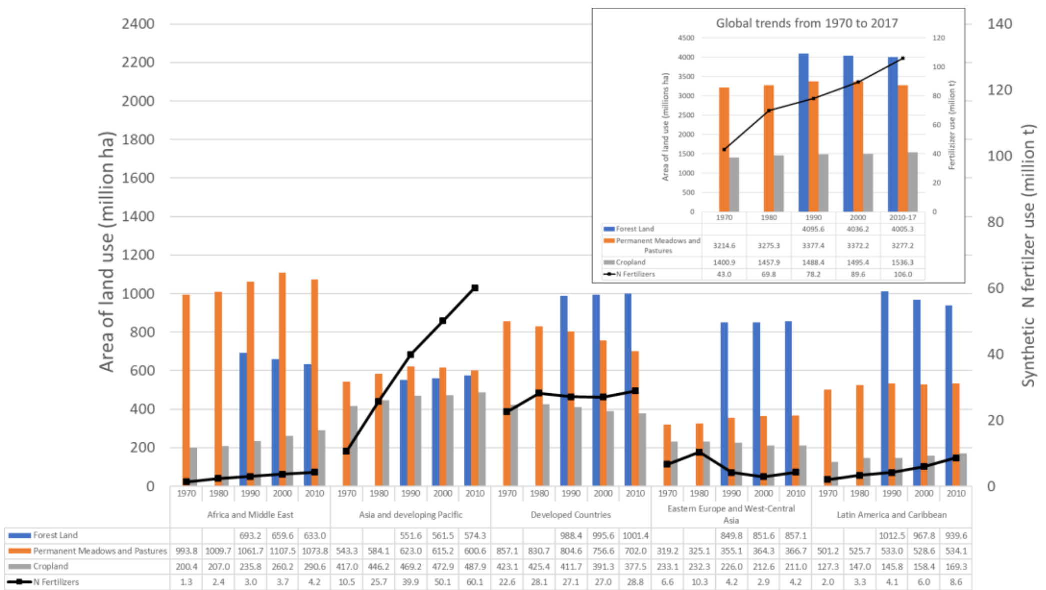
Europe & Southern Scandinavia, parts of Australia, plantations of China and parts of New Zealand, where relative pressure is much higher. Most recent drivers are increases in demand for wood because of a recovery after the 2009 recessions and a strong bio-economy trend, for both harvested wood products, biorefineries and bio-energy. To a certain extent, this increase in demand will also lead to more forest planting and thus more forest growth because of promising wood prices (Wear and Greis 2013, Galik and Abt 2015). This stimulates especially private forest owners to invest in more forest planting. In other regions (central parts of Russia) a too strong bio-economy trend may lead to harvesting moving into primary forest areas.

Another strong driver for the temperate and boreal zones is the apparent increases in natural disturbances like mountain pine beetle in West Canada (Kurz et al. 2009), bark beetle in Central Europe (Hlasny et al, 2018; Nabuurs et al. 2019) and forest fires in Russia (Bartalev 2015) and Australia (see below).

Supply and consumption trends in agriculture

Land and Fertiliser Use

In 2017 world agricultural land (Croplands plus Permanent Meadows and Pastures) occupied 4,813 Mha, an increase of 4% (198 Mha) since the 1970s; an increase in the Croplands area has been primarilly responsible for this increase. However, at the global scale there has been almost no change in the area of land devoted to agricultural activities since 1990. Major recent regional trends are an increase in cropland and a decrease in Forest land in Africa and Middle East and Latin America and Caribbean (Figure 7.7). Between 1970 and 2017 global nitrogen (N) fertiliser use has increased (147%) from 43 to 106 $Mtyr^{-1}$ with the largest regional increase observed in Asia and developing Pacific. When adjusted for land area, the annual quantity of N applied per ha (Croplands + Permanent Meadows and Pastures) is currently lowest in Africa ($3.1kgha^{-1}yr^{-1}$) and the Middle East and highest in Asian and Developing Pacific ($55kgha^{-1}yr^{-1}$) with decade on decade increases in all regions except Developed Countries and Eastern Europe and West Central Asia.



	1970	1980	1990	2000	2010-17
Forest Land			4095.6	4036.2	4005.3
Permanent Meadows and Pastures	3214.6	3275.3	3377.4	3372.2	3277.2
Cropland	1400.9	1457.9	1488.4	1495.4	1536.3
N Fertilizers	43.0	69.8	78.2	89.6	106.0

	Africa and Middle East 1970	1980	1990	2000	2010	Asia and developing Pacific 1970	1980	1990	2000	2010
Forest Land			693.2	659.6	633.0			551.6	561.5	574.3
Permanent Meadows and Pastures	993.8	1009.7	1061.7	1107.5	1073.8	543.3	584.1	623.0	615.2	600.6
Cropland	200.4	207.0	235.8	260.2	290.6	417.0	446.2	469.2	472.9	487.9
N Fertilizers	1.3	2.4	3.0	3.7	4.2	10.5	25.7	39.9	50.1	60.1

	Developed Countries 1970	1980	1990	2000	2010	Eastern Europe and West-Central Asia 1970	1980	1990	2000	2010	Latin America and Caribbean 1970	1980	1990	2000	2010
Forest Land			988.4	995.6	1001.4			849.8	851.6	857.1			1012.5	967.8	939.6
Permanent Meadows and Pastures	857.1	830.7	804.6	756.6	702.0	319.2	325.1	355.1	364.3	366.7	501.2	525.7	533.0	528.6	534.1
Cropland	423.1	425.4	411.7	391.3	377.5	233.1	232.3	226.0	212.6	211.0	127.3	147.0	145.8	158.4	169.3
N Fertilizers	22.6	28.1	27.1	27.0	28.8	6.6	10.3	4.2	2.9	4.2	2.0	3.3	4.1	6.0	8.6

Figure 7.7 Global trends from 1970-2017 in area of land use and amount of synthetic fertilizer use (million tonnes), averaged over each decade. Data sourced from FAOSTAT.

Livestock Populations

Although the total area of land devoted to agricultural activities has changed little since the 1970's there has been an increase in the quantity of food produced from this land area.

Globally, since the 1970s, there has been a 29% increase in the numbers of cattle and buffaloes, a 45% increase in sheep and goats, and increases of 44% and 255% for pigs and poultry respectively. Major regional trends between 1970 and 2017 include increases in large ruminant animals in all regions except Developed Countries and Eastern Europe and West-Central Asia regions, increases in poultry and pig populations of 10.2 billion and 267 million respectively in Asia and developing Pacific and increases in small ruminants of 407 and 475 million respectively in Asia and developing Pacific and Africa and the Middle East (Figure 7.8).

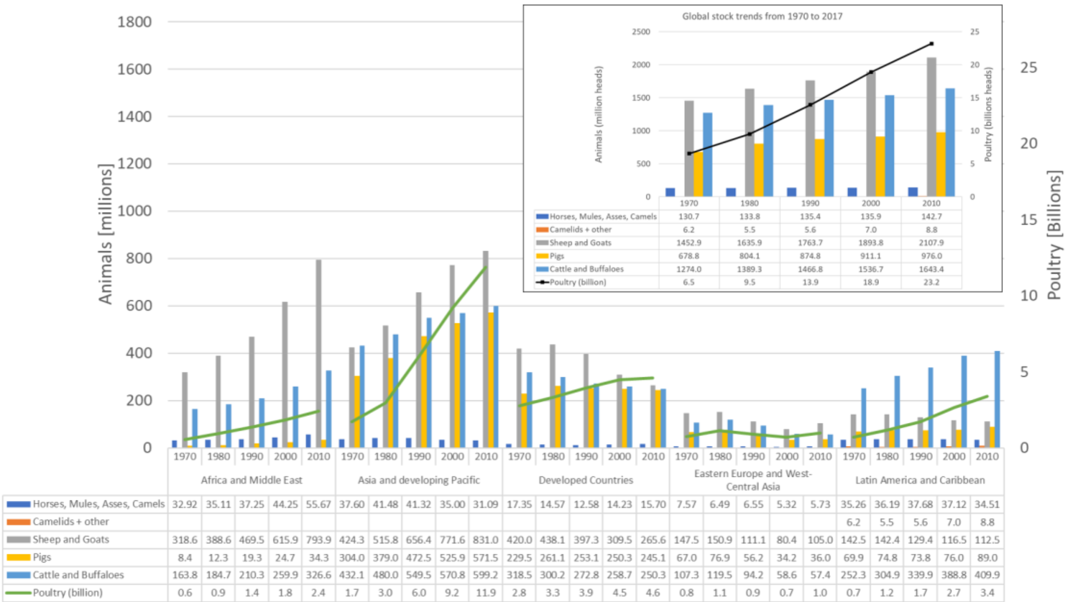


Figure 7.8 Global trends from 1970 to 2017 for number of livestock (million heads) and poultry (billion heads). Data sourced from FAOSTAT.

These changed livestock populations and increases in individual animal performance have resulted in an increased supply of fresh milk and meat. Globally meat production from all categories increased from 112 to 310 million tonnes while the production of whole fresh milk has increased 424 to 778 million tonnes. Except for Eastern Europe and West-Central Asia all regions have increased meat and milk production since the 1970's (Figure 7.9)

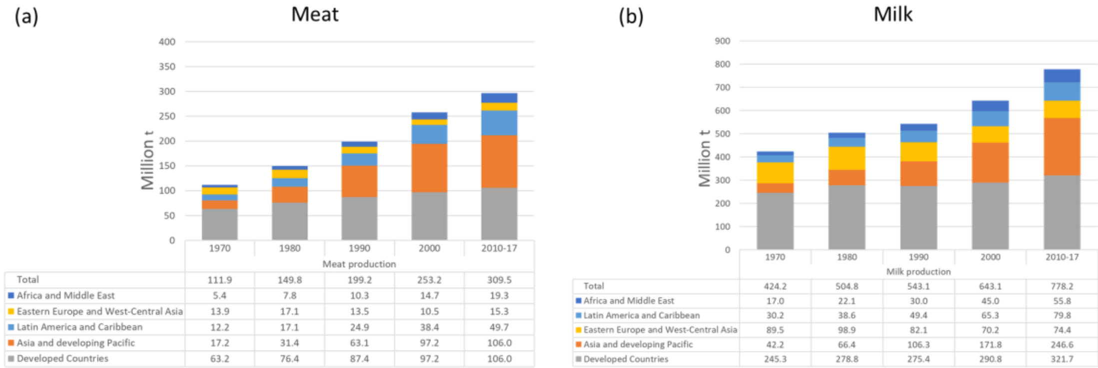
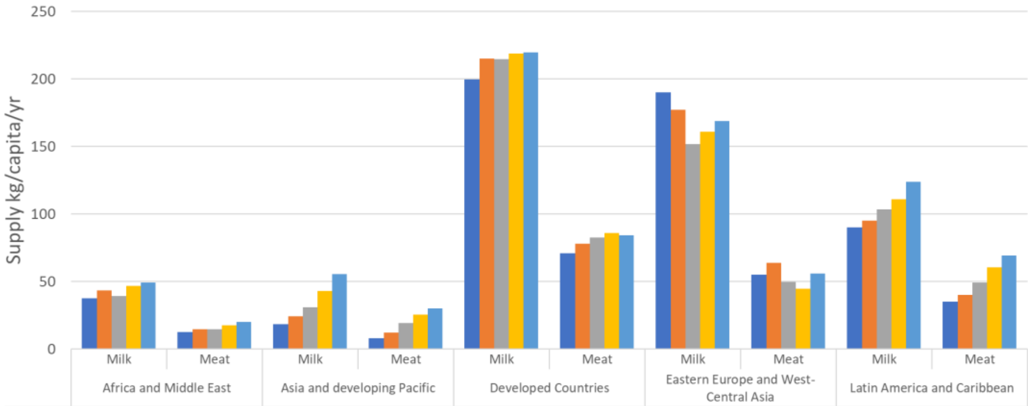


Figure 7.9 Global trends of (a) meat and (b) milk produced. Data sourced from FAOSTAT (2013).

The production of livestock products has increased at a faster rate than that of population and the global supply of milk and meat (kg/capita/yr) increased by 14 and 37% respectively between 1970 and 2013. Although there are large regional differences in the absolute quantity of meat, and particularly milk, consumed per capita, temporal trends are consistent; all regions (except for Eastern Europe and West-Central Asia) showing increases since 1970's. However, since 1990 both milk and meat consumption in Developed countries has remained static.



	Africa and Middle East – Milk	Africa and Middle East – Meat	Asia and developing Pacific – Milk	Asia and developing Pacific – Meat	Developed Countries – Milk	Developed Countries – Meat	Eastern Europe and West-Central Asia – Milk	Eastern Europe and West-Central Asia – Meat	Latin America and Caribbean – Milk	Latin America and Caribbean – Meat
1970	37.68	12.38	18.43	8.08	199.76	71.02	190.19	54.98	89.95	35.00
1980	43.28	14.75	24.00	12.02	215.16	78.11	177.19	63.65	95.15	39.87
1990	39.13	14.71	30.94	19.12	214.68	82.46	151.71	49.52	103.16	49.06
2000	46.49	17.31	43.09	25.59	218.55	85.71	160.96	44.47	110.66	60.62
2010-13	49.30	19.87	55.34	30.04	219.56	84.27	168.90	56.03	123.65	69.36

Figure 7.10 Global trends of weighted (for population) mean supply of milk and meat kg/capita/yr. Data sourced from FAOSTAT.

Changes in agriculture and bioenergy demands

Changes are manifested as expansion or contraction of agricultural land as well as in changes of land management intensity, landscape elements, agricultural land use activity, and specialization/diversification. However, the difference between increase in area and increase in intensity is not always clear, and likewise for decrease in intensity and decrease in agricultural land area. An analysis of 137 studies indicated that agricultural land use in Europe has changed considerably in the last decades with 76 cases of intensification and 143 cases of disintensification (Van Vliet et al. 2015). Economic, technological, institutional and location factors were frequently identified as underlying drivers, while demographic drivers and sociocultural drivers were mentioned less often. Farmers were very important as moderators between underlying drivers and manifestations of agricultural land use change. Major land use change trajectories were related to globalization of agricultural markets, the transition from a rural to an urban society, and the shift to post-socialism in central and Eastern Europe.

Technological changes are also important drivers in agricultural land use. A significant example is the expansion of soybean in Brazil. In a few decades, soybean evolved from being a localized crop (restricted to regions with long photoperiods) to being the most cultivated crop in the country (Abrahão and Costa, 2018). This was due to the development of new varieties with fewer photoperiod limitations that, in the 1980s, gradually made double-cropping possible in central–northern Brazil by lengthening the planting period and the use of a larger portion of the rainy season. As a result, extensive areas of native vegetation were converted in the Brazilian Cerrado and Cerrado-Amazon transition.

In Asia, technological development in change in agriculture since the 1960s led to significant improvements in yields of traditional crops and the composition of agricultural output of developing Asia has shifted from traditional to high-value products (Briones and Felipe, 2013). Agriculture is the largest employer in developing Asia but not the largest sector in any Asian country by GDP. Both agricultural labor productivity and land productivity in Asia have grown faster than in other developing regions (Briones and Felipe, 2013). The analysis

of agricultural datasets from 1960–2015 in South and Southeast Asia countries points out to both agricultural expansion as well as intensification (Vadrevu et al. 2018). Increasing trends are observed in area harvested, food production, cereal yield, nitrogenous fertilizer consumption, and irrigated areas from 1960–2014.
The production of biomass for energy has emerged as a controversial driver of land-use change (Strapasson et 2017). However, the analysis of country-level data (1961–2011) indicated that the area used for bioenergy was shown to be relatively small, but formed a substantial contribution (36%) to net agricultural expansion in the most recent period. Nevertheless, in comparison to dietary shifts in animal products, bioenergy accounted for less than a tenth of the increase in demand for agricultural land (Alexander et al. 2015). Increasing deployment of bioenergy in the future in response to climate change mitigation actions might have impacts on agricultural use of land. As bioenergy becomes more valuable, the competition for the land puts pressure on the price of all agricultural commodities, including food although the increase in food prices is strongly correlated with carbon prices (Muratori et al 2016).

Land tenure, land grabbing and green grabbing
Land tenure defines the way people hold, own and enjoy rights to land reflecting local realities in legal and social terms. In most developing countries, land ownerships and land rights are insecure and conflicts arise with the implementation of traditional land-use planning (Chigbu et al 2017). Land tenure security affects land use and outcomes as it underpins landholders' decision-making, which then influences the value of different management decisions (Robinson et al. 2017).

The value of land has shifted from being measured in terms of what its production potential to its increasingly multifunctional, complex, and market transaction values making the identification of best land management practices increasingly difficult (Spalding 2017). Land grabbing is considered a prominent driver of land system change in certain parts of the globe, especially in the Global South whose lands are increasingly perceived as a potential factor of production for the increasing global demand for alternative energy (primarily biofuels), food crops, mineral deposits and reservoirs of environmental services. The unequal global distribution of population growth and the abundance of land resources taken into account, this development will, all other things being equal, increase incentives for cross-national/continental land deals.

Notably, Africa has become an attractive destination for land investments (Mbow 2010) because of its relatively low population density. Land deals in the region involved an area as large as 51 to 63 million ha and in ten of the identified recipient countries the deals ranged from more than 5% (Uganda) to more than 48% (DR Congo) of the agricultural land (Friis and Rennenberg 2010).

The literature on land grabbing in Southeast Asia has grown substantially pointing out to social and ecological impacts (Davis et al., 2015; Leuprecht, 2004; Neef et al., 2013; Scheidel, 2016; Schoenberger 2017).

The concept of green grabbing addresses a sub-set of cases, in which a convergence of environmental aims with processes of land grabbing occurs (Fairhead et al., 2012) drawing attention to the role that 'green' factors in restricting local users' access to land (for example, the role of a large reforestation project in Cambodia as reported by Scheidel and Work 2018).

Human population, behavior and migration
Demography is one key factor in land change. On the global scale, population pressure on land resources has risen as the world population has increased. From 1987 to 2007, global population grew 34% and it is estimated that the population will increase further from approximately 6.8 billion people in 2010 to 9.2 billion in 2050. As a consequence, the

average amount of land per person has decreased from around 7.9 ha in 1900 to around 2 ha in 2005 (Gitay et al. 2007) and further decline to approximately 1.6 ha is predicted to 2050. Population expansion has been the largest driver for agricultural land use change, but dietary changes are a significant and growing driver. Considering country-level data (1961–2011), the production of animal products dominates agricultural land use and land use change over the 50-year period, accounting for 65% of land use change (Alexander et al. 2015). The rate of extensification of animal production was found to have reduced more recently, principally due to the smaller effect of population growth. Future dietary changes will become the principal driver for land use change, pointing to the potential need for demand-side measures to regulate agricultural expansion.

South and Southeast Asia, population pressure together with rapid economic development is causing immense pressure to convert land from forest to agriculture and from agricultural areas to residential and urban uses. Several countries are transitioning from largely agrarian to urban societies due to increased industrialization (Vadrevu et al. 2018). Trends in forest area from 1985–2014 suggest that in Sri Lanka, Pakistan, Nepal and Bangladesh, forest area decreased considerably whereas it is reported to have increased in India and Bhutan (FAOSTAT 2017). In Southeast Asia, Timor-Leste, Myanmar, Indonesia, Cambodia and Brunei, forest area decreased from 1985–2014 whereas it increased in Vietnam, Thailand and Philippines (Vadrevu et al. 2018). However, these changes cannot be attributed only to population growth; as the forest cover in India did not decline in spite of significant population increase.

Migration is also a significant social and economic phenomenon in historic and contemporary societies. Linked to growing mobility and growing human population, the stock of migrants in the world now is greater than at any point in the past, with the dominant flows of people being from rural areas to urban settlements over the past decades (Adger et al. 2015).

Infrastructure (mining, dams, roads, urbanization)

The impact of mining on deforestation varies considerably across minerals and countries. Mining causes significant changes to the environment including also mining infrastructure establishment, urban expansion to support a growing workforce and development of mineral commodity supply chains (Sonter et al. 2015).

The increasing consumption of gold in developing countries, increase in price, and uncertainty in financial markets are associated with induced deforestation by gold mining the Amazon region (Alvarez-Berríos and Aide 2015, Dezécache et al. 2017, Asner and Tupayachi 2017, Espejo et al. 2018). The total estimated area of gold mining throughout the region increased by about 40% between 2012 and 2016 (Asner and Tupayachi 2017). In the Brazilian Amazon, mining significantly increased forest loss up to 70 km beyond mining lease boundaries, causing 11,670 km^2 of deforestation between 2005 and 2015 what represents 9% of all Amazon forest loss during this time (Sonter et al. 2015).

Mining is also a significant driver of deforestation in African and Asian countries. In the Democratic Republic of Congo, the location of the second-largest tropical forest in the world, deforestation related to mining is particularly significant when associated with violent conflicts (Butsic et al. 2015). In India, mining has contributed to deforestation at a district level, and within the minerals considered, coal, iron, and limestone have had the most adverse impact on forest area loss (Rajan, 2019). Gold mining is also identified as a deforestation driver in Myanmar (Papworth et a. 2017).

Roads are one of the most consistent and largest factors in deforestation, particularly, in tropical frontiers (Pfaff et al. 2007, Rudel et al. 2009, Ferretti-Gallon and Busch 2014). Projections of the International Energy Agency projects indicate that by 2050 another 25 million kilometres of paved roads will be built globally. Nine-tenths of these roads will be located in developing nations, mostly in the tropics and subtropics, where the expansion of road networks increases access to remote forests that act as refuges for biodiversity and provide globally important ecosystem services (Campbell et al. 2017).

Unsustainable and illegal logging is one of the main drivers of road construction in tropical forests. (Kleinschroth and Healey, 2017). Besides the clearings associated with the construction of logging road, more severe impacts include increased fire incidence, soil erosion, landslides, and sediment accumulation in streams, wildlife poaching, illicit land colonization, illegal logging and mining, land grabbing, and land speculation (Laurance et al. 2009; Alamgir et al. 2017). Some roads, initially built for logging, become permanent, public roads with subsequent in-migration and conversion of forest to agriculture. Strategic landscape planning should design road networks that concentrate efficient forest exploitation and conserve roadless areas (case study).

Urbanization is one of the most remarkable features of social development and also has effects on forest resources and land use (Unal et al. 2018). In the Amazon rapid urbanization is an ongoing process, as more rural people move to cities in search of better public services and more favourable job opportunities (Becker, 2001, 2004). In the Brazilian Amazonia, urbanization occurred at a more rapid pace than in the country as a whole, increasing from 37% in 1970 to 73% in 2010 (IBGE, 2011).

Studies showed that most rapid urban growth in the region is occurring within cities that are located near rural areas that produce commodities (minerals or crops) and are connected to export corridors (Richards and VanWey, 2015) and that urbanization is not significantly associated with positive changes in human welfare indicators at the regional level (Caviglia-Harris et al. 2016).

Box 7.1 Reducing the Impacts of Rapidly Proliferating Roads on Deforestation

Conservationists are facing a daunting dilemma. Although the number and extent of protected areas has increased markedly in recent decades (Watson et al. 2014), many other indicators reveal that nature is in broad retreat. For example, the total area of intact wilderness is declining rapidly worldwide (Watson et al. 2016), 70% of the world's forests are now less than 1 kilometer from a forest edge (Haddad et al. 2015), the extent of tropical forest fragmentation is accelerating exponentially (Taubert et al. 2018). The most direct and immediate driver of deforestation and biodiversity decline is the dramatic expansion of roads and other transportation infrastructure (Laurance et al. 2014; Alamgir et al. 2017; Laurance & Burgues 2017).

We are witnessing the most explosive era of road expansion in human history. From 2010 to 2050, the total length of paved roads is projected to increase by 25 million kilometres (Dulac 2013). This is occurring both because of massive infrastructure-expansion schemes—such as China's One Belt One Road initiative (Laurance & Burgues 2017; Lechner et al. 2018) and the IIRSA program in South America (Laurance et al. 2001; Killeen 2007)—as well as widespread illegal or unplanned road building (Laurance et al. 2009; Barber et al. 2014).

The net effect can be catastrophic for forests. In Amazonia, 95% of all deforestation occurs within 5.5 kilometers of a road, and for every kilometer of legal road there are nearly three kilometers of illegal roads (Barber et al. 2014). New roads have allowed ivory poachers to invade the greater Congo Basin in recent years, slaughtering two-thirds of all forest elephants (Maisels et al. 2013). More than any other proximate factor, the dramatic expansion of roads is determining the pace and patterns of habitat disruption and its impacts on biodiversity (Laurance et al. 2009; Laurance & Burgues 2017).

Unfortunately, much road expansion is chaotic or poorly planned. Environmental Impact Assessments (EIAs) for roads and other infrastructure are typically too short-term and superficial to detect rare species or assess long-term or indirect impacts of projects (Flyvberg 2009; Laurance & Burgues 2017). Most EIAs are myopic, considering each project in isolation from other existing or planned developments (Laurance et al. 2014). Hence, EIAs alone are inadequate for planning infrastructure projects and assessing their broader environmental, social, and financial impacts and risks (Laurance et al. 2015a; Alamgir et al. 2017, 2018).

A vital tactic for managing the modern infrastructure tsunami is to use large-scale, proactive land-use planning. Approaches such as the "Global Roadmap" scheme (Laurance & Balmford 2013; Laurance et al. 2014) or Strategic Environmental Assessments (Fischer 2007) can be used to evaluate the relative costs and benefits of infrastructure projects, and to spatially prioritize land-uses to optimize human benefits while limited new infrastructure in areas of intact or critical habitats. For example, the Global Roadmap strategy has been used in parts of Southeast Asia (Sloan et al. 2018), Indochina (Balmford et al. 2016), and sub-Saharan Africa (Laurance et al. 2015b) to devise land-use zoning that can help optimize the many risks and rewards of planned infrastructure projects. The great challenge is to use such tools in practice to blunt the environmental impacts of the modern infrastructure tsunami.

7.3. Historical and current trends in GHG emission and removals

7.3.1 Global net GHG flux due to anthropogenic activities

The land is a source and sink of CO_2 and a source of CH_4 and N_2O due to both natural and anthropogenic processes that happen simultaneously. The IPCC has in the past noted that it is impossible with any direct observation to separate direct anthropogenic effects from non-anthropogenic effects in the land sector (IPCC, 2010; IPCC, 2019). The processes responsible for fluxes from land have been divided 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) (IPCC, 2010). 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 (refs). The methodologies range from estimates based on country level statistics and emissions factors, to approaches including complex modelling and remote sensing information, and are described in more detail in the IPCC SRCCL (IPCC, 2019).

We estimate the total global net GHG emissions from AFOLU to be approximately 12.0 ± 2.9 $GtCO_2e^{-1}yr^{-1}$ or around 23% of total global anthropogenic GHG emissions over the period 2007-2016[1] (Table 7.1). This AFOLU flux is the net of anthropogenic emissions of CO_2, CH_4 and N_2O, and anthropogenic removals of CO_2. The estimate is similar to that reported in AR5 and SRCCL. Emissions from AFOLU have remained relatively constant over the past few decades, although their relative contribution to anthropogenic emissions has decreased due to increases in emissions from the energy sector. The emissions estimates remain subject to large uncertainties due to the difficulties in attribution, the different methodologies applied and large uncertainties in the underpinning data.

Broadly following national Greenhouse Gas inventory (GHGI) reporting, the anthropogenic AFOLU flux is separated into: CO_2 net anthropogenic flux from Land Use Land Use Change and Forestry (LULUCF) (due to both change in land cover and land management), also referred to as FOLU in previous IPCC reports (see section 7.3.2); and CH_4 and N_2O emissions from agriculture including biomass burning (see section 7.3.3). In addition to the direct anthropogenic CO_2 net emissions from LULUCF/FOLU, we also present the net flux due to indirect effects i.e. the natural response of land to human-induced environmental change in 7.3.2. As can be seen in Table 7.1, the land provided a natural sink service in removing 11.2 ± 2.6 $GtCO_2\ yr^{-1}$ from the atmosphere during 2007-2016[1].

[1] Note this number is not final as we are getting updated FAOSTAT and EGDAR data sets to use along with the updated GCP data and will then give numbers for the most recent years possible

Table 7.1 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. [Placeholder-to be updated with FAOSTAT and new EDGAR database when available, then will also include latest GCP numbers up to 2018 for the next draft]

		Direct Anthropogenic							
Gas	Units	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 (LULUCF)	**Agriculture**	**Total AFOLU**					
		A	**B**	**C = A + B**	**D**	**E = C + D**	F = (C/E) *100	G	A + G
CO_2[2]	Gt CO_2 y^{-1}	5.2 ± 2.6		5.2 ± 2.6	33.9 ± 1.8	**39.1 ± 3.2**	13%	-11.2 ± 2.6	-6.0 ± 3.7
CH_4[3,8]	Mt CH_4 y^{-1}	19.2 ± 5.8	141.6 ± 42	160.8 ± 43	201.3 ± 100.6	**362 ± 109**			
	Gt CO_2e y^{-1}	0.5 ± 0.2	4.0 ± 1.2	4.5 ± 1.2	5.6 ± 2.8	**10.1 ± 3.1**	44%		
N_2O[3,8]	Mt N_2O y^{-1}	0.3 ± 0.1	8.3 ± 2.5	8.7 ± 2.5	2.0 ± 1.0	**10.6 ± 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**	81%		
Total (GHG)	**Gt CO_2e y^{-1}**	**5.8 ± 2.6**	**6.2 ± 1.4**	**12.0 ± 2.9**	**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 postproduction	Total global food system emissions			
CO_2 (land-use and land-use change)[4]	Gt CO_2e y^{-1}	4.9 ± 2.5							
CH_4 Agriculture[3, 8, 9]	Gt CO_2e y^{-1}		4.0 ± 1.2						
N_2O Agriculture[3, 8, 9]	Gt CO_2e y^{-1}		2.2 ± 0.7						
CO_2 (other sectors)[5]	Gt CO_2e y^{-1}				2.6 – 5.2				
Total[10]	Gt CO_2e y^{-1}	**4.9 ± 2.5**	**6.2 ± 1.4**		**2.6 – 5.2**	**10.8 – 19.1**	**21-37%**		

[1] Estimates are only given until 2016 as this is the latest date when data are available for all gases [Placeholder-to be updated for AR6 Second order draft]; [2] Net anthropogenic flux of CO_2 due to land use change (such as deforestation, afforestation, cropland conversion), and land management including wood harvest and regrowth, as well as peatland drainage and fires, based on two bookkeeping models as used in the Global Carbon Budget and SRCCL. Agricultural soil carbon stock change due to cropland or grassland management are also considered under LULUCF in the National GHGIs and in FAOSTAT, but is not modelled in the bookkeeping models, and only in some Dynamic Global Vegetation Models (DGVMs).

[3] Estimates show the mean and assessed uncertainty of two databases, FAOSTAT and USEPA 2012

[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.

[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. agrifood 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%.

[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.

[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.

[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 CH_4 (30 per AR5), then total anthropogenic CH_4 emissions expressed in CO_2e 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

7.3.2 Anthropogenic (FOLU) and non-anthropogenic fluxes of CO_2

7.3.2.1 Global direct anthropogenic fluxes of CO_2

Anthropogenic land use change and land management resulted in *likely* net global emissions of 5.5 ± 2.6 $GtCO_2$ yr^{-1} for 2009–2018, approximately 14% of total anthropogenic CO_2 emissions) (Friedlingstein et al., 2019). The flux is the mean of two estimates from bookkeeping (carbon accounting) models (Hansis et al. 2015; Houghton and Nassikas 2017). This net flux was predominately due to tropical deforestation emissions (see forest area change Figure 7.11), but also fluxes due to afforestation/reforestation, forest management (e.g. wood harvest) and peatland draining and burning. Houghton and Nassikas (2017) estimated gross FOLU emissions to be 20.2 $GtCO_2$ yr^{-1} (Figure 7.11), with a gross FOLU sink of 15.x $GtCO_2$ yr^{-1}, although these numbers themselves are based on FAOSATAT data of net change in forest area in different countries (Tubiello et al. 2013), so gross deforestation and afforestation regrowth fluxes could be higher (limited evidence, low agreement), indicating the potential for future emissions reduction and sink enhancement. 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).

While the bookkeeping mean global CO_2 net emissions have remained relatively constant, at around 4.8 ± 2.6 $GtCO_2$ yr^{-1} over the past 50 years (Friedlingstein et al, 2019) the individual models each suggest an opposite trend. Different methodological approaches also show different trends (Figure

7.11 Friedlingstein et al, 2019), and while we can explain some of the reasons for this it means we have *low confidence* in the trend in global FOLU CO_2 emissions.

While there is *high certainty* that FOLU activities have resulted in emissions of CO_2 over recent decades there is a wide range of estimates from different methods and approaches (Fig 7.11) (SRCCL, Smith et al. 2014; Houghton et al. 2012; Gasser and Ciais 2013; Pongratz et al. 2014; Tubiello et al. 2015; Grassi et al. 2018). In addition to differences in land-cover data sets, there are different definitions of land-cover type, including forest, 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). 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).

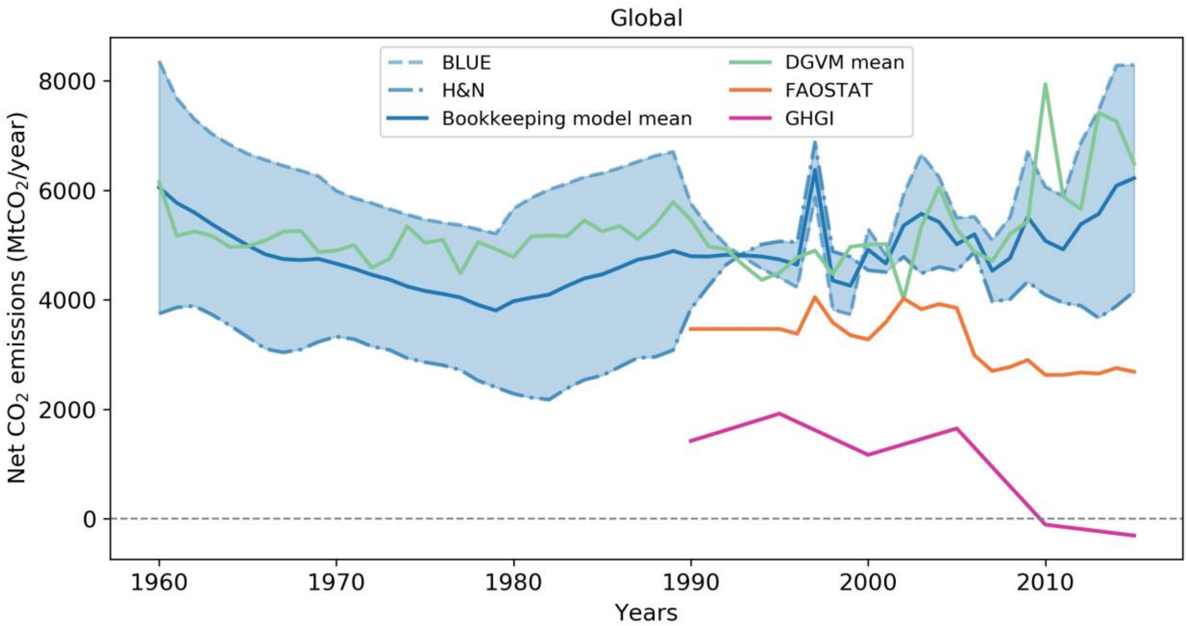


Figure 7.11 Global net CO_2 emissions due to AFOLU from different approaches (in $GtCO_2$ yr^{-1}). Blue line: the mean and individual estimates from two bookkeeping models (Houghton and Nassikas 2017; Hansis et al. 2015). Green: the mean from DGVMs run with the same driving data for the Global Carbon Budget (Friedlingstein et al, 2019) ***[note: currently does not show the with shading showing the ±1 standard deviation range as made the figure hard to read, but could add]*****. Orange: data downloaded from FAOSTAT website (Tubiello et al. 2013) including emissions from peat fires and peat draining. Pink line: National Greenhouse Gas Inventories (GHGI) based on country reports to UNFCCC (Grassi et al. 2018), reporting in many developing countries became more consistent/reliable after this 2005. For more details see notes in table 7.1**

The mean results from an ensemble of Dynamic Global Vegetation models (DGVMs) are similar to the bookeeping models, except during the last decade when they show an increase in emissions to around 7.3 ± 1.8 $GtCO_2$ yr^{-1} (Fig 7.11). The DGVMS model the indirect effect of environmental change on runs with and without land use change, and estimate the net flux due to land use change as the difference between these runs. This approach means they include the Lost Additional Sink Capacity – land that is cropland takes up less carbon that it would have done if it was still forests (as in the no land use run), this "lost sink" appears as apparent emissions in the differencing methodology, adding about 1.5 ± 1.1 $GtCO_2$ yr^{-1}. Some of the DGVMs and the BLUE model use the LUH2 data set for land cover change and wood harvest (Hurtt et al., 2019) based on HYDE 3.2 (Klein Goldewijk et al, 2017), which is in turn based on FAOSTAT cropland area change data. Other DGVMs use the HYDE 3.2 directly data set directly, while Houghton and Nassikas (2017) primarily use FAO FRA forest area change data (FAO 2015). [Placeholder-add fig/table of difference in primary and secondary forest areas over time and wood harvest data from these different data sets]. Other details of model methods and differences in processes are described in the SRCCL and Frieldingstein et al, (2019) (see also Pongratz et al, 2014). 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). There have been advances since AR5 in estimating the GHG effects of different types of forest management (e.g. (Valade et al. 2017). Grazing management has been found to have large effects (Sanderman et al. 2017a), and is not included in most DGVMs (Pugh et al. 2015; Pongratz et al., 2018).

FAOSTAT data is derived from country reported data provided to FAO, and application of an IPCC (2006) Tier 1 type approach of net land area change and change in carbon stocks (Tubielo et al, 2013). It also includes wood harvest and peatland burning and draining. FAO data does not distinguish natural and managed forests, and depending on data provided, may be capturing the indirect effects of environmental change in unmanaged lands if countries include these lands in their reported carbon stock changes (Federici et al, 2015).

There are large differences globally, between estimates of net anthropogenic land-atmosphere fluxes of CO_2 from national GHGIs and from global models (Grassi et al, 2017, 2018). The major reasons have been identified as the inclusion of larger areas of forests as anthropogenic under the "managed land proxy" than is typical in the global models, and the sink in these lands due to the indirect effect of environmental change, that the models treat as non-anthropogenic (Grassi et al, 2018). The reasons for the differences, as well as the implications for the global stocktake and approaches to reconciliation are discussed in more detail in 7.8.

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 differences in spatial resolution, processes included (e.g. Baccini et al include degradation and regrowth), The 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. The satellite-based estimates do not include changes in soil carbon.

7.3.2.2 Natural response of land to environmental change and the net land-atmosphere flux CO_2

The natural response of land to human-induced environmental change (e.g., climate change, and the fertilising effects of increased atmospheric CO_2 concentration nitrogen deposition) on unmanaged and managed lands provided a net flux of -11.7 ± 2.2 $GtCO_2$ yr^{-1} during 2009-2018, a sink of around 29% of global anthropogenic emissions of CO_2 (*robust evidence, high agreement*) (Friedlingstein et al, 2019). Unlike in AR5, where this number was determined as the residual, here (consistent with WGI and SRCCL) it is estimated directly by DGVMs. The land sink has generally increased since 1900. 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). Data from forest inventories around the world corroborate the modelled land sink (Pan et al. 2011).

Climate change 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, there 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). [Placeholder-to be further coordinated with WGI and II].

When combining the anthropogenic AFOLU net source with the non-AFOLU net sink, the total net land-atmosphere flux was -6.2 ± 3.3 $GtCO_2$ yr^{-1} (net sink) during 2009-2018, (*robust evidence, high agreement*) (Friedlingstein et al, 2019). Data from global model is corroborated by invernsion methods based on worldwide atmosphseric measurements of CO_2 giving an range from -4.0 to -8.1 $GtCO_2$ $yr^{-1.}$ (Van Der Laan-Luijkx et al., 2017; Rödenbeck, 2005, Rödenbeck et al., 2018; Chevallier et al., 2005).

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. 2018b).

7.3.2.3 Regional direct anthropogenic fluxes of CO_2

Regional CO_2 fluxes are shown in Figure 7.12 [Placeholder-there are ongoing efforts to explain some of the regional differences between methods and regional forest are and wood harvest data that match these 10 regions that will be added for the next draft]. While forest area continues to decline in some tropical regions, some countries and regions have seen increases in forest area such as India, China, the USA and Europe (FAO-FRA 2015 - Placeholder-FRA 2020 should be out for next draft, also draw from NYDF global assessment 2019). 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 (SRCCL, Chapter 1).

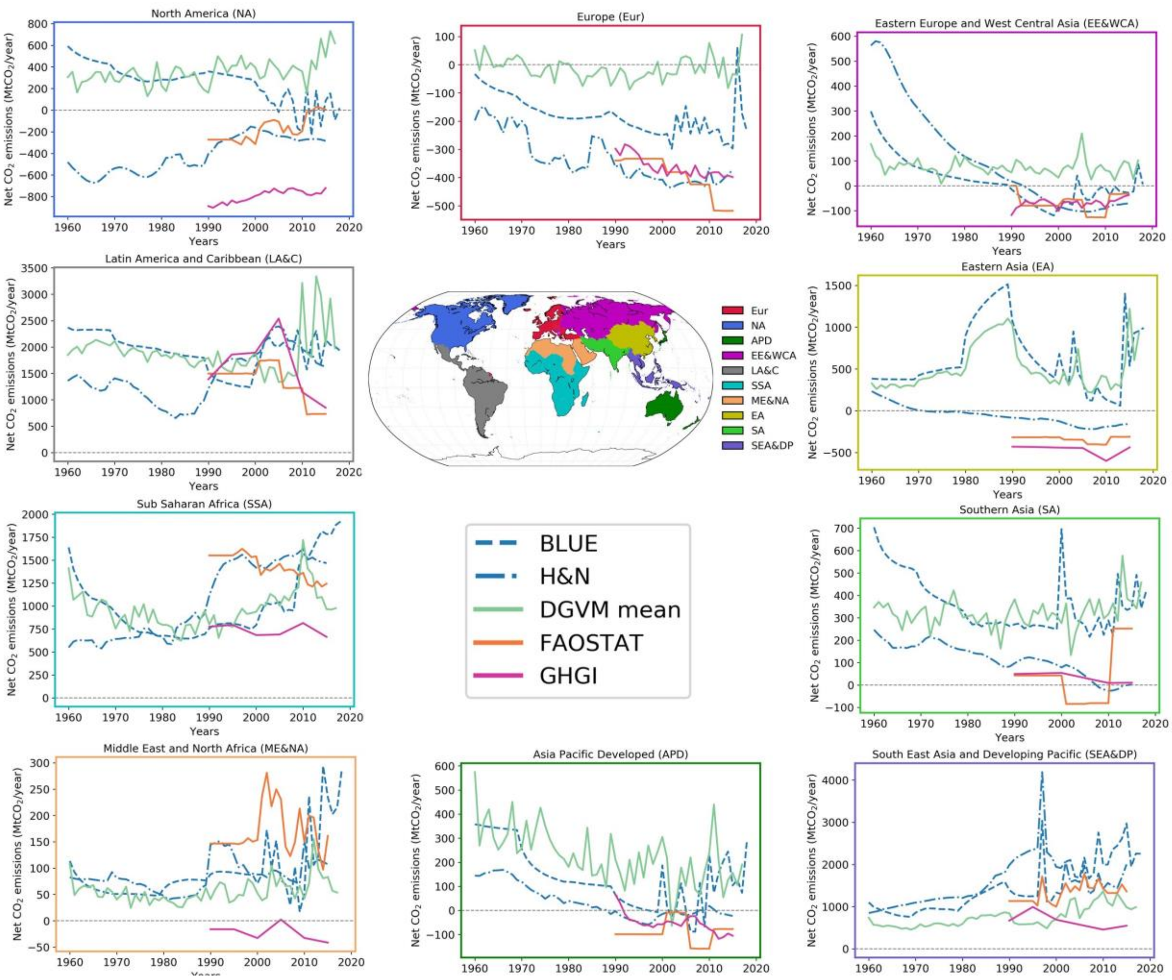


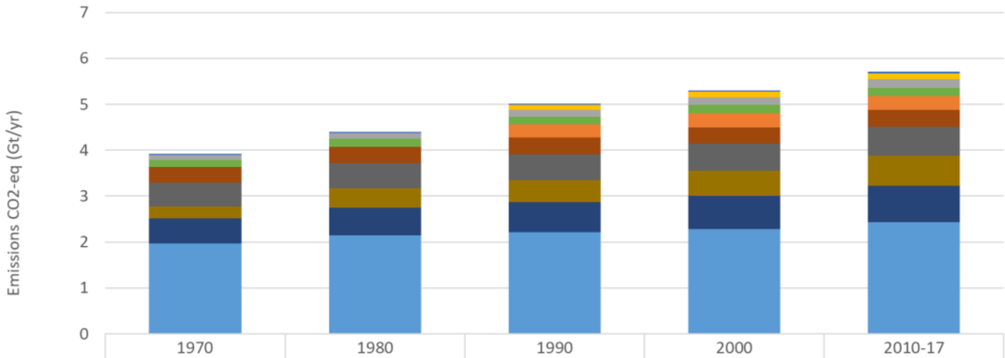
Figure 7.12 Regional trends in net AFOLU CO_2 flux from a range of different approaches (in $GtCO_2\ yr^{-1}$). Blue line: the mean and individual estimates from two bookkeeping models (Houghton and Nassikas 2017; Hansis et al. 2015). Green: the mean from DGVMs run with the same driving data for the Global Carbon Budget (Placeholder-*data actually from Le quere 2018 but will be updated with Firedlingstein et al 2019 when regional data available*). Orange: data downloaded from FAOSTAT website (Tubiello et al. 2013) including emissions from peat fires and peat draining. Pink line: National Greenhouse Gas Inventories (GHGI) based on country reports to UNFCCC (Grassi et al. 2018), reporting in many developing countries became more consistent/reliable after 2005. For more details see notes in table 7.1

7.3.2.4 Agricultural emissions of CH_4 and N_2O

Comprehensive global time series data for Agricultural non-CO_2 emissions from different sectoral activities are only available from a limited number of sources that adopt similar simplified emission-factor type approaches. FAOSTAT (Tubiello et al. 2013) and EDGAR (Ref)[2] utilise much of the same input data and thus, unsurprisingly, the results from these two sources are generally consistent. Emissions data are also available from national GHG inventory reports to the UNFCCC, but the variability in methods and definitions applied, and the data gaps for many developing countries, make them a less comprehensive and consistent data source for global and regional aggregations, and they are only available since 1990 in some countries. Based on data from GHGI aggregated for the IPCC SRCCL agricultural emissions appear to be similar to the values reported by EDGAR and FAOSTAT. The FAOSTAT database provides non-CO_2 GHG emissions from agriculture at global, regional and national level and are based on FAOSTAT activity data and IPCC Tier 1 approaches. FAOSTAT

[2] Note we are expected updated EDGAR data to 2018, and will also update FAOSTAT data.

reports emissions from enteric fermentation, manure deposited on pasture, synthetic fertilizers, rice cultivation, manure management, crop residues, biomass burning, and manure appilied to soils. Enteric fermentation, biomass burning and rice cultivation are reported separately under IPCC inventory guidelines, with the remaining categories aggregrated into 'agricultural soils'. FAOSTAT estimates of global trends in total GHG CO_2e emissions from agricultural activities between 1970 and 2017 are shown in Figure 7.13.



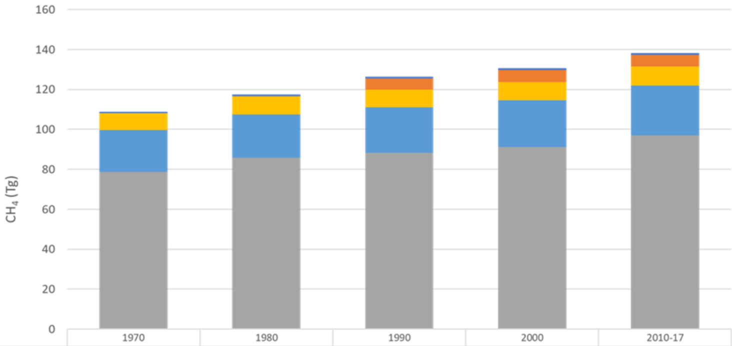
	1970	1980	1990	2000	2010-17
Total	3.918	4.405	5.021	5.298	5.706
Burning - Crop residues	0.026	0.027	0.028	0.029	0.033
Cultivation of Organic Soils			0.116	0.118	0.122
Crop Residues	0.113	0.135	0.150	0.167	0.199
Manure applied to Soils	0.150	0.169	0.168	0.168	0.179
Burning - Savanna			0.285	0.317	0.295
Manure Management	0.329	0.358	0.359	0.357	0.375
Rice Cultivation	0.524	0.543	0.566	0.583	0.622
Synthetic Fertilizers	0.266	0.433	0.485	0.556	0.657
Manure left on Pasture	0.540	0.597	0.655	0.722	0.797
Enteric Fermentation	1.968	2.144	2.209	2.282	2.426

Figure 7.13 Agricultural non-CO_2e emissions per decade by sub-sector since 1970. For the agricultural sub-sectors, emissions are shown for separate categories, based on FAOSTAT. Emissions from crop residues, manure applied to soils, manure left on pasture, cultivated organic soils, and synthetic fertilizers are typically aggregrated to the category 'agricultural soils' for IPCC reporting. Data sourced from FAOSTAT (2013).

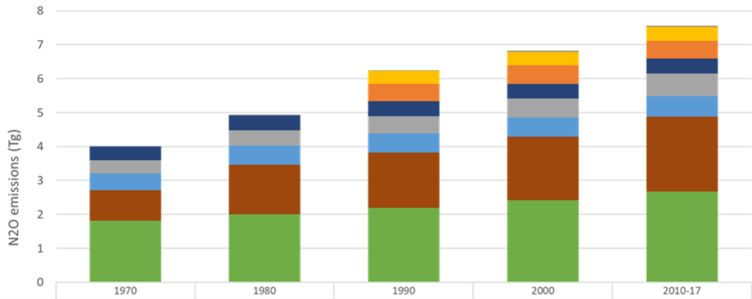
Annual total non-CO_2 GHG emissions from agriculture in 2017 are estimated to be 5.71 $GtCO_2$e compared with 3.92 $GtCO_2$e in the 1970s. In the period 2010-2017 the largest single source of emissions was enteric methane 2.43 $GtCO_2$e (43%) with livestock sourced manures collectively contributing a further 1.35 $GtCO_2$e (24%). Emissions from rice production comprise 0.62 $GtCO_2$e (11%) while those from synthetic N fertiliser application add a further 0.66 $GtCO_2$e (18%). Emissons from all categories have increased in the decades since 1990 with emissions from enteric methane, livestock sourced manures, rice cultivation and synthetic N fertiliser applications rising by 10, 37, 10 and 36% respectively.

FAOSTAT also provide emissions projections for both 2030 and 2050 by emissions category. Total CO_2e emissions are projected to increase by a further 1.18 Gt (18%) above 2017 values. Emissions from livestock activities and synthetic fertiliser are projected to the largest increases (20-30%) while emissions from rice production are projected to fall by 7%.

At the global scale methane emissions have risen from 109 $Tgyr^{-1}$ in the decade beginning in 1970 to 138 $Tgyr^{-1}$ in the 2010-2017 period (27% increase); in the 2010-2017 period enteric methane emissions were 70% of total methane emissions with emissions from rice making up 18% (Figure 7.14, Top graph). Since the 1970s N_2O emissions have risen from 4 $Tgyr^{-1}$ to 7.5 $Tgyr^{-1}$ (88% increase); in the 2010-2017 period 49% of N_2O emissions came from livestock sourced manures while 29% came from synthetic N applications (Figure 7.14, Bottom graph)



	1970	1980	1990	2000	2010-17
Total	108.85	117.38	126.35	130.71	138.28
Burning - Crop residues	0.79	0.83	0.86	0.90	1.02
Burning - Savanna			5.47	6.10	5.67
Manure Management	8.36	9.10	9.02	9.11	9.65
Rice Cultivation	20.97	21.71	22.62	23.33	24.89
Enteric Fermentation	78.73	85.74	88.38	91.26	97.05



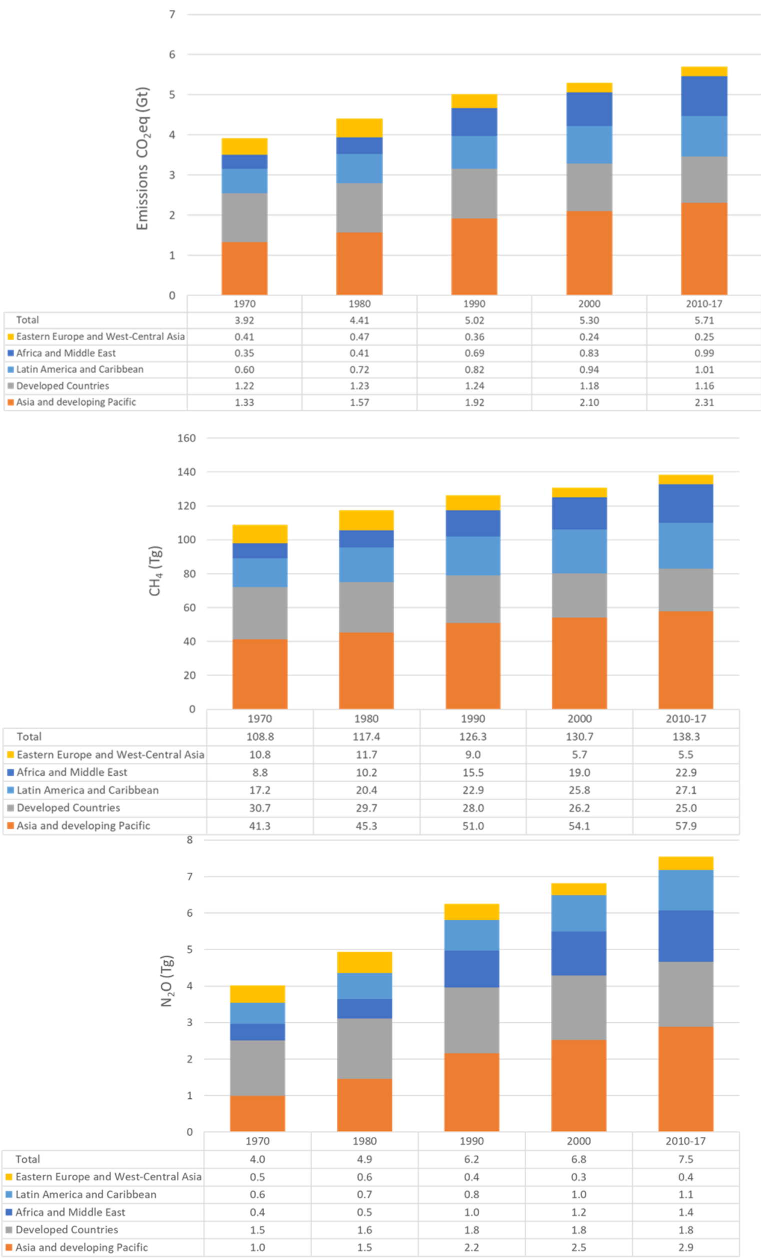
	1970	1980	1990	2000	2010-17
Total	4.015	4.936	6.249	6.814	7.548
Burning - Crop residues	0.021	0.022	0.022	0.023	0.027
Cultivation of Organic Soils			0.391	0.396	0.411
Burning - Savanna			0.497	0.552	0.513
Manure Management	0.403	0.439	0.449	0.432	0.447
Crop Residues	0.380	0.453	0.502	0.560	0.668
Manure applied to Soils	0.505	0.566	0.565	0.563	0.602
Synthetic Fertilizers	0.894	1.452	1.627	1.864	2.206
Manure left on Pasture	1.812	2.005	2.196	2.422	2.673

Figure 7.14 AFOLU emissions for the last five decades detailed per individual GHGs. (Top) CH_4 and (Bottom) N_2O. For the agricultural sub-sectors, emissions are shown for separate categories. Data sourced from FAOSTAT

Regional trends in agricutlural Non-CO_2 emissions

Just under half of all CO_2e emissions are estimated to arise from the Asia and developing Pacific region (40% of annual emissions in 2010-2017). Developed countries contributed 20% with Africa and Middle East contributing 17% and Latin America and Caribbean 18% (Figure 7.15, Top graph). These percentage figures are essentially the same for both CH_4 and N_2O emissions (Figure 7.15, Middle and Bottom graph).

Developed countries and those in the Eastern European and Western Asia regions have recorded a decline in CH_4 emissions since the 1970's and a stabilisation in N_2O emissions since the 1990's resulting in an overall deline in total CO_2e emissions of 6% and 41% respectively. In contrast the other three regions have recorded rises in both CH_4 and N_2O emissions such that total CO_2e emissions are currently 183%, 74% and 68% above their 1970-1979 average for Africa and Middle East, Asia and developing Pacific and Latin America and Caribbean respectively.



	1970	1980	1990	2000	2010-17
Total	3.92	4.41	5.02	5.30	5.71
Eastern Europe and West-Central Asia	0.41	0.47	0.36	0.24	0.25
Africa and Middle East	0.35	0.41	0.69	0.83	0.99
Latin America and Caribbean	0.60	0.72	0.82	0.94	1.01
Developed Countries	1.22	1.23	1.24	1.18	1.16
Asia and developing Pacific	1.33	1.57	1.92	2.10	2.31

	1970	1980	1990	2000	2010-17
Total	108.8	117.4	126.3	130.7	138.3
Eastern Europe and West-Central Asia	10.8	11.7	9.0	5.7	5.5
Africa and Middle East	8.8	10.2	15.5	19.0	22.9
Latin America and Caribbean	17.2	20.4	22.9	25.8	27.1
Developed Countries	30.7	29.7	28.0	26.2	25.0
Asia and developing Pacific	41.3	45.3	51.0	54.1	57.9

	1970	1980	1990	2000	2010-17
Total	4.0	4.9	6.2	6.8	7.5
Eastern Europe and West-Central Asia	0.5	0.6	0.4	0.3	0.4
Latin America and Caribbean	0.6	0.7	0.8	1.0	1.1
Africa and Middle East	0.4	0.5	1.0	1.2	1.4
Developed Countries	1.5	1.6	1.8	1.8	1.8
Asia and developing Pacific	1.0	1.5	2.2	2.5	2.9

Figure 7.15 Emissions from AFOLU for each region using data from FAOSTAT (2013). Top: total emissions ; Middle: CH_4; Bottom: N_2O.

Need a section on emissons per unit of product and emissions per capita – unlikely that we can get emissions per unit of product on a regional basis but we can get a global picture from FAO (update graph in AR5) plus we can simply use the FAOSTAT emissions data with regional populations to derive a per capita figure for each region over time.

7.3.2.5. Short lived climate forcers and biophysical effects

[Placeholder-For SOD-summarise and update from the SRCCL and WGI]

7.4. Policy and socioeconomic contexts related to historical trends

7.4.1 Historical Trends

Several policy instruments have been deployed over the years to increase carbon storage in the world's ecosystems. These policies range from land use planning to regulatory approaches. Table 7.2 presents a list of various alternatives that have been applied in different contexts in the past several decades.

Table 7.2 Policies that could be used to reduce emissions or increase carbon stored on the landscape.

Policy	***Example***	***Region***	***Scale***	***Effect on carbon***	***Citations***
Land Use Planning [micro/macro zoning]	Land use regulations that prevent conversion of standing forest to developed uses	State of Oregon, USA	State-wide	Increased carbon in forests	(Cathcart et al., 2007)
Land Use Planning [micro/macro zoning]	Land use planning and regulation	Guyana	Country-wide	Increased carbon in forests	(Roopsind et al., 2019)
Regulations	Prohibit the conversion of land to soy production	Brazilian Amazon	Region-wide	Reduced Emissions from deforestation	(Nepstad et al., 2014; Soterroni et al., 2019)
Certification [value-chain]	Soy supply chain	Brazilian Amazon	Region-wide	Reduced Emissions from deforestation	(Nepstad et al., 2014)
Certification [value-chain]	FSC certification of timber harvesting in concessions	Sumatra, Indonesia	Region-wide	No change compared to non certified forests	(Griscom et al., 2014)
Certification [value-chain]	FSC certification of timber harvesting in Ejidos	Mexico	Country-wide	No change in deforestation	(Blackman et al., 2018)
Community forest management	CFR in Maya Biosphere Reserve	Guatemala	Region-wide	Reduced deforestation	(Blackman, 2015; Fortmann et al., 2017)
Protected Areas	Protected areas and parks	Costa Rica	Country-wide	Reduced deforestation, but potential for spillovers/leakage	Andam et al. (2008) Robalino et al. (2017)
Subsidies/PES	Conservation Reserve Program	United States	Country-wide	Increased grassland and forestland, but spillover/leakage = 20%	Pfaff and Robalino (2017) Haight et al. (2019) Wu (2000)
Subsidies /PES	Pagos por servicios ambientales (PSA)	Costa Rica	Country-wide	Reduced deforestation	Robalino and Pfaff (2013)
Subsidies/PES	National Conservation Policies	China	Country-wide	Reduced deforestation, forest conservation and afforestation	Ouyang et al. (2016)
Governance	Enforcement of Brazilian Forest Code	Brazilian Amazon	Region-wide	Reduced Emissions from deforestation	(Nepstad et al., 2014)
Finance	REDD+ payments for forest conservation	Brazil	Project	Reduced Emissions from deforestation	Simonet et al. (2018)

Before 2007, many carbon-specific efforts focused largely on developing protocols and methodologies in afforestation, reduced impact logging, methane capture in farming, and soil carbon.

Many projects were funded through the Clean Development Mechanism, which is estimated to have sequestered an additional 11.3 million tons of CO_2 in forests and 21.8 million tons of CO_2 in agriculture through 2015 (Table 7.3).

Numerous protocols for carbon sequestration in agriculture and forestry have been developed by various groups, including the California Climate Action Reserve (CAR), American Carbon Registry, Verra (formerly Verified Carbon Standard), Gold Standard, or Plan Vivo. As an example, CAR has been developing forest carbon and agricultural carbon protocols continuously since the early 2000s. The forest carbon protocol is in its 5th version, with the latest revision occurring in 2019 (see http://www.climateactionreserve.org/how/protocols/forest/), and there are several protocols for agricultural emission reductions or offsets.

These programs provide credits that can feed into regulatory programs, such as the California cap and trade program, or voluntary markets (Hamrick and Gallant, 2017). Voluntary markets have continued to grow, and it is estimated that over 100 million tons CO_2 have been stored in projects that have been sold on voluntary carbon markets (Table X). The largest share of annual sales of voluntary carbon credits occurs in Latin America, followed by Africa, Asia and North America. Europe and Oceania have smaller voluntary carbon markets. Most of the volume lies in avoided deforestation projects, with some volume accruing to afforestation and improved forest management. Prices for these offsets in the period 2014-2016 ranged from \$4.90 to \$5.40 per ton CO_2 (Hamrick and Gallant, 2017). Prices are higher in Europe, North America, and Oceania.

Some countries are in the process of developing carbon trading mechanisms that could include forest or agricultural carbon offsets. The US State of California and the country of Australia have already begun implementing carbon trading systems that include offsets from forests and agriculture. In California, 137 million tons CO_2 have been sequestered in forestry and agricultural projects between 2007 and 2018. All forest projects used as offsets in California currently are located in the US, but the California Air Resources Board just adopted their tropical forest carbon standard, potentailly allowing for avoided deforestation projects from outside the US to enter the California market (https://ww3.arb.ca.gov/cc/ghgsectors/tropicalforests/ca_tropical_forest_standard_english.pdf).
Australia and New Zealand also have regulatory markets for buying and selling carbon credits, including provisions for land-based credits. New Zealand now treats carbon storage in forests not only as a potential sink for carbon but also as a source when harvesting occurs.

After the COP meeting in Bali, significant effort shifted to develop methodologies to reduce deforestation and forest degradation (REDD+) According to Simonet et al. (2018), nearly 65 million hectares have been enrolled in REDD+ type projects funded through a variety of mechanisms, including the UNDP, the World Bank Forest Carbon Partnership Facility, and bi-lateral agreements between countries (e.g., Norway). Quantification of the carbon benefits of this funding is starting to emerge with a recent paper illustrating that REDD+ efforts in Guyana saved 12.8 million tons of CO_2 emissions. In addition, a number of countries have claimed REDD+ successes in terms of emissions reductions due to reducing deforestation or reducing forest degradation (see UNFCCC Biennial Reviews). Based on analysis of existing biennial reviews, we estimate that more than 7.5 Gt CO_2 of emissions, or 0.7 Gt CO_2 yr^{-1}, have been avoided through reducing deforestation and forest degradation (Table 7.3). The largest share of these emissions reductions have occurred in Latin America.

Table 7.3 Achieved emissions reductions achieved in AFOLU so far.

Name of fund	Tons reduced to date (12/31/2018)	Time period	Years	tCO_2yr^{-1}
CDM-forest	11,328,560	2007-2015	9	1,258,729
CDM-agriculture	21,835,793	2007-2015	9	2,426,199
REDD+ (Guyana)	12,800,000	2010-2015	6	2,133,333
REDD+ Brazil	6,894,522,601	2006-2017	12	574,543,550
REDD+Indonesia	244,900,000	2013-2017	5	48,980,000
REDD+Argentina	165,172,705	2014-2015	3	55,057,568
REDD+Others	162,784,210	2010-2017	8	20,348,026
GCF				
Voluntary Market	100,370,000	2010-2016	7	14,338,571
IKI Bonn Challenge (Restoration)				
Agriculture other				
Australia ERF	33,685,951	2012-2018	7	4,812,279
California	137,168,294	2007-2018	12	11,430,691
New Zealand Carbon Trading	55,090,497	2010-2017	8	6,886,312
TOTAL	7,839,658,611	2007-2018	12	653,304,884

Additionality, Permanence and Leakage

The issues of additionality, permanence and leakage have been widely discussed in forestry and agricultural offset literature (see Murray et al., 2007). Additionality has largely been addressed as a counterfactual issue. Additionality is established when the project developers project the baseline future conditions for the site, and present evidence that those conditions would likely have held in the future. To do this, they can use either using historical management actions, timber management plans and documents, or other sources of information. It is plausible to test for additionality after the fact using impact analysis techniques which have been used relatively widely in the literature in recent years to assess a range of policies (e.g., Andam et al., 2008; Blackman, 2015; Fortmann et al., 2017; Roopsind et al., 2019).

Permanence requirements require the project to maintain quantifiable carbon on the site over an extended period. When considering permanence, the critical issue is the crediting of the carbon stocks. As shown in Van Kooten et al. (1995), if the carbon gains are fully credited when they occur, then project developers should relinquish those credits, less any permanent storage in wood products, when the carbon is lost of the site due to disturbance (harvest, fire, etc.). On the other hand, if the credits are only partially paid in any given year, e.g., they are rented, then project developers may not need to relinquish their credits (see Favero et al., 2019). Most project systems to date appear to have taken first approach, assuming that carbon gains are fully credited during the project period, so that when losses occur, the project partners are required to make up the difference.

That is, most project guidelines provide incentives for project managers to maintain forest carbon stock over time by requiring them to establish a reserve requirement. A reserve requirement is an insurance pool of credits maintained by the developer or someone else. For example, The Climate Action Reserve (CAR) protocol for forests requires carbon to remain on the site for 100 years. The carbon on the site will be verified at pre-determined intervals over the life of the project. If carbon is diminished on a given site, the credits for the site have the relinquished and the project developer has to use credits from their reserve fund (either other projects or purchased credits) to make up for the loss.

Permanence and the carbon neutrality of biomass energy are similar issues in that they concern the treatment of carbon emissions from forests at the time of harvest. Forest carbon sequestration projects concern carbon that either already has been credited to a developer, or carbon in a project that is being developed. This carbon receives protections in that the crediting organization or agency has a reserve requirement for forest projects. Forest biomass energy is carbon that is stored in forests, typically with no carbon protocol protections. Current forest biomass protocols (EU, US states, etc.) have acknowledged this difference by requiring forest biomass to be sustainably harvested in systems that are replanted after harvest for biomass energy.

Leakage associated with timber harvesting has been handled in various ways. Murray et al., (2004) suggests that it can range from 10% to over 90% in the US, while Sohngen and Brown (2004) founds effects in the 20-50% range in the tropics of Latin America. The Climate Action Reserve (CAR) assumes it is 20%. One of the voluntary protocols (Verra) uses specific information about the location of the project to calculate a location specific leakage factor.

The literature suggests that there are two economic pathways for leakage (see Roopsind et al., 2019), either through a shift in output price that occurs when outputs are affected by the policy or program implementation, as described in (Gan and McCarl, 2007; Murray et al., 2004b; Sohngen and Brown, 2004b; Wear and Murray, 2004), or through a shift in input prices and markets, such as for labor or capital, as analyzed in Alix-Garcia et al. (2012), Andam et al. (2008), Fortmann et al. (2017), and Honey-Rosés et al. (2011). Estimates of leakage have generally been larger in the first type of projects discussed above (e.g., output market leakage) than in the input market leakage.

The main response to potential leakage in projects has been to assume it exists, based on the literature above, and adjust the carbon credit issues for the project accordingly. Few studies attempt to measure leakage, as in Roopsind et al. (2019), who found that it was not statistically observable in the case of the Guyana-Norway REDD+ program. Roopsind et al. (2019), however, acknowledged that leakage could occur, and suggested that policies need to focus on continuing to draw more forests under carbon policy initiatives in order to reduce the impacts of leakage.

Box 7.2 The challenge: growing disconnect between the climate crisis and historical solutions

Any effort to develop practical ideas for improving climate friendly policies in the forestry and agriculture sectors must confront a troubling conundrum that vexes problem focused forest and conservation environmental scientists: despite a plethora of policy tools and innovations engaging the private sector, civil society, and global governance in the last quarter century, the state of the planet's environment is rapidly deteriorating. From the climate crisis that threatens unparalleled catastrophic ecological impacts, to related ongoing rapid extinctions of flora and fauna around the word, in large part owing to forest biodiversity loss, the overwhelming amount of scientific evidence tells us that humans are not doing enough, at almost any scale, to significantly dent these ecological crises. These effects in turn, have profoundly impacted the livelihoods and cultures of forest dependent peoples.

Yet, the world has never seen such an impressive scale of policy experimentation and instruments from which to choose. These include the development of a rich suite of innovative "finance and market" (FMD) driven interventions, ranging from international financing mechanisms such as the Global Environmental Facility (GEF) to climate bonds, to a plethora of non-state market driven (NSMD) eco-labeling programs governing commodity production, to corporate social responsibility initiatives (Park 2007, Auld, Bernstein, and Cashore 2008, Clapp 1998). Taken together, a generation of research and scholarship has indeed found some positive accounts regarding the implementation of FMD tools, but also strong empirical evidence that they usually end up falling short of what creators and supporters had hoped (Grabs Forthcoming 2020, Buntaine, Parks, and Buch 2017). From deforestation and biodiversity loss that is primarily responsible for the one million species being threatened with extinction to growing inequality and the marginalization of local peoples, to the accelerating climate crisis, transnational actors are increasingly frustrated by the profoundly troubling accelerations of many of the problems FMD tools were designed to ameliorate (United Nations 2019, Piketty 2015, Intergovernmental Panel on Climate Change 2019).

However, instead of responding to this evidence by eschewing FMD interventions, creators and supporters end up going back to the drawing board in an attempt to either tinker with, or create new, FMD policy tools. Indeed, these policy experiments have largely been advocated by the international community, including previous IPCC reports, as offering promise in making a difference when, historically, they have been unable to address the crises they were created to ameliorate, despite purposeful intensions. Hence, any effort to identify practical tools, as I do below, must be consistent with analysis about how to understand, and overcome, these conundrums.

The explanation: shifting world view, power, and "good governance" norm complexes

Cashore (2018) recently explored three related explanations for the causes of this policy innovation/environmental degradation paradox (van der Ven, Rothacker, and Cashore 2018). First, I argue that there has been a tendency of (US based) professional environmental institutes and schools, created during the first two waves of environmentalism, to have slowly shifted from an emphasis on "bioenvironmentalist" world views to largely reinforcing anthropogenic needs in general, and utility in particular, through the domination of market-liberal world views and likeminded institutionalist perspectives (Cashore 2018). These trends are reinforced through their graduates, who are trained to emphasize market friendly solutions, and who populate international agencies, businesses championing sustainable development, and international environmental groups. The result has been a generation of students trained to treat cost benefit analysis and "optimization" strategies as providing the answer as to whether, rather than how, the climate crisis might be ameliorated. Second, and related, powerful interests have sought to emphasize market solutions over other policy innovations by arguing they are more effective and efficient. Resulting "feasibility" logics further reduce consideration of regulatory approaches in favour of finance and private governance initiatives. Third, and as a result, a "good governance norm complex" has come to treat a range of procedural goals, such as transparency, inclusion, and balance, and substantive outcomes such as livelihoods, environment and economic growth as synergistic with each other (Cashore and Nathan 2019). Evidence that these goals are inversely related is treated as a policy design challenge rather than confronting the inherent paradox of the norm complex itself. Since the "good governance norm complex" is cognitively engrained in the minds of policy makers and multiple levels, rendering this phenomenon explicit, and understood, is critical if specific problems championed by the Sustainable Development Goals, themselves the product of "better designed" Millennium Development Goals, are to effectively implemented.

The policy analysis solution for Climate Friend Agriculture and Forestry Policy

As a corrective, three related solutions are offered.

Explicit problem conceptions

First, those designing, and justifying, policy innovations must be required to distinguish the problem in question according to 4 different conceptualizations: Type 1 win/win collective action such as "tragedies of the commons"; Type 2 win/lose optimization that prioritizes utility as a moral philosophy; Type 3 win/lose compromise orientations in which tradeoffs are internalized, and Type 4 win/lose prioritization in which, to be solved, the specific problem in question must be given priority over others (Cashore and Bernstein 2018).

Forward looking policy design

Second, policy designers must be required to project forward multiple causal change processes their innovations might be expected to unleash (Cashore and Goyal 2019). This requires the international policy design community to spend much more effort tapping into the "anticipatory policy design" and "policy mix" work being advanced among the comparative public policy scholarly community (Howlett 2019a, b, Howlett, Mukherjee, and Rayner 2018, Howlett 2018, Howlett, Mukherjee, and Rayner 2014, Cashore 2019). **Box 7.2, Table 1. Elements of a policy**

This framework expanded Hall's (1993) first, second, and third order typology of policy change into six "elements or components" (Cashore and Howlett 2007: 535).

		Policy Level		
		High level abstraction (policy orientation)	**Operationalization (program)**	**On the ground specification (measures)**
Policy Content	**Policy ends (aims)**	**GOALS** **What general types of ideas govern policy development?** e.g. environmental protection, economic development, Social cohesion	**OBJECTIVES** **What does policy formally aim to address?** e.g. saving wilderness or species habitat, reducing greenhouse gas emissions	**SETTINGS** **What are the specific "on the ground" requirements of the policy?** e.g. size of protected areas, level of carbon tax
	Policy means (instruments)	**INSTRUMENT LOGIC** **What general norms guide policy instrument preferences?** e.g. coercive "command and control", voluntary, markets, neoliberal norms	**TOOLS** **What types of instruments are utilized?** e.g. tax incentives, loans, public enterprise, cap and trade carbon markets	**CALIBRATIONS** **What are the specific ways in which the instrument is applied?** e.g. qualification for tax incentives, rules governing cap and trade markets such as specifics on leakage, allocation of resources and approach to enforcement

Source: Adapted from Cashore and Howlett (2007: 536) and iterations since then

Cashore and Howlett's (2007) application of these six "elements or components" uncovered historical patterns of endogenous and exogenous policy development that were inconsistent with Hall's theories of policy learning and policy change. For these reasons, this table is now fostered greater conceptual and empirical work about measuring policy change, and how to conduct forward looking policy design including identification of policy mixes. This is important, since decisions made within each cell often affect, inversely, decisions made in other cells. Hence, generalizations and attention to policy design that fail to take into account for (both positive and negative interactions) across the cells help explain frustrations of policy designers in developing durable and meaningful climate friendly policies. The simultaneous desire to produce clear and pithy "lessons learned" recommendations to government officials tragically reinforces these failures. Unfortunately the simple world of policy briefs does not conform with the complex policy analysis needed to address the climate crisis (Stirling 2010).

Such forward looking policy design itself needs to be shepherded by causal frameworks capable of helping analysts and stakeholders identify policy mixes that might unleash historical processes. Two processes are gaining much attention now: how to nurture global to local pathways that can help governments reinforce, and meet, their NDC commitments (Bernstein and Cashore 2012, 2000, Cashore et al. 2015, Cashore et al. 2019) and, likewise, how to trigger "bottom up" pathways capable of being entrenched, and diffused, over time (Cashore, Auld, et al. 2016) (Levin et al. 2012) (Rosenbloom, Meadocroft, and Cashore 2019) (Geels 2018)

Stakeholder policy learning about causal processes

Third, relevant stakeholders must be engaged in policy dialogues around the causal process they can help nurture (Cashore et al. 2019). For these reasons the international community needs to spent much greater attention in fostering policy tools focused on problem focused "anticipatory policy design" in which expected "cause and effect" impacts are carefully projected beforehand. Likewise, strategic implications for fostering these pathways cannot wait for the "experiments to be run", but must be articulated as part of a forward looking "game plan" if results are to be nurtured. For these reasons, I envision a new "**Learning Dialogues for Effective climate smart Forest and Agriculture" governance need to be established to help the world's community "right the course. They would need to be** institutionalized stakeholder and scholarly learning processes on two related themes: identifying "cause and effect" relationships between policy design and clearly defined environmental problems; how trigger and nurture, multiple step causal processes through which transformative impacts might occur (Levin et al. 2012, Bernstein and Cashore 2012, Yona, Cashore, and Schmitz 2019).

The Policy Window

The international window is certainly present. The global community, and the EU, is devoting considerable attention, and resources, to targeting specific gaps in the SDGs implementation including the climate and biodiversity crisis.

Positive Examples

Given this analysis, it is clear that the vast majority of policy design to date has been developed in ways that have failed to meaningfully address the climate crisis in general, and the role of agriculture and forests in particular. These include billions spent on what were now widely understood as sanguine expectations (Streck et al. , Parker et al. 2009) of REDD+ efforts which, over a decade alter, have failed to materialize in any significant manner. They also include previous efforts at supply chain governance that were asserted to have positive effects on climate (Forest Stewardship Council Working Group Germany 2010, Subak 2002) and likewise, sanguine beliefs that protecting community forestry will almost always benefit climate challenges (Lawlor et al. 2013, Duchelle et al. 2013).At the same time, we can identify a number of cases around the world that illustrate the benefits of the policy analysis techniques reviewed above, that carry historical lessons for making a difference.

Example #1: 1990s British Columbia Protected Areas

During the mid-1990s a newly elected government promised to implement Brundtland inspired norms of 12% protection of land from commodity interests. The approach drew on both top down and bottom up processes to populate the six cells above. The "top down" approach included mandated the doubling of protected areas from 6-12% of the provinces' land based, and to implementing a "instrument logic" (bottom left cell) a "command and control" "lines on map" regulatory approach for doing so. The "bottom up" approach included creating local stakeholder processes for deliberating, including drawing on ecosystem science, for deciding **where** to protect. Finally a "micro level" design that appeared to have significant path dependent effects that both led to decisions that were also highly durable 25 years later, was the instructions to the local stakeholder processes that they had two years to achieve a solution. They were further told that if they did not agree within two years, a solution would be imposed on them. This "micro level" design instructions shifted power balances, as powerful interests were unable to impose their resources to shift problem definitions or limit protected area changes. At the same time the deliberative processes that ensued were focused on projecting forward expectations of the effects of protecting various types of land. These deliberations over causal impact, rather than simply focused on compromise interest based approaches, appears to have created the conditions in which legitimacy and norms of appropriateness permeated the deliberative arenas and help account for what are durable change processes 25 years later (Cashore et al. 2001).

Example #2: Peruvian "Rights to Resources" for Indigenous Peoples

A group of forest focused scholars applied the policy learning protocol to reflect on how Indigenous communities in Peru might draw on global “legality verification” efforts to help promote their rights to resources. This was an important case, because transnational actors were becoming disillusioned with REDD+ initiatives as failing to make a serious dent on the biodiversity crisis or Indigenous rights (de Jong and Humphreys 2016), and there were concerns elsewhere that legality verification simply reinforced the rights of transnational corporations who had ben granted legal timber concessions. Following two years of deliberations with stakeholders, and engaging forward looking policy mixes and projecting path dependent processes, a multi-step process was offered that had significant “plausible logics” in reversing the negative potential effects of legality verification on Indigenous communities. The policy design included championing wood produced by Indigenous communities as “most favoured” legal sources, and developing strategic coalitions with Peruvian wood exporters to the US market, which, owing to the Lacy Act and the US-Peruvian free trade agreement, were increasingly concerned with ensuring wood came from legal sources (Humphreys et al. 2017, Cashore, Visseren-Hamakers, et al. 2016).

Example #3: Canadian boreal forest carbon cycles

A third example has been applied to the Canadian forest carbon cycle relationship. Ecologists Leroux and Schmitz and colleagues found that the introduction of mining and logging “punctuated up” carbon emissions cycle of predator prey relationships – highlighted by the interaction of Caribou, Moose and Wolves (Leroux and Schmitz 2015). Moose benefit from open areas, and their grazing leads to soild disturbance that increases carbon emissions. However, Caribou are also endangered owing to extractive practices. This had led to policies culling wolves while limit hunting of Caribou. These, in turn, have caused political conflict with hunters and Indigenous communities. In response, Yona, Schmitz and Cashore deliberated over forward looking policy mixes that might find “easy to pull” levers that might cause durable change. They focused on the role of local managers as “street level” bureaucrats that might be able to punctuate lower carbon emissions by allowing for Moose hunting. This would be expected to create durable coalitions of “Bootleggers and Baptists” hunting and environmental groups coalitions, both of whom would supporting moose hunting for different reasons. It was further anticipated that Moose hunting would reinforce and created, cultural traditions of local communities around sustainable protein supplies, while simultaneously helping Canada meet its NDC under the Paris Accord (Yona, Cashore, and Schmitz 2019).

Lessons

The lessons from these examples is that they could be applied to a wide variety of cases, from conservation efforts in Southeast Asia, Latin American and Africa. This is because while they take into account historical political economic differences, they also apply micro level design to macro level transformative expectations.

Factors influencing Successes and Failures in AFOLU Programs and Projects

Governance Successes

Good governance practices play a key role in the success of AFOLU programs and projects. A study in Indonesia found that a REDD+ agency succeeded in transforming the governance structure in place into one that does not enable deforestation and forest degradation (Kaisa et al., 2017). A greater ownership of the transformed governance structure was strengthened by dissolving the REDD+ agency and reassigning the mandates to the ministries. Another area of progress is the increasing involvement of non-state actors in promoting good climate governance in climate change projects, programs and initiatives from the local to the international levels (Bäckstrand, Kuyper, Linnér, & Lövbrand, 2017). States are delegating authority to private sector actors and in some cases the private sector develops their own rules and standards(Kuchler, 2017). Examples are the increasing authority of the private sector in the Kyoto Protocol’s Clean Development Mechanism and REDD+ voluntary carbon market.

Governance Barriers

There are many governance barriers affecting the implementation of agriculture and forestry programs and projects. They include amongst others tenure insecurity, inequitable benefit sharing and inadequate coordination across ministries and sectors. Even though policymakers, practitioners and different actors have long recognized these barriers and have repeatedly called for them to be addressed, they however still exist (Gupta, Pistorius, & Vijge, 2016). Several studies show that unclear property rights and tenure insecurity can undermine the incentives to improve productivity, lead to food insecurity, undermine REDD+ objectives, discourage tree planting, and result in conflict between different land users (Antwi-Agyei, Dougill, & Stringer, 2015; Borras & Franco, 2018; Etongo et al., 2015; Felker, Bong, DePuy, & Jihadah, 2017; Kansanga & Luginaah, 2019; Mbatu, 2015; Paudel, Vedeld, & Khatri, 2015; Riggs, Langston, & Sayer, 2018; Scheidel & Work, 2018; William D. Sunderlin et al., 2018; Thaler & Anandi, 2017).

Multilevel governance challenges are related to poor vertical and horizontal intersectoral coordination, insufficient information sharing, concerns over accountability as well as an understanding of the interests and power relations among actors at different levels within REDD+ projects and programs (Ravikumar, Larson, Duchelle, Myers, & Tovar, 2015). An analysis of REDD+ in seven countries showed that the problem with coordination is due to the lack of recognition and integration of REDD+ at multiple existing national and subnational policy levels. This situation often leads to overlapping regulations and unequal resource allocation among the different sectors (Korhonen-Kurki et al., 2016). Another study found that multiple actors shaped REDD+ programs and projects to some extent, but REDD+ and its advocates are unable to shape land-use dynamics or landscape governance especially in the short term (Rodriguez-Ward, Larson, & Ruesta, 2018). In some cases, governments are increasingly centralizing REDD+ governance and limiting the distribution of governance functions between state and non-state actors (Phelps, Webb, & Agrawal, 2010; Zelli, Möller, & van Asselt, 2017). FLEGT and REDD+ governance regimes are in some cases acting with overlaps and duplication. This situation has raised questions whether FLEGT and REDD+ regimes act in isolation or in coordination in ways that build effective and enduring forest governance(Gupta et al., 2016).

Institutional successes

Institutional interaction determines how institutions may exert causal influence on each other's development and effectiveness (Tegegne, Ochieng, Visseren-Hamakers, Lindner, & Fobissie, 2014). In Cameroon and the Republic of Congo, several interactions occurred between the institutions in charge of REDD+ and forest law, enforcement, governance and trade (FLEGT) voluntary partnership agreement (VPA). In both countries, the overlap in similar issues and between actors triggered interactions between FLEGT VPA and REDD+. The process for developing the VPA served as a policy model for designing elements of REDD+(Tegegne et al., 2014) . To make adaptation or REDD+ a priority in the forestry and agriculture sectors in Cameroon, a study (Somorin, Visseren-Hamakers, Arts, Tiani, & Sonwa, 2016) found that synergetic institutional interaction occurred in sharing of ideas and knowledge to promote interinstitutional learning, cooperative behavior and effective communication. A study that explored vertical institutional interactions between European Union and national level forest-based bioenergy policies in five EU countries observes a high degree of institutional interaction at the level of policy objectives and instruments (Lindstad et al., 2015).

Institutional barriers

Institutional complexity has different shapes and underlying causes (Bäckstrand et al., 2017). It represents a major challenge in integrating adaptation and mitigation in agriculture, forest and other land uses. Current institutional practices in implementing adaptation and mitigation projects and programs are limited to seeking co-benefits, which are necessary but insufficient steps towards promoting synergies at landscape scale (Duguma, Minang, & Van Noordwijk, 2014). Another aspect of institutional complexity is the different climate and non-climate values as well as the public and private financial means involved in the architecture and implementation of REDD+ and other initiatives (Zelli et al., 2017).

Policy approach successes

Polycentric policy approach provide great potentials for climate change adaptation and mitigation. The approach is known to increase cooperation, communication, interaction among local, national and international actors and instigates learning- by-doing to improve policies at different levels of decision making and overtime (Cole, 2015; Ostrom, 2010, 2012). A polycentric system consists of several autonomous units that act together to foster cooperation and facilitate conflict resolution (Galaz, Crona, Österblom, Olsson, & Folke, 2012).

Polycentric systems have their limits however. Most polycentric models downplay the powerful role of power dynamic at the national and subnational levels(Morrison et al., 2017). Empirical research from six countries shows that many systems that are described as polycentric are critically shaped by power. Some of the polycentric systems have not performed well, for example, in reducing deforestation and forest degradation through REDD+ programs. Forest policies under the REDD+ programs are in many cases embedded within established hierarchies of centralized control and state ownership of forest land (W. D. Sunderlin et al., 2016).

Policy approaches barriers

A frequent question and challenge in climate, agriculture, forestry, trade, investment, tenure, and land use policies is how to replicate successful projects, programs and policies across different communities, jurisdictions, and locations (Ravikumar et al., 2015). For this to happen, enabling conditions should be created to promote learning, experimentation and policy entrepreneurship in different locations (Brown & Cohen, 2019). Empirical evidence from the assessment of national policy settings to address deforestation and forest degradation in thirteen countries finds the combination of powerful transformational coalition, strong ownership and leadership and performance-based funding as necessary conditions for success (Korhonen-Kurki et al., 2019). Similarly, empirical assessment of the evolution of drivers of deforestation and forest degradation in the Congo basin shows that institutional and policy factors are more important than any other type of underlying factors and should therefore be considered in the long term design of land use policies (Tegegne, Lindner, Fobissie, & Kanninen, 2016).

Safeguards (social and environmental) successes

- [Placeholder-for SOD]

7.5. Assessment of AFOLU mitigation measures

Land based mitigation can be delivered through a variety of management practices that reduce greenhouse gas emissions or increase carbon sequestration in forests, wetlands, grasslands, and agricultural lands. The types of management practices can broadly be categorized as activities in Agriculture, Forests and other ecosystems, Bioenergy and other land-based energy technologies, and Consumer behaviour. If implemented at scale and in a sustainable way, these land based mitigation practices have the capacity to sequester billions of tons of carbon from the atmosphere in the coming decades, while also feeding millions of people, building more fertile soils, enhancing water quantity and quality, and contributing to ecosystem health (Toensmeier 2016; Francis 2016; Smith et al. 2019).

Since the AR5, there have been numerous new global assessments of ‘bottom-up’ mitigation potential (climate impact of a single practice) (Roe et al. 2019; Fuss et al. 2018; Griscom et al. 2017; Smith et al. 2016) as well as ‘top-down’ mitigation potential (climate impact of multiple and interlinked practices from integrated assessment models) (Frank et al. 2019; Rogelj et al 2018; Popp et al. 2017; Riahi et al. 2017). ‘Bottom-up’ mitigation potentials include a larger suite of practices and activities, and represent a broader body of literature, whereas ‘top-down’ potentials can provide more detail on the economic costs and cross-sector impacts of a smaller suite of practices. Recently published studies on land-based mitigation potential were described in the Special Report on Land, and are synthesized in Figure 7.16 and Table 7.4. Integrated assessment models and the emissions, land cover, and economic dynamics of land-based practices are further discussed in Section 7.6.

Land-based mitigation can deliver about 30% of global mitigation between 2020-2050 (IPCC 2019, Roe et al. 2019, Griscom et al. 2017). However, the technical, economic and political feasibility of mitigation activities range widely. Forest management practices (e.g. avoided deforestation and natural regeneration) are the cheapest and technically easiest options, while activities in agriculture and bioenergy tend to be the most expensive and require enhanced capacities to deploy (Table 7.4). Political will, available finance, governance capacities as well the variable co-benefits and trade-offs are likely to inform the political feasibility of individual land-based mitigation practices (IPCC 2019, Roe et al. 2019). Geographically, cost-constrained mitigation potential is highest in tropical regions (Latin America and the Caribbean, South-East Asia and Africa) because of the lower cost of avoided deforestation and degradation, however there is also considerable potential in developed and emerging countries in temperate regions (Eastern Asia, Europe, Southern Asia, Eurasia and North America) (Figure 7.17).

Table 7.4 Annual carbon mitigation potential ($GtCO_2e$) by category and carbon price across bottom-up literature and integrated assessment models, based on data from Roe et al. (2019). Estimates represent the average, and full range of potential for the years 2030-2050. Note that numbers are cumulated over the price ranges

	<\$20/tCO_2-eq	**<\$50/tCO_2-eq**	**<\$100/tCO_2-eq**	**Technical Potential**
Agriculture only	2.4 (0.5-4.3)	2.4 (0.5-4.6)	5.2 (3-8.7)	11.9 (5.2-26.3)
Forestry and other ecosystems	3.5 (2.4-4.7)	3.6 (2.4-4.7)	9.2 (5.7-13.5)	15.7 (12.9-26.9)
AFOLU (Ag + Forestry)	5.9 (2.9-9.8)	6 (2.9-10.1)	14.4 (8.7-23.7)	27.6 (18.1-60.3)
BECCS	0 (0-0)	0 (0-0)	2.2 (0-0)	5.8 (0.4-16.1)
Demand-side practices	ND	ND	ND	7.7 (1.8-14.3)

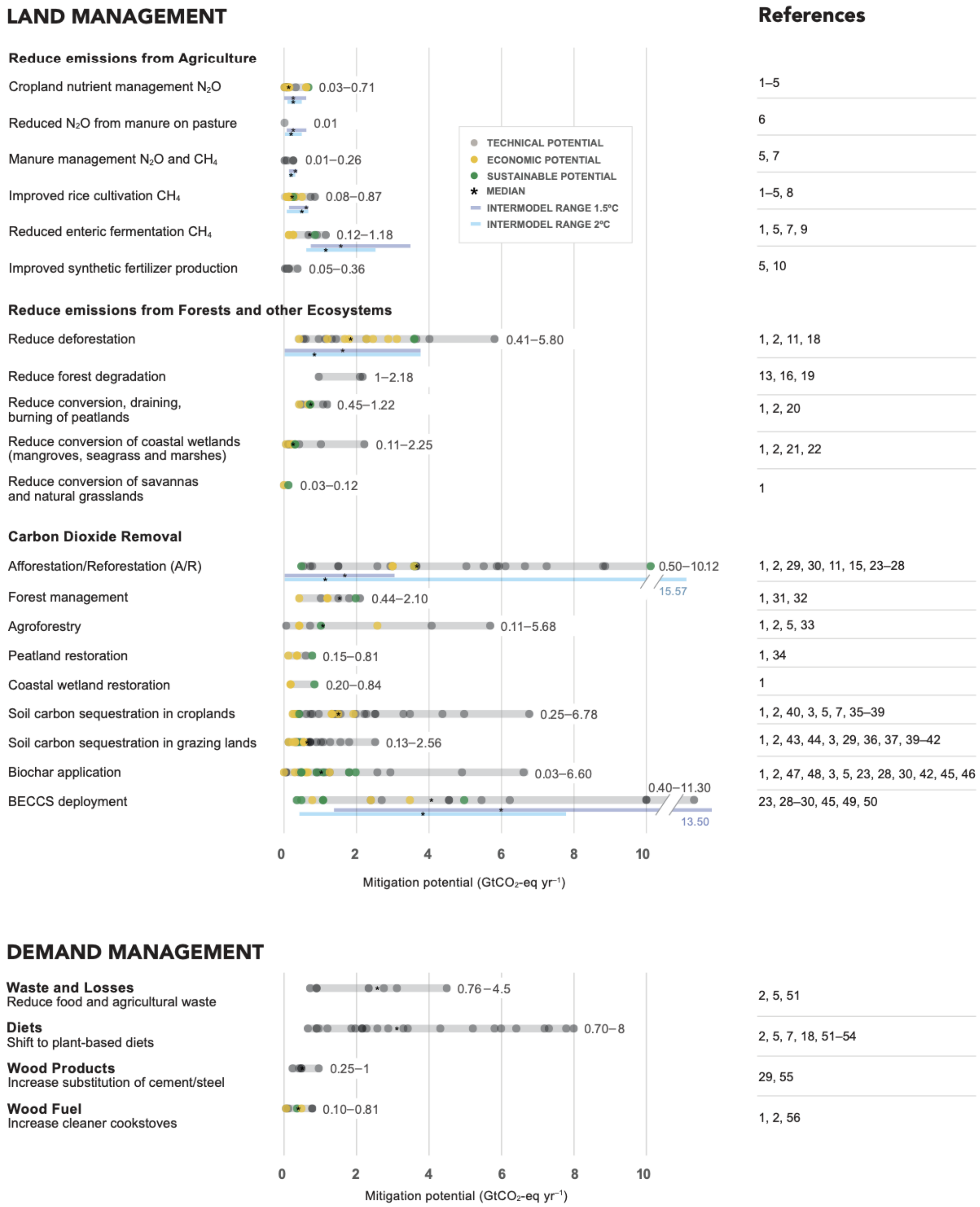


Figure 7.16 Mitigation potential of response options in 2020–2050, measured in $GtCO_2$-eq yr^{-1}, adapted from Roe et al. (2019). Mitigation potentials reflect the full range of low to high estimates from studies published after 2010, differentiated according to technical (possible with current technologies), 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 more than four data points. We only include references that explicitly provide mitigation potential estimates in CO_2-eq yr^{-1} (or a similar derivative) by 2050. Not all options for 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 IAMs 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'.

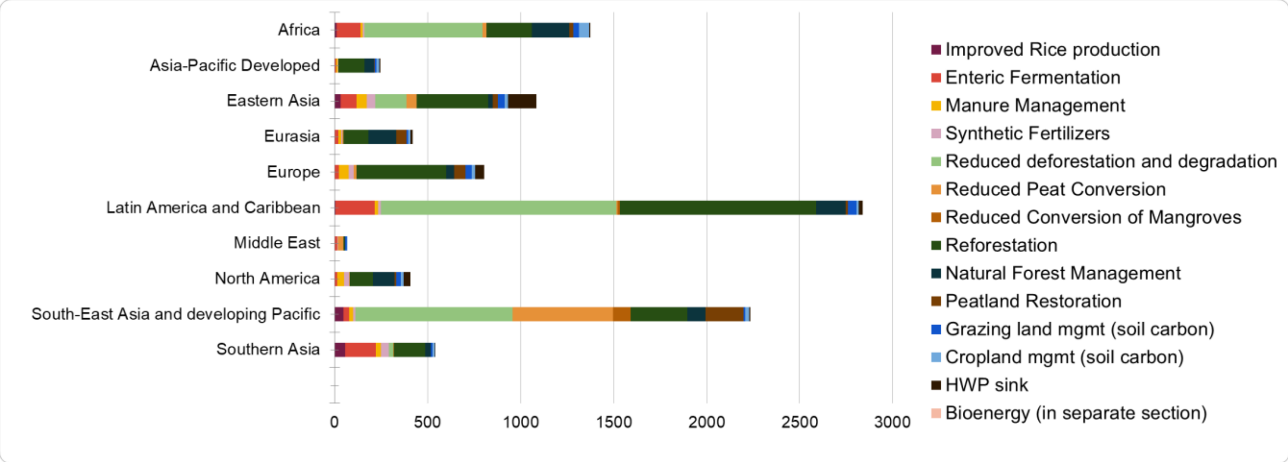


Figure 7.17 Regional and cost-constrained mitigation potential of response options in 2020–2050, measured in $MtCO_2$-eq yr^{-1}, adapted from Roe et al. (2019) and Griscom et al. (2017). Mitigation potentials only reflect land-based mitigation practices with available country-level data, thus the regional potentials do not necessarily add up to the global potential ranges in Figure 7.16. [Placeholder-Figure will be revised with updated and additional data for SOD]

7.5.1. Forest management interventions

Forests provide a wide variety of ecosystem services and international conventions and national policies for climate change mitigation and biodiversity conservation recommend forest protection and restoration.

Afforestation and reforestation

Afforestation is defined by UNFCCC (FCCC/CP/2001/13/Add.1, Marrakesh Accord) as the direct human-induced conversion of land that has not been forested for a period of at least 50 years to forested land through planting, seeding and/or the human-induced promotion of natural seed sources. It is important to differentiate (i) conversion of non-forest land to forest, in places that were originally forested (i.e. in a forest ecoregion) but has not been forested for over 50 years and (ii) conversion of non-forest land to forest that is a native non-forest ecosystem (e.g. planting eucalyptus in a native savanna ecosystem). Although both of these phenomena are referred to as afforestation by the UNFCCC definition, they have very different ecological implications.

According to the UNFCCC, reforestation is the direct human-induced conversion of non-forested land to forested land through planting, seeding and/or the human-induced promotion of natural seed sources, on land that was forested but that has been converted to non-forested land more recently than that classified as afforestation.

Forest restoration, or forest landscape restoration often employs reforestation and afforestation to restore degraded land. The most effective place in terms of carbon sequestration to plant trees is in the tropics and subtropics because of rapid growth and little effect on the albedo (reflectivity) of the land surface, unlike at high latitudes, where trees obscure snow that would otherwise reflect solar energy and help to cool the planet (Lewis et al. 2019). Considering the countries in the Bonn Challenge and other schemes that have published detailed restoration plans, three main approaches are planned. Natural regeneration is the cheapest and technically easiest option. Just over one-third (34%) of the total area allocated is to be managed in this way. Plantations represent 45% of all commitments involve planting vast monocultures of trees as profitable enterprises (mainly in large countries) while agroforestry represent 21% (Lewis et al. 2019).

A review of 154 ongoing and planned restoration projects in Latin America and Caribbean indicated that most projects are located in the humid tropics and less attention is paid to drylands. Additionally, restoration activities were diverse and were related to the type and source of funding that projects receive (for example, Forest Investment Program (FIP), the Global Environment Facility (GEF), and Clean Development Mechanism) (Romjin et al. 2019).

Since 1950s the area of planted forests has increased globally (Szulecka et al. 2014). Recognizing the substantial potential of well-managed forest plantations, the new generation plantations (NGP) platform was launched in 2007 (Silva et al. 2019). NGP encourages well-managed planted forests in the right places to conserve biodiversity and meet human needs. The information from NGP participants and others over 10 years shows that NGP participants are responsible for c.11.1 million ha of land, much of it previously degraded or abandoned; 43% is managed as timber plantations, with the remainder being wildlife reserves, restored natural forest, grassland and agriculture (Silva et al. 2019).

Reduced deforestation and degradation

Forests are one of the most cost-effective ways to sequester carbon and carbon stocks in vegetation have a key role in the climate system (Mader 2019, Erb et al. 2018). Deforestation and other land-cover changes are responsible for 53–58% of the difference between current and potential biomass stocks. Land management effects (the biomass stock changes induced by land use within the same land cover) contribute 42–47% (Erb et al. (2018). These results indicate that avoiding deforestation is necessary but not sufficient for mitigation of climate change.

Tropical regions harbour more than 40% of the world's remaining 4 billion hectares of forests. While deforestation shows signs of decreasing in several countries, in others it continues at a high rate or is increasing (Turubanova et al., 2018). Overall, the area of intact forests is decreasing in all tropical regions with about 3.7 million hectares lost each year (Poker and MacDicken, 2016). In addition, tropical forests are subjected to different drivers of forest degradation as forest fires, severe storms, flooding, and earthquakes but also to the impacts of human interventions in production forests, protected areas and parks.

Improved forest management practices

Besides stopping deforestation or enhancing afforestation, mitigation options include options ranging from improved natural and plantation forest management, improved fire management, and avoided woodfuel (Griscom et al. 2017). Improved Forest Management options maintain or increase forest carbon stocks through a variety of mechanisms, including (but not limited to) increasing forest productivity, avoiding emissions from logging activity, and increasing forest age. Studies that have examined improved forest management have mostly dealt only with e.g. rotation length extension, reduced impact logging methods (e.g. Putz et al. 2008), the influence of alternative silvicultural treatments on stand growth (Davis et al. 2009; Hoover and Stout 2007) or an admixture of species (Paquette and Messier, 2010).

Understanding the effects of management practices on ecosystem values is critical for managing tropical forests sustainably. For example, in Australian tropical forests increasing intensity of management led to greater time required for recovery of species diversity, composition and compositional similarity (Hu et al. 2018). Currently, more than 400 Mha of tropical forests worldwide are being managed for timber production but due to accessibility problems, only parts of the production forests are available for harvest (Poker and MacDicken, 2016). An important commodity is fuelwood, although the statistics on this product are neither complete nor precise because few tropical countries are able to report on the actual amount and value of non-timber forest products (Poker and MacDicken, 2016).

Recent literature also considers the mitigation potential of integrating supply side and demand side including harvested wood products, optimizing the bio-energy chain and taking into account local circumstances and adaptation to climate change. This approach, called climate smart forestry (CSF)

(Nabuurs et al. 2017), has the advantage of hinging on those global forest areas that are under management already, making a change in management much easier because these forests are accessible already and often have a clear ownership structure. CSF builds upon three main objectives; (i) reducing and/or removing greenhouse gas emissions; (ii) adapting and building forest resilience to climate change; and (iii) sustainably increasing forest productivity and incomes. Through CSF, European forests and harvested wood could mitigate an additional 420 Mton CO_2yr^{-1} by 2050 (Nabuurs et al. 2017).

European temperate and boreal forests sequester up to 12% of Europe's annual carbon emissions (Yousefpour et al. 2018). Forest carbon density can be manipulated through management to maximize its climate mitigation potential, and fast-growing tree species may contribute the most to Climate Smart Forestry (CSF) compared to slow-growing hardwoods. This type of CSF takes into account not only forest resource potentials in sequestering carbon, but also the economic impact of regional forest products and discounts both variables over time. A simulation of the implementation of CSF for 18 European countries encompassing 68.3 Mha of forest (42.4% of total EU-28 forest area) found a potential sequestration of 7.3–11.1 billion tons of carbon over the whole 21st century (Yousefpour et al. 2018).

Improved wood utilization

Harvested wood products (HWPs) are means of carbon storage. Three primary wood products (sawn wood, wood-based panels, and paper and paperboards) amounted approximately 5360 Tg C (19,671 Gt CO_2e) in 2013 (FAO, 2016a; Miner and Gaudreault, 2016) and represented 73, 21 and 6% of estimated stored carbon, respectively (FAO, 2016a; Palma et al., 2016). There was a trend of increasing share of the two latter products.

Wood and wood-based can substitute for energy-intensive products made of conventional materials. For example, in China, the quantification substitution benefits of wood furniture demonstrated that from the less to the more wood intensity products with the equivalent function, the proportion of energy-intensive materials decreased 24% while the proportion of their greenhouse gas (GHG) emissions decreased 34% (Geng et al. 2019).

Wood use for piles, check dams, paved walkways, guardrails, and noise barriers, in civil engineering also results in avoided GHG emissions (Kayo and Noda 2018) by carbon storage, material substitution, and energy substitution, with the greatest contributions coming from carbon storage through the use of log piles.

On the other hand, the climate benefits of wood products depend on how and where wood products are sourced (Keith et al. 2015, Schlesigner, 2018). For example, rapid expansion of timber plantations in Indonesia is the largest driver of the loss of higher carbon native forests in that country (Abood et al. 2014). Large differences in emissions per cubic meter of wood produced depend on logging practice used, management intensity, and the relationship between logging activity and subsequent forest conversion (Griscom et al. 2018). These aspects point to the importance of robust safeguards, including improved chain-of-custody tracking and regulations on climate smart wood sourcing.

7.5.2. Restoration of degraded lands

Peatland restoration

Soil carbon sequestration and avoidable emissions through peatland restoration are both strategies to tackle climate change. Peatlands only account for ~3% of the terrestrial surface, predominately occurring in boreal and temperate ecosystems, with a smaller proportion in tropical regions but may store ~644 Gt of C or 21% of the global total soil organic C stock of ~3000 Gt (0–3m). In addition, peatlands are large stores of organic N. Northern peatlands, have accumulated 8–15 Gt N, whereas the N stock in tropical peatlands has not yet been reviewed.

In the event that no further areas are exploited, drained peatlands will cumulatively release 80.8 Gt carbon and 2.3 Gt nitrogen corresponding to a contemporary annual greenhouse gas emission of 1.91 (0.31–3.38) Gt CO_2-eq (Leifeld and Menichetti 2018). Restoring peatlands is 3.4 times less nitrogen costly and involves a much smaller land area demand than mineral soil carbon sequestration. Restoration through rewetting can significantly reduce GHG emissions, restore vegetation communities, and recover biodiversity, while still allowing for extensive management such as paludiculture (Leifeld and Menichetti 2018).

7.5.3. Agricultural interventions

Grassland management

Grasslands occupy 3.4 billion ha and account for about one-fourth of potential C sequestration in the world (FAO, 2016) while also providing other essential ecosystem services such as maintenance of biodiversity and natural resources protection (Lemaire, 2007). However, land use management exerts a substantial influence on C balance in grassland ecosystems (Borges et al. 2019). Management practices such as fertilization, species selection, grazing, and harvesting regimes can have both positive or detrimental impacts on soil C and the overall ecosystem C balance (Lal, 2018). Historically, extensive management of grasslands predominates. However, increasing demand for food production and the massive increasing population are related to the intensification of grasslands management or converted into more intensive agriculture (see Drivers section).

In tropical regions, pastures-based animal production systems generally use low stocking rates or mechanical harvest of above-ground biomass associated with relatively low levels (or absence) of fertilization. In the long-term, poor pasture management without proper soil conservation practices leads to further soil degradation, with significant adverse impacts on soil C dynamics (Silveira et al., 2013). On the other hand, proper soil management improves soil fertility conditions and soil C accumulation (Follett and Reed, 2010).

Total soil C is a standard indicator of changes in response to management (Jeong et al., 2016). Carbon loss after grassland conversion to cropland is often rapid (estimated mean of 1.81 t ha^{-1} yr^{-1}), and a new equilibrium of soil organic carbon occurs after 17 years (Khalil et al. 2019). On the other hand, the maintenance or enhancement soil organic C density can occur with the conversion of cultivated land to pasture, restoration of degraded land, and integrated production systems (for example, agro-silvopastoral) (Khalil et al. 2019). Carbon sequestration improved with integrated crop-livestock farming systems in different regions (see Franzluebbers et al. 2014 for a review). Practices as the inclusion of productive grass varieties, incorporation of legumes in N deficient systems, appropriate fertilization, rotational grazing, and other climate-resilient approaches could improve the sustainability of livestock farming systems (Khalil et al. 2019, Maia et al. 2009, Coonan et al. 2019).

Improved grazing management

Overgrazing is one of the leading causes of desertification in semiarid grasslands. Grazing exclusion (GE) is an effective management practice globally to restore degraded grasslands and improves SOC significantly (Wang et al., 2018, Chai et al. 2019).

The adoption of improved grazing management practices has the potential to reduce the adverse impacts of intensive farming on climate. Model simulations indicate that adequately managing the timing and duration of rest and grazing periods can enhance environmental benefits from subtropical pastures through increased soil C storage without compromising beef production (Gomez-Casanovas et al. 2018).

Table 7.5 Annual SOC density changes (t $Cha^{-1}yr^{-1}$) in grazing grassland/pasture and the associated management pracices (*this table is a placeholder; ha scale data to be replaced by regional/contineneta l assessments and to be incorporated in Figure 7.16)*

Biomes/regions	Livestock category	Livestock density (LSU: L, M, and H) * and management practices	Fertilization (N, P) and liming (kg ha^{-1})	SOCρ changes (t C ha^{-1} $year^{-1}$)
The EU and France (managed grassland) [23]	Cattle	L = LSU <0.6; HUI <0.3	Zero and ≥100 N	1.27 ± 0.40*
		M = LSU 0.6–1.3, HUI 0.3–0.7	Zero and ≥100 N	1.12 ± 0.32*
		H = HUI 0.7–1	Zero	−0.57 ± nd
			>100 kg N	0.74 ± 0.30
Australia (perennial and annual pasture) [24–28]	Mixed (cattle/ sheep)	M	—	0.50 ± nd
			Liming ± nutrients	0.40 ± 0.06*
			Liming ± phosphate	0.35 ± nd*
		Density = 0%, L to H = 50–200%	—	0.10 ± 0.10 −0.45 ± 0.53*
Rotational grazing [29]		M	—	0.35 ± nd
Hungary (extensive grazing) (Grazing +1 cut) [20]	Cattle (mowed)	L = 0.64 NLSU, HUE: 0.4	—	0.86 ± 0.74
		L = HUE 0.6	—	−1.23 ± 0.35
Hungary [30]	Mixed	L (Cattle, sheep, goat, and horses)	—	0.0013 ± nd
Mediterranean, Spain (extensive grazing)		L = 0.7–2.5 ewe eq ha^{-1} (cattle, sheep, pigs, and goats)	—	0.05–0.10
USA (mixed prairie) [5] (Grazed Bermuda grass) [31, 32] -Mixed prairie - Short-grass steppe [21, 22]	Steers	H + M	Inorg. (N: 200–270), inorg-org. (73.6), and broiler litter	0.03 ± 0.00
	Cattle Angus Steers	L H	—	1.17 ± nd 0.51 ± nd
		M	—	0.65 ± nd
	Sheep	L H	—	0.20 ± nd* 1.51 ± nd*
		L = YH (20–35% utilization) H = YH (60–75% utilization)	Long-term grazing Heavy grazing	0.55 ± nd 2.27 ± nd

*LSU = Livestock density unit (ha^{-1} $year^{-1}$); L = Low; M = Medium; H = High; nd = Not determined; YH = Yearling heifers; SOCρ = SOC density; * = Pooled/Averaged.*

Table 7.6 Annual SOC density changes (t C$ha^{-1}yr^{-1}$) in integrated farming and the associated management pracices (*this table is a placeholder; ha scale data to be replaced by regional/contineneteal assessments and to be incorporated in Figure 7.16)*

Biomes/regions	Livestock category	Livestock density and/or No. LSU*	Fertilization (N and P) and liming (kg ha^{-1})	SOCρ changes (t C ha^{-1} $year^{-1}$)
Queensland, Australia (grassland and planted tree legumes mixed) [39]	Cattle	L to M = 0.45 AE ha^{-1}; 1 AE = 400 kg steer	P = 22 S = 28	0.28 ± 0.00
Southern Amazon, Brazil (integrated crop-livestock-forestry) [37]	Cattle	H = 21.27 AU ha^{-1}	370 NPK; 318 SP; 105 KCl; 324; NPK; 86 urea +10 KCl; 400 SP + 69 KCl; Pasture: 30 SP	0.60 ± 0.12 1.30 ± 0.23
Amazon, Brazil (nominal)	Beef cattle	M	—	−0.03 to 0.72 ± nd
Improved with legume and productive varieties [40]			Fertilization, lime, and irrigation	0.61 ± nd
Mediterranean, Italy (silvopasture) [41]	Sheep	L = 3–4 sheep ha^{-1}	50–39 (N-P)	0.71 ± 0.13
		L = 6 sheep ha^{-1}	50–39 (N-P)	1.20 ± 0.07
Inner Mongolia (semi-arid steppe) [9]	Cattle/sheep grazing exclusion	L	Season long grazing	0.10 ± 0.00
China (degraded grassland) [42]		M	0–30 cm 0–100 cm	0.23 ± 0.03 0.19 ± 0.04
Northern China (semi-arid, grassland) [43]		M	—	0.10 ± 0.00
China (desert steppe) [44]		H	—	1.43 ± 0.00

LSU = Livestock density unit (ha^{-1} $year^{-1}$); L = Low; M = Medium and H = High; SOCρ = SOC density change; nd: Not determined.

Table 7.7 Annual SOC density changes (t $Cha^{-1}yr^{-1}$) in land use change and the associated management pracices (*this table is a placeholder; ha scale data to be replaced by regional/contineneta l assessments and to be incorporated in Figure 7.16*)

Biomes/regions	Livestock category	Livestock density (LSU*)	Fertilization (N and P); liming (kg ha^{-1}) and slurry (t C ha^{-1} $year^{-1}$)	SOCρ changes (t C ha^{-1} $year^{-1}$)
Germany (grassland to cropland) [49]	Cattle	M = 2.2, 0–1 cut	~46 N, ~7 P (7 yrs); Slurry: ~0.1–0.2 t C ha^{-1} $year^{-1}$	-2.77 ± 1.79[a]
		M = 1.9; 2–3 cuts	~64 N (1 year); Slurry: ~0.2–0.4	-27.2 ± 11.70[b]
Ireland [51]	—	—	87–316 N (mineral and organic fertilizers)	-12.88[c]
Temperate zone [50]	—	—	—	-1.81 ± 0.55[d]
Europe [52]	—	—	—	-19.00 ± 7.00
Europe (cropland to pasture) [50–53]	—	—	—	1.99 ± 0.55[e]
				0.56 ± 0.34[f]
				0.22 ± nd
Australia (cropland to pasture) [2, 37, 54]	—	—	—	0.33–0.70 ± nd

LSU = Livestock density unit (ha^{-1} $year^{-1}$); L = Low; M = Medium and H = High; SOCρ = SOC density change; nd: Not determined.
[a]7-year average.
[b]1-year average.
[c]2.5-year average.
[d]20-year average.
[e]20-year average but reaching an equilibrium may take >100 years.
[f]32-year average, yet to reach equilibrium.

Animal production

Enteric fermentation

Good agricultural practices can lower methane emissions in tropical regions. Beef cattle breeding systems in the Amazonian biome were compared by Mandarino et al. (2019). Methane emissions per kg of live weight were lower in intensified areas. Intensification generated better economic indicators, with gross margin differences of US$ 318.89. The results in the intensified areas, when compared to the one in the conventional systems, presented a 16% increase in productive weight in live weight.ha^{-1}, with a 2.05% increase in investment return rates and US$ 142.94 in net present value. Good practices have provided economic, social and environmental gains, such as an increase of almost 5% in investment return rates, generating higher production, continuing employment in farms with historical negative gross margins and reducing the methane emissions per kg of live weight in the dry season by 54%.

Placeholder for SOD:
Feeds and feeding management
Manure and manure management
Animal husbandry

Integrated production systems

Land and the resources it provides is fundamental for humanity. Worldwide, more than 2.5 billion people depend directly on agriculture for their livelihoods (Steiner, 2018). However, the overexploitation and unsustainable use of land, water, nutrients and energy to meet the demands of unprecedented population growth are major challenges to a sustainable future. Biomass productivity has declined on approximately 29% of Earth's land area (Le et al. 2016; Tripathi et al. 2017). The proportion is 36% for cropland, forest and pasture systems. The share of cropland degradation seems especially high in Asia (30 %), North Africa and Near East (45 %), the regions with extensive irrigated agriculture (Le et al., 2016). Land degradation and desertification significantly affect food availability and distribution and constitutes a key driver of food insecurity and hunger in different parts of the world (IPCC, 2019; Nkonya et al. 2016).

This situation is further exacerbated by anthropogenic land degradation and climate change. The IPCC Special Report on 1.5°C highlights the urgent need to keep global temperature rise to below 2°C above preindustrial levels as one of the actionable goals of the Paris Agreement. The report also intimated the role of land in achieving a low warming future (Chapman et al. 2018). Additionally, the IPCC Special Report on climate change and land has concluded that there are opportunities for land to contribute to mitigate and adapt to climate change but this potential could be undermined if fossil fuel emissions reduction is delayed (IPCC 2019). Given the scale of the pressure on land, water, biodiversity and ecosystems, sustainable and efficient use of land-based resources (termed integrated production systems) to meet the demand for food, fiber and energy has become even more pressing. Integrated production systems also play a crucial role in climate change adaptation and reducing GHG emissions from the agricultural sector, as their emissions intensities are generally lower than the combined total of those from specialized systems, Table 7.1 (Andrade et al. 2014). In integrated systems, adaptive capacity is enhanced or reduced by the nature and trade-offs between the components of the system and their level of integration (Dixon et al. 2014; Smith et al. 2019). Integrated production systems are region specific and the success of these approaches depend on the existing human needs, prevailing environmental, cultural and socio-economic conditions as well as indigenous and local knowledge (FAO 2011). This section of the report covers some AFOLU-related CDR measures including agro-forestry (combining crops with trees for fuel and timber); crop-live systems; livestock-fish and rice-fish farming.

International and national research have revealed that agro-ecological farming and livestock systems, including regenerative, organic, sustainable, conservation agriculture, sylvopasture and agroforestry, can both sequester and reduce direct agricultural GHG emissions. In other terms, applying and adopting climate smart agricultural production systems have the potential to mitigate or curb climate change trends (Smith et al. (2019a). In order to achieve the climate goals of Paris COP21 to limit the global warming to 1.5°C, the world should shift and/or adopt technique of removing CO2 from the atmosphere or implement negative emission technologies (NET) (Fuss et al., 2016, 2018; Williamson, 2018). However, in order for these technique to deliver such targets and at the scale needed depend on efficiency, viability, feasibility, acceptability, safety and costs/benefits (Kartha and Dooley, 2016; Wezel et al., 2014). In addition, the options should both assure CO_2 removal or storage and non-climatic impacts such as healthy ecosystems, biodiversity protection, food security and environmental sustainability (IPBES, 2019; Smith et al., 2016; Wezel et al., 2014).

Soil carbon sequestration

Carbon sequestration in soil is an important NET option with numerous co-benefits of enhancing agricultural production, improving water resources, and strengthening biodiversity. Total SOC to 2 meters depth has been estimated at 2400 Pg, which is three times the amount of CO_2 currently in the atmosphere (~830 Pg C) and 240 times current annual fossil fuel emissions (~10 Pg). Thus, increasing net soil C storage by even a few percent represents a significant C sink potential. Soil C sequestration is one of a few strategies that could be applied on a large scale and at low cost (Paustian et al., 2016). According to recent estimate by Lal et al. (2018), carbon sequestration in the terrestrial biosphere, with a technical cumulative C sink capacity of 155 Pg C (158.6 × 109 t C) in vegetation and 178 Pg C (182.1 × 109 t C) in soil by 2100, is equivalent to drawdown of atmospheric CO_2 by 156 ppm.

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 (Banwart et al., 2015). 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.

Agricultural practices that address major land challenges were studied and compared by several authors in terms of their mitigation potentials as well as environmental and societal implications (Brandt et al. 2019, Minx et al. 2018, Smith et al., 2019a ; 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; Sanchez et al. 2017; Pereira et al. 2010; Griscom et al. 2017; Rogelj et al. 2018, Nemet et al., 2018). In fact, the most practical and cost-effective way to remove excess CO_2 from the atmosphere is through living plants and soils. Farmers and landowners can sequester tons of C per hectare in soil and perennial biomass through best management practices for soil health, crop and livestock production, and agroforestry. The amount of GHG emitted from an agricultural operation or option depends on its system of management (Griscom et al., 2017; Roe et al, 2019). Smith et al. (2019b) summarized that soil carbon sequestration options have mitigation potential ranging from 0.4 to 8.6 $GtCO_2e\ yr^{-1}$. Niles et al. (2018) presented soil carbon sequestration values ranging from 1.3 to 8.0 Gt $CO_2eq\ yr^{-1}$. Jia et al (2019) and Roe et al. (2019) distinguished soil carbon sequestration in agricultural lands from grazing land summarized potential as 0.25-6.78 and 0.13-2.56 $GtCO_2$-eq yr^{-1}.

Natural climate solutions such as conservation, restoration and improved land management that increase carbon storage and/or avoid greenhouse gas emissions in forests, wetlands, grasslands, and agricultural land, are estimated to contribute to a third of climate change mitigation (Griscom et al., 2017) Brandt et al. (2019) showed for East Africa that an intensification of dairy production can lead to a lower emission intensity per litre of milk, partly because of reduced land area required and thus preservation of forest.

Griscom et al. (2017) concluded by noting that existing knowledge provides a solid basis for immediately prioritizing land-base mitigation options as a cost-effective way to provide 11 $PgCO_2e$ yr^{-1} of climate mitigation within the next decade - a terrestrial ecosystem opportunity not fully recognized by prior roadmaps for decarbonization (Friedlingstein, 2015). In other terms, for Griscom et al. (2017), natural climate solutions can provide 37% of cost-effectiveness CO_2 mitigation needed through 2030 for a greater than 66% chance of holding warming to below 2°C. Enhancing soil carbon sequestration through sustainable land management, conservation and restoration of ecosystems, can also improve soil nutrient levels, soil fertility and food security.

Despite the progress in science (Lal 2018) and technologies (Batjes 2018), the scientific knowledge to enhance soil carbon pools has not been effectively translated into an action plan at local, regional, national, or global scales. In addition, some authors suggested some cautious about the potential of soil to mitigate global warming (Amundson and Biardeau 2018; Schlesinger and Amundson 2018; Kartha and Dooley, 2016). Hence, a widespread adoption of proven technologies would also involve incentivization of researchers, farmers and land managers on the numerous co-benefits of SOC sequestration and through payments for provisioning of ecosystem services.

Agricultural C sequestration cannot be expected to offset anthropogenic GHG emissions indefinitely because of the phenomenon of soil C saturation (Wiesmier et al., 2019). Implementation of improved soil health management practices on cropland soils typically leads to steady increases in total SOC over a period of 10 to 40 years, after which it reaches a new steady state or plateau. But assuming SOC reaches a plateau after 25 to 50 years of best practices (Lal et al., 2018) estimated the technical potential for global SOC sequestration at about 52 billion tons (range 21 to 83 billion tons); the median value might reduce atmospheric CO_2 levels in 2100 by 22 ppm. Minasny et al. (2017) consider the 4 per mille initiative—that is, raising the content of soil organic matter by 0.4% per year—as an optimistic and aspirational challenge to maintain and improve soil health and provide food security for the world's peoples. The authors suggested that the carbon storage would off-set 20-35 % of global GHG emissions.

Because of the large variation in global agro-ecosystems (i.e., soil, climate, terrain, crops, farming system, and the human dimensions), there is no single technology that can be universally applicable (Bustamante et al., 2014). In the following we address potential of selected options in addressing CO_2 emission mitigation and other co-benefits as well as barriers for their upscaling. Figure 2 presents a summary for drivers and indicator of soil carbon sequestration and which may explain the variation in potentials from Table 7.8.

Table 7.8 Soil carbon sequestration potential by land-based negative emission options

Authors	Estimated Potential Soil C storage (Pg Cyr^{-1})
Smith et al. (2008)	1.6
Zomer et al. (2017)	0.90–1.85
Lal (2018)	2.45
Griscom et al., 2017	0.41
Minasny et al. (2017)	9

7.5.4. Conservation agriculture

Improvement of cropland management comprises a range of practices and systems which include integrated crop management, conservation agriculture, organic farming, irrigation management, nutrient management, improved rice management and biochar application (Smith et al., 2019b). The authors reported a mitigation potential of this collection of practices to vary from 1.4 to 2.3 Gt CO_2e yr^{-1}.

Conservation Agriculture (CA), comprising minimum mechanical soil disturbance and direct seeding, organic mulch cover, and crop diversification, is now practiced on more than 180 million ha in all continents and all ecologies. There is worldwide scientific evidence from research and empirical evidence from farmer practice to show that CA is an effective strategy for climate change adaptability and mitigation (Kassam et al., 2019; Lal, 2015). Even though, an increasing number of countries and regions are adopting CA systems, but the dynamics, scale and pace should be enhanced (Mrabet et al., 2012).

Changing or shifting from conventional agricultural practices and cropping systems to conservation agriculture can have varying effects on crop yields (Pittelkow et al., 2015); soil carbon contents and conservation (Ogle et al., 2019). Conservation agriculture cropping systems that integrate cover crops, diversified crop rotations, organic amendments, no-till, and limited use of synthetic fertilizers and herbicides show significant C sequestration potential, estimated at 600 to 1,000 lb SOC/ac-year (Lal, 2016), and have substantially improved soils from cold-temperate semi-arid regions like the US northern Great Plains to tropical regions in Africa (Montgomery, 2017). The impact of no till farming and conservation agriculture on soil carbon stocks is often positive (Gonzalez-Sanchez et al. 2012), but can be neutral or even negative, depending on the amount of crop residues returned to the soil (Baker et al., 2007; Powlson et al., 2014; Haddaway et al., 2017; Ogle et al., 2019). However, such practices can influence non-CO_2GHG emissions. Soil N_2O emissions from conservation agriculture systems have been reported to decrease, to be unaffected by or to increase relative to those from CT systems (Six et al., 2004).

If soil organic carbon stocks were increased by increasing fertilizer inputs to increase productivity, emissions of nitrous oxide from fertilizer 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 (Cheesman et al. 2016; Frank et al. 2017; Gao et al. 2018; Keesstra et al 2016; Lambin and Meyfroidt 2011; de Moraes Sá et al. 2017; Powlson et al. 2016, Smith et al., 2016, Soussana et al. 2019; VandenBygaart 2016; Hijbeek et al., 2017; Schjønning et al., 2018).

Use of cover crops is a widely applied CA practice to reduce fertilizer inputs and limit risk of water contamination due to leaching (Gonzalez-Sanchez et al., 2012). Soil biological activity is also enhanced. They are also used to reduce wind and water erosion and to build-up soil organic carbon of soils (Gonzalez-Sanchez et al., 2017 (Table 7.9); Poeplau and Don, 2015). Accumulated soil organic matter also assumes reduction of CO_2 emissions (Basche et al., 2014). Particularly, leguminous cover crops are important sources of easily absorbed nitrogen for crops in rotations and for promoting microbial diversity and soil structure and stability.

CA systems, over large landscapes and watersheds, provide a basis for rehabilitating the soil productive capacity, water resource base and watershed ecosystem services, and landscape biodiversity of the degraded semi-arid environments. However, experience across many countries has shown that the adoption and spread of CA requires a change in commitment and behavior of all concerned stakeholders. Technological progress is uneven among regions and countries. Hence, policies need to be flexible enough to allow technology transfer, adoption and dissemination. In addition, policy momentum across various levels of government, as well as a surge in adoption commitments by non-state actors and business sector, create opportunities for countries to enhance the ambition for wide dissemination of bio-sequestration options.

Table 7.9 Soil organic matter accumulation in soils as affected by conservation agriculture practices over European eco-regions (Gonzalez-Sanchez et al., 2017).

Bio-geographical region	CA practice	Increase of soil organic carbon (t ha^{-1} Yr^{-1})
Boreal	No-tillage	0.02
	Groundcovers	ND
Continental	No-tillage	0.42
	Groundcovers	0.40
Atlantic	No-tillage	0.32
	Groundcovers	0.40
Mediterranean	No-tillage	0.81
	Groundcovers	1.30

According to Kassam et al., (2019), farmers in almost 20 African countries are practicing CA in about 1.8 Mha. From a meta-analysis by Gonzalez-Sanchez et al. (2019), potential sequestration by CA in various regions in Africa is reported in Table 7.10. The authors reported that potential estimate of annual carbon sequestration in African agricultural soils through CA amounts to 143 Tg C yr^{-1}, that is 524 Tg CO_2 yr^{-1}. This figure represents about 93 times the current sequestration figures. In addition, this potential is almost 3 times higher that found for Europe by Gonzalez-Sanchez et al. (2017), which amounts to 189 Tg CO_2 yr^{-1}.

In Indo-Gangetic Plains, annual increases in SOC stock compared to conventional practice were between 0.16 and 0.49 Mg C ha^{-1} yr^{-1}. In Sub-Saharan Africa increases were between 0.28 and 0.96 Mg C ha^{-1} yr^{-1}, but with much greater variation and a significant number of cases with no measurable increase (Powlson et al., 2016). Conservation agriculture can serve to mitigate greenhouse gas (GHG) emissions from agriculture by enhancing soil C sequestration but also through improving soil quality, N-use efficiency and water use efficiencies, and reducing fuel consumption.

Table 7.10 Carbon sequestration rates in conservation agriculture (CA) for each climatic zone in Africa (Gonzalez-Sanchez et al., 2019).

Region	Carbon sequestration rate for CA in annual crops (Mg Ha-1 yr-1)	Carbon sequestration rate for CA in woody crops (Mg Ha-1 yr-1)
Mediterranean	0.44	1.29
Sahel	0.50	0.12
Tropical	1.02	0.79
Equatorial	1.56	0.26

Organic farming (OF)

Organic agriculture or farming is a production system that sustains the health of soils, ecosystems and people. It relies on ecological processes, biodiversity and cycles adapted to local conditions, rather than the use of inputs with adverse effects. Organic agriculture combines tradition, innovation and science to benefit the shared environment and promote fair relationships and a good quality of life for all involved.

OF refers to a process that uses methods respectful of the improve crop quality and reduce GHG emissions. Organic generally has lower energy use and GHG emissions per ha, better energy input/output ratios per unit of product, but variable results for energy use and GHG emissions per unit of product. With some variability in results for field crops, hogs and some fruits and vegetables, organic systems are consistently more energy efficient, beyond a 20% threshold, than conventional systems, measured by land area and production (Lynch et al., 2011). OF systems contribute to climate change mitigation through better management of nutrients and, hence, the reduction of N_2O emissions from soils. Nitrous oxide and carbon dioxide emissions were clearly lower on organic farms, with much higher C sequestration (Gomiero et al., 2008). Organic systems also foster greater carbon sequestration by excluding toxic pesticides.

By 2030 soil carbon sequestration and the avoidance of mineral fertilizers in organic agriculture could reduce or offset emissions equivalent to about 35% of total EU agricultural emissions in the baseline projections, for which the emissions are now forecast to stay at around 465 $MtCO_2$-eq per year till 2030 (Muller et al., 2016). Globally, the emission reduction potential by abstention from mineral fertilizers is about 20% and the compensation potential by carbon sequestration is 40–72% of the world's annual agricultural greenhouse gas emissions.

Crop nutrient management

On croplands, there is a mitigation potential of 0.03–0.71 $GtCO_2$-eq yr^{-1} for cropland nutrient management (fertilizer application) (Griscom et al. 2017; Hawken 2017; Paustian et al. 2016; Dickie et al. 2014; Beach et al. 2015).

Nitrous oxide is one of the main source of emissions from cultivated lands (36%), with methane (53%) but well ahead of carbon dioxide (11%). Improving nitrogen fertilizer management is one of the most effective GHG reduction strategies that farmers can adopt. Hence, identifying and adopting technologies and practices that can make fertilizer use more efficient can help significantly reduce emissions of nitrous oxide in agriculture (Shcherbak et al., 2014). The authors found that the N_2O response to N inputs increase significantly faster than linear for synthetic fertilizers and for most crop types.

The natural processes of nitrification and denitrification produce N_2O in soils. A variety of agricultural activities increase mineral nitrogen availability in soils directly or indirectly and, thereby, increase the amount available for nitrification and denitrification, ultimately leading to increases in the amount of N_2O emitted.

Better timing of fertilization means that the crop need/uptake and the applying of fertilizer and manure are more in line with each other. A timely application of fertilizers, especially nitrogenous fertilizers, has several beneficial effects for the environment (Roy et al., 2014). Splitting N fertilizer application is an effective means of improving N use efficiency in agricultural crop production (Wezel et al. 2014).

Fertilizers with coatings or inhibitors to delay nitrogen availability to crops are useful for N_2O emission reduction but are not highly affordable to most farmers. Nitrification inhibitors (NI) can be applied to slow down the transformation of ammonium into other forms that result in nitrogen losses and have adverse effects on the environment. The objective of using NI is to control leaching of nitrate by keeping nitrogen in the ammonia form for a longer time, preventing denitrification of nitrate and reducing N_2O emissions caused by nitrification and denitrification. Thus, via NI, crops have a better opportunity to absorb nitrate, which increases nitrogen-use efficiency and at the same time reduces N_2O emissions from mineral fertilizers (Lam et al., 2015; Ruser and Schulz, 2015).

Integrated plant nutrient management (IPNM) and 4R approach proposed by IPNI can be used to reduce significantly the emission of N_2O from fields and at the same time guaranty better production and environmental stewardship (IPNI, 2012; Johnston and Bruulsema. 2014). IPNM implies several management options: crop rotations, reduced tillage, use of cover crops, manure application, soil testing and comprehensive nitrogen management plan.

7.5.5. Bioenergy

Many countries have moved quickly to set up targets for fossil fuel substitution by bioenergy. In fact, global biofuel consumption is growing steadily with average annual growth of 5.2 and 11% for ethanol and biodiesel, respectively (Kline et al., 2017). China has announced a target of 20% petroleum substitution by 2017, the European Union 10% by 2020, and different states in the USA have announced different targets ranging from 7% to 20% over different periods. It was reported that at global scale, the land required in order to substitute 10% of fossil fuels with biofuel by 2020 would vary from 142 to 600 Mha (Ravindranath et al., 2009). 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).

Currently, less than 1% of global agricultural land is used for cultivating biofuel crops and land use change (LUC) associated with bioenergy represents a very small percentage of overall changes in land use. Food, fiber and bioenergy crops can be grown in integrated production systems, mitigating displacement effects and improving the productive use of land. Lignocellulosic feedstocks for bioenergy can decrease the pressure on prime cropping land. The targeting of marginal and degraded lands can mitigate land use change associated with bioenergy expansion (Elbersen et al. 2019). Elbersen et al. (2019) in this way identify 69 Mha of land as marginal lands, possibly suitable for industrial crops. In addition, bioenergy does often not entail LUC, although high demands may change management in existing land uses. It is therefore very important to always carefully analyse where the biomass comes from, which biomass it is and how sustainable it was produced, etc. The use of post-consumer organic residues and by-products from the agricultural and forest industries does not cause LUC if these biomass sources are wastes, i.e. were not utilized for alternative purposes. Bioenergy demand can provide opportunities for cultivating new types of crops and integration of bioenergy production with food and forestry production in ways that improve overall resource management (Elbersen et al. 2019).

Globally, 14 Gt of forestry residue[3] and 4.4 Gt residues from crop production (mainly barley, wheat, corn, sugarcane and rice) are generated every year. This is a significant amount of biomass which can

[3]

be combusted to generate 26 $EJyr^{-1}$ and achieve a 2.8 Gt of negative CO_2 emission. Utilizing residues for carbon capture will provide social and economic benefits to rural communities. Using waste from crops and forestry is a way to avoid the ecological and social challenges of BECCS (Pour et al., 2017).

The deployment of bioenergy, on a very large scale as envisaged in some mitigation scenarios, could have significant negative impacts on biodiversity and food security through land use change (Searchinger et al., 2017). Drastic mitigation scenarios that rely on large amounts of negative emissions thus require large areas of productive land, with estimates in the literature ranging from 100 million to almost 3,000 million hectares (Mha). The upper end of this range is equivalent to twice the world's currently cultivated land (Kartha and Dooley, 2016).

In fact, biomass production for energy when aimed at huge amounts is subject to a range of sustainability constraints, such as: scarcity of arable land and fresh water, loss of biodiversity, competition with food production, deforestation and scarcity of phosphorus (IPBES, 2019). Using large volumes of bio-energy crops as feedstock will not only cause sustainability concerns but also require the use of more fertilizer leading to soil contamination and water pollution. Large scale energy crop production may also lead to altering ecosystem function at scale, diminishing biodiversity and depleting scarce resources (Dooley and Kartha, 2018).

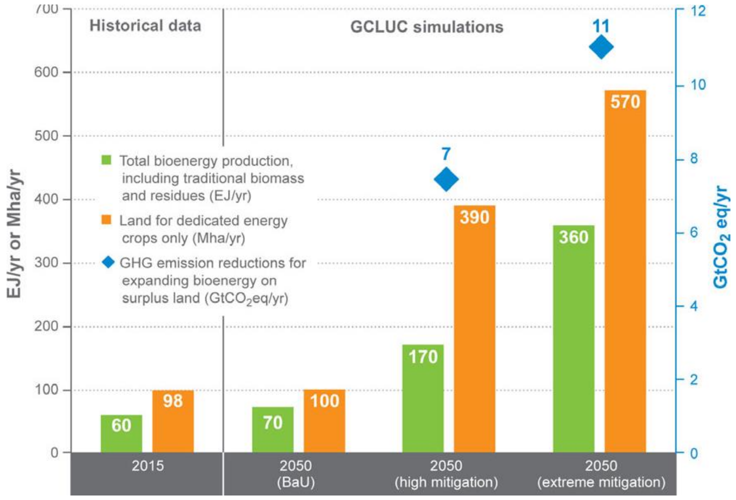


Figure 7.18 Global bioenergy production, land used, and emissions reductions estimated by 2050, from a business-as-usual scenario to an extremely high GHG mitigation pathway for the agriculture, forestry and other land use (AFOLU) sector (Strapasson et al., 2017).

Driven by growing population, urbanization, demand for food and energy, as well as land degradation, competition for land is expected to accentuate land scarcity in the future. However, Kline et al. (2017) presented a more thorough analysis on integrating bioenergy, food security and resource management and they were more optimistic and advocated for site-specific projects and increased implication of sciences and technologies.

Limiting the global temperature rise to 2 °C, with any confidence, would require the removal of some 600 Gt of CO_2 over this century the median estimate of what is needed). Using BECCS, this would probably require crops to be planted solely for the purpose of CO_2 removal on between 430 million and 580 Mha of land — around one-third of the current total arable land on the planet, or about half the land area of the United States, although some states that biophysically, a billion ha of extra tree cover is feasible without harming food production (Bastin et al 2019). How the afforestation is done will determine very much the net GHG balance.

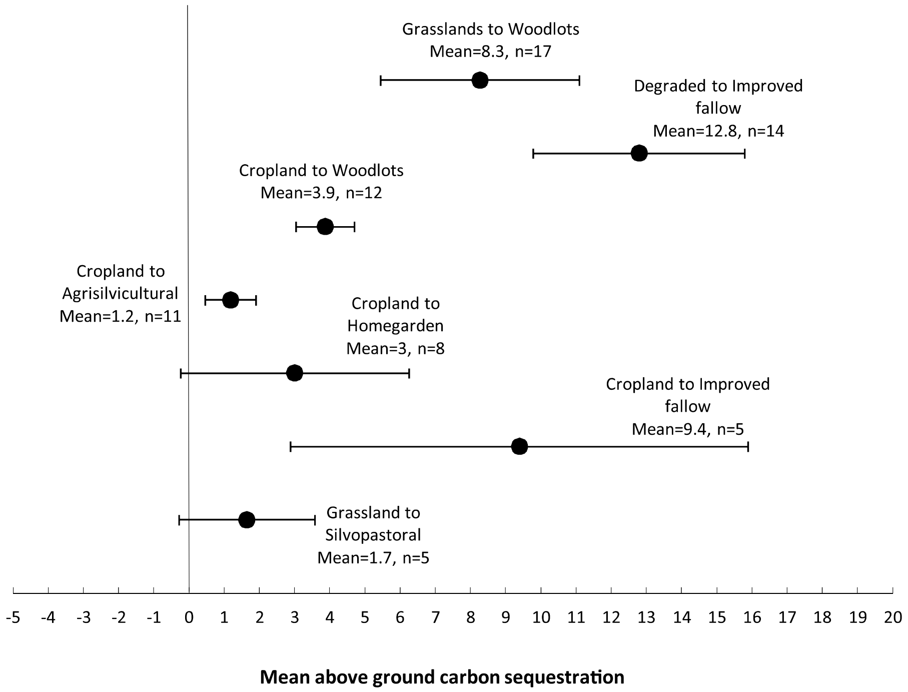
Planting at such scale could involve more release than uptake of greenhouse gases, at least initially, as a result of land clearance, soil disturbance and increased use of fertilizer. When such effects are taken into account, the maximum amount of CO_2 that can be removed by BECCS (under the RCP2.6 scenario) is estimated to be 391 Gt by 2100. This is about 34% less than the median amount assumed to be needed to keep the temperature rise below 2°C. If less optimistic but not unrealistic assumptions are made about where the land for bioenergy crops would come from, a net release of 135 Gt of CO_2 could occur by 2100 (Wiltshire and Davies-Barnard, 2015) with additional uncertainty about the effect of future climatic conditions on the yields of bioenergy crops (Gough and Vaughan, 2015).

7.5.6. Agroforestry systems

As an integrated production system, agroforestry is well positioned to provide agronomic solutions to improve resource efficiency for land, water, nutrients and energy . In this report (and consistent with other IPCC products), the term agro-forestry represents land-use systems which deliberately integrates woody perennials (e.g. trees and shrubs) and crops or grasses and/or animals on the same parcel of land with some form of spatio-temporal arrangement. This term may be used with variations this of meanings in other major global reports. There are three main types of agroforestry systems depending on the shared arrangement established with the trees: agrosilvicultural systems (integration of woody perennials with crops); silvopastorial system (integration of trees and shrubs in pasture with animals); and agrosilvopastoral systems (a combination of perennial crops, herbaceous crops and livestock) (Gebre 2016; FAO 2017).

Agroforestry provides responses to the two major objectives of climate change, adaptation and mitigation. Agroforestry C sequestration in the trees is highly significant under various ecosystems and if combined with soil carbon it become is the second best carbon sequestration option after forestry, with the advantage to address food security and various resilience aspects (Newaj et al. 2016; Mbow et al. 2014). Thus, the importance of agroforestry as a land-use system is receiving wider recognition not only in terms of agricultural sustainability but also in issues related to climate change.

According to a global study, more soil carbon sequestration occurs in agroforestry systems classified as silvopastoral (average of 4.38 tC ha^{-1} yr^{-1}), and more above ground carbon sequestration occurs in improved fallows (average of 11.29 tC ha^{-1} yr^{-1}) (Feliciano et al. 2017). The same study shows that on average, carbon benefits are greater in tropical agroforestry systems as compared to agroforestry other climatic zones. Biggest carbon potential in term of land sue transition occurs in above ground carbon sequestration (12.8 tC ha^{-1} yr^{-1}) when degraded land is replaced by improved fallow. As for the soil C and the greatest sequestration (4.38 tC ha^{-1} yr^{-1}) occurs when land is converted from a grassland system to a silvopastoral system (Feliciano et al. 2017). Rates of carbon storage for tropical tree crops production range from 1.8 to 10 Mg $ha^{-1}yr^{-1}$, compared with 0.6 Mg $ha^{-1}yr^{-1}$ for conservation agriculture. It is not only timber and agroforestry on croplands that sequester carbon. Tree crops provide the same service, while yielding food and other products.



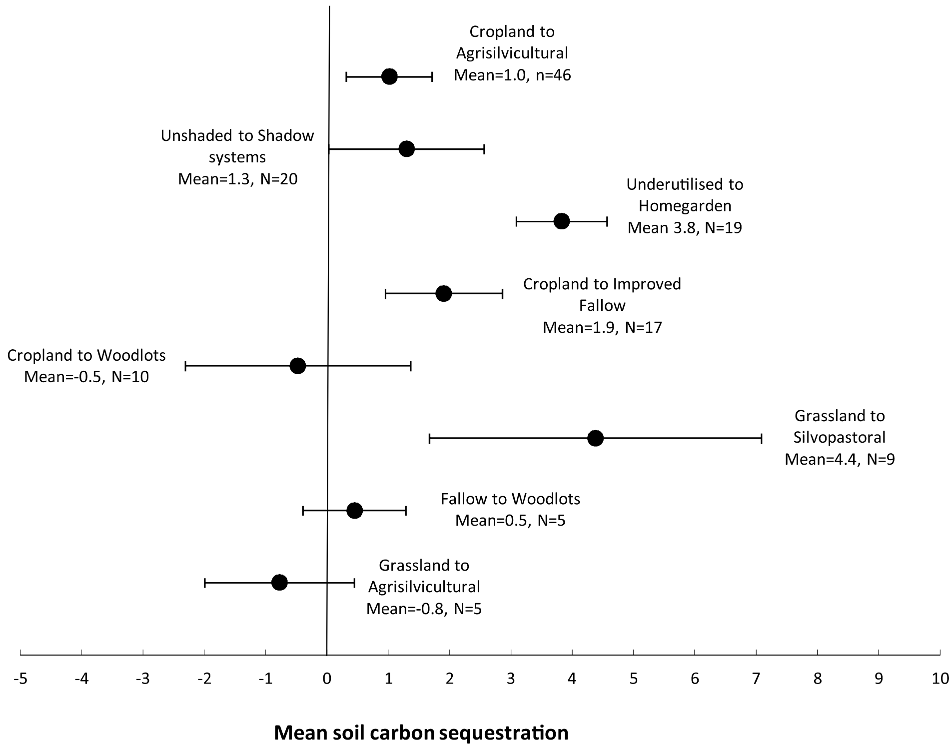


Figure 7.19 Mean absolute change in above ground carbon and below ground carbon sequestration resulting from the implementation of an agroforestry system. (Feliciano et al. 2017).

In terms of the consequences of land use transition on SOC change, (De Stefano and Jacobson 2018) noted a decrease in SOC stocks of 26 and 24% in the land-use change from forest to agroforestry at 0–15 and 0–30 cm respectively. Inversely, the study reported that the transition from agriculture to agroforestry significantly increased SOC stock of 26 % (at 0–15 cm depth), 40 % (at 0–30 cm), and 34% (at 0–100 cm).

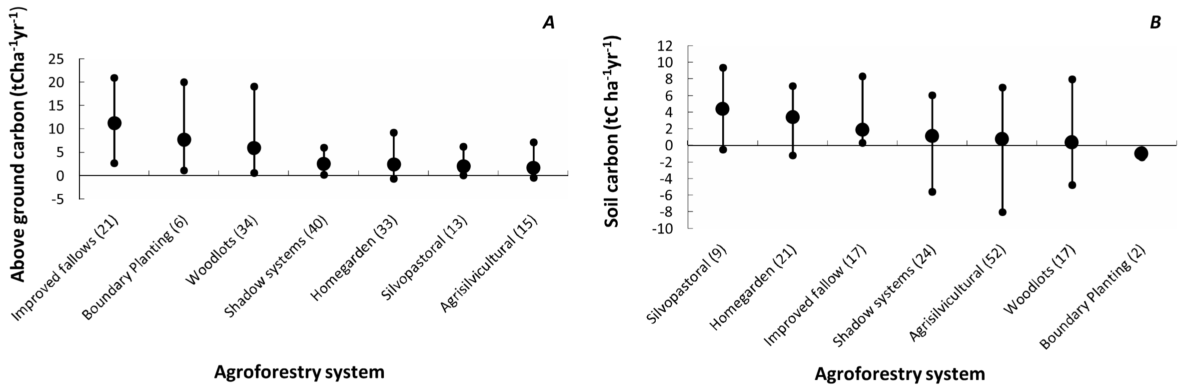


Figure 7.20 Mean, maximum and minimum above ground (A) and soil (B) carbon sequestration in different agroforestry systems. Number of observations (n) is presented in brackets. (Feliciano et al. 2017).

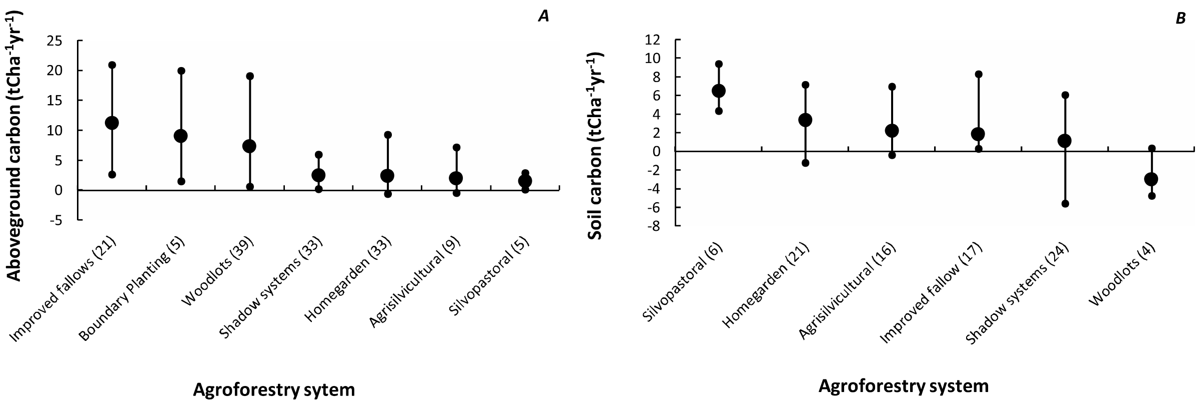


Figure 7.21 Mean, maximum and minimum above ground (A) and soil (B) carbon sequestration in agroforestry systems implemented in Tropical climates. Number of observations (n) is presented in brackets. (Feliciano et al. 2017).

Carbon sequestration in agroforestry systems can slow or even reverse the increase in atmospheric concentration of CO_2 by storing some SOC for a very long time (millennia). Carbon stocks, the global sequestration potential of increased adoption of tropical staple tree crops (trees producing protein, carbohydrates, and oils) has been estimated on degraded grassland (Hawken, 2017). Their model calculated a global sequestration impact of 20.2-46.7 Pg of CO_2 between 2020 and 2050.

According to Lorenz and Lal (2014), SOC storage in agroforestry systems is uncertain but may amount up to 300 Mg C ha^{-1} to 1 m depth and the above and below ground carbon can be estimated to up to 2.2 Pg C (1 Pg = 10^{15} g) over 50 years in agroforestry systems.

To maintain agroforestry SOC function in the global carbon budget, soil disturbance must be minimized while tree species with a high root biomass-to-aboveground biomass ratio and/or nitrogen-fixing trees should be encouraged in various agroforestry systems.

Special formation such as Long fallows over 25 years of Acacia Senegal can have carbon stocks over 100 t C ha^{-1} and soil with values ranging from 67.78 tC ha^{-1} to 89.24 tC ha^{-1}(Temgoua et al. 2018). For comparison, total carbon from cocoa plantation (soil+biomass+dead biomass) is estimated to be 117±47Mgha^{-1}, with 51Mgha^{-1} in the soil and 49Mgha^{-1} (42% of total carbon) in aboveground biomass (cocoa and canopy trees) (Somarriba et al. 2013).

7.5.7. Integrated crop-livestock systems

Integrated crop-livestock systems is a significant component of global livestock production and are associated with delivering copious environmental and economic benefits. Global livestock supply chains represent a significant contribution to total GHG emissions (Robb et al. 2017; Grossi et al. 2019). Nevertheless, it is estimated that emissions from CH_4, N_2O, and NO could be reduced by up to 30% if the most efficient integrated crop-livestock and integrated manure management practices are adopted globally (FAO 2015).

Integrated rice-fish systems comprise a range of aquatic species, including finfish, crustaceans, mollusks, insects, reptiles, amphibians and aquatic plants (Refs). Conventional rice farming is a major source of CH_4, N_2O emissions due to heavy fertilizer usage and anaerobic field conditions (Boateng et al. 2017; Susilawati and Setyanto 2018; Robb et al. 2017). On the other hand, integrated rice-fish systems are expected to deliver on reductions in CH_4, N_2O emissions through reduced fertilizer application and improved soil aeration by fish activity. However, data from these systems have largely been mixed (Fang et al. 2019; Hu et al. 2016).

7.5.8. Biochar

Biochars are produced by heating organic matter in an oxygen-limited environment, a process known as pyrolysis. Biochars can be made from a range of biomass feedstocks, including wood, urban greenwaste, manure, biosolids and straw. When used as a soil amendment biochars can offer benefits for climate change mitigation, and also enhance soil properties. The properties and effects of biochar vary depending on the feedstock and pyrolysis conditions.

Biochars are resistant to decomposition compared with unpyrolysed biomass and those 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 (*robust evidence, high agreement*) (Singh et al. 2012; Wang et al. 2016). Biochar stability is influenced by soil properties: biochar carbon can be further stabilized 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).

Furthermore, biochars can induce "negative priming", enhancing soil carbon stocks through stabilization of rhizodeposits via sorption of labile C on biochar surfaces and formation of biochar-organo-mineral complexes (Weng et al. 2015, 2017, 2018; Wang et al. 2016). Negative priming has been observed particularly in loamy and clayey soils (Ventura et al. 2015; Wang et al. 2016).

Additional climate change mitigation through application of biochars can result from decrease in nitrous oxide (N_2O) emissions from soil (*robust evidence, moderate agreement*). However, this impact varies widely: meta-analyses have found average decrease in N_2O emissions from soil of 9 to 54% (Cayuela et al. 2014, 2015; Song et al. 2016; He et al. 2017; Schirrmann et al. 2017; Verhoeven et al. 2017; Borchard et al. 2019). Effectiveness is inversely related to the molar H/C ratio of the biochar and therefore greatest for wood-derived biochars, with average 73% decrease (Cayuela et al. 2015). Substantial reductions, of 20-40% on average, have also been found in N_2O emitted from rice paddies (Song et al. 2016; Awad et al. 2018, Liu et al. 2018a). Biochar has also been observed to reduce methane emissions from rice paddies, though the effect is small on average, and again, results vary between studies, with increases observed in some situations (Song et al. 2016; He et al. 2017; Kamman et al. 2017; Awad et al. 2018).
Biochars can provide indirect climate benefits through increased yields (Biederman and Harpole, 2013), particularly in sandy soils and acidic tropical soils (Jeffrey et al. 2017); enhanced biological nitrogen fixation (Van Zwieten et al. 2015); avoided GHG emissions from manure handling, in-field burning of crop residues and landfilling of organic wastes; and reduced GHG emissions from compost

when biochar is added (Agyarko-Mintah et al. 2017; Wu et al. 2017a). Furthermore, biochars, especially those from woody feedstocks, can reduce requirements for GHG-intensive N fertilizer, due to reduced losses of N through leaching and/or volatilization (Liu et al., 2019; Borchard et al., 2019).

Biochar is acknowledged as a carbon dioxide removal strategy: the conversion of biomass to biochar stabilizes the carbon, delivering long term C storage when applied to soil; pyrolysis gases can be combusted for heat or power as a renewable energy source, displacing fossil fuels, and could be captured and sequestered through carbon capture and storage. Studies of the life cycle climate change impacts of biochar systems generally show emissions reduction in the range 0.4 -1.2 $tCO_2e\ t^{-1}$ (dry) feedstock (Cowie et al. 2015). Pyrolysis of biomass can deliver greater benefits than use for bioenergy alone, if used 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 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.

7.5.9. Demand-side measures

Harvested wood products

Harvested wood products (HWPs) are increasingly recognized as a means of carbon storage. Three primary wood products (i.e. sawn wood, wood-based panels as well as paper and paperboards) were estimated to store 73, 21 and 6% of carbon, respectively, with a trend of increasing share of the two latter products (FAO, 2016a; Palma et al., 2016). In 2013, the total amount of carbon stored in the three products was estimated as approximately 5360 Tg C (19,671 Gt CO_2e) (FAO, 2016a; Miner and Gaudreault, 2016).

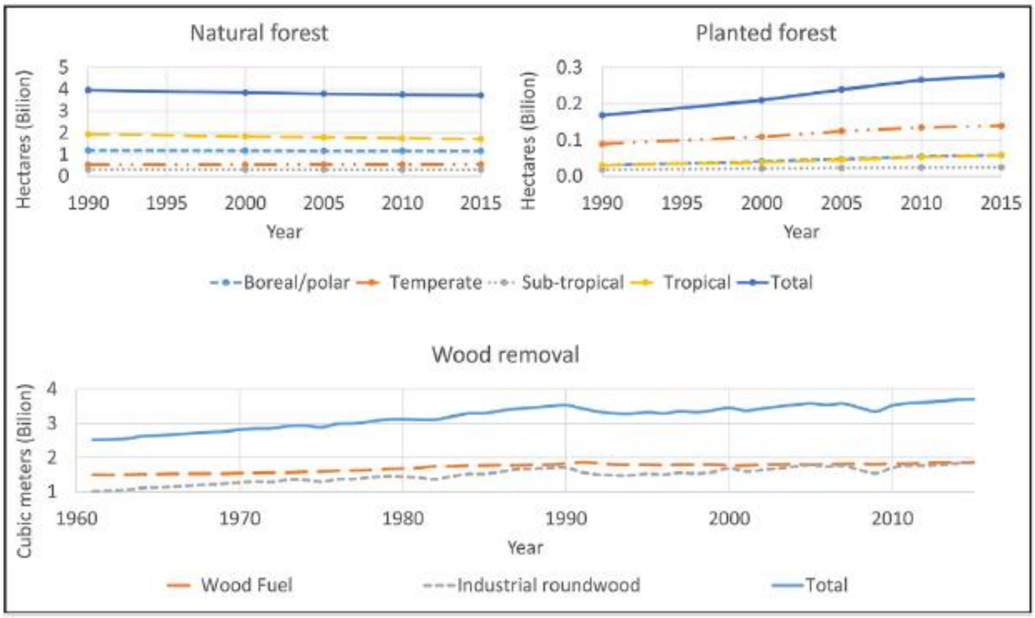


Figure 7.22 Natural forest area, planted forest area and global wood removal trends in the last decades Source: Marchi et al. (2018)

Wood and wood-based products have positive effects on climate change mitigation. These products can substitute for energy-intensive products made of conventional materials. For example, the quantification substitution benefits of wood furniture for China demonstrated that wood materials exerted low environmental impacts (Geng et al. 2019). From the less to the more wood intensity products with the equivalent function, the proportion of energy-intensive materials decreased 24% while the proportion of their greenhouse gas (GHG) emissions decreased 34%. The overall displacement factor for the wood material was 2.67 kg CO_2eq/kg or 1.46 tC/tC (Geng et al. 2019).

Focusing on wood use for piles, check dams, paved walkways, guardrails, and noise barriers, Kayo and Noda (2018) quantified the nationwide potential for climate change mitigation in civil engineering in Japan through 2050. A maximum nationwide avoided GHG emissions potential of 9.63 million tCO_2eqyr^{-1} could be achieved in 2050, which is equivalent to 0.7% of Japan's current GHG emissions. The breakdown of avoided emissions is 73%, 19%, and 8% for carbon storage, material substitution, and energy substitution, respectively, with the greatest contributions coming from carbon storage through the use of log piles.

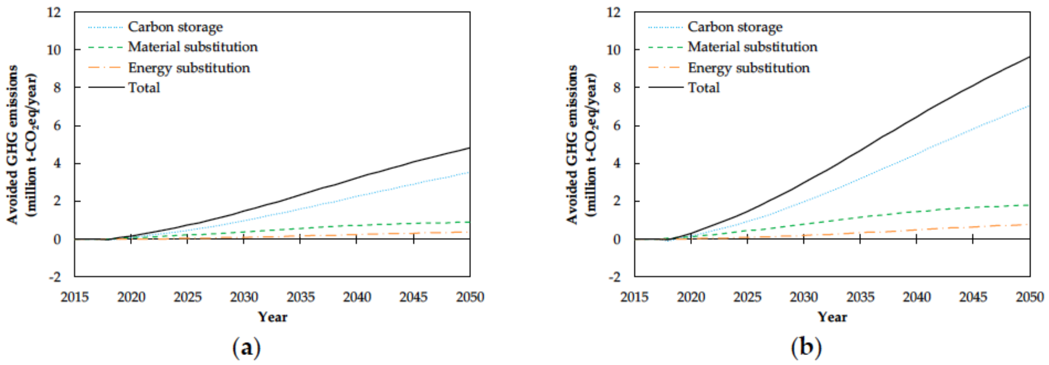


Figure 7.23 Climate change mitigation potential by using wood in civil engineering in Japan through 2050: (a) likely potential; and (b) maximum potential.

On the other hand, it cannot be assumed that wood demand confers climate benefits. In the absence of best practices implemented as part of a CSF approach, increased demand for wood can have perverse climate outcomes. For example, rapid expansion of timber plantations in Indonesia is the largest driver of the loss of higher carbon native forests in that country (Abood et al. 2014). The assumption that wood products confer climate benefits has been questioned by Keith et al. (2015) and Schlesigner (2018), indicating that much depends on how and where wood products are sourced. Likewise, Griscom et al. (2018) find large differences in emissions per m3 of wood produced depending on logging practice used, management intensity, and the relationship between logging activity and subsequent forest conversion. All of these studies point to the importance of robust safeguards, including improved chain-of-custody tracking and regulations on climate smart wood sourcing.

Measures in food systems

Increasing food production to meet the demands of the ever-growing population either by expanding croplands or the intensification of production contributes significantly to GHGs, deteriorating soil quality and biodiversity loss (Johnson et al., 2014; Smith et al., 2013). Approaches to mitigating climate change impacts and achieving food security in an environmentally sustainable way have considered the role of reducing food loss and waste (Hall et al., 2009; Gustavsson et al., 2011; Smith, 2013); diet shifts and changes and other consumer demand-related measures (Lamb et al., 2016; Stehfest et al., 2009).

Reducing losses in the food supply chain

Gustavsson et al., (2011) report that nearly one-third of global food is lost or wasted from production to consumption, and Porter and Reay (2016) estimate that the quantity of this waste has tripled between 1960 and 2011. Many studies have established that food loss and waste contribute considerably to GHG emissions and therefore a better management of FLW may play a substantial role in climate change mitigation (e.g. Kummu et al., 2012; Foley et al., 2011; Smith, 2013; West et al., 2014). Because of cumulative food losses, the proportion of global agricultural dry biomass that is consumed as food estimated at only 6%, and around 25% of harvest biomass (Alexander et al., 2017). Further, losses of harvested crops are also substantial while nearly 44% of crop dry matter is lost before human consumption. Another study by Porter et al. (2016) show that the annual global food loss and waste have grown during the past five decades from 540Mt in 1961 to 1.6 Gt in 2011. Over the same period, global capita food loss and waste related emissions increased from 225 kg CO_2e to 323 kg CO_2e. From a regional perspective, the results of Porter at al. (2016) point out that the share of developed countries in global GHG emissions due to food loss and waste declined from around 48% in 1961 to about 24% in 2011. In contrast, GHG emissions from developing countries due to food loss and waste have grown remarkably, with China and Latin American countries contributing significantly to this growth. Cumulatively, these emissions added around 70 Gt CO_2e to the atmospheric GHG stock, which correspond to the total of two years' emissions from all anthropogenic sources at the current rates. At the commodity groups' level, a study by FAO (2013) reveals that cereals, vegetables and meat represent the main food groups where food loss and waste matter the most to GHG emissions. Springmann et al. (2018) estimate that halving FLW would reduce environmental pressures by 6–16% compared with the baseline projection for 2050, and that reducing food loss and waste by 75% would reduce environmental pressures by 9–24%.

With regard to low-income countries, Niles et al. (2018) show that the most of food loss and waste take place during the pre-consumption stages (e.g. production, post-harvest, transportation and processing). Improper storage, poor infrastructure and lack of cold-chain refrigeration, transportation means and inefficient processing facilities and techniques are the main causes for food losses in these countries. Therefore, reducing the amount of food loss and waste in developing and low-income countries can be achieved by improving post-harvest, transportation and processing infrastructure. In developed and high-income countries, food is mostly wasted at the retail and consumption stages (Blanke, 2015). According to Quested et al. (2013) reducing food waste in developed countries needs to consider human behavioral aspects by changing consumer perceptions about food validity, reducing overstocking and portion sizes in restaurants, by utilizing processing and packaging technologies (Niles at al, 2018; Schanes et al., 2016; Wilson et al., 2017). Further, landfills where food is disposed is another source of methane emissions and it is ranked as the third largest source of global methane emissions. Diverting food waste from human consumption for animal feed, and composting and employing methane capture technologies present opportunities to reduce methane emissions from landfills (Krause et al., 2016).

Shifting diets

In many regions of the world, diets are consistently shifting toward more animal-source foods, processed and packaged foods, and more energy intake. Thus, several studies have suggested that a better management of human diets can be more effective in reducing GHG emissions than technical agricultural mitigation options (Pradhan et al., 2013; Nemecek et al., 2016; Macdiarmid, 2013; Smith et al., 2013; Bajzelj et al., 2014). For instance, it is estimated that low-meat diets would reduce the mitigation costs to the recommended healthy levels can halve mitigation costs needed to achieve the 450 ppm CO_2 eq. target compared to business as usual (Stehfest et al., 2009). Alexander et al. (2017) show that the replacement of meat products by cultured meat, imitation meat or insects could potentially reduce the demand for agricultural land by up to 38% based on current consumption levels of 2011, assuming a 50% replacement of meat products by these substitutes. Springmann et al. (2016) reported that compared with a reference scenario in 2050, a shift toward more plant-based diets that are in line with standard dietary guidelines could decrease global mortality by up to 10% and reduce greenhouse gas emissions from food sources between 30% and 70%. Moreover, Griscom et al. (2017) reveal that around 40% of the global maximum reforestation mitigation potential relies on improving the efficiency of beef production and promoting lower beef consumption, and hence reducing the demand for pasture. Ritchiea et al. (2018) assessed the implications of dietary guidelines on global GHG emissions and found that a wide disparity in the emissions intensity of recommended healthy diets, ranging from 687 kg of carbon dioxide equivalents (CO_2e) capita^{-1} yr^{-1} for the guideline Indian diet to the 1579 kg CO_2e capita^{-1} yr^{-1} in the USA. Most of this variability was introduced in recommended dairy intake. Global convergence towards the recommended USA or Australian diet would result in increased GHG emissions relative to the average business-as-usual diet in 2050. The majority of current national guidelines are highly inconsistent with a 1.5 °C target, and incompatible with a 2 °C budget unless other sectors reach almost total decarbonization by 2050. Effective decarbonization will require a major shift in not only dietary preferences, but also a reframing of the recommendations which underpin this transition. According to Springmann et al. (2018), dietary changes towards healthier diets could reduce GHG emissions and other environmental impacts- compared with the baseline projection for 2050- by 29% and 5–9%, respectively, for the dietary-guidelines scenario, and by 56% and 6–22%, respectively, for the more plant-based diet scenario.

7.6. AFOLU Integrated Models and Scenarios

This section assesses the literature and data available on potential future GHG dynamics in the AFOLU sector, the cost-effectiveness of different mitigation measures, and consequences of climate change mitigation pathways on land-use dynamics as well as relevant sustainable development indicators at the regional as well as global level.

Land-based mitigation options interact, and thus need to be assessed together and in addition in the interaction with mitigation options in other sectors and in combination with other sustainability goals (Popp et al. 2014, Popp et al 2017, Obersteiner et al. 2016, Humpenöder et al. 2017, Hasegawa 2019, Roe et al 2019, van Vuuren et al 2019). Integrative land-use models (ILMs) combine different land-based mitigation options and are partially included in Integrated Assessment Models (IAMs) which combine insights from various disciplines in a single framework and cover the largest sources of anthropogenic GHG emissions from different sectors. Over time, ILMs and IAMs have extended their system coverage (Johnson et al 2019), however, the explicit modeling and analysis of integrated land-use systems is relatively new compared to other sectoral assessments such as the energy system (IPCC 2019). In consequence, ILMs as well as IAMs differ in their portfolio and representation of land-based mitigation options, the representation of sustainability goals other than climate action as well as the interplay with mitigation in other sectors (Johnson et al 2019, van Soest et al 2019). These structural differences have implications for the regional and global deployment of different mitigation options as well as their sustainability consequences.

As a consequence of the relative novelty of land-based mitigation assessment in ILMs and IAMs, the portfolio of land-based mitigation options does not cover the full option space as outlined in 7.5, at least not in all assessments. The inclusion and detail of a specific mitigation measure differs across models and is influenced by the availability of data for its techno-economic characteristics and future prospects as well as the computational challenge, e.g. in terms of spatial and process detail, to represent the measure. Terrestrial Carbon Dioxide Removal (CDR) options are only partially included in ILM and IAM analyses, which mostly rely on afforestation and bioenergy with CCS (BECCS). For example, most of the ILM and IAM scenarios based on the Shared Socio-economic Pathways (SSPs) (Riahi et al 2017) which are the basis for the CMIP6 exercise, the IPCC Special Report on Land (SRCCL) (IPCC 2019) and the IPBES global assessment provide five different stories of future socio-economic development, including possible trends in agriculture and land use, cover a reduced set of land-based mitigation options. This set usually contains

- 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 of biofuels,
- avoided deforestation,
- afforestation,
- bioenergy and BECCS (Popp et al. 2017, van Meijl et al. 2018, Frank et al 2019).

Hence, there are mitigation options not being broadly included in integrated pathway modelling, especially nature based solutions (Griscom et al 2017, Roe et al 2019) such as soil carbon management or wetland management which have the potential to alter the contribution of land-based mitigation in terms of timing, potential and sustainability consequences. In contrast to the IPCC Special Report on Land (SRCCL) (IPCC 2019) as well as chapter 3 in WG3 AR6 this subchapter assesses future GHG dynamics in the AFOLU sector, the contribution of the AFOLU sector to climate change mitigation pathways and its consequences on land-use dynamics as well as relevant sustainable development indicators not only for the global dimension but also at the level of five world regions: Asia (ASIA), Latin America and Caribbean (LAM), Middle East and Africa (MAF), Developed Countries (OECD 90 and EU) (OECD) and Reforming Economies of Eastern Europe and the Former Soviet Union (REF) In addition, this chapter investigates the relevance and value of single mitigation options in the interplay with underlying drivers as well as with other mitigation options.

In addition to a general evaluation of the scenarios available to this assessment (Ref to AR6 database), a set of possible mitigation pathways have been identified which are illustrative of a range of possibilities in their GHG and land-use consequences (especially related to their use of terrestrial CDR such as bioenergy) as well as their consequences for sustainable development at both the global as well as the regional level. They vary due underlying socio-economic and policy assumptions, mitigation options considered, 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.

Regional GHG emissions and land dynamics

In most of the assessed mitigation pathways, the land sector is of great importance for climate change mitigation as it (i) turns from a source into a sink of atmospheric CO_2 due to large-scale afforestation and reforestation, (ii) provides high amounts of biomass for bioenergy or BECCS and (iii), even under improved agricultural management, still delivers residual non-CO_2 emissions from agricultural production and (iv) interplay with sustainability dimensions other than climate action (Popp et al 2017, Rogelji et al. 2017, van Vuuren et al. 2018, Frank et al. 2018, van Soest et al 2019). Regional AFOLU GHG emissions in the Baseline scenarios as shown in Fig 7.24 are shaped by CH_4 and N_2O emissions, mainly from ASIA, MAF and OECD. CH_4 emissions from enteric fermentation are largely caused by MAF, followed by ASIA, while CH_4 emissions from paddy rice production are almost exclusively caused by ASIA. N_2O emissions from animal waste management are more equally distributed across region, while N_2O emission from soils come mainly from ASIA and OECD.

CH_4 and N_2O emission are both lower in 1.5°C and 2°C mitigation pathways. However, the reduction of CH_4 emissions, in particular from enteric fermentation in ASIA and MAF, is more profound. Land-related CO_2 emissions, which include emissions from deforestation as well as from afforestation, are slightly negative compared to rather zero in the Baseline scenarios. Carbon sequestration via BECCS is most prominent in ASIA, LAM and OECD, which are also the regions with the highest bioenergy area.

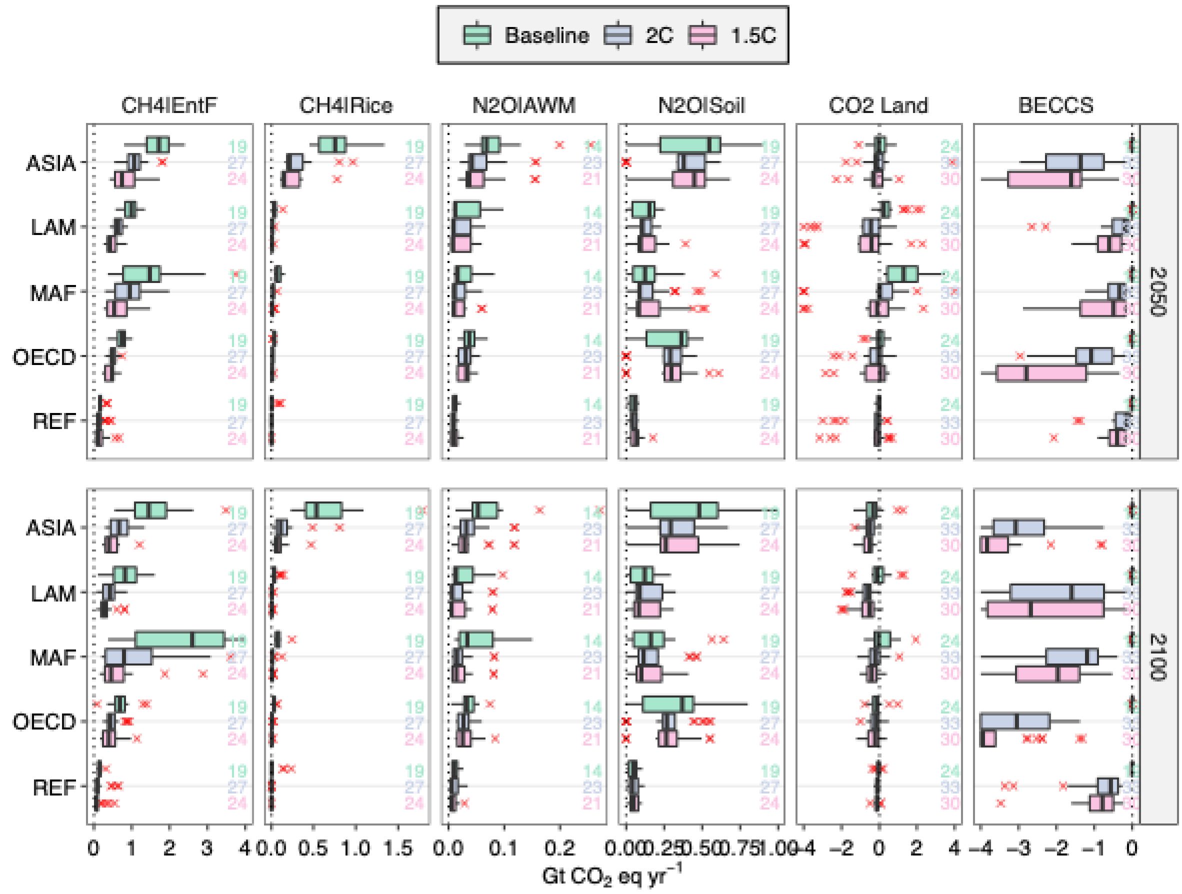


Figure 7.24 Land-based regional GHG emissions and removals in 2050 and 2100 for Baseline, 1.5°C and 2°C scenarios based on the Shared Socioeconomic Pathways (SSP) (Popp et al. 2017; Rogelj et al. 2018; Riahi et al. 2017). Data is from an update of the IAMC Scenario Explorer developed for the SR15 (Huppmann 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 shown include CH_4 emissions from enteric fermentation (EntF) and rice production (RICE), N_2O emissions from animal waste management (AWM) and fertilization (SOIL). The category CO_2 Land includes GHG emissions from land-use change as well as negative emissions due to afforestation/reforestation. BECCS reflects the CO_2 emissions captured from bioenergy use and stored in geological deposits. ASIA = Asia, LAM = Latin America and Caribbean, MAF = Middle East and Africa, OECD = Developed Countries (OECD 90 and EU), REF = Reforming Economies of Eastern Europe and the Former Soviet Union. [Currently based on SSP database - will be updated to AR6 database]

Figure 7.25 indicates that regional land use dynamics in the Baseline scenarios are characterized by decreasing agricultural land (i.e. cropland and pasture) in ASIA, rather static agricultural land in LAM, OECD and REF, and increasing agricultural land in MAF. Bioenergy area is relatively small in all regions. Agricultural land in MAF expands at the cost of forests and other natural land.

The overall land dynamics in 1.5°C and 2°C mitigation pathways are shaped by land-demanding mitigation options such as bioenergy and afforestation, in addition to the demand for other agricultural and forest commodities. The most important regions for bioenergy production and afforestation are ASIA, LAM and OECD. Bioenergy and afforestation area expand at the cost of agricultural land for food production (cropland and pasture) and other natural land. For instance, bioenergy and afforestation area together increase by X Mha between 2010 and 2100 in ASIA, while agricultural land and other natural land decline by Y and Z Mha respectively. Such large-scale transformations of land use have repercussions on biogeochemical cycles (e.g. fertilizer and water) but also on the economy (e.g. food prices) (see subsection on SDGs below).

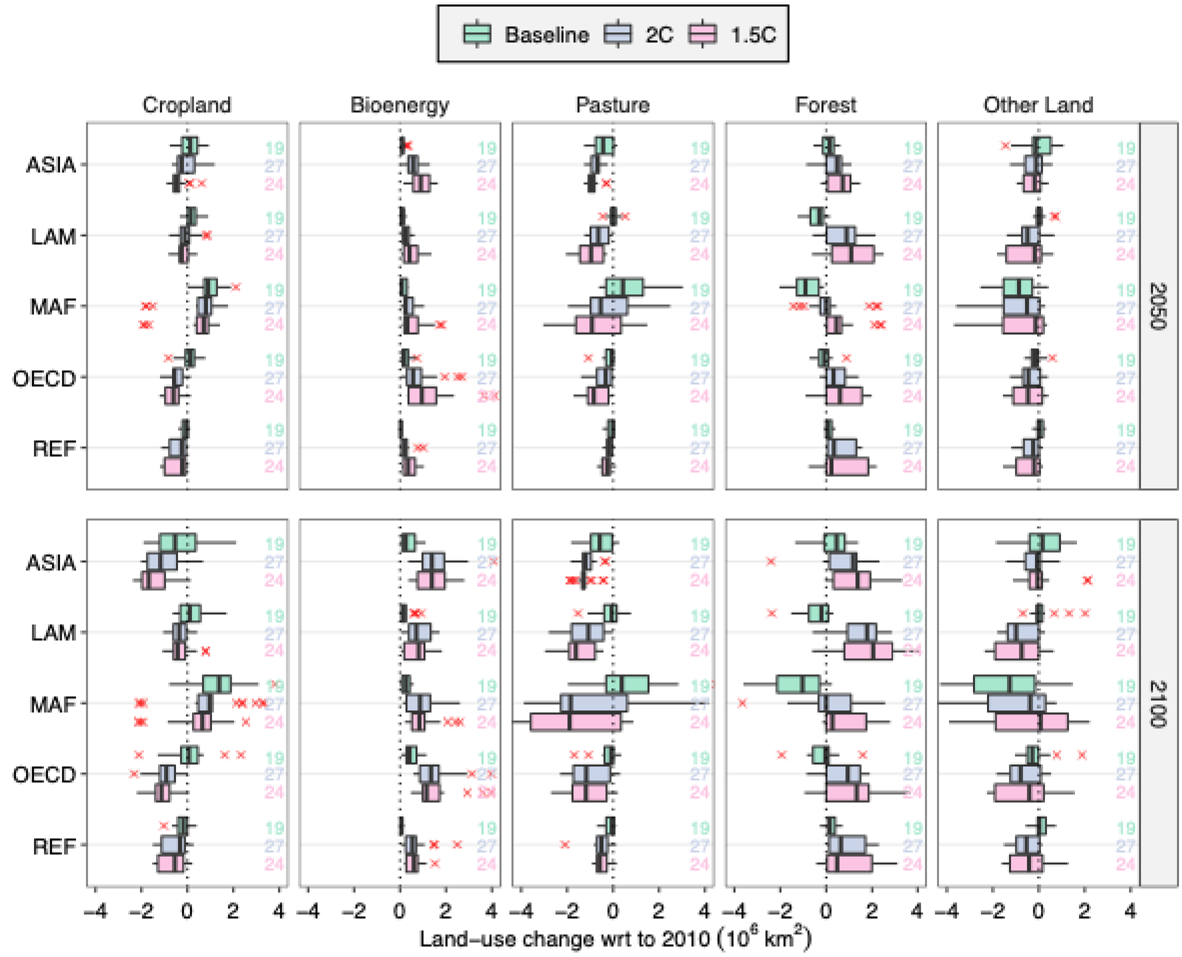


Figure 7.25 Regional change of major land cover types by 2050 and 2100 relative to 2010 for Baseline, 1.5°C and 2°C scenarios based on the Shared Socioeconomic Pathways (SSP) (Popp et al. 2017; Rogelj et al. 2018; Riahi et al. 2017). Data is from an update of the IAMC Scenario Explorer developed for 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 1 (Popp et al. 2017). ASIA = Asia, LAM = Latin America and Caribbean, MAF = Middle East and Africa, OECD = Developed Countries (OECD 90 and EU), REF = Reforming Economies of Eastern Europe and the Former Soviet Union [Currently based on SSP database - will be updated to AR6 database]

Top down marginal abatement cost curves

In our analysis we use the IAM results from the AR6 database to derive some top-down marginal abatement cost curves (MACCs) to reflect on the economic mitigation potential of different measures at the global level and for the R5 regions.

For this purpose, we calculate the following variables from the AR6 database:

- Reduction of agricultural non-CO_2 emissions (CH_4 + N_2O in CO_2eqyr^{-1}) in Scenario S_i with Carbon price P_i in comparison with a no policy scenario from the same set of scenarios run by the same model (i.e., for example all CD-Links scenarios run by REMIND-MAGPIE in comparison with "CD-LINKS_NoPolicy" run by REMIND-MAGPIE)

- CO_2 sequestration from Bioenergy with Carbon Capture & Storage (BECCS) in Scenario S_i with Carbon price P_i in comparison with a no policy scenario from the same set of scenarios
- Emissions reduction / sink enhancement from land use change (CO_2) in Scenario S_i with Carbon price P_i in comparison with a no policy scenario from the same set of scenarios
- Emissions reduction / sink enhancement from AFOLU+BECCS as an aggregation of the other three individual categories.

These variables are plotted against the respective carbon price P_i (compare Figure 7.26). At this stage the figures include pooled data from scenarios SSP1 – 5, the ADVANCE scenarios and the CD_Links scenarios, further scenario sets will be included as they become available in the AR6 database.

From the mentioned scenarios only variables have been included that fulfill the requirement that the sum of the 5 regions equals the global results for that specific variable. Furthermore, we only use results up to the threshold of a carbon price of 1000US$(2010) per ton of CO_2eq since a higher price is considered as unrealistic for the AFOLU sector.

The error bounds in the graphs reflect significance levels of 33% (least transparent part), 66%, 90% and 99% (most transparent part) in line with the thresholds used for likelihood assessment in IPCC reports. MACCs have been calculated for four snapshots in time: 2030, 2050, 2070 and 2100.

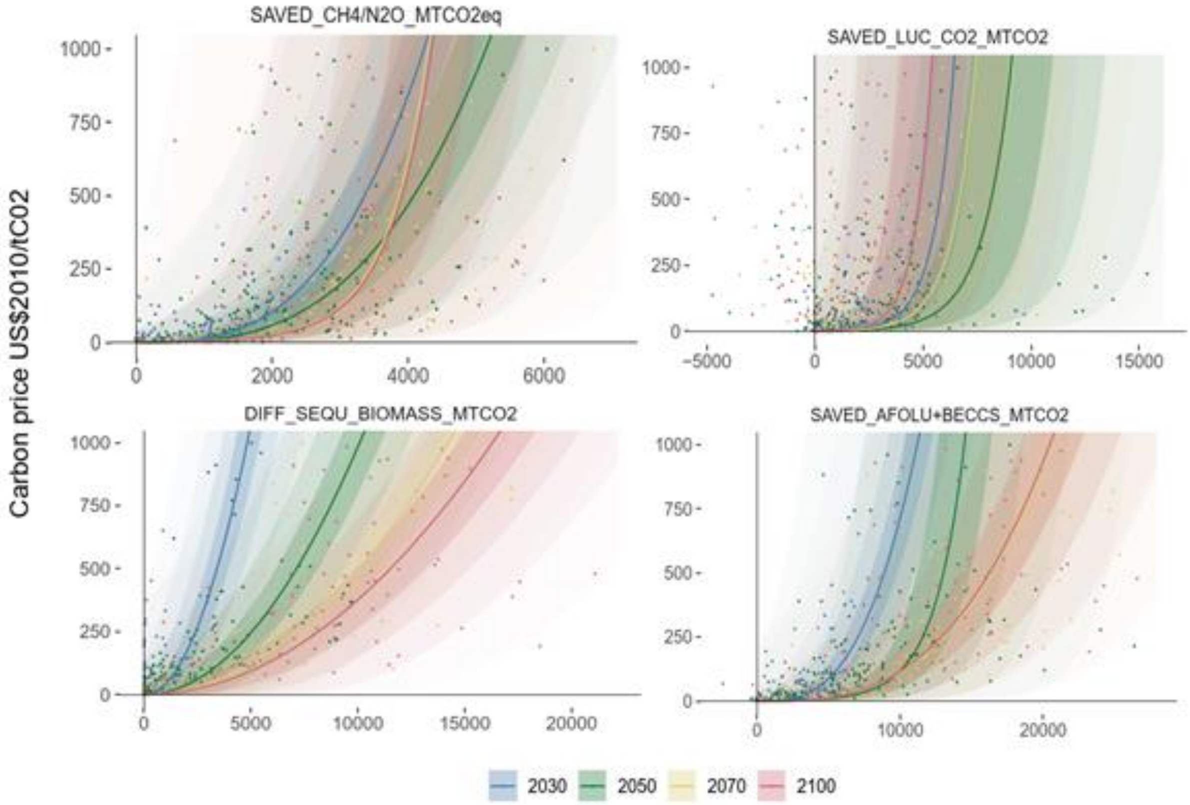


Figure 7.26 Global MACCs derived from AR6 database

No clear distinction pattern between the different time steps can be recognized for the mitigation potentials of non-CO_2 emissions from the agricultural sector. Looking at the trend lines, it seems that in later years a certain threshold of mitigation cannot be exceeded, but with the given overlapping confidence intervals, no robust conclusion can be drawn regarding the mitigation potential of different time steps.

Similarly, no robust finding can be revealed from the MACCs for the land use change emissions. However, the trend line representing the MACC for the year 2100 shows the lowest potential, which would coincide with expectations since most baselines show already a declining trend in land use change emissions and thus, the potential to reduce emissions declines towards the end of the century. A clearer trend can be observed for the sequestration potential from BECCS. It seems that most models assume decreasing costs for this emerging technology towards the end of the century and hence increased economic potential.

From the graph combining the three single variables 'mitigation of land use change emissions', 'mitigation of agricultural non-CO_2 emissions' and 'sequestration via BECCS' (bottom right), the largest potential - given a certain carbon price - can be observed in the later years (2070 and 2100) due to the domination of the BECCS effect in the aggregate.

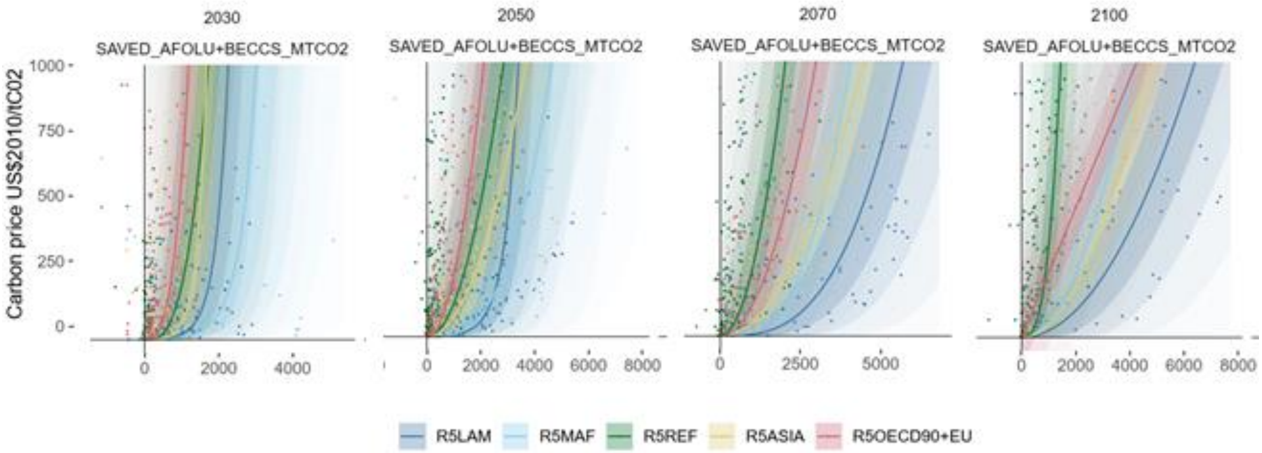


Figure 7.27 Regional MACCs derived from AR6 database showing the combined mitigation potential for agriculture, land use and BECCS

In Figure 7.27 MACCs are presented for 5 global regions and 4 different time steps. While in the earlier years curves are relatively steep, indicating a limited mitigation potential, for most regions (except for R5REF) the curves are getting flatter, indicating a higher mitigation potential over time. Especially in Latin America the mitigation potential seems to increase.

Several caveats of our analysis must be mentioned. We do not use the full set of available scenarios, yet. Adding more scenarios has been started, however, it seems to increase the level of uncertainty even more. For example, the EMF33 scenarios seem to have set-ups that allow for BECCS in some runs and do not allow for BECCS in others, thus generating systematically 2 results for the same carbon price.

Furthermore, ad-hoc testing has shown that results are sensitive to adding or removing certain models. The removal of GCAM from the sample would change some of the relations of the yearly curves. Here, more systematic testing is necessary to understand the underlying relations better. A suggestion for further work would be to calculate the cumulative mitigation potential to get a clearer picture of the time dynamics of mitigation potentials.

Illustrative pathways

Different mitigation strategies can achieve the net emission reductions that would be required to follow a Pathway that limits global warming, with very different consequences on the land system. Figure 7.28 shows illustrative pathways for achieving 1.5C climate targets highlighting AFOLU mitigation strategies, resulting GHG and land use dynamics as well as required agricultural intensification and consequences for food prices. For consistency this chapter discusses alternative pathways aligned to the IPCC special report on 1.5 degree as well as SRCCL (IPCC 2019) but focusing besides global emission and land-use consequences on regional dynamics and consequences on food security. All pathways are assessed by different IAMs but are all based on an RCP 1.9 mitigation pathway (Rogelj et al. 2018). All scenarios use Carbon Dioxide Removal (CDR), but the amount varies across pathways, as do the relative contributions of different land-based CDR options.

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.

Illustrative Pathway 2 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 $GtCO_2$).

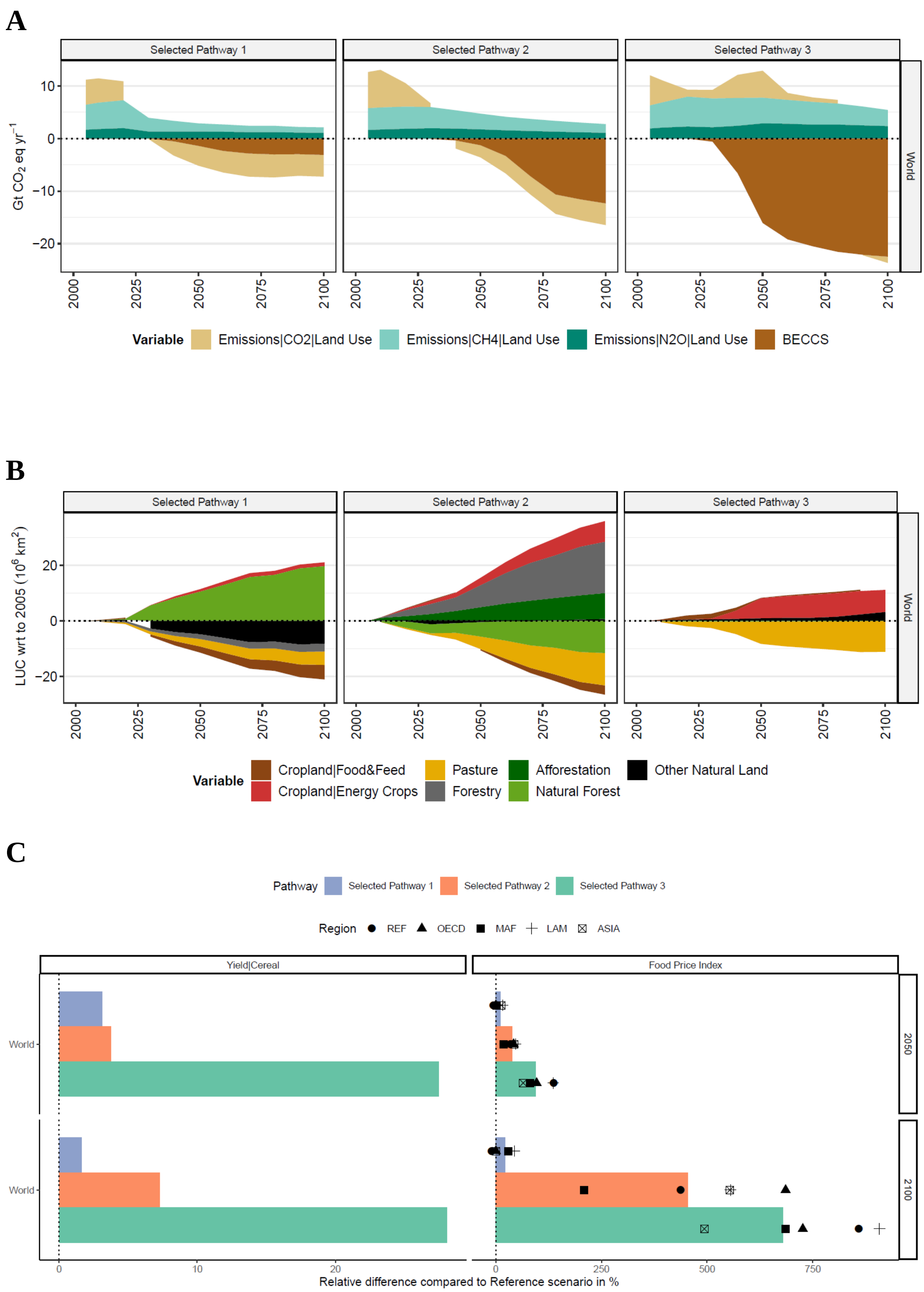


Figure 7.28 Evolution and break down of (A) global land use dynamics, (B) global land-based GHG emissions and removals and (C) regional and global needs for intensification as well as food security consequences under three illustrative mitigation pathways, which illustrate the differences in timing and magnitude of land-based mitigation approaches including afforestation and BECCS. All pathways are

based on different IAM realisations for RCP1.9 (Rogelj et al. 2018): Pathway 1: SSP1 from AIM (Fujimori et al. 2017); Pathway 2: SSP2 from MESSAGE-GLOBIOM (Fricko al. 2017); Pathway 3: REMIND-MAgPIE (Kriegler et al. 2017); In panel A 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. CH_4 and N_2O emissions are converted to CO_2-eq using GWP factors of 28 and 265 respectively.

7.7. Assessment of economic, social and policy responses

7.7.1. Success of policies in the past 20 years

We have quantified around 7.8 Gt CO_2 sequestered directly due to policies and measures implemented for climate change mitigation through the CDM, REDD+, and other policies. This amounts to 8% of the net carbon sink between 1992 and 2012, which was estimated to be 94.3 Pg CO_2 (IPCC WG1 Chapter 6 AR5). In addition to the direct policy drivers that have increased forest carbon stocks, economic drivers have also contributed to the increase in global forest carbon stocks. Mendelsohn and Sohngen (2019) for instance, show how forest management responded to market incentives to add 13.8 Gt CO_2 to global forest stocks between 1992 and 2012.

Numerous policy approaches have either been used directly to sequester carbon, or have indirectly led to increased carbon sequestration in forests. In tropical settings, perhaps the most widely used approaches for reducing deforestation are the establishment of parks or protected areas, and Payments for Ecosystem Services (PES) programs. The evidence on whether these programs have reduced deforestation, and hence, forest carbon emissions, are mixed (Alix-Garcia et al., 2012; Andam et al., 2008; Robalino and Pfaff, 2013; Sims and Alix-Garcia, 2017; Wunder and Albán, 2008).

Data in temperate regions suggests that PES programs can improve carbon sequestration. For example, in China, increases in carbon sequestration have also been attributed to implementation of a number of policies aimed at increasing ecosystem services, including payments for ecosystem services and establishment of parks and protected areas (Liu et al., 2008; Ouyang et al., 2016). In the US, conservation programs through the US Department of Agriculture, like the Conservation Reserve Program are PES programs that increase carbon sequestered in soils and plant material. These programs likely have contributed to the large and persistent US land-based carbon sink in cropland, grassland and forests (US Environmental Protection Agency, 2019), although the size of their contribution to the current sink has not been estimated. Other countries, like those of the European Union, also have policies that have influenced forest and land based carbon outcomes, including rules and regulations related to harvesting, planting, and forest fire management.

Regulatory approaches have also been used. There is evidence that regulatory approaches and forest and agricultural supply chain management helped slow deforestation in Brazil between 2004 and 2010 from over 2 Mha yr^{-1} to around 0.5 Mha yr^{-1} (Nepstad et al., 2014). Similarly, Roopsind et al. (2019) show that country-level REDD+ payments to Guyana encouraged regulatory and other policies in the country that reduced deforestation over the payment period and increased carbon storage. Efforts to reduce deforestation and subsequent carbon emissions in both of these countries appear to have been sustained during a period of policy implementation, which was later relaxed.

In tropical regions, property rights have long been insecure because the governments that own forests do not have the resources to control all activities on widespread forested ecosystems. Over the last 20 years, however, property rights and efforts to provide property rights, especially to groups that will engage in community forest management, have expanded across the globe. According to the Rights and Responsibilities Initiative (RRI, 2014), the area of forests under community management increased globally by 127 Mha from 2002 to 2012, with over 500 million hectares under community management (of some sort) in 2012. A number of studies have now shown that improved property rights with community forest management can reduce deforestation and hence increase carbon relative to a counter-factual (e.g., Alix-Garcia, 2007; Alix-Garcia et al., 2005; Blackman, 2015; Deininger and Minten, 2002, 1999; Fortmann et al., 2017). Given this, it is likely that efforts to expand property rights, especially community forest management, have reduced carbon emissions from deforestation in tropical forests in the last two decades. The amount has been quantified in specific locations (e.g., Fortmann et al., 2017), but not globally.

Forest certification programs, such as Forest Sustainabiltiy Council (FSC) or Programme for the Endorsement of Forest Certification (PEFC) may also help reduce emissions with reduced impact logging and other approaches. These programs involve the manager of a forest concession working with one of the major certification groups to develop management plans for forests that reduce the impact of harvesting on ecosystems. These initiatives are largely consumer driven and voluntary. They have expanded globally according to the UN FAO to over 415 Mha (MacDicken et al., 2016). As the area of land devoted to certification has increased, the amount of timber produced from certified land has increased. State of the World's Forests (FAO, 2018) indicates that in 2018, FSC accounted for harvests of 427 million m^3 and jointly FSC and PEFC accounted for 689 million m^3 in 2016, when adjusted for double counting.

There is evidence that sustainable timber harvesting, such as reduced impact logging, reduces forest carbon emissions from logging sites relative to other practices (e.g., Griscom et al., 2014; Nasi et al., 2011; Putz et al., 2012; Roopsind et al., 2018), suggesting that the current implementation of these practices have already reduced emissions, and more widespread implementation of these practices could provide long-term benefits to the atmosphere. A key reason for the reduction in carbon emissions is that reduced impact logging practices reduce emissions from damage caused to parts of the forest that are not logged (Pearson et al., 2014). The costs for implementing these practices (e.g., Cubbage et al., 2009), however appear greater than the market benefits currently (e.g., Yamamoto et al., 2014), suggesting that despite the widespread implementation of certification programs, the net costs are positive. Although these practices reduce carbon emissions from timber harvesting there is mixed evidence that certification programs reduce carbon emissions from deforestation (Blackman et al., 2018; Miteva et al., 2015).

Assessment of Current Policies

The role of forests for climate change mitigation has strongly been recognized in the Paris Climate Agreement. The Agreement endorses Reduced Emissions from Deforestation and Forest Degradation (REDD+), sustainable forest management, allows for alternative (non-market) policy approaches such as joint mitigation and adaptation, and emphasizes the importance of non-carbon benefits and equity for sustainable development (Martius et al. 2016). A review of the diverse and evolving approaches to REDD+ pointed out that the level of financing was insufficient (Streck, 2012). While significant resources have been invested in CDM projects, REDD+ projects, voluntary and compliance markets for land-based carbon offsets, there is a large funding gap between these efforts and the scale of efforts necessary to meet 1.5 or 2.0°C targets. Current funding is less than $1 billion per year. In order to achieve the goals outlined in the Special Report on warming of 1.5 degrees C, Austin et al. (2019) suggest that forestry actions that could achieve up to 5.8 $GtCO_2\ yr^{-1}$ would cost $431 billion yr^{-1}. Over half of this investment is expected to occur in Latin America, with 13% in SE Asia and 17% in Sub-Saharan Africa. Other studies have suggested similar ranges for the average cost of carbon sequestration in tropical countries (e.g., Griscom et al., 2017 and Busch et al., 2019). Although current efforts have achieved significant outputs, society will need to quickly ramp up investments in order to achieve carbon sequestration levels consistent with high levels of mitigation. Only 2.5% of climate mitigation funding goes to land-based mitigation options, an order of magnitude below the potential proportional contribution (Buchner et al. 2015).

Compliance markets have the potential to increase the demand for forest-based sequestration. For instance, California, Australia, New Zealand and Canada have compliance markets in place, but these have provided limited funding thus far for forest-based credits (see Ruseva et al. 2017 for a review on

California´s cap-and-trade program). One reason for this is that the compliance markets themselves have thus far had only limited overall carbon caps in place.

Significant investments have thus far been made in REDD+ funding. These funds have been allocated through a variety of international organizations. REDD+ investments through the United Nations REDD+ program to date are $318 million in 25 countries. The World Bank Forest Carbon Partnership Facility has allocated $1.3 billion to date for REDD readiness projects and results-based carbon initiatives in 47 countries (Simonet et al., 2018a). Across the many projects developed, Simonet et al. (2018a) document 64.6 million hectares of forests that have been included in REDD-type programs since 2008. Several countries have also developed bilateral agreements with Norway to engage in REDD+ (Guyana, Peru, Indonesia). Simonet et al. (2018b) provide evidence that at the project and household level in a project in Brazil, REDD+ payments have resulted in reduced deforestation. Roopsind et al (2019) have shown that the Guyana-Norway REDD+ agreement reduced deforestation by 35% during the period of implementation. The expectations that carbon-centered REDD+ would be a simple and efficient mechanism for climate mitigation has not been met (Turnhout et al. 2017), however, significant progress has occurred to date.

Most available low emission scenarios at least temporarily exceed the 1.5 °C limit before 2100. The legacy of temperature overshoots and the feasibility of limiting warming to 1.5 °C, or below, thus become central elements of a post-Paris science agenda. As near-term mitigation targets set by countries for the 2020–2030 period are insufficient to achieve the temperature goal, an increase in mitigation ambition and implementation for this period will determine the Paris Agreement's effectiveness (Schleussner et al. 2016).

One of the approaches to limit warming below 1.5 degrees C is to use an increasing amount of biomass energy derived from wood and agricultural products. A debate has emerged about whether biomass energy derived from wood is carbon neutral, and whether efforts to ramp up bioenergy production in forests would cause environmental damages in natural forests (Searchinger et al., 2009; Buchholtz et al., 2016; and DeCicco and Schlesinger, 2018). Khanna et al. (2017) review the literature and argue that the determination of carbon neutrality depends on the underlying assumptions made by modelers, and perhaps most importantly on how forest investments are modelled. If foresters are assumed to respond to rising prices by investing in forest resources, then emissions will be offset by growth due to investments.

In response to the potential that biomass energy from wood products is not carbon neutral, Searchinger et al. (2009) call for a tax on carbon emissions when forest inputs are used for biomass energy. Favero et al. (2020) illustrate that a carbon tax on emissions can lead to inefficient outcomes, and potentially greater carbon emissions, if policies do not address the maintenance or enhancement of forest carbon stocks as well. Furthermore, efforts to increase wood production for biomass energy without policies that protect natural forests could lead to increased harvests in natural forests, with a resulting increase in carbon emissions.

If biomass energy production expands, and shifts to carbon capture and storage (e.g., BECCS), studies suggest that there could be a significant increase the area of land used for biomass energy production (e.g. Favero and Mendelsohn, 2014). BECCS is not projected to be used widely for a number of years, but in the meantime, policies are expected to increase the storage of carbon in forests through a variety of approaches that reduce deforestation and increases afforestation and management. Favero et al. (2017) and Baker et al. (2019) illustrate that carbon sequestration policies and biomass energy policies are largely complementary. Thus, efforts to sequester carbon in forests and provide carbon benefits in the near-term may lower the costs of biomass energy production in the future, and that future biomass energy production can lower the costs of sequestration in forests. However, at high levels of biomass energy and carbon sequestration demand, they may become substitutes and thus compete with each other for the same land.

Voluntary actions and agreements

The New York Declaration of Forests (NYDF) aims at reducing emissions from deforestation and forest degradation (REDD) and has set quantified reduction targets (Roelfsema et al. 2018). With the declaration, 26 national governments, 23 large multinationals and more than 50 civil society and indigenous organizations endorse a global timeline to halve natural forest loss by 2020, and strive to end it by 2030 (New York Declaration of Forests, 2014). In addition, the declaration calls for restoring 150 million hectares of forests and croplands by 2020 and an additional 200 million hectares by 2030 (New York Declaration of Forests, 2014). The participants in the NYDF represented 20% of global CO_2 deforestation emissions in 2010. It was assumed that ending forest loss implies zero emissions from biomass burning. The baseline emissions for the New York Declaration of Forests from deforestation are already projected to decrease from 1.0 $GtCO_2eq$ in 2010 to 0.4 $GtCO_2eq$ in 2030 for the countries that participate in this initiative. On top of that, the emission reduction from the NYDF is estimated at 0.7 $GtCO_2$ in 2030, of which 0.5 $GtCO_2$ is the result of ending natural forest loss, and 0.2 $GtCO_2$ is the result of reforestation and restoration (Roelfsema et al. 2018).

Other voluntary actions have been implemented by companies and corporations in the agricultural sector through supply chain management. For instance, the soy moratorium and cattle agreement in Brazil both are credited with contributing to the reduction in deforestation that occurred in that country from 2004 to the present (Nepstad et al., 2013, 2014; Newton et al., 2013). In the Amazon, these agreements occurred when companies were pressured by environmental groups to agree not to purchase soy or cattle from land that had been deforested in the Amazon Basin. Despite efforts by NGOs and companies to improve supply chain management for goods that potentially can cause deforestation, if these efforts are not combined with other public policy actions, including regulations, supply chain efforts are much less successful (Lambin et al., 2018).

Monitoring and Verification

Development of satellite technologies to assess potential deforestation has grown in recent years with the release of 30 m data by Hansen et al. (2013), however, it is important to recognize that this data only captures tree cover loss. These losses could be due to many different factors, including natural disturbances like fires and traditional timber harvests in regions where forest management is significant. Furthermore, these datasets are less well developed for case of reforestation and afforestation. As Mitchell et al. (2017) point out there has been significant improvement in the ability to measure changes in tree and carbon density on sites using satellite data, but these techniques are still evolving and improving. They are not yet available for widespread use globally.

Ground-based forest inventory measurements have been developed for the US with the US Forest Service Inventory and Analysis database, which is freely available to anyone in the world online (see https://www.fia.fs.fed.us/). These data are collected on plots that are remeasured every 5-10 years. Canada similarly provided significant information online (https://nfi.nfis.org/en). Many European countries provide data from their forest inventories, but the online resources there are less well developed. Similarly, Russia and China have not provided forest inventory data online. Additional efforts to make forest inventory data available to the scientific community would improve confidence in forest statistics, and changes in forest statistics over time.

7.7.2. Constraints and opportunities across different contexts and regions

Sector specific barriers and opportunities

Most climate mitigation scenarios involve negative emissions, especially those that aim to limit global temperature increase to 2°C or less. However, the carbon uptake potential in land-based climate change mitigation efforts is highly uncertain depending on the assumptions related to land use and land management in the models including model assumptions regarding bioenergy crop yields and simulation of soil carbon response to land-use change. Differences between land-use models and DGVMs regarding forest biomass and the rate of forest regrowth also have an impact, albeit smaller, on the results(Krause et al. 2017).

The mitigation potential of land-based negative emissions technologies (NETs) is constrained by critical social objectives and ecological limits. Three types of risks were identified in relation to NETs: (1) that NETs will not ultimately prove feasible; (2) that their large-scale deployment involves unacceptable ecological and social impacts; and (3) that NETs prove less effective than hoped, due to irreversible climate impacts, or reversal of stored carbon (Dooley and Kartha 2018).

Scenarios that limit global warming to below 2 °C by 2100 assume significant land-use change to support large-scale carbon dioxide (CO_2) removal from the atmosphere by afforestation/reforestation, avoided deforestation, and Biomass Energy with Carbon Capture and Storage (BECCS). The more ambitious mitigation scenarios require even greater land area for mitigation and/or earlier adoption of CO_2 removal strategies. Additional land-use change to meet a 1.5 °C climate change target could result in net losses of carbon from the land (Harper et al. 2018). If BECCS involves replacing high-carbon content ecosystems with crops, then forest-based mitigation could be more efficient for atmospheric CO_2 removal than BECCS.

Table 7.11 Risks to successful deployment of land based NETs (Dooley and Kartha 2018)

Risks to deployment of land-based NETs		
Technical and biophysical infeasibility	Unacceptable social and ecological impacts	Ineffectiveness
Technological problems and constraints could prevent full-scale BECCS operation	Demand for land could compete with food production and security	Land-based carbon sinks may be prone to reversal, negating mitigation benefits
Sink saturation may limit sequestration potential of NETs NPP may limit expansion of crop production	Demand for land (exacerbated by lack of clear land rights) could cause dispossession of local communities and livelihoods	Temperature overshoot may lead to irreversible impacts, including on food production, and biodiversity, threatening adaptive capacity
Expected yield increases may not be achieved, particularly in the light of climate change	Demand for land could drive loss of natural ecosystems, weakening resilience and adaptive capacity, and result in conversion of forests, wilderness areas and biodiversity Replacing natural ecosystems with monocultures causes loss of biodiversity Increased inputs for energy crops and forests may cause land and water pollution	Temperature overshoot may cross tipping points preventing further temperature reduction

Table 7.12 Synergies and trade-offs between land based NETs and SDGs (Dooley and Kartha 2018)

NETs	SDGs		
	SDG 2: zero hunger	SDG 13: climate action	SDG 15: life on land
Avoided deforestation	+ Resilient ecosystems, sustainable food production for local communities	+ Avoided emissions + Increased resilience + Secure tenure, greater forest protection	+ Disproportionally large benefits to biodiversity + Sustainable livelihoods
Reforestation	+ Careful restoration can reduce environmental degradation, improve ecosystem resilience and food production − Quick-growing species can increase nutrient input	+ Significant mitigation potential from closed canopy reforestation − Reduced albedo effect at high latitudes	± Biodiversity impacts vary dependent on method of reforestation + Community-managed reforestation has greater livelihood, climate and biodiversity benefits
Forest ecosystem restoration	− Livelihoods threatened if subsistence ag. targeted + Resilient ecosystems, sustainable food production for local communities	+ Greater carbon density, increased mitigation potential + Increased resilience − Climate benefits reduced if (traditional) subsistence ag. targeted, or HWP[a] overly restricted	+ Correlation between carbon density and biodiversity − Local access threatened if subsistence ag. targeted − Constrained by existing land uses
BECCS	− Dedicated use of land for bioenergy competes with land for food production − Use of residues for bioenergy can reduce soil carbon, lowering productivity	± Carbon neutral bioenergy (hence negative emissions from BECCS), only if sequestered carbon exceeds net emissions (including initial and indirect land-use change emissions)	- HANPP[b] for bioenergy can compete with food, biodiversity and subsistence livelihoods − Large-scale bioenergy increases input demand, resulting in environmental degradation and water stress

[a]Harvested wood products

[b]Human appropriation of net primary production

Uncertainties in future socio-economic land use drivers indicate a wide range of potential future land-use dynamics and consequences for land-based ecosystem services. Greenhouse gas emissions from land use and land use change associated to alternatives land-use and agricultural systems, differed strongly across socioeconomic narratives, and cumulative land use change emissions (2005- 2100) ranged from -54 to 402 Gt CO_2 (Popp et al. 2017).

While there are some promising emerging options available that can reduce GHG emissions from the red meat sector, a carbon neutral status will involve investments and commitments by the industry to manage vegetation for the mutual benefit of production, reducing emissions and ecosystems services (Mayberry et al. 2019). There are large countries with export-oriented livestock industries where grazing livestock production is closely associated with deforestation. In many of these places, incentive mechanisms for reducing emissions could contribute to the success of mitigation strategies in agriculture.

Ensuring good governance and accountability is crucial for the implementation of forest-based mitigation options. Implementation of the Paris Agreement would require large-scale estimation, modelling, monitoring, reporting and verification of GHG inventories, mitigation actions and their implications and co-benefits, along with reporting on climate change impacts and adaptation. Most developing countries with large forest cover and intense land use changes have insufficient capacity to address research needs, modelling, monitoring, reporting and data requirements (e.g. Ravindranath et al. 2017 for India) compromising transparency, accuracy, completeness, consistency and comparability. In spite of the many synergies between climate policy instruments and biodiversity conservation, current policies often fall short of realizing this potential (Essl et al. 2018).

Opportunity for political participation of local stakeholders is also a critical factor because in many nations with the highest deforestation rates, forest ownership rights often are not sufficiently documented and secured (Essl et al. 2018). Incentives for self-enforcement can be relevant in the future considering that weak governance insecure property rights are significant barriers to introduction of forest carbon offset projects in developing countries, where many of the low-cost options for such projects exist (Gren and Aklilu, 2016). Despite their comparatively small volume, voluntary offsets have an outsized impact on compliance markets and on emission reductions activities in general with the value of the forestry and land use offsets market more than triple that of the renewables market, corresponding to 46% of the total value of the voluntary carbon markets in 2016 (Hamrick and Gallant, 2017).

The response of the private sector to climate change will be key for setting and achieving the commitments made by Parties to the UNFCCC (Gnych et al. 2016). For example, the number of private commitments to reduce deforestation from supply chains has greatly increased in recent years but there is the concern that corporate commitments will exclude already marginalized groups as smallholders, who often operate within broader informal economies, resulting in negative social and environmental impacts (Gnych et al., 2016). Therefore, the effectiveness of supply-chain initiatives by private actors, as stated by Lambin et al. (2018), also depends on public policies to enable the adoption of sustainable practices, creation and maintenance of key infrastructure, and measures and safeguards to address adverse effects on small producers.

One relevant uncertainty in projections of forest mitigation potential concerns the impacts of climate change on the world's forests that can result in future potential changes in terrestrial ecosystem productivity, climate-driven vegetation migration, wildfires, forest regrowth and carbon dynamics. A set of simulations to assess the impact of climate change on global forests resulted in an increase of the carbon stocks of most forests around the world, with the greatest gains in tropical forest regions (Kim et al. 2017). Temperate forest regions were projected to see strong increases in productivity offset by carbon loss to fire in the boreal zone. The drivers of forest changes varied regionally, associated with differing mechanisms as expansion or contraction of forests, with further loss of area to wildfire; and changes in vegetation productivity. These results contrast with previous studies that pointed to the likelihood of reduced forest carbon stocks due to climate feedback, even when constrained by CO_2 fertilization (Cox et al. 2013, Friedlingstein, 2015).

Afforestation may have minor to severe consequences for surface water acidification, depending on site-specific factors and the exposition to air pollution and sea-salts (Futter et al. 2019). It may also reduce runoff due to increased root uptake and higher evapotranspiration. Afforestation will increase average deposition rates slightly due more effective atmospheric scavenging of dry deposition. The potential effects of coastal afforestation on sea-salt related acidification events must be evaluated in each case as this could lead to re-acidification and damage on aquatic biota.

Forest ecosystems are critical to mitigating greenhouse gas emissions, but climate change is affecting forest ecosystem functioning. The increasing terrestrial sink might to be linked to changes in the global environment (e.g., increased atmospheric CO_2 concentrations, N deposition, or changes in climate) (Ballantyne et al., 2012). However, it is uncertain if the terrestrial carbon sink will continue in the future (see, for example, Aragão et al. 2018 on impacts of severe droughts impacts on the Amazon, and Lovejoy and Nobre, 2018 on the negative synergies between deforestation, climate change, and widespread use of fire leading the Amazon system to a tipping point). Houghton and Nassikas (2017) pointed out that possibly the growing sink has been enabled by large areas of secondary forests, and as a consequence, the potential for additional carbon accumulation would be reduced if the world's forests recover to their undisturbed state. On the other hand, secondary forest carbon saturation might take many decades (Zhu et al 2018). The unknown spatio-temporal response of ecosystem functions to global change adds further uncertainty about the future mitigation potential of forests.

Biodiversity may improve resilience to climate change impacts on a) biodiversity itself, as more-diverse systems could be more resilient to climate change impacts, and b) ecosystem functioning through the positive relationship between diversity and ecosystem functioning (Hisano et al. 2018).

The shift in species/functional diversity and losses in plant species diversity may impair the positive effects of diversity on ecosystem functioning. Forest management strategies based on biodiversity and ecosystems functioning interactions have strong potential for augmenting the effectiveness of the roles of forests in reducing climate change impacts on ecosystem functioning.

Assessment of social and policy responses (public and private)

Sectoral policies

Climate policy goals need to be integrated into the wider context of land-use relevant policies. Land provides vital resources to society, such as food, fuel, fiber and many other ecosystem services that support production functions, regulate risks of natural hazards, or provide cultural and spiritual services (Foley et al. 2005).

Economic incentives

The role of forests for climate change mitigation has strongly been recognized in the Paris Climate Agreement. The Agreement endorses Reduced Emissions from Deforestation and Forest Degradation (REDD+), sustainable forest management, allows for alternative (non-market) policy approaches such as joint mitigation and adaptation, and emphasizes the importance of non-carbon benefits and equity for sustainable development (Martius et al. 2016). However, a review of the diverse and evolving approaches to REDD+ pointed out that the level of financing is insufficient (Streck, 2012), while current compliance markets for forest carbon credits are limited (see Ruseva et al. 2017 for a review on California´s cap-and-trade program). The expectations that carbon-centered REDD+ would be a simple and efficient mechanism for climate mitigation are not currently being met (Turnhout et al. 2017). Further, only 2.5% of climate mitigation funding goes to land-based mitigation options, an order of magnitude below the potential proportional contribution (Buchner et al. 2015).

At the same time, most available low emission scenarios at least temporarily exceed the 1.5 °C limit before 2100. The legacy of temperature overshoots and the feasibility of limiting warming to 1.5 °C, or below, thus become central elements of a post-Paris science agenda. As near-term mitigation targets set by countries for the 2020–2030 period are insufficient to achieve the temperature goal, an increase in mitigation ambition and implementation for this period will determine the Paris Agreement's effectiveness (Schleussner et al. 2016).

Socio-economic barriers and opportunities

Uncertainties in future socio-economic land use drivers indicate a wide range of potential future land-use dynamics and consequences for land-based ecosystem services. Possible land-use changes and their consequences included in five alternative Integrated Assessment Models were evaluated for the translation of the Shared Socio-Economic Pathways (SSPs) narratives into quantitative projections (Popp et al. 2017). Greenhouse gas emissions from land use and land use change associated to alternatives land-use and agricultural systems, differed strongly across socioeconomic narratives, and cumulative land use change emissions (2005- 2100) ranged from -54 to 402 Gt CO_2. Popp et al. (2017) concluded, under the "sustainability scenario (SSPI), that more efficient food production systems and globalized trade have the potential to enhance the extent of natural ecosystems, lead to lowest greenhouse gas emissions from the land system and decrease food prices over time.

While there are some promising emerging options available that can reduce GHG emissions from the red meat sector, that attaining a carbon neutral status will not be possible unless significant investment is made and signals sent to industry to manage vegetation for the mutual benefit of production, reducing emissions and ecosystems services (Mayberry et al. 2019). There are large countries with export-orientated livestock industries where grazing livestock production is closely associated with deforestation. In many of these places, livestock farmers are also the custodians of large swaths of forests or reforestable areas and incentive mechanisms for reducing emissions could contribute to the success of mitigation strategies in agriculture.

Institutional barriers and opportunities

Ensuring good governance and accountability is crucial for the implementation of forest-based mitigation options. Implementation of the Paris Agreement would require large-scale estimation, modelling, monitoring, reporting and verification of GHG inventories, mitigation actions and their implications and co-benefits, along with reporting on climate change impacts and adaptation. Most developing countries with large forest cover and intense land use changes have insufficient capacity to address research needs, modelling, monitoring, reporting and data requirements (e.g. Ravindranath et al. 2017 for India) compromising transparency, accuracy, completeness, consistency and comparability. In spite of the many synergies between climate policy instruments and biodiversity conservation, current policies often fall short of realizing this potential (Essl et al. 2018).

Opportunity for political participation of local stakeholders is also a critical factor because in many nations with the highest deforestation rates, forest ownership rights often are not sufficiently documented and secured (Essl et al. 2018). Incentives for self-enforcement can be relevant in the future considering that weak governance insecure property rights are significant barriers to introduction of forest carbon offset projects in developing countries, where many of the low-cost options for such projects exist (Gren and Aklilu, 2016). Despite their comparatively small volume, voluntary offsets have an outsized impact on compliance markets and on emission reductions activities in general with the value of the forestry and land use offsets market more than triple that of the renewables market, corresponding to 46% of the total value of the voluntary carbon markets in 2016 (Hamrick and Gallant, 2017).

The response of the private sector to climate change will be key for setting and achieving the commitments made by Parties to the UNFCCC (Gnych et al. 2016). For example, the number of private commitments to reduce deforestation from supply chains has greatly increased in recent years, with at least 760 public commitments by 447 producers, processors, traders, manufacturers and retailers as of March 2017 (Donofrio et al. 2017 cited by Lambin et al. 2018). One concern is that corporate commitments related to sustainable and 'deforestation free' supply chains will exclude already marginalized groups as smallholders, who often operate within broader informal economies, resulting in negative social and environmental impacts (Gnych et al., 2016). Therefore, the effectiveness of supply-chain initiatives by private actors, as stated by Lambin et al. (2018), also depends on public policies to enable the adoption of sustainable practices, creation and maintenance of key infrastructure, and measures and safeguards to address adverse effects on small producers.

Ecological barriers and opportunities

One relevant uncertainty in projections of forest mitigation potential concerns the impacts of climate change on the world's forests that can result in future potential changes in terrestrial ecosystem productivity, climate-driven vegetation migration, wildfires, forest regrowth and carbon dynamics. A set of simulations to assess the impact of climate change on global forests was analysed (Kim et al. 2017), where a dynamic global vegetation model was run with climate simulations from the MIT Integrated Global System Model-Community Atmosphere Model (IGSM-CAM). Climate simulations performed under two emissions scenarios (a business-as-usual reference scenario, analogous to the RCP8.5 scenario, and a greenhouse gas mitigation scenario, between RCP2.6 and RCP4.5 scenarios) resulted in an increase of the carbon stocks of most forests around the world, with the greatest gains in tropical forest regions. Temperate forest regions were projected to see strong increases in productivity offset by carbon loss to fire in the boreal zone. The drivers of forest changes varied regionally, associated with differing mechanisms as expansion or contraction of forests, with further loss of area to wildfire; and changes in vegetation productivity. These results contrast with previous studies that pointed to the likelihood of reduced forest carbon stocks due to climate feedback, even when constrained by CO_2 fertilization (Cox et al. 2013, Friedlingstein, 2015).

In a recent study, Houghton and Nassikas (2018) indicated that without a massive, global programme of forest expansion, the potential for negative emissions from ending deforestation and degradation is on the order of 100 PgC. The authors highlighted that negative emissions are possible because many forest ecosystems are recovering from past disturbances, and much of that recovery will have occurred before 2100. In spite of being considered modest, this potential for negative emissions was significant compared to the total carbon emissions allowable for staying within a warming of 2°C (Rogelj et al., 2016) indicating that land management is vital for the 2°C goal.

Afforestation may have minor to severe consequences for surface water acidification, depending on site-specific factors and the exposition to air pollution and sea-salts (Futter et al. 2019). It may also reduce runoff due to increased root uptake and higher evapotranspiration. Afforestation will increase average deposition rates slightly due more effective atmospheric scavenging of dry deposition. The potential effects of coastal afforestation on sea-salt related acidification events must be evaluated in each case as this could lead to re-acidification and damage on aquatic biota.

Forest ecosystems are critical to mitigating greenhouse gas emissions, but climate change is affecting forest ecosystem functioning. The increasing terrestrial sink might to be linked to changes in the global environment (e.g., increased atmospheric CO_2 concentrations, N deposition, or changes in climate) (Ballantyne et al., 2012). However, it is uncertain if the terrestrial carbon sink will continue in the future (see, for example, Aragão et al. 2018 on impacts of severe droughts impacts on the Amazon, and Lovejoy and Nobre, 2018 on the negative synergies between deforestation, climate change, and widespread use of fire leading the Amazon system to a tipping point). Houghton and Nassikas (2017) pointed out that possibly the growing sink has been enabled by large areas of secondary forests, and as a consequence, the potential for additional carbon accumulation would be reduced if the world's forests recover to their undisturbed state. On the other hand, secondary forest carbon saturation might take many decades (Zhu et al 2018). The unknown spatio-temporal response of ecosystem functions to global change adds further uncertainty about the future mitigation potential of forests.

Biodiversity may improve resilience to climate change impacts on a) biodiversity itself, as more-diverse systems could be more resilient to climate change impacts, and b) ecosystem functioning through the positive relationship between diversity and ecosystem functioning (Hisano et al. 2018). The shift in species/functional diversity and losses in plant species diversity may impair the positive effects of diversity on ecosystem functioning. Forest management strategies based on biodiversity and ecosystems functioning interactions have strong potential for augmenting the effectiveness of the roles of forests in reducing climate change impacts on ecosystem functioning.

Technological barriers and opportunities

Most climate mitigation scenarios involve negative emissions, especially those that aim to limit global temperature increase to 2°C or less. However, the carbon uptake potential in land-based climate change mitigation efforts is highly uncertain depending on the assumptions related to land use and land management in the models including model assumptions regarding bioenergy crop yields and simulation of soil carbon response to land-use change. Differences between land-use models and DGVMs regarding forest biomass and the rate of forest regrowth also have an impact, albeit smaller, on the results (Krause et al. 2017).

7.7.3. Linkages to ecosystem services, human well-being and adaptation (incl. SDGs)

The Millennium Ecosystem Assessment acknowledges the close linkages between biodiversity, ecosystem services, human well-being and sustainable development (MEA, 2005). Loss of biodiversity and ecosystem services will have an adverse impact on good quality of life, human well-being and sustainable development. It will not only affect current economic growth but also impede the capacity of the economy to sustain future economic growth.

Anthropogenic factors have been a major driving force behind land use and land cover changes and their role has been significant in the recent decades. Seventy-five per cent of the land surface is significantly altered, 66%of the ocean area is experiencing increasing cumulative impacts, and over 85%of wetlands (area) has been lost (IPBES 2019). Land-use change is driven primarily by agriculture, forestry and urbanization, all of which are associated with air, water and soil pollution (IPBES 2019). Over one third of the world's land surface and nearly three-quarters of available freshwater resources are devoted to crop or livestock production (IPBES 2019). Agricultural expansion is the most widespread form of land-use change, with over one third of the terrestrial land surface being used for cropping or animal husbandry. This expansion, alongside a doubling of urban area since 1992 and an unprecedented expansion of infrastructure linked to growing population and consumption, has come mostly at the expense of forests (largely old-growth tropical forests), wetlands and grasslands. In freshwater ecosystems, a series of combined threats that include land-use change, including water extraction, exploitation, pollution, climate change and invasive species, are prevalent.

Between 1990 and 2015 clearing and wood harvest contributed to a total reduction of 290 Mha in native forest cover, while the area of planted forests grew by 110 Mha (IPBES 2019). While the rate of forest loss has slowed globally since 2000, this is distributed unequally. Across much of the highly biodiverse tropics, 32 Mha of primary or recovering forest were lost between 2010 and 2015 (IPBES 2019). The extent of tropical and subtropical forests is increasing within some countries, and the global extent of temperate and boreal forests is increasing. A range of actions – from restoration of natural forest to planting of monocultures – contribute to these increases but have very different consequences for biodiversity and its contributions to people. On land, particularly sensitive ecosystems include old-growth forests, insular ecosystems, and wetlands; and only around 25% of land is sufficiently unimpacted that ecological and evolutionary processes still operate with minimal human intervention (*established but incomplete*) (IPBES 2019). In terrestrial "hotspots" of endemic species, natural habitats have generally undergone greater reductions to date in extent and condition, and tend to be experiencing more rapid ongoing decline, on average than other terrestrial regions. Globally, the net rate of forest loss has halved since the 1990s, largely because of net increases in temperate and high latitude forests; high-biodiversity tropical forests continue to dwindle, and global forest area is now approximately 68 per cent of the estimated pre-industrial level *(established but incomplete)* (IPBES 2019). Forests and natural mosaics sufficiently undamaged to be classed as "intact" (defined as being larger than 500 km^2 where satellites can detect no human pressure) were reduced by 7%(919, 000 km^2) between 2000 and 2013, shrinking in both developed and developing countries (IPBES 2019). Across regions and sub-regions, one finds diverse trends. For instance, in the Asia Pacific region from 1990 to 2015 South East Asia showed a reduction in forest cover by 12.9%largely due to an increase in timber extraction, largescale bio-fuel plantations and expansion of intensive agriculture and shrimp farms (IPBES 2018c). However, over the same period North East Asia and South Asia have shown an increase in forest cover of 22.9%and 5.8%respectively, through policies and instruments such as joint forest management, payment for ecosystem services, and the restoration of degraded forests (IPBES 2018c). All mining on land has increased dramatically and, while still using less than 1%of the Earth's land, has had significant negative impacts on biodiversity, emissions of highly toxic pollutants, water quality and water distribution, and human health (IPBES 2019).

Ecosystem Services

Trends since 1970 indicate that four out of eighteen ecosystem services assessed, namely agricultural production, fish harvest, bioenergy production and harvest of materials has increased, whereas the remaining ecosystem services-mostly regulating and non-material contributions- have declined (see Figure 7.29) (IPBES 2019). The value of agricultural crop production ($2.6 trillion in 2016) has increased approximately threefold since 1970, and raw timber harvest has increased by 45%, reaching some 4 billion cubic metres in 2017, with the forestry industry providing about 13.2 million jobs (IPBES 2019). However, indicators of regulating contributions, such as soil organic carbon and pollinator diversity, have declined, indicating that gains in material contributions are often not sustainable. Currently, land degradation has reduced productivity in 23%of the global terrestrial area, and between $235 billion and $577 billion in annual global crop output is at risk as a result of pollinator loss (IPBES 2019). Land degradation has had a pronounced impact on ecosystem functions worldwide *(well established)* (IPBES 2018e). Net primary productivity of ecosystem biomass and of agriculture is presently lower than it would have been under natural state on 23%of the global terrestrial area, amounting to a 5%reduction in total global net primary productivity *(established but incomplete)* (IPBES 2018e). Over the past two centuries, soil organic carbon, an indicator of soil health, has seen an estimated 8%loss globally (176 GtC) from land conversion and unsustainable land management practices *(established but incomplete)* (IPBES 2018e). Projections to 2050 predict further losses of 36 Gt C from soils, particularly in sub-Saharan Africa. These future losses are projected to come from the expansion of agricultural land into natural areas (16 Gt C), degradation due to inappropriate land management (11 Gt C) and the draining and burning of peatlands (9 Gt C) and melting of permafrost *(established but incomplete)* (IPBES 2018e). According to Costanza et al (2014), between 1997 to 2011 the loss of global ecosystem services due to land use change is valued at between US$ 4.2-20.2 trillion per year (in 2007 US $) depending on which unit value one adopts.

Loss of coastal habitats and coral reefs reduces coastal protection, which increases the risk from floods and hurricanes to life and property for the 100 million–300 million people living within coastal 100-year flood zones (IPBES 2019). Climate change is a direct driver that is increasingly exacerbating the impact of other drivers on nature and human well-being. Land use change is a major driver behind loss of biodiversity and ecosystem services in Africa, America, Asia-Pacific, Europe and Central Asia regions (IPBES 2018a; IPBES 2018b; IPBES 2018c; IPBES 2018d). Unsustainable extension and intensification of agriculture and forestry in many regions of the world is putting immense stress on biodiversity and ecosystem services resulting in their degradation. Projected impacts of land use change and climate change on biodiversity and ecosystem services (material and regulating contributions to people) between 2015 to 2050 are seen to have relatively less negative impacts under global sustainability scenario as compared to regional competition and economic optimism scenarios (Figure 7.30) (IPBES 2019). These projected impacts are based on a subset of Shared Socioeconomic Pathway (SSP) scenarios and greenhouse gas emissions trajectories (RCP) developed in support of IPCC assessments.

Human Well-being and Sustainable Development Goals

Conservation of biodiversity and ecosystem services is closely linked and critical to ensuring good quality of life, human well-being and realizing the sustainable development goals. Several examples illustrate the interdependencies between nature (biodiversity and ecosystem services) and good quality of life. For example, nature and its contributions may play an important role in reducing vulnerability to climate-related extreme events and other economic, social and environmental shocks and disasters, although anthropogenic assets are also involved (*established but incomplete*) (IPBES 2019). Many of nature's contributions to people are essential for human health (*well established*) and their decline thus threatens a good quality of life (*established but incomplete*) (IPBES 2019). Nature provides a broad diversity of nutritious foods, medicines and clean water *(well established)* (Sustainable Development Goal 3), can help to regulate and reduce levels of certain air pollutants (*established but incomplete*) and improve mental and physical health through exposure to natural areas *(inconclusive),* among other contributions (Sustainable Development Goal 3) (IPBES 2019). Nature's underpinning of specific health targets varies across regions and ecosystems, is influenced by anthropogenic assets and remains understudied. The relationship can be positive or negative, as in the case of certain aspects of biodiversity and infectious diseases.

Nature directly underpins the livelihoods of indigenous peoples and local communities and the rural and urban poor, largely through direct consumption of, or income generated by, trade in material contributions such as food and energy (*well established*) (IPBES 2019). Such contributions are generally underrepresented in poverty analyses (*established but incomplete*) (IPBES 2019). Nature and its contributions are also relevant to goals for education, gender equality, inequalities and peace, justice and strong institutions (Sustainable Development Goals 4, 5, 10 and 16), but the current focus and wording of targets obscures or omits their relationship to nature (*established but incomplete*) (IPBES 2019).

Current negative trends in biodiversity and ecosystem services will undermine progress towards 80%(35 out of 44) of the assessed targets of goals related to poverty, hunger, health, water, cities, climate, oceans and land (Sustainable Development Goals 1, 2, 3, 6, 11, 13, 14, and 15) (IPBES 2019). The Sustainable Development Goals for poverty, health, water and food security and sustainability targets are closely linked through the impacts of multiple direct drivers, including climate change, on biodiversity and ecosystem functions and services, nature and nature's contributions to people and good quality of life (IPBES 2019). In a post-2020 global biodiversity framework, greater emphasis on the interactions between Sustainable Development Goal targets may provide a way forward for achieving multiple targets, as synergies (and trade-offs) can be considered. Future targets are expected to be more effective if they consider impacts of climate change, including on biodiversity, and action to mitigate and adapt to climate change (IPBES 2019). Important positive synergies between nature and goals on education, gender equality, reducing inequalities and promoting peace and justice (Sustainable Development Goals 4, 5, 10 and 16) were found (IPBES 2019). Land or resource tenure insecurity, as well as declines in nature, have greater impacts on women and girls, who are most often negatively impacted. There is a critical need for future policy targets, indicators and datasets to more explicitly account for aspects of nature and their relevance to human well-being in order to more effectively track the consequences of trends in nature on Sustainable Development Goals (IPBES 2019). Some pathways chosen to achieve the goals related to energy, economic growth, industry and infrastructure and sustainable consumption and production (Sustainable Development Goals 7, 8, 9 and 12), as well as targets related to poverty, food security and cities (Sustainable Development Goals 1, 2 and 11), could have substantial positive or negative impacts on nature and therefore on the achievement of other Sustainable Development Goals (IPBES 2019).

Land-based Mitigation and Adaptation

Land-based mitigation and adaptation to the risks posed by climate change and extreme weather events can have several co-benefits as well as help promote development and conservation goals. The conservation of biodiversity and ecosystems enhances adaptive capacity, strengthens resilience and reduces vulnerability to climate change, thus contributing to sustainable development (IPBES 2018a). Land-based mitigation and adaptation will not only help in reducing greenhouse gas emissions in the AFOLU sector but also help augment its role as a carbon sink by increasing the forest and tree cover through afforestation and agro-forestry activities and other nature-based solutions. Land acts as a natural carbon sink with carbon stored in the soil and above ground biomass (forests and plants) (Keramidas et al., 2018). In the central 2°C scenario, improved management of land and more efficient forest practices, in the form of a drastic reduction of deforestation and an increased effort in afforestation, would account for 10% of the total mitigation effort over 2015–2050 (Keramidas et al., 2018). If managed and regulated appropriately, the Land Use, Land Use Change and Forestry (LULUCF) sector could become carbon-neutral as early as 2020–2030, being a key sector for emissions reductions beyond 2025 (Keramidas et al., 2018). These developments would occur with the simultaneous expansion of the use of biomass as an energy source, and thus an increase in the surfaces of managed forests for its provision. Nature-based solutions with safeguards are estimated to provide 37%of climate change mitigation until 2030 needed to meet 2°C goals with likely co-benefits for biodiversity (IPBES 2019). Therefore, land-use actions are indispensable, in addition to strong actions to reduce greenhouse gas emissions from fossil fuel use and other industrial and agricultural activities. However, the large-scale deployment of intensive bioenergy plantations, including monocultures, replacing natural forests and subsistence farmlands, will likely have negative impacts on biodiversity and can threaten food and water security as well as local livelihoods, including by intensifying social conflicts (IPBES 2019). Land-based mitigation and adaptation can also help improve incomes and employment and benefit the poor and vulnerable sections. The report of the Global Commission on Adaptation (2019) notes that investing US$ 1.8 trillion between 2020 to 2030 in five areas namely, early warning systems, climate-resilient infrastructure, dryland agriculture crop production, global mangrove conservation and investing in making water resources more resilient can generate net benefits of US$ 7.1 trillion, i.e. a benefit-cost ratio of over 3.9 (GCA, 2019). The report further states that without adaptation, climate change may depress global agricultural yields by up to

30 per cent by 2050 and the 500 million small farmers around the world will be most affected. The report also notes that climate change may push more than 100 million people in developing countries to below the poverty line by 2030. Among adaptation measures access to crop insurance can be effective in insuring the poor and vulnerable farmers from the risks posed by climate change and extreme weather events (Panda et al, 2013). A recent study notes that in the absence of adaptation efforts climate change will not only have an adverse impact on agricultural yields in India but also aggravate the extent, depth and intensity of rural poverty in India as measured through the headcount ratio, poverty gap index and squared poverty gap index (Ninan 2019).

Avoiding, reducing and reversing land degradation can contribute substantially to the mitigation of climate change, but land-based climate mitigation strategies must be implemented with care if unintended negative impacts on biodiversity and ecosystem services are to be avoided (*well established*) (IPBES 2018e). Between 2000 and 2009, land degradation was responsible for annual global emissions of 3.6–4.4 billion tonnes of CO_2 (*established but incomplete*) (IPBES 2018e). The main processes include the loss and degradation of forests, the drying and burning of peatlands, and the decline of carbon content in cultivated soils and rangelands as a result of excessive disturbance and insufficient return of organic matter to the soil (IPBES 2018e). Land degradation will also weaken the potential of land as a carbon sink (IPBES 2018e).

Figure 7.29 Global trends in the capacity of nature to sustain contributions to good quality of life from 1970 to the present which show a decline for 14 of the 18 categories of nature's contributions to people analysed. Data supporting global trends and regional variations come from a systematic review of over 2000 studies. Indicators were selected based on availability of global data, prior use in assessments and alignments with 18 categories. For many categories of nature's contributions, two indicators are included that show different aspects of nature's capacity to contribute to human well-being within that category. Indicators are defined so that an increase in the indicator is associated with an improvement in nature's contributions.

Source: SPM Figure 1 (IPBES 2019)

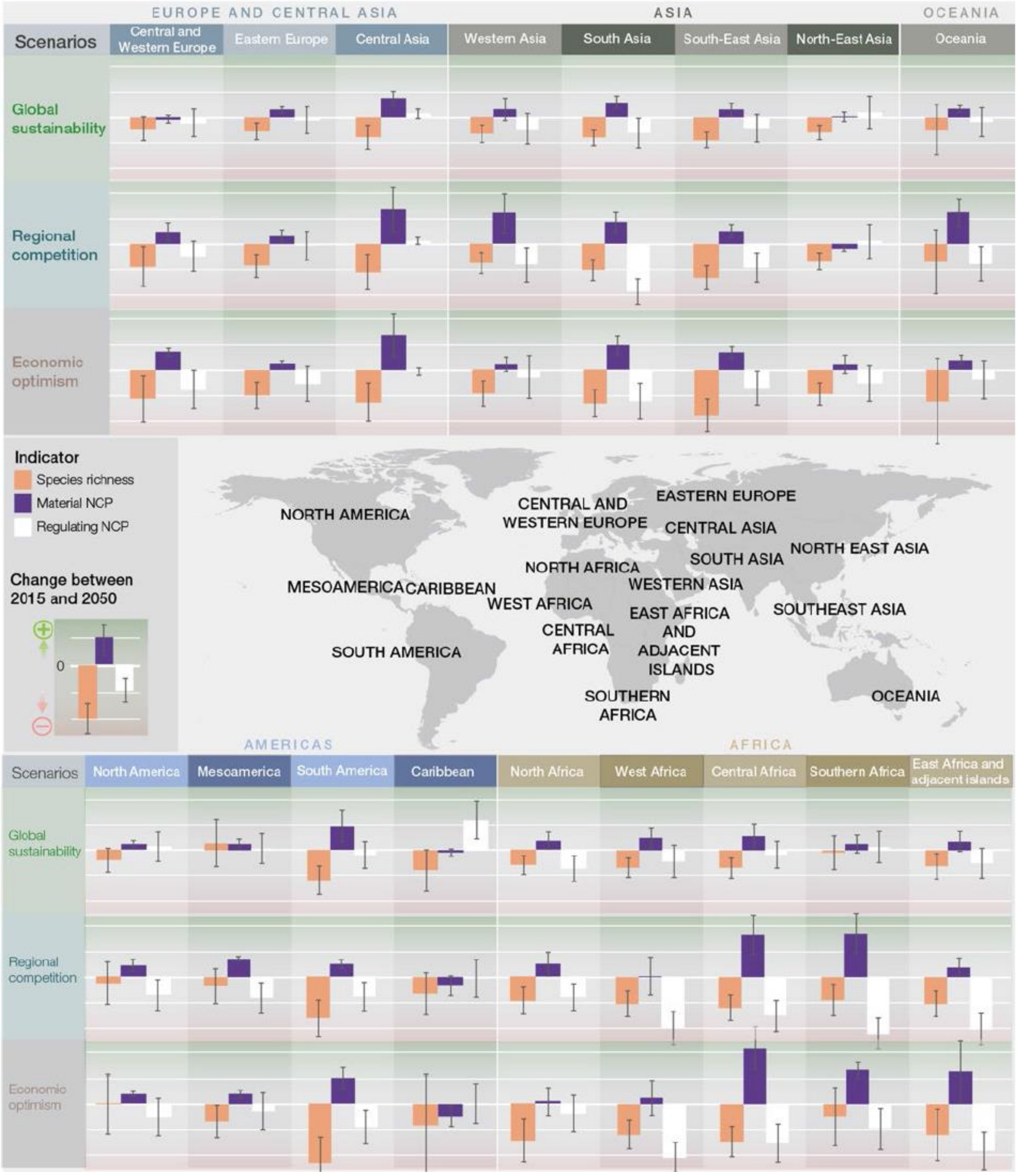


Figure 7.30 Projections of impacts of land use and climate change on biodiversity and nature's material and regulating contributions to people between 2015 and 2050. This figure illustrates three main messages: i) impacts on biodiversity and regulating nature's contributions to people (NCP) are the lowest in Global Sustainability scenario in nearly all sub-regions, ii) regional differences in impacts are high in the regional competition and economic optimism scenarios, and iii) material NCP increase the most in the regional competition and economic optimism scenarios, but this comes at the expense of biodiversity and regulating NCP. Projected impacts are based on a subset of Shared Socioeconomic Pathway (SSP) scenarios and greenhouse gas emissions trajectories (RCP) developed in support of IPCC assessments. This is does not cover scenarios that include transformative change.

Note: (1) The 'Global Sustainability' scenario combines proactive environmental policy and sustainable production and consumption with low greenhouse gas emissions ((SSP1, RCP2.6: top rows in each panel. (2) The 'Regional Competition' scenario combines strong trade and other barriers and a growing gap between rich and poor with high emissions (SSP3, RCP6.0: middle rows). (3) The 'Economic Optimism' scenario combines rapid economic growth with low environmental regulation with very high greenhouse gas emissions (SSP%, RCP8.5; bottom rows). (4) Multiple models were used with each of the scenarios to generate the first rigorous global-scale model comparison estimating the impact on biodiversity (changes in species richness across a wide array of terrestrial plant and animal species at regional scales; orange bars), material NCP (food, feed, timber and bioenergy; purple bars), and regulating NCP (nitrogen retention, soil protection, crop pollination, crop pest control and ecosystem carbon; while bars). The bars are the normalised means of multiple models and whiskers indicate the standard errors.

Source: SPM Figure 8 (IPBES 2019)

7.7.4. Emerging solutions using new technologies

[Placeholder-For SOD]

7.8. Comparing AFOLU estimates from global models and countries: implications for assessing collective climate progress

The Paris Agreement includes a periodic global stocktake (every five years starting in 2023) to assess the countries' collective progress towards the long-term goals of the Paris Agreement "in the light of ... the best available science". Any identified emission gap between collective progress and the 'well-below 2 °C trajectory' is expected to motivate increased mitigation ambition by countries in successive rounds of Nationally Determined Contributions (NDCs).

The anthropogenic AFOLU CO_2 emissions are included in the "Land Use Change" flux by global models and in the Land Use, Land-Use Change and Forestry ("LULUCF") sector by the country GHG inventories reported to the UNFCCC. As illustrated in section XXX, the AFOLU CO_2 emissions reported under the Land Use Change flux by global models (bookkeeping models and DGVMs) are 5.1 ± 2.6 $GtCO_2$ yr^{-1} globally during 2005 to 2014, much higher than the 0.1 ± 1.0 $GtCO_2$ yr^{-1} reported for LULUCF by country GHG inventories over the same time period.

This large difference, that holds also for the future AFOLU mitigation pathways modelled by IAMs, indicates a possible lack of comparability that could jeopardize the assessment of the collective progress under the global stocktake.

Reconciling the differences in AFOLU CO_2 emissions between global models and country GHG inventories is therefore important to support consistency, transparency and accuracy of the global stocktake's assessment of collective progress towards the long-term goals of the Paris Agreement.

7.8.1 Causes of the difference in AFOLU CO_2 flux between global models and countries

Based on the available scientific literature, the causes of the ≈ 5 $GtCO_2yr^{-1}$ gap between global models and countries' GHG inventories can be identified in the following:

(i) Coarse and incomplete representation of land-use change and management in global models (Pongratz et al. 2018).
(ii) Inaccurate and incomplete estimation of LULUCF fluxes in country GHG inventories, especially in developing countries (Grassi et al. 2017, other refs).
(iii) Conceptual differences in estimating "anthropogenic" land CO_2 flux, especially for the forest CO_2 sink, which make global models and countries hardly comparable from a conceptual point of view (Grassi et al. 2018).

The impacts of (i) and (ii) are difficult to quantify, and are expected to be minimized only in the in the medium-long term, linked to the expected improvements by both models and countries' GHG inventories. By contrast, the impact of (iii) is estimated to be at least 3 $GtCO_2yr^{-1}$, and short-term solutions to minimize it have been proposed (Grassi et al. 2018).

Due to differences in purpose and scope, the largely independent scientific communities supporting the global land flux modelling (bookkeeping models, IAMs and DGVMs) and the country GHG inventory compilation have developed different approaches to identify anthropogenic CO_2 fluxes for the land sector (Grassi et al. 2018, IPCC SRCCL). Both approaches are valid in their own specific contexts, yet both incomplete.

As summarized in Figure 7.31, the different approaches relate to the attribution of the impact of human-induced environmental changes (indirect human-induced effects) and to the forest area considered managed.

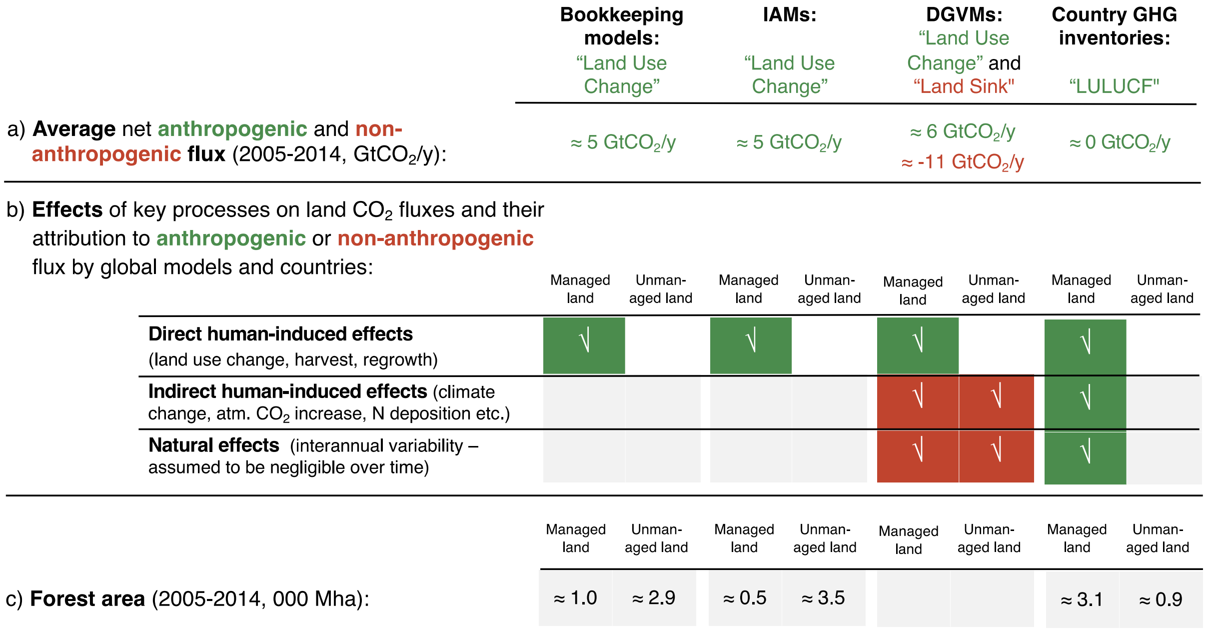


Figure 7.31 Summary of the main quantitative and conceptual differences between global models (bookkeeping models, IAMs and DGVMs) and country GHG inventories in considering what is the "anthropogenic land CO_2 flux" (i.e. "Land Use Change" by global models and "LULUCF" by countries) and "non anthropogenic land CO_2 flux" (i.e. "Land Sink" by DGVMs, considered to be the natural response of land to human-induced environmental change). Adapted from Grassi et al. 2018 and IPCC SRCCL, including: a) Averages land net CO_2 flux for 2005-2014 from Bookkeeping models and DGVMs (Le Quere et al. 2018), IAMs (SSP database) and country GHG inventories (Grassi et al. 2018); b) Effects of key processes on the land flux as defined by IPCC (2010), where these effects are captured (in managed and/or unmanaged lands), and their attribution to anthropogenic or non-anthropogenic flux by global models and countries; (c) Forest area considered "managed" and "unmanaged" by bookkeeping models, IAMs and country GHG inventories. Note that the figure may in some case be an oversimplification, e.g. a minority of IAMs do include indirect effects in the Land Use Change component, the way Land Use Change is estimated by DGVMs imply that conceptually some limited indirect effect is taken into account, and not all country GHG inventories include all recent indirect effects.

Bookkeeping models and IAMs estimate the anthropogenic land CO_2 flux (Land Use Change) considering only the impact of "direct human induced effects" (i.e., land use change, harvest, regrowth) and a relatively small area of managed forest. The DGVMs estimate the non-anthropogenic land CO_2 flux (Land Sink) considering the impact of "indirect human-induced effects" (environmental changes such as climate change, atmospheric CO_2 increase, N deposition etc.) and of "natural effects" (e.g., interannual variability) in both managed and unmanaged lands. The anthropogenic land CO_2 flux in country GHG inventories (LULUCF) include the impact of direct effects, and in most cases of indirect effects, from all lands considered "managed"; furthermore countries use a much bigger area of managed forests than those used by bookkeeping models and IAMs.

The approach used by countries on the area follows the methodological guidance provided by the IPCC for estimating GHG inventories (IPCC 2006, 2019). The separation of anthropogenic from non-anthropogenic effects on the land CO_2 sink is impossible with direct observation (IPCC 2010). Since most country GHG inventories are fully or partly based on direct observations, such as national forest inventories, the IPCC adopted the 'managed land' concept as a pragmatic proxy to facilitate GHG inventory reporting. Anthropogenic land GHG fluxes (direct and indirect effects) are defined as all those occurring on managed land, that is, where human interventions and practices have been applied to perform production, ecological or social functions (IPCC 2006, 2019). The contribution of natural effects on managed lands is assumed to be negligible over time. GHG fluxes from unmanaged land are not reported in GHG inventories because they are assumed to be non-anthropogenic. The definition of managed land used in country GHG inventories is typically much broader than the one used by bookkeeping models and IAMs. For example, managed forests in GHG inventories may include parks and protection forests, while global models include only those areas that were subject to intense and direct management such as harvest.

In summary, global models consider as managed forest those lands that were subject to harvest whereas, consistent with IPCC guidelines, country GHG inventories define managed forest more broadly. On this larger area, inventories often consider the natural response of land to human-induced environmental changes (indirect effects) as anthropogenic, while the global model approach treats this response as part of the non-anthropogenic sink.

7.8.2 Reconciling the differences in AFOLU CO_2 flux between global models and countries

Reconciling the differences in AFOLU CO_2 emissions between global models and country GHG inventories is important to build confidence in land-related CO_2 estimates and in the associated mitigation potential (Grassi et al. 2018). In particular, such reconciliation would facilitate the use of land use emission results by IAMs to assess, under the global stocktake, the collective gap between the country mitigation ambition and a well-below-2 °C pathway. Even if the current discrepancies between global models and country GHG inventories can be harmonized (Rojeli et al. 2011) or corrected for, this may increase the uncertainty of the future emission gap (Rojeli et al. 2016).

Although the full reconciliation of the conceptual differences outlines above would require a long-term workplan, some short-term solutions have been proposed and successfully implemented for the historical period (Grassi et al. 2018). Such solutions imply a post-processing of current global models' results, reallocating the impact of human-induced environmental change from managed land (indirect effects, considered "non-anthropogenic" and thus part of the Land Sink component) to the "anthropogenic" net land flux.

This reallocation of global models' results, conceptually illustrated in table 7.13, would increase their comparability with country GHG inventories for the historical period and with country climate pledges (NDCs) for the future. In the context of the global stocktake, such reallocation would help reducing the uncertainty of the future emission gap without affecting the decarbonization pathways that are consistent with the Paris Agreement (Rockström et al. 2017).

Table 7.13 Reallocation of anthropogenic and non-anthropogenic land CO_2 flux categories by global models to facilitate the comparison with country GHG inventories. The split of the Land Sink (left column) into fluxes occurring in managed and unmanaged lands (right column) increases consistency

with country GHG inventories (d = a+b = "LULUCF") while maintaining consistency with previous categories by global models (a: "Land Use Change"; b+c: "Land Sink"). Adapted from Grassi et al. (2018).

Anthropogenic and non-anthropogenic land CO_2 flux categories by global models	**Reallocation of anthropogenic and non-anthropogenic land CO_2 flux categories compatible with country GHG inventories**
Land Use Change by bookkeeping models and IAMs (land use change, harvest and regrowth, only in managed land)	a) CO_2 from direct effects in managed land
Land Sink by DGVMs (natural response of land to environmental change, in both managed and unmanaged land)	b) CO_2 from indirect effects in managed land
	c) CO_2 from indirect effects in unmanaged lands
	d) **LULUCF** in country GHG inventories: a + b

By applying this re-allocation to the historical period (2005-2014), Grassi et al. (2018) found that the impact of indirect effects in managed forests (derived from DGVMs) accounted for most of the difference on the anthropogenic CO_2 flux between the bookkeeping models and the GHG inventories, especially when differences in managed area between global models and countries were taken into account.

7.9. Knowledge gaps

Placeholder-For SOD:

- Understanding functioning of the biosphere
- Effects of management/ effectiveness
- Permanence
- Future impact disturbances
- Cost efficiency
- Drivers and policy effectiveness
- monitoring
- baselines
- approach for GST

7.10. Case studies

Case 1. The climate-smart village approach

Aim - The climate-smart villages (CSV) approach aims to generate local knowledge on climate change adaptation and mitigation while improving productivity, food security, and farmers' livelihoods (Aggarwal et al., 2018). The knowledge generated at the local level with farmers, researchers, practitioners, and governments, feeds a global network of knowledge that includes 36 climate-smart villages in Asia, Africa, and Latin America. The CSV approach is holistic and seeks for an integrated vision, so that rural sustainable development is the final goal for rural communities while they understand climate and implement actions to adapt to climate changes and mitigate GHG emissions as much as possible. Rural communities and local stakeholders are the leaders of this process, where scientists facilitate their knowledge to be useful for the communities and learn at the same time about challenges but also the capacity those communities have built through time.

Process - Understanding the priorities, context, challenges, capacity, and characteristics of the territory and the communities regarding climate, as well as, the environmental and socioeconomic dimensions is the first step. Then, understanding climate vulnerability in their agricultural systems based on scientific data but also listening to their experience will set the pathway to identify climate-smart agriculture options (practices and technologies) to reduce such vulnerability. Those practices and technologies are part of a farm adaptation plan, which also includes costs and a timeline to implement them. Rural communities understand that putting this practices and technologies in place will contribute to sustained and diversified production even in drier periods, which means food for their families and even an income increase.

Building capacity is also a key element of the CSV approach, rural families learn about the practices and technologies in a neighbor's house while they are being implemented, and as part of the process families commit to sharing their knowledge with other families, in order to star a scaling-out process within the communities. Understanding the relationship between climate and their crop is key, as well as the use of weather forecasts to plan their agricultural activities. Also, an assessment of the implementation of the CSA options is done together with community leaders in order to understand changes in livelihoods and climate vulnerability.

Results - As mentioned above, the CSV approach involves a variety of stakeholders, including rural communities but also local governments, which are key for scaling purposes and sustainability in the long term. In a CSV located in Cauca, Colombia, while carrying out the process explained above, local government was part of it, initially just as observer but after as funder to implement the approach in other municipalities (Ortega & Martínez-Barón, 2018b). In adittion, knowledge appropriation in Cauca by community leaders has led to famer-to-farmer knowledge exchange within and outside the community (Ortega & Martínez-Barón, 2018a).

The CSV process includes a strong component on participatory research, which has enabled dialogue between scientist and farmers to co-generate knowledge useful to implement CSA. As part of this process, Acosta-Alba et al. (2019) found in Cauca CSV that the introduction of compost allowed an improvement in mitigation of 22% to 41% for the coffee crop systems of all types of farms. Productivity was also improved by between 30% and 60% thanks to reduced production costs. The introduction of compost also made it possible, for all types of farms combined, to reduce GHGs by between 3% and 33%. As expected, this analysis applied to mitigation showed that the practice of compost had limited effects on farms where livestock units exist. Moreover, farmers are using research results to promote their products as climate-smart leading to increases in their income.

According to Acosta-Alba et al. (2019), the analysis of the contribution of emissions by item for the indicators in tension (Climate change potential, Acidification and Terrestrial Eutrophication) made it possible to see which part of the coffee production process contributed to the different potential impacts before and after the introduction of compost (Figure 7.32). GHG emissions that occurred upstream from the farm came mainly from the manufacture of fertilizers and lime used for growing coffee. These represented between 30% and 52% of total emissions and corresponded to orders of magnitude encountered in the literature (van Rikxoort et al., 2014). Compost was therefore a favourable alternative in this respect because it rendered it possible to reduce this type of emissions occurring upstream of production, which only accounted for 11% to 22% of total emissions.

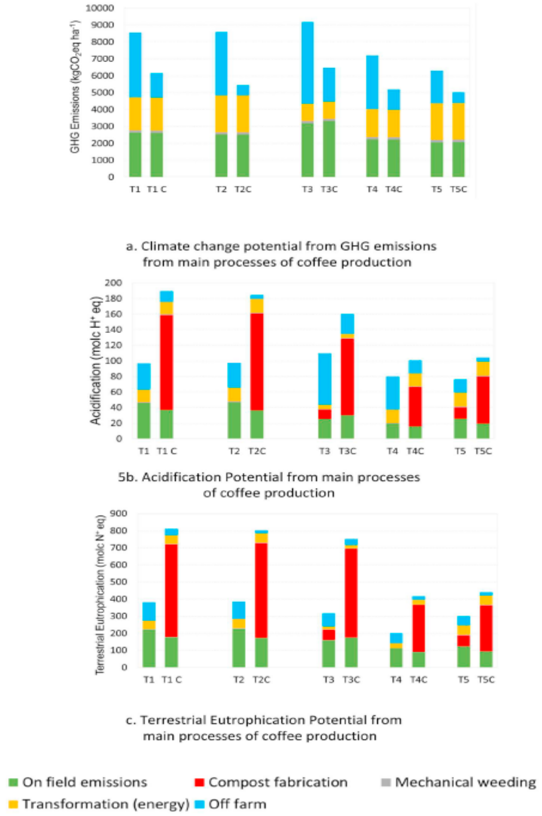


Figure 7.32 Analysis at the coffee crop system level (productive year), of the main spots of contribution to (a) potential climate change, (b) terrestrial eutrophication and (c) acidification, for the baseline (T) and compost (TC) scenarios and for the 5 types of farms.

Additional CSA practices implemented in the CSV are describe below, however the emphasis is mainly in adaptation and productivity, which is expected in small-holder farmers, although an effort is made to identify potential co-benefits with mitigation.

Improved bean/maize varieties resistant to climate stresses/biofortifed: this option is combined with the organic fertilizer production, which allows reducing production costs. Also, farmers are able to know local agroclimatic forecasts can plan their crop avoiding additional costs or potential losses. Beans production also incentivized its consumption in villages where they were not used to eat it, such change in behavior led to a shift in consumption patterns leading to savings in animal protein household expenditure. Families are saving approximately USD$24.oo/month of the expenditure of their basic shopping basket in the CSV located in Cauca, Colombia.

Climate-smart home gardens (circular, vertical, traditional): Rural households were both buying vegetables from the closest urban area or not eating vegetables at all. Vegetable home gardens were built in the past, however in strong rainy or drought season the harvest was lost. This is why the climate-smart home gardens include water harvesting and drip-irrigation so that farmers can collect water during rainy seasons and use it efficiently to irrigate the vegetables. In addition, the climate-smart home gardens include a protective cover in case of hail, strong wind or rain. Rural households in Cauca shift from buying almost 22 food products, including vegetables, in the urban market to less than 10, which are the ones they do not produce (e.g. sugar, salt, etc.). Women farmers are leading the implementation of this CSA practice, they have organized and now they sell vegetables in the urban area making additional income for the household. Organic compost for crop fertilization is also used in the climate-smart vegetable home gardens, which helps to reduce production costs and produce organic food.

Water-related CSA options (*Optimizer Hydraulic water pump, rainwater harvesting, and efficient irrigation mechanisms and water reservoirs and efficient irrigation mechanisms*) were designed so that, rural families could keep producing throughout the year despite climate behavior. As they now know how weather and climate behave in their locality and have access to agroclimatic forecast, they are able to make a more efficient use of water resources. This has reduced additional costs due to potential losses due to climate and have now a more secure maintenance or even an increase in their productivity and income.

Crop management with crop rotation (Legumes/ No legumes)**:** This practice consists of rotating the crops that are no more than two continuous cycles of the same crop in the same plot. In addition, legumes allow nitrogen fixation in the soil, improving its quality and reducing nitrogen fertilizer use, which generates emissions. This practice allows increase suitability of crops because crop rotation disease cycles can be interrupted.

Crop management with intercropping (Legumes/ No legumes)**:** To sow different crops in the same plot in the same cycle, to diversify production and take advantage of the benefits that one crop can generate over another on input use intensity. By diversifying the crops, farmers can harvest and generate income even in excessive rain or drought seasons. Legumes allow nitrogen fixation in the soil, improving its quality and reducing nitrogen fertilizers used, which generate GHG emissions.

Case 2. India: Mitigation Options and Costs in the Agricultural Sector

Introduction:

India is one of the largest GHG emitter in the world and the agricultural sector alone accounts for almost 18%of India's total GHG emissions (Sapkota et al, 2019). India has therefore identified the agricultural sector as one of the priority sectors for emission reductions in its National Determined Contribution (NDC) (Sapkota et al., 2019). There are considerable cost-effective mitigations options in the agriculture sector which needs to be tapped. Sapkota et al (2019) in their study use large data sets and information along with stakeholder engagement and expert opinion to assess the technical mitigation potentials of Indian agriculture and costs under a Business as Usual scenario and Mitigation scenario up to 2030.Their study shows that by 2030 under Business as Usual scenario GHG emissions from the agricultural sector in India would be 515 Megatonne CO_2 equivalent ($MtCO_2e$) yr^{-1} with a technical mitigation potential of 85.5 $MtCO_2e$ yr^{-1} through adoption of various mitigation practices. About 80% of the technical mitigation potential could be achieved by adopting only cost-saving measures. Their analysis shows that three mitigation options, i.e. efficient use of fertilizer, zero-tillage and rice-water management, could deliver more than 50% of the total technical abatement potential.

Data and Methodology:

In the study by Sapkota et al (2019) the authors adopted a bottoms-up approach to analyse GHG emissions. They used the large data set of India's Cost of Cultivation surveys collected from across India covering different crops and seasons; the nineteenth livestock data, along with soil, climate and management data for each location. Mitigation potential and costs and benefits of adoption of mitigation practices were gathered from the literature, stakeholder engagement and expert opinion etc.

GHG emissions from crops were calculated using the Cool Farm Tool (CFT) (Hillier et al., 2011). The CFT is a GHG emission calculator that allows users to estimate GHG emission associated with the production of crops or livestock products (Hillier et al., 2011). Sapkota et al (2019) used a version of the CFT scripted in Matlab to calculate the emissions for on-farm plots across India. GHG emissions for rice production was estimated using the method of Yan et al (2005) which bases estimates of CH_4 emission on several variables (soil pH, climate, organic amendment, pre-water regime, water regime). These factors were available at the plot level for India in this study. Background and fertilizer-induced N_2O emissions were calculated based on the updated nitrogen model of Stehfest and Bouwman (2006). Emissions from crop residues returned to the field were estimated using IPCC N_2O emission factors. Emissions from the production and transportation of fertilizer was based on the Ecoinvent database (Ecoinvent Centre, 2007). Changes in soil C due to tillage, manure and residue management are based on IPCC methodology as in Ogle et al (2005) and Smith et al (1997). Emissions of CO_2 from soil resulting from urea application or liming were estimated using the IPCC methodology.

GHG emissions from livestock husbandry were calculated using the approach of Herrero et al. (2013) which provided data on GHG emissions from enteric fermentation and manure management for several animal groups (i.e. ruminants, small ruminants, pigs and poultry), which was tailored to various livestock management systems under different agro-ecologies in India. National GHG emissions were calculated based on the average body weight of the livestock for different regions. Emissions arising from feed production were not included in this analysis as livestock feeding in India largely depends on crop by-products and concentrate, the environmental footprint of which is included in crop emissions. The study accounted for only GHG emissions related to farm management, and did not account for processing, marketing or consumption post farm-gate. GHG emissions up to the farm-gate are reported in CO_2 equivalent (CO_2e) ha^{-1} of crops and per head for livestock using the 100 year global warming potentials For each crop and livestock type, state-level mean emission and standard deviation were obtained from the spatial model run using all available data-points within a state. The state means were then multiplied by state-level total area (for crops) and number of animals (for livestock) to obtain state totals. Emissions from all the states were summed-up to obtain total national emissions.

The study also held stakeholders' workshops involving participants from different subsectors of Indian agriculture including crops, livestock and natural resource management to discuss about the technical mitigation potentials and barriers to realise them.

Results:

Mitigation potentials and costs for Indian agriculture (for crop production, livestock production and restoration of degraded lands) for different mitigation options is indicated in Table 7.14. Figure 7.33 presents the information on GHG emissions and mitigation options for Indian agricultural sector up to 2030 under business as usual and mitigation scenarios. Sapkota et al (2019) note that as with the 2012 baseline, the most important sources of projected emissions under the BAU scenario were cattle followed by rice, buffalo and small ruminants. Although livestock production and rice cultivation are the major contributors of agricultural emissions, the highest mitigation potential was observed in rice (~36$MtCO_2e$ yr^{-1}) followed by buffalo (~14 $MtCO_2e$ yr^{-1}), wheat (~11 $MtCO_2e$ yr^{-1}) and cattle (~7 $MtCO_2e$ yr^{-1}). Cotton and sugarcane each offered mitigation potential of about 5 $MtCO_2e$ yr^{-1}. Technical mitigation potential from goat/sheep was about 2 $MtCO_2e$ yr^{-1}.

Land-based mitigation measures provide a low-cost option to mitigate the adverse effects of climate change. Figure 7.34 and Figure 7.35 present information on the marginal abatement cost curve of Indian agriculture without considering additional yield benefits or alternatively considering the additional yield benefits associated with the adoption of mitigation measures. Using the bottom up approach Sapkota et al (2019) estimated the total mitigation potential for Indian agriculture at 85.5 $MtCO_2e$ per year. Figure 7.34 and Figure 7.35 show the magnitude of GHG savings per year through adoption of various mitigation measures, together with the total cost (Figure 7.34) and net cost (Figure 7.35) per unit of CO_2e abated. Many of the mitigation measures employ currently available technologies and can be implemented immediately. The cost-beneficial measures have negative cost and appear below the x-axis on the left-hand side of the graph, whereas the cost-incurring measures appear above the x-axis, on the right-hand side of the graph. When the additional benefits of increased yield due to adoption of the mitigation measures were considered, about 80% of the technical mitigation potential (67.5 out of 85.5 $MtCO_2e$) could be achieved by cost-saving measures (Figure 7.35). When yield benefits were considered, green fodder supplement to ruminant diets was the most cost-effective mitigation measure, followed by vermicomposting and improved diet management of small ruminants. Mitigation measures such as fertigation and micro-irrigation, various methods of restoring degraded land and feed additives in livestock appear to be cost-prohibitive even when considering the yield benefits, if any.

Table 7.14 GHG mitigation options along with their mitigation potential and cost of adoption in Indian agricultural sector[@]

GHG abatement options	Mitigation potential[a]	Gross cost of mitigation[b]	Net cost of mitigation[b]
Crops			
Improved water management in rice	2760	−1378	−3445
Adoption of zero-tillage	518 to 1796	−963 to −308	−1690 to 208
Stop residue burning	−3 to 522	−6278 to − 498	−6278 to − 498
Fertilizer production	57 to 529	Not considered	Not considered
Fertilizer consumption	47.83 to 198.46	−710 to −2327	−710 to −2327
Laser Land levelling	1284 to 3055	1000	−5188
Increase NUE through fertigation	170 to 4999	25,000	21,750
Sprinkler/micro-sprinkler irrigation	163 to 1276	10,000	8700
Livestock			
Green fodder supplement for large ruminants	32.23 to 38.84	2957 to 4106	−14,783 to −5493
Increased concentrate feeding for large ruminants	116.77 to 139.82	4654 to 6894	−2340 to 128
Monensin pre-mix for large ruminants	32.23 to 38.84	61,685	57,973 to 60,316
Molasses Urea Product (MUP) for large ruminants	116.77 to 139.82	1460	−5964 to −1278
High fiber diet for pigs	121.75	675	−325
Improved diet management for small ruminants	21.36	189	−1411
Improved Manure management of large ruminants	30.63	13,358	−2235
Biogas from large ruminants' manure	500.23	2960	−1751
Restoration of degraded lands			
Reclamation of salinity/Alkalinity through chemical amendment	495	85,000	85,000
Reclamation of water logged soil through sub-surface drainage	183	76,000	76,000
Restoration of wind/water eroded land through Jatropha plantation	275	1833	−2000
Restoration of wind/water eroded land through plantation	275	71,500	71,500
Controlling wind/water erosion through contour farming/wind breaks/water flow breaks etc.	275	45,500	26,000

Note:

@ The range of values indicate the mitigation potential and costs when mitigation options are applied to multiple crops or livestock. When mitigation options are applied to a single crop or livestock, a single value of mitigation potential and cost is given.

a. $kgCO_2eha^{-1}yr^{-1}$ for options related to crop management and restoration of degraded land and $kg\ CO_2eha^{-1}yr^{-1}$ for the options related to livestock management.

b. Indian Rupees (INR)ha^{-1} for options related to crop management and restoration of degraded land and INR/head for the option related to livestock management.

c. 1 USD= Indian Rupees (INR) 68.41 in 2018 (Yearly average exchange rate).

Source: Sapkota et al (2019)

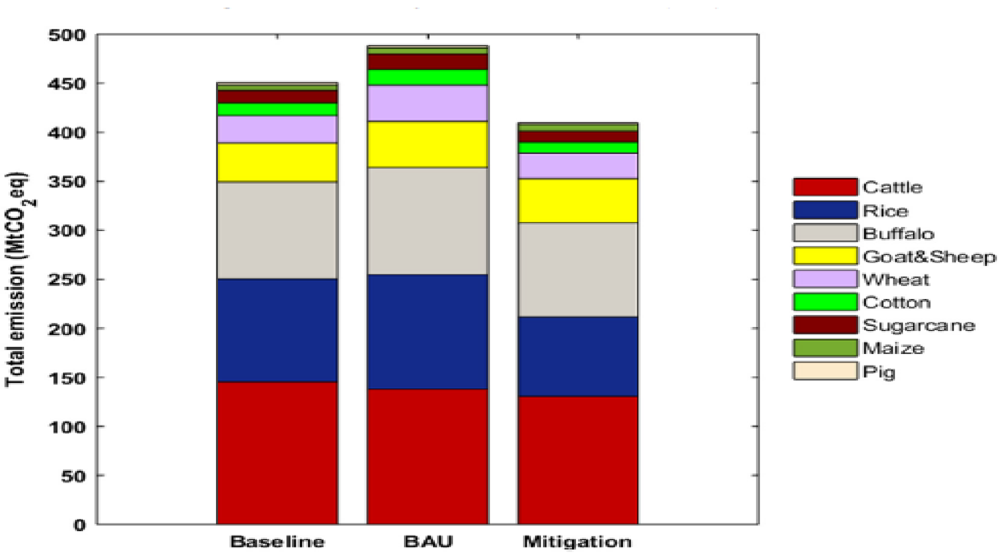


Figure 7.33 Contribution of various crops and livestock species total agricultural emission in 2012 (baseline) and by 2030 under business as usual (BAU) and mitigation scenarios for Indian Agricultural sector. Source: Sapkota et al (2019)

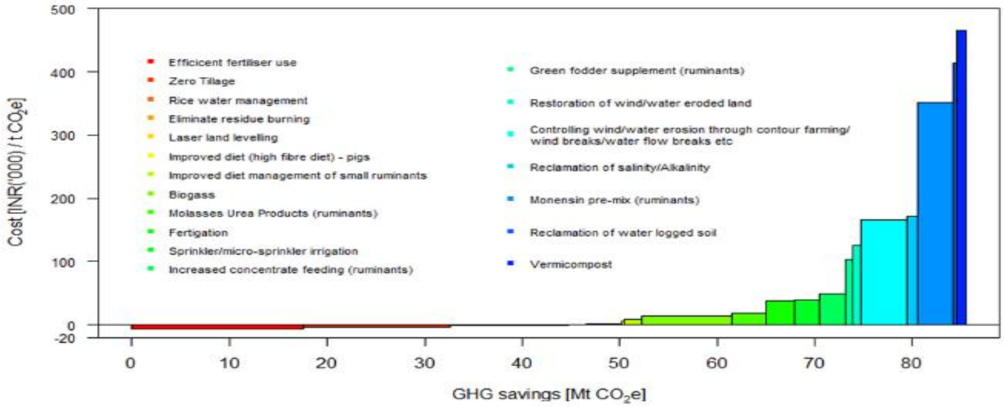


Figure 7.34 Marginal Abatement Cost Curve of Indian agriculture without considering additional income from increased yield associated with the adoption of mitigation measures. The width of the bar represents the abatement potential from the mitigation option whereas height of the bar represents the average cost per unit of CO_2e abated. The area (height x width) of the bar represents the total cost of the action i.e. how much would it cost to altogether in order to deliver all of the CO_2e savings from the action. For description of the colours used in the figure see the web version of Sapkota et al (2019). Source: Sapkota et al (2019)

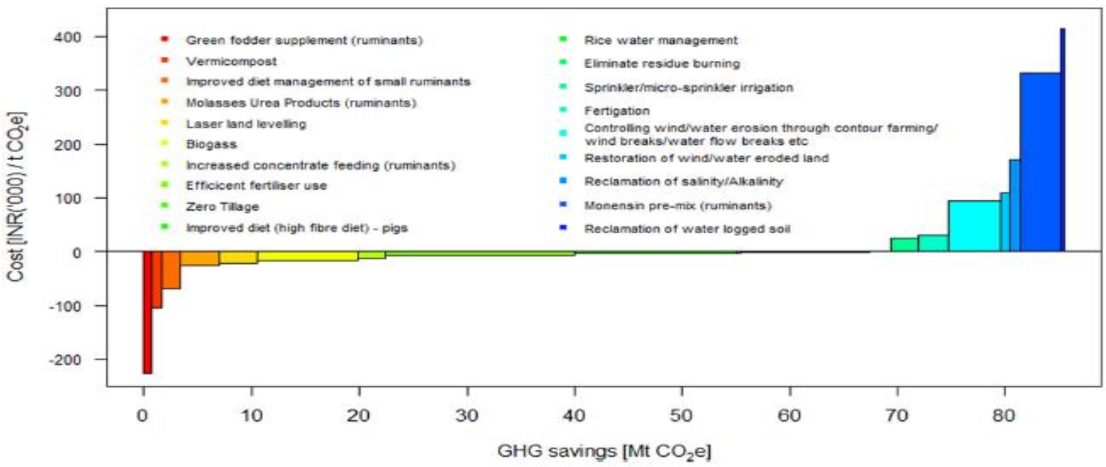


Figure 7.35 Marginal Abatement Cost Curve of Indian Agriculture considering additional yield benefit of adopting mitigation measures. The width of the bar represents the abatement potential from the mitigation option whereas height of the bar represents the average cost per unit of CO_2e abated. The area (height x width) of the bar represents the total cost of the action i.e. how much would it cost to altogether in order to deliver all of the CO_2e savings from the action. For description of the colours used in the figure see the web version of Sapkota et al (2019)

Case 3. Climate Smart Forestry in Europe

European forests have been regarded as prospering and increasing for last 5 decades. However these views also changed recently. Climate change is putting a large pressure on Norway spruce stocks in Central Europe (Nabuurs et al. 2019) with estimates of mortality reaching 200 million m^3, biodiversity under pressure, the Mediterranean area showing a weak sector and harvesting pressure in the Baltics and north reaching maxima achievable. In addition, future demand will increase the forest carbon sink needs to be strengthened.

A European strategy for unlocking the EU's forests and forest sector potential based on the concept of "Climate Smart Forestry" (CSF) was elaborated (Nabuurs et al. 2017). CSF is a more specific (climate-oriented) form of the Sustainable Forest Management paradigm. The idea behind CSF is that it considers the whole value chain from forest to wood products and energy, and illustrates that a wide range of measures can be applied to provide positive incentives for more firmly integrating climate objectives into the forest and forest sector framework. CSF is more than just storing carbon in forest ecosystems; it builds upon three main objectives; (i) reducing and/or removing greenhouse gas emissions; (ii) adapting and building forest resilience to climate change; and (iii) sustainably increasing forest productivity and incomes. These three CSF objectives can be achieved by tailoring policy measures and actions to regional circumstances in Member States forest sectors.

The current annual mitigation effect of EU forests via contributions to the forest sink, material substitution and energy substitution is estimated at 569 Mt CO_2/year, or 13% of total current EU emissions. With the right set of incentives in place at EU and Member States levels, it was found that the EU has the potential to achieve an additional combined mitigation impact through the implementation of CSF goals, of 441 Mt CO_2/year by 2050. Climate Smart Forestry is now taking shape across Europe in many countries with various research and implementation projects. Also with Commissioner Timmermans' new climate law, more emphasis will be placed on forests, forest management and the provision of renewables. It is in a diversity of measures, taking into account the local state of forest resources and local needs that will determine the success. Only with co-benefits in a.o. nature conservation, soil protection, and provision of renewables and income, the mitigation and adaptation measures will be successful. There is no doubt that a closer collaboration between industry and forest owners will be needed to make these measures happen. And especially the larger (often) public owners will have to be in the forefront. They will have to establish examples and take care of outreach to small owners. However, the right triggers and incentives are often still lacking. E.g. adapting the spruce forest areas in Central Europe to climate change requires knowledge about different species and different management options. It simply also requires alternative species to be available from the nurseries.

Further, better monitoring will be needed; this is the only way forward when carbon credits from land are to be accepted. Today, the monitoring is too slow, and too fragmented. Member States do not feel obliged to send detailed data to common databases, and thus estimates of harvesting levels fluctuate and are seen as unreliable with discussions about biomass for bioenergy mounting on this uncertainty. Also estimates of spruce mortality fluctuate a lot, even after two years.

Finalising: a joint effort between Commission, Member States, industry, research and large public owners will be needed to tackle the challenges as outlined above. Only then Climate smart forestry will make its way into a large roll out and into practice.

Case 4. Sustainable rice management

Rice systems provide food for more than 3.5 billion people which is about half of the world population and accounted for 19% of dietary globally. More than 50 kg of rice being consumed per capita per year globally. In addition it is expected that rice cultivation need to be increase 46 % by the end of 2030s to meet the increasing demand of global population (FAO). Ninety five percent of the rice production is from developing country which 90 percent of the total production produced in Asia. Rice cultivation area cover 167 Mha of harvested area and contribute around 18.3 +/-0.1 -38.8 +/-1.0 Tg $CH_4 yr^{-1}$ (Zhang et al. 2016) depending on different water management practices. This emission account for 10% of the total global emission. In addition, global methane emission form rice field increased by 85%from 1901-2010.

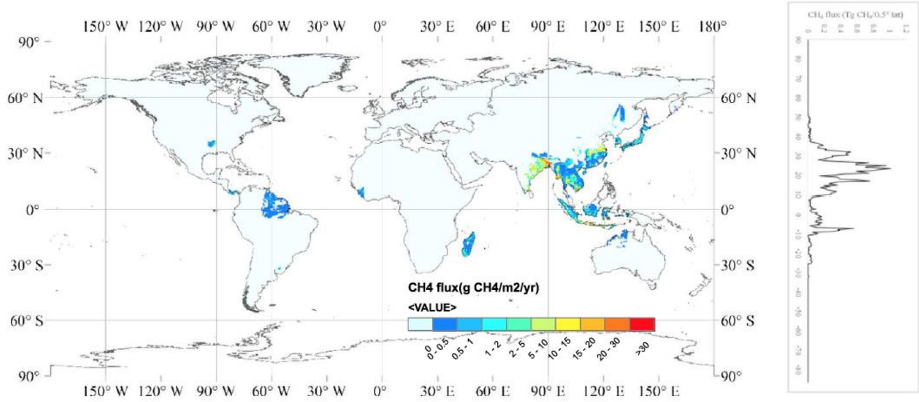


Figure 7.36 Spatial distribution of estimated mean annual CH_4 emissions from global rice fields during 1993-2007. (Source: Zhang et.al. 2016)

Mitigation options

Emissions from rice field are mainly from methane under anaerobic condition through methanogenesis process and nitrous oxide under anoxic condition through nitrification and denitrification process. Intensity of emission is usually due to several influential factors some of which are degradation of organic matter in the soil, management of water level in the field, application of various types and amount of fertilizers, rice variety plantation and local cultivation practices.

Current mitigation options are focused on techniques and management that related to influential factors with less impact or promote rice yield. Since rice cultivation involve with farmers mostly in developing countries, reduction of GHG from rice cultivation should integrate farmers concerns and promising options of real implementation with less interfere livelihood of farmers. They are for example, water management including single drainage and multiple drainage, soil amendment including biochar and organic amendment, appropriate fertilizer application including slow release fertilizer and nutrient specific application.

Integrative mitigation of GHG reduction, rice yield and water use

Water management is the promising technology as it can integrate the issue of GHG reduction, rice yield and water productivity together. In general, rice is an aqua plant that need appropriate amount of water consumption. Level of water in the field can be managed to reduce methane emission with the single drainage and multiple drainage in the middle of growing period. Alternative wetting and drying system is one of the multiple drainage that are promoted recently as one of the promising option. Meta-analysis of AWD indicated that under the mild AWD condition, AWD with irrigation management, can promote sustainable intensified goals by reduce both methane emission (20-30%) and water used (25.7 %) despite slightly less yield production (5.4%) (Carrijo et al. 2017). Nevertheless, analysis of AWD implementation in 5 countries of ASEAN include (Yagi et al. 2019) Cambodia, Indonesia, Philippines, Thailand and Vietnam showed that rice yield can be maintained while methane emission is reduced. However, due to condition of wet and dry in the soil which can induce oxygen and cause nitrification and denitrification occurred leading to increasing of N_2O emission in the field. The quantity of N_2O emission can not offset the total GWP when combine both gases together.

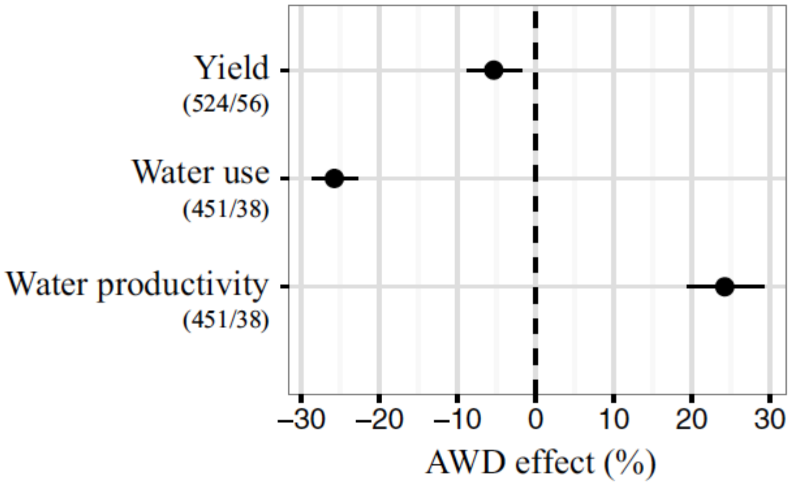


Figure 7.37 Effect of AWD on yield, water use (irrigation+ rainfall) and water productivity (grain yield/water use). Mean effect sizes (●) and bootstrapped confidence intervals(-) are represented. The number of observations/number of studies included in each data set are indicated in parenthesis.
(Source: Carrija et al 2017)

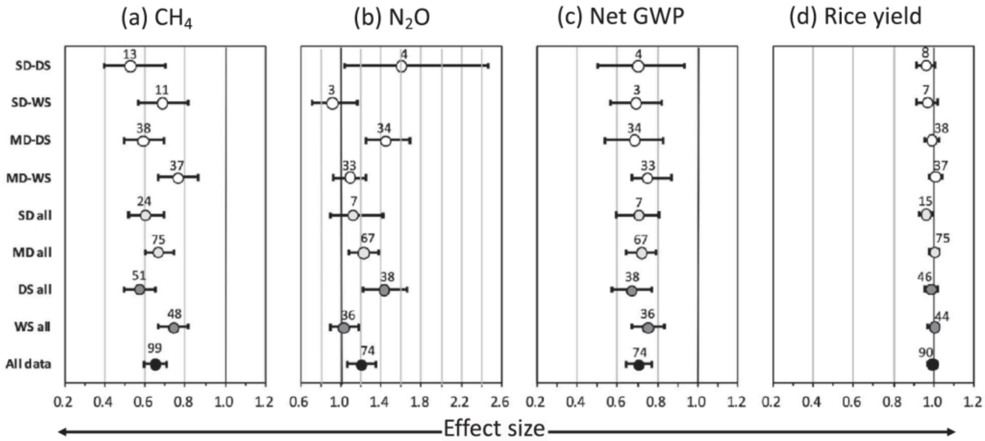


Figure 7.38 Effects of water management options on CH_4 emissions (a), N_2O emissions(b), net GWP(c) and rice yield (d) from rice cultivation in Southeast Asian countries. Mean effects and 95% confidence intervals are shown by symbols and bars, respectively. Numerals indicate numbers of data. SD; Single drainage; MD: multiple drainage including AWD; DS: dry season; WS: wet season.
(Source: Yagi et al. 2019)

Implementation of sustainable rice in some Southeast Asian countries

Major rice harvested area are located in Asian countries contributed for 87%of the total global area (167 Mha in 2017). The area is 8.6%increase since 2000. As for Southeast Asian countries, in 2017, rice harvested area cover approximately 50 M ha (30% of the Asian harvested area) with the increasing rate of 17% since 2000. Total emission from rice field from Southeast Asian countries is approximately 150,000 $GgCO_2e$. Out of these, emission from Indonesia, Philippines, Thailand and Vietnam cover approximately 78%of emission from rice field in ASEAN country.

Implementation in Vietnam

Vietnam is among top five countries in the rank of global rice exporter. Rice is grown throughout the country with irrigation land cover around 80% of rice area. Since 2005, water management system in term of AWD has been officially introduced to rice farmers by local government as part of the 1M5R (One must do 5 reduction) agrarian campaign in order to increase the efficiency of rice cultivation

(Lampayan et al., 2015). The safe AWD, with 5 cm of water level in the field and 15 cm dry below the soil surface indicated by plastic pipes, is introduced.

An Giang is the first province that AWD first diffusion in 2009 with the rate of 18%of the total rice cultivation area. In 2015, the diffusion rate increases to 52%while the adoption rate of farmers household to AWD is 54% (Yamaguchi et al., 2019). Also some communes of Phu Tan and Cho Moi districts had more than 75% AWD adoption rate in 2015. However, there are some communes in the Tri Ton district including Ba Chuc and Tan Tuyen that AWD adoption rate is declined due to restriction factors of each area, which are natural factors including different percolation and seepage rates as resulted of different elevation of paddy plots and fluctuation of precipitation , agro-engineering factors including density and quality of water canals, pump ownership status and paddy surface level, and social factors including farmer's understanding of AWD, contracted paddy cultivation and synchronizing water management with neighbouring plot (Yamaguchi et al., 2017). There are also others barrier identify by Quynh and Sander (2015) such as poor irrigation system, level and size of rice field, different type of soil, conflict on benefits between farmers and pumping stations etc.

GHG reduction and water use

Rice cultivation under AWD (safe AWD) and AWDS (site specific AWD) in Huong Tra district, Thua Thien Hue Province, can reduce CH_4 and N_2O emissions by 29% to 30% and 26% to 27%, respectively with the combination of net GWP about 30% as compared to continuous flooding (Tran et al. (2018)). Water use is also accounted for 15% reduction. In addition the system increase water productivity from 0.556 kg grain m^{-3} to 0.727 kg grain m^{-3}, accounted for a 31% increase.

Impact on yield and cost

Three years consecutive study of AWD implemented in Thua Thien Hue Province showed the development of grain yield 10-11% higher than conventional field (Tran et al. 2018). Yields increasing are different in season. Implementation of AWD system in dry season can increase yield by 6-15 % in An Giang Province (Ha et al., 2014) while during spring season and summer season at Nam Sach district, Hai Duong province, yield improvement is found at 8 % and 20 % increasing , respectively, when compare to conventional practice (Quynh and Sander 2015). The achievement of higher yield may resulted from decreasing plant disease, insect damage, and poor grain filling, as well as promoting root spread (Yamaguchi et al. 2017).

In term of economic benefit, farmers gain more incomes (22%) due to the reduction of production cost including seed (14%), pesticide (35%), pumping and labors (5%), while increased fertilizer cost by 12 % (Quynh and Sander 2015). In addition farmers can save the pumping cost and harvest cost (Yamaguchi et al., 2017). The economic benefit depends on many factors including site specific constrains and farmer's practice related to their understanding.

Implementation in Thailand

Rice is the most important economic crops that significantly contribute to Thai socioeconomic development in the past decades with its harvested land cover more than 50 % of total country agricultural land. Contribution of GHG emission from agricultural sector is accounted for 18 % of the national total while emission from rice cultivation contributed 51 % of the agricultural sector.

In 2018, The Thai government has initiated the Thai Rice NAMA project with the collaboration of Ministry of Natural Resources and Environment (MoNRE) and Ministry of Agriculture and Cooperatives (MoAC) and supported by Deutsche Gesellschaft für Internationale Zusammenarbeit (GIZ) GmbH Germany. The project aims to enables farmers to implement low-emission rice farming, as well as supports entrepreneurs in providing mitigation services (land laser levelling, alternate wetting and drying, site-specific nutrient management & straw/stubble management) to farmers, finally to dedicate to policy formulation and supporting measures promoting low-emission production at the national political level. Six provinces in the central part of Thailand including Chainat, Suphanburi, Singburi, Angtong, Ayuthaya and Pathumthani are the target area with 100,000 farmer household and 420 mitigation service providers. The total areas involved are 455,420 ha of rice cultivation.

Mitigation Potential

Four technologies are implemented in the target area including alternate wetting and drying (AWD), site-specific nutrient management (SSNM), land laser levelling (LLL) and mechanized straw/stubble management (SSM). The baseline of emission is calculated using IPCC methodology at 3,152,858 tCO_2e annually. The annual mitigation potential is estimated at 918,820 tCO_2e in Year 5 when 80%of 100,000 target farmer households have adopted the basic mitigation practices. Therefore, the cumulative mitigation potential over the five-year implementation period is estimated at 1,730,106 tCO_2e. The technologies with the highest mitigation potential are AWD (721,661 $tCO_2e\ yr^{-1}$) and SSNM (74,603 $tCO_2e\ yr^{-1}$).

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