Chapter 4. Terrestrial and Inland Water Systems

Coordinating Lead Authors
Robert Scholes (South Africa), Josef Settele (Germany)

Lead Authors
Richard Betts (UK), Stuart Bunn (Australia), Paul Leadley (France), Daniel Nepstad (USA), Jonathan Overpeck (USA), Miguel Angel Taboada (Argentina)

Contributing Authors
Craig Allen (USA), Bill Anderegg (USA), Celine Bellard (France), Paulo Brando (Brazil), Franck Courchamp (France), Wendy Foden (South Africa), Dieter Gerten (Germany), Scott Goetz (USA), Nicola Golding (UK), Patrick Gonzales (USA), Ed Hawkins (UK), Thomas Hickler (Germany), Josh Lawler (USA), Heike Lischke (Switzerland), Georgina Mace (UK), Melodie McGeoch (Australia), Camille Parmesan (USA), Richard Pearson (USA), Beatriz Rodriguez-Labajos (Spain), Rebecca Shaw (USA), Stephen Sitch (UK), Klement Tockner (Germany), Marten Winter (Germany)

Review Editors
Andreas Fischlin (Switzerland), Jose Moreno (Spain), Terry Root (USA)

Volunteer Chapter Scientist
Marten Winter (Germany)

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There is high confidence that the magnitude of future climate change could approach that of many of the largest climatic changes observed in Earth history. There is also high confidence that the planet’s biota, carbon cycle and associated feedbacks and services responded to this climate change, even when the rates of past global climate change were slower than implied by higher warming scenarios (e.g., RCP 8.5). [4.2.2] However, the impacts of climate change on terrestrial and freshwater ecosystems must also be considered in the context of non-climatic influences, both naturally-occurring and directly driven by humans. There is high confidence that most ecosystems change over time, due to biological interactions and natural environmental disturbances, even under climate change slower than projected for low- to medium-range warming scenarios (e.g., RCP2.6 to RCP6.0). [4.2.2]
Few ecosystems can now be considered to be unaffected by human activity. Vegetation cover classed as “primary” and essentially undisturbed by humans is estimated with high confidence to have decreased from approximately 95% of the global land area in 1500 AD to less than 40% at the start of the 21st Century, with half of this reduction being at the expense of primary forest. [4.2.1, 4.2.3, 4.2.4] There is medium confidence that significant feedbacks exist between terrestrial ecosystems and the climate. Thus local, regional and global climate may be affected as ecosystems are altered, through climate change itself or other mechanisms, such as conversion to agriculture or human settlement. These climate feedbacks are driven by changes in surface albedo, evapotranspiration and greenhouse gas emissions. The regions where the climate is affected may be different from the location of the ecosystem change. [e.g. 4.3.3.4]

There is high confidence for freshwater ecosystems and medium confidence for terrestrial ecosystems that direct human impacts such as land-use change, pollution and water resource development will continue to dominate the threats to ecosystems, with climate change becoming an increasing additional stress through the century, especially for high-warming scenarios such as RCP6.0 and RCP8.5. [4.3.3] Model-based projections imply that direct land cover change will continue to dominate over climate-induced change for low to moderate warming scenarios at global scales (e.g., RCP2.6 to RCP6.0). However, in many areas not subject to intensive human disturbance, there is high confidence that even lower levels of projected future climate changes will result in changes in large-scale ecosystem character depending on the nature of regional climate changes. Such changes may not be fully apparent for several decades after reaching the critical regional climate state, due to long response times in ecological systems (medium confidence). For higher warming scenarios, some model projections imply climate-driven large-scale ecosystem changes which become comparable with direct human impacts at the global scale (medium confidence). There is high confidence that rising water temperatures will lead to shifts in freshwater species distributions and compound water quality problems, especially in those systems experiencing high anthropogenic loading of nutrients. There is medium confidence that changes in precipitation will significant alter ecologically important attributes of flow regimes in rivers and wetlands and exacerbate impacts from human water use in developed river basins. [Box CC-RF, 4.3.3.3]

There is high confidence that the capacity of many species to respond to climate change will continue to be constrained by non-climate factors, including but not limited to the simultaneous presence of land-use changes, habitat fragmentation and loss, competition with alien species, exposure to novel pests and diseases, nitrogen loading, increasing carbon dioxide and tropospheric ozone. [Figure 4-1, 4.2.4-4.2.4.6, 4.3.3-4.3.3.5]

There is high confidence that a changing climate exacerbates other threats to biodiversity. In some systems, such as high altitude and latitude freshwater and terrestrial ecosystems, there is high confidence that climate changes exceeding those projected under RCP2.6 will lead to major changes in species distributions and ecosystem function. [4.3.2.5, 4.3.3.1, 4.3.3.3, 4.4.1.1] Since the specific changes in individual regions depends on the nature of the projected regional climate change, the confidence in specific future ecosystem changes is limited by the confidence assigned to regional climate change projections by Working Group I.

There is high confidence that plants and animal species have moved their ranges, altered their abundance and shifted their seasonal activities in response to climate change in the past. There is also high confidence that they are doing so now in many regions and will continue to do so in response to projected future climate change. [4.2.2, 4.3.2, 4.3.2.1, 4.3.2.5, 4.3.3] The broad patterns of species and biome movement towards the poles and higher in altitude in response to a warming climate are well established for the distant (very high confidence) and recent past (medium confidence). High confidence in these past responses, coupled with projections from a diversity of models and studies provide high confidence that such responses will be the norm with continued warming. [4.2.1, 4.2.2, 4.3.2, 4.3.2.1, 4.3.2.5] There is high confidence that these shifts in species ranges will cause large changes in local abundance under all climate change scenarios: abundance declining in areas where climate becomes unfavorable and potentially increasing in areas where climate becomes more favorable. [4.3.2.5] There is high confidence that such changes in species abundance lead to changes in community composition and ecosystem function. There is high confidence, based on extensive ground and satellite-based measurements that the seasonal activity of species has responded to warming over the last several decades. [4.3.2, 4.3.2.1, 4.4.1.1] Observations and models of the seasonal activities of species indicate that climate warming disrupts species life cycles and
interactions between species, as well as altering ecosystem function. [4.3.2.1, 4.4.1.1] At local scales, observed and
modelled species responses sometimes differ from qualitative predictions based on global scale indices of warming;
this can often be explained by large variation in local scale climate response to global warming, changes in climate
factors other than average temperature, non-climatic determinants of species distributions, interactions between
climate and other simultaneous global change factors such as nitrogen deposition, and species interactions. [4.3.2.1,
4.3.2.5, 4.4.1.1] There is high confidence that no past climate changes is a precise analog to the current and
projected climatic changes, so species responses inferred from the past only give indications, especially at the local
scale. [4.2.2]

There is high confidence that climate change is increasing the likelihood of the establishment, growth, spread
and survival of some invasive alien species populations in some regions. Alien invasive species are more likely
than native species to have traits that favor their survival and reproduction under changing climates. Species
movement into areas where they were not present historically will be driven both by climate change and by
increased dispersal opportunities associated with human activities. [4.2.4.6]

There is medium confidence that even for mid-range rates of climate change (i.e., RCP4.5 and RCP6.0
scenarios) many species will be unable to move fast enough to track suitable climates. Over the last several
decades many species have tracked changes in climate. However, there is medium confidence that some species have
been unable to track recent climatic changes. [4.3.2.5] Populations of species that cannot track future climate change
by migrating will find themselves in unfavorable climates and are unable to expand into newly climatically suitable
areas. Species in large flat areas are particularly vulnerable because they must migrate over longer distances to keep
up with climate change than species in mountainous regions. Species with low migration capacity will also be
especially vulnerable: examples include most trees, many plants, and some small mammals. Combinations of low
migration capacity and large flat areas are projected to pose the most serious problems for tracking climate; for
example, there is medium confidence that even the maximum observed and modelled migration rates for mid- and
late-successional tree species will be insufficient to track climate change in flat areas even at moderate rates of
climate change. There is high confidence that barriers to migration such as mountain ranges, dams, habitat
fragmentation and occupation of habitat by competing species substantially reduce the ability of species to migrate
to more suitable climates. There is high confidence that outlier populations (e.g., collections in botanical gardens or
parks), as well as intentional and accidental anthropogenic transport will speed migration. [4.3.2.5, 4.3.3.3]

There is high confidence that large magnitudes of climate change will negatively impact species with
populations that are primarily restricted to protected areas, mountaintops or mountain streams, even those
that potentially migrate fast enough to track suitable climates. [4.3.2.5, 4.3.4.1] Climate change is projected with
high confidence to either create unsuitable climates for species that remain in these areas, or force species out of
protected areas and off mountaintops. These effects are foreseen to be modest for low magnitudes of climate change
(e.g., RCP 2.6) and very high for the highest magnitudes of projected climate change (e.g., RCP 8.5). There is high
confidence that species have already started to migrate out of protected areas and towards mountaintops over the last
several decades due to a warming climate. [4.3.2.5, 4.3.4.1]

There is very high confidence that projected climate changes imply increased extinction risk for a substantial
fraction of species during and beyond the 21st century, especially as climate change interacts with other
pressures. [4.3.2.5] Uncertainties in regional climate projections, highly variable estimates from comparisons of
paleontological extinctions in response to past climate changes, different methods of estimating present and future
extinction risk, and the variable adaptive capacity of wild species all contribute to an extremely broad range of
estimates of future extinction risk due to climate change. There is high confidence that current extinction pressures
from habitat modification, over-exploitation and invasive species are continuing, and will interact with growing
impacts of climate change. When combined with methodological challenges in modelling future extinction risks,
this means there is low confidence that global extinction risks due to climate change can be accurately quantified.
There is, however, a strong consensus that current climate change pressures and their interactions with other global
changes will increase extinction risk for many terrestrial and freshwater species. There is very low confidence that
observed species extinctions can be attributed to recent climate warming given the very low fraction of species for
which global extinction has been ascribed to climate change and tenuous nature of most attributions. However, in
the specific case of Central American amphibians, there is medium confidence that recent warming has played a role in their extinctions. [4.3.2, 4.3.2.5]

It is virtually certain that the carbon stored in land and freshwater ecosystems in the form of plant biomass and soil organic matter has increased over the past two decades in what is known as the terrestrial carbon sink. There is low confidence that the transfer of carbon dioxide from the atmosphere to the land will continue at a similar rate for the remainder of the century. There is high confidence that the terrestrial carbon sink is offset to a large degree by carbon released to the atmosphere through forest conversion to farm and grazing land and through forest degradation [4.2.4.1, 4.3.3.1]. There is medium confidence that the carbon stored thus far in terrestrial ecosystems is vulnerable to loss back to the atmosphere as a result of climate change (including indirect effects such as increased risk of fires and pest outbreaks) and land-use change. Terrestrial and freshwater ecosystems have been responsible for the uptake of about a quarter of all anthropogenic CO\textsubscript{2} emissions in the past half century. [4.3.2.2 and 4.3.2.3] The net fluxes out of the atmosphere and into plant biomass and soils show large year-to-year variability. As a result there is low confidence in the ability to determine whether the net fluxes into or out of terrestrial ecosystems at the global scale have increased or decreased over the past two decades. There is high confidence that the factors causing the current increase in land carbon include the positive effects of rising CO\textsubscript{2} on plant productivity, a warming climate, and recovery from past disturbances, but low confidence regarding the relative contribution by each of these and other factors. [4.2.4.1, 4.2.4.2, 4.2.4.4, 4.3.2.2] Experiments and modelling studies provide medium confidence that increases in CO\textsubscript{2} up to about 600 ppm will continue to enhance photosynthesis and plant water-use efficiency, but at a diminishing rate. [4.2.4.4]. There is also medium confidence that other factors associated with global change, including high temperatures, rising ozone concentrations and in some places drought, decrease plant productivity by comparable amounts [4.2.4.3, Box 4-3]. Models provide high confidence that nitrogen availability will limit the response of many natural ecosystems to rising CO\textsubscript{2} [4.2.4.2]. There are few field-scale experiments on ecosystems at the highest CO\textsubscript{2} concentrations projected by RCP8.5 for late in the century, and none of these include the effects of other potential confounding factors [4.3.2.2, 4.2.2.3].

There is medium confidence that increases in the frequency or intensity of ecosystem disturbances due to fires, pest outbreaks, wind-storms and droughts have been detected in many parts of the world. There is high confidence that such changes, which are by definition beyond the range of historical natural variability, will alter the structure, composition and functioning of ecosystems. There is medium confidence that such changes will often be manifested as relatively abrupt and spatially-patchy transitions following disturbances, rather than gradual and spatially-uniform shifts in location or abundance. [Box 4-3, Box 4-4, 4.2.4.6, 4.3.3, 4.3.2.5, Figure 4-12]

There is evidence of an increase in tree mortality in many regions over the last decade, but there is low confidence in the detection of a global trend in increased mortality or in the attribution of such a global trend to climate change. In some regions, increased tree mortality is sufficiently intense and widespread as to result in forest dieback. Forest dieback constitutes a major risk because of its large impacts on biodiversity, wood production, water quality, amenity, economic activity and the climate itself [4.3.3.1, 4.3.3.4, 4.3.4, Box 4-2, Box 4-3]. In detailed regional studies, particularly in western and boreal North America, there is high confidence that observed tree mortality is detectable and can be attributed to the direct effects of high temperatures and drought, or to changes in the distribution and abundance of insect pests and pathogens related, in part, to warming.

Recent experimental, observational and modelling studies provide medium confidence that forests may be more sensitive to future climate change than reported in IPCC AR4 assessment and that tree mortality and forest dieback could become a problem in many regions much sooner than previously anticipated. [4.3.3.1] There is high confidence that future climate change impacts on tree mortality and tree ranges could be large, but experimental, observational and modelling studies also indicate that there is low confidence associated with model-based projections of the details of these impacts. As such, projections of increased tree growth and enhanced forest C sequestration mediated by increasing growing season length, rising CO\textsubscript{2} concentrations and atmospheric N deposition are being viewed with increasingly greater caution due to the counter-balancing effects of mortality and dieback. [4.3.3.1] The consequences for the provision of timber and other wood products are projected to be highly variable between regions and products depending on the balance of the positive vs. negative effects of global change. [4.3.3.1, 4.3.3.5, 4.3.4]
In the Amazon, there is medium confidence that climate change alone will not lead to widespread forest loss during this century. However, there is also medium confidence that a combination of severe drought, land use and fire act together to transform mature forests to less dense, fire prone woodland and shrublands.

Modelling studies project a range of changes in the regional climate of the Amazon, and very few indicate a state that cannot support rainforest. Nevertheless, many models still project a general drying tendency, and coupled with warming this may increase fire risk. Human land use is associated with fire, so a combination of direct human influence and climate change may still affect the Amazon. [4.3.3.1.3, Box 4-3]

There is high confidence that terrestrial and freshwater ecosystems can, when pushed by climate change, cross “tipping points” and abruptly change in composition, structure and function. There is medium confidence that the crossing of these tipping points will result in significant increases in carbon emissions to the atmosphere. This has happened many times in Earth history. [4.2.2]. There are plausible mechanisms, supported by experimental evidence and model results, for the existence of ecosystem tipping points in both boreal-arctic systems and the rainforests of the Amazon basin [Boxes 4-3 and 4-4], others may exist. There is a plausible but low confidence case that continued climate change could push the boreal-arctic system across such a tipping point in this century and cause an abrupt transformation of the ecology and albedo of this region, as well as the release of greenhouse gases from the thawing permafrost and burning forests. There is high confidence that adaptation measures will be unable to prevent substantial change in the boreal-arctic system. There is also a plausible but low confidence case that continued climate change together with land-use change and fire activity could cause much of the Amazon forest to transform abruptly to more open, dry-adapted ecosystems, and in doing so, put a large stock of biodiversity at elevated risk, and create a large new net greenhouse gas source to the atmosphere. [4.3.3.1]. There is medium confidence that the combination of climate change and land-use change in the Amazon will cause accelerated drying and drought frequency in the region and low confidence that these Amazon changes will affect rainfall in agricultural regions elsewhere on the planet. [4.3.3.1]. There is medium confidence that rigorously applied adaptation measures could lower the risk of abrupt change in the Amazon, as well as the impacts of that change. [Box 4-3, 4.3.3-4.3.3.4, Figure 4-10]. There is very high confidence that policy and market-driven interventions have caused a steep decline in deforestation in the Amazon since 2005 that have decreased anthropogenic carbon emissions to the atmosphere by 1.5%. [4.2.4.1, 4.3.3.1]

There is high confidence that management actions can reduce, but not eliminate, exposure to climate-driven ecosystem impacts, as well as increase ecosystem adaptability. [4.4] The capacity for natural adaptation by ecosystems and their constituent organisms is substantial, but for many ecosystems and species there is medium confidence that this is insufficient to cope without substantial loss of species and ecosystem services, given the rate and magnitude of climate change projected under medium-range warming (e.g., RCP6.0) or high–range warming scenarios (e.g., RCP8.5). The capacity for ecosystems to adapt to climate change can be increased by reducing the other stresses operating on them; reducing the rate and magnitude of change; reducing habitat fragmentation and increasing connectivity; maintaining a large pool of genetic diversity and functional evolutionary processes; assisted translocation of slow moving organisms or those whose migration is impeded, along with the species on which they depend; and manipulation of disturbance regimes to keep them within the ranges necessary for species persistence and sustained ecosystem functioning. [4.4.1-4.4.1.2, 4.4.3]

There is medium confidence that management adaptation responses to climate change in some sectors will lead to unintended and unwanted outcomes for terrestrial and freshwater ecosystems. For example, adaptation responses to counter increased variability of water supply for urban and agricultural use, such as building more and larger impoundments and increased water abstraction, will compound the direct effects of climate change in freshwater ecosystems. [4.3.3.3, 4.3.4.5] There is very high confidence that the use of the terrestrial biosphere in climate mitigation actions, such as through introduction of fast-growing tree species for carbon sequestration or the conversion of forest to biofuel plantations, may lead to negative impacts on ecosystems and biodiversity. The land use scenario accompanying the mitigation scenario RCP2.6, intended to avoid 2°C global warming, features large expansion of biofuel production, displacing natural forest cover. [4.2.4.1]
4.1. Past Assessments

The topics assessed in this chapter were last assessed by the IPCC in 2007, principally in the Working Group II report chapters 3 (Freshwater resources and their management; Kundzewicz et al., 2007) and 4 (Ecosystems, their properties, goods and services; Fischlin et al., 2007). Together they found that ‘Observational evidence from all continents and most oceans shows that many natural systems are being affected by regional climate changes, particularly temperature increases’. Although circumstantial evidence was offered that anthropogenic climate change was a cause of the observed changes, it was not possible to attribute a relative proportion of the changes to this cause with any confidence. An important finding was that 20-30% of the plant and animal species that had been assessed to that time were considered likely to be at increased risk of extinction if the global average temperature increases exceeded 2-3°C. Fischlin et al. (2007) also stated that substantial changes in structure and functioning of ecosystems (terrestrial, marine and other aquatic) are very likely under a warming of more than 2-3°C above pre-industrial levels. No timescale was associated with these findings. The report noted that vulnerability to climate change could be exacerbated by the presence of other stresses, such as pollution.

4.2. A Dynamic and Inclusive View of Ecosystems

Ecologists no longer view ecosystems as inherently unchanging unless perturbed by an externally originating disturbance (Hastings, 2004). A growing body of both theory and observation (Scheffer, 2009) supports the emerging view that most ecosystems vary substantively over time in the relative magnitude of their components and fluxes, even under a relatively constant environment. Furthermore, attempts to restrict this intrinsic variation - or that resulting from imposed disturbances - are frequently futile, and may damage the capacity of the ecosystem to adapt to changing environments (Folke et al., 2004). Contemporary treatments of ecosystems treat humans as an integral part of the system, rather than as an external perturbation to the natural state without any feedback to the human system itself (Gunderson and Holling, 2001). The emerging view is of relatively tightly coupled social-ecological systems, which means that their joint dynamics are governed not only by biophysical processes such as energy flows, material cycles, competition and predation, but also by social processes such as economics, politics, culture and individual preferences (Walker and Salt, 2006).

4.2.1. Ecosystems as Adaptive Entities

There is increasing focus on how ecosystems behave in the vicinity of their environmental limits, rather than solely on their behavior near the middle of their domain of their inferred preferred state (Scheffer et al., 2009). The notion of thresholds has become a prominent ecological and political concern (Lenton et al., 2008; Knapp et al. 2008, Scheffer et al., 2009; Leadley et al., 2010). Some thresholds reflect a human preference that the ecosystem stays within certain bounds. Others reflect fundamental biophysical properties, for instance the thawing of frozen soils (see Box 4-4); or the physiological tolerance limits of constituent species. A third category of threshold relates to system dynamics under externally-driven change: the point at which the net effect of all the positive and negative feedback loops regulating the ecosystem is positive, causing a small perturbation to become amplified sufficiently to lead to a change in ecosystem state, from which it will not spontaneously return in the short term. The new state will typically exhibit different dynamics, mean composition, sensitivity to environmental drivers and flows of ecosystem services relative to the prior state. This type of threshold is called a tipping point (see glossary).

The early detection and prediction of ecosystem thresholds, particularly tipping points, is an area of active research. There are indications (Scheffer, 2009) that an increase in ecosystem variability signals the impending approach of a threshold. In practice, such signals may not be detectable against background noise and uncertainty until it is too late to avoid the threshold (Biggs et al., 2009). The dynamics of ecosystems are complex and our present level of knowledge is often inadequate to predict an ecosystem outcome with confidence, even if the future climate was precisely known.

The term ‘resilience’ (see glossary) has attracted a range of meanings in different disciplines. In ecology it is used both in the sense of ‘a measure of the ability of these systems to absorb changes of state variables, driving variables
and parameters, and still persist’ (Holling, 1973), and also in a broader sense of being able to adapt to change, without necessarily remaining within the same dynamical range (Walker et al., 2004) and has been applied in the context of climate change (e.g. Morecroft et al., 2012). One source of such adaptability is evolution within the populations of organisms making up the ecosystem. Natural selection operates on genetic variability against those individuals less able to survive, compete and successfully reproduce in the altered environment, leading to a shift, over generations, in the tolerance range of the population. Organisms also typically exhibit physiological, developmental or behavioral plasticity, within limits, that does not require genetic change. A particular case occurs when humans in the ecosystem use learning and technology to alter how they interact with the system. A third form of adaptation occurs when the configuration and functional form of interactions within an ecosystem change, in some cases leading to a new stable state (www.regimeshifts.org). It occurs when the abundance of one or more species are greatly altered or are replaced by other species with similar functional attributes. In the extreme, this would constitute a change in ecosystem type.

Field observations over the past century in numerous locations in boreal, temperate and tropical ecosystems have detected shifts of plant species at the biome level upwards in elevation and to higher latitudes (Gonzalez et al., 2010). These shifts have been formally or informally attributed to climate change (Table 4-1, Figure 4-1). Projections of potential future vegetation indicate substantial susceptibility of ecosystems to biome shifts (Warszawski et al., submitted; Figure 4-2, Box 4-3, Box 4-4).

Table 4-1: Biome shifts from 1700 to the present from published field research that examined trends over periods > 30 y for biomes in areas where climate (rather than land-use change or other factors) predominantly influenced vegetation, derived from a systematic analysis of published studies (Gonzalez et al. (2010). Pre-AR4 publications are included to provide a comprehensive review. Shift type: elevational (E), latitudinal (L), examined but not detected (N). Biomes (and abbreviations), from poles to equator: tundra and alpine (UA), boreal conifer forest (BC), temperate conifer forest (TC), temperate broadleaf forest (TB), temperate shrubland (TS), tropical grassland (RG), tropical woodland (RW). Rate of change in temperature (Temp.) and fractional rate of change in precipitation (Precip.) are derived from linear least squares regression of 1901-2002 data (Mitchell and Jones, 2005; Gonzalez et al., 2010). The table provides general regional climate trends at 50 km spatial resolution because the references do not give uniform site-specific climate data to compare across locations. The regional trends are consistent with local trends reported in each reference. * rate significant at P ≤ 0.05.)

Figure 4-1: Biome shifts from 1700 to the present at sites in Table 4-1, derived from Gonzalez et al. (2010). Temperature change is the rate from linear least squares regression of 1901-2002 temperatures (Mitchell and Jones, 2005; Gonzalez et al., 2010). Numbers refer to the sites in Table 4-1. Arrows indicate general direction of shifts.

Figure 4-2: Fraction of land surface at risk of severe ecosystem change as a function of global mean temperature change for all ecosystems models, global climate models and RCPs. The colors represent the different ecosystems models, which are also horizontally separated for clarity. Results are collated in unit-degree bins, where the temperature for a given year is the average over a 30-year window centered on that year. The median in each bin is denoted by a black horizontal line. The grey boxes span the 25th and 75th percentiles across the entire ensemble. The short, horizontal stripes represent individual (annual) data points; the curves connect the mean value per ecosystem model in each bin. The solid (dashed) curves are for models with (without) dynamic vegetation composition changes. Right: Fraction of global land surface subject to severe ecosystem change as a function of global mean temperature (left panels) and time (right panels) for the JeDi (upper) and JULES (lower) dynamic global vegetation model driven by the HadGEM2-ES global climate model. The colors represent the different RCPs used to drive the climate model. Good agreement of results at different levels of global warming demonstrates that results are independent of the emissions scenario. (Reproduced from Warszawski et al., submitted)
4.2.2. What the Paleoecological Record Can and Cannot Tell Us

Paleoclimatic observations and modelling indicate that the Earth’s climate has always changed on a wide range of time-scales. In many cases, particularly over the last million years, it has changed in ways that are well-understood in terms of both patterns and causes, (Jansen et al., 2007; IPCC AR5 WGI Chapter 5). Independent paleoecological records demonstrate with high confidence (robust evidence, high agreement) that the planet’s biota (both terrestrial and aquatic), carbon cycle and associated feedbacks and services have responded to this climatic change, particularly when the climatic change was as large as projected future mid- to high-end change (e.g., MacDonald et al., 2008; Claussen, 2009; Arneth et al., 2010; Willis and MacDonald, 2011; Dawson et al., 2011). Excellent examples of past large climate change events that drove large ecological change include the events that led to the Earth’s five mass extinctions in the distant past (i.e., during the Ordovician, ca. 443 Ma, the Devonian, ca. 359 Ma, the Permian, ca. 251 Ma, the Triassic, ca. 200 Ma, and the Cretaceous, ca. 65 Ma; Barnosky et al., 2011). Major ecological change was also driven by climate change during the Paleocene-Eocene Thermal Maximum (PETM, 56 Ma; Wing et al., 2005; Jaramillo et al., 2010), the early Eocene Climatic Optimum (EECO, 53-50 Ma; Woodburne et al., 2009), the Pliocene (5.3 to 2.6 Ma; Haywood and Valdes, 2006; Haywood et al., 2011), and the Last Glacial Maximum (LGM; 21 ka) to Holocene (11 ka) transition (MacDonald et al., 2008; Clark et al., 2009; Gill et al., 2009; Williams et al., 2010b; Prentice et al., 2011; Daniau et al., 2012). The paleoecological record thus provides high confidence (robust evidence, high agreement) that large climate change, such as that projected for the 21st century, can result in large ecological changes, including large scale biome shifts, reshuffling of communities and, potentially, species extinctions.

Rapid, regional warming before and after the Younger Dryas cooling event (11.7-12.9 ka) provides a relatively recent analogy for climate change at a rate of a speed that approaches that projected for the 21st century (Alley et al., 2003; Steffensen et al., 2008). Ecosystems and species responded rapidly during this period by shifting distributions and abundances, and there were some notable large animal extinctions, probably exacerbated by humans activities (Gill et al., 2009; Dawson et al., 2011). In some regions, species became locally or regionally extinct, but there is no evidence for climate-driven large-scale extinctions during this period of rapid climate change (Botkin et al., 2007; Willis et al., 2010c), except for a small number of plant species (Jackson and Weng, 1999; Birks and Willis, 2008). However, these climate excursions differ from those projected for the future because they were regional rather than global; may have only regionally exceeded rates of warming projected for the future; and started from a baseline substantially colder than present (Alley et al., 2003). The mid-Holocene around ca. 6 ka provides a very recent example of the effects of modest climate change, because regional warming during this period (ca. 0.5-1.5°C above pre-industrial temperatures in some regions) was the same order of magnitude as the warming the Earth has experienced over the last century. Ecological effects were small compared to periods with larger climate excursions, but even this small degree of warming was characterized by frequent fires in a drier Amazon (Mayle and Power, 2008), development of lush vegetation and lakes in a wetter Sahara (Watrin et al., 2009), temperate deciduous forests in Europe expanding further north and up to higher elevations (Adamik and Kral, 2008), and large-scale migration of Boreal forest into a warmer tundra (Jackson and Overpeck, 2000). Past climate change, even more modest than mid-range projected future change, has also clearly impacted inland water systems (e.g., Smol and Douglas, 2007b; Battarbee et al., 2009; Beillman et al., 2009). However, there are no exact analogues for future climate change: none of the well-studied past periods of large climate change involved simultaneously the rates, magnitude and spatial scale of climate change projected for the next century and beyond (Jansen et al., 2007; Schulte et al., 2010; IPCC AR5 Chapter 5). Direct analogy with the paleoecological record is also unwarranted because future climate change will interact with other global changes such as land-use change, invasive species, pollution and overexploitation of natural resources (Pereira et al., 2010). There is high confidence (robust evidence, high agreement) that these interactions will also be important: the paleoecological record provides medium confidence (medium evidence, high agreement) that exploitation by humans helped drive many large mammal species to extinction during periods of climate change in past (Lorenzen et al., 2011).

It has been demonstrated that state-of-the-art vegetation models are able to simulate much of the biome-level equilibrium response of terrestrial to large paleoclimate change (Prentice et al., 1996; Salzmann et al., 2008; Prentice et al., 2011). These same types of models predict large changes in species ranges, ecosystem function and carbon storage when applied to 21st century climate change (Sitch et al., 2008; Cheaib et al., 2012). Thus, the paleoecological record, and models that have been tested against that record provide a coherent message that biomes...
will move, species mixtures will change, novel new plant communities will emerge, and significant carbon changes will take place, all in response to changing and often novel future climates (Williams and Jackson, 2007; Williams et al., 2007b; MacDonald, 2010; Prentice et al., 2011; Willis and MacDonald, 2011; see Section 4.3.2.5 for more specific information on migration rates). As such, the paleoecological record and models provide high confidence (robust evidence, high agreement) that it will be difficult or impossible to maintain many ecological systems in their current states if global warming exceeds 2 to 3°C, raising questions about the long-term viability of some current protected areas and conservation schemes, particularly where the objective is to maintain present-day species mixtures (Jackson and Hobbs, 2009; Hickler et al., 2012).

Much of the complex, time-dependent change at regional scales has not yet been simulated by models. The paleoecological record indicates that vegetation in many parts of the world has the potential to respond within years to a few decades to climate change (e.g., Watrin et al., 2009; Williams et al., 2009; Mueller et al., 2009a; Harrison and Goni, 2010), and this same record thus provides a critical model evaluation opportunity that should be more thoroughly exploited to gain confidence in time-dependent simulations of future change, particularly given the complex role that interacting climate change and vegetation disturbance has played in the past (e.g., Marlon et al., 2009; Jackson et al., 2009; Williams et al., 2009; Daniau et al., 2010; Dawson et al., 2011).

The paleoclimatic record also reveals that past radiative climate forcing change was slower than that anticipated for the 21st century (IPCC AR5 Chapter 12), but even these slower changes often drove surprisingly abrupt, or non-linear, regional change in terrestrial and aquatic systems (e.g., Harrison and Goni, 2010; Williams et al., 2011), as did even slower orbitally-driven change during the most recent Holocene interglacial (e.g., Booth et al., 2005; Kropelin et al., 2008; Williams et al., 2010b; Williams et al., 2011). In all cases, specific periods of abrupt ecological response were regionally distinct in nature and were less synchronous for small, slow changes in forcing (e.g., during the Holocene) than for more global rapid changes listed at the start of this section. State-of-the-art climate and Earth system models are unable to simulate the full range of abrupt change observed in many of these periods (e.g., Valdes, 2011) and thus there is medium confidence (medium evidence, medium agreement) that these models may not capture some aspects of future climate change and associated ecosystem impacts (Leadley et al., 2010).

4.2.3. Landscapes and Social-Ecological Systems

Ecosystems do not exist in isolation from one another or from human systems. Landscapes, which consist of multiple interacting ecosystems, all experience some degree of interaction with people. The responses of terrestrial and freshwater ecosystems to climate change are therefore mediated by human activities at a range of spatial and temporal scales, and to varying degrees. Together, the social and ecological elements form a linked system (a ‘social-ecological system, SES) in which the degree of coupling can range from very close to slight (Berkes et al., 2003).

4.2.4. Multiple Stressors and the Role of Climate Change

While isolated effects of the main drivers of global change (e.g. climate change, habitat loss, chemical pollution and biological invasions) are increasingly well documented (Millennium Ecosystem Assessment, 2005b; Settele et al., 2010a) there is much less knowledge about their consequences when acting in combination. Multiple stressors can act in a non-additive way (Settele et al., 2010b; Shaw et al., 2002; Larsen et al., 2011), potentially invalidating findings and interventions based on single-factor analysis. In a multifactorial climate change experiment Larsen et al., 2011), demonstrated that non-additive interactions among the climate factors were frequent but most often antagonistic leading to smaller effects than predicted from the single factor effects. Leuzinger et al., (2011) and Dieleman et al., (2012) have synthesized such multifactor interactions and demonstrated that in general, the effect size appears to be reduced when more factors are involved. Leuzinger et al. (2011) further showed that models tend to show the opposite trend.
Climatic and non-climatic drivers of ecosystem change must be distinguished if the joint and separate attribution of changes to their causes is to be performed – of paramount importance if the processes of change are to be understood and future changes predicted. Within this section we elaborate on drivers and stressors which often act in concert with climate change, ranging from a high degree of relatedness to climate change (rising CO₂; tropospheric ozone; total, diffuse and UV radiation), over intermediate (land-use change) to low degrees of relatedness (invasive species and N deposition).

4.2.4.1. Land-Use and Cover Change (LUCC)

Land use and land cover continues to change worldwide. In tropical and subtropical areas of Asia, Africa, Oceania and South America, the most widespread land-use changes are conversion of forests and woodlands to annual and perennial agriculture, grazing pastures and commercial plantations, followed by conversion of grasslands and pastures to annual agriculture. Afforestation at the expense of agricultural lands is more frequent in Europe (Rounsevell and Reay, 2009; Schwaiger and Bird, 2010; Miyake et al., 2012) and loss of agricultural land to urban settlements is common in both the developed and developing world. LUCC is both a cause and consequence of climate change. The underlying drivers of LUCC include rising demand for food, fiber and bioenergy, and changes in lifestyle and technologies – some of which are also drivers of climate change. In the future climate change is projected to become a major driver of land cover change and ecosystem functioning, with more than a quarter of the land surface projected to be at risk of severe ecosystem change at global warming of 4°C or above (Figure 4-2).

Land cover change can also be caused by change in non-climate environmental drivers such as nutrient loading, pollution and disturbance regimes.

The cause-and-effect entanglement of climate change and LUCC make attribution of consequences to one or the other very difficult. In 13 of the 24 regional land use studies reviewed for this chapter, local-to-regional climate change was at least partly attributed to LUCC, generally with limited evidence and low confidence. (Direct climate effects attributed to LUCC: Tseng and Chen, 2008; Cui and Graf, 2009; Li et al., 2009; McAlpine et al., 2009; Zhang et al., 2009; Fall et al., 2010; Graiprab et al., 2010; Jin et al., 2010; Mishra et al., 2010; Schwaiger and Bird, 2010; Wu et al., 2010; Gao and Liu, 2011; Carmo et al., 2012. No climate effects attributed: Suarez et al., 1999, Saurral et al., 2008; SuFen et al., 2008; Cochrane and Barber, 2009; Jia et al., 2009a; Rounsevell and Reay, 2009; Martin et al., 2010; Wiley et al., 2010; Clavero et al., 2011; Dai et al., 2011; Viglizzo et al., 2011; Yoshikawa and Sanga-Ngoie, 2011).

LUCC contributes to climate change through changes in the sources or sinks of greenhouse gases and through changes in the absorption of solar energy by the land surface. These factors can have opposite effects and operate at different scales. For instance, conversion of forest to non-forest generally releases carbon dioxide from biomass and soils to the atmosphere, causing warming globally; but also often results in an increase in seasonally-averaged albedo, causing cooling locally and globally (Davin et al., 2007), and a decrease in transpiration, causing warming locally but not globally. Four out of five global climate models suggested local cooling effects following forest conversion to croplands and pastures (Pitman et al., 2009). Equilibrium experiments with the GFDL climate model (Findell et al., 2007) concluded that the land-use change driven climatic impacts not mediated by greenhouse gases were generally minor, but significant in a few regions (Eastern Europe, Northern India, and Eastern China). More substantial global effects were projected by van der Molen et al. (2011) who found little temperature response to change from forests to pasture or crops in the tropics, increasing towards the Northern Hemisphere mid-latitudes. In high latitudes, changes in snow cover exposure and duration contribute to the albedo increase following forest clearing (Mishra et al., 2010, Levis, 2010).

Afforestation programs are often recommended to promote carbon sequestration (e.g. Fiorese and Guariso, 2013; Singh et al., 2013) but over the long term afforestation may lead to a slight net global warming when the albedo increase effect neutralizes the carbon uptake effect (Schwaiger and Bird, 2010). Wickham et al., (2012) found significant positive slopes in regressions of average annual surface temperatures versus the proportion of forest and argue that the value of temperate afforestation as a potential climate change mitigation strategy is unclear. Changes in albedo, other biophysical effects and some indirect greenhouse effects (such as changes in soil erosion), are not
currently included in the formal rules for accounting for the climate effects of land use activities (Schwaiger and Bird, 2010; Kirschbaum et al., 2012).

The reported effects on precipitation of conversion of forest to pastures and crops vary. Many studies report rainfall decreases, soil moisture decreases and changes in rainfall pattern (e.g. Nepstad et al., 2008; McAlpine et al., 2009; Mendelsohn and Dinar, 2009; Eliseev and Mokhov, 2011). In some cases forest clearing has been reported to enhance local rainfall. Garcia-Carreras and Parker (2011) suggest that enhanced local precipitation can occur over deforested patches due to breezes between forest and cropland. Enhanced rainfall over vegetation boundaries has been shown to occur both in the Amazon (e.g. Knox et al., 2011) and in West Africa (e.g. Garcia-Carreras et al., 2010) as a result of mesoscale processes that are currently not resolved in General Circulation Models.

In China, Han et al., (2012) compared historical data from 244 weather stations predominantly surrounded by cultivated land and 195 stations surrounded by with >70% natural cover. They observed a greater decrease in potential evaporation at stations with significant agricultural influence. In arid and semi-arid regions this was associated with a decrease in wind speed, increase in relative humidity and increase in air temperature compared with natural stations; while in humid and sub-humid regions it was associated with increased wind speed and decreased solar radiation.

Several assessments conclude that LUCC, acting through habitat loss and fragmentation, is the largest cause of contemporary terrestrial and freshwater biodiversity loss, with climate change becoming an increasingly important factor later in the 21st century (Millennium Ecosystem Assessment, 2005a; http://www.cbd.int/gbo3/).

Box 4-1. Future Land Use Pathways and Their Uncertainties

Assessment of climate change effects on terrestrial and inland freshwater ecosystems is incomplete in the absence of information on the direct impacts of people on these systems through land use. The world is undergoing important shifts in land-use patterns. Growth in demand for food, feed, fiber, and fuel has accelerated in recent years, placing new pressures on terrestrial and freshwater ecosystems and outpacing the growth in supply. The principal driver of this acceleration in demand is the rate at which per capita consumption is growing in emerging economies (Tilman et al., 2011). Policy shifts in OECD countries favoring the expansion of biofuel production have contributed to this acceleration (Lapola et al., 2010; Searchinger et al., 2008; Miyake et al., 2012). Growth in demand that outpaces growth in supply, climate-related crop failure (Lobell et al., 2011), declines in the growth of agricultural productivity (Ray et al., 2012) and possibly a shortage of arable land that is not already under cultivation, especially in the temperate zone (Lambin and Meyfroidt, 2011) are all contributing to a rise in commodity prices that may persist through 2020 (OECD/FAO, 2010) and growing pressure on as-yet uncultivated ecosystems on soils that are potentially suitable for cultivation and that are concentrated in tropical latitudes, especially South America and Africa.

Emerging economies start from a relatively low per capita consumption base. The growth of these economies will therefore play an important role in future land use trends. Deforestation in developing countries is now correlated with the export of agricultural commodities (DeFries et al., 2010), providing an early sign of the growing role of agricultural trade with emerging economies as a driver of land-use change. The trajectory of future growth in demand for land- and freshwater-based production remains uncertain, since it depends on economic trends and policies, which are themselves dependent upon complex political and social processes. One of the uncertainties about future land use trends is climate policy. Deforestation rates in the Brazilian Amazon declined 76% (as of 2012) below its ten-year average as a result of policy and market signals (Soares-Filho et al., 2010); this single trend represents a 1.5% reduction in global anthropogenic carbon emissions (Nepstad et al., in press).

These land use trends and the global land cover patterns that they promote have been simulated and mapped within a global grid as input to the Regional Concentration Pathways (RCPs) in preparation for the IPCC AR5. The 4 main RCPs used for future climate projections are accompanied by spatially-explicit future land use scenarios consistent with the emissions scenario and the associated socio-economic scenario (Hurtz et al., 2011) generated by Integrated
Assessment Models using simple methodologies intended to capture the main processes at continental and global scales.

Table 4-2: Summary of drivers and outcomes of land use / cover scenarios associated with Representative Concentration Pathways (Hurtt et al., 2011).

In the RCPs, land cover change is driven by a number of socioeconomic drivers, including climate policy. The nature of climate policy effects on land use differs between RCPs, with some climate mitigation policy scenarios including conservation of primary vegetation (natural, mature ecosystems) as carbon sinks while others included replacement of primary vegetation with biomass and/or biofuel plantations. The extent to which primary vegetation is replaced by secondary vegetation, crops or pasture also varies between the RCPs (Figure 4-3), but there is no simple linear relationship between the extent of vegetation change and the level of total radiative forcing arising from all sources. Larger reductions in primary vegetation cover are seen in RCP8.5 (the highest emissions scenario) due to a general absence of pro-active measures to control land cover change, but large reductions were also seen in RCP2.6 (an aggressive mitigation scenario with low emissions) due to widespread conversion of land to biomass and biofuel crops coupled with carbon capture and storage. Smaller reductions were seen in RCP6.0 and RCP4.5, with the latter involving conservation of primary forest or regrowth of secondary forest as a mitigation measure. Hence both the highest and lowest emissions scenarios were associated with greater extents of land cover change.

Figure 4-3: Proportion of global land cover occupied by primary and secondary vegetation (forest and non-forest), cropland, pasture and urban land, from satellite data and historical reconstructions up to 2005 (Klein Goldewijk, 2001), and from scenarios associated with the RCPs from 2005 to 2100 (Hurtt et al., 2011).

Spatial patterns of the future land cover change in all RCPs were largely similar to patterns of historical change (Figure 4-4) as most future change is projected to be a continuation of past change. By definition, primary vegetation cover cannot increase on the timescales considered here, so in scenarios where cropland and pasture are projected to decrease, these are replaced with secondary vegetation. Both tropical and boreal forest regions are projected to undergo declining primary forest cover in all 4 RCPs, but in RCP6.0 total forest area remains approximately constant.

Figure 4-4: Fractional cover of primary vegetation at 1850, 2005 based on satellite data and historical reconstructions (Klein Goldewijk, 2001) and at 2100 in scenarios associated with the RCPs (Hurtt et al., 2011).

4.2.4.2. Nitrogen Deposition

The global nitrogen (N) cycle has been strongly perturbed by human activity over the past century (Gruber and Galloway, 2008; Sutton et al., 2011). Human activities currently transform 160 TgN/year of nitrogen gas in the atmosphere into reactive forms of N that can be readily used by plants and microorganisms in land and in the ocean (Galloway et al., 2008). This is close to the pre-industrial global flux of about 255 TgN/year from all natural sources. The human-caused flow from land to the coastal oceans in rivers is about 50 TgN/year, additional to the estimated natural flux of 30 TgN/year (Galloway et al., 2008). Many of the sources of additional reactive nitrogen share root causes with changes in the carbon cycle - such as increased use of fossil fuels and expansion and intensification of global agriculture - so N deposition, rising CO₂ concentrations and warming are increasing together at global scales (Steffen et al., 2011). This large addition of reactive N into the biosphere can reinforce global warming (e.g., though emission of N₂O associated with fertilizer manufacture and application), or diminish warming (e.g., by increasing C storage in many terrestrial ecosystems). The balance of these effects is difficult to estimate (Erisman et al., 2011; WG1 chapter 6; Hanson et al., 2005). Regional trends in N fluxes differ substantially: N fertilizer use and N deposition are stable or declining in some regions, such as Western Europe; but
N deposition and its impacts on biodiversity and ecosystem functioning are projected to increase substantially over
the next several decades in other regions, especially in regions dominated by rapidly-growing emerging economies
(Galloway et al., 2008).

Experiments and observations, most of which are in temperate and boreal Europe and North America, show a
consistent pattern of increase in the dominance of a few nitrogen-loving plant species and loss of overall plant
species richness at N deposition loads exceeding between 5 and 20 kgN/ha/year (Bobbink et al., 2010; but see
Stevens et al., 2010a). N deposition is currently above these limits in much of Europe, eastern North America, and
Southern Asia (Galloway et al., 2008), including in many protected areas (Bleeker et al., 2011). Experiments
examining the interactive impacts of N deposition and elevated CO2 or climate change on natural systems have
increased in number since the AR4, but remain few (Rustad, 2008). Recent studies show that interactions between
nitrogen and other global change factors are widespread, strong and complex (Thompson et al., 2008; Langley and
Megenigal, 2010; Gaudnik et al., 2011; Hoover et al., 2012; Eisenhauer et al., 2012; but see Zavaleta et al., 2003a
for evidence of additive effects). For example, combinations of treatments mimicking future N deposition, elevated
CO2 concentrations and warming had larger negative impacts on pollinator populations than could be predicted from
individual treatments (Hoover et al., 2012). Analyses using the multi-factor biodiversity change model GLOBI03
suggest that N deposition will continue to be a significant contributing factor to terrestrial biodiversity loss in the
first third of the century but will be a less important factor than climate change in this period, and a much smaller
driver than habitat loss due expansion of agricultural lands (Alkemade et al., 2009). Models that explicitly take into
account interactive effects of climate change and N deposition on plant communities project that N deposition
impacts will continue to be important, but climate change effects will begin to dominate other factors by the middle
of the 21st century (Belyazid et al., 2011).

The impacts of N deposition are often first manifested in freshwater ecosystems, since they collect and concentrate
the excess N (and phosphorus) from the land, as well as from sewage and industrial effluents. Primary production in
freshwater ecosystems can be either N and P limited or both (Elser et al., 2007), but the biodiversity and capacity of
freshwater ecosystems to deliver high quality water, recreational amenity and fisheries services is severely reduced
by the addition of nutrients beyond their capacity to process them. Excessive loading of N and P is widespread in the
lakes of the northern hemisphere (Bergstrom and Jansson, 2006). The observed symptoms include a shift from
nitrogen limitation of phytoplankton in lakes to phosphorus limitation (Elser et al., 2009).

Since the AR4 report there has been an increase in the number of studies using models, observations and
experiments to understand and predict the interactive effects of N deposition, climate change and CO2 on ecosystem
function. Broadly, this research shows that ecosystem function is mediated by complex interactions between these
factors, such that many ecosystem responses remain difficult to understand and predict (Norby and Zak, 2011;
Churkina et al., 2010). For example, experiments, observations and models suggest that the observed increase in
productivity and carbon storage in forests in many parts of the world is due to combinations of N deposition, climate
change, fertilization effects of rising CO2, and forest management (Magnani et al., 2007; Huang et al., 2007; Pan et
al., 2009; Churkina et al., 2010; Bontemps et al., 2011; Bellassen et al., 2011; de Vries and Posch, 2011; Eastaugh
et al., 2011; Norby and Zak, 2011; Shanin et al., 2011; Lu et al., 2012). Despite difficulties in quantifying the
relative contributions of these factors and their interactions, it appears that the effects of N deposition and rising CO2
have generally dominated to date in much of the Northern hemisphere, however, the direct effects of rising
temperature and changes in precipitation are may exceed N and CO2 as key drivers of ecosystem primary
productivity in a few decades time. On the other hand, experiments in grasslands show that plant productivity is
increased more by N addition (within the projected range for this century) than by elevated CO2, also within its
projected range; and that N effects increase with increasing precipitation (Lee et al., 2010). In contrast to forests and
temperate grasslands, N deposition and warming can have negative effects on productivity in other terrestrial
ecosystems, such as moss-dominated ecosystems (Limpens et al., 2011). The interactions between N deposition and
climate change remain difficult to understand and predict (Ma et al., 2011; Menge and Field, 2007), in part due to
shifts in plant species composition (Langley and Megenigal, 2010) and the complex dynamics of coupled C, N and
P cycles (Menge and Field, 2007; Niboyet et al., 2011).
4.2.4.3. Tropospheric Ozone

The concentration of ozone in the troposphere (the part of the atmosphere adjacent to terrestrial ecosystems) has been rising over the past 160 years (Horowitz, 2006; Oltmans et al., 2006; Cooper et al., 2010). This is due to increasing anthropogenic emissions of gases which react in the atmosphere to form ozone (Denman et al., 2007), but is also partly due to the increased mixing of stratospheric ozone into the troposphere as a result of climate change (Hegglin and Shepherd, 2009). The key ozone-precursor gases are volatile organic compounds (VOC) and oxides of nitrogen (NOx). Intercontinental transport of these precursors is contributing to rising global background ozone concentrations, including in regions where local emissions are decreasing (Dentener et al., 2010).

Negative effects of the current levels of ozone have been widely documented (Mills et al., 2011). A meta-analysis of over 300 articles addressing the effect of ozone on tree growth (Wittig et al., 2009) - largely focussed on northern-hemisphere temperate and boreal species - concluded that current levels of tropospheric ozone are suppressing growth by 7% relative to pre-industrial levels. Modelling studies extrapolating measured dose-response functions suggest a 5 to 30% contemporary reduction in Net Primary Productivity (NPP) worldwide (Sitch et al., 2007) and 1-16% in temperate forests (Ainsworth et al., 2012).

The mechanisms by which ozone affects plant growth are now better known (Hayes et al., 2007; Ainsworth et al., 2012). Chronic exposure to ozone at levels above about 40 ppb generally reduces stomatal conductance and impairs the activity of photosynthetic enzymes (The Royal Society, 2008) although some evidence also suggests ozone exposure may increase stomatal conductance (Wilkinson and Davies, 2010). For the species studied, carbon assimilation rates are generally reduced and respiration increases, leaf area is reduced and leaf senescence accelerated - all leading to a reduction in NPP and biomass accumulation. Gymnosperms are less sensitive than angiosperms. In a modeling study, lower stomatal conductance due to ozone exposure increased river runoff by reducing the loss of soil moisture through transpiration (Huntingford et al., 2011), although studies that measured runoff are contradictory on this issue (Wittig et al., 2007; McLaughlin et al., 2007; Mills et al., 2009; Huntingford et al., 2011).

A modeling study (Sitch et al., 2007) suggests that the negative effects of rising O$_3$ on plant productivity could offset 17% of the projected increase in global carbon storage due to increasing CO$_2$ concentrations, but the possible interactive effects between CO$_2$ and O$_3$ are poorly understood (The Royal Society, 2008). Reduced stomatal conductance widely observed under elevated CO$_2$ should help protect plants from ozone damage. Some chamber experiments (Bernacchi et al., 2006) and model studies (Klingberg et al., 2011) suggest this to be the case. The one plot-scale FACE study of CO$_2$ and O$_3$ interactions in a temperate forest (Karnosky et al., 2005; Hofmockel et al., 2011) suggests that the effects of O$_3$ and CO$_2$ are not independent and may partly compensate for one another.

There is genotypic variation in sensitivity to O$_3$ (Ainsworth et al., 2012). Other than changing cultivars or species, there is not believed to be much scope for management actions promoting adaptation to higher levels of O$_3$ (Teixiera et al., 2011; Wilkinson and Davies, 2010). Research into developing ozone resistant varieties and/or chemical protectants against damage may provide management options in the future (Ainsworth et al., 2012, Wilkinson and Davies, 2010).

4.2.4.4. Rising CO$_2$

The effects of rising CO$_2$ on the global carbon cycle are discussed in chapter 6 of the WG1 report [Box 6-3, in the SOD]. The discussion here is focused on impacts on terrestrial ecosystems and inland water systems.

Since AR4, investigation of elevated CO$_2$ effects on plants and ecosystems has focused mainly on Free Air CO$_2$ Enrichment (FACE) techniques, which have now been in use for approximately 20 years (Leakey et al., 2009) and are considered more realistic than earlier approaches using enclosed chambers because plant community and atmospheric interactions and below-ground conditions are more like those of natural systems. C3 plant species, which includes nearly all tree species, show a general increase in photosynthesis under elevated CO$_2$, although the response varies between species. Although FACE experiments indicate lower responses of crop yields to elevated
CO₂ than chamber studies, FACE experiments initially suggested greater biomass production of trees (Leakey et al., 2009). But newer results from older stands show that different system respond differently, and only at the Duke forest, a loblolly pine (Pinus taeda) stand in North Carolina has biomass accumulation been enhanced several years after canopy closure on rather nutrient poor soils (McCarthy et al., 2010). At most other tree FACE experiments, most of the additional carbon gain has been invested into below-ground pools with short residence time, such as fine roots (Körner et al., 2005; Finzi et al., 2007), and the initial positive response of NPP at the Oak Ridge experiment with sweetgum (Liquidambar styraciflua) has disappeared due to progressive nitrogen limitation with stand development (Norby et al., 2010). Acclimation (“down-regulation”) occurs under long-term exposure, leading to cessation of effects in some studies (Norby and Zak, 2011) but with on-going stimulation of photosynthetic carbon assimilation due to elevated CO₂ persists at a level of 19 to 46% for approximately 600 ppm CO₂ relative to 370 ppm reported elsewhere (Leakey et al., 2009). Kongstad et al. (2012) did not find such increases for a temperate heathland. Paleo records over the Late Quaternary also indicate effects of CO₂ at the ecosystem scale across the globe (Prentice and Harrison, 2009). Generally, nitrogen use efficiency and dark respiration are also both increased under higher CO₂ (Leakey et al., 2009), but in some tree FACE experiments, productivity increases as a result of enhanced CO₂ were sustained by increased nitrogen uptake rather than increased nitrogen use efficiency (Finzi et al., 2007). Transpiration is decreased due to reduced opening of stomatal apertures which does not acclimate to higher CO₂ in the medium term, i.e. the scale of several years leading to greater water use efficiency (Leuzinger and Körner, 2010; Leakey et al., 2009; De Kauwe et al., 2013) – this is corroborated by additional evidence from studies of stable carbon isotopes (Barbosa et al., 2010; Koehler et al., 2010; Silva et al., 2010; Maseyk et al., 2011). C4 plant species, which includes most tropical grasses and some crops, undergo photosynthesis via a different biochemical pathway which is not directly affected by elevated CO₂. However, CO₂ rise generally increases water use efficiency of C4 plants which can exert indirect effects on growth (Leakey et al., 2009). Experiments have also shown that elevated CO₂ has potential to stimulate root growth (e.g. Finzi et al., 2007; McMurtrie et al., 2012), mineralisation of soil organic matter (Drake et al., 2011; Carney et al., 2007, but see Finzi et al., 2007) and soil microbial communities (Carney et al., 2007), although Norby and Zak (2011) suggests that elevated CO₂ effects on the microbial community structure are minor. Impacts of CO₂ physiological forcing may extend beyond the organisms themselves – increased water use efficiency may significantly influence hydrological impacts (see Box CC-VW).

Elevated CO₂ can assist in maintaining resilience of individual species against other stresses, such as high temperature, drought and ozone, as ozone exposure decreases with lower stomatal conductance (Sitch et al., 2007). It is also suggested that faster growth rates under higher CO₂ can allow woody plants to become tall enough to withstand ground fires, and hence alter the fire/vegetation regime in rangelands (Bond and Midgley, 2001; Scheiter and Higgins, 2009) It is also suggested that differential species responses to elevated CO₂ are altering competition (Dawes et al., 2011), for example, increasing the likelihood of faster-growing species such as lianas out-competing slower-growing species such as trees (Potvin et al., 2007; Lewis et al., 2009a).

The interactive effects of elevated CO₂ and other global changes (such as climate change, nitrogen deposition and biodiversity loss) on ecosystem function are extremely complex and as yet poorly predictable (Potvin et al., 2007). For example, in one ten-year temperate grassland experiment in Minnesota, elevated CO₂ halved the loss of species richness due to nitrogen addition (Reich, 2009), whereas no such change was reported for a similar experiment in annual grassland on serpentine soils in California (Zavaleta et al., 2003a) or an alpine grassland in France (Bloor et al., 2010) or a Danish heathland ecosystem (Kongstad et al., 2012). Some of the mechanisms of community and production changes under multi-factor forcing may be mediated through changes in plant litter quality, although in the temperate grassland experiment described above the changes in litter decomposition rates were less marked than the changes in the chemical composition of tissues (Knops et al., 2007). Hyvönen et al. (2007) review the combined effects of elevated CO₂, temperature, nitrogen deposition and management on carbon sequestration in temperate and boreal forests and conclude that findings extrapolated from experiments that only consider one of these factors in isolation can be misleading. Since FACE experiments either examine CO₂ effects in isolation, or in conjunction with a very limited set of imposed climate changes such as elevated temperature, the extent to which CO₂ responses are affected by a range of temperature changes or other climatic or environmental factors remains poorly studied. A further, broader analysis supports this, and further suggests that the magnitude of CO₂ effects decreases as the scale of study increases (Leuzinger et al., 2011).
Inclusion of CO₂ effects can be a first-order influence on model projections of ecosystem and hydrological responses to anthropogenic climate change. For example, most CMIP3 climate models project a drying, warming trend in Amazonia, and this leads to widespread forest loss being simulated under most climate projections in a model that ignores CO₂ effects (Salazar et al., 2007). However, when CO₂ effects are included, the same model projects forest loss under fewer climate projections (Lapola et al., 2009). The current generation of dynamic global vegetation models (DGVMs; Sitch et al., 2008), which includes those used within the CMIP5 Earth System Models, uses formulations based on experimental work that pre-dates FACE experiments (e.g., Farquhar et al., 1980; Collatz et al., 1992). However, the DGVMs simulate present-day global carbon budgets that agree with those inferred from observations such as the atmospheric CO₂ record (Sitch et al., 2008).

The effect of CO₂ on plant physiology independent of its role as a greenhouse gas has implications for the definition of “dangerous climate change” in terms of levels of global mean temperature rise, and for the use of “CO₂ equivalent” metrics for comparing different greenhouse gases. Aside from uncertainties in the response to a given change in CO₂ concentration, the relative importance of radiative forcing and “CO₂ physiological forcing” depend on uncertainties in the response of global climate to radiative forcing and also on the relative proportions of CO₂ and other greenhouse gases (Betts et al., 2012). The impacts of, for example, a 2°C rise in global mean temperature may have a very different net impact on ecosystems depending on the change in CO₂ concentration accompanying this rise. A high climate sensitivity and/or a higher proportion of non-CO₂ GHGs would imply a relatively low CO₂ rise at 2°C global warming, so the offsetting effects of CO₂ fertilization and increased water use efficiency would be smaller than for low climate sensitivity and/or a lower proportion of non-CO₂ GHGs. Similarly, inertia in the climate system means that warming would be expected to continue for many decades after stabilisation of GHG concentrations, so the relative contribution of CO₂ physiological effects would diminish over time. Comparison of the effects of CO₂ with other GHGs in terms of the current radiative forcing-based metrics may therefore not be representative of the relative effects of different GHGs on ecosystems and hydrology (Huntingford et al., 2011).

There remain still important limitations on understanding of CO₂ responses. Large-scale FACE experiments have still only been conducted for up to two locations for any particular ecosystem type, and these are in temperate and boreal regions (e.g., Hickler et al., 2008); there is currently no tropical FACE experiment. The scale of controlled experiments is limited to approximately 100 m², and extrapolation to larger scales may not be appropriate as other constraints may be important, for example large-scale atmospheric feedbacks (Körner et al., 2007). Similarly, the limited size of FACE experiments means that influences on catchment-scale hydrology have not yet been studied experimentally. The majority of FACE experiments examine CO₂ concentrations of approximately 550 ppmv, which has been projected to be reached by the end of the 21st Century under a number of scenarios including RCP6.0 (van Vuuren et al., 2011), whereas higher concentrations are projected under higher emissions scenarios; RCP8.5 reaches a CO₂ concentration of approximately 900 ppmv by 2100 (van Vuuren et al., 2011). However, the basic physiology of photosynthesis suggest that CO₂ effects saturate at levels of approx. 700 ppmv (Long et al., 2004). Moreover, FACE experiments impose a sudden increase of CO₂ concentration as opposed to the continuous rise measured in the present-day atmosphere. The extent to which photosynthesis and nitrogen and water use efficiency continue to increase with ongoing CO₂ rise therefore remains poorly-constrained. Overall, there is medium confidence (much evidence, medium agreement) that increases in CO₂ up to about 600 ppm will continue to enhance photosynthesis and plant water-use efficiency, but at a diminishing rate.

4.2.4.5. Diffuse Versus Direct Radiation

The quantity and size distribution of aerosols in the atmosphere alters both the amount of solar radiation reaching the Earth’s surface and its distribution between direct and diffuse radiation. There are observed trends in diffuse and direct radiation in many parts of the world, usually in the direction of overall ‘dimming’. Locally, direct radiation is reduced by up to 30 W m⁻², with an accompanying increase in diffuse radiation of up to 20 W m⁻² (Kvalevåg and Myhre, 2007). The global mean direct and diffuse radiation changes due to aerosols are -3.3 and +0.9 W m⁻², respectively (Kvalevåg and Myhre, 2007). For a constant total radiation, an increased fraction received as diffuse radiation theoretically increases net photosynthesis, (Knohl and Baldocchi, 2008; Kanniah et al., 2012) but potentially could alter the hydrological balance and soil temperature. In a global model which included this process, an increase in diffuse fraction of solar radiation due to volcanic and anthropogenic aerosols and cloud cover was...
simulated to lead to approximately a 25% increase in the strength of the global land carbon sink between 1960 and 1999; but under a scenario of climate change and decreased anthropogenic aerosol concentration, this enhancement declined to near zero by the end of the 21st Century (Mercado et al., 2009). The influence of form of radiation on plant growth and the land carbon budget is a potentially important unintended consequence of solar radiation management schemes that involve the injection of aerosols into the stratosphere to reduce radiant forcing (Boucher et al., in preparation) but is presently insufficiently researched for adequate assessment.

4.2.4.6. Invasive and Alien Species

Since the IPCC AR4, the number of observations of the spread and establishment of alien species attributed to climate change has increased for several taxa (e.g. Walther et al., 2009) and for particular areas, including mountain tops and polar regions (McDougall et al., 2011; Chown et al., 2012). The aggressive expansion of plant and animal species beyond their historical range is having increasingly negative impacts on ecosystem services and biodiversity (high confidence; Brook, 2008; Burton et al., 2010; McGeoch et al., 2010; Simberloff et al., 2013). Climate change will exacerbate some invasion impacts and ameliorate others (Peterson et al., 2008; Bradley et al., 2009; Britton et al., 2010). The spread of alien species has several causes, including habitats made favorable by climate change (Walther et al., 2009), deliberate species transfer and accidental transfer due to increased global movement of goods.

In most cases climate change increases the likelihood of the establishment, growth, spread and survival of invasive species populations (Dukes et al., 2009; Walther et al., 2009; Bradley et al., 2010; Huang et al., 2011; Chown et al., 2012). However, divergent alien species responses and local consequences are expected (e.g. Rahel and Olden, 2008; Frelich et al., 2012; Haider et al., 2012; West et al., 2012). Invasive species compared to native species might have traits that favor their survival, reproduction and adaptation under changing climates; invasive plants in particular tend to have faster growth rates and are particularly favored when resources are not limited (medium to high confidence; Buswell et al., 2011; Davidson et al., 2011; Zerebecki and Sorte, 2011; van Kleunen et al., 2010; Willis et al., 2010a; Haider et al., 2012; Matzek, 2012). Invasive plants tend to be more drought tolerant (Crous et al., 2012; Matzek, 2012; Perry et al., 2012), and on average have higher overall metabolic rates, foliar nitrogen concentrations and photosynthetic rates than their native counterparts (Leishman et al., 2007).

Extreme climate events may provide opportunities for invasion by generating disturbances and resource pulses (Diez et al., 2012). Current warming has already enabled many alien species, including plant, vertebrate, invertebrate and single cell taxa, to extend their distributions into new areas (high confidence for plants and insects; Walther et al., 2009; Smith et al., 2012). Population declines and range contractions are however predicted for some invasive species in parts of their ranges (Bradley et al., 2009; Bertelsmeier et al., 2012; Sobek-Swant et al., 2012; Taylor et al., 2012). The expansion of invasive species in some areas and contraction in others will contribute to community re-organisation and the formation of novel ecosystems and interactions in both terrestrial and freshwater habitats (high confidence; e.g., Kiesecker, 2011; Britton et al., 2010; Martinez, 2012; see also chapter 4.3.2.5.). For example, invasive grasses will probably be favored over native ones with increasing temperatures (Parker-Allie et al., 2009; Chuine et al., 2012; Sandel and Dangremond, 2012).

Benefits to biodiversity and society as a result of interactive effects of climate change and invasive species will be experienced in individual cases, such as increases in resources available to some threatened species (Caldow et al., 2007), forest structural recovery (Bolte and Degen, 2010) and available biomass for timber and fuel (van Wilgen and Richardson, 2012). The effect of invasions on net changes in carbon stocks are situation specific, and may be either positive or negative (Williams et al., 2007a). Rising CO₂ levels will increase the growth rates of most invasive plant species (Mainka and Howard, 2010), but the effectiveness of invasive species management for sequestering carbon is uncertain and context specific (Peltzer et al., 2010). Longer term, indirect effects of invasive species will be more important than direct, short-term effects, as a result of changes in soil carbon stocks and tree community composition (low-medium confidence; Peltzer et al., 2010).

Threat syndromes are formed by synergistic interactions between climate change and invasive species, along with landscape change, habitat disturbance and human-facilitated breakdown of dispersal barriers (Brook, 2008; Angeler and Goedkoop, 2010; Bradley et al., 2010; Winder et al., 2011; Cahill et al., 2013). Climate change and invasive
species change the risk and properties of fire, and the interaction is being reported more frequently as a direct result of higher temperatures and increased invasive plant biomass (high confidence; Abatzoglou and Kolden, 2011). In freshwater systems, alien species establishment and survival, species interactions and disease virulence will change as a result of changes in frequency of high-flow events, increasing water temperature, water properties and water demand (medium confidence; Schnitzler et al., 2007; Rahel and Olden, 2008; Britton et al., 2010).

A range of climate change-related variables (extreme events, changes in precipitation, temperature and CO$_2$) will continue to exacerbate the establishment and spread of pests, vectors and pathogens and negatively impact on crop production (medium confidence; Robinet and Roques, 2010; Clements and Ditommaso, 2011). Warming has led to the spread of many invasive insect species such as the mountain pine bark beetle and resulted in forest destruction (high confidence; Kurz et al., 2008). The interactions between crop growth, climate change and pest dynamics are however complex to predict (West et al., 2012). Management strategies may become less effective as a consequence of decoupling of biocontrol relationships and less effective mechanical control as biomass of invasive species increases (low to medium confidence; Hellmann et al., 2008).

4.3. Vulnerability of Terrestrial and Freshwater Ecosystems to Climate Changes

Vulnerability of terrestrial ecosystems to climate change is defined as the combination of three things: the degree to which their climatic environment changes relative to conditions under which the ecosystem evolved; the sensitivity of ecosystem processes to the elements of climate undergoing change; and the degree to which the system (including its coupled social elements) can maintain its structure, composition and function in the presence of such change, either by tolerating the change or adapting to it. See WGII chapter 19 for a fuller discussion on vulnerability concepts.

4.3.1. The Importance of Changes in Disturbance Regime

The species composition at a given location is determined by three factors: the physiological tolerance of the various species that have access to the site in relation to the range of conditions experienced there; interactions with other species, including competitors, symbionts, predators, prey and pathogens; and the ability of species to reach the location in evolutionary or recent times. Occasional disturbances that relieve competition and create opportunities for establishment and success of less-dominant species are important in maintaining diversity. The average environmental conditions are often less of a determinant of species ranges than the extremes, such as the occurrence of exceptionally cold or hot days or droughts exceeding a certain duration, which are often defined as ‘disturbances’. The change in probability of such extremes is typically disproportionately larger than the change in the mean (see IPCC, 2012, but also Diffenbaugh et al., 2005). Fire, floods and strong winds are all examples of biodiversity-sustaining events, provided that their frequency and intensity does not deviate greatly from the regime that prevailed at the location in evolutionary time; outside this range they have adapted to (Connell, 1978).

It is thought that widespread compositional shifts resulting from climate change in interaction with other factors will often be associated with changes in the disturbance regime, rather than physiological tolerance (Thonicke et al., 2008). A ‘disturbance regime’ refers to the totality of different types of disturbance events in a system, each characterized by their probability-intensity function and other relevant attributes, such as their seasonal distribution. A corollary is that disturbance-related change is likely to abrupt rather than gradual.

Change in the fire disturbance regime, in particular, is emerging as a key, mechanism of terrestrial ecosystem change and an early indicator of change that has been detected and in some cases attributed to climate change (Westerling et al., 2006; Littell et al., 2009; Westerling et al., 2011; Moritz et al., 2012).
4.3.2. Evidence of Change in Ecosystems

This section highlights key indices of change in terrestrial and freshwater ecosystems over the recent past. It assesses the degree of confidence that change can be detected and the extent to which changes can be attributed to changes in climate (Figure 4-5). Confidence in detection of change is considered to be very high when there is high agreement between many independent studies, species, ecosystems or regions and where there is robust evidence that the changes over time are statistically significant (see WGII, Chapter 18; Mastrandrea et al., 2010). Note that a slightly different definition than Chapter 18 for detection is used, because detection here is based solely on the presence of a temporal trend and does not attempt to distinguish natural from climate related variation. Confidence in attribution to climate change is very high when three criteria are satisfied: changes correspond to a sound mechanistic understanding of responses to climate change; time series of observations are sufficiently long to detect trends correlated with climate change; and confounding factors can be accounted for or are of limited importance. We have provided estimated levels of confidence even in cases where the capacity for detection or attribution capacity is low or very low, because changes in these ecosystem properties or processes could have large impacts on biodiversity or ecosystem services at regional to global scales. It is also important to account for the spatial distribution of climate change trends: for example, warming trends are very high in the Arctic and small or negative in parts of South America (Burrows et al., 2011). Statements of confidence for detection and attribution are given without references, as detailed traceability is provided in the section 4.3.2, which follows this introduction.

Some of the subsections referred to include assessments of model-based projections of future change, but this analysis of detection and attribution does not cover projected changes in ecosystems. The absence of observed changes does not preclude high levels of confidence in projections of change in the future because 21st century climate change is projected to substantially exceed changes experienced over the last century, and because ecosystem responses to climate change may be non-linear and may appear after considerable time lags.

[INSERT FIGURE 4-5 HERE]

Figure 4-5: Confidence in detection of change and attribution of observed responses of terrestrial ecosystems to climate change. Confidence levels have are based on expert judgment of the available literature following the IPCC uncertainty guidance (Mastrandrea et al., 2010), and detection and attribution criteria outlined in Chapter 18. The points in the figure represent global and cross-taxon assessments; the positioning may be different for specific taxa or regions.]

Phenology (see 4.3.2.1) - Since the AR4 report there has been a significant increase in the spatial, temporal and taxonomic coverage through ground-based phenological observation networks, data mining and remote sensing. These observations show that phenology, i.e., the timing of key life history events such as flowering in plants or nesting for birds, has shifted for many but not all species over the last several decades (high confidence due to robust evidence but only moderate agreement when examined across all species). Attribution of temporal shifts in phenology to qualitative expectations of responses due to global warming has given way to more refined analyses and models that account for regional differences in warming trends, urban heat island effects, confounding effects of other global change drivers, and non-linear responses of phenology to warming. These advances notwithstanding, attribution can only be made with high confidence because there are a wide range of confounding factors, a substantial fraction of species that have not responded in a predictable manner and discrepancies between phenological shifts in experiments vs. observations.

Primary Productivity (see 4.3.2.2) and Biomass and C Stocks (see 4.3.2.3) - Evidence has accumulated from a wide range of sources including eddy flux towers, inversion of atmospheric CO₂ concentrations, remote sensing and models that, once the effects of deforestation are accounted for, terrestrial ecosystems are currently net sinks for carbon over much of the Northern hemisphere and parts of the Southern hemisphere (high confidence). Measurements of increased tree growth over the last several decades, a large sink for carbon, are consistent with this but confounding factors such as N deposition, afforestation and land management make attribution of these trends to climate change difficult (low confidence). Most studies speculate that rising CO₂ concentrations are contributing to this trend through stimulation of photosynthesis, but there is no clear, consistent signal of a climate change contribution.
Evapotranspiration (see 4.3.2.4) - Evapotranspiration is now being monitored at a variety of scales ranging from individual plants to large regions. A variety of studies suggest that evapotranspiration increased over the period 1970-1990, but there is substantial uncertainty in this estimate due to low agreement across studies leading to low confidence in detection of a trend. The trend of rising evapotranspiration from 1970-1990 has been attributed to global warming, and the absence of a continued trend in from the 1990s onward has been attributed to increased water stress. However, there is low confidence in these attributions because of the large number of confounding factors that influence transpiration including N deposition, land management and irrigation.

Species Distributions (see 4.3.2.5) - Depending on the species and species group, there are varying degrees of confidence in the detection of species range shifts and that these range shifts can be attributed to climate change. The number of species studied has considerably increased since the AR4 report as have the number of studies examining the mechanisms underlying range shifts. There is a wide range of evidence from models, field studies, and quantitative comparisons between observed and expected range shifts that some species and species groups, especially arthropods, have shifted their ranges significant distances (10s of km or more) over the last several decades (high confidence). Range shifts — that must be distinguished from shifts in community composition — in other key species groups such as plants have been much more difficult to detect, meaning there is only medium confidence in detection in range shifts when examined across all species groups. Range shifts in some species groups, e.g., arthropods, correlate well with warming over the last several decades and can be linked to changes in climate through understanding of mechanisms. But for many other species groups range shifts are more difficult to attribute to changes in climate because the climate signal is small, there are many confounding factors, differences between expected and observed range shifts are large, or variability within or between studies is high. Thus there is only medium confidence in attribution when examined across all species. An importance advance since the AR4 report is the recognition changes in climate over the last several decades have led to range shifts that are frequently not towards the poles or up in altitude due to many factors including interactions between warming, land use and changes in precipitation (high confidence).

Global Species Extinctions (see 4.3.2.5) - There is high confidence that global species extinctions are at or above the highest rates of species extinction in the fossil record, and high confidence that global extinctions have increased in some species groups, e.g., amphibians, over the last several decades. However, across all taxa there is only low confidence that rates of species extinctions have increased over the last several decades. Only a small fraction of observed species extinctions have been attributed to climate change — most have been ascribed to invasive species, overexploitation, habitat loss or habitat modification. As such, there is very low confidence that observed species extinctions can be attributed to recent climate warming given the small fraction of species for which global extinction has been ascribed to climate change and tenuous nature of most attributions. However, in the specific case of Central American amphibians, there is medium evidence that recent climate warming has contributed to extinctions and medium confidence in this attribution. There is low agreement among investigators concerning the importance of climate compared to other factors such as invasive disease and habitat loss.

Invasive and Alien Species (see 4.2.4.6) - There is very high confidence that species invasions are increasing over the last several decades. There is high confidence that recent climate trends have contributed to the establishment, growth, spread and survival of some alien species populations in some regions, but only low confidence that the species invasions have generally been assisted by recent climatic trends because of the overwhelming importance of human facilitated dispersal in mediating invasions.

Tree Mortality (see 4.3.3.1, Box 4-2) - Extensive tree mortality appears to be increasing globally, but there is low confidence in detection due to a lack of temporal and spatial coverage of tree mortality. In particular, the increase in reporting of mortality may reflect greater observation effort rather than a real increase in mortality. In many cases, tree mortality can clearly be linked directly to climate impacts, especially heat and drought. However, attribution to directional climate change over the last several decades is difficult (low confidence) due to lack of time series measurements for a large range of forested systems and the large number of confounding factors mediating tree mortality.

Boreal Forest Regime Shift (see 4.3.3.1.1, Box 4-4) - An increase in tree mortality is observed in many boreal forests, with particularly widespread dieback related to insect infestations in North America. Tree health in boreal
forests varies greatly among regions, which coupled with insufficient temporal coverage means that there is low confidence in the detection of a clear temporal trend in morality at the global scale. Models and paleobotanical data indicate that regression of boreal forest at its southern edge is a predictable response to warming. High magnitudes of climate change over most of boreal forest also facilitate attribution to global warming. In some cases, tree mortality can be linked directly or indirectly (e.g., due to changes in insect and pathogen ranges) to fluctuations in climate. However, the lack of long-term data sets and high spatial heterogeneity means that attribution to climate change is of low confidence.

Amazon Forest Regime Shift (see 4.3.3.1.3, Box 4-3) - Long-term increases in tree turnover (increased mortality and growth) in the humid tropical forests of the Amazon basin have been documented with moderate confidence. In addition, two episodes of extreme drought over the last decade have led to increased mortality of large trees. There is low confidence that this represents a trend towards degradation of humid tropical forest, since structural changes in intact forests have been minor. Observed changes in tree turnover have been ascribed to a variety of mechanisms including rising CO₂ effects on vines, recovery from past disturbance and changing climate; the attribution to these various drivers is difficult due to limited temporal and spatial coverage. Models and experiments lend credence to a contribution of climate to increased tree mortality and growth. There is very low confidence in attribution of observations to climate change because of the short period of intensive monitoring records and because warming trends over the last several decades are weak in this region.

Tundra Regime Shift (see 4.3.3.4) - Field and satellite measurements indicate an increase in shrub growth and permafrost thawing in many areas of the Arctic tundra. Lack of temporal coverage makes it difficult to know if this is a strong directional signal leading to medium confidence in detection of a trend. It is not clear to what extent this presages boreal forest encroachment into tundra, which would have much larger impacts on ecosystem structure and function. Experiments, models and paleontological responses of tundra to past warming all indicate that the increase in shrub coverage and permafrost thawing are predictable responses to climate warming trends. High magnitudes of climate change over the last 50 years for most of the Arctic facilitate attribution. The lack of long-term monitoring data makes it difficult to attribute these changes to climate change with greater than medium confidence.

Cultural Landscapes (see 4.3.3.5) - There is very high confidence that species composition and landscape structure are changing in cultural landscapes such as Satoyama landscapes in Japan or mixed forest, agricultural landscapes in Europe. Models and experiments suggest that climate change should be contributing to these observed changes. The land use and land management signal is so strong in these landscapes, that there is very low confidence that we can attribute these observations to climate change.

Evolutionary and Genetic Adaptation (see 4.4.1.2) - While there are an increasing number of observations of "rapid evolutionary" responses to climate variation, the number of species studied is extremely limited. In addition, it remains difficult to separate phenotypic from genotypic adaptation meaning that there is only low confidence that we are detecting a widespread signal of evolutionary or genetic adaptation to climate. An increasing number of models and experiments illustrate that "rapid evolutionary" responses are possible and even probable response to climate change. In a few cases, strong correlations between climate trends and changes in gene frequencies underlying phenological responses to climate trends suggest that climate change is an important driver of rapid evolution. The limited number of well-studied cases means there is only low confidence in attribution of adaptation as a generalized response to changes in climate over the last several decades.

4.3.2.1. Phenology

A large number of site-specific studies, and more recently satellite imagery, allow monitoring of phenology. The advance in spring events (e.g. breeding, bud burst, breaking hibernation, flowering, migrating,) has been reported for many regions, especially from the northern hemisphere but fewer in the southern hemisphere and for many plant and animal taxa. Since the AR4 many new studies showing phenological adaptations of species as response to climate change (e.g., amphibians: Kusano and Inoue, 2008; birds: Pulido, 2007; Thorup et al., 2007; mammals: Adamik and Kral, 2008; Lane et al., 2012; plants: Cleland et al., 2007; Gordo and Sanz, 2009; plants and animal taxa: Haye et al., 2007; Primack et al., 2009) and meta analyses were published (observed data: Parmesan, 2006; Parmesan, 2007;...
Large-scale changes and variations in vegetation have been monitored by satellite since 1982 (White et al., 2009). The Advanced Very High Resolution Radiometer (AVHRR) quantifies the differences in visible and near-infrared wavebands of outgoing radiation (NDVI) which is related to the quantity of photosynthetically-active vegetation. The times of maximum and minimum rate of change of NDVI provide a metric of the growing season. From 1982 to 2008 the NDVI-defined start of the growing season advanced by 5.4 days at the hemispheric scale, and the end was delayed by 6.6 days (Jeong et al., 2011). Rates of change varied over time and at smaller scales. Changes were found to be strongly related to temperature trends, although changes in precipitation may also be key drivers, especially in more arid regions (McPherson, 2007). Studies with the more recent instrument, Moderate Resolution Imaging Spectrometer (MODIS), and other metrics such as Leaf Area Index (the ratio of leaf area to ground area; Ahl et al., 2006) provide a similar picture at large scales, although results can differ at local scales due to the use of different instruments, indices and trend detection methods. The relatively short duration of satellite observations makes trend detection particularly sensitive to the choice of analysis period. Nevertheless, the general emerging picture of an advancement of the start of the growing season length is consistent with ground-based phenology studies. At a regional scale, a combination of species-level observations, meta-analysis, NDVI measurements and phenology modeling to reveal geographically different advancements in spring phenology of Chinese plant communities in different Ecoregions and Biomes between the 1960s and 2000s (Ma and Zhou, 2012).

Changes in breeding phenology are reported from various regions and different taxa (insects, birds, mammals, amphibians; e.g. Parmesan, 2006; Parmesan, 2007; Post et al., 2008; Primack et al., 2009). Various factors can be attributed to changes on breeding phenology, e.g. food availability, increasing spring temperatures. In the northern hemisphere several studies show advancements of egg laying dates in birds (e.g. Parmesan, 2007: -3.70 days/decade ± 0.7, n= 41 species in a meta-analysis). In mammals, Réale et al. (2003) found for the female North American squirrel (Tamiasciurus hudsonicus) an advancement of 18 days of the mean parturition day in 10 years (1992-2002) in the Yukon area, Canada, coinciding with increasing abundance of white spruce cones, the major food source. For the southern hemisphere the opposite pattern, a significant delay of the mean breeding date by 2.8 to 3.7 days between 1950 and 2004 for two of nine seabirds in the Eastern Antarctic (Barbraud and Weimerskirch, 2006).

Patterns of changes in autumn migration in birds are mostly not consistent (delayed, advanced, no change) across analyzed species and regions and appear to be highly related to non-climatic variables (e.g. Sokolov, 2006; Adamik and Pietruszkova, 2008). Phenological response can differ with migration strategy in birds, such as short distant migrants show greater advancements in spring arrivals than long distant migrants (e.g. Saino et al., 2009; but see Parmesan, 2006 for different patterns). Insects show also a variety of phenological responses to changing temperatures (e.g. reviewed in Robinet and Roques, 2010). Development rates speed up and for polyvoltine species an increase in number of generations attributed to increasing temperatures and a lengthening of the growing seasons due to earlier timing of last winter frost and later timing of first fall frosts are reported (e.g. Menzel et al., 2003; Kunkel et al., 2004; Sunley et al., 2006; Robinet and Roques, 2010).

A variety of environmental drivers could affect behavioral changes in species. Experimental-manipulation shows that higher temperatures lead to significantly earlier egg laying dates of the great tit (Parus major) in 5 out of 6 years (Visser et al., 2009), but rates of seasonal warming may also be important (Schaper et al., 2012). In a temperate region (Massachusetts, USA), declining sizes of migration cohorts or of populations appeared to account for a large amount of the variation in previously documented changes in migration times between 1970 and 2002 (Miller-Rushing et al., 2008). After accounting for changes in migration cohort size, migration distance, climatic variables, and migration date explained portions of the variation in migratory changes over time. Changes in migration phenology cannot only be attributed to changes in temperature changes but also to differing variability of changed feathers during moulting times, food availability at stop-over places and differing health conditions of individual species (Gordo, 2007).

Snow cover and snow depth can also be important factors. Interactions between temperature precipitation determining snowmelt changes, which is reported to lead to earlier flowering and appearances of plants and arthropods in Greenland between 1996 and 2005 (Høye et al., 2007) and earlier flowering in an alpine plant in the
Rocky Mountains, USA between 1975 and 2008 (Lambert et al., 2010; Hülber et al., 2010). Earlier snow melts are reported to decrease floral resources and can hence affect survival of dependent insects in mountain ranges in the USA across 1980, 1985, 1986 and 1989 (Boggs and Inouye, 2012). Mammals show also changes in phenology as effects of climate change induced changes on snowmelt pattern, especially hibernating species, ranging from advancements of emergence dates from hibernation (Ozgul et al., 2010) due to earlier snowmelts or to delayed emergence dates (Lane et al., 2012) due to delayed snowmelts. Delayed emergence from hibernation was also associated with decreased population growth rate (Lane et al., 2012).

Experiments are an important source of information on physiological and phenological responses to changing climate, but some caution should be exercised when translating their results into future projections using models. Wolfovčik et al. (2012), in a large meta-analysis, suggested that warming experiments substantially underpredict advances in the timing of flowering and leafing of plants. They also showed that experimentally-derived temperature sensitivities of species did not match the observations in the wild. However, in contrast, Gunderson et al. (2012) found good agreement between experimental warming and in situ observations. Experiments on wild species may also directly affect results; for example, flipper-tagging of penguins can alter their migratory behavior (Saraux et al., 2011).

Generally, there is a great interspecific variability in phenological responses to changing climatic factors (e.g.: Gunderson et al., 2012). Changes in interspecific interactions, such as predator-prey or interspecific competition for food, stemming from changes in phenological characteristics and breakdown in synchrony between species have been observed. Parmesan (2007) found taxonomic groups to be advancing at significantly different rates, with amphibian breeding date advancing by eight times as much as other groups, birds and butterflies were not significantly different from each other, but both showed spring advancement three times stronger than for herbs and grasses. Since most butterfly species eat herbs and grasses as host plants, this suggests an increasing asynchrony between these two interacting groups. Thackeray et al. (2010) draw similar conclusions. Species unable to adjust their behavior, i.e. advancement of spring events to keep track with changing climates could be negatively affected. This may be particularly important in highly seasonal habitats (Both et al., 2010). The degree, direction and strength of the asynchrony due to changing climatic variables depends on differences in the phenology of the interacting species (van Asch and Visser, 2007). Increasing temperatures may either bring species more into synch or more out of synch, depending on their respective starting positions (Singer and Parmesan, 2010), although evidence is more towards a loss of synchrony (Visser and Both, 2005). Changes in (a)synchrony of interacting species is assumed to effect dynamical features of ecological communities such as trophic cascades, competitive hierarchies, and species coexistence (Nakazawa and Doi, 2012). Ahol et al. (2007) could show that the competition between the resident P. major and the migratory pied flycatcher (Ficedula hypoleuca) is increasing when the onset of breeding of both is becoming closer to each other and the abundances of both species are increasing (data: 1953–2005; South-Western Finland). This pattern resulted in a relative host change to more long-distance hosts. van Asch and Visser (2007) report increasing asynchrony of the winter moth (Operophtera brumata) and its feeding host oak tree (Quercus robur) due to increasing spring temperatures and not changing winter temperatures (based on a descriptive model, oak data: 1975-1999, Netherlands, moth data: 1995-1999; Oosterhout, Warnsborn, Doorwerth, Arnhem; Netherlands). The edible dormouse (Glis glis), a nest predator advanced its hibernation termination significantly with -8 days per decade in the Czech Republic between 1975-2005 due to increasing annual spring air temperatures, leading to increased nest predation in three out of four bird surveyed species (Adamik and Kral, 2008)

Cross taxa observations show high variation in species- and location specific responses to increasing temperatures in both direction and magnitude (e.g. Parmesan, 2007; Primack et al., 2009; Thackeray et al., 2010). Uncertainties and biases can be introduced in research that compares different taxonomic groups or geographic regions by often incomplete or non-overlapping time series, averaged geographical scales, lack of consideration of effects of local climatic variability (e.g. wind speed, climatic conditions at stop-over places) and mostly unknown pressures in winter ranges for migratory species (Hudson and Keatley, 2010). With too short time series long term trends in phenological changes cannot be detected, although responses to annual climate variability can often be characterized. However, some studies now extend over several decades (Cook et al., 2008) or even 100 years (Naef-Daenzer et al., 2012). Given the geographical coverage and length over time over which studies are now available, there is high confidence (robust evidence, high agreement) that phenological shifts are occurring as a response to a warming climate.
4.3.2.2. Primary Productivity

Primary production is the process of plant growth that underpins energy flow through all ecosystems, enabling provisioning services such as crop, forest, livestock and fisheries harvest. It is also fundamental to the regulation of global carbon cycle (see 4.3.2.3. below). Trends in the amount, seasonal timing, variability, location and type of primary production are therefore important indicators of ecosystem function. Well-established theory, experimentation and observation all agree that primary production is directly and indirectly sensitive to almost all aspects of climate change as well as to many of the other changes simultaneously taking place in the world, such as the factors described in section 4.2.4. The complexity of the interactions between driving factors and the frequently non-linear form of the responses to them means that at a given location the net outcome can be an increase in productivity, no change or a decrease.

At continental-to-global scale two main sources of information on primary productivity are available. The first is precise and frequent measurements of atmospheric carbon dioxide, which once the known signal of fossil fuel emissions is removed, show clear patterns in space and time largely related to the primary productivity of the land and oceans. The relative contribution by the land, which consists of a sink term due to increased net productivity plus a source term due to land-use change, can be estimated using isotope measurements, emission databases and models (Canadell et al., 2007). During the decade 2000 to 2009, land net primary productivity at the global scale continued to be enhanced about 5% relative to the postulated pre-industrial level, leading to a land sink of 2.6 ± 0.7 PgC/y (see WG1 chapter 6; Raupach et al., 2008; Le Quere et al., 2009). This net uptake of carbon by the land is highly variable year-to-year, in response to global climate models such as ENSO and major volcanic eruptions (Peylin et al., 2005; Sitch et al., 2008; Mercado et al., 2009). It is currently not possible to state with any degree of confidence that the net uptake of carbon on land has either increased or decreased over the past two decades (Raupach et al., 2008).

It is possible to estimate the net uptake on land at subglobal scales, using inversion modelling techniques and the growing network of precision atmospheric observations. There is broad agreement that the net land uptake in largely-untransformed terrestrial ecosystems is globally distributed, almost equally between forested and non-forested ecosystems, but is offset in the tropics by a large carbon emission flux resulting from land-use change, principally deforestation (Pan et al., 2011). Space-based remote sensing can in principle monitor trends in vegetation cover and leaf area, from which primary productivity can be inferred. A commonly-used metric is the Normalised Difference Vegetation Index (NDVI), for which over 30 years of observations are available, covering most of the global land surface. The observed trends are discussed under the various ecosystem-specific discussions below. In some cases the trends are sufficiently strong and consistent to support a confident statement about the underlying phenomenon, but in many cases they are not. This may mean that no change has occurred, but simply reflect inadequacies in the indicator, method of analysis and length of the record. The AR4 reported a trend of increasing seasonally-accumulated NDVI (“greening”) at high northern latitudes (Fischlin et al., 2007; based on Sitch et al., 2007), but later observations show a lower rate and no geographical uniformity (Goetz et al., 2007). More than 25% of North American forest areas, excluding areas recently disturbed by fire, showed a decline in greenness and no systematic change in growing season length, particularly after 2000. NDVI trend analysis studies in rangelands show varying patterns around the world, with substantial areas of disagreement between studies (Millennium Ecosystem Assessment, 2005b; Bai et al., 2008; Beck et al., 2011a; Fensholt et al., 2012). There is agreement that the Sahel showed widespread NDVI increase between the mid 1980s and about 2000, along with an increase in rainfall, but no consensus on whether the detected signal represents increased productivity by grasses, trees or forbs; and to what degree it reveals land management efforts or responses to climate (Anyamba and Tucker, 2005; Hickler et al., 2005; Prince et al., 2007; Hellden and Tottrup, 2008; Seaquist et al., 2009). In the period 2000 to 2009 no NDVI trend was apparent in the Sahel (Samanta et al., 2011).

Tree rings record changes in tree growth over approximately the past millennium. Ring width and the wood density within a ring indicate tree productivity during the period represented by the ring. Isotopes within the ring reveal
changes in the environment to which the tree was exposed. Variations in tree ring parameters arise from many factors, including temperature, moisture stress, CO₂ fertilization and ozone damage. The most recent rings in most published tree ring chronologies date from before the 1990s (Gedalof and Berg, 2010) so tree ring-based conclusions for more recent dates are based on a smaller body of evidence. Sites which are not sensitive to environmental changes may therefore be under-represented. Direct repeated measurements of tree girth increment in forest monitoring plots (discussed in 4.3.2.3) are an alternate data source for recent decades. Tree ring studies may not be representative of forests in general, as most studies were specifically designed to examine growth in response to environmental changes (Gedalof and Berg, 2010).

Many tree ring records show a tendency of accelerated tree growth during much of the 20th century (Briffa et al., 2008), which often correlates with rising temperature. Direct CO₂ effects, inferred from the ring record once the effects of drought and temperature have been accounted for, have been proposed for approximately 20% of the sites in the International Tree Ring Data Base (Gedalof and Berg, 2010) and studied in detail at some sites (Koutavas, 2008). Since the 1980s a number of tree ring records show a decline in tree growth (Wilson et al., 2007). Several possible causes have been suggested for this, including increasing water stress and ozone damage.

4.3.2.3. Biomass and Carbon Stocks

Forests are routinely monitored for management and research purposes, typically by recording species composition and stem size distribution in hundreds of small plots. Forest carbon stock can be estimated from this and forest area data, usually derived from remote sensing. Repeated forest inventories in principle allow the growth rate and change in carbon stock to be estimated. Forest inventories were designed to track timber volumes; inferring total biomass and ecosystem carbon stocks requires further information and assumptions, which make absolute values less certain, but have a lesser effect on trend detection. Forest inventory systems are well-developed for northern hemisphere temperate and boreal forest (Nabuurs et al., 2010; Ryan et al., 2010; Wang et al., 2010a). Data for tropical and southern hemisphere forests and woodlands also exist (Maniatis et al., 2011; Tomppo et al., 2010) but are typically less available and comprehensive (Romijn et al., 2012). More and better data may become available due to advances in remote sensing (e.g. Baccini et al., 2012) and the REDD+ initiative.

Forests have increased in biomass and carbon stocks over the past half century in Europe (Ciais et al., 2008; Luyssaert et al., 2010) and the USA (Birdsey et al., 2006). Canadian managed forests increased in biomass only slightly over 1990-2008, due to fires and beetle outbreaks (Stinson et al., 2011). Several dozen sites across the moist tropics have been monitored to estimate forest biomass changes. In the Amazon (Malhi et al., 2009a; Phillips et al., 2009) forest biomass has generally increased in recent decades, dropping after a drought in 2005. Globally, mature forests are withdrawing from the atmosphere one fourth of human CO₂ emissions while regrowth forests withdraw an additional 17%. This global terrestrial sink is offset by the losses of forest carbon to the atmosphere through tropical deforestation and land-use (Pan et al., 2011).

The carbon stock in global soils, including litter, peatlands and permafrost, is estimated to be in the order of 3300 PgC (Davidson et al., 2011), which is about ten times more than is contained in forest biomass (Kindermann et al., 2008). The size of this stock represents the balance of inputs from primary productivity and soil respiration. Soil respiration increases with increasing temperature, but the sensitivity remains uncertain (Davidson and Janssens, 2006). An analysis of long-term soil respiration measurements from around the world suggests that soil respiration has increased over the past two decades by an amount of 0.1PgC/y, some of which may be due to increased productivity (Bond-Lamberty and Thomson, 2010). Flooded coastal soils also show increases in respiration, larger than increases in production (Kirwan and Blum, 2011).

4.3.2.4. Evapotranspiration and its Role in the Terrestrial Water Cycle

Transpiration is affected by multiple factors (Luo et al., 2008) including temperature, humidity, soil moisture, solar radiation and CO₂ concentration. Total evapotranspiration (ET: transpiration through plant stomata plus evaporation from the ground surface and leaf surfaces) is monitored with flux towers, but since most records are at most 15 years
Observed Species Range Shifts - The number of studies looking at observed range shifts and the breadth of species examined has greatly increased since the AR4 report. The most important advances since AR4 concern improvements in understanding the relationship between range shifts and changes in climate over the last several decades. The "uphill and poleward" view of species range shifts in response to recent warming (Parmesan and Yohe, 2003; Parmesan, 2006; Fischlin et al., 2007; Chen et al., 2011) is an useful simplification of species response to changing climate. However, responses to warming are conditioned by changes in precipitation, land use, species interactions and many other factors. Investigations of the mechanisms underlying observed range shifts show that climate signals can often be detected, but the impacts of and interactions between changing temperature, precipitation and land use often result in range shifts that are downhill or away from the poles (Rowe et al., 2010; Crimmins et al., 2011; Hockey et al., 2011; McCain and Colwell, 2011; Rubidge et al., 2011; Pauli et al., 2012;
Tingley et al., 2012; Zhu et al., 2012). There are large differences in the ability of species groups and species within species groups to track changes in climate through range shifts (Angert et al., 2011; Mattila et al., 2011; Chen et al., 2011). For example, butterflies appear to be able track climate better than birds (community shifts - Devictor et al., 2012; but see Chen et al. (2011 for range shifts) while some plants appear to be lagging far behind climate except in mountainous areas (Bertrand et al., 2011; Doxford and Freckleton, 2012; Gottfried et al., 2012; Zhu et al., 2012). There is also growing evidence that responses at the "trailing edge" of species distributions (i.e., local extinction in areas where climate has become unfavorable) are often less pronounced than responses at the "leading edge" (i.e., colonization of areas where climate has become favorable), which may be related to differences in the rates of local extinction vs. colonization processes (Doak and Morris, 2010; Chen et al., 2011; Brommer et al., 2012; Sunday et al., 2012) and difficulties in detecting local extinction (Thomas et al., 2006).

Rates of recent climate change have varied greatly across the globe ranging from rapid warming to cooling (Burrows et al., 2011; Dobrowski et al., 2013), so taking into account this spatial variation should enhance the ability to detect climate related range shifts. A recent synthesis of range shifts indicates that terrestrial animal species have moved at rates that correspond better with changes in temperature when climate is measured in the regions where the range shifts were observed (Chen et al., 2011), providing greater confidence in attributing range shifts than from range data alone. Average range shifts across taxa and regions in this study were 17 km poleward and 11 m up in altitude per decade, which are 2 to 3 times greater than previous estimates (compare with Parmesan and Yohe, 2003; Fischlin et al., 2007). However, even this approach is an oversimplification since key climate drivers of species range modifications, e.g., temperature and precipitation, frequently have shifted in divergent geographical directions (Dobrowski et al., 2013). Disentangling these conflicting climate signals can help explain complex responses of species ranges to changes in climate (Tingley et al., 2012). Overall, studies since the AR4 report show that species range changes are complex responses to interactions among climate drivers and between climate and non-climate factors. It is the greater understanding of these interactions, combined with increased geographical scope and breadth of species studied that provides medium confidence that species ranges have responded to recent changes in climate.

**Future Range Shifts** - Projections of future species range shifts since AR4 report have been dominated by studies using ecological niche models (ENMs) that estimate future ranges based on current relationships between climate and species distribution (also called “species distribution models”, or “bioclimatic models” when models depend entirely on climate factors; Peterson et al., 2011). A variety of process-based species distribution models are also being developed and more widely used (Kearney et al., 2009; Dawson et al., 2011; Cheaib et al., 2012). Recent model comparisons suggest that in some instances bioclimatic models predict larger range shifts than process-based models (Morin and Thuiller, 2009; Kearney et al., 2010; Cheaib et al., 2012) although in other cases process-based models predict larger shifts (Buckley et al., 2012). Projected future species shifts are broadly coherent with species responses to climate change in the paleontological record and with observed recent species shifts (see section 4.2.2 and above). There has been some progress in model validation, such as the use of hindcasting of observed recent or paleontological range shifts, but validation still needs to be reinforced using a wide array of data types (Green et al., 2008; Pearman et al., 2008; Nogues-Bravo et al., 2010; Dawson et al., 2011). A major limitation of these models is that the vast majority do not account for realistic species displacement rates (i.e., rates at which species are able to shift their ranges); as such, they only indicate changes in areas of favorable and unfavorable climate from which possible shifts in species distribution can be inferred.

A small, but growing number of new analyses and models developed since the A4 report permit the estimation the ability of a large range of species to track climate change. These projections are based on i) past and future climate velocity, which is a measure of the rate of climate displacement across a landscape and provides an indication of the speed at which an organism would need to move in order to keep pace with the changing climatic conditions (Loarie et al., 2009; Burrows et al., 2011; Chen et al., 2011; Sandel et al., 2011; Schippers et al., 2011; Feeley and Rehm, 2012; Schloss et al., 2012; Dobrowski et al., 2013) and ii) species displacement rates across landscapes with and without habitat fragmentation for very broad range of species (e.g., Stevens et al., 2010b; Nathan et al., 2011; Barbet-Massin et al., 2012; Kappes and Haase, 2012; Meier et al., 2012; Schloss et al., 2012; see additional references in Figure 4-6 legend). Figure 4-6 provides a simplified comparison of these rates as means of illustrating the projected abilities of species to track climate. This simplified analysis is coherent with more realistic projections of climate induced species displacement across landscapes, some of which include additional constraints such as...
habitat fragmentation or competition (Barbet-Massin et al., 2012; Meier et al., 2012; Schloss et al., 2012).

Comparison of these rates indicates whether species are projected to be able to track climate as it changes, i.e., when species displacement capacity exceeds climate velocity. When displacement rates are lower than projected climate velocities this indicates that species may be unable to keep pace with climate change.

Rates of climate change over the 20th and projected for the 21st century are shown in Figure 4-6A. Rates of climate change for global land surfaces are given for IPCC AR5 climate projections for a wide range of greenhouse gas emissions scenarios (i.e., IPCC RCP scenarios, Knutti and Sedlacek, 2012). Rates of global warming for land surfaces have averaged ca. 0.03 °C/yr since 1980. At the low end of projected future rates of warming, rates decrease over time reaching near zero by the end of the century (RCP 2.6). At the high end, projected rates increase over time and culminate above 0.06 °C/yr by the end of the century (RCP 8.5), and can exceed 0.08 °C/yr at the upper bound for this scenario. Background colors in Figure 4-6A indicate the degree to which species may be able or unable to track suitable climates based on corresponding climate velocities (Figure 4-6B) and rates of displacement for a broad range of species groups (Figure 4-6C). Blue background color indicates that nearly all species will be able to track climate change in all areas; orange indicates that some species will be unable to track climate especially in flat areas; and red indicates that many species groups will be unable to track climate except in mountainous regions.

Climate velocity depends on regional rates of climate change and the degree of altitudinal relief (Figure 4-6B, Loarie et al., 2009; Dobrowski et al., 2013). For example, climate velocity is low in mountainous areas, because climatic gradients are strong, and in areas where climate change is low (Figure 4-6B, leftmost axis). Climate velocity is high (Figure 4-6B, rightmost axis) in flat areas (e.g., Amazon basin) or areas where projected climate change is high (e.g., Arctic regions). In flat areas, climate velocity may exceed 8 km/yr for the highest rates of projected climate change (SRES A2 or RCP 8.5).

Rates of displacement vary greatly within and among species groups (Figure 4-6C). Some species groups, notably plants and especially trees, have very low displacement capacity. Other species groups such as butterflies, birds and large vertebrates generally have a very high capacity to disperse across landscapes. Small-ranged species across several groups (amphibians, mammals and birds) are markedly absent from areas with high climate velocities in the past, suggesting a low dispersal capacity (Sandel et al., 2011). Current and future rates of climate change correspond to climate velocities that exceed rates of displacement for several species groups for most climate change scenarios. This is particularly true for mid- and late-successional trees that have maximum displacement rates that are on the order of 10s to a few 100s of m/yr. Overall, many plant species are foreseen to be able to track climates only in areas of mountainous areas, though there is uncertainty concerning the potential role of long-distance dispersal (Pearson, 2006). Primates have substantially higher dispersal capacity than trees; however, a large fraction of primates are found in regions with very high climate velocities (e.g., the Amazon basin), thereby putting them at high risk of being unable to track climates even at relatively low rates of climate change. On a global average, many rodents, as well as some carnivores and freshwater mollusks are projected to be beyond their limits to track climate at high rates of climate change (i.e., >0.06°C/yr). These projected differences in species ability to keep pace with future climate change are broadly coherent with observations of species ability or inability to track recent global warming (see above).

Humans can enhance species displacement by intentionally or unintentionally dispersing individuals or propagules (arrow on right of Figure 4-6C pointing upwards). For example, many economically important tree species may be deliberately moved on large scales as part of climate adaptation strategies in forestry in some regions (Lindner et al., 2010). Human activities can also substantially reduce displacement rates. For example, habitat fragmentation typically reduces displacement rates, sometimes substantially (Eycott et al., 2012; Hodgson et al., 2012; Meier et al., 2012; Schloss et al., 2012; arrow on right pointing downwards). The degree to which habitat fragmentation slows displacement depends on many factors including the spatial pattern of fragmentation and corridors, maximum dispersal distances, population dynamics and the suitability of modified habitats as stepping-stones (Pearson and Dawson, 2003).

Species that cannot move fast enough to keep pace with the rate of climate change will lose favorable climate space and potentially experience large range contractions, whereas displacement that keeps pace with climate change may
allow some species to increase their range size (Menéndez et al., 2008; Pateman et al., 2012). Mountains will
provide an extremely important climate refuge for many species because the rate of displacement required to track
climate is low (Figure 4-6B, Colwell et al., 2008; Engler et al., 2011; Gottfried et al., 2012; Pauli et al., 2012; but
see Dullinger et al., 2012). An important exception is for species that are already at the tops of mountains (or near
other boundaries) – they are among the most threatened by climate change because they cannot move upwards
(Thuiller et al., 2005; ; Raxworthy et al., 2008; Engler et al., 2011; Sauer et al., 2011). The consequences of losing
favorable climate space are not yet well understood. A variety of lines of evidence reviewed below indicate that in
the absence of adaptation this leads to declining abundance and eventually local extinction in areas of unfavorable
climate, with potentially large effects on biodiversity and ecosystem services (see Box 4-2). Adaptive responses of
species are discussed in section 4.4.1.

Significant uncertainties arising from translating rates of warming into climate velocities and from estimating
displacement rates mean that the ability to track future climate can only be projected with low confidence across all
species examined. For trees there is medium confidence in these projections due to the large number of studies of
past, current and future displacement rates (robust evidence and medium agreement). Climate velocities are
calculated using a variety of methods and spatial resolutions making direct comparisons difficult, so climate
velocities in Figure 4-6B are based on semi-quantitative comparisons of several studies (see Figure 4-6 legend) and
have only low confidence (limited evidence and medium agreement). For the global average climate velocity (Figure
4-6B, center axis), lowest estimates are about one third of the values on the axis (e.g. Loarie et al., 2009; 0.42 km/yr
≈ 0.03 °C/yr for projected SRES A1B 21st century global land warming). Highest estimates are about four times the
values on the axis (Burrows et al., 2011; 2.7 km/yr ≈ 0.02 °C/yr global land warming from 1960-2009), but these
high estimates may be artifacts of very large spatial resolutions used in this analysis (Dobrowski et al., 2013). In
addition, the rates used in this analysis are based on temperature alone and most analyses suggest that including
more climate factors increases climate velocity (Feeley and Rehm, 2012; Dobrowski et al., 2013). Species
displacement rates are calculated based on a very wide range of methods including rates of displacement in the
palaeontological record, rates of current range shifts due to climate warming, models of dispersal and establishment,
maximum observed dispersal distances and genetic analyses (e.g., Kinlan and Gaines, 2003; Stevens et al., 2010b).
There are often large differences in estimates of dispersal rates across methods (e.g., Kinlan and Gaines, 2003;
Stevens et al., 2010b) due to intrinsic uncertainties in the methods and differences in the mechanisms included. For
example, estimates of tree displacement rates are frequently based on models or observations that explicitly or
implicitly include both dispersal of seeds and biotic and abiotic factors controlling establishment of adult trees.
Displacement rates of trees are often more strongly limited by establishment than dispersal (Higgins et al., 2003;
Meier et al., 2012). It is reasonable to expect that limits on establishment could also be important for other species
groups, but often only dispersal rates have been calculated (e.g., mammals, birds and phytophagous insects). As
such, the displacement rates for these species may be optimistic. For some species groups such as trees, there are a
wide range of published estimates of displacement rates leading to high confidence in the estimates in Figure 4-6C.
Considerably less is known about displacement rates for other species groups in response to climate change (e.g.,
some mammal groups, mollusks), so we have only low to medium confidence in estimates of their dispersal capacity.

[INSERT FIGURE 4-6 HERE]

Figure 4-6: (A) Rate of climate change, (B) corresponding climate velocities and (C) rates of displacement of
several terrestrial and freshwater species groups in the absence of human intervention. The thin dotted red arrows
give an example of interpretation. Rates of climate change of 0.03 °C/yr correspond to ca. 1.1 km/yr global average
climate velocity. When compared to rates of displacement, this would exceed rates for most plants, many primates
and some rodents. (A) Observed rates of climate change for global land areas are derived from CRUTEM4 climate
data reanalysis, all other rates are calculated based on the average of the CMIP5 climate model ensembles for the
historical period and for the future based on the four RCP emissions scenarios. The lower bound (17% of model
projections are outside this bound) is given for the lowest emissions scenario and the upper bound for the highest
emissions scenario. Data were smoothed using a 20-year sliding window, and rates are based means of between 17
and 30 models using one member per model. Global average temperatures at the end of the 21st century
corresponding to each of the RCP scenarios are from Knutti and Sedlacek (2012). Colors in the background
synthesize the ability of species to track climate through displacement. (B) Estimates of climate velocity were semi-
quantitatively synthesized from seven studies using a diversity of analytical approaches and spatial resolutions
(historical - Burrows et al., 2011; Chen et al., 2011; Dobrowski et al., 2013; projected future - Loarie et al., 2009;
Sandel et al., 2011; Schippers et al., 2011; Feeley and Rehm, 2012; Schloss et al., 2012). The three axes represent estimated climate velocities are representative of mountainous areas (left), averaged across all global land area (center, usually geometric means), and regions that are flat or have high rates climate change (right). (C) Rates of displacement for terrestrial plants - predominantly based on dispersal and establishment - were derived from paleobotanical records, modern plant invasion rates and genetic analyses (Kinlan and Gaines, 2003). The box represents ~95% of the observations in Kinlan and Gaines (2003) and the bar represents a qualitative estimate of the median. Displacement estimates for trees are based on reconstructed rates of tree migration during the Holocene (Clark, 1998; Clark et al., 2003; Kinlan and Gaines, 2003; McLachlan et al., 2005; Nathan, 2006; Pearson, 2006) and modeled tree dispersal and establishment in response to future climate change (Higgins et al., 2003; Iverson et al., 2004; Epstein et al., 2007; Goetz et al., 2011; Nathan et al., 2011; Meier et al., 2012; Sato and Ise, 2012). The box represents ~95% of the estimates and the bar is a qualitative estimate of the median. Displacement rates for mammals were based on modeled dispersal rates of wide range of mammal species from Schloss et al. (2012). The box represents 1.5 x the inter-quartile range of the estimates in Schloss et al. (2012) and the bar is the median.

Several groups of mammals have not been plotted because they include a relatively small number of species. Estimates for birds are based on observed rates of displacement (Chen et al., 2011; La Sorte and Jetz, 2010) and on modeled dispersal rates in response to projected climate change (Schippers et al., 2011; Barbet-Massin et al., 2012), but very low confidence in these estimates makes it difficult to set bounds or indicate central tendency. Displacement rates for phytophagous insects are based on observed dispersal rates (Kinlan and Gaines, 2003; Stevens et al., 2010b; Sekar, 2012). The qualitative estimate of median dispersal rate exceeds the highest rates on the axis and is on the order of 100 km/yr. These dispersal rates do not take into account limitations imposed by host plants and habitats. Displacement estimates for freshwater mollusks correspond to the range of dispersal given for passive plus active dispersal mechanisms for upstream movement in Kappes and Haase (2012).

**Observed Changes in Abundance and Local Extinctions** - Observations of range shifts imply changes in abundance; i.e., colonization at the "leading edge" and local extinction at the "trailing edge" of ranges. Evidence that the attribution of these responses to recent changes in climate can be made with *high confidence* is reviewed above, in the AR4 report and by Cahill et al. (2013). Changes in abundance, as measured by changes in the population size of individual species or shifts in community structure, also have occurred within existing range limits in response to recent global warming (*high confidence*) (Thaxter et al., 2010; Bertrand et al., 2011; Naito and Cairns, 2011; Rubidge et al., 2011; Devictor et al., 2012; Tingley et al., 2012; Cahill et al., 2013; Ruiz-Labourdette et al., 2013). More confident attribution to recent global warming is hindered by confounding factors such as disease, land use change and invasive species (Cahill et al., 2013). A new tentative conclusion since the AR4 is that climate related local extinctions appear to be more strongly related to species interactions than to physiological tolerance limits (*low confidence*) (Cahill et al., 2013). This gives weight to concerns that biological interactions, which are poorly understood and modeled, may play a critical role in mediating the impacts of future climate change on species abundance and local extinctions (Dunn et al., 2009; Bellard et al., 2012; Urban et al., 2012; Hannah, 2012).

A few examples illustrate the types of changes in abundance and local extinctions that are being observed and the challenges in attributing these to recent global warming. Some of the clearest examples of climate-related changes in species populations come from high latitude ecosystems where non-climate drivers are of lesser importance. For example, both satellite data and a large number of long-term observations indicate that shrub abundance is generally increasing over broad areas of Arctic tundra, which is coherent with predicted shifts in community structure due to warming (Epstein et al., 2007; Goetz et al., 2011; Myers-Smith et al., 2011). In the Antarctic, two native vascular plants, Antarctic pearlwort (*Colobanthus quitensis*) and Antarctic hair grass (*Deschampsia antarctica*) have become more prolific over recent decades, perhaps because they benefit more from warming of soils than do mosses (Hill et al., 2011). Penguin populations have declined in several areas of the Antarctic, including a recent local extinction of an Emperor penguin population that has been attributed to regional changes in climate (Trathan et al., 2011). The attribution to regional climate change is well supported, but the link to global warming is tenuous (Barbraud et al., 2011).

Mountains also provide good examples of climate signals in abundance and local extinction because of very strong climate gradients. The AR4 report already highlighted these responses and the case for changes in abundance, in particular plants, has become stronger since then. For example, Pauli et al. (2012) reported an increase in species richness from plant communities of mountain tops in the European boreal and temperate zones due to increasing
temperatures and a decrease in species richness on the Mediterranean mountain tops, probably due to a decrease in
the water availability in Southern Europe. An increase in the population size of warm adapted species at high
altitudes also appears to be attributable to increasing temperatures (Gottfried et al., 2012). However, these
attributions are complicated by other anthropogenic influences such as changes in grazing pressure, atmospheric
nitrogen deposition, and forest management practices (Gottfried et al., 2012). Altitudinal gradients in local and
global extinctions of amphibians also contributed to the attribution of these extinctions to recent global warming,
although this attribution remains controversial (see below).

Projected Changes in Abundance and Local Extinction - Ecological niche models do not predict population changes,
but the shifts in suitable climates can be used to infer areas where species populations might decline or increase.
Projected areas of local extinction from these types of models ranging from near 0% of the current range to more
than 95% (Settele et al., 2008; Bellard et al., 2012), and projected local colonization rates are equally variable.
There has been progress in coupling species distribution and species abundance models for a wide range of
organisms (Keith et al., 2008; Midgley et al., 2010; Matthews et al., 2011; Schippers et al., 2011; Oliver et al.,
2012; Renwick et al., 2012). These approaches have the substantial advantage of predicting extinction risk directly,
rather than by inference from changes in climate suitability (Fordham et al., 2012). The main conclusion from these
studies is species response to climate change is determined by a combination of factors, including its environmental
niche, demographics and life history traits, as well as interactions among these factors (Aiello-Lammens et al., 2011;
Conlisk et al., 2012; Fordham et al., 2012; Swab et al., 2012).

Changes in abundances will also be accompanied by changes in genetic diversity. At an intraspecific level, future
climate change projections are projected to induce severe losses of genetic diversity (Balint et al., 2011). In addition,
there is theoretical and observational evidence that range contractions will reduce genetic diversity (Pauls et al.,
2013), but that the loss of genetic diversity will depend on rates of migration and range contraction (Arenas et al.,
2012). Reductions in genetic diversity may then decrease the ability of species to adapt to further climate change or
other global changes. Although future projections of climate change impacts do not appear to result in the loss of
more phylogenetic diversity than expected by chance, projected species losses are projected to lead to phylogenetic
homogenization across European plant, bird and mammal communities (Thuiller et al., 2011, but see Davies et al.,
2011). Climate change may also compound losses of genetic diversity that already occurring due other global
changes such as the introduction of alien species or habitat fragmentation (Winter et al., 2009; section 4.2.4.6).

Observed Global Extinctions - Global species extinctions, many of them caused by human activities, are now close
to the upper limits of observed natural rates of extinction in the fossil record (Barnosky et al., 2011). Most
extinctions over the last several centuries can be attributed to habitat loss, overexploitation or invasive species
(Millennium Ecosystem Assessment, 2005a; Hofmann and Todgham, 2010; Cahill et al., 2013). Of the more than
800 global extinctions documented by the IUCN, only 20 have been tenuously linked to recent climate change
(Cahill et al., 2013). Mollusks, especially freshwater mollusks, have by far the highest rate of documented
extinctions of all species groups (Barnosky et al., 2011). Mollusk extinctions are primarily attributed to invasive
species, habitat modification and pollution — changes in climate are rarely evoked as a driver (Lydeard et al., 2004;
Regnier et al., 2009; Chiba and Roy, 2011; but see a few cases in Kappes and Haase, 2012; Cahill et al., 2013).
Freshwater fish have the highest documented extinction rates of all vertebrates, and again very few have even been
tenuously attributed to changing climate (Burkhead, 2012; Cahill et al., 2013). In contrast, changes in climate have
been identified as one of the key drivers of extinctions of amphibians (Pounds et al., 2006). There have been more
than 160 probable extinctions of amphibians documented over the last two decades, many of them in Central
America (Pounds et al., 2006; Kiesecker, 2011). The most notable cases have been the golden toad and Monteverde
harlequin frog (Atelopus varius) of Central America, which belong to a group of amphibians with high rates of
extinction ascribed to global warming with "very high confidence" (Pounds et al., 2006; Fischlin et al., 2007). This
case has raised a number of important issues about attribution since i) the proximate causes of extinction of these
and other Central American frogs appear to be an extremely virulent invasive fungal infection and land use change,
with regional changes in climate as a potential contributing factor and ii) changes in regional climate may have been
related to natural climate fluctuations rather than anthropogenic climate change (Sodhi et al., 2008; Lips et al., 2008;
Collins, 2010; Anchukaitis and Evans, 2010; Vredenburg et al., 2010; Kiesecker, 2011; McKenzie and Peterson,
2012; McMenamin and Hannah, 2012). While this case highlights difficulties in attribution of extinctions to recent
global warming, it also points to a growing consensus that it is the interaction of climate change with other global
change pressures that poses the greatest threat to species (Pereira et al., 2010; Hof et al., 2011; Brook et al., 2008).
Overall, there is very low confidence that observed species extinctions can be attributed to recent climate warming due to the very low fraction of species for which global extinction has been ascribed to climate change and tenuous nature of most attributions.

Projected Future Species Extinctions - Projections of future extinctions due to climate change have received considerable attention since the AR4 report. The AR4 report stated with medium confidence "that approximately 20-30% of the plant and animal species assessed to date are at increasing risk of extinction as global mean temperatures exceed a warming of 2-3°C above pre-industrial levels" (Fischlin et al., 2007). All model-based analyses since the AR4 broadly confirm this concern, leading to very high confidence that climate change will contribute to increased extinction risk for terrestrial species over the coming century (Pereira et al., 2010; Sinervo et al., 2010; Pearson, 2011; Warren et al., 2011; Hannah, 2012; Ihlow et al., 2012; Bellard et al., 2012; Sekercioglu et al., 2012; Wearne et al., 2012). More generally, there is broad agreement that climate change will result in shifts in species’ ranges and abundances, and that in the context of habitat fragmentation and other global change pressures this will contribute substantially to increased extinction risk of terrestrial species over the coming century (Pereira et al., 2010).

However, work since the AR4 also calls into question the ability of models to accurately quantify future extinction risk. Modeling studies have used a wide range of methods to estimate future extinction risk, generally inferring extinction risk indirectly from changes in availability of habitat and rarely taking into account demographic factors such as dispersal ability and generation time (Akcakaya et al., 2006). All studies project increased extinction risk by the end of the 21st century due to climate change, but these estimates range from below 1% to above 50% of species in the groups that have been studied (Pereira et al., 2010; Bellard et al., 2012; Cameron, 2012). However, most models do not incorporate key processes that link climate change to population change. For example, models rarely account for genetic and phenotypic adaptive capacity, dispersal, the effects of habitat fragmentation and loss, and community interactions as well as, in plants, the effects of rising CO2 concentrations. A general paucity of approaches to model validation, the large variation in the climate sensitivity of species groups and uncertainties about timescales involved in range reduction to extinction all lead to very low confidence that models currently predict extinction risk accurately (Kuussaari et al., 2009; Pereira et al., 2010; Dawson et al., 2011; Pearson, 2011; McMahon et al., 2011; Araujo and Peterson, 2012; Bellard et al., 2012; Fordham et al., 2012; Kramer et al., 2012; Hannah, 2012; Zurell et al., 2012; Halley et al., 2013). In addition, evidence from the paleontological record indicating very low extinction rates over the last several hundred thousand years of substantial natural fluctuations in climate - with a few notable exceptions such as large land animal extinctions during the Holocene - has also led to concern that forecasts of very high extinction rates due entirely to climate change may be overestimated (Botkin et al., 2007; Willis and Bhagwat, 2009; Pereira et al., 2010; Dawson et al., 2011; Hof et al., 2011; Willis and MacDonald, 2011; Wake, 2012). On the other hand, processes not accounted for in most models, such changes in biotic interactions such as pollination or predator-prey networks or non-linear changes in ecosystems with widespread consequences for resident species, mean that future extinction risks may also have been substantially underestimated, at least for certain areas and species groups (Schweiger et al., 2008; Bellard et al., 2012; Urban et al., 2008; Nakazawa and Doi, 2012; Hannah, 2012) or overestimated (Menéndez et al., 2008; Pateman et al., 2012). Finally, there is growing awareness that most modeling analyses have not accounted for climatic variation at fine spatial scales (microclimate) that may increase (Randin et al., 2009; Gillingham et al., 2012; Suggitt et al., 2012; Dobrowski et al., 2013; Franklin et al., 2013) or decrease (Trivedi et al., 2008; Engler et al., 2011) the likelihood of persistence of small populations under future climate change.

4.3.3 Impacts on Major Systems

This section covers impacts of climate change on broad types of terrestrial and freshwater ecosystems of the world. We have placed a particular emphasis on those areas of high vulnerability due to high exposure to climate change or that may be pushed past thresholds or “tipping points” by climate change.

Figure 4-7 provides an overview of two perspectives on estimating ecosystem vulnerability to climate change at the global scale. Figures 4-7A and 4-7B show projected future climate change in relationship to current inter-annual variability (Diffenbaugh and Giorgi, 2012; see similar analyses by Beaumont et al., 2011; Williams et al., 2007b).
This analysis highlights those areas where interannual variability is low compared to projected changes in climate, i.e., where future climate is likely to be outside of temperature fluctuations already experienced by ecosystems (Beaumont et al., 2011; Diffenbaugh and Giorgi, 2012). In this analysis, it is the species and ecosystems around the equator that are projected to be the most exposed to the largest climate changes. High latitude systems do not appear as exposed to climate change in this analysis because they already experience high interannual variability. A number of lines of evidence suggest that this index of climate change is an appropriate way to view species and ecosystem vulnerability. In particular, there is evidence that tropical species have evolved to have narrow thermal limits due to low inter and intra-annual climate variability, and are also already near their upper thermal limits (ectotherms - Deutsch et al., 2008; Huey et al., 2012; birds - Sekercioglu et al., 2012; trees - Corlett, 2011). On this basis, tropical species and ecosystems are predicted to be more sensitive to climate change than species and ecosystems where intra- and inter-annual variability in climate is high (e.g., Arctic and boreal ecosystems). However, there a number of physiological, evolutionary and ecological arguments that species and ecosystem vulnerability to climate change is considerably more complex (Corlett, 2011; Laurance et al., 2011; Gunderson and Leal, 2012; Walters et al., 2012, Gonzalez et al., 2010). Figure 4-7C compares future and contemporary climate focusing on the absolute differences between future and current climates. In this case, high northern latitude systems have the greatest exposure to rising temperature (often exceeding 6°C in the RCP 8.5 scenario, Figure 4-7C), which all else being equal would make them more vulnerable to climate change. Indeed, the greatest degree of recent climate warming has occurred at high Northern latitudes (Burrows et al., 2011) and the strongest and clearest signals of recent climate warming impacts on ecosystems come from these regions. A comparison of modeled biome level vulnerability indicated that temperate and high northern latitude systems are also the most vulnerable in the future (Gonzalez et al., 2010). There has been a tendency to oppose these two points of view; i.e., tropics are most vulnerable due to limited capacity of species to adapt to moderate global warming vs. high northern latitude systems are the most vulnerable because temperature increases are projected to the greatest at these latitudes. There are, however, there are good indications that both types of systems are highly vulnerable, but for different reasons (Corlett, 2011). This analysis highlights some of the reasons why there is a particularly strong focus on climate change impacts on tropical and high latitude systems in the following sub-chapters.

Figure 4-7: Two climate indexes related to the vulnerability of terrestrial biomes to future climate change based on the RPC 8.5 scenario and the CMIP5 climate model ensemble (from Diffenbaugh and Giorgi, 2012). Panels A and B show the evolution of June-August temperatures in two ca. 20 year time slices around 2055 and 2090 in comparison with the hottest year during the period 1980-2005. Climate change is projected to result in many areas of the tropics and sub-tropics exceeding the temperatures during the 1980-2005 period 100% of the time by 2055. High northern latitude systems exceed this limit less frequently especially in the 2055 time slice. Panel C shows the absolute temperature differences between the 1980-2005 reference period and the 2090 time slice. High northern latitudes are exposed to much greater temperature increases than other regions and this is representative of the regional patterns of climate change for all time slices and emissions scenarios.]

An issue that emerged in the AR4 report and has grown substantially in prominence since then is that of thresholds or ‘tipping points’ (Lenton et al., 2008; Scheffer et al., 2009; Leadley et al., 2010). Tipping points occur when positive feedback processes dominate over negative feedback loops, and the system begins to change as a result of both its internal dynamics and the external forcings, leading to a change in state that is difficult or slow to reverse (Lenton et al., 2008; Scheffer et al., 2009; Leadley et al., 2010). In the AR5, tipping points are considered “a level of change in system properties beyond which a system reorganizes, often abruptly, and persists in its new state even if the drivers of the change are abated” (Glossary). Many examples of such thresholds have now been identified (Scheffer, 2009). Most are at the local scale, but the potential existence of ecological thresholds at a sufficient scale to have regional or global consequences is high. Posited examples include a drought and temperature threshold in the Amazon beyond which forest dieback and fires could contribute to large C fluxes to the atmosphere, or thawing of Arctic permafrost leading to emissions of CH4 and CO2 in sufficient quantities to further warm the Earth and amplify the permafrost warming (Lenton et al., 2008; Leadley et al., 2010). Ecosystem water balance thresholds that are surpassed through changes in temperature and precipitation, both gradual and through extreme events, may be a widespread phenomenon across a range of biomes (Knapp et al., 2008a). Large-scale tipping points have not occurred in the recent past, but there is good evidence for tipping points in the distant past (section 4.2.2) and the sub-sections below highlight reasons for concern that they could occur in the near future. A thorough analysis of all
the highlighted tipping points is not possible in this section, so we have highlighted some of the most important and well-studied tipping points. Others are discussed in Lenton et al. (2008) and Leadley et al. (2010).

4.3.3.1. Forests and Woodlands

Forests and woodlands, are the principal providers of timber and pulp, biofuel and an important and steady source of good-quality water worldwide. They also supply other services, including food and, cultural services such as recreation. Forests are the habitat of a large fraction of the earth's terrestrial biodiversity, especially in tropical regions (Gibson et al., 2011). Climate change and forests interact strongly. Climate and atmospheric CO$_2$ concentrations are major drivers of forest productivity and forest dynamics. At the same time forests play an important role in controlling climate through the large amounts of carbon they can store or release, and through direct effects on the climate such as the absorption or reflection of solar radiation (albedo), cooling through evapotranspiration and the production of cloud-forming aerosols (Arneth et al., 2010; Pan et al., 2011; Pielke et al., 2011).

Combinations of ground based observations, atmospheric carbon budgets and satellite measurements indicate that forests are currently a net sink for carbon at the global scale. It is estimated that intact and regrowing forests currently contain 861±66 PgC and stored 4.0 ± 0.7 Pg C year$^{-1}$ globally between 2000 and 2007 (Pan et al., 2011; WG1 Chapter 6; Canadell et al., 2007; Le Quere et al., 2009). The carbon stored in intact and regrowing forests was counterbalanced by a release due to land-use change of 2.8±0.4 Pg C year$^{-1}$ over this same period due to tropical deforestation, resulting in a net C balance for global forests of 1.2±0.8 Pg C year$^{-1}$.

The phrase 'biophysical effects' is shorthand for the effect forests have on the climate other than through being sources and sinks of greenhouse gases. These effects are now well documented, significant and are increasingly included in models of global and regional climate change. Observations and models indicate that boreal and tropical forests have the largest influence on climate, but for different reasons: boreal forests have low albedo (i.e., reflect less solar radiation); and tropical forests pump more water and aerosols into the atmosphere than non-forest systems in similar climates (Davin and de Noblet-Ducoudre, 2010; Delire et al., 2011; Pielke et al., 2011). Overall, this means that the biophysical effect of boreal forests is to warm the atmosphere and tropical forests to cool the atmosphere, compared to non-forest systems (Davin and de Noblet-Ducoudre, 2010). It has also been shown that forests influence regional precipitation through biophysical effects: deforestation in the tropics can alter regional precipitation (Butt et al., 2011) and there is growing evidence of similar effects in temperate forests (Pielke et al., 2011).

The future of the interaction between climate and forests is unclear. The carbon stored by intact and regrowing forests appears have stabilized compared to the 1990's, after having increased in the 1970's and 1980's (Canadell et al., 2007; Pan et al., 2011). The drivers behind the forest carbon sink vary greatly across regions. They include forest regrowth and simulation of C sequestration by climate change, rising atmospheric CO$_2$ concentrations and N deposition (Pan et al., 2011). Most models suggest that rising temperatures, drought and fires will lead to forests becoming a weaker sink or a net carbon source before the end of the century (Sitch et al., 2008; Bowman et al., 2009). Fires play a dominant role in driving forest dynamics in many parts of the world and the susceptibility to fire is projected to increase little for the lowest emissions scenario (RCP 2.6), but substantially for the high emissions scenario (RCP 8.5, Figure 4-8). Climate change-mediated disease and insect outbreaks could precipitate this transition (Kurz et al., 2008). The greatest risks for large feedbacks from forests to climate arise from widespread tree mortality and fire in tropical forests and low latitude areas of boreal forests, as well as northward expansion of boreal forests into arctic tundra (Lenton et al., 2008; Kriegler et al., 2009; Good et al., 2011).

[INSERT FIGURE 4-8 HERE] Figure 4-8: Forest Fire Danger Index (FFDI) and its changes simulated with the HadGEM2-ES Earth System Model (Jones et al., 2011). Left column: 30-year mean FFDI (MacArthur, 1973; Noble et al., 1980) for 1980-2010 (top) and 2070-2100 under RCP2.6 (middle) and RCP8.5 (bottom). Right column: differences in 30-year mean FFDI relative to 1980-2010 for 2070-2100 under RCP2.6 (middle) and RCP8.5 (bottom). Maps show 30-year means of
FFDI values calculated using daily model data, following the methodology of (Golding and Betts, 2008). FFDI is not shown where the simulated vegetation cover is less than 10% for the respective time period (colored grey).]

Recent evidence suggests that the stimulatory effects of global warming and rising CO₂ concentrations on tree growth may have already peaked in many regions (Charru et al., 2010; Silva et al., 2010; Silva and Anand, 2013) and that warming and changes in precipitation are increasing tree mortality in a wide range of forest systems, acting via heat stress, drought stress, pest outbreaks and a wide range of other indirect impact mechanisms (Allen et al., 2010a; Box 4-2). Detection of a coherent global signal is hindered by the lack of long-term observations in many regions, and attribution to climate change is difficult due to the multiplicity of mechanisms mediating mortality (Allen et al., 2010a).

Deforestation has slowed over the last decade (Meyfroidt and Lambin, 2011). This includes substantial reductions in tropical deforestation in some regions, such as the Brazilian Amazon, where deforestation rates declined rapidly after peaking in 2005 (Secretariat of the Convention on Biological Diversity, 2010). It is unclear if the global trend to reduced forest loss will continue - there are substantial pressures to deforest for the production of food and biofuels (Wise et al., 2009; Meyfroidt and Lambin, 2011). However, there are a number of signs that tropical deforestation could slow substantially over the next few decades, although this is thought to require continued and substantial technological, institutional and policy innovations (Wise et al., 2009; Meyfroidt and Lambin, 2011; Westley et al., 2011).

Box 4-2. Tree Mortality and Climate Change

Extensive tree mortality and widespread forest die-off linked to drought and temperature stress have been documented recently on all vegetated continents (Allen et al., 2010a; Figure 4-9). Plot level datasets with appropriate spatial and temporal spans to detect long-term tree mortality trends (>20 years) are currently lacking for many regions (Anderegg et al., 2013a), leading to low confidence in the ability to detect a global trend. Nevertheless, long-term increasing tree mortality rates associated with temperature increases and drought have been documented in boreal and temperate forests in western North America (van Mantgem et al., 2009; Peng et al., 2011). Increased levels of tree mortality following drought episodes have also been detected across widespread plot networks in multiple tropical forests (Kraft et al., 2010; Phillips et al., 2010) and Europe (Carnicer et al., 2011).

Episodes of widespread die-off (high mortality rates of dominant tree species at a landscape scale) have been observed in multiple vegetation types, particularly in western North America, Australia, and southern Europe (Raffa et al., 2008; Carnicer et al., 2011; Anderegg et al., 2013b). Some widespread die-off events have occurred concomitant with infestation outbreaks (Raffa et al., 2008), where pest populations are also directly influenced by climate, such as population release by warmer winter temperatures (Bentz et al., 2010). While strong attribution of extensive tree mortality to recent warming has been made in a few studies, the paucity of long-term studies of the mechanisms driving mortality means that there is low confidence that this attribution can be made at the global scale.

Widespread die-off has influenced the species composition, structure, and age demographics, and successional trajectories in these forests, and in some cases led to decreased plant species diversity and increased risk of invasion (Kane et al., 2011; Anderegg et al., 2012). Widespread mortality also has multiple effects on biosphere-atmosphere interactions and could play an important role in future carbon-cycle feedbacks through complex effects on forest biophysical properties and biogeochemical cycles (Breshears et al., 2005; Kurz et al., 2008; Anderson et al., 2011).

Projections of tree mortality due to climate stress and potential thresholds of widespread forest loss are currently highly uncertain (McDowell et al., 2011). Most current global scale vegetation models have little-to-no mechanistic representation of tree mortality (Fisher et al., 2010; McDowell et al., 2011). Nonetheless, a global analysis of tree hydraulic safety margins found that more than 70% of surveyed tree species operate close to their limits of water stress tolerance (Choat et al., 2012), indicating that vulnerability to drought and temperature stress will not be limited to arid and semi-arid forests. Furthermore, timescales of tree and plant community recovery following drought are largely unknown, but preliminary evidence from several systems indicates recovery times may be shorter than recent drought return intervals, leading to “compounding” effects of multiple droughts (Mueller et al.,...
4.3.3.1.1. Impacts of climate change on boreal forests

Most projections suggest a poleward expansion of forests into tundra regions, accompanied by a general shift in composition towards more temperate plant functional types (e.g., evergreen needleleaf being replaced by deciduous broadleaf; or in colder regions, deciduous needleleaf replaced by evergreen needleleaf; Lloyd et al., 2011). At the same time, projections of climate-driven changes in boreal forests over the next few centuries remain divergent on some issues, partly as a result of different processes of change being considered in different models. In particular, the inclusion or exclusion of fire and insects makes a big difference, possibly making the boreal forest more susceptible to a rapid, non-linear or abrupt decline in some regions of the circum-Northern Hemisphere (Bernhardt et al., 2011; Mann et al., 2012; Scheffer et al., 2012). Recent change (Box 4-2) and dynamic vegetation modeling (e.g., Sitch et al., 2008) provide high confidence that regions of the boreal forest could witness widespread forest dieback. These shifts put at risk the boreal carbon sink, estimated at 0.5 Pg year⁻¹ in 2000-2007 (Pan et al., 2011; Mann et al., 2012).

Whereas boreal forest productivity is widely expected, with medium confidence, to increase as a result of warming (Hari and Kulmata, 2008; Bronson et al., 2009; Zhao and Running, 2010; Van Herk et al., 2011), and early analyses of satellite observations confirmed this trend in the 1980s, more recent and longer-term assessments indicate with high confidence that many areas of boreal forest have instead experienced productivity declines (Goetz et al., 2007; Parent and Verbyla, 2010; Beck et al., 2011b; de Jong et al., 2011). The reasons for these “browning” trends have been debated but they are not directly related to fire disturbance (because fires produce random rather than systematic trends in time series). The best evidence to date indicates that warming-induced drought, specifically the greater drying power of air (vapor pressure deficit), induces photosynthetic down-regulation of boreal tree species, particularly conifer species, most of which are not adapted to the warmer conditions (Welp et al., 2007; Bonan, 2008; Van Herk et al., 2011). Satellite evidence for warming-induced productivity declines has been corroborated by tree-ring studies (Barber et al., 2000; Hogg et al., 2008; Beck et al., 2011b; Porter and Pisaric, 2011; Griesbauer and Green, 2012) and long-term tree demography plots in more continental and densely forested areas (Peng et al., 2011; Ma et al., 2012). Conversely, productivity has increased at the boreal-tundra ecotone where more mesic conditions support an expected warming-induced growth response (Rupp et al., 2001; McGuire et al., 2007; Goldblum and Rigg, 2010; Beck et al., 2011b).
There is high confidence that warming and drying, coupled with productivity declines, insect disturbance and associated tree mortality, also favor greater fire disturbance. The boreal biome fire regime has intensified in recent decades, exemplified by increases in the extent of area burned but also a longer fire season and more episodic fires that burn with greater energy output or intensity (Girardin and Mudelsee, 2008; Macias Fauria and Johnson, 2008; Kasischke et al., 2010; Turetsky et al., 2011; Mann et al., 2012). The latter is particularly important because more severe burning consumes soil organic matter to greater depth often to mineral soil, providing conditions that favor recruitment of deciduous species that, in the North American boreal forest, replace what was previously evergreen conifer forest (Johnstone et al., 2010; Bernhardt et al., 2011). Fire mediated composition changes in post-fire succession influence a host of ecosystem feedbacks to climate, including changes net ecosystem carbon balance (Bond-Lamberty et al., 2007; Goetz et al., 2007; Welp et al., 2007; Euskirchen et al., 2009) as well as albedo and energy balance (Randerson et al., 2006; Jin et al., 2012; O’Halloran et al., 2012). The most comprehensive study to date (Randerson et al., 2006) focused on a set of flux tower measurements across a composition gradient, but this study was necessarily limited to a few sites in central Alaska. The extent to which the net effect of these feedbacks will exacerbate or mitigate additional warming is not well known over the larger geographic domain of the boreal biome, except via modeling studies that are relatively poorly constrained due to sparse in situ observations.

There is high confidence that the vulnerability of the boreal biome to this cascading series of interacting processes (Wolken et al., 2011), and their ultimate influence on climate feedbacks, differs between North America and northern Eurasia. The latter is dominated by deciduous conifer (larch) forest, extending from western Russia across central to eastern Siberia — a region more than twice the size of the North American boreal biome, most of it underlain by permafrost conditions of the Siberian environment. In terms of post-fire succession analogous to the North American boreal biome, larch function more like deciduous species than evergreen conifers, with greater density and biomass gain in more severely burned areas, given adequate seed survival through fire events or post-fire seed dispersal (Zyryanova, 2007; Osawa et al., 2010; Alexander et al., 2012). Although the fire regime has intensified in the last 100 years in Siberia, as well as in North America (Soja et al., 2007; Mann et al., 2012; Ali et al., 2012; Marlon et al., 2013), the likelihood of regime shifts in larch forests is currently unknown, partly because larch are self-replacing (albeit at different densities) and partly because it is largely dependent on the fate of permafrost across the region.

The vulnerability of permafrost to thawing and degradation with climate warming is critical not only for determining the rate of a boreal-tundra biome shift, and its associated net feedback to climate, but also for predicting the degree to which the mobilization of very large carbon stores frozen for centuries will exacerbate additional warming (Schuur et al., 2008; 2009; Tarnocai et al., 2009; Romanovsky et al., 2010; Schaefer et al., 2011; see also Section 4.3.3.4). The extent and rate of permafrost degradation varies with temperature gradients from warmer discontinuous permafrost areas to colder, more continuous areas, but also with the properties of the soil composition and biology (e.g., Mackelprang et al., 2011). The degree of thermokarsting associated with different substrates, and associated topographic relief, is tremendously variable because boreal vegetation in latter successional stages (evergreen conifers in North America) insulate permafrost from air temperature increases; soils with differing silt and gravel content tend to have different ice content that, when melted, produces different degradation and deformation rates; and other factors such as fire disturbance interactions with both vegetation cover and soil organic depth as insulating layers (Jorgenson et al., 2010; Grosse et al., 2011). This variability, and vulnerability, is poorly represented in earth system models (McGuire et al., 2012) and is thus the emphasis of research initiatives currently underway. Carbon management strategies to keep permafrost intact, for example by removing forest cover to expose the land surface to winter temperatures (Zimov et al., 2009) are impractical not only because of the vast spatial domain underlain by permafrost, but also because of the broad societal and ecological impacts that would result.

4.3.3.1.2. Impacts of climate change on temperate forests

The largest areas of temperate forest are found in eastern North America, Europe and eastern Asia. The overall trend for forests in these regions has until recently been an increase in growth rates of trees and in total carbon stocks. This has been attributed to a combination of increasing growing season length, rising atmospheric CO$_2$ concentrations, nitrogen deposition and forest management – specifically regrowth following formerly more intensive harvesting regimes (Ciais et al., 2008). The relative contribution of these factors has been the subject of
substantial and unresolved debate (Boisvenue and Running, 2006). Most temperate forests are managed such that
any change is and will be to a large extent anthropogenic. With the exception of Europe (Ciais et al., 2008), the
relative contribution of anthropogenic and environmental factors to forest growth and carbon uptake rates remains to
be quantified.

The world’s temperate forests act as an important carbon sink (high confidence due to robust evidence; high
agreement), absorbing 0.7±0.08 Pg C year$^{-1}$ from 1990 to 1999 and 0.8±0.09 from 2000 to 2007 (Pan et al.,
2011). This represents 34% of global carbon accumulation in intact forests and 65% of the global net forest carbon
sink (total sink minus total emissions from land use).

Recent indications are that temperate forests and trees are beginning to show signs of climate stress, including: a
reversal of tree growth enhancement in some regions (N America - Silva et al., 2010; Silva and Anand, 2013;
Europe - Charru et al., 2010; Bontemps et al., 2011; Kint et al., 2012); increasing tree mortality (Allen et al., 2010a,
Box 4-2); and changes in fire regimes, insect outbreaks and pathogen attacks (Adams et al., 2012; Edburg et al.,
2012). In north-eastern France, widespread recent declines in growth rates of European beech (Fagus sylvatica L.)
have been attributed to decreasing water availability (Charru et al., 2010). These trends threaten the substantial role
of temperate forests as net carbon sinks, but it is still unclear to what extent the observations are representative for
temperate forests as a whole. Several studies find that tree growth rates in temperate forests passed their peak in the
late 20$^{th}$ century and that the decline in tree growth rates can be attributed to climatic factors, especially drought or
heatwaves (Charru et al., 2010; Silva et al., 2010). Extreme climate events have had a major impact on temperate
forests over the last decade (Witte et al., 2011; Kasson and Livingston, 2012; Ciais et al., 2005). Extensive forest
fires occurred in Russia during the exceptionally hot and dry summer of 2010 (Witte et al., 2011). The complex
interactions between climate and forest management in determining susceptibility to extreme events make it difficult
to unequivocally attribute these events to recent climate warming (Allen et al., 2010a). There is low confidence
(limited evidence; medium agreement) that climate change is threatening the temperate forest carbon sink directly or
indirectly.

At the biome level, there remains considerable uncertainty in the sign and the magnitude of the carbon cycle
response of temperate forests to climate change. A comparison of DGVM models showed that for identical end of
21$^{st}$ century climate projections, temperate forests are variously projected to substantially increase in total (biomass
plus soil) carbon storage, especially as a result of potential gains in tree cover in eastern North America and Europe;
or decrease due to reductions in total carbon storage per hectare and loss of tree cover (Sitch et al., 2008).
Projections for eastern Asia are less variable: temperate forests remain carbon sinks over the coming century, with
carbon storage generally peaking by mid-century and then declining (He et al., 2007; Sitch et al., 2008; Ni, 2011;
Peng et al., 2009). However, regional vegetation models for China predict a substantial northward shift of temperate
forest (Weng and Zhou, 2006; Ni, 2011). There is little indication from either models or observations that the
responses of temperate forests to climate change are characterized by tipping points. The feedback mechanisms
operating in boreal and tropical forests which lead to tipping point behavior appear to be less dominant in temperate
forests (Bonan, 2008). There is low confidence (medium evidence; low agreement) on long-term, climate-driven
changes in temperate forest biomass and geographical range shifts.

At the species level, models predict that the potential climatic space for most tree species will shift poleward and up
in altitude in response to climate change (Dale et al., 2010; Ogawa-Onishi et al., 2010; Hickler et al., 2012).
Associated long-term projected range shifts generally vary from several km to several tens of km per decade, most
probably cannot achieve such high rates by natural migration (e.g. Chmura et al., 2011, see also 4.3.2.5). Therefore,
assisted migration has been suggested as an adaptation measure (see 4.4.2.4). Such shifts would alter biodiversity
and ecosystem services from temperate forests (e.g. Dale et al., 2010). Multi-model comparisons for temperate
forests, however, illustrate that there are differences in species response and that models differ greatly in the severity
of projected climate change impacts on species ranges (Kearney et al., 2010; Kramer et al., 2010; Morin and
Thuiller, 2009; Cheaib et al., 2012). Tree growth models project increased tree growth at the poleward and high
altitudinal range limits over most of the next century in China (Ni, 2011). New approaches to modelling tree
responses, based on the sensitivity of key life-history stages, suggest that climate change impacts on reproduction
could be a major limitation on temperate tree distributions (Morin et al., 2007). Comparisons with paleoecological
data have helped improve confidence in the ability of models to project future changes in species ranges (Pearman et
4.3.3.1.3. Impacts of climate change on tropical wet and dry forests

The responses of tropical forests to variability and change in climate and atmospheric CO\textsubscript{2} concentration are superimposed upon the direct influences of human activities. Humans modify the composition and structure of tropical forests through the harvest of individual plants or animals and through management interventions to favor some species or structural features over others. Humans also convert forests to cropland or grazing land and influence forest exposure to ignition sources. Forest conversion is the best understood change that is underway in tropical forests. Climate change effects on tropical forests interact with the direct influences of humans and are understood largely through field studies of the responses of forests to extreme weather events and through models that are able to simulate a growing number of ecological and atmospheric processes.

A key uncertainty in our understanding of future impacts of climate change on tropical forests is the strength of direct CO\textsubscript{2} effects on photosynthesis and transpiration (see section 4.3.2.4). These responses will play an important role in determining tropical forest trends as temperatures and atmospheric CO\textsubscript{2} concentrations rise. There is a physiological basis for arguing that photosynthesis will increase sufficiently to offset the inhibitory effects of higher temperatures on forest productivity (Lloyd and Farquhar, 2008) although heightened photosynthesis does not necessarily translate into an increase in overall forest biomass, (Körner and Basler, 2010). DGVMs and the current generation of Earth System Models, including those used within CMIP5 (e.g., Jones et al., 2011) generally use formulations for CO\textsubscript{2} effects on photosynthesis and transpiration based on laboratory-scale work (Jarvis, 1976; Farquhar et al., 1980; Ball et al., 1987; Stewart, 1988; Collatz et al., 1992; Leuning, 1995; Haxeltine and Prentice, 1996; Cox et al., 1998) that pre-dates larger ecosystem-scale studies, although some models have been optimized on the basis of more recent data (Jones et al., 2011).

A second important source of uncertainty is the rate of future CO\textsubscript{2} and temperature rise (Betts et al., 2012). Modelled simulations of future climate in tropical forest regions indicate with high confidence (robust evidence, high agreement) that temperature will increase. Future precipitation change, in contrast, is highly uncertain and varies considerably between climate models (WG1 Annex A), although there is medium confidence (medium evidence, medium agreement) that some tropical regions, such as the eastern Amazon Basin, will experience lower precipitation (Malhi et al., 2009b). The range of possible shifts in the moist tropical forest envelope is large, sensitive to the responsiveness of water use efficiency (WUE) to rising concentrations of atmospheric CO\textsubscript{2}, and varies depending on the climate and vegetation model that is used (Scholze et al., 2006; Zelazowski et al., 2011) (Sitch et al., 2008) Recent model medley studies (Cox et al., 2013; Huntingford et al., 2013) indicate that the future geographical range of moist tropical forests as determined by its shifting climatological envelope is less likely to undergo major retractions or expansions by 2100 through climate change alone as previously predicted (Cox et al., 2000).

Changes in the species composition and biomass of moist tropical forests observed in networks of long-term forest plots in moist tropical forests are generally consistent with increasing WUE associated with the rising concentration of CO\textsubscript{2}. Lianas and fast-growing tree species are increasing, as is forest biomass (Phillips et al., 2002; Phillips et al., 2005; Lewis et al., 2009a; Lewis et al., 2009b; Lewis et al., 2011). Faster tree growth could also result from changes in solar radiation and the ratio of diffuse to direct radiation (Mercado et al., 2009; Lewis et al., 2009a; Brando et al., 2010). There is low confidence (limited evidence, medium agreement) that the composition and biomass of Amazon and African forests are changing through the rise in atmospheric CO\textsubscript{2}. The potential suppression of photosynthesis and tree growth in tropical forests through rising air temperatures is supported by physiological and eddy covariance studies (Doughty and Goulden, 2008; Lloyd and Farquhar, 2008; Wood et al., 2012), but is not yet observed as changes in forest biomass (except Clark et al., 2003).
Since AR4, there is new experimental and observational evidence of ecological thresholds of drought and fire in moist tropical forests that points to an important indirect role of climate change in driving large-scale changes in moist tropical forests (see Box 4-3). Forest tree mortality increased abruptly above a critical level of soil moisture depletion in two rainfall exclusion experiments (Nepstad et al., 2007; Fisher et al., 2008) and above a critical level of weather-related fire intensity in a prescribed burn experiment (Brando et al., 2012). These experimental results were corroborated by observations of increased tree mortality during the extremely severe 2005 drought in the Amazon (Phillips et al., 2009) and extensive forest fire (Alencar et al., 2006; Aragão et al., 2008; Alencar et al., 2011; Box 4-3). There is medium confidence (medium evidence, high agreement) that moist tropical forests have many tree species that are vulnerable to drought- and fire-induced mortality during extreme dry periods. There is low confidence (limited evidence, low agreement) that the severe droughts that have affected the Amazon region are associated with climate change (Shiogama et al., 2011).

There is also a growing body of evidence that severe weather events interact with land use to influence moist tropical forest fire regimes. Many moist tropical forests are not susceptible to fire during typical rainfall years because of high moisture content of fine fuels Cochrane, 2003. Selective logging, drought, and fire itself can reduce this fire resistance by killing trees, thinning the canopy and allowing greater heating of the forest interior; Uhl and Kauffman, 1990; Ray et al., 2005; Curran et al., 2004; Box 4-3). Land use often increases the ignition sources in tropical landscapes (Silvestrini et al., 2011). These relationships are not yet represented fully in coupled climate-vegetation models. There is high confidence (robust evidence, high agreement) that forest fire frequency and severity is increasing through the interaction between severe droughts and land use.

There are an estimated one million square kilometres of dry tropical forests in the world (Miles et al., 2006), defined as forests occurring in tropical regions characterized by pronounced seasonality in rainfall distribution (Mooney et al., 1995). More than half of the remaining tropical dry forests (54%) are located in South America. Approximately one third of the remaining tropical forests in the Americas are predicted to experience “severe climate change” by 2055, defined as an increase in air temperature of at least 2.5 deg. C and/or a decrease in precipitation of at least 50 mm a⁻¹. Less than one fifth of forests in other tropical dry forest formations are faced with this level of climate risk (Miles et al., 2006). Climate change, deforestation, fragmentation, fire, or human pressure place virtually all (97%) of the remaining tropical dry forests at risk (Miles et al., 2006).

____ START BOX 4-3 HERE ____

Box 4-3. A Possible Amazon Basin Tipping Point

Since AR4, our understanding of the potential of a large-scale, climate-induced, self-reinforcing transition of Amazon forests to a dry stable state (known as the Amazon “forest dieback”) has improved. Modeling studies indicate that the likelihood of a purely climate-driven dieback in 2100 is lower than previously thought (Malhi et al., 2009b; Good et al., 2013; Cox et al., 2013; Huntingford et al., 2013) and there is now medium confidence (robust evidence, medium agreement) that climate change alone will not lead to widespread forest loss during this century. Meteorological fire danger is projected to increase (Golding and Betts, 2008; Betts et al., submitted; Figure 4-8). Field studies and regional observations, have provided new evidence of critical ecological thresholds and positive feedbacks between climate change and land-use activities that could drive a fire-mediated, self-reinforcing dieback in this century. There is now medium confidence (medium evidence, medium agreement) that severe drought episodes, land use, and fire interact synergistically to drive the transition of mature Amazon forests to low-biomass, low-statured pyrophilic woody vegetation.

Most primary forests of the Amazon Basin have damp fine fuel layers and low susceptibility to fire, even during annual dry seasons (Uhl and Kauffman, 1990; Ray et al., 2005). Forest susceptibility to fire increases through canopy thinning and greater sunlight penetration caused by tree mortality associated with selective logging (Uhl and Kauffman, 1990; Ray et al., 2005; Barlow and Peres, 2008), previous forest fire (Balch et al., 2008; Brando et al., 2012), severe drought (Alencar et al., 2006), or drought-induced tree mortality (Nepstad et al., 2007; da Costa et al., 2010). The impact of fire on tree mortality is also weather-dependent. Under very dry, hot conditions, fire-related tree mortality can increase sharply (Brando et al., 2012). Under some circumstances, tree damage is sufficient to allow light-demanding, flammable grasses to establish in the forest understory, increasing forest susceptibility to
further burning (Veldman and Putz, 2011). There is high confidence (robust evidence, high agreement) that logging,
severe drought, and previous fire increase Amazon forest susceptibility to burning.

Landscape level processes further increase the likelihood of forest fire. Ignition sources are more common in
agricultural and grazing lands than in forested landscapes (Silvestrini et al., 2011) (high confidence: robust evidence, high agreement), and forest conversion to grazing and crop lands can inhibit regional rainfall through changes in
albedo and evapotranspiration (Costa et al., 2007; Butt et al., 2011; Knox et al., 2011) (low confidence: medium
evidence, low agreement) or through smoke, that can inhibit rainfall under some circumstances (Andreae et al.,
2004) (low confidence: medium evidence, low agreement). Apart from these landscape processes, climate change
could increase the incidence of severe drought episodes (Shiogama et al., 2011).

In sum, fire-mediated changes in Amazon forests are self-reinforcing through increases in tree mortality, forest
susceptibility to fire, ignition sources, and regional rainfall inhibition (Figure 4-10). If current patterns of
deforestation, logging, severe drought, and forest fire continue into the future, 55% of the region’s forests will be
cleared or degraded by 2030, even without invoking positive feedbacks with regional climate, releasing 20±10 Pg of
carbon to the atmosphere (Nepstad et al., 2008) (low confidence: low evidence, medium agreement). The likelihood
of a tipping point being reached may decline if extreme droughts (such as 1998, 2005, and 2010) (Marengo et al.,
2011) become less frequent, if land management fires are suppressed, if forest fires are extinguished (Soares-Filho et
al., 2012), if deforestation declines, or if cleared lands are reforested (Nepstad et al., 2008). The 76% decline in
deforestation in the Brazilian Amazon with 80% of the region’s forest still standing (INPE, 2012) demonstrates that
large-scale mitigation of a fire-mediated tipping point could be feasible.

4.3.3.2. Rangelands and Drylands, including Mediterranean-type Systems

The following sections treat a wide range of terrestrial ecosystems covering a large part of the terrestrial surface,
whose unifying features are that they are not classified as forests and they typically exhibit strong water stress for
several months each year. Grasses, grass-like plants and herbs are usually a major part of their vegetation cover. The
principle land use often involves either wild or domestic herbivores.

4.3.3.2.1. Savannas

Savannas are defined as mixtures of coexisting trees and grasses. The geographical distribution of tropical savannas
is determined by temperature, the seasonal availability of water, fire and soil conditions (Ellery et al., 1991; Walker
and Langridge, 1997; Staver et al., 2011) and is therefore inferred to be susceptible to climate change. The
proportion of trees and grasses in savannas is considered inherently unstable under some conditions (DeMichele et
al., 2011; Staver et al., 2011; Wake, 2012). Savannas are characterized by annual to decadal fires of relatively low
intensity, which are an important factor in maintaining the tree-grass proportions (Beerling and Osborne, 2006), but
also constitute a major global source of pyrogenic emissions from land to atmosphere (Schultz et al., 2008; van der
Werf et al., 2010). The differential effects of climate change, rising CO₂, fire and herbivory on trees and grasses
have the potential to alter the structure, composition and ecosystem service delivery capacity of mixed tree-grass
systems (e.g. Hughes et al., 2006). Since tropical savannas occupy about an eighth of the global land surface, and
mixed tree-grass systems outside the tropics cover a similar area, small changes in their structure or fire regime can have globally-significant effects on the emission or uptake of radiatively-active gases and particles.

There is evidence from many parts of the world that the tree cover and biomass in savannas has increased over the past century and in some places, on all continents, continues to do so (high agreement, robust evidence: Cabral et al., 2009; Angassa and Oba, 2008; Moleele et al., 2002; Wigley et al., 2009; Witt et al., 2009; Lunt et al., 2010; Rohde and Hoffman, 2012). The general consequences are more carbon stored per unit land area as tree biomass and soil organic matter (Hughes et al., 2006; Liao et al., 2006; Knapp et al., 2007; Throop and Archer, 2008; Boutton et al., 2009), changes in hydrology (Muñoz-Robles et al 2011) and reduced grazing potential (Scholes and Archer, 1997). Increasing tree cover in savannas has been attributed to changes in land management (Joubert et al., 2008; Van Aukcn, 2009), rising CO₂ (Bond and Midgley, 2012; Buitenwerf et al., 2012), climate variability and change (Eamus and Palmer, 2008; Fensham et al., 2009) or two or more of these factors acting in combination (Ward, 2005; Bond et al., 2008). As yet there are no studies that definitively attribute the relative importance of the various causes of woody plant biomass increase in savannas and grasslands, but there is medium agreement and robust evidence that climate and atmospheric changes are contributing factors in many cases. The increased growth rate of C3 trees relative to C4 grasses under by rising CO₂ could relieve the demographic bottleneck that keeps trees trapped within the flame zone of the grasses, a hypothesis supported by elevated CO₂ experiments with savanna saplings (Kgope et al., 2010). A vegetation model of grasslands, savannas and forests suggests that rising CO₂ does increase the likelihood of abrupt shifts to woody states, but the transition will take place at different CO₂ concentrations in different environments (Higgins and Scheiter, 2012). On the other hand, observation of contrasts in the degree of savanna thickening between land parcels with the same CO₂ exposure but different land use histories, topographic position or soil depth (Wu and Archer, 2005; Wiegand et al., 2005) imply that land management, water balance and microclimate are also important. Tree cover in savannas is rainfall-constrained (Sankaran et al., 2005), suggesting that increases in rainfall could lead to increased tree biomass. In some places the savanna boundary is moving into former grasslands on elevation or climate gradients (Breshears, 2006).

It has been projected that drying and greater seasonality, acting in conjunction with increased fire, could lead to more savanna-like conditions in parts of the Amazon basin (Malhi et al., 2009a; Box 4-3). In parts of Central Africa, forests have been observed to be moving into adjacent savannas and grasslands (Mitchard et al., 2009), possibly due to depopulation and fire changes. In northern Australia, forest is expanding into former savanna areas (Brook and Bowman, 2006; Tng et al., 2012; Bowman et al., 2011)

4.3.3.2.2. Grasslands and shrublands

Much evidence from around the world shows that dry grasslands and shrublands are highly responsive in terms of primary production, species composition and carbon balance to changes in water balance (precipitation and evaporative demand) within the range of projected climate changes (e.g., Sala et al., 1988; Snyman and Fouché, 1993; Fay et al., 2003; Peñuelas et al., 2004, 2007; Prieto et al., 2009; Wu and Chen, 2013) Other factors being equal, grasslands and shrublands are in cool areas are expected to respond to warming with increased primary production, while those in hot areas are expected to show decreased production. Grasslands are exposed to rising CO₂, which is postulated to stimulate plant production, but more so for temperate grasses, herbs and woody plants with a C3 photosynthetic system than for C4 tropical grasses (see section 4.2.4.4). Some grasslands are simultaneously exposed to elevated levels of nitrogen deposition, which alters species composition, increases primary production up to a point and decreases it thereafter (see section 4.2.4.2, Bobbink et al., 2010; Cleland and Harpole, 2010). Most rangelands are subject to grazing and fire. All these factors may have interactive effects, and species composition changes may overwhelm or alter the physiological climate effects (Suttle et al., 2007).

Drawing on published studies, Ravi et al. (2010) found that changing climate and land use have resulted in increased aridity and a higher frequency of droughts in drylands around the world. They predict increasing dominance of abiotic controls of land degradation (in contrast to direct human- or herbivore-driven degradation); in particular highlight changes in hydrology and the erosion of soil by wind.
Rainfall amount and timing have large effects on a wide range of biological processes in grasslands and shrublands, including seed germination, seedling establishment, plant growth, flowering time, root mass, community composition, population and community dynamics production, decomposition and respiration, microbial processes and carbon assimilation (e.g. Peñuelas et al., 2004, 2007; Beier et al., 2008; Sowerby et al., 2008; Albert et al., 2011; Albert et al., 2012; Miranda et al., 2009; Selsted et al., 2008; Walter et al., 2012). In a controlled experiment with communities of tallgrass prairie species in Kansas, Fay et al. (2008) showed that changes in the interval between rainfall events and the rainfall delivered by individual storms altered carbon assimilation and respiration rates independently of changes in the total amount of water supplied. A similar experiment on a Mediterranean semiarid grassland community (Miranda et al., 2009) found that delayed watering led to decreases in plant productivity and to delayed flowering. Species diversity was not affected by delayed onset of rain; however, it was reduced by changes in the frequency, amount and seasonality of wetting. In the Mongolian steppe (Shinoda et al., 2010), drought reduced the above ground plant mass but did not substantially affect the root mass. A subset of species did not recover to pre-drought levels once precipitation returned to normal levels. A water and temperature manipulation experiment in China indicated that changes in water balance had a stronger effect on grassland microbial processes leading to carbon storage than changes in temperature (Liu et al., 2009). Engler et al. (2011) emphasized that precipitation changes were as important for mountain flora in Europe as temperature changes, and the greatest composition changes will probably occur when decreased precipitation accompanies warming. Vohland and Barry (2009) reviewed the effectiveness of in situ rainwater harvesting (RWH) as an adaptation to climate change in sub-Saharan Africa and concluded that it conferred higher food security and income, but may strengthen conflicts between nomadic and sedentary populations.

Experimental manipulation of six European shrublands on a latitudinal gradient showed the response to warming and drought to depend on site, year and plant species (Peñuelas et al., 2004, 2007). Droughts decreased aboveground net primary production two thirds of the sites. Responses to warming were weaker in the presence of aridity. Gao and Giorgi (2008) identified the southern Mediterranean as particularly vulnerable to water stress and desertification processes under climate change conditions. Analyzing 168 site-years of eddy covariance measurements, Wu and Chen (2013) found summer drought to reduce both summer gross primary production and net ecosystem production in grasslands and crops of North America.

A European gradient study (Emmett et al., 2004), showed that N mineralization in shrublands under either arid or wet conditions is more sensitive to periodic droughts than systems under more mesic conditions. An increase in drought frequency, without an increase in drought severity, leads to loss of soil carbon in moist, carbon-rich moorlands, due to changes in soil structure or soil microbial community leading to increased hydrophobicity and soil respiration (Sowerby et al., 2008, 2010). Responses of shrublands to drought may partly be driven by changes in the soil microbial community (Jensen et al., 2003) or changes in soil fauna (Maraldo et al., 2008).

Beier et al. (2008) found that nitrogen limitation reduces acclimation in plant production, as a result of an asymmetrical response to warming between the carbon and nitrogen cycles. Decreased tissue concentrations of phosphorus was also associated with warming and drought (Peñuelas et al., 2004; Beier et al., 2008; Peñuelas et al., 2012). strong interactions of warming with disturbances have been observed leading to increased nitrogen leaching from shrubland ecosystems (Beier et al., 2004).

N fertilization and nighttime warming caused soil and plant community responses within one season in a Chihuahuan semi-arid desert (Collins et al., 2010). In contrast, Grime et al. (2008) found little species composition change after 13 years of manipulation of water and temperature in an infertile grassland in England. The community composition of subalpine grasslands in Switzerland have changed relatively little over the past fifty years (Vittoz et al., 2009). The changes that did occur were largely associated with changes in land management rather than climate, although they were consistent with warming trends.

Sommer et al. (2010) used global empirical relationships between plant species richness and climate variables such as temperature, moisture and radiation to project richness changes under future climates. This method is independent of the usual species-by-species or biome-based climate niche approach, but its predictions apply to the long-term equilibrium outcome, rather than changes in the next few decades. In most temperate and arctic regions, the capacity...
to support richer (i.e. more diverse) communities increases with rising temperature, while the projections indicate a
strong decline in capacity to support species-rich communities in most tropical and subtropical regions.

Many grasslands, shrublands and savannas are characterized by relatively frequent but low-intensity fires.
The fire frequency, intensity and seasonality in southern Africa are primarily under climate control, and secondarily
influenced by management (Archibald et al., 2009). In South America, El Niño-Southern Oscillation climate modes
strongly influence on area burned in Colombian grasslands (Armenteras-Pascual et al., 2011). Fire frequency in
grasslands and forests in Australia is projected to increase, due to climate change (Pitman et al., 2007). Since it is
well-established that plant species composition is sensitive to the fire regime (e.g. Gibson and Hulbert, 1987; Uys et
al., 2004; Gill et al., 1999 and many other studies) this provides an indirect mechanism by which climate change can
lead to species composition change in grasslands and rangelands.

Relatively little is known regarding the combined effect of climate change, other global change factors and
increasing use pressure by large mammal herbivores in rangelands, nor on the consequences for pastoral livelihoods
that depend on rangelands (Thornton et al., 2009).

4.3.3.2.3. Deserts

The deserts of the world, defined as land areas with an arid or hyperarid climate regime, occupy 35 % of the global
land surface. Deserts are sparsely populated, but the people who do live there are amongst the poorest in the world
(Millennium Ecosystem Assessment, 2005b). There is medium agreement but limited evidence that the present
extent of deserts will increase in the coming decades, despite the projected increase in rainfall at a global scale, as a
result of the strengthening of the Hadley circulation. The descending air on the poleward side of the Hadley
circulation results in decreased rainfall (Mitas and Clement, 2005; Seidel et al., 2008; Johanson and Fu, 2009; Lu et
al., 2009; Zhou et al., 2011) and which determines the location of the broad band of warm deserts lands
approximately 15-30° N and S of the equator. Recent satellite data show a greening trend in the Sahel, associated
with a return to wetter climates following a multidecade dry spell. There may be a feedback to the global climate
from an increase in desert extent (Alkama et al., 2012), which differs in sign between deserts closer to the equator
than 20° and those closer to the poles. In model simulations, extension of the ‘warm deserts’ causes warming, while
extension of the ‘cold deserts’ causes cooling, in both cases largely through albedo-mediated effects. Two special
circumstances of desert areas are important with respect to climate change. Deserts are expected to become warmer
and drier at faster rates than other terrestrial regions (Lapola et al., 2009). Most deserts are already extremely hot,
and therefore further warming will probably be physiologically injurious rather than beneficial. Secondly, the
ecological dynamics in deserts are rainfall event-driven (Holmgren et al., 2006), often involving the concatenation
of a number of quasi-independent events. According to a conceptual model elaborated by Lapola et al. (2009) some
desert tolerance mechanisms (e.g., biological adaptations by long-lived taxa) may be outpaced by global climate
change.

Deserts are expected to become warmer and drier at a faster rate than other terrestrial regions (Stahlschmidt et al.,
2011). The warm deserts are already extremely hot, and therefore further warming is generally physiologically
injurious rather than beneficial. Ecological dynamics in deserts are rainfall event-driven (Holmgren et al., 2006),
often involving the concatenation of a number of quasi-independent events. According to a conceptual model
elaborated by Stahlschmidt et al. (2011) some desert tolerance mechanisms (e.g., biological adaptations by long-
lived taxa) may be outpaced by global climate change.

4.3.3.3. Rivers, Lakes, Wetlands and Peatlands

Freshwater ecosystems are considered to be among the most threatened on the planet (Dudgeon et al., 2006;
Vörösmarty et al., 2010). Fragmentation of rivers by dams and the alteration of natural flow regimes have led to
major impacts on freshwater biota (Pringle, 2001, Bunn and Arthington, 2002; Nilsson et al., 2005). Floodplains and
wetland areas have become occupied for intensive urban and agricultural land use to the extent that many are
functionally extinct from their rivers (Tockner et al., 2008). Pollution from cities and agriculture, especially nutrient
loading, has resulted in declines in water quality and the loss of essential ecosystem services (Allan, 2004). As a
direct consequence of these and other impacts, freshwater systems have the highest rates of extinction of any
ecosystem (as much as 4% pa for some groups - e.g. crayfish, mussels, fishes and amphibians in North America),
with estimates that at least 10,000-20,000 freshwater species are extinct or at risk (Strayer and Dudgeon, 2010). This
is particularly significant considering that freshwater habitats support 6% of all described species, including
approximately 40% of fish diversity and a third of the vertebrate diversity (Dudgeon et al., 2006; Balian et al.,
2008).

Peatlands contain large stocks of carbon that are vulnerable to change through land use and climate change.
Although peatlands cover only about 3% of the land surface, they hold the equivalent of half of the atmosphere’s
carbon (as CO2), or one third of the world’s soil carbon stock (Limpens et al., 2008; Page et al., 2011). About 14-
20% of the world’s peatlands are currently used for agriculture (Oleszczuk et al., 2008) and many, particularly peat
swamp forests in Southeast Asia, are still undergoing major transformations through drainage and burning in
preparation for oil palm and other crops or through unintentional burning (Limpens et al., 2008; Hooijer et al.,
2010). Deforestation, drainage, and burning in Indonesian peat swamp forests can release 1,400 Mg CO2 ha\(^{-1}\)
(Murdiyarso et al., 2010), contributing significantly to global GHG emissions, especially during periods of intense
drought associated with ENSO when burning is more common (Page et al., 2002). Anthropogenic disturbance has
changed peatlands from being a weak global carbon sink to a source (Frolking et al., 2011) with large inter-annual
variability.

It is very likely that these stressors to freshwater ecosystems and peatlands will continue to dominate as human
demand for water resources grows, accompanied by increased urbanization and intensification of agriculture
(Vörösmarty et al., 2000; Malmqvist et al., 2007; Dise, 2009). However, climate change will have significant
additional impacts on freshwater ecosystems (high confidence), from altered thermal regimes, altered precipitation
and flow regimes and, in the case of coastal wetlands, sea level rise. Specific aquatic habitats that are most
vulnerable to these direct climate effects, especially rising temperatures, are those at high altitude and high latitude,
including arctic and subarctic bog communities on permafrost, and alpine and arctic streams and lakes (see 4.3.4.4;
Smith et al., 2005; Smol and Douglas, 2007a). It is noteworthy that these high latitude systems currently experience
a relatively low level of threat from other human activities (Vörösmarty et al., 2010). Dryland rivers and wetlands,
many already experiencing severe water stress from human consumptive use, are also likely to be further impacted
by decreased and more variable precipitation and higher temperatures. Headwater stream systems in general are also
vulnerable to the effects of warming because their temperature regimes closely track air temperatures (Caissie,
2006).

Evidence of rising stream and river temperatures over the past few decades across several continents continues to
grow (Chessman, 2009; Kaushal et al., 2010; Langan et al., 2001; Morrison et al., 2002; Ormerod, 2009; van Vliet
et al., 2011; Webb and Nobilis, 2007), and has been linked by observational and experimental studies to shifts in
invertebrate community composition, including declines in cold stenothermic species (Brown et al., 2007;
Chessman, 2009; Durance and Ormerod, 2007; Ormerod, 2009). Rising water temperatures are also implicated in
changes in the composition of river fish communities (Buisson et al., 2008; Daufresne and Boet, 2007), especially in
headwater streams where species are more sensitive to warming (e.g. Buisson and Grenouillet, 2009), and this will
probably restrict the distribution of cool-water species such as salmonids within their current range (Bartholow,
2005; Bryant, 2009; Ficke et al., 2007; Hague et al., 2011). While these changes in river temperature regimes may
also open up new habitat at higher latitudes (or altitudes) for migratory (Reist et al., 2006) and cool- and warm water
species of fish (Tisseuil et al., 2012), there is high confidence that range contraction threatens the long term
persistence of some fully aquatic species, including mountain-top endemics; e.g. species of spiny crayfish
(Parastacidae: Eustacus) in eastern Australia (Ponniha and Hughes, 2004).

The surface and epilimnetic waters in many lakes also show increases in temperature over the past four decades,
with warming trends detected in North America, Eurasia and Africa (Adrian et al., 2009). Rising temperatures
resulting in early onset and increased duration and stability of the thermocline in temperate lakes during summer
(Winder and Schindler, 2004) are projected to favor a shift in dominance to smaller phytoplankton (Parker et al.,
2008; Yvon-Durocher et al., 2011) and cyanobacteria (Johnk et al., 2008; Paerl et al., 2011; Wiedner et al., 2007),
especially in those ecosystems experiencing high anthropogenic loading of nutrients (Wagner and Adrian, 2009);
with impacts to water quality, food webs and productivity (Gyllström et al., 2005; O'Reilly et al., 2003; Parker et al., 2008; Shimoda et al., 2011; Verburg et al., 2003). Emergent aquatic macrophytes are likely to expand their northward distribution and percentage cover in boreal lakes and wetlands, posing an increasing overgrowth risk for sensitive macrophyte species (Alahatti et al., 2011). Long-term shifts in macroinvertebrate communities have also been observed in European lakes where temperatures have increased (Burgmer et al., 2007), noting that warming may increase species richness in smaller temperate waterbodies, especially those at high altitude (Rosset et al., 2010). While less studied, it has been proposed that tropical ectotherms will be particularly vulnerable because they will approach critical maximum temperatures proportionately faster than species in high latitude environments, despite lower rates of warming (Deutsch et al., 2008; Hamilton, 2010; Laurance et al., 2011).

There is growing evidence that climate induced changes in precipitation will significantly alter ecologically important attributes of hydrologic regimes in rivers and wetlands, and exacerbate impacts from human water use in developed river basins (see Box CC-RF; Aldous et al., 2011; Xenopoulos et al., 2005). Freshwater ecosystems in Mediterranean-montane ecoregions (e.g. Australia, California and South Africa), are projected to experience a shortened wet season and prolonged, warmer summer season (Klausmeyer and Shaw, 2009), increasing the vulnerability of fish communities to drought (Magalhães et al., 2007; Hermoso and Clavero, 2011) and floods (Meyers et al., 2010). Shifts in hydrologic regimes in snow-melt systems, including earlier runoff and declining base flows in summer (Stewart et al., 2005; Stewart, 2009), are projected to impact on freshwater ecosystems, through changes in physical habitat and water quality (Bryant, 2009). Declining rainfall and increased inter-annual variability will probably increase low-flow and dry-spell duration in dryland regions, leading to reduced water quality in remnant pools (Dahm et al., 2003), reduction in floodplain egg- and seed-banks (Capon, 2007; Jenkins and Boulton, 2007), the loss of permanent aquatic refugia for fully aquatic species and water birds (Bond et al., 2008; Johnson et al., 2005; Sheldon et al., 2010), and terrestrialization of wetlands (Davis and Thompson, 2010).

Climate induced changes in precipitation will probably be an important factor altering peatland vegetation in temperate and boreal regions, with decreasing wetness during the growing season generally associated with a shift from a Sphagnum dominated to vascular plant dominated vegetation type and a general decline of C sequestration in the longer term (Limpens et al., 2008). Mire ecosystems (i.e. bogs, transition bogs and fens) in central Europe face severe climate risk, with increased summer temperatures being particularly important (Essl et al., 2012). Declines in precipitation and longer dry seasons in major tropical peatland areas in southeast Asia, are projected to result in lower water tables more often and for longer periods, with an increased risk of fire (Li et al., 2007; Rieley et al., 2008; Frolking et al., 2011).

Sea level rise will lead to direct losses of coastal wetlands with associated impacts on waterbirds and other wildlife species dependent on fresh water (BMT WBM, 2010; Pearlstine et al., 2010; Traill et al., 2010) but the impact will probably be relatively small compared with the degree of direct and indirect human-induced destruction (Nicholls, 2004). River deltas and associated wetlands are particularly vulnerable to rising sea level, and this threat is further compounded by trapping of sediment in reservoirs upstream and subsidence from removal of oil, gas and water (Svyitski et al., 2009; see Chapter 5.3.1.4). Lower river flows might exacerbate the impact of sea level rise and thus salinization on freshwater ecosystems close to the ocean (Ficke et al., 2007).

4.3.3.4. Tundra, Alpine, Permafrost Systems

The region of the High Arctic, and its tundra-dominated landscapes, has warmed more than the global average over the last century (Kaufman et al., 2009; Hartmann et al., in press. Changes consistent with warming are also evident in the freshwater systems, ecosystems (both plants and animals) and permafrost of the region (Hinzman et al., 2005; Axford et al., 2009; Jia et al., 2009b; Post et al., 2009; Prowse and Brown, 2010; Romanovsky et al., 2010; Walker et al., 2012). Change is widespread, with most of the Arctic experiencing recent change in vegetation photosynthetic capacity and other aspects, particularly in areas adjacent to areas of the Arctic with rapidly retreating sea ice (Bhatt et al., 2010).

There is high confidence that continued climate change is projected to cause the terrestrial vegetation and lake systems of the Arctic to change substantially in the future, with an ongoing expansion in woody vegetation cover
projected in tundra regions over the 21st Century in the CMIP5 Earth System Models (Ciais et al., in press) by
dynamic global vegetation models driven by other climate model projections, and by observationally-based
statistical models (Pearson et al., 2013). Changes may be complex (see Box 4-4) and in some cases involve non-
linear and threshold responses to warming and other climatic change (Hinzman et al., 2005; Mueller et al., 2009b;
Bonfils et al., 2012). Due to long response times of vegetation to both warming and increased CO₂ (Ciais et al., in
press; Falloon et al., 2012) Earth System Models project Arctic vegetation change to continue long after any
stabilization of global mean temperature. In some regions, reduced surface albedo due to increased vegetation cover
is projected to cause further local warming even in scenarios of stabilized global radiative forcing due to greenhouse
gases (Falloon et al., 2012).

In the arctic tundra biome, vegetation productivity has systematically increased over the past few decades, as
documented across a range of scales in both North America and northern Eurasia – from the plot level (Myers-Smith
et al., 2011; Elmdorf et al., 2012) to the region (Stow et al., 2007) to continental (Jia et al., 2009b) and the
circumpolar arctic (Goetz et al., 2007; de Jong et al., 2011). This phenomenon is amplified by retreat of coastal sea
ice (Bhatt et al., 2010) and has been widely discussed in the context of increased shrub growth consistent with
documented shrub expansion over the last half century through repeat photography and satellite observations
(Forbes et al., 2010; Myers-Smith et al., 2011). Deciduous shrubs (i.e., dwarf birch, alder and willow species), in
particular, respond to warming with increased growth (Euskirchen et al., 2009; Lantz et al., 2010), but this response
is shared with other plant functional types, particularly graminoids (Walker, 2006; Epstein, 2008). Analyses of
satellite time series data show the increased productivity trend is not unique to shrub-dominated areas (Jia et al.,
2009b; Beck and Goetz, 2011), thus greening is a response shared by multiple vegetation communities and
continued changes in the tundra biome can be expected irrespective of shrub presence. Shrub expansion and height
changes remain particularly important, however, because they trap snow, mediate winter soil temperature and
summer moisture regimes, increase nutrient mineralization, and produce a positive feedback for additional shrub
growth (Sturm et al., 2005; Lawrence et al., 2007; Bonfils et al., 2012). Although increased shrub cover and height
produce shadowing that reduce ground heat flux and active layer depth, they also reduce surface albedo, increase
energy absorption and evapotranspiration (Chapin III et al., 2005; Blok et al., 2010), and produce feedbacks that
reinforce shrub densification and regional warming (Lawrence and Swenson, 2011; Bonfils et al., 2012). On
balance, these feedbacks can act to partially offset one another, but when coupled with warmer and wetter conditions
they act to increase active layer depth and permafrost thaw (Yi et al., 2007; Bonfils et al., 2012).

The arctic tundra biome is also experiencing shifts in two additional phenomena; fire disturbance and permafrost
degradation. Both of these processes facilitate conditions for woody species establishment in tundra areas, either
through incremental migration or via more rapid “leap-froging” to areas reinitialized by burning (Epstein et al.,
2007; Goetz et al., 2011). When already present at the boreal-tundra ecotone, even sparsely, shrub and tree species
show increased productivity with warmer conditions (Devi et al., 2008; Andreu-Hayles et al., 2011; Elmdorf et
al., 2012). Tundra fires not only emit large quantities of combusted carbon formerly stored in vegetation and organic
soils (Mack et al., 2011; Rocha and Shaver, 2011) but also increase active layer depth during summer months
(Racine et al., 2004; Liljedahl et al., 2007) and produce thermokarsting associated with thawing of ice-rich
permafrost. There is tremendous variability in the degree of thermokarsting associated with fire disturbance,
depending upon the substrate and ice content (Jorgenson et al., 2010) but the combination of thermokarsting and
seasonal cryoturbation create conditions that alter vegetation succession (Racine et al., 2004; Lantz et al., 2009;
Higuera et al., 2011).

There is high confidence that the area of permafrost is projected to continue to decline over the first half of the 21st
Century in all emissions scenarios (Figure 4-11). In the RCP2.6 scenario of an early stabilization of CO₂
concentrations, permafrost area is projected to stabilize at a level approximately 20% below the 20th Century area,
and then begin a slight recovering trend. In RCP4.5, in which CO₂ concentration is stabilized at approximately
550ppmv by the mid-21st Century, permafrost continues to decline for at least another 250 years. In the RCP8.5
scenario of ongoing CO₂ rise, permafrost area is simulated to approach zero by the middle of the 22nd Century
(Figure 4-11).
Frozen soils and permafrost currently hold more than twice the carbon than the atmosphere and thus represent a particularly large vulnerability to climate change (i.e., warming). Although the Arctic is currently a net carbon sink, continued warming will act to soon turn the Arctic to a net carbon source, that will in turn create a potentially strong positive feedback to accelerate Arctic (and global) warming with additional releases of CO$_2$, CH$_4$, and perhaps N$_2$O, from the terrestrial biosphere into the atmosphere (Schuur et al., 2008; Schuur et al., 2009; Maslin et al., 2010; McGuire et al., 2010; O'Connor et al., 2010; Schaefer et al., 2011). Moreover, this feedback is already accelerating due to climate-induced increases in fire (McGuire et al., 2010; O'Donnell et al., 2011). The rapid retreat of snowcover, and resulting spread of shrubs and trees into areas currently dominated by tundra has also already begun, and will continue to serve as a positive feedback accelerating high latitude warming (Chapin III et al., 2005; Bonfils et al., 2012).

There is medium confidence that rapid change in the Arctic is also affecting its animals. For example, seven of 19 sub-populations of the polar bear are declining in number, while four are stable, one is increasing and the remaining seven have insufficient data to identify a trend (Vongraven and Richardson, 2011). Declines of two of the sub-populations are linked to reductions in sea ice (Vongraven and Richardson, 2011). Polar bear populations are projected to decline greatly in response to continued Arctic warming (Hunter et al., 2010), and it is expected that the populations of other Arctic animals (e.g., fox and caribou) will be affected dramatically by climate change (Post et al., 2009; Sharma et al., 2009). Simple niche-based, or climatic envelope models have difficulty in capturing the full complexity of these future changes (MacDonald, 2010).

There is high confidence that alpine systems are already showing a high sensitivity to on-going climate change and will be highly vulnerable to change in the future. In western North America, warming, glacier retreat, snowpack decline and drying of soils is already causing a large increase in mountain forest mortality, wildfire and other ecosystem impacts (e.g., Westerling et al., 2006; Kelly and Goulden, 2008; Crimmins et al., 2009; van Mantgem et al., 2009; Pederson et al., 2010; Muhlfeld et al., 2011; Schwikl and Keeley, 2012; Park Williams et al., 2012), and disturbance will continue to be an important agent of climate-induced change in this region (Littell et al., 2010). Alpine ecosystems already appear to already be changing in response to climate change in Africa, Tibet, the Alps, the tropics and elsewhere (Cannone et al., 2007, 2008; Lenoir et al., 2008, 2010; Cui and Graf, 2009; Britton et al., 2009; Normand et al., 2009; Chen et al., 2009, 2011; Allen et al., 2010a; Eggermont et al., 2010; Kudo et al., 2011; Laurance et al., 2011; Engler et al., 2011; Dullinger et al., 2012). For example, in a study of permanent plots from 1994 to 2004 in the Austrian high Alps, a range contraction of subnival to nival vascular plant species was indicated at the downslope edge, and an expansion of alpine pioneer species at the upslope edge (Pauli et al., 2007). Thermophilous vascular plant species were observed to colonize in alpine mountain-top vegetation across Europe during the past decade (Gottfried et al., 2012). As with the Arctic, permafrost thawing in alpine systems could provide a strong positive feedback in those systems (e.g., Tibet; Cui and Graf, 2009).

Box 4-4. Boreal – Tundra Biome Shift

Changes in a suite of ecological processes currently underway across the broader arctic region are consistent with earth system model predictions of climate-induced geographic shifts in the range extent and functioning of the tundra and boreal forest biomes (Figure 4-12). Thus far these changes appear to be not so much threshold responses as gradual and systematic shifts across temperature and moisture gradients. Responses are expressed through gross and net primary production, microbial respiration, fire and insect disturbance, vegetation composition, species range expansion and contraction, surface energy balance and hydrology, active layer depth and permafrost thaw, and a range of other inter-related variables. Because the high northern latitudes are warming more rapidly than other parts of the earth, due at least in part to arctic amplification (Serreze and Francis, 2006), the rate of change in these ecological processes are sufficiently rapid that they can be documented in situ (Hinzman et al., 2005; Post et al.,...
Gradual changes in composition resulting from decreased evergreen conifer productivity and increased mortality, as well as increased deciduous species productivity, are facilitated by more rapid shifts associated with fire disturbance (Mack et al., 2008; Johnstone et al., 2010). Each of these interacting processes, as well as insect disturbance and associated tree mortality, are tightly coupled with warming-induced drought (Ma et al., 2012; Anderegg et al., 2013a; Choat et al., 2012). Similarly, gradual productivity increases at the boreal-tundra ecotone are facilitated by leap-frog dispersal into areas disturbed by tundra fire and thermokarsting (Tchebakova et al., 2009; Hampe, 2011; Brown, 2010). In North America these coupled interactions set the stage for changes in ecological processes, already documented, consistent with a biome shift characterized by increased deciduous composition in the interior boreal forest and evergreen conifer migration into tundra areas that are, at the same time, experiencing increased shrub densification. The net feedback of these ecological changes to climate is multi-faceted, complex, and not yet well known across large regions except via modeling studies, which are often poorly constrained by observations.

Figure 4-12: Tundra-Boreal Biome Shift. Earth system models predict a northward shift of Arctic vegetation with climate warming, as the boreal biome migrates poleward into what is currently tundra. Observations of shrub expansion in tundra, increased tree growth at the tundra-forest transition, and tree mortality at the southern extent of the boreal forest in recent decades are consistent with model projections. Vegetation changes associated with a biome shift, which is facilitated by intensification of the fire regime, will modify surface energy budgets and net ecosystem carbon balance. Arrows indicate the magnitude of albedo and net ecosystem carbon balance (NECB) for boreal forest and tundra biomes, and albedo for sea ice and open water.

4.3.3.5. Highly Human-Modified Systems - Plantations, Bioenergy Systems, Urban Ecosystems, Cultural Landscapes

As the majority of ice-free land surface is dominated by highly human modified systems, we assess their vulnerability to climate change particularly for those systems not dealt with elsewhere, i.e. except agriculture (WGII, chapter 7) and fisheries (WGII, chapter 3).

4.3.3.5.1. Plantation forestry

While a majority of general aspects on forests are already dealt with in section 4.3.3.1, here we focus on issues that are specifically relevant to plantation forests. Forest plantations are established through afforestation of recent non-forest land and reforestation of forest land, often with tree crop replacement (Dohrenbusch and Bolte, 2007; FAO, 2010b). With 7%, they currently cover only a minor fraction of the global forest area, and the largest plantation areas exist in Asia, Europe (excl. Russian Federation) and North and South America (FAO, 2010a). However, the area of forest plantations has grown rapidly by about five million ha from 2000 to 2010 (FAO, 2010b). Most recent plantations have been established in the tropics and subtropics, but also in some temperate regions, in particular in China (Kirilenko and Sedjo, 2007; FAO, 2010a), mostly by afforestation of non-forest area (FAO, 2010b).

In most areas with forest plantations, forest growth rates have generally increased during the last decades - in Europe also because of formally more intensive harvesting (Ciais et al., 2008; Lindner et al., 2010), but the variability is large, and in some areas production has decreased (see 4.3.3.1). In forests that are not highly water-limited, these trends are consistent with higher temperatures and extended growing seasons, but, as in the case of forests in general, clear attribution is difficult because many environmental drivers and changes in forest management interact (e.g. Boisvenue and Running, 2006; Ciais et al., 2008; Dale et al., 2010; see also 4.3.3.1).
Regarding future climatically driven changes, results from several studies with forest yield models suggest increases in forest production (Kirilenko and Sedjo, 2007), but these results may be overly optimistic because many models may overestimate potential positive effects of elevated CO₂ (Kirilenko and Sedjo, 2007; see 4.2.4.4) and the effects of disturbances, such as wildfires, forest pests, pathogens and windstorms, which are major drivers of forest dynamics, are often either poorly or not at all represented (Kirilenko and Sedjo, 2007). Using a model that accounts for fire effects and insect damage, Kurz et al. (2008), for example, showed that recent insect outbreaks might have caused a transition in the Canadian forest sector from a sink to a source of carbon. Future projections for particular stands or regions are also uncertain because results from different models often differ substantially, both regarding forest productivity (e.g. Sitch et al., 2008; Keenan et al., 2011) and potential species ranges (see 4.3.3.1.2).

Nevertheless, decreased production is expected in particular in already dry forest regions for which further drying is projected, such as the south-western U.S. (Williams et al., 2010a), and extreme drying might also decrease forest yields in currently not water-limited forests (e.g. Sitch et al., 2008; see 4.3.3.1). Plantations in cold-limited areas could benefit from climate change and their productivity could increase if associated changes in disturbances, pests and pathogens do not outweigh the potential positive climatic effects.

Many plantation forests are monospecies stands or sometimes even include only a limited number of clones of one species. In the temperate and boreal zone, native species are commonly used (but in some cases beyond their native range), while in the tropics, conventional tree planting is mainly based on exotic species from a few genera such as Pinus, Eucalyptus and Acacia grown in single-species stands. Low species (and often also genetic) diversity compared with natural stands might render plantation forests particularly vulnerable to climate change (e.g. Hemery, 2008). Choosing provenances that are well adapted to current and future climates is extremely difficult because of uncertainties in climate projections (Broadmeadow et al., 2005). Furthermore, it is highly uncertain how forest pests and pathogens will spread as a result of climate change and trade and new pathogen-tree interactions might occur (e.g. Brasier and Webber, 2010). Nevertheless, adaptive forest management can decrease the vulnerability of plantations to climate change (Hemery, 2008; Bolte et al., 2009; Seppälä, 2009; Dale et al., 2010. For example, risk spreading by promoting multi-species mixed stands and natural regeneration, which can increase genetic diversity (Kramer et al., 2010), has been advocated as a plausible adaptation strategy for temperate forests (Hemery, 2008; Bolte et al., 2010). Also in the tropics, recent approaches highlight the use of native species in mixed stands (Erskine et al., 2006; Petit and Montagnini, 2006; Hall et al., 2011), but missing information on the ecology of many of the tropical tree species and little experience in managing mixed tropical tree plantations remains to be a major problem (Hall et al., 2011). At least at the southern border of cold-adapted species, such as Norway spruce (Picea abies) in Europe, climate change will very likely lead to a shift in the main tree species used for forest plantations (Iverson et al., 2008; Bolte et al., 2010).

4.3.3.5.2. Bioenergy systems

Bioenergy sources include traditional forms such as wood and charcoal and more modern forms such as the industrial burning of biomass wastes, the production of ethanol and biodiesel. While traditional biofuels have been in general decline as users switch to fossil fuels or electricity, they remain dominant energy sources in many less-developed parts of the world, such as Africa, and retain a niche in developed countries. The production of modern bioenergy is growing rapidly throughout the world in response to climate mitigation policies that incentivise their use, or as a strategy to decrease oil dependence and thus increase energy security (Coehrane and Barber, 2009). The WG III chapter on energy addresses their potential as a climate mitigation strategy, while the sensitivity of biofuel crops to climate change should be quite similar to those previously mentioned for plantation forestry (which to a large extent are grown for bioenergy purposes) and/or agricultural systems (WGII, chapter 7). In a review on climate change impacts on biofuel yields in temperate environments (Luckman and Kavanagh, 2000) it was concluded that elevated CO₂ might contribute to increase drought tolerance of bioenergy crops (as it is paralleled by improved plant water use), which may lead to constantly high yields. Generally, potentials of bioenergy production under climate change might be high, but are very uncertain (Ma and Zhou, 2012). An important part to deal with here is the ecosystem impacts of large-scale land-use changes related to the growing of bioenergy biofuels. Policy shifts in OECD countries favor the expansion of biofuel production, sometimes at the expense of food crop production, and placing new pressures on terrestrial and freshwater ecosystems (Searchinger et
It is, for example, unclear if the global trend to reduced forest loss will continue - there are substantial pressures to deforest for the production of food and biofuels (Wise et al., 2009; Meyfroidt and Lambin, 2011). Moiseyev et al. (2011) found that in a 20 years perspective it is very little difference between the IPCC scenarios A1 and B2 regarding harvest level in Europe, and that the EU RES policy may only moderately influence the EU forestry and forest industries as long as the wood price paid by the bioenergy producers are below 50-60 US$ per m³ wood. Under the assumption of doubling the growth rate of demand for bioenergy until 2030, a scenario by Martin et al. (2010) would lead to severe consequences for the global forest sector with a global reduction of forest stocks of 2% or a 4% reduction for Asia.

Bioenergy potentials are strongly influenced by human food requirements (incl. feed required for livestock), thus integrated approaches to optimize food and bioenergy are needed (Ma and Zhou, 2012). Such considerations ignore areas for the conservation of biodiversity and ecosystems. These are more explicitly dealt with by McAlpine et al. (2009) and Millar et al. (2004), while in both papers the usage of abandoned land, which was previously under agricultural use, is regarded as an option for biomass production which reduces net warming. However, this ignores for example that such habitats may be core elements in cultural landscapes of high conservation value (many species rich grasslands in Europe have been croplands before and later abandoned), and that the productivity of such areas could be too limited for biofuels (Mishra et al., 2010). As an alternative, Fargione (2011) discusses biofuel crop yield increases on existing cropland, with the aim to avoid expansion of agriculture.

4.3.3.5.3. Cultural landscapes

“Cultural landscapes are at the interface between nature and culture, tangible and intangible heritage, biological and cultural diversity—they represent a closely woven net of relationships, the essence of culture and people’s identity” (Rössler, 2006, p. 334). They are characterized by a long history of human-nature interactions, where man is the main driver and has often created open landscapes, rich in structures and often also in species. A UNESCO Flagship programme focusses especially on cultural landscapes (Rössler, 2006). These landscapes nowadays are often about to lose their roles as recreational species rich entities due to agricultural intensification and extensification (Huntington et al., 2012). Well researched examples are grassland or mixed agriculture landscapes in Europe or rice landscapes in Asia (Kuldna et al., 2009), while such landscapes may well exist across the globe (e.g., Rössler, 2006; Heckenberger et al., 2007).

In such landscapes, conservation efforts - as all across the globe - often focus on the conservation of ecosystems that contain endangered biotic communities. However, in such cultural landscapes this aim might be hard to achieve due to the very dynamic nature of systems. Keeping species in a favorable conservation status in cultural landscapes (one aim e.g. of European policies; EU Council, 1992), can normally only be achieved through appropriate management, as the vast majority of endangered species in the wider countryside depend on certain types of land use for their survival. This requires profound knowledge of the systems and species involved, and as this is rarely the case, conservation success was limited (see Kirdyanov et al., 2012 for a notable exception).

As in many other cases, population and niche models are available and partly already validated (Kirdyanov et al., 2012). This shows where future challenges are to be found: particularly in the quantification of relative importance of climate change in comparison with the habitat and its management (Settele and Kühn, 2009). So far the majority of changes can be attributed to land use as the most obvious driver (Nowicki et al., 2007), while the impact of climate change can be readily detected in few examples (Devictor et al., 2012), combined effects seem to better explain the overall picture (Schweiger et al., 2008; Schweiger et al., 2012), where attribution to climate change seems low to medium. As a consequence, it has been suggested to adjust the European Natura 2000 protected area network to take into account changing climatic conditions and to enable migration or dispersal of species across the landscape (Heubes et al., 2011; Jin et al., 2010).
4.3.3.5.4. Urban ecosystems

Over half of humanity lives in urban areas (see for definition in chapter 8) with a yearly increase of ca. 74 million people (United Nations et al., 2012). Although urban areas cover only ca. 0.5 % of Earth’s terrestrial surface (Schneider et al., 2009) they harbor a large variety of species (McKinney, 2008). Urban areas are themselves drivers of climatic change as they are accounted for a significant proportion of total anthropogenic greenhouse gas emissions (Satterthwaite, 2008) and many urban centers are also heavily impacted by increased minimum and sometimes decreased maximum air temperatures, reduced or increased precipitation and altered biogeochemical cycles (Grimm et al., 2008). An important threat to cities comes from extreme events. Many cities are located at coasts or rivers. However since there is medium confidence (based on physical conditions) that projected increases in heavy rainfall events would contribute to increases in local flooding (IPCC, 2012) there is limited evidence that urban areas will be over-proportionally affected by floods in the future, however, it is very likely that sea level will contribute to rise in future and hence affect coastal urban areas (IPCC, 2012). For the future it is virtually certain that the frequency and magnitude of warm days and nights (heat waves) will increase globally (IPCC, 2012).

However Leonelli et al. (2011) projected with the integration of an urban land-surface model in the HadAM3 Global Climate Model a significant higher increase in the frequency of hot nights in urban areas compared to rural areas. Hence also the social aspects in urban areas (health status, outdoor activities) and urban infrastructures will be increasingly affected (IPCC, 2012). There is high agreement among scientists that urban climate effects (e.g., increased temperatures) nowadays are similar to projected changes of climatic variables at a local scale. Similar to all other ecosystems, ecological impacts of changing climatic factors in urban ecosystems will change species compositions as well as compositions of traits. Knapp et al. (2008b) found that traits compositions of urban plant communities are changing during urbanization towards adaptive characteristics of dry and warm environments. With increasing temperatures, this effect might to continue also in the future and will hence be exacerbated compared to non-urban surroundings. Urban areas are one of the main starting points for the establishment and dispersal of alien species (e.g. for plants through urban gardening; Li et al., 2009). With increasing air temperatures especially in winter time and the warming effect of urban areas, the number of established alien plants in urban areas might increase (see also 4.2.4.6.). With increasing numbers of alien species, also the BVOC (Biogenic Volatile Organic Compounds) emissions from gardened alien plants will increase (Lloyd and Fastie, 2003; see also 4.2.4.6.).

4.3.4. Impacts on Key Services

Ecosystem services are the benefits which people derive from ecosystems [see glossary]. The classification system proposed by the Millennium Ecosystem Assessment (Millennium Ecosystem Assessment, 2003) is widely used. It recognises provisioning services such as food, fiber and water (also known as ‘goods’ in the economic literature, and which have their own chapters in this assessment); regulating services such as climate regulation, pollination, pest and disease control and flood control; supporting services (habitat services in TEEB) which are used by people indirectly, through other services, and include items such as primary production and nutrient cycling; and cultural services which include recreation, aesthetic and spiritual benefits. The list of ecosystem services is long and growing; and almost all are potentially vulnerable to climate change. The provisioning services are comprehensively dealt with in chapter 3 and chapter 7, and the tourism-based services in chapter 13. This chapter focusses on those regulating services where the link to climate change has been examined.

4.3.4.1. Habitat for Biodiversity

Climate change can alter habitat for species by inducing i) shifts in habitat distribution that are not followed by species, ii) shifts in species distributions that move them outside of their preferred habitats and iii) changes in habitat quality (Urban et al., 2012; Dullinger et al., 2012). There is some evidence that these climate change impacts have already occurred. For example, loss of sea ice is altering habitats for polar bears and may partially explain declines in polar bear populations in some area (see also Chapter 28). However, this is not yet a widespread phenomenon. Models of future shifts in biome, vegetation type and species distributions, on the other hand, suggest that many species could be outside of their preferred habitats with the next few decades due to climate change (Urban et al., 2012).
Hole et al. (2009) report that the majority of African birds are projected to move large distances over the next 60 years resulting in substantial turnover of species within protected areas (>50% turnover in more than 40% of Important Bird Areas of Africa) and migration across unfavorable habitats. Many birds may find suitable climate in the large network of protected areas, but will be forced to cope with new habitat constraints. Similarly, Araujo et al., 2011) indicate that approximately 60% of plants and vertebrate species would no longer have favorable climates within European protected areas, often pushing them into unsuitable or less preferred habitats. Wiens et al. (2011) project similar effects in the western US, but also find that climate change may open up new opportunities for protecting species in areas where climate is currently unsuitable. Indeed, in some changes climate change may allow some species to move into areas of lower current or future land use pressure including protected areas (Bombard et al., 2005). These and other studies strongly argue for a rethinking of protected areas networks and of the importance of the habitat matrix outside of protected areas as a key to migration and long-term survival of species.

Over sufficiently long periods, biomes or habitat types may shift their distributions or disappear entirely due to climate change. Non-analog climates are projected to occur in the future (Williams et al., 2007b; Wiens et al., 2011), and in the past climate shifts have resulted in vegetation types that have no current analog (section 4.2.2). The impacts of this on species abundance and extinction risk is difficult to evaluate because species may show substantial capacity to adapt to novel habitats (Prugh et al., 2008; Willis and Bhagwat, 2009; Oliver et al., 2009). This high uncertainty in habitat specificity for many species means it is not possible to predict if species extinctions due to climate induced loss of habitat will be below or far above current extinction rates caused by non-climate factors (Malcolm et al., 2006). This does, however, also reinforce the idea that habitat quality across all components of landscapes will increase in importance for biodiversity conservation in the future.

Effects of climate change on habitat quality are less well studied than shifts in species or habitat distributions. However, several recent studies indicate that climate change may have and probably will alter habitat quality (Iverson et al., 2011; Matthews et al., 2011). For example, climate change induced changes in habitat quality due to decreasing snowfall may partially explain declining songbird populations in southwestern US (Martin and Maron, 2012).

4.3.4.2. Pollination, Pest and Disease Regulation

It can be inferred that global change will result in new communities (Schweiger et al., 2010). As these will have experienced a much shorter (or even no) period of coevolution, drastic changes of ecological interactions like shifts in the use of certain plants by herbivores, in the range of prey of predators or in pollination networks are to be expected (Tylianakis et al., 2008; Schweiger et al., 2012). This might generally result in drastic changes in the provision of services (Montoya and Raffaelli, 2010).

Among the regulating services most strongly related to biodiversity, pollination and biocontrol of pests have to be highlighted. Climate change tends to increase the abundance of pest species particularly in previously cooler climates, but assessments of changes in impacts are hard to make (Payette, 2007). Insect pests are directly influenced, e.g. through the quality of food plants (Payette and Filion, 1985) or via the effects on their natural enemies (predators and parasitoids). Direct impacts are via the relatively high temperature optima of insects, which lead to increased vitality and reproduction (Allen et al., 2010a). Mild winters in temperate areas promote frost susceptible pests. For the vast majority of indirect effects, e.g. spread of insect borne diseases, information is scarce (for further assessments on climate change effects on pest and disease dynamics see WGII, chapters 7 and 11).

Climate change, after land-use changes, can be regarded as the second most relevant factor responsible for the decline of pollinators (Potts et al., 2010; for other factors see Biesmeijer et al., 2006; Brittain et al., 2010a; Brittain et al., 2010b). While the potential influence of climate change on pollination can be manifold (compare Hegland et al., 2009; Roberts et al., 2011; Schweiger et al., 2010), there are only few observations which mostly relate to the de-coupling of plants and their pollinators – especially in relation to phenology (Gordo and Sanz, 2005). While Peñuelas and Boada (2003) states that these phenological effects may be less than feared, an analysis of
phonological observations in plants Wolkovich et al. (2012) shows that experimental data on phenology may grossly underestimate phenological shifts. As Willmer’s (2003) view is partly based upon experimental observations, it has to be seen whether field evidence might proof something different (compare Phenology Chapter 4.3.2.1.).

In relation to honeybees Le Conte and Navajas (2008) state that the generally observed decline is a clear indication for an increasing susceptibility against global change phenomena, with pesticide application, new diseases and stress (and a combination of these) as the most relevant causes. Climate change might contribute by modifying the balance between honeybees and their environment (incl. diseases). Honeybees also have shown a large capacity to adjust to a large variety of environments and their genetic variability should allow them to also cope with climatic change, that’s why the preservation of genetic variability within honeybees is regarded as a central aim to mitigate climate change impacts (Le Conte and Navajas, 2008).

4.3.4.3. Climate Regulation Services

Ecosystems moderate the local climate through a range of mechanisms, including reducing the near-ground wind velocity, cooling the air through shading and the evaporation of water and ameliorating low temperatures through releasing heat absorbed during the day or summer. This service is widely recognized and valued, for instance in the establishment of windbreaks, gardens and urban parks. The focus of this section is on processes operating at much larger scales – the region to the whole globe. Terrestrial ecosystems affect climate at large scales through their influence on the physical properties of the land surface and on the composition of the atmosphere with respect to radiatively-active gases and particles. In the decade 2000-2010, approximately one fourth of the CO₂ emitted to the atmosphere by human activities was taken up by terrestrial ecosystems, reducing the rate of climate change proportionately (4.3.2.3, WG 1 Chapter 6).

One study (Arora and Montenegro, 2011) suggests that the overall effect of tropical afforestation on global temperature is up to 75% greater than would be expected on the basis of increased carbon storage alone, due to physical processes such as changes in evapotranspiration; but in other circumstances the cooling due to land-cover change may be less than that estimated from carbon uptake alone, due to decreased albedo (4.2.4.1). Observations and model evidence indicates that, broadly speaking, forests make warm areas cooler and cold areas warmer, through the competing effects of evapotranspiration (cooling) and surface albedo (in dark-canopied forests, generally warming). Model simulations suggest that if more than 40% of the pre-1700 extent of the Amazon forest were to be cleared, rainfall in the region would be reduced (Sampaio et al., 2007). According to satellite observations, the effect of conversion of the Brazilian savannas (cerrado) to pasture is to induce a local warming, (Loarie et al., 2011), which is partly offset when the pasture is converted to sugarcane. It has been suggested (Ridgwell et al., 2009) that planting large areas of crop varieties with high-albedo leaves could help regional cooling. Model analysis indicates this strategy could be marginally effective at high latitudes, but have undesirable climate consequences at low latitudes, and measurements show that the current range of leaf albedo in major crops is insufficient to make a meaningful difference (Doughty et al., 2011).

4.3.4.4. Potable Water Supply

Freshwater systems provide a range of ecosystem services that benefit society and are likely to be affected by climate change (see Table 4-1; Palmer and Filoso, 2009). Many of the implications of climate change on the provisioning services of water for food production and domestic supply are discussed in detail in WGII Chapter 3. Over 80% of the world’s population is currently exposed to high levels of threat to water security because of land use change, pollution and water resource use (Vörösmarty et al., 2010). Under climate change, reliability of surface water supply is likely to decrease due to increased temporal variations of river flow that are caused by increased variability in precipitation and decreased snow/ice storage (see WGII Chapter 3). In some Mediterranean regions, e.g. the southwest of Western Australia, significant reductions in surface flows have already forced governments to pursue alternative water sources (desalination, recycled wastewater) (see http://www.watercorporation.com.au/watersupply/index.html). In snow dominated regions, warming will mean that less winter precipitation falls as snow and melting of winter snow occurs earlier in spring (high confidence; Barnett
et al., 2005). This means a shift in peak discharge to winter and early spring and away from summer and autumn when human demand is highest. In some locations, water storages may not be sufficient to capture winter runoff, particularly if they are also used to safeguard downstream communities from flooding (Barnett et al., 2005). Climate change not only poses risks to the quantity of water available for human society but also the quality. Warming will exacerbate many of the symptoms of eutrophication, including the increased frequency and intensity of harmful cyanobacterial blooms (Paerl and Paul, 2012). Reduced summer flows in rivers increases residence time for algal growth, increases the risk of low dissolved oxygen events and associated poor water quality, and limits the ability to dilute pollution from point sources (Jeppesen et al., 2010).

4.4. Adaptation and its Limits

Wild organisms and ecosystems can adapt to climate change to some degree using intrinsic mechanisms such as migration, change in behavior and evolution: this is known as autonomous adaptation. These mechanisms rely on the availability of suitable habitat and a clear pathway to it; climate change that is sufficiently gradual to allow response by all organisms; and the retention of genetic variability. There are also biological limits to the magnitude of change which can be adapted to; for instance the tolerable body temperature of all warm-blooded animals falls within a narrow range (Clarke and Rothery, 2008). Where autonomous adaptation mechanisms are inadequate, human-assisted adaption is indicated as a supplementary approach.

4.4.1. Autonomous Adaptation by Ecosystems and Wild Organisms

Autonomous adaptation (see glossary) refers to the adjustments made by ecosystems, including their human components, without external intervention, in response to a changing environment (Smit et al., 2000); also called ‘spontaneous adaptation’ (Smit et al., 2007). In the context of human systems it is sometimes referred to as ‘coping capacity’. The capacity for autonomous adaptation is necessary for resilience but is not synonymous with that term as it is now used (Walker et al., 2004). Here we focus on adaptation to a changing climate in all its manifestations, recognizing that climate change in the narrow sense is necessarily associated with changing CO₂ and ozone, and in practice is usually accompanied by changing land use, abundance of competing or mutualistic organisms and other environmental stressors, such as pollution.

All systems have some capacity for autonomous adaptation, or they would not exist at all. Ecosystems which have persisted for a long time can reasonably be inferred to have a high capacity for autonomous adaptation, at least with respect to the variability which they have experienced in the past. Adaptability to one set of challenges does not automatically confer greater adaptive capacity to a different set of challenges, especially if those challenges are novel (i.e. outside the range of experience of the system). Furthermore, an environmental change that is more rapid than in the past or is accompanied by other stresses may exceed the previously-demonstrated adaptive capacity of the system. Adaptation at one level, for instance by organisms in a community, can confer greater resilience at higher organization levels, such as the ecosystem (Morecroft et al., 2012). The mechanisms of autonomous adaptation of organisms and ecosystems consist of changes in the physiology, behavior, phenology or physical form of organisms, within the range permitted by their genes and the variety of genes in the population; changes in the genetic composition of the populations; and change in the composition of the community, either through in- or out-migration, or local extinction.

4.4.1.1. Phenological and Physiological

The ability to project future impacts of climate change on ecosystems is complicated by the potential for adaptations by species, which may alter apparent relationships between climate variables and species / ecosystem viability. Adaptation by individual species may increase their ability to survive and flourish under different climatic conditions, possibly leading to lower risks of extinction than predicted from statistical correlations between current distribution and climate (Botkin et al., 2007), but may also affect their interdependence with other species leading to disruption of community interactions (Visser and Both, 2005).
Phenology is typically highly adapted to the climate seasonality of the environment in which the organism evolved. The phenology of any species also needs to be keyed to the phenology of other species with which it interacts, such as competitors, food species and pollinators. Thus change in phenology is a sensitive indicator of adaptation (Menzel et al., 2006), but also a potential mechanism of ecosystem disruption if adaptation is insufficiently rapid or coordinated between interdependent species, or is cued to environmental signals that are not affected by climate change, such as day-length (Bradley et al., 1999; Both and Visser, 2001; Parmesan, 2006). The environmental cues for phenological events are complex and multi-layered (Körner and Basler, 2010; Singer and Parmesan, 2010). For instance, late-succession temperate trees require a chilling period in winter, followed by a threshold in daylength, and only then are sensitive to temperature. As a result, projecting current phenological trends forward is risky, since the relative importance of cues can change (Cook et al., 2012). The effects are complex and sometimes apparently counterintuitive, such as the increased sensitivity of flowering in high-altitude perennial herbs in the Rocky mountains to frost as a result of earlier snowmelt (Inouye, 2008).

The importance of adaptation and potential for disruption from different rates of phenological change are illustrated by the consequences of timing of migratory bird activities (see 4.3.2.1.). Breeding is most effective when synchronized with the availability of food, so changes in the phenology of food supplies can exert a selective pressure on birds. In a study of 100 European migratory bird species, those that advanced their arrival showed stable or increasing populations, while that did not adjust their arrival date on average showed declining populations (Møller et al., 2008). In a comparison of nine Dutch populations of the migratory pied flycatcher (Ficedula hypoleuca) over two decades, populations declined by 90% in areas where food peaked early in the season and the arrival of the birds was mis-timed, unlike areas with a later food peak which can still be exploited by early-breeding birds (Both et al., 2006). Selection for earlier laying dates has been shown to stronger in warmer years when food supply (caterpillars) peaked early (Coppack and Both, 2002). Systematic cross-taxa studies in the USA and UK indicate different rates of phenological change for different species and trophic levels (Parmesan, 2007; Cook et al., 2008; Thackeray et al., 2010).

It has been suggested that shorter generation times would give greater opportunity for autonomous adaptation through natural selection (Rosenheim and Tabashnik, 1991; Bertaux et al., 2004), but a standardized assessment of 25,532 rates of phenological change for 726 UK taxa indicated that generation time only had limited influence (Thackeray et al., 2010).

The physiological processes in organisms can either adapt through plasticity or genetically (i.e., through evolution). The former is generally regarded as rapid but limited, and the latter as less constrained, but slow (see 4.4.1.2) although long-term studies of a few organisms indicate rapid genetic adaptation to a changing climate (Bradshaw and Holzapfel, 2006). Acclimation to higher temperatures and/or higher CO₂ concentrations is seen in many organisms, but to very widely varying extents and general understanding remains limited (Hofmann and Togdham, 2010). A physiological process with large potential consequences for the global carbon cycle and thus the climate system is the sensitivity of the relationship between organism temperature and respiration rate, particularly in decomposer organisms (Jones et al., 2003). The shape of the relationship varies between locations with different climates and shows apparent acclimation within months or years to a changing climate (Giardina and Ryan, 2000; Luo et al., 2001; Rustad, 2001). The relationship is also dependent on C substrate type and the presence of other respiration-controlling environmental factors, complicating the determination of the inherent rate and its adaptation to a changing climate. Several competing hypotheses can explain the observed reduction in respiration rates in soil warming experiments after an initial peak – physiological adaptation; depletion of readily available substrate (Kirschbaum, 2004); and varying temperature sensitivities in different soil carbon pools (Knorr et al., 2005).

### 4.4.1.2. Evolutionary and Genetic

Since the AR4 report there has been substantial progress in defining the concepts and tools necessary for documenting and predicting evolutionary and genetic responses to recent and future climate change, often referred to as "rapid evolution". Evolution can occur through many mechanisms including selection of existing resistant genes or genotypes within populations, hybridization, mutation and selection of new adaptive genes and perhaps...
even through epigenetics (Chevin et al., 2010; Chown et al., 2010; Lavergne et al., 2010; Paun et al., 2010; Hoffmann and Sgro, 2011; Anderson et al., 2012b; Donnelly et al., 2012; Franks and Hoffmann, 2012; Hegarty, 2012; Merilä, 2012; Bell, 2013; Zhang et al., 2013). Mechanisms such as selection of existing genes and genotypes, hybridization and epigenetics can lead to adaptation in very few generations, while others, notably mutation and selection of new genes, typically take at least many tens of generations. This means that species with very fast life cycles, e.g., bacteria, should in general have greater capacity to respond to climate change than species with long life cycles such as large mammals and trees. Unfortunately, there remains a paucity of observational or experimental data that can be used for detection and attribution of climate signals and for validation of models and theory.

**Observed Evolutionary and Genetic Responses to Rapid Changes in Climate** - There is a small, but growing body of observational evidence supporting studies reviewed in the AR4 report that some species have adapted to recent climate warming or to climatic extremes through genetic responses (e.g., plants - Franks and Weis, 2008; Anderson et al., 2012a; Hill et al., 2011; vertebrates - Ozgul et al., 2010; Husby et al., 2011; Phillimore et al., 2010; Karell et al., 2011; insects - Buckley et al., 2012; van Asch et al., 2012). For example, Karell et al. (2011) found increasing numbers of brown genotypes of the tawny owl in Finland over the course of the last 28 years and attributed it to fewer snow-rich winters, which creates strong selection pressure against the heritable white genotype. Phillimore et al. (2010) showed for the common frog in Britain, that population differences in earlier spawning due to increasing spring temperatures could be attributed largely to local genetic adaptation. Using a combination of models and observations Visser and colleagues have built a case for detection and attribution of genetic adaptation in an insectivorous bird (Husby et al., 2011), and in a herbivorous insect that has tracked warming related changes in the budburst timing of its host tree (van Asch et al., 2012). In contrast, many species appear to be maladapted to changing climates or to respond to recent warming through phenotypic plasticity, in part because factors such as limited standing genetic variation, weak heritability of adaptive traits or conflicting constraints on adaptation create low potential for rapid evolution (Knudsen et al., 2011; Ketola et al., 2012; Mihoub et al., 2012; Merilä, 2012). Most studies of rapid evolution suffer from methodological weakness making it difficult to clearly demonstrate a genetic basis underlying observed phenotypic responses to environmental change (Gienapp et al., 2008; Franks and Hoffmann, 2012; Hansen et al., 2012; Merilä, 2012). When combined with recent progress on conceptual frameworks, rapid advances in quantitative genetics, genomics and phylogenetics will substantially improve the detection and attribution of genetic responses to changing climate over the next few years (Davis et al., 2010; Salamin et al., 2010; Hoffmann and Sgro, 2011).

The ability of species to adapt to new environmental conditions through rapid evolutionary processes can also be inferred from the degree to which environmental niches are conserved when environment is changed. There is good evidence that environmental niches are conserved for some species under some conditions (plants - Petitpierre et al., 2012; birds - Monahan and Tingley, 2012; review - Peterson et al., 2011), but also evidence suggesting that environmental niches can evolve over time scales of several decades following invasion or changes in climate (Broennimann et al., 2007; Angetter et al., 2011; Konarzewski et al., 2012; Leal and Gunderson, 2012; Lavergne et al., 2013). The paleontological record also provides insight into past evolutionary response in the face of natural climate variation. In general, environmental niches appear to be broadly conserved through time although there is insufficient data to determine the extent to which genetic adaptation has attenuated range shifts and changes in population size (Peterson et al., 2011; Willis and MacDonald, 2011). Phylogeographic reconstructions of past species distributions suggest that hybridization may have helped avoid extinctions during cycles of glaciation and could also play a key role in future adaptation (Soliani et al., 2012; Hegarty, 2012). There is also new evidence that epigenetic mechanisms, such as DNA methylation, can play a role in heritable and potentially very rapid adaptation to climate (Paun et al., 2010; Zhang et al., 2013), but understanding of these mechanisms is too preliminary to know how important they are for adaptation to climate change.

**Mechanisms Mediating Rapid Evolutionary Response to Future Climate Change** - Studies of extent genetic variability across species ranges and models that couple gene flow with spatially-explicit population dynamics suggest that populations may respond to climate change in ways that are counterintuitive. In some cases, too much or too little gene flow to populations at range margins may have created fragile, maladapted populations, which is in contrast to the current wisdom that populations at the range margins may be best adapted to global warming (Bridle et al., 2010; Hill et al., 2011). Conversely, there is also evidence from experiments, models and observations that populations in the center of species ranges may in some cases be more sensitive to environmental change than those...
at range boundaries (Bell and Gonzalez, 2009). Generalization will be complicated by the interactions between local adaptation, gene flow and population dynamics (Bridle et al., 2010).

Substantial progress has been made since the AR4 report in developing models that can be used to explore whether genetic adaptation will be fast enough to track climate change. Models of long-lived tree species suggest that existing genetic variation may be sufficient to significantly attenuate negative impacts of future climate change (Kuparinen et al., 2010; Kremer et al., 2012), which is coherent with observations and experiments (Jump et al., 2006, Jump et al., 2008). However, these studies also indicate that adaptive responses will lag far behind even modest rates of projected rates of climate change, in large part due to the very long generation time of trees. In a species with much shorter generation times, the great tit (Parus major), Gienapp et al. (2013) found that modeled breeding times tracked climate change, but only at low to moderate rates of change. For a species with an even faster life cycle, van Asch et al. (2007, 2012) predicted that rapid evolution of the phenological response of a herbivorous insect should have allowed it to track recent warming, which it has. This suggests substantial capacity for future adaptation in this insect since current and moderate future rates of future climate change are similar. Kearney et al. (2009) on the other hand found that rapid evolutionary responses only modestly affected modeled range shifts in a mosquito species in response to climate change. More broadly, Walters et al. (2012) found that modeled extinction risk in eutherms - species that do not internally regulate their body temperature - is substantially reduced by genetic adaptation at rates of climate change that are roughly less than 0.02°C per generation. Based on these assumptions, species with short generation times such as most insects potentially have the capacity to genetically adapt to even the most rapid rates of projected climate change (Figure 4-6), but species with longer generation times could be at risk of extinction at moderate to high rates of climate change. Recent experimental work on "evolutionary rescue" shows that long-term avoidance of extinction through genetic adaptation to hostile environments can occur under certain conditions (Bell, 2013). Generalizations coming from experiments and theoretical work are that evolutionary rescue requires large initial genetic variation and population sizes and is accompanied by substantial loss of genetic diversity, reductions in population size and range contractions over many generations before population recovery (Bell, 2013; Schiffers et al., 2013). Model-based projections must, however, be viewed with considerable caution because there are many evolutionary and ecological mechanisms that can either speed up or inhibit heritable adaptation to climate change that are not accounted for in most models (Cobben et al., 2012; Norberg et al., 2012; Kovach-Orr and Fussmann, 2013). In some cases, accounting for evolutionary processes in models even leads to predictions of enhanced maladaptation to climate change and more rapid population declines under certain conditions (Ferriere and Legendre, 2013). Phenotypic plasticity is thought to generally improve the odds of adaptation to climate change, but the extent to which it contributes to adaptive capacity depends a great deal on the costs of plasticity in terms of fitness (Chevin et al., 2010). High plasticity in the face of climate change that has low costs can greatly improve the odds of adapting to climate change; however, plasticity that has high fitness costs can have much more modest effects on the odds of adaptation.

The AR4 report concluded that "projected rates of climate change are very likely to exceed rates of evolutionary adaptation in many species (high confidence)". Work since then provides a similar, but more nuanced view of rapid evolution in the face of climate change. There are few observational studies of rapid evolution and difficulties in detection and attribution, so there is only medium confidence that some species have responded to recent changes in climate through genetic adaptations, and insufficient evidence to determine if this is a widespread phenomenon (thus low confidence across all species). The lack of adaptation in some species to recent changes in climate, broad support for niche conservatism and models showing limited adaptive capacity especially in species with long generation times, all indicate that high rates of climate change (RCP8.5) will exceed the adaptive capacities of many species (high confidence). On the other hand, evidence from observations and models also indicates that there is substantial capacity for genetic adaptation to attenuate phenological shifts, population declines and local extinctions in many species, especially for low rates of climate change (RCP2.6) (high confidence). Projected adaptation to climate change is frequently characterized by population declines and loss of genetic diversity for many generations (medium confidence), thereby increasing species vulnerability to other pressures and potentially impacting ecosystem functioning and services.
4.4.1.3. Migration of Species

This mode of adaptation has been extensively dealt with in section 4.3.2.5. It is anticipated that the movement of species – individually and collectively – will continue in response to shifting climate patterns. Its effectiveness as an adaptation mechanism is constrained by three factors. First, the rate of migration for many species, in many regions of the world, is slower than the rate of movement of the climate envelope (see Figure 4-6). Second, the ecosystem interactions can only remain intact if all parts of the ecosystem migrate simultaneously and at the same rate. Thirdly, the contemporary landscape and drainage systems contain many barriers to migration, in the form of habitat fragmentation, roads, human settlements and dams.

4.4.2. Human-Assisted Adaptation

Human-assisted adaptation means a deliberate, external intervention with the intent of increasing the capacity of the target organism, ecosystem or social-ecological system to survive and function at an acceptable level in the presence of climate change (see glossary). It is also known as ‘planned adaptation’ (Smit et al., 2007). This chapter focuses less on the adaptation of people, human communities and infrastructure, since they are the topics of WGII chapters 8 to 17, and more on non-human organisms and ecosystems, while acknowledging the importance of the human elements within the ecosystem. “Assistance” in this context means a range of actions, from ensuring the presence of suitable habitat and dispersal pathways and the reduction or removal of other stressors, to physically moving organisms, storing them and establishing them in new places. In addition to the other approaches assessed in this section, “Ecosystem-based Adaptation” provides an option that integrates the use of biodiversity and ecosystem services into climate change adaptation strategies in ways that can optimize co-benefits for local communities and carbon management, as well as reduce the risks associated with possible maladaptation (see Box CC-EA). Note that there are risks associated with all forms human-assisted adaptation (see 4.4.4), particularly in the presence of far-from-perfect predictive capabilities (Willis and Bhagwat, 2009).

4.4.2.1. Reduction of Non-Climate Stresses and Restoration of Degraded Ecosystems

The alleviation of other stresses acting on ecosystems is suggested to increase the capacity of ecosystems to survive, and adapt to, climate change, since the effects are generally either additive or compounding. Ecosystem restoration is one way of alleviating such stresses while increasing the area available for adaptation (Harris et al., 2006). Building the resilience of at-risk ecosystems by identifying the full set of drivers of change and most important areas and resources for protection is the core of the adaptation strategy for the Arctic (Christie and Sommerkorn, 2012). ‘Ecosystem-based adaptation’ is the phrase increasingly used to cover a package of protective and restorative actions aimed at increasing resilience (e.g. Colls et al., 2009).

4.4.2.2. The Size, Location and Layout of Protected Areas

Additions to, or reconfigurations of, the protected area estate are commonly suggested as pre-adaptations to projected climate changes (Heller and Zavaleta, 2009). This is because for most protected areas, under plausible scenarios of climate change, a significant fraction of the biota will no longer have a viable population within the present protected area footprint. It is noted that the extant geography of protected areas is far from optimal for biodiversity protection even under the current climate; that it is cheaper to acquire land proactively in the areas of projected future bioclimatic suitability than to correct the current non-optimality and then later add on areas to deal with climate change as it unfolds (Hannah et al., 2007); and that the existing protected area network will still have utility in future climates, even though it may contain different species. Hickler et al. (2012) analyzed the layout of protected areas in Europe and concluded that under projected 21st century change and concluded that a third to a half of them would potentially be occupied by different vegetation than they currently represent. The new area that needs to be added to the existing protected area network to ensure future representativeness is situation-specific, but some general design rules apply: orientation along climate gradients (e.g., altitudinal gradients) is more effective than orientation across them (Roux et al., 2008); regional scale planning is more effective than treating each local case
independently (Heller and Zavaleta, 2009); since it is the network of habitats and protected areas that confers
resilience rather than any single element; and better integration of protected areas with a biodiversity-hospitable
landscape outside is more effective than treating the protected areas as islands (Willis and Bhagwat, 2009).

4.4.2.3. Landscape and Watershed Management

The general principles for management adaptations in United States were summarized from a major literature review
by West et al., 2009. They suggest that in the context of climate change a successful management of natural
resources will require cycling between ‘managing for resilience’ and ‘managing for change’. This requires the
anticipation of changes that can alter the impacts of grazing, fire, logging, harvesting, recreation, and so on. At the
national level, the necessary changes in practice to facilitate adaptation include: (1) management at appropriate
scales, and not necessarily the scales of convenience or tradition; (2) increased collaboration among agencies; (3)
rational approaches for establishing priorities and applying triage; and (4) management with the expectation of
ecosystem change, rather than keeping them as they have been. Barriers and opportunities were divided into four
categories: (1) legislation and regulations, (2) management policies and procedures, (3) human and financial capital,
and (4) information and science.

In 14,000 ha of forested watersheds in central Nova Scotia, Canada, the adaptation to climate change was assessed
using the landscape disturbance model LANDIS-II, (Steenberg et al., 2011). The study simulated the impact of three
components of timber harvesting: the canopy-opening size of harvests, the age of harvested trees within a stand, and
the species composition of harvested trees within a stand. The combination of all three adaptation treatments
allowed target species and old forest to remain reasonably well represented without diminishing the timber supply.
This minimized the trade-offs between management values and climate adaptation objectives.

4.4.2.4. Assisted Migration

Assisted migration has been proposed when fragmentation of habitats limits migration potential or when natural
migration rates are outstripped by the pace of climate change, (Hoegh-Guldberg et al., 2008; Vitt et al., 2010;
Chmura et al., 2011; Loss et al., 2011). The options for management can be summarized as: i) try to maintain or
improve existing habitat so that species don’t have to move, ii) maintain or improve migration corridors, including
active management to improve survival along the moving margin of the distribution (Lawson et al., 2012), or iii)
intervene heavily by translocating species (Hoegh-Guldberg et al., 2008; Loss et al., 2011). There is low agreement
in the scientific community whether it is best to increase the resilience of ecosystems to climate change, thus help to
preserve existing communities, or to enhance the capacity of ecosystems to transform in the face of the
overwhelming forces of species migration and modifications of ecosystem function by climate change.

There is high agreement among the scientific and conservation community that maintaining or improving migration
corridors or ecological networks is a low-regret strategy, partly because it is also seen as useful in combatting the
negative effects of habitat fragmentation on population dynamics (Hole et al., 2011; Jongman et al., 2011). This
approach has the benefit of improving the migration potential for large numbers of species and is therefore a more
ecosystemic approach than assisted migration for individual species. However, observational and modeling studies
show that increases in habitat connectivity do not always improve the population dynamics of target species, may
decrease species diversity, and may also facilitate the spread of invasive species (Cadotte, 2006; Brisson et al., 2010;
Matthiessen et al., 2010).

There is medium agreement that the practice of assisted migration of targeted species is a useful adaptation option
(Hoegh-Guldberg et al., 2008; Loss et al., 2011; Vitt et al., 2009; Willis and Bhagwat, 2009; Hewitt et al., 2011).
The speed of 21st century climate change and substantial habitat fragmentation in many areas of the world mean that
many species will be unable to migrate or adapt fast enough to keep pace with climate change. If this results in
significant reductions in range size it will pose problems for long-term survival of the species. Some ecologists
believe that careful selection of species to be moved would minimize the risk of undesirable impacts on existing
communities or ecosystem function (Minteer and Collins, 2010), but others argue that the history of intentional
species introductions shows that the outcomes are unpredictable and in many cases have had disastrous impacts (Ricciardi and Simberloff, 2009). The number of species that could potentially require assisted migration that could easily overwhelm funding capacity (Minteer and Collins, 2010). The degree and magnitude of phenotypic responses or genetic adaptation to climate change are very variable among species, making decisions regarding which species should be translocated complex and debatable.

4.4.2.5. Ex Situ Conservation

Conservation of plant and animal genetic resources outside of their natural environment, in gardens, zoos, breeding programmes, seed-banks or gene-banks has been widely advocated as an ‘insurance’ against both climate change and other sources of biodiversity loss and impoverishment (Khoury et al., 2010). There are many examples of existing efforts of this type, some very large and with global scope (e.g. Millennium Seed Bank, Svalbard vault). The Dixon National Tallgrass Prairie Seed Bank is an example of prioritization of species for seed banking, both for restoration purposes and for potential assisted migration in the future (Leishman et al., 2007). A genome scan approach identified four potentially adaptive loci in important grassland species *Arrhenatherum elatium*. Knowledge on adaptive loci might in the long run also help to adapt ecosystems to adverse climate change effects through assisted migration of ecotypes rather than introduction of new species (Colautti et al., 2010).

Several issues remain largely unresolved (Li and Pritchard, 2009). The physiological, institutional and economic sustainability of such efforts into the indefinite future is unclear. The fraction of the intra-specific variation that needs to be preserved for future viability and how much genetic bias is introduced by founder effects and selection pressures applied during ex-situ maintenance is unknown. It remains uncertain whether it is possible to reintroduce species (especially animals) successfully into the wild after generations of ex-situ conservation.

4.4.3. Consequences and Costs of Inaction and Benefits of Action

Failure to act to assist ecosystem adaptation to climate change will plausibly leads to ecological, social and economic damages. The necessary actions to cope with unavoidable damages generate adaptation costs, while mitigation costs are associated with actions to tackle undesired future damages. Examples of these costs, based on recent literature, are shown in Table 2-5 in Rodriguez-Labajos (submitted). The timing of the action also has cost implications: increasing costs that result from must be weighed against the risks associated with premature measures (Szlavik and Csete, 2012). In addition to the direct financial costs of action, further costs may appear through trade-offs between services: e.g. afforestation for climate mitigation is costly in terms water provision (Chisholm, 2010).

Traditional agriculture preserves soil carbon sinks, supports on-site biodiversity and uses less fossil fuel than high-input agriculture (Martinez-Alier, 2011), but due to the lower per hectare yields, requires a larger area to be dedicated to cropland.

A comprehensive cost estimate of the effects of climate change on ecosystem service provision is not available. ten Brink et al. (2008) report the monetary cost of not meeting the 2010 biodiversity goals in terms of lost value of ecosystem services. Their model incorporates climate change, among other pressures, and concludes that cumulative losses of welfare due to losses in ecosystem service provision could reach an annual amount of 14 trillion Euro (based on values of 2007), in 2050 equivalent to 7% of projected global GDP for that year.

Economic calculations are appropriate when at least one component of the ecosystem services is traded in markets (such as for biofuels; Mishra et al., 2010). If climate regulation services are translated into a tradable item, through carbon markets or payments for avoiding deforestation, then their value can be expressed via market prices (Shaw et al., 2011).

The market price for carbon is volatile, linked to speculation and political agreements on emission reduction. There is a range of estimates from $23/metric ton of carbon to $371, in current 1995 and 2007 dollars respectively (Tol, 2008; Watkiss and Downing, 2008). In the South-African Fynbos region, economic viability of afforestation proved to be highly sensitive to the value of carbon (Chisholm, 2010). The realized prices of the traded carbon have been on
average lower than the figures mentioned above. The proposed global REDD programs, financed by carbon-offset trading, rely on the premises on which market-based payment for ecosystem services (PES) are founded. However, a decade of PES experience demonstrates a clash between market-efficiency criteria and poverty reduction and therefore a risk of regressive wealth redistribution (McAfee, 2012).

Uncertainty about the potential consequences of abrupt changes advises against valuation of ecosystems close to critical thresholds (TEEB, 2009). Similar difficulties can be argued in the presence of irreversible changes, as in the case of irreversible biodiversity losses or damage to cultural services such as World Heritage sites (Viles and Cutler, 2012).

4.4.4. Unintended Consequences of Adaptation and Mitigation Actions in This and Other Sectors

Some issues relevant to this section are also found in 4.2.2 and in Chapter 14.

Several of the alternatives to fossil fuel require extensive use of the land surface. Bio-energy requires land to either be taken from food production or from natural ecosystems. Many renewables require significant land areas. Hydroelectricity usually involves the impoundment of large bodies of water behind dams, leading to flooding of pre-existing ecosystems. Solar energy effectively involves decreasing the albedo of areas of ground surface. Large-scale wind energy involves the location of arrays of turbines across the landscape. As an illustration, the “aggressive mitigation” scenario RPC2.6 relies heavily on both bio-energy and renewables as major components of the energy mix (van Vuuren et al., 2011), so there is clear potential for unintended consequences for terrestrial and inland water systems.

In RCP2.6, although food production remains dominant, the scenario still involves some land use / land-cover change (Box 4-1) in order to facilitate the use of bio-energy. By 2100, bio-energy crops occupy approximately 4 million km², approximately 7% of global cultivated land projected at the time. Modification of the landscape and the fragmentation of habitats are major influences on extinction risk (Fischer and Lindenmayer, 2007), especially if native vegetation cover is reduced or degraded, human land use is intensive and “natural” areas become disconnected. Hence, additional extensification of cultivated areas for energy crops may pose extinction risks.

RCP2.6 also includes a substantial increase in renewables above current levels of deployment, either as hydropower, solar or wind. Damming of river systems for hydropower can cause fragmentation of the inland water habitat with implications for fish species, and monitoring studies indicate that flooding of ecosystems behind the dams can lead to declining populations, e.g., of amphibians (Brandão and Araújo, 2007). Large dams may also result in CH₄ emissions due to decay of flooded vegetation in anaerobic conditions and the subsequent release of CH₄ from the water surface and especially from water passing through turbines and spillways (Fearnside, 2005; Lima et al., 2008), so dams may act as sources of greenhouse gas emissions.

Concern is often raised over wind turbines posing a danger to birds and bats, but estimating mortality rates is complex and difficult (Smallwood, 2007). It has been estimated that wind farms in Europe and the USA cause between 0.3 and 0.4 wildlife fatalities per gigawatt-hour (GWh) of electricity, and while nuclear and fossil-fuel power stations are estimated to cause approximately 5.2 fatalities per GWh (Sovacool, 2009), although this quantification has been criticized as having key methodological flaws (Willis et al., 2010b). One study found on-site bird populations to be generally affected more by windfarm construction than subsequent operation, with some populations recovering after construction (Pearce-Higgins et al., 2012).

Solar energy systems, by design, aim to maximize the absorption of solar energy and hence widespread deployment of photovoltaics (PV) has the potential to exert a positive radiative forcing on climate through decreased surface albedo. In a scenario of widespread deployment of solar PV, such a forcing is estimated to be 30 times smaller than the avoided radiative forcing arising from the use of PV to substitute fossil fuels, so overall PV still exerts a net reduction of climate warming (Nemet, 2009). Nevertheless, at the local scale there are some plausible circumstances in which the decreased surface albedo substantially reduces the effectiveness of PV as a negative climate forcing (Nemet, 2009).
Adaptation measures may also result in unintended consequences. Relocation of agricultural areas as a climate change adaptation measure could pose risks of habitat fragmentation and loss similar to those discussed above in the context of mitigation through bio-energy. Deliberate relocation of vulnerable and important species – assisted migration – may also be a potential conservation measure in the context of facilitating adaptation to climate change (Maclachlan et al., 2007). However this may directly conflict with other conservation priorities, for example by actually facilitating the introduction of invasive species (Maclachlan et al., 2007).

Proactive adaptation measures may also have unforeseen consequences politically. Carey et al. (2011) argue that improved water management in a glacial lake in the Peruvian Andes in 1985 led to increased tensions over many years and ultimately local civil action against the authorities in 2008. Introduction of a tunnel to reduce the risk of a glacial lake outburst flood facilitated proactive human management of the lake levels, but in the context of changes in the wider political situation, power struggles arose between different stakeholders due to different and conflicting interests. This may provide an example of how attempts by society to exert more direct control over the environment can lead to unintended impacts.

4.5. Emerging Issues and Key Uncertainties

The presence of thresholds in ecosystem response to climate change, and specifically the type of thresholds characterized as ‘tipping points’, is a growing concern but remains a major source of uncertainty. In general (Field et al., 2007), negative feedbacks currently dominate the climate-ecosystem interaction, but in several areas, such as the boreal ecosystems, positive feedbacks could dominate, even under moderate warming. For most ecological processes, increasing magnitude of warming shifts the balance towards positive rather than negative feedbacks (Field et al., 2007). For positive feedbacks to propagate into ‘runaway’ processes leading to a new ecosystem state, the strength of the feedback has to exceed that of the initial perturbation. This has not as yet been demonstrated for any large-scale, plausible and immanent ecological process; but the risk is non-negligible and the consequences if it did occur would be severe: thus further research is needed. The issue of biophysical feedbacks between ecosystem state and the climate, over and above the effects mediated through greenhouse gases, is emerging as significant in many areas. Such effects include those caused by changes in surface reflectivity (albedo) or the partitioning of energy between latent energy and sensible heat.

Uncertainty in predicting the response of ecosystems to climate and other perturbations remains a major impediment to determining prudent levels of permissible change. A significant source of this uncertainty stems from the inherent complexity of ecosystems, especially where they are coupled to equally-complex social systems. The high number of interactions can lead to cascading effects (Biggs et al., 2011). Some of this uncertainty can be removed by understanding the systems better, but some will remain irreducible because of the failure of predictive models when faced with mathematical bifurcations – a problem that is well-known in climate science. Probabilistic statements about the range of outcomes are possible in this context, but ecosystem science is mostly as yet unable to conduct such analyses routinely and rigorously. One consequence is the ongoing difficulty in attributing observed changes unequivocally to climate change. More comprehensive monitoring is a key element of the solution.

Studies of the combined effects of multiple simultaneous elements of global change, such as the effects of elevated carbon dioxide and rising tropospheric ozone on plant productivity - which has critical consequences for the suture sink strength of the biosphere, since they of similar magnitude but opposite sign – are needed as a supplement to the single-factor experiments.

Understanding of the tolerable rate of climate change is as important as understanding the tolerable magnitude. Despite being explicitly required under Article 2 of the UNFCCC, rate studies are currently less developed and more uncertain than magnitude (equilibrium) studies. This includes evidence for the achievable migration rates of a range of species as well as the rate of micro-evolutionary change.

The costs of assisted adaptation ecosystems, biodiversity and ecosystem services to a changing climates is poorly known, as are the costs of failing to do so.
Frequently Asked Questions

FAQ 4.1: How does climate change contribute to species extinction?
Species extinctions are of considerable concern because they represent irreplaceable losses of unique life forms and because species loss has been shown to degrade the functioning of ecosystems in some cases. Most species extinctions in the recent past can be attributed to habitat destruction, invasive species, overexploitation and pollution. Climate change may have already contributed to the extinction of a small number of species, such as frogs and toads in Central America, but the role of climate change in these current extinctions is the subject of considerable debate. Over the coming century, climate change is foreseen to increase extinction risk because some species will not be able to adapt to new environments, nor move areas where climate becomes suitable. There is consensus that minimizing climate change will help protect species from extinctions. Under the highest rates of projected climate change, observations and models indicate that many species will be unable to move fast enough to track suitable climate or their ranges will shrink considerably. There is concern that this will substantially increase extinction risk for a large fraction of species, especially when combined with other global change pressures such as habitat destruction. We are, however, currently unable to accurately quantify this risk. Under the lowest projected rates and amounts of climate change and with the assistance of effective conservation actions, the large majority of species are foreseen to be able to adapt to the new climates, or respond by moving to higher latitudes or altitudes.

FAQ 4.2: Why does it matter that ecosystems are altered by climate change?
Ecosystems are the machinery that delivers things essential for all people, everywhere in the world – including obvious items like food, clean water and timber, but also less-obvious ones such as control of pests and diseases, regulation of the climate and a life-supporting atmosphere and pollination of flowers. When ecosystems change, their capacity to supply these services changes as well, for better or worse. Humans are directly affected, as are the millions of other species dependent on ecosystems. "Ecosystem change" includes the species it contains and in what amounts and proportions, how the ecosystem looks (e.g. tall or short, open or dense) and how it works (e.g. whether it is productive or unproductive). Climate change, among other factors, has an effect on all these aspects of ecosystems, as well as on the total area occupied by particular ecosystems and its geographical location. Ecosystem change has knock-on effects on many sectors, including human health through altering where diseases are found, agriculture through grazing supply and pollinators, infrastructure through changing risks of flooding. Although in the long-term not all ecosystem changes are detrimental to all people or all species, rapidly-changing ecosystems will require adaptation, perhaps in excess of their adaptive capacity, by both the people and organisms dependent on them.

FAQ 4.3: What are the non-greenhouse gas effects of rising carbon dioxide?
Some greenhouse gases exert influences on the environment other than their impact on radiative forcing. The concentration of carbon dioxide affects photosynthesis and transpiration in plants, with photosynthesis generally being enhanced under higher CO₂ concentrations while transpiration is generally decreased. The growth of plants and/or the efficiency with which they use water can therefore be increased. The response varies considerably between species. Increased atmospheric CO₂ concentration also increases the acidity of water, with negative consequences for organisms with calcium carbonate shells, which cannot be grown under low pH.

FAQ 4.4: What costs are caused by changes in ecosystem services due to climate change?
Climate change will certainly alter the services provided by ecosystems, and on balance, for high degrees of change, in an overall negative way. For instance, coral reefs protect the coast against extreme storm events, which are predicted to increase under climate change – but the reefs are at risk due to bleaching induced by temperature rise, as well as due to ocean acidification. This service alone has been valued at several 1000 $/ha per hour. Other costly damages could include a decrease in pollination services due to climate change related asynchrony of flowering time of plants and activity periods of pollinators. This service is worth worth EUR 153 billion per year (a value that nearly doubles if you take into account purchasing power parity of the affected countries).
FAQ 4.5: What are the opportunities for better managing ecosystems under climate change?

Management of ecosystems can help to mitigate climate change. Forests, for example deliver a wide set of services, including climate regulation. Ecosystems can also be managed to increase their ability to cope with climate change. Reducing the many other stresses on ecosystems – such as excessive harvest, habitat fragmentation and pollution is an important strategy. Maintaining biological diversity and near-natural disturbance regimes also helps. We may need to take drastic action such as assisting species to migrate, or protecting them out of the wild until such time as they can be reintroduced.

FAQ 4.6: Can land-use and land-cover changes cause changes in climate?

Land-use change can affect the local climate, and even the global climate. It does so by changing the emission or uptake of greenhouse gases, but also by changing how much solar energy is absorbed by the land surface. Depending on the type of change and where it occurs, the effects can be either warming or cooling. The effects on precipitation are equally variable and context-dependent. At a global scale, the conversion of forests to croplands, pastures and urban areas is currently contributing around a fifth of the overall emissions of the greenhouse gases which are causing climate change.

FAQ 4.7: Will the number of alien species increase due to climate change?

An alien species, also called exotic or non-native species is one that has been introduced to a region outside of its native geographic range as a deliberate or accidental consequence of human activity. They are called invasive when they cause ecological and/or economic damages as well as if they spread fast and produced a high amount of fertile offspring. The number of new species and the abundance of some already established alien species will increase in certain areas, such as polar regions, due to climate change improving conditions for these species, e.g. higher winter temperatures or longer growing seasons. Increased globalization of goods and human transportation and increased global land use changes are both likely to increase the frequency, establishment and damages of alien invasive species throughout the world although management actions to prevent the transfer of alien species (e.g. through treatment of ballast water and wood products and strict quarantine protocols applied to crop and horticultural products and embargos on trade and deliberate introductions of known invaders) are more and more applied. Some invasive plants and insects have already been shown to benefit from climate change and will likely establish and spread into new regions, once they are introduced there. However, other invasive species will suffer from climate change and, as for most of the biodiversity, are expected to decrease in range and population size in some regions. Within just the Family of ants, some highly invasive species are expected to benefit from climate change and increase their potential invasive areas, while some others will likely much recede.

Generally, increased establishment success and spread will be most visible for those alien species that have characteristics favored by the changing climate, such as those that are drought tolerant or able to take advantage of higher temperatures.

Cross-Chapter Boxes

Box CC-EA. Ecosystem Based Approaches to Adaptation - Emerging Opportunities

[Rebecca Shaw (USA), Jonathan Overpeck (USA), Guy Midgley (South Africa)]

Ecosystem-based approaches to adaptation (also termed Ecosystem-based Adaptation, EBA) integrate the use of biodiversity and ecosystem services into climate change adaptation strategies (e.g., CBD, 2009; Munroe et al., 2011; Munroe et al., 2011). EBA is implemented through the sustainable management of natural resources, as well as conservation and restoration of ecosystems, to provide and sustain services that facilitate adaptation both to climate variability and change (Colls et al., 2009). The CBD COP 10 Decision X/33 on Climate Change and Biodiversity states further that effective EBA also “takes into account the multiple social, economic and cultural co-benefits for local communities”.

The potential for EBA is increasingly being realized (e.g., Munroe et al., 2011), offering opportunities that integrate with or even substitute for the use of engineered infrastructure or other technological approaches. Engineered defenses such as dams, sea walls and levees, may adversely affect biodiversity, resulting in maladaptation due to damage to ecosystem regulating services (Campbell et al., 2009, Munroe et al., 2011). There is some evidence that
the restoration and use of ecosystem services may reduce or delay the need for these engineering solutions (CBD, 2009). Well-integrated EBA is also more cost effective and sustainable than non-integrated physical engineering approaches, and may contribute to achieving sustainable development goals (e.g., poverty reduction, sustainable environmental management, and even mitigation objectives), especially when they are integrated with sound ecosystem management approaches. EBA also offers lower risk of maladaptation than engineering solutions in that their application is more flexible and responsive to unanticipated environmental changes.

EBA provides opportunities particularly in developing countries where economies depend more directly on the provision of ecosystem services (Vignola et al., 2009), to reduce risks to climate change impacts and ensure that development proceeds on a pathways that are resilient to climate change (Munang et al., ). In these settings, ecosystem-based adaptation projects may be readily developed by enhancing existing initiatives, such as community-based adaptation and natural resource management approaches (e.g., Khan et al., 2012, Midgley et al., 2012; Roberts et al., 2012)

Examples of ecosystem based approaches to adaptation include:

- Sustainable water management, where river basins, aquifers, flood plains, and their associated vegetation are managed or restored to provide resilient water storage and enhanced baseflows, flood regulation services, reduction of erosion/siltation rates, and more ecosystem goods (e.g., Midgley et al., 2012, Opperman et al., 2009).
- Disaster risk reduction through the restoration of coastal habitats (e.g., mangroves, wetlands and deltas) to provide effective measure against storm-surges, saline intrusion and coastal erosion;
- Sustainable management of grasslands and rangelands to enhance pastorallivelihoods and increase resilience to drought and flooding;
- Establishment of diverse and resilient agricultural systems, and adapting crop and livestock variety mixes to secure food provision. Traditional knowledge may contribute in this area through, for example, identifying indigenous crop and livestock genetic diversity, and water conservation techniques;
- Management of fire-prone ecosystems to achieve safer fire regimes while ensuring the maintenance of natural processes.

It is important to assess the appropriate and effective application of EBA as a developing concept through learning from work underway, and to build understanding of the social and physical conditions that may limit its effectiveness. Application of EBA, like other approaches, is not without risk, and risk/benefit assessments will allow better assessment of opportunities offered by the approach.

[INSERT FIGURE EA-1 HERE

Figure EA-1: Adapted from Munang et al. (2013). Ecosystem based adaptation approaches to adaptation can utilize the capacity of nature to buffer human systems from the adverse impacts of climate change through sustainable delivery of ecosystems services. A) Business as Usual Scenario in which climate impacts degrade ecosystems, ecosystem service delivery and human well-being B) Ecosystem-based Adaptation Scenario which utilizes natural capital and ecosystem services to reduce climate-related risks to human communities.]

CC-EA References


Box CC-RF. Impact of Climate-Change on Freshwater Ecosystems due to Altered River Flow Regimes

[Petra Döll (Germany), Stuart E. Bunn (Australia)]

It is widely acknowledged that the flow regime is a primary determinant of the structure and function of rivers and their associated floodplain wetlands, and flow alteration is considered to be a serious and continuing threat to freshwater ecosystems (Bunn and Arthington, 2002; Poff and Zimmerman, 2010; Poff et al., 2010). Most species distribution models do not consider the effect of changing flow regimes (i.e. changes to the frequency, magnitude, duration and/or timing of key flow parameters) or they use precipitation as proxy for river flow (Heino et al., 2009).

There is growing evidence that climate change will significantly alter ecologically important attributes of hydrologic regimes in rivers and wetlands, and exacerbate impacts from human water use in developed river basins (Aldous et al., 2011; Xenopoulos et al., 2005). By the 2050s, climate change is projected to impact river flow characteristics like long-term average discharge, seasonality and statistical high flows (but not statistical low flows) more strongly than dam construction and water withdrawals have done up to the year 2000 (Figure RF-1; Döll and Zhang, 2010). For one climate scenario, 15% of the global land area may suffer, by the 2050s, from a decrease of fish species in the upstream basin of more than 10%, as compared to only 10% of the land area that has already suffered from similar decreases due to water withdrawals and dams (Döll and Zhang, 2010). Climate change may exacerbate the negative impacts of dams for freshwater ecosystems but may also provide opportunities for operating dams and power stations to the benefit of riverine ecosystems. This is the case if total runoff increases and, like in Sweden, the annual hydrograph becomes more similar to variation in electricity demand, i.e. with a lower spring flood and increased run-off during winter months (Renofalt et al., 2010).

Figure RF-1: Impact of climate change on the ecologically relevant river flow characteristics mean annual river flow and monthly low flow Q90 as compared to the impact of water withdrawals and dams on natural flows, as computed by a global water model (Döll and Zhang, 2010). Impact of climate change is the percent change of flow between 1961-1990 and 2041-2070 according to the emissions scenario A2 as implemented by the global climate model HadCM3. Impact of water withdrawals and reservoirs is computed by running the model with and without water withdrawals and dams that existed in 2002.

Because biota are often adapted to a certain level of river flow variability, the larger variability of river flows that is due to increased climate variability is likely to select for generalist or invasive species (Ficke et al., 2007). The relatively stable habitats of groundwater-fed streams in snow-dominated or glacierized basins may be altered by reduced recharge by meltwater and as a result experience more variable (possibly intermittent) flows (Hannah et al., 2007). A high-impact change of flow variability is a flow regime shift from intermittent to perennial or vice versa. It is projected that until the 2050s, river flow regime shifts may occur on 5-7% of the global land area, mainly in semi-arid areas (Döll and Müller Schmied, 2012; see Chapter 3, Table 3-2).

In Africa, one third of fish species and one fifth of the endemic fish species occur in eco-regions that may experience a change in discharge or runoff of more than 40% by the 2050s (Thieme et al., 2010). Eco-regions containing over 80% of Africa’s freshwater fish species and several outstanding ecological and evolutionary
Phenomena are likely to experience hydrologic conditions substantially different from the present, with alterations in long-term average annual river discharge or runoff of more than 10% due to climate change and water use (Thieme et al., 2010).

Due to increased winter temperatures, freshwater ecosystems in basins with significant snow storage are affected by higher river flows in winter, earlier spring peak flows and possibly reduced summer low flows (chapter 3.2.3). Strongly increased winter peak flows may lead to a decline in salmonid populations in the Pacific Northwest of the USA of 20-40% by the 2050s (depending on the climate model) due to scouring of the streambed during egg incubation, the relatively pristine high-elevation areas being affected most (Battin et al., 2007). Reductions in summer low flows will increase the competition for water between ecosystems and irrigation water users (Stewart et al., 2005). Ensuring environmental flows through purchasing or leasing water rights and altering reservoir release patterns will be an important adaptation strategy (Palmer et al., 2009).

Observations and models suggest that global warming impacts on glacier and snow-fed streams and rivers will pass through two contrasting phases (Burkett et al., 2005; Vuille et al., 2008; Jacobsen et al., 2012). In the first phase, when river discharge is increased due to intensified melting, the overall diversity and abundance of species may increase. However, changes in water temperature and stream-flow may have negative impacts on narrow range endemics (Jacobsen et al., 2012). In the second phase, when snowfields melt early and glaciers have shrunken to the point that late-summer stream flow is reduced, broad negative impacts are foreseen, with species diversity rapidly declining once a critical threshold of roughly 50% glacial cover is crossed (Figure RF-2).

River discharge also influences the response of river temperatures to increases of air temperature. Globally averaged, air temperature increases of 2°C, 4°C and 6°C are estimated to lead to increases of annual mean river temperatures of 1.3°C, 2.6°C and 3.8°C, respectively (van Vliet et al., 2011). Discharge decreases of 20% and 40% are computed to result in additional increases of river water temperature of 0.3°C and 0.8°C on average (van Vliet et al., 2011). Therefore, where rivers will experience drought more frequently in the future, freshwater-dependent biota will suffer not only directly by changed flow conditions but also by drought-induced river temperature increases, as well as by related decreased oxygen and increased pollutant concentrations.

CC-RF References


Increased intrinsic water use efficiency (iWUE) of plants, as less water is transpired per unit of carbon assimilated. Reports of stable carbon isotopes in woody plants (Peñuelas et al., 2011; Reddy et al., 2004) also showed different runoff trends in some areas. Detection of ecosystem influences on terrestrial water flows, hence, critically depends on the availability and quality of hydrometeorological observations (Haddeland et al., 2011; Lorenz and Kunstmann, 2012). There are conflicting views on whether the direct CO$_2$ effects on plants already have a significant influence on evapotranspiration and runoff at global scale. AR4 reported work by Gedney et al., (2006) which suggested that physiological CO$_2$ effects (lower transpiration) contributed to a supposed global increase in runoff seen in reconstructions by (Labat et al., 2004). However, a more recent dataset (Dai et al., 2009) showed different runoff trends in some areas. Detection of ecosystem influences on terrestrial water flows, hence, critically depends on the availability and quality of hydrometeorological observations (Haddeland et al., 2011; Lorenz and Kunstmann, 2012). A key influence on the significance of increased iWUE for large-scale transpiration is whether overall leaf area of
primary vegetation has remained approximately constant (Gedney et al., 2006) or has increased in some regions due to structural CO₂ effects (as assumed in models by Piao et al., 2007; Gerten et al., 2008). While field-based results vary considerably between sites, tree ring studies suggest that tree growth did not increase globally since the 1970s in response to climate and CO₂ change (Peñuelas et al., 2011; Andreu-Hayles et al., 2011). However, basal area measurements at over 200 plots across the tropics suggest that biomass and growth rates in intact tropical forests have increased in recent decades (Lewis et al., 2009), which is also confirmed for 55 temperate forest plots, with a suspected contribution of CO₂ rise (McMahon et al., 2010). The net impact of CO₂ on global-scale transpiration and runoff therefore remains poorly constrained.

Moreover, model results differ in terms of the importance of CO₂ effects for historical runoff relative to other drivers such as climate, land use change and irrigation water withdrawal. Other than Gedney et al., (2006), Piao et al., (2007) and Gerten et al., (2008) found that CO₂ effects on global runoff were small relative to effects of precipitation, and that land use change (which often acts to decrease evapotranspiration and to increase runoff) was of second-most importance, as also supported by Sterling et al., (2012) data and model analysis. By contrast, using a shorter time period and a smaller selection of river basins, Alkama et al., 2011(2011) suggested that global effects of land use change on runoff have been negligible. Oliveira et al., 2011(2011) furthermore point to the importance of changes in incident solar radiation and the mediating role of vegetation; their global simulations demonstrate, for example, that a higher diffuse radiation fraction during 1960–1990 increased evapotranspiration in the tropics by 3% due to increased photosynthesis from shaded leaves. Since the anthropogenic component of the precipitation and temperature contributions (i.e. of the radiative CO₂ effect) to runoff trends is not yet established, a full attribution of anthropogenic emissions of CO₂ (and other greenhouse gases) is still missing.

Analogously, there is uncertainty about how vegetation responses to future increases in CO₂ will modulate effects of climate change on the terrestrial water balance. 21st-century continental- and basin-scale runoff is projected by some models to either increase more or decrease less when CO₂-induced increases in iWUE are included in addition to climate change (Betts et al., 2007; Murray et al., 2012), potentially reducing an increase in water stress due to rising population or climate change (Wiltshire et al., submitted) – although other models project a smaller response (Cao et al., 2009). Direct effects of CO₂ on plants have been modelled to increase future global runoff by 4–5% (Gerten et al., 2008) up to 13% (Nugent and Matthews, 2012), depending on the assumed CO₂ trajectory and whether feedbacks of changes in vegetation structure and distribution to the climate are accounted for. The model analysis by Alkama et al., (2010) suggests that although the physiological CO₂ effect will be the second-most important factor for 21st-century global runoff and although both physiological and structural effects will amplify compared to historic conditions, runoff changes will still primarily follow the projected climatic changes. Using a large ensemble of climate change projections, Konzmann et al., 2013 put hydrological changes into an agricultural perspective and suggest that direct CO₂ effects on crops reduce their irrigation requirements (Fig. CC-VW-1). Thus, adverse climate change impacts on crop yields might be partly buffered as iWUE improves (Fader et al., 2010), but only if proper management abates limitation of plant growth by nutrient availability or other factors. Lower transpiration under rising CO₂ may also affect future regional climate change itself (Boucher et al., 2009) and may enhance the contrast between land and ocean surface warming (Joshi et al., 2008).

Application of a soil-vegetation-atmosphere-transfer model indicates complex responses of groundwater recharge to changes in different climatic variables mediated by vegetation, with computed groundwater recharge being always larger than would be expected from just accounting for changes in rainfall (McCallum et al., 2010). In a warmer climate with increased atmospheric CO₂ concentration, iWUE of plants increases and leaf area may either increase or decrease, and even though precipitation may slightly decrease, groundwater recharge may increase as a net effect of these interactions (Crosbie et al., 2010). Depending on the type of grass in Australia, the same change in climate is suggested to lead to either increasing or decreasing groundwater recharge in this location (Green et al., 2007). For a location in the Netherlands, a biomass decrease was computed for each of eight climate scenarios indicating drier summers and wetter winters (A2 emissions scenario), using a fully coupled vegetation and variably saturated hydrological model. The resulting increase in groundwater recharge up-slope was simulated to lead to higher water tables and an extended habitat for down-slope moisture-adapted vegetation (Brolsma et al., 2010).

Future anthropogenic and climate-driven land cover and land use changes will also affect regional evapotranspiration, surface and subsurface water flows, with the direction and magnitude of these changes...
depending on the direction and intensity of the changes in vegetation coverage, as shown e.g. for a river basin in Iowa (Schilling et al., 2008) or for the Elbe river basin (Conradt et al., 2012). Removal of vegetation acting as source of atmospheric moisture can change regional water cycling and decrease potential crop yields by up to 17% in regions otherwise receiving this moisture in the form of precipitation (Bagley et al., 2012). Changes in vegetation coverage and structure due to long-term climate change or shorter-term extreme events such as droughts (Anderegg et al., 2013) also affect the partitioning of precipitation into evapotranspiration and runoff, sometimes involving complex feedbacks with the climate system such as in the Amazon region (Port et al., 2012; Saatchi et al., 2013). As water, carbon and vegetation dynamics evolve synchronously and interactively under climate change (Heyder et al., 2011) in that e.g. vegetation structure and composition can dynamically adapt to changing climatic and hydrologic conditions (Gerten et al., 2007), it remains a challenge to disentangle the effects of future land cover changes on the water cycle.

[INSERT FIGURE VW-1 HERE]

Figure VW-1: Percentage change (ensemble median across 19 GCMs used to force a vegetation and hydrology model) in net irrigation requirements of 12 major crops by the 2080s, assuming current extent of irrigation areas and current management practices. Top: impacts of climate change only; bottom: additionally considering physiological and structural crop responses to increased atmospheric CO₂ concentration. Taken from Konzmann et al. (2013).]

CC-VW References


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Green, T.R., Bates, B.C., Charles, S.P., and Fleming, P.M., 2007: Physically Based Simulation of Potential Effects of Carbon Dioxide–Altered Climates on Groundwater Recharge All rights reserved. No part of this periodical may be reproduced or transmitted in any form or by any means, electronic or mechanical, including photocopying, recording, or any information storage and retrieval system, without permission in writing from the publisher. * Vadose Zone J.*, 6(3), 597-609.


Water, energy, and food are linked through numerous interactive pathways and subject to a changing climate, as depicted in Figure CC-WE-1. The depth and intensity of those linkages vary enormously between regions and production systems. Some energy technologies (biofuels, hydropower, thermal power plants), transportation fuels and modes and food products (from irrigated crops, in particular animal protein produced by feeding irrigated crops) require more water than others (Chapter 3.7.2, 7.3.2, 10.2, 10.3.4, McMahon and Price, 2011, Macknick et al, 2012a, Cary and Weber 2008). In irrigated agriculture, climate, crop choice and yields determine water requirements per unit of produced crop, and in areas where water must be pumped or treated, energy must be provided (Kahn and Hajra 2009, Gerten et al. 2011). While food production and transport require large amounts of energy (Pelletier et al 2011), a major link between food and energy as related to climate change is the competition of bioenergy and food production for land and water (7.3.2, Diffenbaugh et al 2012, Skaggs et al, 2012).

Box CC-WE. The Water-Energy-Food Nexus as Linked to Climate Change

[DoDouglass J. Arent (USA), Petra Döll (Germany), Ken Strzepek (UNU/USA), Ferenc Toth (IAEA/Hungary), Blanca Elena Jimenez Cisneros (Mexico), Taikan Oki (Japan)]

Water, energy, and food are linked through numerous interactive pathways and subject to a changing climate, as depicted in Figure CC-WE-1. The depth and intensity of those linkages vary enormously between regions and production systems. Some energy technologies (biofuels, hydropower, thermal power plants), transportation fuels and modes and food products (from irrigated crops, in particular animal protein produced by feeding irrigated crops) require more water than others (Chapter 3.7.2, 7.3.2, 10.2, 10.3.4, McMahon and Price, 2011, Macknick et al, 2012a, Cary and Weber 2008). In irrigated agriculture, climate, crop choice and yields determine water requirements per unit of produced crop, and in areas where water must be pumped or treated, energy must be provided (Kahn and Hajra 2009, Gerten et al. 2011). While food production and transport require large amounts of energy (Pelletier et al 2011), a major link between food and energy as related to climate change is the competition of bioenergy and food production for land and water (7.3.2, Diffenbaugh et al 2012, Skaggs et al, 2012).

Most energy production methods require significant amounts of water, either directly (e.g. crop-based energy sources and hydropower) or indirectly (e.g., cooling for thermal energy sources or other operations) (Chapter 10.2.2 and 10.3.4, and Davies et al 2013, van Vliet et al 2012). Water is also required for mining, processing, and residue disposal of fossil fuels. Water for biofuels, for example, has been reported by Gerbens-Leenes et al. 2012 who computed a scenario of water use for biofuels for transport in 2030 based on the Alternative Policy Scenario of the IEA. Under this scenario, global consumptive irrigation water use for biofuel production is projected to increase from 0.5% of global renewable water resources in 2005 to 5.5% in 2030, resulting in increased pressure on freshwater resources, with potential negative impacts on freshwater ecosystems. Water for energy currently ranges from a few percent to more than 50% of freshwater withdrawals, depending on the region and future water requirements will depend on electric demand growth, the portfolio of generation technologies and water management options employed (WEC 2010, Sattler et al., 2012). Future water availability for energy production will change due to climate change (Chapter 3.5.2.2).

Water may require significant amounts of energy for lifting, transport and distribution, treatment or desalination. Non-conventional water sources (wastewater or seawater) are often highly energy intensive. Energy intensities per m³ of water vary by about a factor of 10 between different sources, e.g. locally produced or reclaimed wastewater vs. desalinated seawater (Plappally and Lienhard 2012, Macknick et al, 2012b). Groundwater (35% of total global water withdrawals, with irrigated food production being the largest user, Döll et al. 2012) is generally more energy intensive than surface water – in some countries, 40% of total energy use is for pumping groundwater. Pumping from greater depth (following falling groundwater tables) increases energy demand significantly – electricity use
(kWhr/m^3) increases by a factor of 3 when going from 35 to 120 m depth (Plappally and Lienhard 2012). A lack of water security can lead to increasing energy demand and vice versa, e.g. over-irrigation in response to electricity or water supply gaps.

Other linkages through land use and management, e.g. afforestation, can affect water as well as other ecosystem services, climate and water cycles (4.4.4, Box 25-10). Land degradation often reduces efficiency of water and energy use (e.g. resulting in higher fertilizer demand and surface runoff), and many of these interactions can compromise food security (3.7.2, 4.4.4). Only a few reports have begun to evaluate the multiple interactions among energy, food, land, and water (McCornick et al., 2008, Bazilian et al., 2011, Bierbaum and Matson, 2013), addressing the issues from a security standpoint and describing early integrated modeling approaches. The interaction among each of these factors is influenced by the changing climate, which in turn impacts energy demand, bioproductivity and other factors (see Figure WE-1 and Wise et al, 2009), and has implications for security of supplies of energy, food and water, adaptation and mitigation pathways, air pollution reduction as well as the implications for health and economic impacts as described throughout this Assessment Report.

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Table 4-1: Biome shifts from 1700 to the present from published field research that examined trends over periods > 30 y for biomes in areas where climate (rather than land-use change or other factors) predominantly influenced vegetation, derived from a systematic analysis of published studies (Gonzalez et al. (2010). Pre-AR4 publications are included to provide a comprehensive review. Shift type: elevational (E), latitudinal (L), examined but not detected (N). Biomes (and abbreviations), from poles to equator: tundra and alpine (UA), boreal conifer forest (BC), temperate conifer forest (TC), temperate broadleaf forest (TB), temperate shrubland (TS), tropical grassland (RG), tropical woodland (RW). Rate of change in temperature (Temp.) and fractional rate of change in precipitation (Precip.) are derived from linear least squares regression of 1901-2002 data (Mitchell and Jones, 2005; Gonzalez et al., 2010). The table provides general regional climate trends at 50 km spatial resolution because the references do not give uniform site-specific climate data to compare across locations. The regional trends are consistent with local trends reported in each reference. * rate significant at P ≤ 0.05).

<table>
<thead>
<tr>
<th>Location</th>
<th>Reference</th>
<th>Plots</th>
<th>Time Period</th>
<th>Shift type</th>
<th>Retracting biome</th>
<th>Expanding biome</th>
<th>Temp. change (°C century⁻¹)</th>
<th>Precip. change (century⁻¹)</th>
</tr>
</thead>
<tbody>
<tr>
<td>1. Alaska Range, Alaska, USA</td>
<td>Lloyd and Fastie, 2003</td>
<td>18</td>
<td>1800-2000</td>
<td>L</td>
<td>UA</td>
<td>BC</td>
<td>1.1*</td>
<td>0.03</td>
</tr>
<tr>
<td>2. Baltic Coast, Sweden</td>
<td>Walther et al., 2005</td>
<td>7</td>
<td>1944-2003</td>
<td>L</td>
<td>TC</td>
<td>TB</td>
<td>0.6*</td>
<td>0.08</td>
</tr>
<tr>
<td>3. Becca di Viou, Italy</td>
<td>Leonelli et al., 2011</td>
<td>1</td>
<td>1700-2008</td>
<td>E</td>
<td>UA</td>
<td>BC</td>
<td>0.9*</td>
<td>-0.06</td>
</tr>
<tr>
<td>4. Garibaldi, British Columbia, Canada</td>
<td>Brink, 1959</td>
<td>1</td>
<td>1860-1959</td>
<td>E</td>
<td>UA</td>
<td>BC</td>
<td>0.7*</td>
<td>0.16*</td>
</tr>
<tr>
<td>5. Goulet Sector, Québec, Canada</td>
<td>Payette and Filion, 1985</td>
<td>2</td>
<td>1880-1980</td>
<td>E</td>
<td>UA</td>
<td>BC</td>
<td>1.4*</td>
<td>0.19*</td>
</tr>
<tr>
<td>6. Green Mountains, Vermont, USA</td>
<td>Beckage et al., 2008</td>
<td>33</td>
<td>1962-2005</td>
<td>E</td>
<td>BC</td>
<td>TB</td>
<td>1.6*</td>
<td>0.06</td>
</tr>
<tr>
<td>7. Jasper, Alberta, Canada</td>
<td>Luckman and Kavanagh, 2000</td>
<td>1</td>
<td>1700-1994</td>
<td>E</td>
<td>UA</td>
<td>BC</td>
<td>0.6</td>
<td>0.21*</td>
</tr>
<tr>
<td>8. Kenai Mountains, Alaska, USA</td>
<td>Dial et al., 2007</td>
<td>3</td>
<td>1951-1996</td>
<td>E</td>
<td>UA</td>
<td>BC</td>
<td>0.7</td>
<td>0.06</td>
</tr>
<tr>
<td>9. Kluane Range, Yukon, Canada</td>
<td>Danby and Hik, 2007</td>
<td>2</td>
<td>1800-2000</td>
<td>E</td>
<td>UA</td>
<td>BC</td>
<td>0.7</td>
<td>0.05</td>
</tr>
<tr>
<td>10. Low Peninsula, Québec, Canada</td>
<td>Payette and Filion, 1985</td>
<td>1</td>
<td>1750-1980</td>
<td>N</td>
<td>-</td>
<td>-</td>
<td>1.4*</td>
<td>0.19*</td>
</tr>
<tr>
<td>11. Mackenzie Mountains, Northwest Territories, Canada</td>
<td>Szeicz and Macdonald, 1995</td>
<td>13</td>
<td>1700-1990</td>
<td>N</td>
<td>-</td>
<td>-</td>
<td>1.4*</td>
<td>0.03</td>
</tr>
<tr>
<td>12. Montseny Mountains, Catalonia, Spain</td>
<td>Peñuelas and Boada, 2003</td>
<td>50</td>
<td>1945-2001</td>
<td>E</td>
<td>UA</td>
<td>TB</td>
<td>1.2*</td>
<td>-0.03</td>
</tr>
<tr>
<td>13. Napaktok Bay, Labrador, Canada</td>
<td>Payette, 2007</td>
<td>2</td>
<td>1750-2000</td>
<td>L</td>
<td>UA</td>
<td>BC</td>
<td>1.1*</td>
<td>0.05</td>
</tr>
<tr>
<td>14. Noatak, Alaska, USA</td>
<td>Suarez et al., 1999</td>
<td>18</td>
<td>1700-1990</td>
<td>L</td>
<td>UA</td>
<td>BC</td>
<td>0.6</td>
<td>0.19*</td>
</tr>
<tr>
<td>Location</td>
<td>Reference</td>
<td>Plots</td>
<td>Time Period</td>
<td>Shift type</td>
<td>Retracting biome</td>
<td>Expanding biome</td>
<td>Temp. change (°C century^{-1})</td>
<td>Precip. change (century^{-1})</td>
</tr>
<tr>
<td>----------------------------------------------</td>
<td>--------------------------------</td>
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<td>-------------------------------</td>
</tr>
<tr>
<td>Putorana Mountains, Russia</td>
<td>Kirdyanov et al., 2012</td>
<td>10</td>
<td>1500-2000</td>
<td>E</td>
<td>UA</td>
<td>BC</td>
<td>0.3</td>
<td>0.10</td>
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<td>Rahu Saddle, New Zealand</td>
<td>Cullen et al., 2001</td>
<td>7</td>
<td>1700-2000</td>
<td>N</td>
<td>-</td>
<td>-</td>
<td>0.6*</td>
<td>0.03</td>
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<tr>
<td>Rai-Iz, Urals, Russia</td>
<td>Devi et al., 2008</td>
<td>144</td>
<td>1700-2002</td>
<td>E</td>
<td>UA</td>
<td>BC</td>
<td>0.3</td>
<td>0.35*</td>
</tr>
<tr>
<td>Sahel, Sudan, Guinea zones; Senegal</td>
<td>Gonzalez, 2001</td>
<td>135</td>
<td>1945-1993</td>
<td>L</td>
<td>RW</td>
<td>RG</td>
<td>0.4*</td>
<td>-0.48*</td>
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<tr>
<td>Sahel, Burkina Faso, Chad, Mali, Mauritania, Niger</td>
<td>Gonzalez et al., 2012</td>
<td>14</td>
<td>1960-2000</td>
<td>L</td>
<td>RW</td>
<td>RG</td>
<td>0.01* to 0.8*</td>
<td>-0.31* to 0.09</td>
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<tr>
<td>Scandes, Sweden</td>
<td>Kullman and Öberg, 2009</td>
<td>123</td>
<td>1915-2007</td>
<td>E</td>
<td>UA</td>
<td>BC</td>
<td>0.8*</td>
<td>0.25*</td>
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<td>Sierra Nevada, California, USA</td>
<td>Millar et al., 2004</td>
<td>10</td>
<td>1880-2002</td>
<td>E</td>
<td>UA</td>
<td>TC</td>
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<td>South Island, New Zealand</td>
<td>Wardle and Coleman, 1992</td>
<td>22</td>
<td>1980-1990</td>
<td>E</td>
<td>TS</td>
<td>TB</td>
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<td>0.03</td>
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<td>Yambarran, Northern Territory, Australia</td>
<td>Sharp and Bowman, 2004</td>
<td>33</td>
<td>1948-2000</td>
<td>N</td>
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<td>-</td>
<td>0.06</td>
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</tbody>
</table>
Table 4-2: Summary of drivers and outcomes of land use / cover scenarios associated with Representative Concentration Pathways (Hurtt et al., 2011).

<table>
<thead>
<tr>
<th>RCP</th>
<th>Key assumptions / drivers</th>
<th>Land use / cover outcomes</th>
</tr>
</thead>
<tbody>
<tr>
<td>8.5-MESSAGE</td>
<td>No climate change mitigation actions; radiative forcing still rising at 2100</td>
<td>Increase in cultivated land by about 305 million ha from 2000 to 2100</td>
</tr>
<tr>
<td></td>
<td>Strong increase in agricultural resource use driven by the increasing population (rises to 12 billion people by 2100)</td>
<td>Forest cover declines by 450 million ha from 2000 to 2100</td>
</tr>
<tr>
<td></td>
<td>Yield improvements and intensification assumed to account for most of production increases</td>
<td>Arable land use in developed countries slightly decreased - all of the net increases occur in developing countries.</td>
</tr>
<tr>
<td>6.0-AIM</td>
<td>Mitigation actions taken late in the century to stabilize radiative forcing at 6 Wm(^{-2}) after 2100</td>
<td>Urban land-use increases</td>
</tr>
<tr>
<td></td>
<td>Population growth and economic growth</td>
<td>Cropland area expands</td>
</tr>
<tr>
<td></td>
<td>Increasing food demand drives cropland expansion</td>
<td>Grassland area declines</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Total forested area extent remains constant</td>
</tr>
<tr>
<td>4.5-GCM</td>
<td>Mitigation stabilizes radiative forcing at 4.5 Wm(^{-2}) before 2100</td>
<td>Preservation of large stocks of terrestrial carbon in forests</td>
</tr>
<tr>
<td></td>
<td>Assumes that global GHG emissions prices are invoked to limit emissions and therefore radiative forcing. Emissions pricing assumes all carbon emissions charged an equal penalty price, so reductions in land-use change carbon emissions available as mitigation</td>
<td>Overall expansion in forested area</td>
</tr>
<tr>
<td></td>
<td>Food demand met through crop yield improvements, dietary shifts, production efficiency and international trade.</td>
<td>Agricultural land declines slightly due to afforestation,</td>
</tr>
<tr>
<td>2.6-IMAGE</td>
<td>Overall trends in land use and land cover mainly determined by demand, trade and production of agricultural products and bio-energy</td>
<td>Much agriculture relocates from high income to low income regions</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Increase in bio-energy production, new area for bioenergy crops near current agricultural areas.</td>
</tr>
</tbody>
</table>
Table References Chapter 4 SOD


Figure 4-1: Biome shifts in the past.

Figure 4-2: Fraction of land surface at risk of severe ecosystem change as a function of global mean temperature change for all ecosystems models, global climate models and RCPs.
Figure 4-3: Proportion of global land cover occupied by primary and secondary vegetation (forest and non-forest), cropland, pasture and urban land, from satellite data and historical reconstructions up to 2005, and from scenarios associated with the RCPs from 2005 to 2100.

Figure 4-4: Fractional cover of primary vegetation at 1850 and 2005 based on satellite data and historical reconstructions, and at 2100 in scenarios associated with the RCPs.
Figure 4-5: Confidence in Detection and Attribution of observed responses of terrestrial ecosystems to climate change.
Figure 4-6: Rate of climate change, climate velocity and rate of displacement
Figure 4-7: Vulnerability of terrestrial biomes to future climate change.
Forest Fire Danger Index (FFDI)

- 0-5: Low
- 5-12: Moderate
- 12-24: High
- 24-50: Very High
- 50-100: Extreme

Figure 4-8: Forest Fire Danger Index simulated with the HadGEM2-ES Earth System Model.
Figure 4-9: Forest mortality related to climatic stress.
Figure 4-10: Amazon Die back.
Figure 4-11: Simulations of past and future northern hemisphere permafrost area with a maximum thaw depth less than 3m deep.
Figure 4-12: Tundra biome shift.
Figure EA-1: Adapted from Munang et al. (2013). Ecosystem based adaptation approaches to adaptation can utilize the capacity of nature to buffer human systems from the adverse impacts of climate change through sustainable delivery of ecosystems services. A) Business as Usual Scenario in which climate impacts degrade ecosystems, ecosystem service delivery and human well-being B) Ecosystem-based Adaptation Scenario which utilizes natural capital and ecosystem services to reduce climate-related risks to human communities.
Figure RF-1: Impact of climate change on the ecologically relevant river flow characteristics mean annual river flow and monthly low flow $Q_{90}$ as compared to the impact of water withdrawals and dams on natural flows, as computed by a global water model (Döll and Zhang, 2010). Impact of climate change is the percent change of flow between 1961-1990 and 2041-2070 according to the emissions scenario A2 as implemented by the global climate model HadCM3. Impact of water withdrawals and reservoirs is computed by running the model with and without water withdrawals and dams that existed in 2002.
Figure RF-2: Accumulated loss of regional species richness (gamma diversity) as a function of glacial cover GCC. Obligate glacial river macroinvertebrates begin to disappear from assemblages when glacial cover in the catchment drops below approximately 50%. Each data point represents a river site and lines are Lowess fits. Adapted by permission from Macmillan Publishers Ltd: *Nature Climate Change*, Jacobsen et al., 2012, © 2012.
Figure VW-1: Percentage change (ensemble median across 19 GCMs used to force a vegetation and hydrology model) in net irrigation requirements of 12 major crops by the 2080s, assuming current extent of irrigation areas and current management practices. Top: impacts of climate change only; bottom: additionally considering physiological and structural crop responses to increased atmospheric CO₂ concentration. Taken from Konzmann et al. (2013).
Figure WE-1: The water-energy-food nexus as related to climate change.