Chapter 4. Terrestrial and Inland Water Systems

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Executive Summary

The planet's biota and ecosystem processes were strongly affected by past climate changes at rates of climate change lower than those projected during the 21st century under high warming scenarios (e.g., RCP8.5) (*high confidence*). Most ecosystems are vulnerable to climate change even at rates of climate change projected under low- to medium-range warming scenarios (e.g., RCP2.6 to RCP6.0). The paleoecological record shows that global climate changes comparable in magnitudes to those projected for the 21st century under all scenarios resulted in large-scale biome shifts and changes in community composition; and that for rates projected under RCP6 and 8.5, was associated with species extinctions in some groups (*high confidence*). [4.2.3]

Climate change is projected to be a powerful stressor on terrestrial and freshwater ecosystems in the second half of the 21st century, especially under high-warming scenarios such as RCP6.0 and RCP8.5 (*high confidence*). Direct human impacts such as land-use and land use change, pollution and water resource development will continue to dominate the threats to most freshwater (*high confidence*) and terrestrial (*medium confidence*) ecosystems globally over the next three decades. **Changing climate exacerbates other impacts on biodiversity (*high confidence*).** Ecosystem changes resulting from climate change may not be fully apparent for several decades, due to long response times in ecological systems (*medium confidence*). Model-based projections imply that under low to moderate warming scenarios (e.g., RCP2.6 to RCP6.0), direct land cover change will continue to dominate over (and conceal) climate-induced change as a driver of ecosystem change at the global scale; for higher climate change scenarios, some model projections imply climate-driven ecosystem changes sufficiently extensive to equal or exceed direct human impacts at the global scale (*medium confidence*). In high altitude and high latitude freshwater and terrestrial ecosystems, climate changes exceeding those projected under RCP2.6 will lead to major changes in species distributions and ecosystem function, especially in the second half of the 21st century (*high confidence*). [4.2.4, 4.3.2.5, 4.3.3, 4.3.3.1, 4.3.3.3, 4.4.1.1]

When terrestrial ecosystems are substantially altered (in terms of plant cover, biomass, phenology or plant group dominance), either through the effects of climate change or through other mechanisms such as conversion to agriculture or human settlement, the local, regional and global climates are also affected (*high confidence*). The feedbacks between terrestrial ecosystems and climate include, among other mechanisms, changes in surface albedo, evapotranspiration and greenhouse gas emissions and uptake. The physical effects on the climate can be opposite in direction to the greenhouse gas effects, and can materially alter the net outcome of the ecosystem change on the global climate (*high confidence*). The regions where the climate is affected may extend beyond the location of the ecosystem that has changed. [4.2.4.1, 4.3.3.4]

Rising water temperatures, due to global warming, will lead to shifts in freshwater species distributions and worsen water quality problems, especially in those systems experiencing high anthropogenic loading of nutrients (*high confidence*). Climate change-induced changes in precipitation will substantially alter ecologically important attributes of flow regimes in many rivers and wetlands and exacerbate impacts from human water use in developed river basins (*medium confidence*). [4.3.3.3, Box CC-RF]

Many plant and animal species have moved their ranges, altered their abundance and shifted their seasonal activities in response to observed climate change over recent decades (*high confidence*). They are doing so now in many regions and will continue to do so in response to projected future climate change (*high confidence*). The broad patterns of species and biome shifts towards the poles and higher in altitude in response to a warming climate are well established for periods thousands of years in the past (*very high confidence*). These general patterns of range shifts have also been observed over the last few decades in some well-studied species groups such as insects and birds and can be attributed to observed climatic changes (*high confidence*). Interactions between changing temperature, precipitation and land use can sometimes result in range shifts that are downhill or away from the poles. Certainty regarding past species movements in response to changing climate, coupled with projections from a variety of models and studies, provide *high confidence* that such species movements will be the norm with continued warming. Under all RCP climate change scenarios for the second half of the 21st century, with *high confidence*: a) community composition will change as a result of decreases in the abundances of some species and increases in others; b) the seasonal activity of many species will change differentially disrupting life cycles and interactions between species. Composition and seasonal change will both alter ecosystem function. **[**4.2.1, 4.2.3, 4.3.2, 4.3.2.1, 4.3.2.5, 4.3.3, 4.4.1.1]

Many species will be unable to move fast enough during the 21st century to track suitable climates under mid- and high-range rates of climate change (i.e., RCP 4.5, RCP 6.0 and RCP 8.5 scenarios) (*medium confidence*). The climate velocity (the rate of movement of the climate across the landscape) will exceed the maximum velocity at which many groups of organisms, in many situations, can disperse or migrate, except after mid-century in the RCP 2.6 scenario. Populations of species that cannot keep up with their climate niche will find themselves in unfavourable climates, unable to reach areas of potentially suitable climate. Species occupying extensive flat landscapes are particularly vulnerable because they must disperse over longer distances to keep pace with shifting climates than species in mountainous regions. Species with low dispersal capacity will also be especially vulnerable:

examples include many plants (especially trees), many amphibians and some small mammals. For example, the maximum observed and modelled dispersal and establishment rates for mid- and late-successional tree species are insufficient to track climate change except in mountainous areas, even at moderate projected rates of climate change. Barriers to dispersal, such as habitat fragmentation, prior occupation of habitat by competing species and human-made impediments such dams on rivers and urbanized areas on land, reduce the ability of species to migrate to more suitable climates (*high confidence*). Intentional and accidental anthropogenic transport can speed dispersal. [4.3.2.5, 4.3.3.3]

Large magnitudes of climate change will reduce the populations, vigour and viability of species with spatially-restricted populations, such as those confined to small and isolated habitats, mountaintops or mountain streams, even if the species has the biological capacity to move fast enough to track suitable climates (*high confidence*). The adverse effects on restricted populations are modest for low magnitudes of climate change (e.g., RCP 2.6) but very severe for the highest magnitudes of projected climate change (e.g., RCP 8.5). [4.3.2.5, 4.3.3.4, 4.3.4.1]

The capacity of many species to respond to climate change will be constrained by non-climate factors (*high confidence*), including but not limited to the simultaneous presence of inhospitable land-uses, habitat fragmentation and loss, competition with alien species, exposure to new pests and pathogens, nitrogen loading and tropospheric ozone. [4.2.4.6, 4.3.3.5, Figure 4-1]

The establishment, growth, spread and survival of populations of invasive alien species has increased (*high confidence*), but the ability to attribute alien species invasion to climate change is low in most cases. Some invasive alien species have traits that favour their survival and reproduction under changing climates. Future movement of species into areas where they were not present historically will continue to be mainly driven by increased dispersal opportunities associated with human activities and by increased disturbances from natural and anthropogenic events, in some cases facilitated and promoted by climate change. [4.2.4.6, Figure 4-4]

A large fraction of terrestrial and freshwater species face increased extinction risk under projected climate change during and beyond the 21st century, especially as climate change interacts with other pressures, such as habitat modification, over-exploitation, pollution and invasive species (*high confidence*). The extinction risk is increased under all RCP scenarios, and the risk increases with both the magnitude and rate of climate change. While there is *medium confidence* that recent warming contributed to the extinction of many species of Central American amphibians, there is generally *very low confidence* that observed species extinctions can be attributed to recent climate change. Models project that the risk of species extinctions will increase in the future due to climate change, but there is *low agreement* concerning the fraction of species at increased risk, the regional and taxonomic focus for such extinctions and the timeframe over which extinctions could occur. Modelling studies and syntheses since the AR4 report broadly confirm that a large proportion of species are projected to be at increased risk of extinction at all but the lowest levels of climate warming (RCP2.6). Some aspects leading to uncertainty in the quantitative projections of extinction risks were not taken into account in previous models; as more realistic details are included, it has been shown that the extinction risks may be either under- or overestimated when based on simpler models. [4.3.2.5]

Terrestrial and freshwater ecosystems have sequestered about a quarter of the carbon dioxide emitted to the atmosphere by human activities in the past three decades (*high confidence*). 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 rate at which carbon has been taken up by terrestrial ecosystems at the global scale has changed between the decades 1991-2000 and 2001-2010. There is *high confidence* that the factors causing the current increase in land carbon include the positive effects of rising CO_2 on plant productivity, a warming climate, nitrogen deposition 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, 4.3.2.3, WGI AR5 Sections 6.3.1 and 6.3.2.6]

The natural carbon sink provided by terrestrial ecosystems is partially offset at the decadal timescale by carbon released through the conversion of natural ecosystems (principally forests) to farm and grazing land

and through ecosystem degradation (*high confidence*). Carbon stored in the terrestrial biosphere is vulnerable to loss back to the atmosphere as a result of the direct and indirect effects of climate change, deforestation and degradation (*high confidence*). The net transfer of carbon dioxide from the atmosphere to the land is projected to weaken during the 21st century (*medium confidence*). The direct effects of climate change on stored terrestrial carbon include high temperatures, drought and windstorms, indirect effects include increased risk of fires, pest and disease outbreaks. Experiments and modelling studies provide *medium confidence* that increases in CO_2 up to about 600 ppm will continue to enhance photosynthesis and plant water-use efficiency, but at a diminishing rate; and *high confidence* that low availability of nutrients, particularly nitrogen, will limit the response of many natural ecosystems to rising CO_2. There is *medium confidence* that other factors associated with global change, including high temperatures, rising ozone concentrations and in some places drought, decrease plant productivity by amounts comparable in magnitude to the enhancement by rising CO_2. There are few field-scale experiments on ecosystems at the highest CO_2 concentrations projected by RCP 8.5 for late in the century, and none of these include the effects of other potential confounding factors. [4.2.4, 4.2.4.1, 4.2.4.2, 4.2.4.3, 4.2.4.4, 4.3.2.2, 4.3.3.1, Box 4-3, Box CC-VW, WGI AR5 6.4.3.3]

Increases in the frequency or intensity of ecosystem disturbances such as droughts, wind-storms, fires and pest outbreaks have been detected in many parts of the world and in some cases are attributed to climate change (*medium confidence*). Changes in the ecosystem disturbance regime beyond the range of natural variability will alter the structure, composition and functioning of ecosystems (*high confidence*). Ecological theory and experimentation predicts that ecological change resulting from altered disturbance regimes will be manifested as relatively abrupt and spatially-patchy transitions in ecosystem structure, composition and function, rather than gradual and spatially-uniform shifts in location or abundance of species (*medium confidence*). [4.2.4.6, 4.3.3, 4.3.2.5, Box 4-3, Box 4-4, Figure 4-10]

Increased tree death has been observed in many places worldwide, and in some regions has been attributed to climate change (high confidence). In some places it is sufficiently intense and widespread as to result in forest dieback (*low confidence*). Forest dieback is a major environmental risk with potentially large impacts on climate, biodiversity, wood production, water quality, amenity and economic activity. In detailed regional studies in western and boreal North America, the tree mortality observed over the past few decades has been attributed to the effects of high temperatures and drought, or to changes in the distribution and abundance of insect pests and pathogens related, in part, to warming *(high confidence)***.** Tree mortality and associated forest dieback will become apparent in many regions sooner than previously anticipated (*medium confidence*). Earlier projections of increased tree growth and enhanced forest C sequestration due to increased growing season duration, rising CO_2 concentration and atmospheric N deposition, must be balanced by observations and projections of increasing tree mortality and forest loss due to fires and pest attacks. 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 *versus* negative effects of global change. [4.3.2, 4.3.3.1, 4.3.3.4, 4.3.3.5, 4.3.4, 4.3.4.2, Box 4-2, Box 4-3]

There is a high risk that the large magnitudes and high rates of climate change associated with low-mitigation climate scenarios (RCP4.5 and higher) will result within this century in abrupt and irreversible regional-scale change in the composition, structure and function of terrestrial and freshwater ecosystems, especially in the Amazon and Arctic, leading to substantial additional climate change (*medium confidence*). There are plausible mechanisms, supported by experimental evidence, observations, and model results, for the existence of ecosystem tipping points in both boreal-tundra Arctic systems and the rainforests of the Amazon basin. Continued climate change will transform the species composition, land cover, drainage and permafrost extent of the boreal-tundra system, leading to decreased albedo and the release of greenhouse gases (*medium confidence*). Adaptation measures will be unable to prevent substantial change in the boreal-arctic system (*high confidence*). Climate change alone is not projected to lead to abrupt widespread loss of forest cover in the Amazon during this century a (*medium confidence*), but a projected increase in severe drought episodes, together with land-use change and forest fire, would cause much of the Amazon forest to transform to less dense, drought- and fire-adapted ecosystems, and in doing so, put a large stock of biodiversity at elevated risk, while decreasing net carbon uptake from the atmosphere (*medium confidence*)**.** Large reductions in deforestation, as well as wider application of effective wildfire management, lower the risk of abrupt change in the Amazon, as well as the impacts of that change (*medium confidence*). [4.2.4.1, 4.3.3.1.1, 4.3.3.1.3, 4.3.3.4, Figure 4-8, Box 4-3, Box 4-4]

Management actions can reduce, but not eliminate, the risk of impacts to terrestrial and freshwater ecosystems due to climate change, as well as increase the inherent capacity of ecosystems and their species to adapt to a changing climate (*high confidence*). The capacity for natural adaptation by ecosystems and their constituent organisms is substantial, but for many ecosystems and species it will be insufficient to cope with projected rates and magnitudes of climate change in the 21st century without substantial loss of species and ecosystem services, under medium-range warming (e.g., RCP6.0) or high–range warming scenarios (e.g., RCP8.5) (*medium confidence*). 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 climate 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, 4.4.1, 4.4.2]

Adaptation responses to climate change in the urban and agricultural sectors can have unintended negative outcomes for terrestrial and freshwater ecosystems (*medium confidence*). For example, adaptation responses to counter increased variability of water supply, such as building more and larger impoundments and increased water extraction, will in many cases worsen the direct effects of climate change in freshwater ecosystems**.** [4.3.3.3, 4.3.4.6]

Widespread transformation of terrestrial ecosystems in order to mitigate climate change, such as carbon sequestration through planting fast-growing tree species into ecosystems where they did not previously occur, or the conversion of previously uncultivated or non-degraded land to bioenergy plantations, will lead to negative impacts on ecosystems and biodiversity (*high confidence*). For example, the land use scenario accompanying the mitigation scenario RCP2.6 features a large expansion of biofuel production, displacing natural forest cover. [4.2.4.1, 4.4.4]

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), but also Chapter 1 (Assessment of observed changes and responses in natural and managed systems, Sections 1.3.4 and 1.3.5; Rosenzweig *et al.*, 2007). The WGII SPM said "Observational evidence from all continents and most oceans shows that many natural systems are being affected by regional climate changes, particularly temperature increases", though they noted that documentation of observed changes in tropical regions and the Southern Hemisphere was sparse (Rosenzweig *et al.*, 2007). Fischlin *et al.* (2007) found that 20-30% of the plant and animal species that had been assessed to that time were considered to be at increased risk of extinction if the global average temperature increase exceeds 2-3°C above the pre-industrial level *with medium confidence*, and that substantial changes in structure and functioning of terrestrial, marine and other aquatic ecosystems are *very likely* under that degree of warming and associated atmospheric CO_2 concentration. No timescale was associated with these findings. The carbon stocks in terrestrial ecosystems were considered to be at high risk from climate change and land use change. The report warned that the capacity of ecosystems to adapt naturally to the combined effect of climate change and other stressors is likely to be exceeded if greenhouse emission continued at or above the then-current rate.

4.2. A Dynamic and Inclusive View of Ecosystems

There are three aspects of the contemporary scientific view of ecosystems that are important to know for policy purposes. Firstly, ecosystems usually have imprecise and variable boundaries. They span a wide range of spatial scales, nested within one another, from the whole biosphere, down through its major ecosystem types (biomes) to local and possibly short-lived associations of organisms. Secondly, the human influence on ecosystems is globally pervasive. Humans are regarded as an integral, rather than separate, part of social-ecological systems (Gunderson and Holling, 2001; Berkes *et al.*, 2003). Ecosystems are connected across boundaries through the movement of energy, materials and organisms, and subsidies between terrestrial and freshwater systems are known to be particularly important (Polis *et al.*, 1997; Loreau *et al.*, 2003). As a consequence, human activities in terrestrial systems can significantly impact freshwater ecosystems and their biota (Allan, 2004). The dynamics of social-ecological systems 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). Thirdly, ecologists do not view ecosystems as necessarily inherently static and at equilibrium in the absence of a human disturbance (Hastings, 2004). Ecosystems vary over time and space in the relative magnitude of their components and fluxes, even under a constant environment, due to internal dynamics (Scheffer, 2009). Furthermore, attempts to restrict this intrinsic variation - or that resulting from externally-generated disturbances - are frequently futile, and may damage the capacity of the ecosystem to adapt to a changing environment (Folke *et al.*, 2004). This contrasts with the popular view that ecosystems exhibit a ‘balance of Nature’ and benefit from being completely protected from disturbance.

4.2.1. Ecosystems, Adaptation, Thresholds, and Tipping Points

The term “adaptation” has different meanings in climate policy, ecology and evolutionary biology. In climate policy (see glossary) it implies human actions intended to reduce negative outcomes. In ecology, ecosystems are said to be adaptive because their composition or function can change in response to a changing environment, without necessarily involving deliberate human actions (see Section 4.4.1). In evolutionary biology, adaptation means a change in the genetic properties of a population of individuals as a result of natural selection (Section 4.4.1.2); a possibility seen since the Fourth Assessment Report as increasingly relevant to climate change.

The notion of thresholds has become a prominent ecological and political concern (Lenton *et al.*, 2008; Knapp *et al.*, 2008a; Leadley *et al.*, 2010). To avoid policy confusion, three types of threshold need to be distinguished. The first reflects a human preference that the ecosystem stays within certain bounds, such as above a certain forest cover. These can be, by definition, negotiated. The second type reflects fundamental biological or physical properties, for instance the temperature at which frozen soils thaw (see Box 4-4) or the physiological tolerance limits of species. The third type is caused by system dynamics: the point at which the net effect of all the positive and negative feedback loops regulating the system is sufficiently large and positive that a small transgression becomes sufficiently amplified to lead to a change in ecosystem state called a regime shift (Lenton *et al.*, 2008). The new state exhibits 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” (defined in the glossary as 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) and is important in the context of climate change because its onset may be abrupt, hard to predict precisely and effectively irreversible (Scheffer *et al.*, 2009; Leadley *et al.*, 2010; Barnosky *et al.*, 2012; Brook *et al.*, 2013; Hughes *et al.*, 2013). Many examples of tipping points have now been identified (Scheffer, 2009). Regional-scale ecosystem tipping points have not occurred in the recent past, but there is good evidence for tipping points in the distant past (Section 4.2.3) and there is concern that they could occur in the near future (see Boxes 4-3 and 4-4).

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 the threshold is crossed (Biggs *et al.*, 2009). The dynamics of ecosystems are complex and our present level of knowledge is inadequate to predict all ecosystem outcomes with confidence, even if the future climate was precisely known.

Field observations over the past century in numerous locations in boreal, temperate and tropical ecosystems have detected biome shifts, the replacement at a location of one suite of species by another (*high confidence*). The effect is usually of biomes moving upwards in elevation and to higher latitudes (Gonzalez *et al.*, 2010). These shifts have often been attributed to anthropogenic climate change, since biome distribution is known to broadly reflect climate zones, and the shifts have been observed in area without major human disturbance (*medium confidence*) (Table 4-1). Projections of future vegetation distribution under climate change indicate that many biomes could shift substantially, including in areas where ecosystems are largely undisturbed by direct human land use (Figure 4-2). The extent of the shift increases with increasing global mean warming, without a sudden threshold (Scholze *et al.*, 2006; Pereira *et al.*, 2010; Rehfeldt *et al.*, 2012).

[INSERT TABLE 4-1 HERE
Table 4-1: Biome shifts of the 20th century 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). The biome abbreviations match those in Figure 4-1. 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 $P \leq 0.05$.]

[INSERT FIGURE 4-1 HERE
Figure 4-1: Locations of observed biome shifts during the 20th century, listed in Table 4-1, derived from Gonzalez *et al.* (2010). The color of each semi-circle indicates the retracting biome (top for North America, Europe, Asia; bottom for Africa and New Zealand) and the expanding biome (bottom for North America, Europe, Asia; top for Africa and New Zealand), according to published field observations. Biomes, from poles to equator: ice (IC), tundra and alpine (UA), boreal conifer forest (BC), temperate conifer forest (TC), temperate broadleaf forest (TB), temperate mixed forest (TM), temperate shrubland (TS), temperate grassland (TG), desert (DE), tropical grassland (RG), tropical woodland (RW), tropical deciduous broadleaf forest (RD), tropical evergreen broadleaf forest (RE). The background is the potential biome according to the MC1 dynamic global vegetation model under the 1961-1990 climate.]

[INSERT FIGURE 4-2 HERE
Figure 4-2: Implications of climate change and land use change for biome shifts. (a) Fraction of land covered by primary vegetation in 2005 (Hurtt *et al.*, 2011); (b) Fraction of simulations showing climate change-driven biome shift for any level of global warming between 1990 and 2100, with no direct anthropogenic land use change, using the MC1 vegetation model under 9 CMIP3 climate projections (3 GCMs each forced by the SRES A2, A1B and B1 scenarios; Gonzalez *et al.*, 2010); (c) Fraction of land covered by primary vegetation in 2100 under the RCP2.6 land use scenario with the IMAGE model, with no effect of climate change (Hurtt *et al.*, 2011); (d) Fraction of land covered by primary vegetation in 2100 under the RCP8.5 land use scenario with the MESSAGE model, with no effect of climate change (Hurtt *et al.*, 2011). Comparison of coloured areas in (b) with those in (a) shows where climate- driven biome shifts would occur in current areas of primary vegetation. Comparison of (b) and (d) shows where climate-driven biome shifts would occur in areas of primary vegetation projected under a land use scenario associated with RCP6.0. Comparison of (c) and (a) illustrates a scenario of land use change associated with RCP2.6, in which global climate change is projected to be smaller than that driving the biome shifts in (b) as a result of mitigation measures, some of which involve land use. Further details of the RCP land use / cover scenarios are given in Box 4-1, Figure 4-3 and Table 4-2.]

4.2.2. Methods and Models Used

Analysis of the current and past impacts of climate change on terrestrial and freshwater ecosystems and their projection into the future relies on three general approaches: inference from analogous situations in the past or elsewhere in the present; manipulative experimentation, deliberately altering one of a few factors at a time; and models with a mechanistic or statistical basis. Studies of the relatively distant past are discussed in depth in 4.2.3. Inferences from present spatial patterns in relation to climate is at the core of climate envelope niche modelling, a well-established but limited statistical technique for making projections of the future distribution under equilibrium conditions (Elith and Leathwick, 2009). Representing the rate of change during the non-equilibrium conditions which will prevail over the next century requires a more mechanistic approach, of which there are some examples (e.g., Keith *et al.*, 2008; Kearney and Porter, 2009). Changes in ecosystem function are usually determined by experimentation (see examples in Section 4.3.3) and are modelled using mechanistic models, in many cases with relatively high uncertainty (Seppelt *et al.*, 2011).

4.2.3. Paleoecological Evidence

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; see WGI AR5 Chapter 5). Paleoecological records demonstrate with *high confidence* 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 that projected during the 21st century under mid- to high-end radiative forcing pathways (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, as well as recovery periods in excess of a million years, 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; Wing and Currano, 2013), 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) to Holocene transition between 21 and 6 ka (MacDonald *et al.*, 2008; Clark *et al.*, 2009; Gill *et al.*, 2009; Williams *et al.*, 2010a; Prentice *et al.*, 2011; Daniau *et al.*, 2012). The paleoecological record thus provides *high confidence* that large global climate change, comparable in magnitudes to that projected for the 21st century, can result in large ecological changes, including large scale biome shifts, reshuffling of communities and 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 approaching, for many regions, that projected for the 21st century for all RCPs (Alley *et al.*, 2003; Steffensen *et al.*, 2008). Ecosystems and species responded rapidly during the Younger Dryas 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 (extirpated), but there is no evidence for climate-driven global-scale extinctions during this period (Botkin *et al.*, 2007; Willis *et al.*, 2010a). However, the Younger Dryas climate changes 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. Regional mean warming during this period (mean annual temperature ca. 0.5-1.0°C above pre-industrial in some continental-scale regions; see WGI AR5 5.5.1) 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 warming was characterized by frequent fires in a drier parts of the 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, also clearly impacted inland water systems (e.g., Smol and Douglas, 2007a; Battarbee *et al.*, 2009; Beilman *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 and atmospheric CO_2 change projected for the next century and beyond (Jansen *et al.*, 2007; Schulte *et al.*, 2010; Wing and Currano, 2013; see WGI 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* that these interactions will 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 vegetation to large paleoclimate change (Prentice *et al.*, 1996; Salzmann *et al.*, 2008; Prentice *et al.*, 2011). The same types of models predict large changes in species ranges, ecosystem function and carbon storage when forced by 21st century climate change, although the future situation is complicated by land-use and other factors absent in the paleoenvironmental case (Sitch *et al.*, 2008; Cheaib *et al.*, 2012; see WGI AR5 6.4). Thus, the paleoecological record and models that have been tested against it provide a coherent message that biomes will alter their functioning and composition in response to changing and often novel future climates: they will move as species mixtures change (Section 4.3.2.5 has more specific information on projected migration rates), novel plant communities will emerge and significant carbon stock changes take place (Williams and Jackson, 2007; MacDonald, 2010; Prentice *et al.*, 2011; Willis and MacDonald, 2011). The paleoecological record and models provide *high confidence* 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). This record provides a critical opportunity for model evaluation 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 paleoecological record also highlights the importance of including the direct effects of changing atmospheric CO_2 levels in efforts to simulate future ecosystem functioning and plant species competition (Prentice *et al.*, 2011; Woillez *et al.*, 2011; Bond and Midgley, 2012; Claussen *et al.*, 2013).

The paleoeclimatic record also reveals that past radiative climate forcing change was slower than that anticipated for the 21st century (see WGI AR5 Chapters 5, 8 and 12), but even these slower changes often drove surprisingly abrupt, non-linear, regional-scale change in terrestrial and inland water systems (e.g., Harrison and Goni, 2010; Williams *et al.*, 2011), as did even slower climate change during the most recent Holocene interglacial (e.g., Booth *et al.*, 2005; Kropelin *et al.*, 2008; Williams *et al.*, 2010a; 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 the global-scale 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). Thus there is *high confidence* that these models may not capture some aspects of future abrupt climate change and associated ecosystem impacts (Leadley *et al.*, 2010).

4.2.4. Multiple Stressors Interacting with Climate Change

The climatic and non-climatic drivers of ecosystem change need to be distinguished if the joint and separate attribution of changes to their causes is to be performed (see Chapter 18). In this section we elaborate on factors affecting ecosystems, operating simultaneously with climate change. These factors share underlining drivers with one another and with climate change to varying degrees, together they form a syndrome known as "global change". The individual effects of climate change, habitat loss and fragmentation, chemical pollution, overharvesting and invasive alien species are increasingly well documented (Millennium Ecosystem Assessment, 2005c; Settele *et al.*, 2010a) but much less is known about their combined consequences. Ecosystem changes may occur in cascades, where a change in one factor precipitates increased vulnerability with respect to other factors (Wookey *et al.*, 2009) or propagates through the ecosystem as a result of species interactions (Gilman *et al.*, 2010). 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. For instance, Larsen *et al.* (2011) demonstrated that non-additive interactions among the climate factors in a multifactor experiment were frequent and most often antagonistic, leading to smaller effects than predicted from the sum of single factor effects. Leuzinger *et al.* (2011) and Dieleman *et al.* (2012) have synthesized multifactor experiments and demonstrated that in general, the effect size is reduced when more factors are involved, but Leuzinger *et al.* (2011) suggest that multifactor models tend to show the opposite tendency.

4.2.4.1. Land-Use and Cover-Change (LUCC)

LUCC is both a cause (WGI AR5 6.1.2) and consequence of climate change. It is the major driver of current ecosystem and biodiversity change (Millennium Ecosystem Assessment, 2005b) and a key cause of changes in freshwater systems (Section 4.3.3.3). In tropical and subtropical areas of Asia, Africa, Oceania and South America, the dominant contemporary changes are conversion of forests and woodlands to annual and perennial agriculture, grazing pastures, industrial logging and commercial plantations; followed by conversion of savannas, grasslands and pastures to annual agriculture (Hosonuma *et al.*, 2012; Macedo *et al*. 2012). In Europe there is net conversion of agricultural lands to forest (Rounsevell and Reay, 2009; Miyake *et al.*, 2012). Conversion of peatlands to agriculture has been an important source of carbon to the atmosphere in Southeast Asia (Limpens *et al.*, 2008; Hooijer *et al.*, 2010; see Section 4.3.3.3).

Contemporary drivers of LUCC include rising demand for food, fibre and bioenergy and changes in lifestyle and technologies (Hosonuma *et al.*, 2012; Macedo *et al*. 2012). By mid-century climate change is projected to become a major driver of land cover change (Leadley *et al.*, 2010). Non-climate environmental changes such as nitrogen deposition, air pollution and altered disturbance regimes are also implicated in LUCC. Some of the underlying drivers of LUCC are also direct or indirect drivers of climate change (Cui and Graf, 2009; McAlpine *et al.*, 2009; Mishra *et al.*, 2010; Schwaiger and Bird, 2010; van der Molen *et al.*, 2011; Groisman *et al.*, 2012); this cause-and-effect entanglement of climate change and LUCC can confound the detection of climate change and make attribution to one or the other difficult. Local-to-regional climate change was at least partly attributed to LUCC in 14 of 26 studies reviewed for this chapter, 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; Groisman *et al.*, 2012. No climate effects attributed: Suarez *et al.*, 1999; Saurral *et al.*, 2008; Wang *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 (and land use itself) contributes to changes in the climate through altering the greenhouse gas concentrations in the atmosphere, surface and cloud albedos, surface energy balance, wind profiles and evapotranspiration, among other mechanisms. The phrase “biophysical effects” is shorthand for the effect vegetation has on the climate other than through its role as a source or sink of greenhouse gases. These effects are now well documented, significant and are increasingly included in models of global and regional climate change. The greenhouse gas and biophysical effects of vegetation can be opposite in sign (de Noblet-Ducoudre *et al.*, 2012) 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 may result in an increase in seasonally-averaged albedo (local and global cooling, Davin *et al.*, 2007) and a decrease in transpiration (local, but not global warming). Findell *et al.* (2007) concluded on the basis of model studies that the non-GHG climate impacts of LUCC were generally minor, but nevertheless significant in some regions. Brovkin *et al.* (2013), projecting the overall effect of LUCC on climate change for the 21st century, found LUCC to be small driver globally, but locally important. Most global climate models suggest local average cooling effects following forest conversion to croplands and pastures (Pitman *et al.*, 2009; Longobardi *et al.*, 2012). Satellite observations suggest that the effect of conversion of the Brazilian savannas (*cerrado*) to pasture was to induce a local warming which was partly reversed when the pasture was subsequently converted to sugarcane (Loarie *et al.*, 2011). Several modelling studies suggest that the global surface air temperature response to deforestation depends on the latitude at which deforestation occurs. High latitude deforestation results in global cooling, low latitude deforestation causes global warming and that the mid latitude response is mixed (Bathiany *et al.*, 2010; Davin and de Noblet-Ducoudre, 2010; van der Molen *et al.*, 2011; Longobardi *et al.*, 2012), with some exceptions documented for boreal forests (Spracklen *et al.*, 2008). Boreal and tropical forests influence the climate for different reasons: boreal forests have low albedo (i.e., reflect less solar radiation, especially in relation to a snowy background; Levis, 2010; Mishra *et al.*, 2010; Longobardi *et al.*, 2012) 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). The implications of these findings for afforestation as a climate mitigation action are discussed in Section 4.3.4.5. Forests may also influence regional precipitation through biophysical effects (Butt *et al.*, 2011; Pielke *et al.*, 2011; see Section 4.3.3). In summary, changes in land cover have biophysical effects on the climate, sometimes opposite in direction to greenhouse gas mediated effects, which can materially alter the net outcome of the land cover change on the global climate (*high certainty*).

In summary, changes in land cover have biophysical effects on the climate, sometimes opposite in direction to greenhouse gas mediated effects, which can materially alter the net outcome of the land cover change on the global climate (*high confidence*).

_____ START BOX 4-1 HERE _____

Box 4-1. Future Land Use Changes

Assessment of climate change effects on terrestrial and inland freshwater ecosystems requires the simultaneous consideration of LUCC. The world is undergoing important shifts in land use, driven by accelerating demand for food, feed, fibre, and fuel. The main underlying driver is the rate at which *per capita* consumption is growing, particularly in emerging economies (Tilman *et al.*, 2011). Policy shifts in developed countries favouring biofuel production have also contributed (Searchinger *et al.*, 2008; Lapola *et al.*, 2010; Miyake *et al.*, 2012). Agricultural commodity prices have risen and may stay high through 2020 (OECD/FAO, 2010), due to: a) demand growth outpacing supply growth, exacerbated by climate-related crop failure (Lobell *et al.*, 2011); b) decline in the rate of improvement in agricultural productivity (Ray *et al.*, 2012); c) shortage of arable land not already under cultivation, especially in the temperate zone; d) 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 (Lambin and Meyfroidt, 2011); and e) declining area under cultivation in temperate zones, mainly in developed countries. The shortage of arable land in temperate systems could put pressure on marginal or sensitive landscapes, mainly in Latin America's *cerrados* and grasslands (Brazil, Argentina) and in African savannahs (Sudan, Democratic Republic of the Congo, Mozambique, Tanzania, Madagascar) (Lambin and Meyfroidt, 2011).

Deforestation in developing countries is correlated with the export of agricultural commodities (DeFries *et al.*, 2010). Future LUCC remains uncertain, since it depends on economic trends and policies themselves dependent upon complex political and social processes, including climate policy. By 2012, the deforestation rate in the Brazilian Amazon had declined by 77% below its 1996-2005 average (INPE, 2013; Nepstad *et al*., 2009) 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.*, 2013).

[INSERT TABLE 4-2 HERE
Table 4-2: Summary of drivers and outcomes of LUCC scenarios associated with Representative Concentration Pathways (Hurtt *et al.*, 2011). RCPs are identified with the radiative forcing by 2100 (8.5, 6.0, 4.5 and 2.6 Wm^{-2}) and by the name of the model used to generate the associated land use/cover scenarios (MESSAGE, AIM, GCAM and IMAGE; see Hurtt *et al*. (2011) for further details).]

Each of the four main RCPs used for future climate projections has a spatially-explicit future land use scenario consistent with both the emissions scenario and the underlying associated socio-economic scenario simulated by integrated assessment models, as well as conditions in 2005 (Hurtt *et al.*, 2011; Table 4-2; Figure 4-2; Figure 4-3). In scenarios where cropland and pasture are projected to decrease, they are replaced with secondary vegetation. Tropical and boreal forest regions are both projected to undergo declining primary forest cover in most RCPs, but in RCP6.0 total forest area remains approximately constant and in RCP4.5 total forest area expands due to increased secondary forest. The extent to which primary vegetation is replaced by secondary vegetation, crops or pasture varies between the RCPs (Figure 4-3), with no simple linear relationship between the extent of vegetation change and the level of total radiative forcing. Larger reductions in primary vegetation cover are projected in RCP8.5, due to a general absence of pro-active measures to control land cover change in that scenario. Large reductions are also projected in RCP2.6 due to widespread conversion of land to biofuel crops (Figure 4-2). Smaller reductions are foreseen in RCP6.0 and RCP4.5, with the latter involving conservation of primary forest and afforestation as mitigation measures.

[INSERT FIGURE 4-3 HERE
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 *et al.*, 2010; Klein Goldewijk *et al.*, 2011), and from scenarios associated with the RCPs from 2005 to 2100 (Hurtt *et al.*, 2011).]

_____ END BOX 4-1 HERE _____

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; Canfield *et al.*, 2010). Activities such as fertilizer production and fossil fuel burning currently transform 210 TgN/year of nitrogen gas in the atmosphere into reactive forms of nitrogen (N_r) that can be readily used by plants and microorganisms in land and in the ocean, slightly more than the non-anthropogenic transformation of 203 TgN/year (Fowler *et al.*, 2013). Most of the transformations of anthropogenic N_r are on land (Fowler *et al.*, 2013). The human-caused flow from land to oceans in rivers is 40-70 TgN/year, additional to the estimated natural flux of 30 TgN/year (Galloway *et al.*, 2008; Fowler *et al.*, 2013). 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. N deposition, CO_2 concentrations and temperatures are therefore increasing together at global scales (Steffen *et al.*, 2011). 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 the tropics (Galloway *et al.*, 2008) due to increased needs for food and energy for growing populations in emerging economies (e.g., Zhu *et al.*, 2005).

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 (Power *et al.*, 2006; Clark and Tilman, 2008; 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).

The impacts of N deposition are often first manifested in freshwater ecosystems, since they collect and concentrate the excess N (and phosphorus, P) 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 (Bergström and Jansson, 2006), although reduced N loading including deposition was observed between 1988 and 2003 in Sweden (Weyhenmeyer *et al.*, 2007). The observed symptoms include a shift from nitrogen limitation of phytoplankton in lakes to phosphorus limitation (Elser *et al.*, 2009).

Since the AR4 report, an increasing number of studies have models, observations and experiments to understand and predict the interactive effects of N deposition, climate change and CO_2 on ecosystem function. Interactions between nitrogen and other global change factors are widespread, strong and complex (Rustad, 2008; Thompson *et al.*, 2008; Langley and Megonigal, 2010; Gaudnik *et al.*, 2011; Eisenhauer *et al.*, 2012; Hoover *et al.*, 2012; but see Zavaleta *et al.*, 2003 for evidence of additive effects). In a study of plant-pollinator relationships, the combination of N deposition, CO_2 enrichment and warming resulted in larger negative impacts on pollinator populations than could be predicted from the individual effects (Hoover *et al.*, 2012). In a perennial grassland species, N limitation constrained the response to rising CO_2 (Reich *et al.*, 2006). Broadly, the overall body of research shows that ecosystem function is mediated by complex interactions between these factors, such that many ecosystem responses remain difficult to understand and predict (Churkina *et al.*, 2010; Norby and Zak, 2011).

In forests in many parts of the world, experiments, observations and models suggest that the observed increase in productivity and carbon storage is due to combinations of N deposition, climate change, fertilization effects of rising CO_2, and forest management (Huang *et al.*, 2007; Magnani *et al.*, 2007; Pan *et al.*, 2009; Churkina *et al.*, 2010; Bellassen *et al.*, 2011; Bontemps *et al.*, 2011; de Vries and Posch, 2011; Eastaugh *et al.*, 2011; Norby and Zak, 2011; Shanin *et al.*, 2011; Lu *et al.*, 2012). N deposition and rising CO_2 appear to have generally dominated in much of the Northern Hemisphere. However, the direct effects of rising temperature and changes in precipitation may exceed N and CO_2 as key drivers of ecosystem primary productivity in a few decades time. In grasslands, however, experiments show that plant productivity is increased more by N addition (within the projected range for this century) than by elevated CO_2, 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 (Menge and Field, 2007; Ma *et al.*, 2011), in part due to shifts in plant species composition (Langley and Megonigal, 2010) and the complex dynamics of coupled C, N and P cycles (Menge and Field, 2007; Niboyet *et al.*, 2011).

Analyses using the multi-factor biodiversity change model GLOBIO3 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).

4.2.4.3. Tropospheric Ozone

The concentration of ozone in the troposphere (the part of the atmosphere adjacent to the Earth's surface) has risen over the past 150 years from a global average of 20-30 ppb to 30-50 ppb, with high spatial and temporal variability (Horowitz, 2006; Oltmans *et al.*, 2006; Cooper *et al.*, 2010; WGI AR5 Figure 2.7). This is due to a) increasing anthropogenic emissions of gases which react in the atmosphere to form ozone (Denman *et al.*, 2007) and b) 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 (VOCs) and oxides of nitrogen (NO_x). Intercontinental transport of these precursors contributes to rising global background ozone concentrations, including in regions where local ozone-precursor emissions are decreasing (Dentener *et al.*, 2010). Global sources of VOC are predominantly biogenic (BVOC), and especially forests (Hoyle *et al.*, 2011).

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 focused on Northern-Hemisphere temperate and boreal species - concluded that current levels of tropospheric ozone suppress growth by 7% relative to pre-industrial levels. Modelling studies which extrapolate experimentally-measured dose-response relationships suggest a 14 to 23% contemporary reduction in Gross Primary Productivity (GPP) worldwide, with higher values in some regions (Sitch *et al.*, 2007) and 1-16% reduction of Net Primary Productivity (NPP) 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 in some cases ozone exposure increases stomatal conductance (Wilkinson and Davies, 2010). For the species studied, carbon assimilation rates and leaf area are generally reduced, while respiration increases and leaf senescence accelerated - all leading to a reduction in NPP.

Conifers are less sensitive than broad-leafed species. In a modelling study, lower stomatal conductance due to ozone exposure increased river runoff by reducing the loss of soil moisture through transpiration, but observational studies that measured runoff in relation to ozone exposure show divergent trends on this issue (McLaughlin *et al.*, 2007; Wittig *et al.*, 2007; Mills *et al.*, 2009; Huntingford *et al.*, 2011).

A modelling study (Sitch *et al.*, 2007) suggests that the negative effects of rising O_3 on plant productivity could offset 17 to 31% of the projected increase in global carbon storage due to increasing CO_2 concentrations over the 21^{st} century, 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 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 plant sensitivity to O_3 (Ainsworth *et al.*, 2012). Other than changing cultivars or species, few management actions promoting adaptation to higher levels of O_3 are currently available (Wilkinson and Davies, 2010; Teixiera *et al.*, 2011). Research into developing ozone resistant varieties and chemical protectants against damage may provide management options in the future (Wilkinson and Davies, 2010; Ainsworth *et al.*, 2012).

4.2.4.4. Rising CO_2

Rising atmospheric CO_2 concentrations directly affect ecosystems and through biological and chemical processes. The consequences for the global carbon cycle are discussed in WGI AR5 Box 6.3; the discussion here focusses on impacts on terrestrial and inland water systems. Paleo records over the Late Quaternary (past million years) show that changes in the atmospheric CO_2 content between 180 and 280 ppmv had ecosystem-scale effects worldwide (Prentice and Harrison, 2009).

In contrast to the oceans, changes in CO_2 concentrations in inland waters are primarily influenced by biological processes, such as inputs of terrestrial organic matter (particularly DOC) and bacterial respiration (van de Waal *et al.*, 2010; Aufdenkampe *et al.*, 2011). Carbon can, however, become limiting during intense algal blooms, especially in the surface waters of stratified lakes and reservoirs, and rising atmospheric CO_2 concentrations may stimulate higher algal production under these conditions (van de Waal *et al.*, 2010). Higher CO_2 concentrations can lead to increases in the C:N and C:P ratios of phytoplankton, though the trophic consequences of this are difficult to predict because zooplankton may alter their feeding behaviour to select higher quality forms of algae or increase feeding rate (Urabe *et al.*, 2003; van de Waal *et al.*, 2010).

Over the past two decades, and especially since AR4, experimental investigation of elevated CO_2 effects on plants and ecosystems has mainly used Free Air CO_2 Enrichment (FACE) techniques (Leakey *et al.*, 2009). FACE is 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. Plants with a C3 photosynthetic system, which includes most species but excludes warm-region grasses, show an increase in photosynthesis under elevated CO_2, the precise magnitude of which varies between species. Acclimation ("down-regulation") occurs under long-term exposure, leading to cessation of effects in some (Norby and Zak, 2011) but not all studies (Leakey *et al.*, 2009). The C4 photosynthetic system found in most tropical grasses and some important crops is not directly affected by elevated CO_2, but C4 plant productivity generally increases under elevated CO_2 due to increased water use efficiency (WUE). Transpiration is decreased under elevated CO_2 in many species, due to reduced opening of stomatal apertures, leading to greater WUE (Leakey *et al.*, 2009; Leuzinger and Körner, 2010; De Kauwe *et al.*, 2013). Increasing WUE is corroborated by studies of stable carbon isotopes (Barbosa *et al.*, 2010; Koehler *et al.*, 2010; Silva *et al.*, 2010; Maseyk *et al.*, 2011). The WUE increase does not acclimate to higher CO_2 in the medium term, *i.e.* over several years (Leakey *et al.*, 2009). Satellite observations from 1982-2010 show an 11% increase in green foliage cover in warm, arid environments (where WUE is most important) after correcting for the effects of precipitation variability (Donohue *et al.*, 2013); gas exchange theory predicts 5 to 10% greening resulting from rising CO_2 over this period.

The interactive effects of elevated CO_2 and other global changes (such as climate change, nitrogen deposition and biodiversity loss) on ecosystem function are extremely complex. Generally, nitrogen use efficiency is increased under higher CO_2 (Leakey *et al.*, 2009), although in some tree FACE experiments, productivity increases as a result of enhanced CO_2 were sustained by increased nitrogen uptake rather than increased nitrogen use efficiency (Finzi *et al.*, 2007). In one ten-year temperate grassland experiment in Minnesota, elevated CO_2 halved the loss of species richness expected from nitrogen addition (Reich, 2009), whereas no such benefit was reported for an alpine grassland in France (Bloor *et al.*, 2010) or a Danish heathland ecosystem (Kongstad *et al.*, 2012).

Elevated CO_2 can affect plant response to other stresses, such as high temperature (Lloyd and Farquhar, 2008) and drought. Ozone exposure decreases with lower stomatal conductance (Sitch *et al.*, 2007). In savannas, faster growth rates under higher CO_2 can allow woody plants to grow tall enough between successive fires to escape the flames (Bond and Midgley, 2001; Scheiter and Higgins, 2009). Differential species responses to elevated CO_2 appear to be 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; Mohan *et al.*, 2006).

Experimental studies have shown that elevated CO_2 leads to increased leaf C:N ratios in woody plants, forbs and C3 grasses (but not C4 grasses), which may decrease their quality as food and increase herbivorous insect feeding rates and changes to their density and community structure (Sardans *et al.*, 2012). Plants may also become more toxic to herbivores under elevated CO_2 levels, through increased concentrations of C-based and N-based defences (Lindroth, 2010; Cavagnaro *et al.*, 2011).

Our understanding of ecosystem responses to elevated CO_2 is incomplete in some respects. The majority of FACE experiments apply upper CO_2 concentrations of approximately 550 ppmv, which is below the concentrations projected by 2100 under higher emissions scenarios. The physiology of photosynthesis suggests that direct CO_2 effects saturate at levels of approximately 700 ppmv (Long *et al.*, 2004). Most elevated CO_2 experiments impose a sudden increase of CO_2 concentration as opposed to the gradual rise experienced in reality. Most large-scale FACE experiments have been carried in temperate locations (e.g., Hickler *et al.*, 2008); there are currently no large-scale tropical or boreal FACE experiments. The magnitude of CO_2 effects decreases as the spatial scale of study increases (Leuzinger *et al.*, 2011).The scale of controlled experiments is limited to approximately 100 m^2. Extrapolation to larger scales ignores large-scale atmospheric feedbacks (Körner *et al.*, 2007) and catchment-scale hydrological effects (see Cross-Chapter Box CC-VW). Overall, there is *medium confidence* (*much evidence, medium agreement*) that increases in CO_2 up to about 600 ppm will continue to enhance photosynthesis and plant water-use efficiency, but at a diminishing rate.

CO_2 effects are a first-order influence on model projections of ecosystem and hydrological responses to anthropogenic climate change (Sitch *et al.*, 2008; Lapola *et al.*, 2009; Friend *et al.*, in press).The direct effect of CO_2 on plant physiology, independent of its role as a greenhouse gas, means that assessing climate change impacts on ecosystems and hydrology solely in terms of global mean temperature rise, (or equivalently, expressing greenhouse gas effects solely in terms of radiative forcing) is an oversimplification (Huntingford *et al.*, 2011; Betts *et al.*, 2012). A 2°C rise in global mean temperature, for example, may have a different net impact on ecosystems depending on the change in CO_2 concentration accompanying the rise (e.g., Good *et al.*, 2011a). A high climate sensitivity and/or a higher proportion of non-CO_2 GHGs would imply a relatively low CO_2 rise at 2°C global warming, so the offsetting effects of CO_2 fertilization and increased water use efficiency would be smaller than for low climate sensitivity and/or a lower proportion of non-CO_2 GHGs.

4.2.4.5. Diffuse and 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 the proportions of direct versus diffuse radiation. In some regions, direct radiation has been reduced by up to 30 W m^{-2} over the industrial era, 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 because a smaller fraction of the vegetation canopy is light-saturated, making photosynthesis more light efficient at the canopy scale (Knohl and Baldocchi, 2008; Kanniah *et al.*, 2012). In a global model which included this effect, 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; however, 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), All RCPs project decreased aerosol concentrations due to air quality protection measures, as already seen in some countries. The influence of the 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 (see WGI AR5 7.7); but this topic 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). Species invasions have increased over the last several decades (*very high confidence*), and 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; Bellard *et al.*, 2013). Although there is increasing evidence that some species invasions have been assisted by climate change, there is *low confidence* that species invasions have in general been assisted by recent climatic trends because of the overwhelming importance of human facilitated dispersal in mediating invasions. The spread of alien species has several causes, including habitats made favourable 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). Some degree of climate/habitat match has been found to be a pre-requisite of establishment success across seven major plant and animal groups (Hayes and Barry, 2008). A range of 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, may have traits that favour their survival, reproduction and adaptation under changing climates; invasive plants in particular tend to have faster growth rates and are particularly favoured when resources are not limited (*medium to high confidence*, van Kleunen *et al.*, 2010; Willis *et al.*, 2010b; Buswell *et al.*, 2011; Davidson *et al.*, 2011; Zerebecki and Sorte, 2011; Haider *et al.*, 2012; Matzek, 2012). Some invasive plants are more drought tolerant (Crous *et al.*, 2012; Matzek, 2012; Perry *et al.*, 2012), and on average they have higher overall metabolic rates, foliar nitrogen concentrations and photosynthetic rates than their native counterparts (Leishman *et al.*, 2007).

Extreme climate events provide opportunities for invasion by generating disturbances and redistributing available resources (Diez *et al.*, 2012) and changing connectivity between different ecosystems. Current warming has already enabled many invasive 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). However, population declines and range contractions are 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 Section 4.3.2.5). For example, invasive grasses may be favoured over native ones with increasing temperatures (Parker-Allie *et al.*, 2009; Chuine *et al.*, 2012; Sandel and Dangremond, 2012).

In a few cases, benefits to biodiversity and society may result from the interactive effects of climate change and invasive species, 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 CO_2 levels will increase the growth rates of most invasive plant species (Mainka and Howard, 2010; but see Section 4.2.4.4).The effectiveness of invasive alien species management for sequestering carbon is uncertain and context specific (Peltzer *et al.*, 2010). Longer term, indirect effects of invasive alien species will be more important than direct, short-term effects; for instance as a result of changes in soil carbon stocks and tree community composition (*low-medium confidence*, Peltzer *et al.*, 2010).

Synergistic interactions occur between climate change and invasive alien 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.*, 2011a; Cahill *et al.*, 2013). Climate change and invasive alien plant species generally increase the risk and intensity 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 production systems (*medium confidence*, Robinet and Roques, 2010; Clements and Ditommaso, 2011). Warming has contributed to the spread of many invasive insect species, such as the mountain pine bark beetle, and resulted in forest destruction (*high confidence*, Raffa *et al.*, 2008). The interactions between crop growth, climate change and pest dynamics are difficult to predict (West *et al.*, 2012). Management strategies may become less effective as a consequence of the decoupling of biocontrol relationships and less effective mechanical control as biomass and/or population size of invasive species increases (*low to -medium confidence*, Hellmann *et al.*, 2008).

4.3. Vulnerability of Terrestrial and Freshwater Ecosystems to Climate Change

The vulnerability of ecosystems to climate change, i.e. their propensity to be adversely affected, is determined by the sensitivity of ecosystem processes to the particular 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 or adapting to it. Tolerance and adaptability both interact with exposure, which in the case of terrestrial and freshwater ecosystems means the magnitude and rate of climate change relative to ranges of climatic conditions and rates of change under which the ecosystem developed and its organisms evolved. Chapter 19 provides a full discussion on vulnerability concepts.

4.3.1. Changes in the Disturbance Regime

The species composition at a given location is determined by three considerations: the ability of species to reach the location; the physiological tolerance of the species in relation to the range of conditions experienced there; and interactions with other species, including competitors, symbionts, predators, prey and pathogens. Occasional disturbances relieve competition, create opportunities for the establishment and success of less-dominant species; and may facilitate dispersal. Moderate disturbance is thus important in maintaining diversity and ecosystem function (Connell, 1978). Exposure to disturbances keeps tolerance of disturbance in the population high. Fire, floods and strong winds are all examples of biodiversity-sustaining climate disturbances, provided that their frequency and intensity does not deviate greatly above or below the regime to which the species are adapted. Average environmental conditions may be less of a determinant of species range and abundance than the extreme conditions, such as the occurrence of exceptionally cold or hot days or droughts exceeding a certain duration (Zimmermann *et al.*, 2009). The projected changes in probability of extremes are typically disproportionately larger than the projected changes in the mean (see IPCC, 2012; but also Diffenbaugh *et al.*, 2005). Biotic disturbances, such as pest and pathogen outbreaks are also often implicated in with ecosystem change, and may be enabled by climate change.

It is suggested that ecosystem regime shifts resulting from climate change (alone or in interaction with other factors) will often be triggered by changes in the disturbance regime, rather than by physiological tolerance for the mean conditions (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 of occurrence, intensity and other relevant attributes, such as their seasonal pattern. A corollary is that disturbance-related change is abrupt rather than gradual. Change in the fire disturbance regime is emerging as a key proximal mechanism and early indicator of terrestrial ecosystem change (Girardin *et al.*, 2009; Johnstone *et al.*, 2010). Changes in the fire regime have in some cases been attributed to climate change (Westerling *et al.*, 2006; Littell *et al.*, 2009; Turetsky *et al.*, 2011; Westerling *et al.*, 2011; Moritz *et al.*, 2012). Regional trends in fire occurrence have been observed since 2000 (Giglio *et al.*, 2013), but interpreting their significance requires a longer term perspective (e.g., Bergeron *et al.*, 2010).

4.3.2. Observed and Projected Change in Ecosystems

This section highlights key observed changes in terrestrial and freshwater ecosystems over the recent past, as well as changes projected during the 21st century. For observations, we assess the degree of confidence that change has been detected, and separately the confidence we have in attributing the change to climate change (Figure 4-4). Confidence in detection 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 Chapter 18; Mastrandrea *et al.*, 2010). Note that a slightly different definition of detection is used here than in Chapter 18, 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 tests are satisfied: changes correspond to a sound mechanistic understanding of responses to climate change; the time series of observations is sufficiently long to detect trends correlated with climate change; and confounding factors can be accounted for or are of limited importance. In the sections that provide the details of the assessment of detection and attribution, estimated levels of confidence are given 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. In all cases the estimates of confidence levels are based on global and cross-taxon assessments, so the positioning may be different for specific taxa or regions. Some of the sections include assessments of model-based projections of future change; the confidence assessment of detection and attribution does not extend to these.

A key message arising from the analysis of *detection* and *attribution* is that climate impacts on the functioning of organisms and ecosystems are clearest when temperature is a principal driver, changes are relatively rapid and confounding factors play a small role. At one end of the spectrum, the large warming signal over the last several decades in much of the Arctic tundra combined with minimal human impacts is associated with *high confidence* in detection of an increase in shrubs and permafrost thawing and *high confidence* in the attribution to climate warming (Section 4.3.3.1.1). Likewise, the phenology of most organisms is sensitive to temperature, confounding effects are often small and the response is rapid, leading to *high confidence* in detection and attribution of changes in phenology to warming (Section 4.3.2.1). At the opposite end of the spectrum, species extinctions are very difficult to attribute to climate change (Section 4.3.2.5), in part because other factors dominate recent extinctions. This does not mean that climate has not played an important contributing role; indeed it has been argued that the low level of confidence in attribution is due to the lack of studies looking for climate signals in extinctions (Cahill *et al.* 2013). Similarly there is very good evidence that species composition is changing in cultural landscapes, but the important role of other factors, e.g., land management, nitrogen deposition, makes attribution of a contribution to recent warming difficult. This analysis indicates that responses in most species and ecosystem levels will become more apparent over time because i) observed organism level changes will have long term impacts on ecosystem functioning (*high confidence,* Sections 4.3.2.1, 4.3.2.5, 4.3.3) and ii) warming signals can be detected in ecosystems where the recent warming has been strong and confounding factors are minimal. In addition, the absence of observed changes does not preclude confident projections of future change for three reasons: climate change projected for the 21st century substantially exceeds the changes experienced over the past century in medium to high scenarios (all but RCP 2.6); ecosystem responses to climate change may be non-linear; and change may only be apparent after considerable time lags (Jones *et al.*, 2009).

[INSERT FIGURE 4-4 HERE
Figure 4-4: Confidence in detection of change and attribution of observed responses of terrestrial ecosystems to climate change. Confidence levels are based on expert judgment of the available literature following the IPCC uncertainty guidance (Mastrandrea *et al.*, 2010), attribution criteria outlined in Chapter 18 and detection criteria defined in the text. The symbols in the figure represent global and cross-taxon assessments; the positioning may be different for specific taxa or regions. The following sections provide the details of the assessments that were used in positioning each of the points Phenology, Section 4.3.2.1; Primary Productivity, Section 4.3.2.2; Biomass and C stocks, Section 4.3.2.3; Evapotranspiration, Section 4.3.2.4; Species distributions, Section 4.3.2.5**;** Global species extinctions, Section 4.3.2.5**;** Invasive and alien species**,** Section 4.2.4.6**;** Tree mortality, Section 4.3.3.1, Box 4-2; Boreal forest regime shift, Section 4.3.3.1.1, Box 4-4; Amazon forest regime shift, Section 4.3.3.1.3, Box 4-3; Tundra regime shift, Section 4.3.3.4, Box 4-4; Woody encroachment, Section 4.3.3.2.2; Cultural landscapes, Section 4.3.3.5.3; Evolutionary and genetic adaptation, Section 4.4.1.2.]

4.3.2.1. Phenology

Further evidence from ground-based and satellite studies, mainly focused on the Northern Hemisphere, supports the AR4 conclusion that shifts in phenology have occurred over recent decades. “Spring advancement” - earlier occurrence of spring events, such as breeding, bud burst, breaking hibernation, flowering, migration - is seen in hundreds of plant and animal species in many regions (Menzel *et al.*, 2006; Cleland *et al.*, 2007; Parmesan, 2007; Primack *et al.*, 2009; Cook *et al.*, 2012a; Peñuelas *et al.*, 2013) although magnitudes of change vary considerably and some species show no change (Parmesan, 2007). Apparent discrepancies between two estimates of overall Northern-Hemisphere spring advancement noted in AR4 (2.3 days per decade, Parmesan and Yohe, 2003; 5.1 days per decade, Root *et al.*, 2003) are largely resolved when methodological differences are accounted for, particularly the inclusion of species that do not show phenological changes (Parmesan, 2007). A combined analysis of 203 species suggests Northern Hemisphere spring advancement of 2.8±0.35 days per decade (Parmesan, 2007).

Plants – Spring advancement in is seen across the Northern Hemisphere including North America (e.g., Cook *et al.*, 2008; Cook *et al.*, 2012b), Europe (e.g., Menzel *et al.*, 2006; Cook *et al.*, 2012b), Asia (e.g., Primack *et al.*, 2009; Ma and Zhou, 2012) and the High Arctic (Høye *et al.*, 2007). Changes are generally larger at higher latitudes. A meta-analysis indicates mean spring Northern Hemisphere spring advancement of -1.1 ± 0.16 days per decade for herbs and grasses (85 species), -1.1 ± 0.68 days per decade for shrubs (6 species) and -3.3 ± 0.87 days per decade for trees (16 species), over a record period of 35-132 years, depending on the study. The warming trends detected in the well-mixed surface waters (epilimnion) of many lakes in North America, Eurasia and Africa (Adrian *et al.*, 2009) are associated with the earlier onset of spring phytoplankton blooms (Winder and Schindler, 2004; Winder and Sommer, 2012). Satellite data also indicate a general tendency of spring advancement, though there is variation between satellite studies, especially at local scales, due to the use of different instruments and methods (e.g., White *et al.*, 2009). A study using the Advanced Very High Resolution Radiometer (AVHRR) suggests that for vegetation between 30ºN and 80ºN, the start of the growing season advanced by -5.2 days between 1999 and 1982 and advanced a further -0.2 days by 2008; while the growing season end was delayed by 6.6 days between 1982 and 2008 (Jeong *et al.*, 2011). Studies with a more recent satellite instrument, Moderate Resolution Imaging Spectrometer (MODIS) also show spring advancement (e.g., Ahl *et al.*, 2006). The relatively short duration of satellite observations makes trend detection particularly sensitive to the choice of analysis period.

Animals – Many new studies provide further evidence of changes in animal phenology (e.g., amphibians: Kusano and Inoue, 2008; Phillimore *et al.*, 2010; birds: Pulido, 2007; Thorup *et al.*, 2007; mammals: Adamik and Kral, 2008; Lane *et al.*, 2012; insects: Robinet and Roques, 2010; freshwater plankton: Adrian *et al.*, 2009). Changes in breeding phenology are reported from various regions and different taxa (e.g., Parmesan, 2006; Parmesan, 2007; Post *et al.*, 2008; Primack *et al.*, 2009). In the Northern Hemisphere several studies show advancements of egg laying dates in birds (e.g., Parmesan, 2007: -3.7 ± 0.7 days/decade, in 41 species). In contrast, a delay of the mean breeding date by 2.8 to 3.7 days between 1950 and 2004 was seen for two of nine seabirds in the Eastern Antarctic, linked to decreased sea ice extent (Barbraud and Weimerskirch, 2006). Spring arrival dates have advanced for many migratory birds (e.g., Thorup *et al.*, 2007). 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).

A large body of evidence therefore shows that Northern Hemisphere temperate, boreal and Arctic regions, spring advancement has occurred in many plant and animal species over the last several decades (*high confidence* due to *robust evidence* but only *medium agreement* when examined across all species and regions, Figure 4-4).

Understanding of the drivers of phenological change has also improved further since AR4. Many observational studies find a correlation with higher temperatures (Cook et al 2012a). Experimental manipulation generally supports this (e.g., plants, Cleland *et al.*, 2012; bird egg-laying, Visser *et al.*, 2009; insects, Musolin *et al.*, 2010; Kollberg *et al.*, 2013). Some individual studies find good agreement between experimental warming and *in situ* observations (e.g., Gunderson *et al.*, 2012) although a meta-analysis suggests that experiments can substantially under-predict advances in the timing of flowering and leafing of plants in comparison with observational studies (Wolkovich *et al.*, 2012). Observational data can also be affected by methodological issues; for example, flipper-tagging of penguins can alter their migratory behaviour (Saraux *et al.*, 2011). Rates of warming across a season may also be important (Schaper *et al.*, 2012). Models can be used to explain relationships between observed phenological changes and environmental variables. For example, a model based on water temperature captured the observed temporal and spatial variation in *Daphnia* phenology in Northern Hemisphere lakes (Straile *et al.*, 2012). Other environmental factors related to temperature, such as timing of snowmelt, snow cover and snow depth, can play a role. Snowmelt changes led 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 (Hülber *et al.*, 2010; Lambert *et al.*, 2010). Earlier snowmelts decreased floral resources and hence affected insect population dynamics in mountain ranges in the USA in the years 1980, 1985, 1986 and 1989 (Boggs and Inouye, 2012). In Colorado, USA, the yellow-bellied marmot emerge earlier from hibernation due to snowmelts becoming earlier over 1976-2008 (Ozgul *et al.*, 2010) while in Alberta, Canada, Columbian ground squirrels emerged later over 1992-2012 due to delayed snowmelts associated with increased late-season snowstorms (Lane *et al.*, 2012). Delayed emergence from hibernation was associated with decreased population growth rate (Lane *et al.*, 2012). Food availability can be important; for example, in the Yukon area, Canada, the date of giving birth in North American squirrels (*Tamiascurus hudsonicus*) advanced by an average of -18 days over the period 1989-1998, coinciding with increasing abundance of white spruce cones, their major food source (Réale *et al.*, 2003).

Phenological response can differ with migration strategy in birds, for example short-distance migrants show greater advancements in spring arrivals than long distant migrants (e.g., Saino *et al.*, 2009; but see Parmesan, 2006 for different patterns). In a temperate region (Massachusetts, USA), declining sizes of populations and migrating cohorts of North American Passerine birds account for a large part of the variation in migration times between 1970 and 2002 (Miller-Rushing *et al.*, 2008). The remaining variation was explained by climatic variables, migration distance and date. The variation in bird migration phenology change can also be related to differing patterns of feather changes during moulting times, food availability at stop-over places and differing health conditions of individual species (Gordo, 2007).

Although a number of non-climatic influences on phenology are also identified, an increased number of observational and experimental studies, across many organism types, suggest that warming has contributed to the overall spring advancement observed in the Northern Hemisphere (*high confidence* due to *high agreement* and *medium evidence*).

4.3.2.2. Primary Productivity

Primary production, the process of plant growth, is fundamental to the global carbon cycle (see Section 4.3.2.3. below) and t underpins provisioning ecosystem services such as food, timber and grazing. 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 sensitive to most aspects of climate change, is indirectly affected via the effects of climate on pests and diseases, and is responsive to many of the other changes simultaneously taking place in the world, such as the described in Section 4.2.4. The diverse and frequently non-linear form of responses to the factors influencing primary production, combined with the complexity of interactions between them, means that at a given location the net outcome can be an increase, no change or a decrease in productivity.

The concentration of CO_2 in the atmosphere shows clear patterns in space and time largely related to the primary productivity of the land and oceans. The contribution by terrestrial ecosystems to these patterns can be estimated using isotope measurements, emission databases and models (Canadell *et al.*, 2007). It consists of a sink term, due to increased net ecosystem production, plus a source term due to land-use change. During the decade 2000 to 2009, land net primary productivity at the global scale continued to be enhanced about 5% relative to the estimated pre-industrial level, leading to a land sink of 2.6 ± 1.2 PgC/y (these values are from WGI AR5 6.3.2.6; the uncertainty range is two standard deviations; for the primary literature see also Raupach *et al.*, 2008; Le Quéré *et al.*, 2009). The net uptake of carbon by the land is highly variable year-to-year, mainly in response to climate variation and major volcanic eruptions (Peylin *et al.*, 2005; Sitch *et al.*, 2008; Mercado *et al.*, 2009). Given the uncertainty range, it is not possible to conclude whether the rate of carbon uptake by the residual land sink has increased or decreased over the past two decades (Raupach *et al.*, 2008; WGI AR5 6.3.2.6). CMIP5 model projections, using the RCP scenarios, suggest that the rate of net carbon uptake by terrestrial ecosystem will decrease during the 21st century except under the RCP4.0 scenario, and by the greatest amount under RCP8.5. There is greater uncertainty between models than between scenarios; in some models terrestrial ecosystems become a net source of CO_2 to the atmosphere (WGI AR5 Section 6.4.3.2, especially Figure 6.26).

It is possible to downscale the land sink estimate continentally, using inversion modelling techniques and the growing network of precision atmospheric observations. There is *high agreement and medium evidence* that the net land uptake in natural and semi-natural terrestrial ecosystems is broadly distributed around the world, 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).

The observed trends in NDVI, a satellite proxy for primary productivity, are discussed under various ecosystem-specific discussions above and 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, or simply reflect inadequacies in the indicator, method of analysis and length of the record in relation to the high inter-annual variability. 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 subsequent observations show a lower rate and no geographical uniformity (Goetz *et al.*, 2007). More than 25% of high latitude 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 (Goetz *et al.*, 2007). NDVI trend analyses in rangelands show varying patterns around the world, with substantial disagreement between studies (Millennium Ecosystem Assessment, 2005a; 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 herbs; and to what degree it reveals land management efforts or responses to climate (Anyamba and Tucker, 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. Many tree ring records show accelerated tree growth during much of the 20th century (Briffa *et al.*, 2008), which often correlates with rising temperature. Variations in tree ring width, density and isotopic composition arise from many factors, including temperature, moisture stress, CO_2 fertilization, N deposition and ozone damage, but also stand structure and management. Direct CO_2 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; but 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 the past two decades are based on a relatively small body of evidence and may therefore be biased. Recent tree ring studies were often specifically designed to examine growth in response to environmental changes (Gedalof and Berg, 2010) and may therefore not be representative of global tree growth. Direct repeated measurements of tree girth increment in forest monitoring plots (discussed in Section 4.3.2.3) are an alternate data source for recent decades.

Primary production in freshwater lakes has been observed to increase in some arctic (Michelutti *et al.*, 2005) and boreal lakes, but decrease in Lake Tanganyika in the tropics (O'Reilly *et al.*, 2003). In both cases the changes were attributed by the authors to climate change.

In summary, there is *high confidence* that net terrestrial ecosystem productivity at the global scale has increased relative to the pre-industrial era (Figure 4-4). There is *low confidence* in attribution of these trends to climate change. Most studies speculate that rising CO_2 concentrations are contributing to this trend through stimulation of photosynthesis, but there is no clear, consistent signal of a climate change contribution (Figure 4-4).

4.3.2.3. Biomass and Carbon Stocks

The forest biomass carbon stock can be estimated from the routine forest monitoring that takes place for management and research purposes. Forest inventories were generally 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 increased investment in forest monitoring through initiatives such as the Reduced Emissions from Deforestation and Degradation (REDD) of the UNFCCC.

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 during 1998-2008, because growth was offset by significant losses 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 (Phillips *et al.*, 2009) forest biomass has generally increased in recent decades, dropping temporarily after a drought in 2005. Globally, for the period 2000-2007, recently-undisturbed forests are estimated to have withdrawn 2.30 ± 0.49 PgC/y from the atmosphere, while formerly-cleared tropical forests, now regrowing, withdrew an additional 1.72 ± 0.54 PgC/y (Pan *et al.*, 2011). The global terrestrial carbon sink is partly offset by the losses of forest carbon stocks to the atmosphere through land use change, largely in the tropics, of 1.1 ± 0.8 PgC/y (2000-2009, WGI AR5 6.3.2.6).

The carbon stock in global soils, including litter and peatlands is 1500-2400 PgC, with permanently frozen soils adding another 1700 PgC (Davidson and Janssens, 2006). The soil carbon stock is thus more than ten times greater than the carbon stock in forest biomass (Kindermann *et al.*, 2008). Changes in the size of the soil carbon stock result from changes in the net balance of inputs and losses over a period of many years. Inputs derive from primary production, discussed in Section 4.3.2.2, and are mostly modestly increasing under climate change. Losses result principally through the respiration of soil microbes, which increases with increasing temperature. The present and future temperature sensitivity of microbial respiration remains uncertain (Davidson and Janssens, 2006). An analysis of long-term respiration measurements from the soil around the world suggests that it 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). If soil respiration were to exceed terrestrial net primary production globally and on a sustained basis, the present net terrestrial sink would become a net source, accelerating the rate of CO_2 build up in the atmosphere (Luo, 2007).

The carbon stock in freshwater systems is also quite high in global terms. Annual rates of storage (0.03 – 0.07 Pg C/yr) may be trivial compared with sequestration by soils and terrestrial vegetation, but lake sediments are preserved over longer time scales (+10,000 years compared with decades to centuries), and Holocene storage of C in lake sediments has been estimated at 820 Pg (Cole *et al.*, 2007). Manmade impoundments represent an increasing and short-lived additional carbon store with conservative annual estimates of 0.16 – 0.2 Pg C/yr (Cole *et al.*, 2007).

A short-duration study of the temperature sensitivity of decomposition in flooded coastal soils, extrapolated to the 21st century, suggested that increases in respiration would exceed increases in production in future (Kirwan and Blum, 2011). Further detail on wetland soil carbon stocks can be found in Section 4.3.3.3 on peatlands; and on permafrost carbon stocks in Box 4-4 and in Chapter 28.

In summary, biomass and soil carbon stocks in terrestrial ecosystems are currently increasing (*high confidence*) but are vulnerable to loss to the atmosphere as a result of rising temperature, drought and fire projected in the 21st century (Figure 4-4). 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*).

4.3.2.4. Evapotranspiration and Water Use Efficiency

Evapotranspiration (ET) includes evaporation from the ground and vegetation surfaces, and transpiration through plant stomata. Both are affected by multiple factors (Luo *et al.*, 2008) including temperature, solar (shortwave) and thermal (longwave) radiation, humidity, soil moisture and terrestrial water storage; transpiration is additionally affected by CO_2 concentration through its influence on plant stomatal conductance. Studies using lysimeters, evaporation pans, the balance of observed precipitation and runoff, and model reconstructions, indicate both increases and decreases in ET in different regions and between approximately 1950 and the present (Huntington, 2008; Teuling *et al.*, 2009; Douville *et al.*, 2013). Flux tower records have at most 15 years duration (FLUXNET, 2012) so there are insufficient data to calculate large-scale, long-term trends. ET can also be estimated from meteorological observations or simulated with models constrained by observations. Estimates of ET from 1120 globally (but non-uniformly) distributed stations indicate that global land mean ET increased by approximately 2.2% between 1982 and 2002, a rate of increase of 0.75 mm yr^{-2} (Wang *et al.*, 2010b). Other studies, using data-constrained models indicated global ET rises of between 0.25 - 1.1 mm yr^{-2} during the 1980s and 1990s (Jung *et al.*, 2010; Vinukollu *et al.*, 2011; Zeng *et al.*, 2012), possibly linked with increased surface solar radiation and thermal radiation (Wild *et al.*, 2008) or warming (Jung *et al.*, 2010). There has been no significant ET trend since approximately 2000 (Jung *et al.*, 2010; Vinukollu *et al.*, 2011; Zeng *et al.*, 2012), possibly due to soil moisture limitation (Jung *et al.*, 2010). Overall, there is *low confidence* in both detection and attribution of long-term trends in ET (Figure 4-4).

Experiments show that rising CO_2 decreases transpiration and increases intrinsic water use efficiency (iWUE, the ratio of photosynthesis to stomatal conductance, Leakey *et al.*, 2009). Some modelling studies suggest that over the last century, the effects of CO_2 on decreasing transpiration are of comparable size but opposite to the effects of rising temperature (Gerten *et al.*, 2008; Peng *et al.*, 2013). However the observed general increase in ET argues that reduced transpiration cannot be the dominant factor (Huntington, 2008). A meta-analysis of studies at 47 sites across 5 ecosystem types (Peñuelas *et al.*, 2011) suggests that iWUE for mature trees increased by 20.5% between the 1970s and 2000s. Increased iWUE since pre-industrial times (1850 or before) has also been found at several forest sites (Andreu-Hayles *et al.*, 2011; Gagen *et al.*, 2011; Loader *et al.*, 2011; Nock *et al.*, 2011) and also in a temperate semi-natural grassland since 1857 (Koehler *et al.*, 2010), although in one boreal tree species iWUE ceased to increase after 1970 (Gagen *et al.*, 2011).

4.3.2.5. Changes in Species Range, Abundance and Extinction

Species respond to climate change through genotypic adaptation and phenotypic plasticity; by moving out of unfavourable and into favourable climates; or by going locally or globally extinct (Dawson *et al.*, 2011; Bellard *et* *al.*, 2012; Peñuelas *et al.*, 2013; Section 4.2.3). These responses to climate change can potentially have large impacts on biodiversity and ecosystem services. Genotypic adaptation in the face of strong selection pressure from climate change is typically accompanied by large reductions in abundance (see Section 4.4.1.2). Species range shifts are accompanied by changes in abundance, local extinctions and colonization that can alter ecosystem services when they affect dominant species such as trees, keystone species such as pollinators, or species that are vectors for disease (Zarnetske *et al.*, 2012). Global extinctions result in the permanent loss of unique forms of life.

Substantial evidence has accumulated since AR4 reinforcing the conclusion that the geographical ranges of many terrestrial and freshwater plant and animal species have moved over the last several decades in response to warming and that this movement is projected to accelerate over the coming decades under high rates of climate change. Some changes in species abundances appear to be linked to climate change in a predictable manner, with species abundances increasing in areas where climate has become more favourable and vice versa. In contrast, uncertainties concerning attribution to climate change of recent global species extinctions, and in projections of future extinctions, have become more apparent since the AR4 report.

Observed species range shifts - The number of studies looking at observed range shifts and the breadth of species examined has greatly increased since AR4. 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 a useful simplification of species responses; 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 (i.e., broad taxonomic categories of species) and species within these 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 trends except in mountainous areas (Bertrand *et al.*, 2011; Doxford and Freckleton, 2012; Gottfried *et al.*, 2012; Zhu *et al.*, 2012; Telwala *et al.*, 2013). There is growing evidence that responses at the “trailing edge” of species distributions (i.e., local extinction in areas where climate has become unfavourable) are often less pronounced than responses at the “leading edge” (i.e., colonization of areas where climate has become favourable), 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 with confidence (Thomas *et al.*, 2006).

Rising water temperatures are also implicated in species range shifts in river fish communities (e.g., Comte and Grenouillet, 2013), combined with a decrease in recruitment and survival as well as range-contraction of cold-water species such as salmonids (Bartholow, 2005; Bryant, 2009; Ficke *et al.*, 2007; Jonsson and Jonsson, 2009; Hague *et al.*, 2011). Shifts in freshwater fish species range towards higher elevation and upstream (Hickling et al., 2006; Comte and Grenouillet, 2013) also are not keeping pace with the rate of warming in streams and rivers. 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.

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). Taking this spatial variation into account 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 only in the regions where the range shifts were observed (Chen *et al.*, 2011), providing greater confidence in attribution of the range shifts to climate change. Average range shifts across taxa and regions in this study were ca. 17 km poleward and 11 m up in altitude per decade; velocities which are 2 to 3 times greater than previous estimates (compare with Parmesan and Yohe, 2003; Fischlin *et al.*, 2007), but these responses differ greatly among species groups. However, this approach remains a simplification, since the climate drivers of species range changes, e.g., temperature and precipitation, have frequently shifted in different 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 AR4 show that species range changes result from interactions among climate drivers and between climate and non-climate factors. It is the greater understanding of these interactions, combined with increased geographical scope that leads to *high confidence* that the ranges of several well-studied species groups, such as insects and birds, have shifted their ranges significant distances (10s of km or more) over the last several decades, and that these range shifts can be attributed to changes in climate. 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 detection and attribution when examined across all species and all regions.

Future range shifts - Projections of climate change impacts on future species range shifts since the AR4 report have been dominated by studies using ecological niche models (ENMs) that project future ranges based on correlative models of current relationships between environmental factors and species distribution (Peterson *et al.*, 2011). A variety of process-based models are starting to be more widely used to make projections of future species distributions (Buckley *et al.*, 2010; Beale and Lennon, 2012; Cheaib *et al.*, 2012; Higgins *et al.*, 2012; Foden *et al.*, 2013). Model comparisons show that correlative models generally predict larger range shifts than process-based models for trees (Morin and Thuiller, 2009; Kearney *et al.*, 2010; Cheaib *et al.*, 2012). For other species groups that have been studied, differences in projections between model types show no clear tendency (Kearney *et al.*, 2009; Buckley *et al.*, 2010; Bateman *et al.*, 2012). There has been some progress in model validation: projected 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), but further validation is needed (Green *et al.*, 2008; Pearman *et al.*, 2008; Nogues-Bravo *et al.*, 2010; Dawson *et al.*, 2011). Modelling studies typically do not account for a number of key mechanisms mediating range shifts, such as genetic adaptation and phenotypic plasticity (see Section 4.4.1.2), species interactions or human-mediated effects. An important limitation in most studies is that realistic species displacement rates are not accounted for (i.e., rates at which species are able to shift their ranges through dispersal and establishment); as such, they only indicate changes in the location of favourable and unfavourable climates, from which potential shifts in species distribution can be inferred, but not rates of change (Bateman *et al.*, 2013).

Analyses and models developed since AR4 permit the estimation of the ability of a wide range of species to track climate change. Figure 4-5 provides a synthesis of the projected abilities of several species groups to track climate change. This analysis is 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; Feeley and Rehm, 2012; Dobrowski *et al.*, 2013); and ii) species displacement rates across landscapes for a 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 4-5 legend). Comparisons of these rates indicate whether species are projected to be able to track climate as it changes. When species displacement capacity exceeds climate velocity it is inferred that species will be able to keep pace with climate change; when displacement capacity is lower than projected climate velocities then they will not, within the bounds of uncertainty of both parameters. This simplified analysis is coherent with more sophisticated model analyses of climate induced species displacement across landscapes, some of which have evaluated additional constraints such as demographics, habitat fragmentation or competition (e.g., Meier *et al.*, 2012; Schloss *et al.*, 2012).

Rates of climate change over the 20th century and projected for the 21st century are shown in Figure 4-5A. Rates of climate change for global land surfaces are given for IPCC AR5 climate projections under a wide range of greenhouse gas emissions scenarios (i.e. WGI AR5 Chapter 12; Knutti and Sedláček, 2012). Rates of global warming for land surfaces have averaged ca. 0.03 °C/yr since 1980, but have slowed over the last decade and a half (WGI Chapter 2). 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, exceeding 0.06 °C/yr by the end of the century (RCP 8.5), and perhaps above 0.08 °C/yr at the upper bound for this scenario.

Climate velocity is defined as the rate of change in climate over time (e.g., °C/yr, if only temperature is considered) divided by the rate of change in climate over distance (e.g., °C/km, if only temperature is considered) and therefore depends on regional rates of climate change and the degree of altitudinal relief (Figure 4-5B, Loarie *et al.*, 2009; Dobrowski *et al.*, 2013). For example, climate velocity for temperature is low in mountainous areas because the change in temperature over short distances is large (e.g., Rocky mountains, Andes, Alps, Himalayas, Figure 4-5B, leftmost axis). Climate velocity for temperature is generally high in flat areas because the rate of change in temperature over distance is low (e.g., parts of the US Mid-west, Amazon basin, West Africa, central Australia, Figure 4-5B, rightmost axis). In flat areas, climate velocity can exceed 8 km/yr for the highest rates of projected climate change (RCP 8.5). We have focused on climate velocity for temperature change, but several analyses also account for precipitation change.

Rates of displacement vary greatly within and among species groups (Figure 4-5C). Some species groups, notably herbaceous plants and trees, generally have very low displacement capacity. Other species groups such as butterflies, birds (not shown) and large vertebrates generally have a very high capacity to disperse across landscapes, nonetheless some species in these groups have low dispersal capacity. 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 mountainous areas at medium to high rates of warming, though there is uncertainty concerning the potential role of long-distance dispersal (Pearson, 2006). Primates generally have substantially higher dispersal capacity than trees; however, a large fraction of primates are found in regions with very high climate velocities, in particular the Amazon basin, thereby putting them at high risk of being unable to track climates even at relatively low rates of climate change (Schloss *et al.*, 2012). On a global average, many rodents, as well as some carnivores and freshwater molluscs are projected to be unable to track climate at very 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 *Observed species range shifts* above).

Humans can increase species displacement rates by intentionally or unintentionally dispersing individuals or propagules. 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. In particular, habitat loss and fragmentation typically reduces displacement rates, sometimes substantially (Eycott *et al.*, 2012; Hodgson *et al.*, 2012; Meier *et al.*, 2012; Schloss *et al.*, 2012). The degree to which habitat fragmentation slows displacement depends on many factors, including the spatial pattern of the fragments and corridors, maximum dispersal distances, population dynamics and the suitability of intervening modified habitats as stepping-stones (Pearson and Dawson, 2003). Species and habitat dependencies may also speed or hinder species displacement. For example, host plants are projected to move much more slowly than most herbivorous insects, substantially slowing displacement of the insects if they are unable to switch host plants (Schweiger *et al.*, 2012). Likewise, many habitats are structured by slow moving plants, so habitat shifts are projected to lag behind climate change (Jones *et al.*, 2012; Hickler *et al.*, 2012) which will in turn mediate the movements of habitat specialists.

There are significant uncertainties in climate velocities, measured estimates of dispersal and establishment rates, and model formulations. Climate velocities are calculated using a variety of methods and spatial resolutions, making direct comparisons difficult and leading to *low confidence* in estimates of climate velocities in Figure 4-5B (*limited evidence* and *medium agreement*). The lowest estimates of global average climate velocity (Figure 4-5B, centre axis), are about half the best estimate values we show on the climate velocity axes (Loarie *et al.*, 2009), while the highest estimates are about four times higher (Burrows *et al.*, 2011), but high estimates may be artefacts of using very large spatial resolutions (Dobrowski *et al.*, 2013). In addition, the climate velocities used in Figure 4-5 are based on temperature alone, and recent analyses indicate 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 paleontological 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 due to intrinsic uncertainties in the methods and differences in the mechanisms included (Kinlan and Gaines, 2003; Stevens *et al.*, 2010b). 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, leading to an overestimation of displacement rates. For trees there is *medium confidence* in projections of their displacement rates due to the large number of studies of past, current and future displacement rates (*robust evidence* and *medium agreement*). Less is known for other broad species groups such as mammals, so there is only *low confidence* in estimates of their displacement capacity. Estimates for other groups, such as freshwater molluscs are based on very little data, so estimates of their dispersal capacity are poorly constrained.

Despite large uncertainties in displacement capacity and climate velocity, the rates of displacement required to track the highest rates of climate change (RCP8.5) are so high that many species will be unable to do so (*high confidence*). Moderate rates of projected climate change (RCP4.5 and RCP6.0) would allow more species to track climate, but would still exceed the capacity of many species to track climate (*medium confidence*). The lowest rates of projected climate change (RCP2.6) would allow most species to track climate towards the end of the century (*high confidence*). This analysis highlights the importance of rates of climate change as an important component of climate change impacts on species and ecosystems. For example, differences in the magnitude of climate change between scenarios are small at mid-century (WGI Chapter 12), but the differences in rates of climate change are large. At mid-century, it is projected that species would need to move little at the lowest rates of climate change (RCP 2.6), but will need to move approximately 70 km/decade in flat areas in order to track climate at the highest rates of climate change (RCP 8.5).

Species that cannot move fast enough to keep pace with the rate of climate change will lose favourable climate space and experience large range contractions (Warren *et al.*, 2013), whereas displacement that keeps pace with climate change greatly increases the fraction of species that can maintain or increase their range size (Menéndez *et al.*, 2008; Pateman *et al.*, 2012). Mountains provide an extremely important climate refuge for many species because the rate of displacement required to track climate is low (Figure 4-5B, Colwell *et al.*, 2008; Engler *et al.*, 2011; Gottfried *et al.*, 2012; Pauli *et al.*, 2012; but see Dullinger *et al.*, 2012). However, species that already occur near mountaintops (or other boundaries) are among the most threatened by climate change because they cannot move upwards (Ponniah and Hughes, 2004; Thuiller *et al.*, 2005; Raxworthy *et al.*, 2008; Engler *et al.*, 2011; Sauer *et al.*, 2011). The consequences of losing favourable climate space are not yet well understood. The extent to which adaptive responses might allow persistence in areas of unfavourable climates is discussed in Section 4.4.1.2. In the absence of adaptation, losing favourable climate space is projected to lead to reduced fitness, declining abundance and local extinction, with potentially large effects on biodiversity and ecosystem services (see evidence of early signs of this for trees in Box 4-2).

[INSERT FIGURE 4-5 HERE
Figure 4-5: (A) Rates 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. Horizontal and vertical pink bands illustrate the interpretation of this figure. Climate velocities for a given range of rates of climate change are determined by tracing a band from the range of rates in panel A to the points of intersection with the three climate velocity scalars in panel B. Comparisons with species displacement rates are made by tracing vertical bands from the points of intersection on the climate velocity scalars down to the species displacement rates in panel C. Species groups with displacement rates below the band are projected to be unable to track climate in the absence of human intervention. (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 CMIP5 climate model ensembles for the historical period (grey shading indicates model uncertainty) and for the future based on the four RCP emissions scenarios. Data were smoothed using a 20-year sliding window, and rates are means of between 17 and 30 models using one member per model. Global average temperatures at the end of the 21st century for the four RCP scenarios are from WGI AR5 Chapter 12. (B) Estimates of climate velocity for temperature were synthesized from historical and

projected future relationships between rates of temperature change and climate velocity (historical: Burrows *et al.*, 2011; Chen *et al.*, 2011; Dobrowski *et al.*, 2013; projected future: Loarie *et al.*, 2009; Sandel *et al.*, 2011; Feeley and Rehm, 2012). The three scalars are climate velocities that are representative of mountainous areas (left), averaged across global land areas (centre), and large flat regions (right). (C) Rates of displacement are given with an estimate of the median (black bars) and range (boxes = ca. 95% of observations or models for herbaceous plants, trees and plant-feeding insects or median ± 1.5 inter-quartile range for mammals). Displacement rates for herbaceous plants were derived from paleobotanical records, modern plant invasion rates and genetic analyses (Kinlan and Gaines, 2003). 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 modelled 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). Displacement rates for mammals were based on modelled dispersal rates of a wide range of mammal species (mean of Schloss *et al.*, 2012 for Western Hemisphere mammals and rates calculated from global assessments of dispersal distance by Santini *et al.*, 2013 and generation length by Pacifici *et al.*, 2013). Displacement rates for phytophagous insects are based on observed dispersal distances and genetic analyses (Peterson and Denno, 1998; Kinlan and Gaines, 2003; Schneider, 2003; Berg *et al.*, 2010; Chen *et al.*, 2011). The estimate of median displacement rate for this group exceeds the highest rates on the axis. These displacement rates do not take into account limitations imposed by host plants. Displacement estimates for freshwater molluscs correspond to the range of passive plus active dispersal rates for upstream movement (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* for several species groups is reviewed here (Section 4.3.2.5), in AR4 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 within existing range limits, have also occurred 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; Vadadi-Fulop *et al.*, 2012; Cahill *et al.*, 2013; Ruiz-Labourdette *et al.*, 2013). Confident attribution to recent global warming is hindered by confounding factors such as disease, land use change and invasive species (Cahill *et al.*, 2013). New tentative conclusions since AR4 is that climate related changes in abundance and local extinctions appear to be more strongly related to species interactions than to physiological tolerance limits (*low confidence*, Cahill *et al.*, 2013) and that precipitation can be a stronger driver of abundance change than temperature in many cases (Tian *et al.*, 2011; Tingley *et al.*, 2012). This gives weight to concerns that biological interactions, which are poorly known and modelled, 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; Hannah, 2012; Urban *et al.*, 2012; Vadadi-Fulop *et al.*, 2012).

A few examples illustrate the types of change in abundance 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 *(Aptenodytes forsteri)* population that has been attributed to regional changes in climate (Trathan *et al.*, 2011). The attribution of these declines to changes in regional climate is well supported, but the link to global warming is tenuous (Barbraud *et al.*, 2011).

Mountains also provide good examples of changes in abundance that can be linked to climate because very strong climate gradients are found there. AR4 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 mountaintops 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. These models project that local extinction risk by the end of the 21st century due to climate change will vary widely, ranging from almost no increase in local extinction risk within the current range for some species or species groups to greatly increased risk of local extinctions in more than 95% of the present-day range for others (Settele *et al.*, 2008; Bellard *et al.*, 2012). Projected local colonization rates are equally variable. There has been progress in coupling species distribution models 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.*, 2012a; Renwick *et al.*, 2012). These hybrid approaches predict extinction risk directly, rather than by inference from changes in climate suitability (Fordham *et al.*, 2012). The main conclusions from these studies are that changes in species abundance and local extinction risk as a result of climate change can range from highly positive to highly negative, and are 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; Clavero *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 (see also Section 4.4.1.2). At the intraspecific level, future climate change is projected to induce losses of genetic diversity when it results in range contraction (Balint *et al.*, 2011; Pauls *et al.*, 2013). In addition, there is theoretical and observational evidence this loss of genetic diversity will depend on rates of migration and range contraction (Arenas *et al.*, 2012). In these cases, reductions in genetic diversity may then decrease the ability of species to adapt to further climate change or other global changes. Climate change may also compound losses of genetic diversity that are 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 at rates that approach or exceed the upper limits of observed natural rates of extinction in the fossil record (Barnosky *et al.*, 2011). However, across all taxa there is only *low confidence* that rates of species extinctions have increased over the last several decades (Szabo *et al.*, 2012 - birds; but see Kiesecker, 2011 - amphibians). Most extinctions over the last several centuries have been attributed to habitat loss, overexploitation, pollution or invasive species, and these are the most important current drivers of extinctions (Millennium Ecosystem Assessment, 2005b; 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; see also Hoffmann *et al.*, 2011; Szabo *et al.*, 2012). Molluscs, especially freshwater molluscs, have by far the highest rate of documented extinctions of all species groups (Barnosky *et al.*, 2011). Mollusc 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 been attributed to changing climate, even tenuously (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 (*Bufo periglenes*) and Monteverde harlequin frog (*Atelopus varius*) of Central America, which belong to a group of amphibians with high rates of extinction previously 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; Anchukaitis and Evans, 2010; Bustamante *et al.*, 2010; Collins, 2010; Vredenburg *et al.*, 2010; Kiesecker, 2011; McKenzie and Peterson, 2012; McMenamin and Hannah, 2012). Due to *low agreement* among studies there is only *medium confidence* in detection of extinctions and attribution of Central American amphibian extinctions to climate change. 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 (Brook *et al.*, 2008; Pereira *et al.*, 2010; Hof *et al.*, 2011b). Overall, there is *very low confidence* that observed species extinctions can be attributed to recent climate warming, due to the very low fraction of global extinctions that have 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 AR4. AR4 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 AR4 broadly confirm this concern, leading to *high confidence* that climate change will contribute to increased extinction risk for terrestrial and freshwater species over the coming century (Pereira *et al.*, 2010; Sinervo *et al.*, 2010; Pearson, 2011; Warren *et al.*, 2011; Bellard *et al.*, 2012; Hannah, 2012; Ihlow *et al.*, 2012; Sekercioglu *et al.*, 2012; Wearn *et al.*, 2012; Foden *et al.*, 2013; Warren *et al.*, 2013). Most studies indicate that extinction risk rises rapidly with increasing levels of climate change, but some do not (Pereira *et al.*, 2010). The limited number of studies that have directly compared land use and climate change drivers have concluded that projected land use change will continue to be a more important driver of extinction risk throughout the 21st century (Pereira *et al.*, 2010). There is, however, broad agreement that land use, and habitat fragmentation in particular, will pose serious impediments to species adaptation to climate change as it is projected to reduce the capacity of many species to track climate (see above). These considerations lead to the assessment that future species extinctions are a high risk because the consequences of climate change are potentially severe, widespread and irreversible since extinctions constitute the permanent loss of unique life forms.

There is, however, *low agreement* concerning the overall fraction of species at risk, the taxa and places most at risk, and the time scale for climate-change driven extinctions to occur. Part of this uncertainty arises from differences in extinction risks within and between modelling studies: this uncertainty has been evaluated in AR4 and subsequent syntheses (Pereira *et al.*, 2010; Warren *et al.*, 2011; Bellard *et al.*, 2012; Cameron, 2012). All studies project increased extinction risk by the end of the 21st century due to climate change, but as indicated in AR4 the range of estimates is large. Recent syntheses indicate that model-based estimates of the fraction of species at substantially increased risk of extinction due to 21st century climate change 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; Foden *et al.*, 2013). Differences in modelling methods, species groups, and climate scenarios between studies make comparisons between estimates difficult (Pereira *et al.*, 2010; Warren *et al.*, 2011; Cameron, 2012).

Many papers published since AR4 argue that the uncertainty may be even higher than indicated in syntheses of model projections, due to limitations in the ability of current models to evaluate extinction risk (e.g., Kuussaari *et al.*, 2009; Pereira *et al.*, 2010; Dawson *et al.*, 2011; McMahon *et al.*, 2011; Pearson, 2011; Araujo and Peterson, 2012; Bellard *et al.*, 2012; Fordham *et al.*, 2012; Hannah, 2012; Kramer *et al.*, 2012; Zurell *et al.*, 2012; Halley *et al.*, 2013; Moritz and Agudo, 2013). Models frequently do not account for genetic and phenotypic adaptive capacity, dispersal capacity, population dynamics, the effects of habitat fragmentation and loss, community interactions, micro-refugia and the effects of rising CO_2 concentrations, all of which could play a major role in determining species vulnerability to climate change, causing models to either over- or under-estimate risk. In addition, difficulties in model validation, large variation in the climate sensitivity of species groups and uncertainties about timescales linking extinction risks to range reductions also lead to large uncertainty in model-based estimates of extinction risk.

A variety of studies since AR4 illustrate how accounting for these factors alters estimates of extinction risk. Accounting for biotic interactions such as pollination or predator-prey networks can increase modelled extinction risks, at least for certain areas and species groups (Schweiger *et al.*, 2008; Urban *et al.*, 2008; Hannah, 2012; Nakazawa and Doi, 2012), or can decrease extinction risk (Menéndez *et al.*, 2008; Pateman *et al.*, 2012).

Accounting for climatic variation at fine spatial scales 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; Shimazaki *et al.*, 2012) the persistence of small populations under future climate change. Several recent studies indicate that correlative species distribution models (the type of model most frequently used for evaluating species extinction risk) tend to be much more pessimistic concerning plant species range contractions and the inferred extinction risks due to climate change when compared to mechanistic models that explicitly account for the interactions between climate change and protective effects of rising CO_2 concentrations on plants (Morin and Thuiller, 2009; Kearney *et al.*, 2010; Cheaib *et al.*, 2012). Models that account for population dynamics indicate that some species populations, like those of polar bears (Hunter *et al.*, 2010), will decline precipitously over the course of the next century due to climate change, greatly increasing extinction risk, while others may not (Keith *et al.*, 2008). Phenotypic plasticity in one very well-studied temperate bird population has been estimated to be sufficient to keep extinction risk low even with projected warming exceeding 2-3°C (Vedder *et al.*, 2013), but this and other studies suggest that capacity for adaptation is often substantially lower in species with long generation times (see Section 4.4.1.2). There is evidence that interactions between physiological tolerances and regional climate change will lead to large taxonomic and spatial variation in extinction risk (Deutsch *et al.*, 2008; Sinervo *et al.*, 2010). Even species whose populations are not projected to decline rapidly over the next century can face a substantial "extinction debt"; i.e., will be in unfavourable climates that over a period of many decades to many centuries, leading to large projected reductions in population size (Dullinger *et al.*, 2012). Finally, 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 led to concern that forecasts of very high extinction rates due entirely to climate change may be overestimated (Botkin *et al.*, 2007; Dawson *et al.*, 2011; Hof *et al.*, 2011a; Willis and MacDonald, 2011; Moritz and Agudo, 2013). However, as indicated in Section 4.2.3, no past climate changes are precise analogues of future climate change in terms of speed, magnitude and spatial scale; nor did they occur alongside the habitat modification, over-exploitation, pollution, and invasive species that are characteristic of the twenty-first century. Therefore the paleontological record cannot easily be used to assess future extinction risk due to climate change.

4.3.3. Impacts on and Risks for Major Systems

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

Two geographical regions of particularly high risk have been identified in recent studies: i) tropics, due to the limited capacity of species to adapt to moderate global warming and ii) high northern latitude systems, because temperature increases are projected to be large. There has been a tendency to oppose these two points of view, but there is a high risk in *both* types of systems, albeit for different reasons (Corlett, 2011). Tropical species, which experienced low inter and intra-annual climate variability, have evolved within narrow thermal limits, and are 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 that have evolutionary histories of climatic variability (e.g., Arctic and boreal ecosystems; Beaumont *et al.*, 2011). However, there are physiological, evolutionary and ecological arguments that tropical species and ecosystem sensitivities to climate change are complex and may not be particularly high compared to other systems (Gonzalez *et al.*, 2010; Corlett, 2011; Laurance *et al.*, 2011; Gunderson and Leal, 2012; Walters *et al.*, 2012). High latitude systems have the greatest projected exposure to rising temperatures (WGI AR5 Chapter 12; Diffenbaugh and Giorgi, 2012), which all else being equal would put them at higher risk. 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 modelled biome level vulnerability indicated that temperate and high northern latitude systems are also the most vulnerable in the future (Gonzalez *et al.*, 2010).

Several potential tipping points (see Section 4.2.1) with regional and global consequences have been identified (Scheffer, 2009); two are elaborated in Boxes 4-3 (Amazon die-back) and 4-4 (Tundra-boreal regime shift).

An assessment by the authors of this chapter of the top risks in relation to climate change and terrestrial and freshwater ecosystems, is presented in Table 4-3.

[INSERT TABLE 4-3 HERE
Table 4-3: Key risks for terrestrial and freshwater ecosystems from climate change and the potential for reducing risk through mitigation and adaptation. Key risks are identified based on assessment of the literature and expert judgments by chapter authors, with evaluation of evidence and agreement in supporting chapter sections. Each key risk is characterized as very low to very high. Risk levels are presented in three timeframes: the present, near-term (here, assessed over 2030-2040), and longer-term (here, assessed over 2080-2100). For the near-term era of committed climate change, projected levels of global mean temperature increase do not diverge substantially across emission scenarios. For the longer-term era of climate options, risk levels are presented for global mean temperature increase of 2°C and 4°C above preindustrial levels. For each timeframe, risk levels are estimated for a continuation of current adaptation and for a hypothetical highly adapted state. Relevant climate variables are indicated by icons. For a given key risk, change in risk level through time and across magnitudes of climate change is illustrated, but because the assessment considers potential impacts on different physical, biological, and human systems, risk levels should not necessarily be used to evaluate relative risk across key risks, sectors, or regions.]

4.3.3.1. Forests and Woodlands

Forests and woodlands are principal providers of timber, pulp, bioenergy, water, food, medicines and recreation opportunities and can play prominent roles in cultural traditions. Forests are the habitat of a large fraction of the earth's terrestrial plant and animal species, with the highest concentrations and levels of endemism found in tropical regions (Gibson *et al.*, 2011). Climate change and forests interact strongly; air temperature, solar radiation, rainfall and atmospheric CO_2 concentrations are major drivers of forest productivity and forest dynamics, and forests help control climate through the large amounts of carbon they can remove from the atmosphere or release, through 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 with *high confidence* that forests are currently a net sink for carbon at the global scale. It is estimated that intact and regrowing forests currently contain 860±70 PgC and sequestered 4.0 ± 0.7 Pg C year^{-1} globally between 2000 and 2007 (WGI AR5 Chapter 6; Canadell *et al.*, 2007; Pan *et al.*, 2011; Le Quéré *et al.*, 2012). The carbon taken up by 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 mostly to tropical deforestation and forest degradation associated with logging and fire, resulting in a net C balance for global forests of 1.1±0.8 Pg C year^{-1}.

The future of the interaction between climate and forests is unclear. The carbon taken up by intact and regrowing forests appears to have stabilized compared to the 1990s, after having increased in the 1970s and 1980s (Canadell *et al.*, 2007; Pan *et al.*, 2011). There is *medium confidence* that the terrestrial carbon sink is weakening. The drivers behind the forest carbon sink vary greatly across regions. They include forest regrowth and stimulation of C sequestration by climate change, rising atmospheric CO_2 concentrations and N deposition (Pan *et al.*, 2011; Sections 4.2.4.1; 4.2.4.2; 4.2.4.4). 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; forest susceptibility to fire is projected to change little for the lowest emissions scenario (RCP 2.6), but substantially for the high emissions scenario (RCP 8.5, Figure 4-6). There is *low agreement* on whether climate change will cause fires to become more or less frequent in individual locations (Figure 4-6). Climate change-mediated disease and insect outbreaks could exacerbate climate-driven increases in fire susceptibility (Kurz *et al.*, 2008). The greatest risks for large positive feedbacks from forests to climate through changes in disturbance regimes 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.*, 2011b).

[INSERT FIGURE 4-6 HERE
Figure 4-6: Projected changes in meteorological fire danger, fire probability and fire frequency with different methods and climate models. (a)-(e) 30-year annual mean MacArthur Forest Fire Danger Index (FFDI) and change simulated with the HadGEM2-ES Earth System Model, with areas of no vegetation excluded (Betts *et al.*, 2013; (a) FFDI 1970-2000; (b) FFDI 2070-2100, RCP2.6; (c) change in FFDI by 2070-2100 relative to 1970-2000, RCP2.6;(d) FFDI 2070-2100; RCP8.5 (e) change in FFDI by 2070-2100 relative to 1970-2000, RCP8.5. (f) Change in fire frequency by 2100 relative to 2004, SRES B1, simulated using climate and land cover projections from the GISS GCM and IMAGE IAM (Pechony and Shindell, 2010). (g) Change in fire frequency by 2051-2100 relative to 1951-2000, SRES A1B, simulated with the MC1 vegetation model driven by 3 GCMs (CSIRO-Mk3.0, HadCM3, MIROC 3.2medres; mean over 3 simulations; Gonzalez et al., 2010). (h) Agreement on changes in fire probability simulated with a statistical model using climate projections from 16 CMIP3 GCMs, SRES A2 (i) Change in fire frequency by 2100 relative to 2004, SRES A2, simulated using climate and land cover projections from the GISS GCM (AR4 version) and IMAGE IAM (Pechony and Shindell, 2010). Changes in FFDI (a)-(e) and fire probability (h) arise entirely from changes in meteorological quantities, whereas changes in fire frequency (f) (g) (i) depend on both meteorological quantities and vegetation.]

Recent evidence suggests (*low confidence*) that the stimulatory effects of global warming and rising CO_2 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 (Nepstad *et al.*, 2009; INPE, 2013). Growing pressure for new crop (Section 4.4.4) and grazing land will continue to drive tropical deforestation (*medium confidence*) although recent policy experiments and market-based interventions in land use demonstrate the potential to reduce deforestation (Meyfroidt and Lambin, 2011; Westley *et al.*, 2011; Nepstad *et al.*, 2013).

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Box 4-2. Tree Mortality and Climate Change

Extensive tree mortality and widespread forest dieback (high mortality rates at a regional scale) linked to drought and temperature stress have been documented recently on all vegetated continents (Allen *et al.*, 2010a; Figure 4-7). However, appropriate field data sets are currently lacking for many regions (Anderegg *et al.*, 2013a), leading to *low confidence* in our 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 in multiple tropical forests (Kraft *et al.*, 2010; Phillips *et al.*, 2010) and Europe (Carnicer *et al.*, 2011). Episodes of widespread die-back (high mortality rates at a regional 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.*, 2013a). Some widespread die-back events have occurred concomitant with infestation outbreaks (Hogg *et al.*, 2008; Michaelian *et al.*, 2011; Raffa *et al.*, 2008), where insect 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.

Forest dieback has influenced the species composition, structure and age demographics, and successional trajectories in affected 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 tree 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 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 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 forests indicates that full recovery times may be longer than drought return intervals, leading to "compounding" effects of multiple droughts (Mueller *et al.*, 2005; Anderegg *et al.*, 2013b; Saatchi *et al.*, 2013). Projected increases in temperature are also expected to facilitate expansion of insect pest outbreaks poleward and in altitude which may also cause or contribute to tree mortality (Bentz *et al.*, 2010).

[INSERT FIGURE 4-7 HERE
Figure 4-7: Locations of substantial drought- and heat-induced tree mortality around the globe since 1970 (global forest cover and other wooded regions based on FAO, 2005). Studies compiled through 2009 (red dots) are summarized and listed in Allen *et al.* (2010a). Localities and measurement networks not included in Allen *et al.* (2010a), which are largely from post-2009 publications, have been added to this map (white dots and shapes). New locality references by region: Africa – Mehl *et al.*, 2010, van der Linde *et al.*, 2011; Fauset *et al.*, 2012; Gonzalez *et al.*, 2012; Kherchouche *et al.*, 2012; Asia – Dulamsuren *et al.*, 2009; Kharuk *et al.*, 2013; Liu *et al.*, 2013; Zhou *et al.*, 2013; Australasia – Brouwers *et al.*, 2012; Fensham *et al.*, 2012; Keith *et al.*, 2012; Matusick *et al.*, 2012; Brouwers *et al.*, 2013; Matusick *et al.*, 2013; Europe – Innes, 1992; Peterken and Mountford, 1996; Linares *et al.*, 2009; Galiano *et al.*, 2010; Vennetier and Ripert, 2010; Aakala *et al.*, 2011; Carnicer *et al.*, 2011; Linares *et al.*, 2011; Sarris *et al.*, 2011; Marini *et al.*, 2012; Cailleret *et al.*, 2013; Vilà-Cabrera *et al.*, 2013; North America – Fahey, 1998; Minnich, 2007; Klos *et al.*, 2009; Ganey and Vojta, 2011; Michaelian *et al.*, 2011; Peng *et al.*, 2011; DeRose and Long, 2012; Fellows and Goulden, 2012; Kaiser *et al.*, 2012; Millar *et al.*, 2012; Garrity *et al.*, 2013; Kukowski *et al.*, 2013; Williams *et al.*, 2013; Worrall *et al.*, 2013; South America – Enquist and Enquist, 2011; Lewis *et al.*, 2011; Saatchi *et al.*, 2013.]

_____ END BOX 4-2 HERE _____

4.3.3.1.1. 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; Pearson *et al.*, 2013). Projections of climate-driven changes in boreal forests over the next few centuries remain uncertain 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 (Bernhardt *et al.*, 2011; Mann *et al.*, 2012; Scheffer *et al.*, 2012; see WGI AR5 Chapter 12). Recent observed change (Box 4-2) and dynamic vegetation modelling (e.g., Sitch *et al.*, 2008) suggest that regions of the boreal forest could experience widespread forest dieback, although there is *low confidence* due to conflicting results (Sitch *et al.*, 2008; Gonzalez *et al*, 2010) and poor understanding of relevant mechanisms (WG1 AR5 Section 12.5.5.6.2). If such shifts were to occur, they would put the boreal carbon sink at risk (Pan *et al.*, 2011; Mann *et al.*, 2012).

Whereas boreal forest productivity has been expected 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 (*medium confidence*), more recent and longer-term assessments indicate with *high confidence* that many areas of boreal forest have instead experienced productivity declines (*high confidence*, Goetz *et al.*, 2007; Parent and Verbyla, 2010; Beck *et al.*, 2011b; de Jong *et al.*, 2011). The best evidence to date indicates that these "browning trends" are due to warming-induced drought, specifically the greater drying power of air (vapour pressure deficit, Williams *et al.*, 2013), inducing 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 (moist) conditions may be generating the expected warming-induced positive growth response (Rupp *et al.*, 2001; McGuire *et al.*, 2007; Goldblum and Rigg, 2010; Beck *et al.*, 2011b). The complexity of boreal forest response also involves tree age and size, with younger trees and stands perhaps being more able to benefit from warming where other factors are not limiting (Girardin *et al.*, 2011; Girardin *et al.*, 2012).

Where they occur, warming and drying, coupled with productivity declines, insect disturbance and associated tree mortality, also favour greater fire disturbance (*high confidence*). The boreal biome fire regime has intensified regionally 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; Girardin *et al.*, 2013a). The latter is particularly important because more severe burning consumes soil organic matter to greater depth, often to mineral soil, providing conditions that favour recruitment of deciduous species that in some regions of 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 in 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 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 modelling studies that are relatively poorly constrained due to sparse *in situ* observations.

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 (*high confidence*). 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. 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 parts of North America (Soja *et al.*, 2007; Ali *et al.*, 2012; Mann *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. In summary, an increase in tree mortality is observed in many boreal forests, with the clearest indicators of this in North America. However, tree health in boreal forests varies greatly among regions, which coupled with insufficient temporal coverage means that there is *low confidence* in the detection and attribution of a clear temporal trend in tree morality at the global scale (Figure 4-4).

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 could provide additional warming (*high confidence*; Schuur *et al.*, 2008; 2009; Tarnocai *et al.*, 2009; Romanovsky *et al.*, 2010; Schaefer *et al.*, 2011; see WGI AR5 Chapters 6 and 12; 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 (melting of ice-rich soil) associated with different substrates and associated topographic relief is 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 the reduction of insulation provided by vegetation cover and soil organic layers due to increased fire (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. 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.

The world's temperate forests act as an important carbon sink (*high confidence* due to *robust evidence and 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 (North 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 20th 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 (Ciais *et al.*, 2005; Witte *et al.*, 2011; Kasson and Livingston, 2012). 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 21st century climate projections, temperate forests are variously projected to substantially increase in total (biomass plus soil) carbon storage, especially through gains in forest cover; 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; Peng *et al.*, 2009; Ni, 2011). 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 (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 to higher 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 faster than natural migration (e.g., Chmura *et al.*, 2011; see also Section 4.3.2.5). Therefore, assisted migration has been suggested as an adaptation measure (see Section 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 (Morin and Thuiller, 2009; Kearney *et al.*, 2010; Kramer *et al.*, 2010; Cheaib *et al.*, 2012). Tree growth models project increased tree growth at the poleward and high altitudinal range limits over most of the 21st 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 al.*, 2008; Allen *et al.*, 2010b; Garreta *et al.*, 2010). Model projections are qualitatively coherent with observations that temperate forest species are moving up in altitude, probably due to climate warming at the end of the 20th century (Lenoir *et al.*, 2008). There is *medium confidence (medium evidence, medium agreement)* that temperate tree species are migrating poleward and to higher altitudes.

4.3.3.1.3. 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 (Malhi *et al.*, 2008; Davidson *et al.*, 2012).

A key uncertainty in our understanding of future impacts of climate change on tropical forests is the strength of direct CO_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_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; Powell *et al.*, 2013) generally use formulations for CO_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 calibrated on the basis of more recent data (Jones *et al.*, 2011).

A second important source of uncertainty is the rate of future CO_2 rise and climate change (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 (WGI AR5 Annex 1: Atlas of Global and Regional Climate Projections), although there is *medium confidence* (*medium evidence, medium agreement*) that some tropical regions, such as the eastern Amazon Basin, will experience lower precipitation and more severe drought (Malhi *et al.*, 2009a; Shiogama *et al*. 2011). 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_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 studies (Malhi *et al.*, 2009a; 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 than was suggested in AR4. Since AR4, there is new evidence of more frequent severe drought episodes in the Amazon region that are associated with sea surface temperature increases in the tropical North Atlantic (*medium confidence,* Marengo et al. 2010). There is *low confidence,* however, that these droughts or the observed sea surface temperatures can be attributed to climate change.

Networks of long-term forest plots reveal that 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 is consistent with increasing WUE associated with the rising concentration of CO_2, but also with changes in solar radiation and the ratio of diffuse to direct radiation (Lewis *et al.*, 2009a; Mercado *et al.*, 2009; Brando *et al.*, 2010; Section 4.2.4.5). 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_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 these ecosystems, and to the importance of extreme drought events (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 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 *high 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 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; Curran *et al.*, 2004; Ray *et al.*, 2005; Box 4-3). Land use also 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 is *medium confidence* (*medium evidence, high agreement*) that tree mortality in the Amazon region is increasing through severe drought and increased forest fire occurrence and *low confidence* that this can be attributed to warming (Figure 4-4).

Dry tropical forests are defined by strong seasonality in rainfall distribution (Mooney *et al.*, 1995) and have been reduced to an estimated one million km^2 globally through human activities (Miles *et al.*, 2006). Half of the world's remaining dry tropical forests are located in South America. Using five climate model simulations for the 2040-2069 period under the IS92a "business as usual scenario", Miles et al. (2006) found that approximately one third of the remaining area of tropical dry forests in the Americas will be exposed to higher temperatures and lower rainfall through climate change. Climate change, deforestation, fragmentation, fire, or human pressure place virtually all (97%) of the remaining tropical dry forests at risk of replacement or degradation (Miles *et al.*, 2006). In a regional study a dynamic vegetation model (IBIS) under A2 and B2 scenarios projected by a global climate model (HADRM3) found that most of the dry forests of India would be outside of their climate envelopes later this century (Chaturvedi *et al.*, 2011). There is *low confidence* in our understanding of climate change effects on dry forests globally.

_____ 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-driven, self-reinforcing transition of Amazon forests to a dry stable state (known as the Amazon "forest dieback") has improved. Modelling studies indicate that the likelihood of a climate-driven forest dieback by 2100 is lower than previously thought (Malhi *et al.*, 2009b; Cox *et al.*, 2013; Good *et al.*, 2013; Huntingford *et al.*, 2013), although lower rainfall and more severe drought is expected in the eastern Amazon (Malhi *et al.*, 2009a). There is now *medium confidence* (*medium evidence, medium agreement*) that climate change alone (that is, through changes in the climate envelope, without invoking fire and land use) will not drive large-scale forest loss by 2100 although shifts to drier forest types are predicted in the eastern Amazon (Mahli et al. 2009a). Meteorological fire danger is projected to increase (Golding and Betts, 2008; Betts *et al.*, 2013; Figure 4-6). 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 during the next few decades (Figure 4-8). There is now *medium confidence* (*medium evidence, high agreement*) that severe drought episodes, land use, and fire interact synergistically to drive the transition of mature Amazon forests to low-biomass, low-statured fire-adapted 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. 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) (*medium confidence*: *medium evidence, medium agreement*). Apart from these landscape processes, climate change could increase the incidence of severe drought episodes (Mahli et al. 2009b; Shiogama *et al.*, 2011).

If recent patterns of deforestation (through 2005), logging, severe drought, and forest fire continue into the future, more than half of the region's forests will be cleared, logged, burned or exposed to drought 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*) (Figure 4-8). 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 on a large scale (Soares-Filho *et al.*, 2012), if deforestation declines, or if cleared lands are reforested (Nepstad *et al.*, 2008). The 77% decline in deforestation in the Brazilian Amazon with 80% of the region's forest still standing (INPE, 2013) demonstrates that policy-led avoidance of a fire-mediated tipping point is plausible.

[INSERT FIGURE 4-8 HERE
Figure 4-8: The forests of the Amazon Basin are being altered through severe droughts, land use (deforestation, logging), and increased frequencies of forest fire. Some of these processes are self-reinforcing through positive feedbacks, and create the potential for a large-scale tipping point. For example, forest fire kills trees, increasing the likelihood of subsequent burning. This effect is magnified when tree death allows forests to be invaded by flammable grasses. Deforestation provides ignition sources to flammable forests, contributing to this dieback. Climate change contributes to this tipping point by increasing drought severity, reducing rainfall and raising air temperatures, particularly in the eastern Amazon Basin (*medium confidence; medium evidence, medium agreement).*]

_____ END BOX 4-3 HERE _____

4.3.3.2. Dryland Ecosystems: Savannas, Shrublands, Grasslands, and Deserts

The following sections treat a wide range of terrestrial ecosystems covering a large part of the land surface, whose common features are that they typically exhibit strong water stress for several months each year and grass-like plants and herbs are a major part of their vegetation cover. Thus the principle land use often involves grazing by domestic livestock or wild herbivores.

4.3.3.2.1. Savannas

Savannas are mixtures of coexisting trees and grasses, covering about a quarter of the global land surface, including tropical and temperate forms. Savannas are characterized by annual to decadal fires (Archibald *et al.*, 2009) of relatively low intensity, which are an important factor in maintaining the tree-grass proportions (Beerling and Osborne, 2006), but also constitute a major and climate-sensitive global source of fire-related emissions from land to atmosphere (Schultz *et al.*, 2008; van der Werf *et al.*, 2010). The geographical distribution of 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. 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 changes in the fire regime. In northern Australia, forest is expanding into former savanna areas (Brook and Bowman, 2006; Bowman *et al.*, 2011; Tng *et al.*, 2012). It has been projected that drying and greater seasonality, acting in conjunction with increased fire, could lead to former forested areas becoming savannas in parts of the Amazon basin (Malhi *et al.*, 2009b; Box 4-3) In many places around the world the savanna boundary is moving into former grasslands on elevation gradients, in other words into areas inferred to be formerly too cool for trees (Breshears, 2006).

The proportion of trees and grasses in savannas is considered unstable under some conditions (DeMichele *et al.*, 2011; Staver *et al.*, 2011; Wake, 2012). The differential effects of climate change, rising CO_2, fire and herbivory on trees and grasses have the potential to alter the cover of trees savannas, possibly abruptly. 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*, Moleele *et al.*, 2002; Angassa and Oba, 2008; Cabral *et al.*, 2009; 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 in form of 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 Auken, 2009), rising CO_2 (Bond and Midgley, 2012; Buitenwerf *et al.*, 2012), climate variability and change (Eamus and Palmer, 2007; 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 climate- and non-climate-related causes of woody plant biomass increase in savannas (and the invasion of trees into former grasslands), but there is *medium agreement* and *robust evidence* that climate change and rising CO_2 are contributing factors in many cases. The increased growth rate of C3 photosynthetic system trees relative to C4 grasses under by rising CO_2 could relieve the demographic bottleneck that keeps trees trapped within the flame zone of the grasses, a hypothesis supported by elevated CO_2 experiments with savanna saplings (Kgope *et al.*, 2010).

A model of grasslands, savannas and forests suggests that rising CO_2 does increase the likelihood of abrupt shifts to woodier states, but the transition will take place at different CO_2 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 CO_2 exposure but different land use histories, topographic position or soil depth (Wiegand *et al.*, 2005; Wu and Archer, 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 future increases in rainfall projected for most but not all savanna areas (WGI AR5 Annex I: Atlas of Global and Regional Climate Projections) – could lead to increased tree biomass.

4.3.3.2.2. Grasslands and shrublands

Rangelands (partly overlapping with savannas, above) cover approximately 30% of the earth's ice-free land surface and hold an equivalent amount of the world's terrestrial carbon (Booker *et al.*, 2013). 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 (*robust evidence and confidence*) (e.g., Sala *et al.*, 1988; Snyman and Fouché, 1993; Fay *et al.*, 2003; Peñuelas *et al.*, 2004, 2007; Prieto *et al.*, 2009; Peters *et al.*, 2010; Martí-Roura *et al.*, 2011; Booker *et al.*, 2013; Wu and Chen, 2013). 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, plant and soil nutrient contents (e.g., Fay *et al.*, 2003; Peñuelas *et al.*, 2004; Peñuelas *et al.*, 2007; Beier *et al.*, 2008; Sardans *et al.*, 2008a; Sardans *et al.*, 2008b; Sowerby *et al.*, 2008; Liu *et al.*, 2009; Miranda *et al.*, 2009; Albert *et al.*, 2011; Albert *et al.*, 2012; Selsted *et al.*, 2012; Walter *et al.*, 2012).

Precipitation changes were as important for mountain flora in Europe as temperature changes, and the greatest composition changes will probably occur when decreased precipitation accompany warming (Engler *et al.*, 2011). 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). 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; Sowerby *et al.*, 2010). Simulated increased spring temperature and decreased summer precipitation had a general negative effect on plant survival and plant growth, irrespective of the macroclimatic niche characteristics of the species. Against expectation, species with ranges extending into drier regions did not generally perform better under drier conditions (Bütof *et al.*, 2012).

Changing climate and land use have resulted in increased aridity and a higher frequency of droughts in drylands around the world, with increasing dominance of abiotic controls of land degradation (in contrast to direct human- or herbivore-driven degradation) and changes in hydrology and the erosion of soil by wind (Ravi *et al.*, 2010). In mixed shrub grasslands, the influence of drought periods could produce transient pulses of C that are much larger than the pulses produced by fire (Martí-Roura *et al.*, 2011). Most studies of changes in arid systems between grasslands and shrublands have focused on plant-soil feedbacks that favour shrubs growth. Summers drier than three-quarters of current rainfall decreased grass seedling recruitment to negligible values (Peters *et al.*, 2010). Management cannot reliably increase carbon uptake in arid and semiarid rangelands, which is most often controlled by abiotic factors not easily changed by management of grazing or vegetation (Booker *et al.*, 2013).

Other factors being equal, grasslands and shrublands in cool areas are expected to respond to warming with increased primary production, while those in hot areas are expected to show decreased production (*limited evidence, low confidence*). A shift to more woody vegetation states expected to occur (locally but not globally) in tropical grasslands of the African continent (Higgins and Scheiter, 2012). The response to warming and drought depends on site, year and plant species, as shown by manipulation experiments (Peñuelas *et al.*, 2004, 2007; Gao and Giorgi, 2008; Grime *et al.*, 2008; Shinoda *et al.*, 2010; Wu and Chen, 2013). In most temperate and arctic regions, the capacity to support richer (i.e. more diverse) communities is projected to increase with rising temperature, while decreases in water availability suggest a decline in capacity to support species-rich communities in most tropical and subtropical regions (Sommer *et al.*, 2010). Warming may cause an asymmetrical response of soil C and N cycles, causing N limitation that reduces acclimation in plant production (Beier *et al.*, 2008).

Some grasslands are 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; Gaudnik *et al.*, 2011). In a study of 162 plots over 25 years, N deposition drove grassland composition at the local scale, in interaction with climate, whereas climate changes were the predominant driver at the regional scale (Gaudnik *et al.*, 2011). N mineralization in shrublands under either arid or wet conditions is more sensitive to periodic droughts than systems under more mesic conditions (Emmett *et al.*, 2004). Decreased tissue concentrations of phosphorus were also associated with warming and drought (Peñuelas *et al.*, 2004; Beier *et al.*, 2008; Peñuelas *et al.*, 2012). Strong interactions between warming and disturbances have been observed, leading to increased N leaching from shrubland ecosystems (Beier *et al.*, 2004).

Most grasslands and shrublands are characterized by relatively frequent but low-intensity fires, which affect their plant species composition and demographics (e.g., Gibson and Hulbert, 1987; Uys *et al.*, 2004; Gill *et al.*, 1999; de Torres Curth *et al.*, 2012). Species composition changes may be as important in determining ecosystem impacts as the direct effects of climate on plant (Suttle *et al.*, 2007). Fire frequency, duration and intensity are primarily influenced by climate and secondarily by management (Pitman *et al.*, 2007; Lenihan *et al.*, 2008; Archibald *et al.*, 2009; Giannakopoulos *et al.*, 2009; Armenteras-Pascual *et al.*, 2011), and are therefore sensitive to climate change; the duration of the fire season is also projected to broaden (Clarke *et al.*, 2013). Changes in fire frequency may interact with changes in rainfall seasonality: for instance, if fires are followed by rainy spring periods in in northwestern Patagonia, as occurs with more frequent ENSO phenomena, there are more recruitment windows for shrubs (Ghermandi *et al.*, 2010). Relatively little is known regarding the combined effect of climate change and increased grazing by large mammals, 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. Species composition in desert areas is expected to shift in response to climate warming (Ooi *et al.*, 2009; Kimball *et al.*, 2010), including night-time warming (Collins *et al.*, 2010). Deserts are sparsely populated, but the people who do live there are amongst the poorest in the world (Millennium Ecosystem Assessment, 2005a). 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 which determines the location of the broad band of hot deserts approximately 15-30° N and S of the equator (Mitas and Clement, 2005; Seidel *et al.*, 2008; Johanson and Fu, 2009; Lu *et al.*, 2009; Zhou *et al.*, 2011). There may be a feedback to the global climate from an increase in desert extent, which differs in sign between deserts closer to the equator than 20 ° and those closer to the pole: in model simulations, extension of the near-equator ‘hot deserts’ causes warming, while extension of the near-boreal ”cold deserts” causes cooling, in both cases largely through albedo-mediated effects (Alkama *et al.*, 2012). Deserts are expected to become warmer and drier at faster rates than other terrestrial regions (Lapola *et al.*, 2009; Stahlschmidt *et al.*, 2011). Most deserts are already extremely hot, and therefore further warming is likely to be physiologically injurious rather than beneficial. The ecological dynamics in deserts are rainfall event-driven (Holmgren *et al.*, 2006), often involving the concatenation of a number of quasi-independent events. Some desert tolerance mechanisms (e.g., biological adaptations by long-lived taxa) may be outpaced by global climate change (Lapola *et al.*, 2009; Stahlschmidt *et al.*, 2011).

4.3.3.2.4 Mediterranean-type ecosystems

Mediterranean-type ecosystems occur on most continents, and are characterised by cool, wet winters and hot, dry summers. They were identified as being among the most likely to be impacted by climate change in AR4 and received extensive coverage (Fischlin *et al.*, 2007). Since then, further evidence has accumulated of climate risks to these systems from rising temperature (Giorgi and Lionello, 2008), rainfall change (declining in most but not all cases), increased drought (Section 23.2.3, 25.2) and increased fire frequency (Section 23.4.4). There have been observed shifts in phenology (Gordo and Sanz, 2010), range contraction of Mediterranean species (Pauli *et al.*, 2012), declines in the health and growth rate of dominant tree species (Allen *et al.*, 2010a; Sarris *et al.*, 2011; Brouwers *et al.*, 2012; Section 23.4.4) and increased risk of erosion and desertification, especially in very dry areas (Lindner *et al.*, 2010; Shakesby, 2011). Model projections show further species range contractions in the 21st century under all climate change scenarios. This will result in losses of biodiversity (*medium confidence*) (Maiorano *et al.*, 2011; Kuhlmann *et al.*, 2012; 23.6.4, 25.1).

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; Reidy Liermann *et al.*, 2012). Floodplains and wetland areas have become occupied for intensive urban and agricultural land use to the extent that many are functionally disconnected 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, fresh waters have some of the highest rates of extinction of any ecosystem for those species groups assessed for the IUCN Red List (estimated as much as 4% per decade for some groups, such as crayfish, mussels, fishes and amphibians in North America) (Dudgeon *et al.*, 2006), with estimates that at least 10,000-20,000 freshwater species are extinct or imperilled as a consequence of human activity (Strayer and Dudgeon, 2010). This is a particular concern given that freshwater habitats support 6% of all described species (Dudgeon *et al.*, 2006), including approximately 40% of the world's fish diversity and a third of the vertebrate diversity (Balian *et al.*, 2008).

It is *very likely* that these stressors to freshwater ecosystems will continue to dominate as human demand for water resources grows, accompanied by increased urbanization and expansion of irrigated agriculture (Vörösmarty *et al.*, 2000; Malmqvist *et al.*, 2008; Dise, 2009). However, climate change will have significant additional impacts (*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 Section 4.3.3.4; Klanderud and Totland, 2005; Smith *et al.*, 2005; Smol and Douglas, 2007b). It is noteworthy that these high latitude systems currently experience a relatively low level of threat from other human activities (Vörösmarty *et al.*, 2010). It is *likely* that the shrinkage and disappearance of glaciers will lead to the reduction of local and regional freshwater biodiversity, with 11-38% of the regional macroinvertebrate species pool expected to be lost following complete disappearance of glaciers (Jacobsen *et al.*, 2012; CC-RF2). Shrinkage of glaciers and the loss of small glaciers will most likely reduce beta diversity at the species and the genetic level, as predicted for the Pyrenees (Finn *et al.*, 2013). 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).

There is widespread evidence of rising stream and river temperatures over the past few decades (Langan *et al.*, 2001; Morrison *et al.*, 2002; Webb and Nobilis, 2007; Chessman, 2009; Ormerod, 2009; Kaushal *et al.*, 2010; van Vliet *et al.*, 2011; Markovic *et al.*, 2013– though see Arismendi *et al.*, 2012). Rising water temperature 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 temperature is also implicated in species range shifts (e.g., Comte and Grenouillet, 2013), implying changes in the composition of river fish communities (Daufresne and Boet, 2007; Buisson *et al.*, 2008; Comte *et al.*, 2013), especially in headwater streams where species are more sensitive to warming (e.g., Buisson and Grenouillet, 2009).

Rising temperatures in the well-mixed surface waters in many temperate lakes, resulting in reduced periods of ice formation (Livingstone and Adrian, 2009; Weyhenmeyer *et al.*, 2011) and earlier onset and increased duration and stability of the thermocline during summer (Winder and Schindler, 2004), are projected to favour a shift in dominance to smaller phytoplankton (Parker *et al.*, 2008; Winder *et al.*, 2009; Yvon-Durocher *et al.*, 2011) and cyanobacteria (Wiedner *et al.*, 2007; Jöhnk *et al.*, 2008; Paerl *et al.*, 2011), especially in those ecosystems experiencing high anthropogenic loading of nutrients (Wagner and Adrian, 2009); with impacts to water quality, food webs and productivity (O'Reilly *et al.*, 2003; Verburg *et al.*, 2003; Gyllström *et al.*, 2005; Parker *et al.*, 2008; Shimoda *et al.*, 2011). Prolonged stratification and associated anaerobic conditions near the sediment-water interface can increase the internal loading of phosphorus, particularly in eutrophic lakes (Søndergaard *et al.*, 2003; Wilhelm and Adrian, 2008; Wagner and Adrian, 2009).

In many freshwater ecosystems, the input of dissolved organic carbon through run-off from the catchment has increased, inducing changes in water colour (Hongve *et al.*, 2004; Evans *et al.*, 2005; Erlandsson *et al.*, 2008). Soil recovery from acidification and changed hydrological conditions (partly linked to increased precipitation) appear to be the main factors driving this development (Evans *et al.*, 2005; Monteith *et al.*, 2007). The resulting increased light attenuation can lead to lower algal concentrations and loss of submersed vegetation (Ask *et al.*, 2009; Karlsson *et al.*, 2009).

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 (Alahuhta *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 water bodies, especially those at high altitude (Rosset *et al.*, 2010). While less studied, it has been proposed that tropical ectothermic ("cold blooded") organisms 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 (*high confidence* in detection, *medium confidence* in attribution, see Cross-Chapter Box CC-RF; Xenopoulos *et al.*, 2005; Aldous *et al.*, 2011). 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 alter freshwater ecosystems, through changes in physical habitat and water quality (Bryant, 2009). Declining rainfall and increased inter-annual variability will most likely 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 (Johnson *et al.*, 2005; Bond *et al.*, 2008; Sheldon *et al.*, 2010, altered freshwater food webs (Ledger *et al.*, 2013), and drying out 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 long term (Limpens *et al.*, 2008). Mire ecosystems (i.e. bogs, transition bogs and fens) in Central Europe face severe climate-induced risk, with increased summer temperatures being particularly important (Essl *et al.*, 2012). Decreased dry season 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).

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 CO_2), or one third of the world's soil carbon stock (400-600 Pg) (Limpens *et al.*, 2008; Frolking *et al.*, 2011; 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 undergoing rapid 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 59.4±10.2 Mg CO_2 $ha^{-1}.yr^{-1}$ over 25 years (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), though inter-annual variability is large. Fluvial export can also be a significant contributor to carbon losses that has been largely overlooked to date, with recent estimates of DOC export from degraded tropical peatlands 50% higher than in intact systems (Moore *et al.*, 2013). Conserving peatland areas not yet developed for biofuels or other crops, or re-wetting and restoring degraded peatlands to preserve their carbon store are potential mitigation strategies.

Sea level rise will lead to direct losses of coastal wetlands with associated impacts on water birds 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

(Syvitski *et al.*, 2009; see Section 5.4.2.7). 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, and Permafrost Systems

The High Arctic region, with tundra-dominated landscapes, has warmed more than the global average over the last century (Kaufman *et al.*, 2009; see WGI AR5 Chapter 2). Changes consistent with warming are evident in the freshwater and terrestrial ecosystems 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). Most of the Arctic has experienced recent change in vegetation photosynthetic capacity, particularly adjacent to rapidly retreating sea ice (Bhatt *et al.*, 2010). Changes in terrestrial environments in Antarctica have also been reported. Vieira *et al.* (2010) show that in in the Maritime Antarctic permafrost temperatures are close to thaw. Permafrost warming in has been observed in continental Antarctica (Guglielmin and Cannone, 2012) and for the Palmer archipelago (Bockheim *et al.*, 2013).

Continued warming is projected to cause the terrestrial vegetation and lake systems of the Arctic to change substantially (*high confidence).* Continued expansion in woody vegetation cover in tundra regions over the 21st century is projected by the CMIP5 Earth System Models (Bosio *et al.*, 2012; see WGI AR5 Chapter 6); 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). Arctic vegetation change is expected to continue long after any stabilization of global mean temperature (see WGI AR5 Chapter 6; Falloon *et al.*, 2012). In some regions, reduced surface albedo due to increased vegetation cover is projected to cause further local warming even in scenarios of stabilized greenhouse gas concentrations (Falloon *et al.*, 2012).

In the Arctic tundra biome (in contrast to the boreal forests discussed in Section 4.3.3.1.1) vegetation productivity has systematically increased over the past few decades in both North America and northern Eurasia (Goetz *et al.*, 2007; Stow *et al.*, 2007; Jia *et al.*, 2009b; de Jong *et al.*, 2011; Myers-Smith *et al.*, 2011; Elmendorf *et al.*, 2012). 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 and expansion over the last half century (Forbes *et al.*, 2010; Myers-Smith *et al.*, 2011). Deciduous shrubs and graminoids respond to warming with increased growth (Walker, 2006; Epstein, 2008; Euskirchen *et al.*, 2009; Lantz *et al.*, 2010). Analyses of satellite time series data show the increased productivity trend is not unique to shrub-dominated tundra 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. The very large spatial scale over which these changes are occurring, the strong warming signal over much of the Arctic for the last five decades (Burrows et al. 2011) and the absence of strong confounding factors means that detection these changes in Arctic systems and their attribution to global warming can be made with *high confidence,* despite the relatively short time frame of most observations (Figure 4-4).

Shrub expansion and height changes are particularly important 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 experiencing increasing 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 long-distance dispersal to areas reinitialized by burning (Epstein *et al.*, 2007; Goetz *et al.*, 2011). When already present at the boreal-tundra ecotone, shrub and tree species show increased productivity with warmer conditions (Devi *et al.*, 2008; Andreu-Hayles *et al.*, 2011; Elmendorf *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; Jorgenson *et al.*, 2010), produce landforms associated with thawing of ice-rich permafrost, and can create conditions that alter vegetation succession (Racine *et al.*, 2004; Lantz *et al.*, 2009; Higuera *et al.*, 2011).

It is *virtually certain* that the area of Northern Hemisphere permafrost will continue to decline over the first half of the 21st century (see WGI Chapter 12) in all RCP scenarios (Figure 4-9; Caesar *et al.*, 2013; Koven *et al.*, 2013). In the RCP2.6 scenario of an early stabilization of CO_2 concentrations, the 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 CO_2 concentration is stabilized at approximately 550ppmv by the mid-21st century, the simulations that extend beyond 2100 show permafrost continuing to decline for at least another 250 years. In the RCP8.5 scenario of ongoing CO_2 rise, the permafrost area is simulated to approach zero by the middle of the 22nd century in simulations that extend beyond 2100. RCP8.5 simulations that ended at 2100 showed continued permafrost decline in the late 21st century, although at slower rates in some cases as the remaining permafrost area decreases (Figure 4-9.).

[INSERT FIGURE 4-9 HERE
Figure 4-9: CMIP5 multi-model simulated area of Northern Hemisphere permafrost in the upper 3m of soil, from 1850 to 2100 or 2300 depending on extent of individual simulations. Each panel shows historical (1850-2005) and projected (2005 to 2100 or 2300) simulations for (a) RCP2.6, (b) RCP4.5, and (c) RCP8.5. The observed current permafrost extent is 15×10^6 km^2. (Based on Koven *et al.*, 2013, with analysis extended to 2300 following Caesar *et al.*, 2013).]

Frozen soils and permafrost currently hold about 1700 PgC, more than twice the carbon than the atmosphere, and thus represent a particularly large vulnerability to climate change (i.e., warming) (see WGI AR5 Chapter 6). Although the Arctic is currently a net carbon sink, continued warming will act to 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_2O, from the terrestrial biosphere into the atmosphere (*high confidence*, 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; see WGI AR5 Chapter 6 for detailed treatment of biogeochemistry, including feedbacks). 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 snow cover and resulting spread of shrubs and trees into areas currently dominated by tundra has 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 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; Stirling and Derocher, 2012), and it is expected that the populations of other Arctic animals will be affected dramatically by climate change, often in complex, but potentially dramatic ways (e.g., Post *et al.*, 2009; Sharma *et al.*, 2009; Gallant *et al.*, 2012; Gilg *et al.*, 2012; Post and Brodie, 2012; Gauthier *et al.*, 2013; Nielsen and Wall, 2013; Prost *et al.*, 2013; White *et al.*, 2013). 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 and wildfire, plus other ecosystem impacts (e.g., Westerling *et al.*, 2006; Crimmins *et al.*, 2009; van Mantgem *et al.*, 2009; Pederson *et al.*, 2010; Muhlfeld *et al.*, 2011; Brusca *et al.*, 2013; Williams *et al.*, 2013), and disturbance will continue to be an important agent of climate-induced change in this region (Littell *et al.*, 2010). Globally, tree line altitude appears to be changing, although not always in simple ways (Harsch *et al.*, 2009; Tingley *et al.*, 2012) and may sometimes be due to factors not related to climate change (Schwilk and Keeley, 2012). Responses to climate change in high-altitude ecosystems are taking place in Africa, Asia, Europe and elsewhere (Yasuda *et al.*, 2007; Cannone *et al.*, 2007, Cannone *et al.*, 2008; Lenoir *et al.*, 2008; Britton *et al.*, 2009; Chen *et al.*, 2009; Cui and Graf, 2009; Normand *et al.*, 2009; Allen *et al.*, 2010a; Eggermont *et al.*, 2010; Lenoir *et al.*, 2010; Chen *et al.*, 2011; Engler *et al.*, 2011; Kudo *et al.*, 2011; Laurance *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 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 (e.g., Tibet; Cui and Graf, 2009).

_____ START BOX 4-4 HERE _____

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 4-10). Until now, these changes have been gradual shifts across temperature and moisture gradients, rather than abrupt. 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.*, 2009; Peng *et al.*, 2011; Elmendorf *et al.*, 2012) as well as from satellite observations (Goetz *et al.*, 2007; Beck *et al.*, 2011b; Xu *et al.*, 2013) and captured in Earth system models (McGuire *et al.*, 2010).

Gradual changes in composition resulting from decreased evergreen conifer productivity and increased mortality, as well as increased deciduous species productivity, can be facilitated by more rapid shifts associated with fire disturbance where it can occur (Mack *et al.*, 2008; Johnstone *et al.*, 2010; Roland *et al.*, 2013). Each of these interacting processes, as well as insect disturbance and associated tree mortality, are tightly coupled with warming-induced drought (Choat *et al.*, 2012; Ma *et al.*, 2012; Anderegg *et al.*, 2013a). Similarly, gradual productivity increases at the boreal-tundra ecotone are facilitated by long distance dispersal into areas disturbed by tundra fire and thermokarsting (Tchebakova *et al.*, 2009; Brown, 2010; Hampe, 2011). 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 modelling studies, which are often poorly constrained by observations.

[INSERT FIGURE 4-10 HERE
Figure 4-10: Tundra-Boreal Biome Shift. Earth system models predict a northward shift of Arctic vegetation with climate warming, as the boreal biome migrates 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, permafrost thawing and methane emissions, with net feedbacks to additional climate change.]

_____ END BOX 4-4 HERE _____

4.3.3.5. Highly Human-Modified Systems

About a quarter of the land surface is now occupied by ecosystems highly modified by human activities. In this section we assess the vulnerability to climate change only of those modified systems not dealt with elsewhere, i.e. excluding agriculture (Chapter 7), freshwater fisheries (Chapter 3) and urban areas (Chapter 8).

4.3.3.5.1. Plantation forestry

Plantation forests are established through afforestation or reforestation, often with tree crop replacement (Dohrenbusch and Bolte, 2007; FAO, 2010). They differ from natural or semi-natural forests (Section 4.3.3.1) by generally being even-aged, having a reduced species diversity (sometimes of non-native species) and being dedicated to the production of timber, pulp and/or bioenergy. Plantation forests contribute 7% to the global forest area (FAO, 2010), an increase of five million ha between 2000 and 2010 (FAO, 2010). Most recent plantations have been established by afforestation of non-forest areas in the tropics and subtropics and some temperate regions, particularly China (Kirilenko and Sedjo, 2007; FAO, 2010) Afforestation usually results in net CO_2 uptake from the atmosphere (Canadell and Raupach, 2008; Van Minnen *et al.*, 2008) but does not necessarily result in a reduction in global warming (Bala *et al*., 2007; see Section 4.3.4.5).

Growth rates in plantation forests have generally increased during the last decades but the variability is large. In forests that are not highly water-limited, increased growth is consistent with higher temperatures and extended growing seasons. As in the case of forests in general, clear attribution is difficult because of the interaction of multiple environmental drivers as well as changes in forest management (e.g., Boisvenue and Running, 2006; Ciais *et al.*, 2008; Dale *et al.*, 2010; see also Section 4.3.3.1). In Europe much of the increase has been attributed to recovery following previously more intense harvesting (Ciais *et al.*, 2008; Lindner *et al.*, 2010).

Several studies using forest yield models suggest future increases in forest production (Kirilenko and Sedjo, 2007). These results may over-estimate the positive effects of elevated CO_2 (Kirilenko and Sedjo, 2007; see Section 4.2.4.4). The effects of disturbances such as wildfires, forest pests, pathogens and windstorms, which are major drivers of forest dynamics, are poorly represented in the models (Loustau, 2010; see also Section 4.3.3.1 and Box 4-2). The 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 Section 4.3.3.1.2). Decreased forest production is expected in already-dry forest regions for which further drying is projected, such as the south-western U.S. (Williams *et al.*, 2010b). Extreme drying may also decrease yields in forests currently not water-limited (e.g., Sitch *et al.*, 2008; see Section 4.3.3.1). Plantations in cold-limited areas could benefit from global warming, provided that increased fires, storms, pests and pathogens do not outweigh the potential direct climate effects on tree growth rates.

Low species diversity (and low genetic diversity within species where clones or selected provenances are used) renders plantation forests less resilient to climate change than natural forests (e.g., Hemery, 2008). Choosing provenances that are well adapted to current climates but pre-adapted to future climates is difficult because of uncertainties in climate projections at the timescale of a plantation forest rotation (Broadmeadow *et al.*, 2005). How forest pests and pathogens will spread as a result of climate change and other factors is highly uncertain. New pathogen-tree interactions may arise (e.g., Brasier and Webber, 2010). Adaptive management can decrease the vulnerability of plantation forests to climate change (Hemery, 2008; Bolte *et al.*, 2009; Seppälä, 2009; Dale *et al.*, 2010). For example, risk spreading by promoting mixed stands, containing multiple species or provenances, combined with natural regeneration (Kramer *et al.*, 2010), has been advocated as an adaptation strategy for temperate forests (Hemery, 2008; Bolte *et al.*, 2010) and tropical forests (Erskine *et al.*, 2006; Petit and Montagnini, 2006). Incomplete knowledge of the ecology of tropical tree species and little experience in managing mixed tropical tree plantations remains a problem (Hall *et al.*, 2011). Especially at the equator-ward limits 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

The production of modern bioenergy is growing rapidly throughout the world in response to climate mitigation and energy security policies (Cochrane and Barber, 2009). WGIII AR5 Chapter 7 addresses the potential of bioenergy as a climate mitigation strategy. The vulnerability of bioenergy systems to climate change is similar to that of plantation forestry (Section 4.3.3.5.1) or food crops (Section 7.3): in summary, they remain viable in the future in most but not all locations, but their viability is increasingly uncertain for high levels of climate change (Ma and Zhou, 2012). Oliver *et al.* (2009a) suggested that rising CO_2 might contribute to increased drought tolerance in bioenergy crops (since it leads to improved plant water use efficiency).

The unintended consequences of large-scale land-use changes driven by increasing bioenergy demand are addressed in Section 4.4.4.

4.3.3.5.3. Cultural landscapes

Cultural landscapes are characterized by a long history of human-nature interactions, which results in a particular configuration of species and landscape pattern attaining high cultural significance (Rössler, 2006). Examples are grassland or mixed agriculture landscapes in Europe, rice landscapes in Asia (Kuldna *et al.*, 2009) and many others across the globe (e.g., Rössler, 2006; Heckenberger *et al.*, 2007). Such landscapes are often agricultural, but we deal with them here because their perceived value is only partly in terms of their agricultural products.

It has been suggested that protected area networks (such as Natura 2000 in Europe, which includes many cultural landscape elements) be adjusted to take into account climate change (Jin *et al.*, 2010; Heubes *et al.*, 2011). Conserving species in cultural landscapes (e.g., EU Council, 1992) generally depends on maintaining certain types of land use. Doing so under climate change requires profound knowledge of the systems and species involved, and conservation success so far has been limited (see Kirdyanov *et al.*, 2012 for a notable exception). Understanding the relative importance of climate change and land management change is critical (Settele and Kühn, 2009). To date land use changes have been the most obvious driver of change (Nowicki *et al.*, 2007); impacts have been attributed climate change (with *low* to *medium confidence*) in only a few examples (Devictor *et al.*, 2012). Even in these, combined land use-climate effects explain the pattern of observed threats better than either alone (Schweiger *et al.*, 2008; Clavero *et al.*, 2011; Schweiger *et al.*, 2012).

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 (Figure 4-4).

4.3.3.5.4. Urban ecosystems

Although urban areas (for definition see Section 8.1.2) cover only 0.5 % of Earth’s land surface (Schneider *et al.*, 2009), over half of humanity lives there (increasing annually by 74 million people; United Nations *et al.*, 2012) and they harbour a large variety of species (McKinney, 2008). The frequency and magnitude of warm days and nights (heat waves) is *virtually certain* to increase globally the future it (IPCC, 2012); this trend is higher in urban than rural areas (Leonelli *et al.*, 2011). Heavy rainfall events are also projected to increase (IPCC, 2012), and although the hydrological conditions in urban areas make them prone to flooding *(medium confidence)*, there is *limited evidence* that they will be over-proportionally affected. It is *very likely* that sea level rise in future will contribute to flooding, erosion and salinisation of coastal urban ecosystems (IPCC, 2012). Climate change is projected to increase the frequency of landslides (UN Habitat, 2011). Climate change impacts on urban ecosystems and biodiversity have received comparatively little attention, with water availability being an exception (Hunt and Watkiss, 2011). Changes in water availability and quality either due to changes in precipitation, evaporation or in salinity regimes will especially affect urban freshwater ecosystems (Hunt and Watkiss, 2011). As in other ecosystems, climate change will leaded to a change in species composition, the frequency of traits and ecosystem services from urban ecosystems. Knapp *et al.* (2008b) found that trait composition of plant communities changes during urbanization towards adaptive characteristics of dry and warm environments (see also Sections 4.2.4.6 and 4.3.2.5). Urban areas are one of the main points of introduction of alien species (e.g., for plants through urban gardening; Li *et al.*, 2009). Increased damage by phytophagous insects to plants in urban environments is anticipated (Kollár *et al.*, 2009; Lopez-Vaamonde *et al.*, 2010; Tubby and Webber, 2010; see also Section 8.2.4.5).

4.3.4. Impacts on Key Ecosystem Services

Ecosystem services are the benefits which people derive from ecosystems (see glossary). Many ecosystem services are plausibly vulnerable to climate change. The Millennium Ecosystem Assessment classification (Millennium Ecosystem Assessment, 2003) recognises *provisioning services* such as food (Chapter 7), fibre (Section 4.3.4.2), bioenergy (Section 4.3.4.3) and water (Chapter 3); *regulating services* such as climate regulation (Section 4.3.4.5), pollination, pest and disease control (Section 4.3.4.4) and flood control (Chapter 3); *supporting services* such as primary production (Section 4.3.2.2) and nutrient cycling (Section 4.2.4.2, and indirectly Section 4.3.2.3); and *cultural services*, including recreation, aesthetic and spiritual benefits (Section 10.6). The following section focusses on ecosystem services not already covered in the sections referenced above.

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 (Dullinger *et al.*, 2012; Urban *et al.*, 2012). Climate change impacts on habitats for biodiversity are already occurring (see the polar bear example in Section 28.2.2.1.3) but are not yet a widespread phenomenon. Models of future climate change-induced shifts in the distribution of ecosystems suggest that many species could be outside of their preferred habitats with the next few decades (Urban *et al.*, 2012; see Sections 4.3.2.5, 4.3.3, and Figure 4-1).

Hole *et al.* (2009) report that the majority of African birds would have to move large distances (up to several hundred kilometres) over the next 60 years (under SRES B2a), resulting in substantial turnover of species within protected areas (>50% turnover in more than 40% of Important Bird Areas of Africa). In order to reach suitable climates they will have to migrate across unfavourable habitats. Many may continue to find suitable climate within the protected area network, but will be forced to cope with new habitat constraints (Hole *et al.*, 2009). Araujo *et al.* (2011) estimate that by 2080 approximately 60% (58 ± 2.6%) of plants and vertebrate species will no longer have favourable climates within European protected areas, often pushing them into unsuitable or less preferred habitats (based on SRES A1, A2, B1 and A1FI scenarios). Wiens *et al.* (2011) project similar effects in the western US (until the year 2069, based on SRES A2 scenarios), but also find that climate change may open up new opportunities for protecting species in areas where climate is currently unsuitable. In some cases climate change may allow species to move into areas of lower current or future land use pressure including protected areas (Bomhard *et al.*, 2005). These 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 (see Sections 4.4.2.2 and 4.4.2.3).

In the long term, some habitat types may disappear entirely due to climate change (see Section 4.3.3 and Figure 4-1). Climates are projected to occur in the future which at least in some features do not represent climates that existed in the past (Williams *et al.*, 2007b; Wiens *et al.*, 2011), and in the past climate shifts have resulted in vegetation types that have no current analogue (Section 4.2.3). The impacts of habitat change on species abundance and extinction risk is difficult to evaluate because at least some species are able to adapt to novel habitats (Prugh *et al.*, 2008; Oliver *et al.*, 2009b). The uncertainty in habitat specificity is one reason why quantitative projection of changes in extinction rates is difficult (Malcolm *et al.*, 2006).

The effects of climate change on habitat quality are less well studied than shifts in species or habitat distributions. Several recent studies indicate that climate change may have altered habitat quality already and will continue to do so (Iverson *et al.*, 2011; Matthews *et al.*, 2011). For example, decreasing snowfall in the southwestern US has negatively affected the habitat for songbirds (Martin and Maron, 2012).

4.3.4.2. Timber and Pulp Production

In most areas with forest plantations, forest growth rates have increased during the last decades, but the variability is large, and in some areas production has decreased (see Section 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 Section 4.3.3.1). In Europe a reduction in harvesting intensity has contributed (Ciais *et al.*, 2008; Lindner *et al.*, 2010).

Forest yield models project future increases in forest production under climate change, perhaps over-optimistically (Kirilenko and Sedjo, 2007; see Section 4.2.4.4. Using a model that accounts for fire effects and insect damage, Kurz *et al.* (2008) showed that the Canadian forest sector may have transitioned from a sink to a source of carbon.

4.3.4.3. Biomass-derived Energy

Bioenergy sources include traditional forms such as wood and charcoal from forests (see Section 4.3.3.1) and more modern forms such as the industrial burning of biomass wastes, the production of ethanol and biodiesel and plantations of bioenergy crops. 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. Generally, potentials of bioenergy production under climate change may be high, but are very uncertain (Ma and Zhou, 2012).

4.3.4.4. Pollination, Pest, and Disease Regulation

It can be inferred that global change will result in new communities (Gilman *et al.*, 2010; Schweiger *et al.*, 2010). As these will have had little opportunity for coevolution, changes in ecological interactions, such as shifts herbivore diets, the range of prey of predators or in pollination networks are to be expected (Tylianakis *et al.*, 2008; Schweiger *et al.*, 2012). This may result in temporarily reduced effectiveness of the "regulating services", which generally depend on species interactions (Montoya and Raffaelli, 2010). Burkle *et al.* (2013) show that the loss of species reduces co-occurrence of interacting species and thus reduces ecosystem functions based on them.

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 by climate change, e.g. through a longer warm season during which to breed, and indirectly e.g. through the quality of food plants (Jamieson *et al.*, 2012) or *via* changes in their natural enemies (predators and parasitoids). Insects have well-defined temperature optima; warming towards the optimum leads to increased vitality and reproduction (Allen *et al.*, 2010a). Mild winters in temperate areas promote pests formerly controlled by frost sensitivity. For the vast majority of indirect effects, information is scarce. Further assessments of climate change effects on pest and disease dynamics are found in Sections 7.3.2.3 for agricultural pests and 11.5.1 for human diseases.

Climate change has severe negative impacts on pollinators (including honeybees) and pollination (Kjøhl *et al.*, 2011) (*medium confidence*). After land-use changes, climate change is 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). The potential influence of climate change on pollination can be manifold (compare Hegland *et al.*, 2009; Schweiger *et al.*, 2010; Roberts *et al.*, 2011). There are a few observational studies, which mostly relate to the phenological de-coupling of plants and their pollinators (Gordo and Sanz, 2005; Bartomeus *et al.*, 2011). While Willmer (2012) states, based on experimental studies, that phenological effects may be less important than has been suggested, an analysis of phenological observations in plants by Wolkovich *et al.* (2012) shows that experimental data on phenology may grossly underestimate the actual phenological shifts.

Le Conte and Navajas (2008) state that the generally observed decline in honeybees is a clear indication of an increasing susceptibility to global change phenomena, with pesticide application, new diseases and stress (and a combination of these) as the most relevant causes. Climate change may contribute by modifying the balance between honeybees and their environment (including exposure or susceptibility to diseases). Honeybees show a high capacity to adjust to a variety of environments; their high genetic diversity should allow them to also cope with climatic change (Bartomeus *et al.*, 2011). The preservation of genetic variability within honeybees is regarded as a key adaptation strategy for pollination services (Le Conte and Navajas, 2008).

4.3.4.5. Moderation of Climate Change, Variability, and Extremes

The focus of this section is on processes operating at regional to global scales, rather than the well-known microclimatic benefits of ecosystems in smoothing day-night temperature variations and providing local evaporative cooling. In the decade 2000-2009, the global net uptake of CO_2 by terrestrial ecosystems was a large fraction of the anthropogenic CO_2 emissions to the atmosphere from all sources, reducing the rate of climate change proportionately (Section 4.3.2.3; WGI AR5 Section 6.3.2).

Afforestation or reforestation are a potential climate mitigation options (Van Minnen *et al.*, 2008; Vaughan and Lenton, 2011; Fiorese and Guariso, 2013; Singh *et al.*, 2013), but as discussed in Section 4.2.4.1, the net effect of afforestation on the global climate is mixed and context-dependent. Wickham *et al.* (2012) found significant positive correlations between the average annual surface temperature and the proportion of forest in the landscape and conclude that the climate benefit of temperate afforestation is unclear. Where low-albedo forest canopies replace higher-albedo surfaces such as soil, grassland or snow, the resultant increase in net radiative forcing counteracts the benefits of carbon sequestration to some degree (Arora and Montenegro, 2011). Where the cloud cover fraction is low and the albedo difference is large, i.e. outside the humid tropics, the long-term net result of afforestation can be global warming (Bala *et al.*, 2007; Bathiany *et al.*, 2010; Schwaiger and Bird, 2010). Accounting for changes in albedo and indirect greenhouse effects are not currently required in the formal rules for quantifying for the climate effects of land use activities (Schwaiger and Bird, 2010; Kirschbaum *et al.*, 2012). There are potential negative tradeoffs between afforestation for climate mitigation purposes and other ecosystem services, such as water supply (Jackson *et al.*, 2005) and biodiversity maintenance (CBD, 2012; Russell *et al.*, 2012).

It has been suggested (Ridgwell *et al.*, 2009) that planting large areas of crop varieties with highly reflective leaves could help mitigate global change. Model analyses indicate this “geo-engineering” strategy would be marginally effective at high latitudes, but have undesirable climate consequences at low latitudes. Measurements of leaf albedo in major crops show that the current range of variability is insufficient to make a meaningful difference to the global climate (Doughty *et al.*, 2011).

4.4. Adaptation and its Limits

4.4.1. Autonomous Adaptation by Ecosystems and Wild Organisms

Autonomous adaptation (see glossary under adaptation) 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 called “coping capacity”. The capacity for autonomous adaptation is part of resilience but is not exactly synonymous (Walker *et al.*, 2004).

All social and ecological systems have some capacity for autonomous adaptation. 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. 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, behaviour, 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, through in- or out-migration, or local extinction.

The ability to project impacts of climate change on ecosystems is complicated by the potential for species to adapt. Adaptation by individual species increases 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). It may also affect their interactions with other species leading to disruption of the biotic community (Visser and Both, 2005).

4.4.1.1. Phenological

Changes in phenology are occurring in many species and locations (Section 4.3.2.1). Further evidence since AR4 shows how this can be an adaptation to climate change, but also the limits to phenological adaptation. An organism’s phenology is typically highly adapted to the climate seasonality of the environment in which it evolved. Species unable to adjust their phenological behaviour will be negatively affected, particularly in highly seasonal habitats (Both *et al.*, 2010).

Moreover, 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. Systematic cross-taxa studies indicate different rates of phenological change for different species and trophic levels (Parmesan, 2007; Cook *et al.*, 2008; Thackeray *et al.*, 2010). If adaptation is insufficiently rapid or coordinated between interdependent species, disruption of ecological features such as trophic cascades, competitive hierarchies, and species coexistence is inferred to result (Nakazawa and Doi, 2012). Lack of coordination can occur if one of the species is cued to environmental signals that are not affected by climate change, such as day length (Parmesan, 2006). Increasing temperatures may either bring species more into or out of synchrony, depending on their respective starting positions (Singer and Parmesan, 2010), although evidence is more towards a loss of synchrony (Thackeray *et al.*, 2010).

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. For example, bird 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 date showed stable or increasing populations between 1990 and 2000, while those 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*) between 1987 and 2003, populations declined by 90% in areas where food peaked early in the season and the arrival of the birds was mis-timed, but not in areas with a later food peak which could still be exploited by early-breeding birds (Both *et al.*, 2006). However, compensating processes can exist: for example, in a 4-decade study of great tits (*Parus major*), breeding populations were buffered against phenological mismatch due to relaxed competition between individual fledglings (Reed *et al.*, 2013). Between 1970 and 1990, changes in migration date did not predict changes in population (Møller *et al.*, 2008).

Bird breeding can also be affected by phenological shifts in competing species and predators. Between 1953 and 2005 in south-western Finland, the onset of breeding of the resident great tit *Parus major* and the migratory pied flycatcher (*Ficedula hypoleuca*) became closer to each other, increasing competition between them (Ahola *et al.*, 2007). The edible dormouse (*Glis glis*), a nest predator, advanced its hibernation termination by -8 days per decade in the Czech Republic between 1980 and 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).

Plant-insect interactions have also been observed to change. In Illinois, USA, the pattern of which plants were pollinated by which bees were altered by differing rates of phenological shifts and landscape changes over 120 years, with 50% of bee species becoming locally extinct (Burkle *et al.*, 2013). Increasing asynchrony of the winter moth (*Operophtera brumata*) and its feeding host oak tree (*Quercus robur*) in the Netherlands was linked to increasing spring temperatures but unchanging winter temperatures (van Asch and Visser, 2007). Warmer temperatures shorten the development period of European pine sawfly larvae (*Neodiprion sertifer* Geoffr.), reducing the risk of predation and potentially increasing the risk of insect outbreaks, but interactions with other factors including day length and food quality may complicate this prediction (Kollberg *et al.*, 2013). In North America, the spruce budworm (*Choristaneura fumiferana*) lays eggs with a wide range of emergence timings, so the population as a whole is less sensitive to changing phenology of host trees (Volney and Fleming, 2007).

The environmental cues for phenological events are complex and multi-layered (Körner and Basler, 2010; Singer and Parmesan, 2010). For instance, many late-succession temperate trees require a chilling period in winter, followed by a threshold in day length, and only then are sensitive to temperature. As a result, simple projections of current phenological trends may be misleading, since the relative importance of cues can change (Cook *et al.*, 2012b). 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, since plants begin flowering earlier as a result of earlier snowmelt (Inouye, 2008).

It has been suggested that shorter generation times 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 on adaptation rates (Thackeray *et al.*, 2010).

There is *high confidence* [*much evidence, medium agreement*] that climate change-induced phenological shifts will continue to alter the interactions between species in regions with a marked seasonal cycle.

4.4.1.2. Evolutionary and Genetic

Since AR4 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 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.*, 2012a; 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 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. There is a paucity of observational or experimental data that can be used for detection and attribution of recent climate effects on evolution.

Observed evolutionary and genetic responses to rapid changes in climate - There is a small but growing body of observations supporting the AR4 assessment that some species may have adapted to recent climate warming or to climatic extremes through genetic responses (e.g., plants: Franks and Weis, 2008; Hill *et al.*, 2011; Anderson *et al.*, 2012b; vertebrates: Ozgul *et al.*, 2010; Phillimore *et al.*, 2010; Husby *et al.*, 2011; Karell *et al.*, 2011; insects: Buckley *et al.*, 2012; van Asch *et al.*, 2012). Karell *et al.* (2011) found increasing numbers of brown genotypes of the tawny owl (*Strix aluco*) 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 white genotype. Earlier spawning by the common frog (*Rana temporaria*) in Britain could be attributed largely to local genetic adaptation to increasing spring temperatures (Phillimore *et al.*, 2010). Using a combination of models and observations, Husby *et al.* (2011) have built a case for detection and attribution of genetic adaptation in an insectivorous bird, and in an 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, in part because factors such as limited existing 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 weaknesses, 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). Rapid advances in quantitative genetics, genomics and phylogenetics, combined with recent progress on conceptual frameworks, 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). In sum, 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* for detection and attribution across all species; Figure 4-4).

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 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 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 provides insight into evolutionary responses 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 new evidence that epigenetic mechanisms, such as DNA methylation, could allow very rapid adaptation to climate (Paun *et al.*, 2010; Zhang *et al.*, 2013).

Mechanisms mediating rapid evolutionary response to future climate change - Studies of genetic variability across species ranges, and models that couple gene flow with spatially-explicit population dynamics, suggest counterintuitive responses to climate change. Too much or too little gene flow to populations at range margins can create 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 evidence from experiments, models and observations that populations in the centre of species ranges may in some cases be more sensitive to environmental change than those at range boundaries (Bell and Gonzalez, 2009). Generalization is complicated by the interactions between local adaptation, gene flow, population dynamics and species interactions (Bridle *et al.*, 2010; Norberg *et al.*, 2012).

Substantial progress has been made since AR4 in developing models for exploring whether genetic adaptation is fast enough to track climate change. Models of long-lived tree species suggest that existing genetic variation may be sufficient to slightly 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 climate change, 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 modelled avian breeding times tracked climate change, only at low to moderate rates of change. For a herbivorous insect with an even faster life cycle, van Asch *et al.*, (2007; 2012) predicted that rapid evolution of the phenological response should have allowed it to track recent warming, which it has. More broadly, models suggest that species with short generation times, (one year or less), potentially have the capacity to genetically adapt to even the most rapid rates of projected climate change given large enough present-day populations, but species with longer generation times or small populations could be at risk of extinction at moderate to high rates of climate change (Walters *et al.*, 2012; Vedder *et al.*, 2013). Recent experimental and theoretical work on "evolutionary rescue" shows that long-term avoidance of extinction through genetic adaptation to hostile environments is possible, but 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 be viewed with considerable caution because there are many evolutionary and ecological mechanisms not accounted for in most models that can either speed up or inhibit heritable adaptation to climate change (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 greater maladaptation to climate change, resulting in rapid population declines (Hendry and Gonzalez, 2008; Ferriere and Legendre, 2013). Phenotypic plasticity is thought to generally improve the odds of adaptation to climate change. High plasticity in the face of climate change that has low fitness costs can greatly improve the odds of adaptation; however, plasticity with high costs leads to only modest amounts of adaptation (Chevin *et al.*, 2010).

AR4 concluded that "projected rates of climate change are *very likely* to exceed rates of evolutionary adaptation in many species (*high confidence*)" (Fischlin *et al.*, 2007). Work since then provides a similar, but more nuanced view of rapid evolution in the face of future climate change. The lack of adaptation in some species to recent changes in climate, broad support for niche conservatism, and models showing limited adaptive capacity 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.

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 observed 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 4-5). 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 inland water systems contain many barriers to migration, in the form of habitat fragmentation, roads, human settlements and dams. Mountain ecosystems are less constrained by these factors than flat-land ecosystems, but have additional impediments for species already close to the top of the mountain.

4.4.2. Human-Assisted Adaptation

Human-assisted adaptation means a deliberate, 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. 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 Chapters 8 to 17, and more on non-human organisms and ecosystems, while acknowledging the importance of the human elements within the ecosystem. Intervention in this context means a range of actions, including ensuring the presence of suitable habitat and dispersal pathways; reducing non-climate stressors; physically moving organisms, storing and establishing them in new places. In addition to the other approaches assessed in this section, "Ecosystem-based Adaptation" (see Cross-Chapter Box CC-EA) 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. Note that there are risks associated with all forms of human-assisted adaptation (see Section 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). Protective and restorative actions aimed at increasing resilience can also be cost-effective means as part of an overall adaptation strategy to help people to adapt to the adverse effects of climate change and may have other social, economic and cultural benefits. This is part of "ecosystem-based adaptation" (Colls *et al.*, 2009; Box CC-EA).

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 most biodiversity exists outside rather in protected areas and this between-protected area matrix is as important; that it is usually 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 (Thomas *et al.*, 2012). Hickler *et al.* (2012) analysed the layout of protected areas in Europe and concluded that under projected 21st century climate change a third to a half of them would potentially be occupied by different vegetation than they currently represent. The new areas that need 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 since it is the network of habitats and protected areas that confers resilience rather than any single element (Heller and Zavaleta, 2009); 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). Dunlop *et al.* (2012) assessed the implications of climate change for biodiversity conservation in Australia and found many opportunities to facilitate the natural adaptation of biodiversity, including expanding the network of protected areas and restoring habitat at a large scale.

4.4.2.3. Landscape and Watershed Management

The need to include climate change into the management of vulnerable ecosystems is explicitly included in the strategic goals of the Convention on Biological Diversity. Oliver *et al.* (2012b) developed decision trees based on three scenarios: a) *adversely sensitive*, where areas within the species current geographical range will become climatically unsuitable with a changing climate; b) *climate overlap*, where there are areas that should remain climatically suitable within the species' range; and c) *new climatic space*, which refers to areas outside of the current range that are projected to become suitable. Heller and Zavaleta (2009) reviewed recommendations in the published literature and argue that the majority of them, such as increase habitat heterogeneity of sites and connectivity of habitats across landscapes, lack sufficient specificity to ensure the persistence of many species and related ecosystem services to ongoing climate change. To date, recommendations are overwhelmingly focused on ecological data, neglecting social science insights. Few resources or capacity exist to guide adaptation planning processes at any scale.

Climate-induced impacts to hydrological and thermal regimes in freshwater systems can be offset through improved management of environmental flow releases from reservoirs (Arthington *et al.*, 2006; Poff *et al.*, 2010; Arthington *et al.*, 2010 and references therein). Protection and restoration of riparian vegetation in small stream systems provide an effective strategy to moderate temperature regimes and offset warming, and protect water quality for downstream ecosystems and water supply areas (Davies, 2010; Capon *et al.*, 2013).

General principles for management adaptations were summarized from a major literature review by West *et al.* (2009). They suggest that in the context of climate change, 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, principles 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.

Steenberg *et al.* (2011) simulated the effect on adaptive capacity of three variables related to 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. Manipulation of vegetation composition and stand structure has been proposed as a strategy for offsetting climatic change impacts on wildfires in Canada. Large areas of boreal forests are currently being harvested and there may be opportunities for using planned manipulation of vegetation for management of future wildfire risks. This management option could also provide an additional benefit to the use of assisted species migration since the latter would require introducing non-flammable broadleaves species into forests which are otherwise highly flammable (Girardin *et al.*, 2013b; Terrier *et al.*, 2013). Harvesting practices, like partial cuts that limit the opening of the forest cover created by harvest, will be a key element to maintain diverse forest compositions and age class distributions in boreal forests. Other sound option for decreasing the exposure of silvicultural investments to an increasing fire danger is to use tree species necessitating a shorter rotation (Girardin *et al.*, 2013a).

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; Ste-Marie *et al.*, 2011). The options for management can be summarized as: i) try to maintain or improve existing habitat or environment so that species don't have to move (e.g., Yan *et al.*, 2013), ii) maintain or improve migration corridors, including active management to improve survival along the moving margin of the distribution (Lawson *et al.*, 2012), and iii) directly translocate species or genetically distinct populations within a species (Aitken *et al.*, 2008; Hoegh-Guldberg *et al.*, 2008; Rehfeldt and Jaquish, 2010; Loss *et al.*, 2011; Pedlar *et al.*, 2012). There is *low agreement* whether it is better to increase the resilience to climate change of ecosystems as they currently occur, or to enhance the capacity of ecosystems to transform in the face of climate change (Richardson *et al.*, 2009).

There is *high agreement* 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 ecosystem-wide approach than assisted migration for individual species. However, observational and modelling 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; Vitt *et al.*, 2009; Willis and Bhagwat, 2009; Loss *et al.*, 2011; Hewitt *et al.*, 2011). The velocity of 21st century climate change and substantial habitat fragmentation in large parts of the world means that many species will be unable to migrate or adapt fast enough to keep pace with climate change (Figure 4-5), posing 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 require assisted migration could easily overwhelm funding capacity (Minteer and Collins, 2010). Decisions regarding which species should be translocated are complex and debatable, given variability among and within species and the ethical issues involved (Aubin *et al.*, 2011; Winder *et al.*, 2011b).

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 with global scope (e.g., Millennium Seed Bank, Svalbard vault, Frozen Ark, Global Genome Initiative and others; Leishman *et al.*, 2007; Lermen *et al.*, 2009; Rawson *et al.*, 2011). Knowledge of which genetic variants within a species have more potential for adaptation to climate change could help prioritise the material stored (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 collecting relatively small samples from restricted locations, and then later by the selection pressures inadvertently applied during ex-situ maintenance are unknown. Despite some documented successes, it remains uncertain whether it is always possible to reintroduce species successfully into the wild after generations of *ex-situ* conservation.

4.4.3. Consequences and Costs of Inaction and Benefits of Action

Failure to reduce the magnitude or rate of climate change will plausibly lead to changes (often decreases) in the value of ecosystem services provided, or incur costs in order to maintain or restore the services or adapt to their decline. There are several sources of such costs: administration and assessment, implementation, and opportunity costs, including financial cost. Owing to the number of assumptions made, knowledge gaps and recognized uncertainties, such result should be employed with caution. A systematic review of costs related to ecosystems and climate change by Rodriguez-Labajos (2013), shows that the monetary and non-monetary costs are distributed across all ecosystem service categories. Rodriguez-Labajos (2013) also discusses the potential and limits of monetary cost calculations, and issues of timing, trade-offs and the unequal distribution of costs.

A comprehensive monetary estimate of the effects of climate change on ecosystem service provision is not available. The Millennium Ecosystem Assessment (2005c; 2005d; 2005e) included climate change among the direct drivers of ecosystems change and devoted a chapter to the necessary responses. Building on results of the IPCC, the MA offered some estimated costs of action: complying with the Kyoto protocol for industrial countries would range between 0.2 and 2% of GDP; a modest stabilization target of 450 ppm CO_2 in the atmosphere over the 21st century would range from 0.02 to 0.1% of global-average GDP per year. TEEB (2009) underlined priorities in the ecosystem service-climate change coupling (reduction targets in relation to coral reefs, forest carbon markets and accounting, and ecosystem investment for mitigation), without going in depth into analysis of the cost types involved. The Cost of Policy Inaction (COPI) Project (ten Brink *et al.*, 2008) estimated the monetary costs of not meeting the 2010 biodiversity goals. Their model incorporates climate change, among other pressures, through an impaired quality of land, in terms of species abundance in diverse land-use categories. They conclude that the cumulative losses of welfare due to land-use changes, in terms of loss of ecosystem services, could reach an annual amount of EUR 14 trillion (based on values of 2007) in 2050, which may be equivalent to 7% of projected global GDP for that year. Eliasch (2008) estimates the damage costs to forests as reaching USD 1 trillion a year by 2100. The study used the probabilistic model employed by Stern (2006), which did not value effects on biodiversity or water-related ES.

The studies to date agree on the following points. First, climate change has already caused a reduction in ecosystem services which will become more severe as climate change continues. Second, ecosystem-based strategies to mitigate climate change are cost-effective, although more difficult to implement (i.e., more costly) in intensively managed ecosystems such as farming lands. Third, accurately estimating the monetary costs of reduction in ecosystem services that are not marketed is difficult. The provision of monetized costs tends to sideline the non-monetized political, social, and environmental costs relevant for decision making. Finally, there is a large funding gap between the cost of actions necessary to protect ecosystem services against climate change and the actual resources available.

In addition to direct costs, further costs may result from trade-offs between services: e.g. afforestation for climate mitigation and urban greening for climate adaptation may be costly in terms of water provision (Chisholm, 2010; Jenerette *et al.*, 2011; Pataki *et al.*, 2011). 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 typically lower per hectare yields, may require a larger area to be dedicated to cropland. Leaving aside the contested (Searchinger *et al.*, 2008; Plevin *et al.*, 2010) effectiveness of biofuels as a mitigation strategy, there is evidence of their disruptive effect on food security, land tenure, labour rights and biodiversity in several parts of the world (Obersteiner *et al.*, 2010; Tirado *et al.*, 2010).

4.4.4. Unintended Consequences of Adaptation and Mitigation

Actions taken within the terrestrial and freshwater system domain or in other sectors to mitigate or adapt to climate change can have unintended consequences. Some issues relevant to this section are also found in Section 14.7 and the Working Group III report.

Several of the alternatives to fossil fuel require extensive use of the land surface and thus have a direct impact on terrestrial ecosystems and an indirect impact on inland water systems (Paterson *et al.*, 2008; Turner *et al.*, 2010). As an illustration, the RPC2.6 scenario involves both bioenergy and renewables as major components of the energy mix (Box 4-1; van Vuuren *et al.*, 2011).

Policy shifts in developed countries favour the expansion of large-scale bioenergy production, which places new pressures on terrestrial and freshwater ecosystems (Searchinger *et al.*, 2008; Lapola *et al.*, 2010), either through direct use of land or water or indirectly by displacing food crops, which must then be grown elsewhere. Over the past decade there has been a global trend to reduced rates of forest loss; it is unclear if this will continue in the face of simultaneously rising food and biofuel demand (Wise *et al.*, 2009; Meyfroidt and Lambin, 2011). The EU Renewable Energy Sources Directive is estimated to have only a moderate influence on European forests provided that the price paid by the bioenergy producers remained below 50-60 US$ per m^3 wood (Moiseyev *et al.*, 2011). However, a doubled growth rate for bioenergy until 2030 would have major consequences for the global forest sector, including a reduction of forest stocks in Asia of 2 % to 4% (Martin *et al.*, 2010). By 2100 in RCP2.6, bioenergy crops are projected to occupy approximately 4 million km^2, approximately 7% of global cultivated land projected at the time. Modification of the landscape and the fragmentation of habitats are major influences on extinction risks (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 contribute to extinction risks. Some bioenergy crops may be invasive species (Raghu *et al.*, 2006).

Abandoned former agricultural land could be used for biomass production (McAlpine *et al.*, 2009). However, such habitats may be core elements in cultural landscapes of high conservation value, with European species-rich grasslands often developed from abandoned croplands (Hejcman *et al.*, 2013).

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). Reservoirs can be a sink of CO_2 but also a source of biogenic CO_2 and CH_4 – this issue is discussed in WG III AR5 Section 7.8.1.

Wind turbines can kill birds and bats (e.g., Barclay *et al.*, 2007), and inappropriately-sited wind farms can negatively impact on bird populations (Drewitt and Langston, 2006). Effects can be reduced by careful siting of turbines, for example by avoiding migration routes (Drewitt and Langston, 2006). Estimating mortality rates is complex and difficult (Smallwood, 2007) but techniques are being developed in order to inform siting decisions and impact assessments (Péron *et al.*, 2013). Wind farms in Europe and the USA are estimated to cause between 0.3 and 0.4 wildlife fatalities *per* gigawatt-hour (GWh) of electricity, compared to approximately 5.2 fatalities *per* GWh nuclear and fossil-fuel power stations (Sovacool, 2009; but see Willis *et al.*, 2010c). 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).

Large-scale solar farms could impact local biodiversity if poorly sited, but the impact can be reduced with appropriate planning (Tsoutsos *et al.*, 2005). Solar PV installations can decrease local surface albedo, giving a small positive radiative forcing. There are some plausible local circumstances in which this may be a consideration, but in general the climate effect is estimated to be 30times smaller than the avoided radiative forcing arising from substituting fossil fuels with PV (Nemet, 2009).

Relocation or expansion of agricultural areas and settlements as climate change adaptation measures could pose risks of habitat fragmentation and loss similar to those discussed above in the context of mitigation through bio-energy. Assisted migration (see Section 4.4.2.4) may directly conflict with other conservation priorities, for example by facilitating the introduction of invasive species (Maclachlan *et al.*, 2007).

4.5. Emerging Issues and Key Uncertainties

Detecting the presence and location of thresholds in ecosystem response to climate change, and specifically the type of thresholds characterized as tipping points, remains a major source of uncertainty with high potential consequences. In general (Field *et al.*, 2007), negative feedbacks currently dominate the climate-ecosystem interaction. For most ecological processes, increasing magnitude of warming shifts the balance towards positive rather than negative feedbacks (Field *et al.*, 2007). In several regions, such as the boreal ecosystems, positive feedbacks may become dominant, under moderate warming). 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 interactions 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 terrestrial and freshwater ecosystems to climate and other perturbations, particularly at the local scale, 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 reduced by better systems understanding, but some will remain irreducible because of the failure of predictive models when faced with certain types of complexity (such as those which lead to 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 as yet mostly 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.

The consequences for species interactions of differing phenological or movement-based responses to climate change are insufficiently known and may make projections based on individual species models unreliable.

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 future sink strength of the biosphere, since they are of similar magnitude but opposite sign – are needed as a supplement to the single-factor experiments. For example, uncertainty on the magnitude of CO_2 fertilization is key for forest responses to climate change, particularly in tropical forests, woodlands and savannas (Cox *et al.*, 2013; Huntingford *et al.*, 2013).

The effects of changes in the frequency or intensity of climate-related extreme events, such as floods, cyclones , heat waves, exceptionally large fires on ecosystem change are probably equal to or greater than shifts in the mean values of climate variables. These effects are insufficiently studied, and in particular, are seldom adequately represented in Earth system models.

Understanding of the *rate* of climate change that can be tracked or adapted to by organisms is as important as understanding the *magnitude* of change they can tolerate. 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 capacity for, and limits to, ecological and evolutionary adaptive processes are only known in a few cases. The development and testing of human-assisted adaptation strategies for their cost-effectiveness in reducing risk is a prerequisite for their widespread adoption.

The costs of the loss of biodiversity and ecosystem services as a result of climate change is known for only a few cases, or is associated with large uncertainties; as are the costs and benefits of assisting ecosystems and species to adapt to climate change.

Frequently Asked Questions

FAQ 4.1: How do land-use and land-cover changes cause changes in climate? *[to be placed in Section 4.2.4.1]*
Land use change affects the local as well as the global climate. Different forms of land cover and land use can cause warming or cooling and changes in rainfall, depending on where they occur in the world, what the preceding land cover was, and how the land is now managed. Vegetation cover, species composition and land management practices (such as harvesting, burning, fertilizing, grazing or cultivation) influence the emission or absorption of greenhouse gases. The brightness of the land cover affects the fraction of solar radiation that is reflected back into the sky, instead of being absorbed, thus warming the air immediately above the surface. Vegetation and land use patterns also influence water use and evapotranspiration, which alter local climate conditions. Effective land-use strategies can also help to mitigate climate change.

FAQ 4.2: What are the non-greenhouse gas effects of rising carbon dioxide on ecosystems?
[to be placed in Section 4.2.4.4]
Carbon dioxide (CO_2) is an essential building block of the process of photosynthesis. Simply put, plants use sunlight and water to convert CO_2 into energy. Higher CO_2 concentrations enhance photosynthesis and growth (up to a point), and reduce the water used by the plant. This means that water remains longer in the soil or recharges rivers and aquifers. These effects are mostly beneficial; however, high CO_2 also has negative effects, in addition to causing global warming. High CO_2 levels cause the nitrogen content of forest vegetation to decline and can increase their chemical defences, reducing their quality as a source of food for plant-eating animals. Furthermore, rising CO_2 causes ocean waters to become acidic (see FAQ 6.3), and can stimulate more intense algal blooms in lakes and reservoirs.

FAQ 4.3: Will the number of invasive alien species increase due to climate change?
[to be placed in Section 4.2.4.6]
Some invasive plants and insects have already been shown to benefit from climate change and will establish and spread into new regions (where they are ‘aliens’), once they are introduced. The number of newly-arrived species

and the abundance of some already-established alien species will increase because climate change will improve conditions for them. At the same time, increasing movement of people and goods in the modern world, combined with land use changes worldwide, increases the likelihood that alien species are accidentally transported to new locations and become established there. There are many actions which can be taken to reduce, but not eliminate, the risk of alien species invasions, such as the treatment of ballast water in cargo ships and wood products, strict quarantine applied to crop and horticultural products, and embargos on the trade and deliberate introduction of known invader species. Some invasive species will suffer from climate change and are expected to decrease in range and population size in some regions. Generally, increased establishment success and spread will be most visible for those alien species that have characteristics favoured by the changing climate, such as those that are drought tolerant or able to take advantage of higher temperatures.

FAQ 4.4: How does climate change contribute to species extinction? *[to be placed in Section 4.3.2.5]*

There is a consensus that climate change over the coming century will increase the risk of extinction for many species. When a species becomes extinct, a unique and irreplaceable life form is lost. Even local extinctions can impair the healthy functioning of ecosystems.

Under the fastest rates and largest amounts of projected climate change, many species will be unable to move fast enough to track suitable environments, which will greatly reduce their chances of survival. Under the lowest projected rates and amounts of climate change, and with the assistance of effective conservation actions, the large majority of species *will* be able to adapt to new climates, or move to places that improve their chances of survival. Loss of habitat and the presence of barriers to species movement increase the risk of extinctions as a result of climate change.

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 recent extinctions is the subject of considerable debate.

FAQ 4.5: Why does it matter if ecosystems are altered by climate change? *[to be placed in Section 4.3.4]*

Ecosystems provide essential services for all life; food, life-supporting atmospheric conditions, drinkable water, as well as raw materials for basic human needs like clothing and housing. Ecosystems play a critical role in limiting the spread of human and non-human diseases. They have a strong impact on the weather and climate itself, which in turn impacts agriculture, food supplies, socio-economic conditions, floods and physical infrastructure. When ecosystems change, their capacity to supply these services changes as well; for better or worse. Human wellbeing is put at risk, along with the welfare of millions of other species. People have a strong emotional, spiritual and ethical attachment to the ecosystems they know, and the species they contain.

By “ecosystem change”, we mean changes in some or all of the following: the number and types of organisms present; the ecosystem's physical appearance (e.g., tall or short, open or dense vegetation); the functioning of the system and all its interactive parts, including the cycling of nutrients and productivity. Though in the long-term not all ecosystem changes are detrimental to all people or to all species, the faster and further ecosystems change in response to new climatic conditions, the more challenging it is for humans and other species to adapt to the new conditions.

FAQ 4.6: Can ecosystems be managed to help them and people to adapt to climate change?
[to be placed in Section 4.4.2.3]

The ability of human societies adapt to climate change will depend, in large measure, upon the management of terrestrial and inland freshwater ecosystems. A fifth of global human-caused carbon emissions today are absorbed by terrestrial ecosystems; this important carbon *sink* operates largely without human intervention, but could be increased through a concerted effort to reduce forest loss and to restore damaged ecosystems, which also co-benefits the conservation of biodiversity.

The clearing and degradation of forests and peatlands represents a *source* of carbon emissions to the atmosphere which can be reduced though management; for instance, there has been a three-quarters decline in the rate of deforestation in the Brazilian Amazon in the last two decades. Adaptation is also helped through more proactive detection and management of wildfire and pest outbreaks, reduced drainage of peatlands, the creation of species migration corridors and assisted migration.

FAQ 4.7: What are the economic costs of changes in ecosystems due to climate change?

[to be placed in Section 4.4.3]

Climate change will certainly alter the services provided by most ecosystems, and for high degrees of change, the overall impacts are most likely to be negative. In standard economics, the value of services provided by ecosystems are known as externalities, which are usually outside the market price system, difficult to evaluate and often ignored.

A good example is the pollination of plants by bees and birds and other species, a service which may be negatively affected by climate change. Pollination is critical for the food supply as well as for overall environmental health. Its value has been estimated globally at $350 billion for the year 2010 (The range of estimates is 200 – 500 $ billion).

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 adaptation (EBA) integrates the use of biodiversity and ecosystem services into climate change adaptation strategies (e.g., CBD, 2009; Munroe et al., 2011; see Chapters 3, 4, 5, 8, 9, 13, 14, 15, 16, 19, 22, 24, 25, and 27). EBA is implemented through the sustainable management of natural resources and conservation and restoration of ecosystems, to provide and sustain services that facilitate adaptation both to climate variability and change (Colls et al., 2009). It also sets out to take into account the multiple social, economic, and cultural co-benefits for local communities (CBD COP 10 Decision X/33).

EBA can be combined with, or even a substitute for, the use of engineered infrastructure or other technological approaches. Engineered defenses such as dams, sea walls and levees adversely affect biodiversity, potentially 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). EBA offers lower risk of maladaptation than engineering solutions in that their application is more flexible and responsive to unanticipated environmental changes. Well-integrated EBA can be more cost effective and sustainable than non-integrated physical engineering approaches (Jones et al., 2012), 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. In addition, EBA yields economic, social, and environmental co-benefits in the form of ecosystem goods and services (World Bank, 2009).

EBA is applicable in both developed and developing countries. In developing countries where economies depend more directly on the provision of ecosystem services (Vignola et al., 2009), EBA may be a highly useful approach to reduce risks to climate change impacts and ensure that development proceeds on a pathways that are resilient to climate change (Munang et al., 2013). EBA projects may be 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., Day et al., 2007; 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 (Jonkman et al., 2013)
- Sustainable management of grasslands and rangelands to enhance pastoral livelihoods 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.

Application of EBA, like other approaches, is not without risk, and risk/benefit assessments will allow better assessment of opportunities offered by the approach. The examples of EBA are too few and too recent to assess either the risks or the benefits comprehensively at this stage. EBA is still a developing concept but is should be considered alongside adaptation options based more on engineering works or social change, and existing and new cases used to build understanding of when and where its use is appropriate.

[INSERT FIGURE EA-1 HERE
Figure EA-1: Adapted from Munang et al. (2013). Ecosystem based adaptation (EBA) uses the capacity of nature to buffer human systems from the adverse impacts of climate change. Without EBA, climate change may cause degradation of ecological processes (central white panel) leading to losses in human well-being. Implementing EBA (outer blue panel) may reduce or offset these adverse impacts resulting in a virtuous cycle that reduces climate-related risks to human communities, and may provide mitigation benefits.]

Box CC-EA References

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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 (*medium confidence*) (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 around the year 2000 (Figure RF-1; Döll and Zhang, 2010). For one climate scenario (SRES A2 emissions, HadCM3 climate model), 15% of the global land area may be negatively affected, by the 2050s, by 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 such 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, as occurs in Sweden, the annual hydrograph becomes more similar to variation in electricity demand, i.e. with a lower spring flood and increased runoff during winter months (Renofalt *et al.*, 2010).

Because biota are often adapted to a certain level of river flow variability, the projected 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 Table 3-2 in Chapter 3).

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 (Section 3.2.3 in Chapter 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).

[INSERT FIGURE RF-1 HERE

Figure RF-1: Impact of climate change relative to the impact of water withdrawals and dams on natural flows for two ecologically relevant river flow characteristics (mean annual river flow and monthly low flow Q_{90}), computed by a global water model (Döll and Zhang, 2010). Monthly Q_{90} was defined as the flow that is exceeded in 9 out 10 months. 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. Please note that the figure does not reflect spatial differences in the magnitude of change.]

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.

[INSERT FIGURE RF-2 HERE
Figure RF-2: Accumulated loss of regional species richness (gamma diversity) of macroinvertebrates as a function of glacial cover in catchment. Obligate glacial river macroinvertebrates begin to disappear from assemblages when glacial cover in the catchment drops below approximately 50%, and 9-14 species are predicted to be lost with the complete disappearance of glaciers in each region, corresponding to 11, 16 and 38% of the total species richness in the three study regions in Ecuador, Europe and Alaska. Data are derived from multiple river sites from the Ecuadorian Andes and Swiss and Italian Alps, and a temporal study of a river in the Coastal Range Mountains of southeast Alaska over nearly three decades of glacial shrinkage. Each data point represents a river site or date (Alaska), and lines are Lowess fits. Adapted by permission from Macmillan Publishers Ltd: *Nature Climate Change,* Jacobsen *et al.*, 2012, © 2012.]

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Box CC-VW. Active Role of Vegetation in Altering Water Flows under Climate Change

[Dieter Gerten (Germany), Richard Betts (UK), Petra Döll (Germany)]

Climate, vegetation and carbon and water cycles are intimately coupled, in particular via the simultaneous transpiration and CO_2 uptake through plant stomata in the process of photosynthesis. Hence, water flows such as runoff and evapotranspiration are affected not only directly by anthropogenic climate change as such (i.e. by changes in climate variables such as temperature and precipitation), but also indirectly by plant responses to increased atmospheric CO_2 concentrations. In addition, effects of climate change (e.g. higher temperature or altered precipitation) on vegetation structure, biomass production and plant distribution have an indirect influence on water flows. Rising CO_2 concentration affects vegetation and associated water flows in two contrasting ways, as suggested by ample evidence from Free Air CO_2 Enrichment (FACE), laboratory and modelling experiments (e.g. Leakey *et al.*, 2009; de Boer *et al.*, 2011; Reddy *et al.*, 2010). On the one hand, a *physiological* effect leads to reduced opening of stomatal apertures, which is associated with lower water flow through the stomata, i.e. lower leaf-level transpiration. On the other hand, a *structural* effect (“fertilization effect”) stimulates photosynthesis and biomass production of C3 plants including all tree species, which eventually leads to higher transpiration at regional scales. A key question is to what extent the climate- and CO_2-induced changes in vegetation and transpiration translate into changes in regional and global runoff.

The physiological effect of CO_2 is associated with an increased intrinsic water use efficiency (WUE) of plants, which means that less water is transpired per unit of carbon assimilated. Records of stable carbon isotopes in woody plants (Peñuelas *et al.*, 2011) verify this finding, suggesting an increase in WUE of mature trees by 20.5% between the early 1960s and the early 2000s. Increases since pre-industrial times have also been found for several forest sites (Andreu-Hayles *et al.*, 2011; Gagen *et al.*, 2011; Loader *et al.*, 2011; Nock *et al.*, 2011) and in a temperate semi-natural grassland (Koehler *et al.*, 2010), although in one boreal tree species WUE ceased to increase after 1970 (Gagen *et al.*, 2011). Analysis of long-term whole-ecosystem carbon and water flux measurements from 21 sites in North American temperate and boreal forests corroborates a notable increase in WUE over the two past decades (Keenan *et al.*, 2013). An increase in global WUE over the past century is supported by ecosystem model results (Ito and Inatomi, 2012).

A key influence on the significance of increased WUE for large-scale transpiration is whether vegetation structure and production has remained approximately constant (as assumed in the global modelling study by Gedney *et al.*, 2006) or has increased in some regions due to the structural CO_2 effect (as assumed in models by Piao *et al.*, 2007;

Gerten *et al.*, 2008). While field-based results vary considerably among sites, tree ring studies suggest that tree growth did not increase globally since the 1970s in response to climate and CO_2 change (Peñuelas *et al.*, 2011; Andreu-Hayles *et al.*, 2011). However, basal area measurements at over 150 plots across the tropics suggest that biomass and growth rates in intact tropical forests have increased in recent decades (Lewis *et al.*, 2009). This is also confirmed for 55 temperate forest plots, with a suspected contribution of CO_2 effects (McMahon *et al.*, 2010). Satellite observations analysed in Donohue *et al.* (2013) suggest that an increase in vegetation cover by 11% in warm drylands (1982–2010 period) is attributable to CO_2 fertilization. Owing to the interplay of physiological and structural effects, the net impact of CO_2 increase on global-scale transpiration and runoff remains rather poorly constrained. This is also true because nutrient limitation, often omitted in modelling studies, can suppress the CO_2 fertilization effect (see Rosenthal and Tomeo, 2013).

Therefore, 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 the physiological CO_2 effect (lower transpiration) contributed to a supposed increase in global runoff seen in reconstructions by Labat *et al.* (2004). However, a more recent analysis based on a more complete dataset (Dai *et al.*, 2009) suggested that river basins with decreasing runoff outnumber basins with increasing runoff, such that a small decline in global runoff is *likely* for the period 1948–2004. Hence, detection of vegetation contributions to changes in water flows critically depends on the availability and quality of hydrometeorological observations (Haddeland *et al.*, 2011; Lorenz and Kunstmann, 2012). Overall, the evidence since AR4 suggests that climatic variations and trends have been the main driver of global runoff change in the past decades; both CO_2 increase and land use change have contributed less (Piao *et al.*, 2007; Gerten *et al.*, 2008; Alkama *et al.*, 2011; Sterling *et al.*, 2013). Oliveira *et al.* (2011) furthermore pointed to the importance of changes in incident solar radiation and the mediating role of vegetation; according to their global simulations, a higher diffuse radiation fraction during 1960–1990 may have increased evapotranspiration in the tropics by 3% due to higher photosynthesis from shaded leaves.

It is uncertain how vegetation responses to future increases in CO_2 and to climate change will modulate the impacts of climate change on freshwater flows. 21st century continental- and basin-scale runoff is projected by some models to either increase more or decrease less when the physiological CO_2 effect is included in addition to climate change effects (Betts *et al.*, 2007; Murray *et al.*, 2012). This could somewhat ease the increase in water scarcity anticipated in response to future climate change and population growth (Gerten *et al.*, 2011; Wiltshire *et al.*, in press). In absolute terms, the isolated effect of CO_2 has been modelled to increase future global runoff by 4–5% (Gerten *et al.*, 2008) up to 13% (Nugent and Matthews, 2012) compared to the present, depending on the assumed CO_2 trajectory and whether feedbacks of changes in vegetation structure and distribution to the atmosphere are accounted for (they were not in Nugent and Matthews, 2012). In a global model intercomparison study (Davie *et al.*, in press), two out of four models projected stronger increases and, respectively, weaker decreases in runoff when considering CO_2 effects compared to simulations with constant CO_2 concentration (consistent with above findings, though magnitudes differed between the models), but two other models showed the reverse. Thus, the choice of models and the way they represent the coupling between CO_2, stomatal closure and plant growth is a source of uncertainty, as also suggested by Cao *et al.* (2009). Lower transpiration due to rising CO_2 concentration may also affect future regional climate change itself (Boucher *et al.*, 2009) and enhance the contrast between land and ocean surface warming (Joshi *et al.*, 2008). Overall, although physiological and structural effects will influence water flows in many regions, precipitation and temperature effects are *likely* to remain the prime influence on global runoff (Alkama *et al.*, 2010).

An application of a soil-vegetation-atmosphere-transfer model indicates complex responses of groundwater recharge to vegetation-mediated changes in climate, with computed groundwater recharge being always larger than would be expected from just accounting for changes in rainfall (McCallum *et al.*, 2010). Another study found that even if precipitation slightly decreased, groundwater recharge might increase as a net effect of vegetation responses to climate change and CO_2 rise, i.e. increasing WUE and either increasing or decreasing leaf area (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 site 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).

Using a large ensemble of climate change projections, Konzmann *et al.* (2013) put hydrological changes into an agricultural perspective and suggested that the net result of physiological and structural CO_2 effects on crop irrigation requirements would be a global reduction (Figure VW-1). Thus, adverse climate change impacts on irrigation requirements and crop yields might be partly buffered as WUE and crop production improve (Fader *et al.*, 2010). However, substantial CO_2-driven improvements will only be realized if proper management abates limitation of plant growth by nutrient availability or other factors.

[INSERT FIGURE VW-1 HERE
Figure VW-1: Percentage change in net irrigation requirements of 11 major crops from 1971–2000 to 2070–2099 on areas currently equipped for irrigation, assuming current management practices. Top: impact of climate change including physiological and structural crop responses to increased atmospheric CO_2 concentration (maximum effect in the absence of co-limitation by nutrients). Bottom: impact of climate change only. Shown is the median change derived from climate change projections by 19 GCMs (based on the SRES A2 emissions scenario) used to force a vegetation and hydrology model. Modified after Konzmann *et al.* (2013).]

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 atmosphere such as in the Amazon region (Port *et al.*, 2012; Saatchi *et al.*, 2013). One model in the study by Davie *et al.* (in press) showed regionally diverse climate change effects on vegetation distribution and structure, which had a much weaker effect on global runoff than the structural and physiological CO_2 effects. As water, carbon and vegetation dynamics evolve synchronously and interactively under climate change (Heyder *et al.*, 2011; Gerten *et al.*, in press), it remains a challenge to disentangle the individual effects of climate, CO_2 and land cover change on the water cycle.

Box CC-VW References

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Table 4-1: Biome shifts of the 20th century 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). The biome abbreviations match those in Figure 4-1. 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.

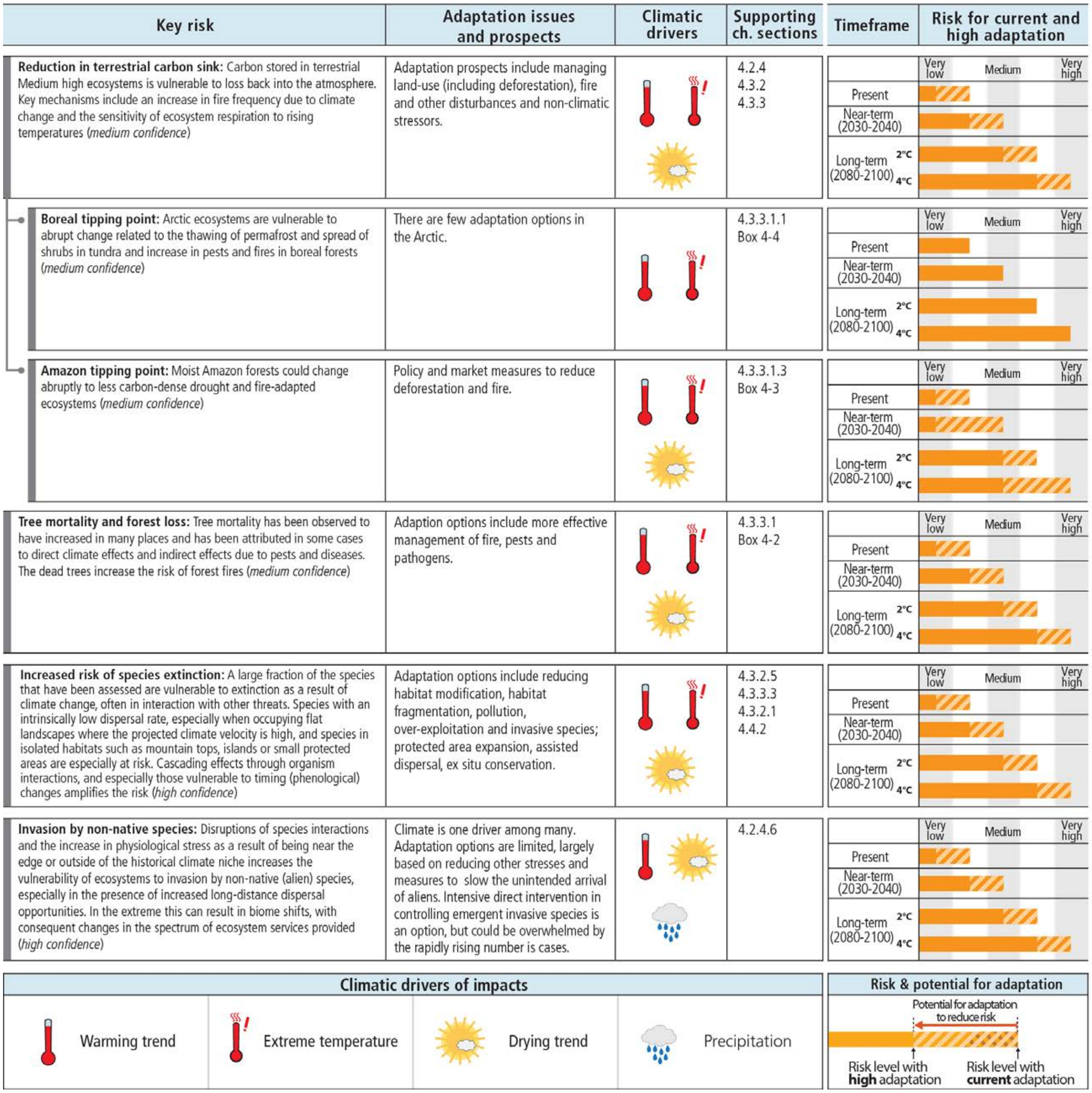
	Location	Reference	Plots	Time Period	Shift type	Retracting biome	Expanding biome	Temp. change (°C century^{-1})	Precip. change (% century^{-1})
1.	Alaska Range, Alaska, USA	Lloyd and Fastie, 2003	18	1800-2000	L	UA	BC	1.1*	3
2.	Baltic Coast, Sweden	Walther *et al.*, 2005	7	1944-2003	L	TC	TB	0.6*	8
3.	Becca di Viou, Italy	Leonelli *et al.*, 2011	1	1700-2008	E	UA	BC	0.9*	-6
4.	Garibaldi, British Columbia, Canada	Brink, 1959	1	1860-1959	E	UA	BC	0.7*	16*
5.	Goulet Sector, Québec, Canada	Payette and Filion, 1985	2	1880-1980	E	UA	BC	1.4*	19*
6.	Green Mountains, Vermont, USA	Beckage *et al.*, 2008	33	1962-2005	E	BC	TB	1.6*	6
7.	Jasper, Alberta, Canada	Luckman and Kavanagh, 2000	1	1700-1994	E	UA	BC	0.6	21*
8.	Kenai Mountains, Alaska, USA	Dial *et al.*, 2007	3	1951-1996	E	UA	BC	0.7	6
9.	Kluane Range, Yukon, Canada	Danby and Hik, 2007	2	1800-2000	E	UA	BC	0.7	5
10.	Low Peninsula, Québec, Canada	Payette and Filion, 1985	1	1750-1980	N	-	-	1.4*	19*

	Location	Reference	Plots	Time Period	Shift type	Retracting biome	Expanding biome	Temp. change (°C century^{-1})	Precip. change (% century^{-1})
11.	Mackenzie Mountains, Northwest Territories, Canada	Szeicz and Macdonald, 1995	13	1700-1990	N	-	-	1.4*	3
12.	Montseny Mountains, Catalonia, Spain	Peñuelas and Boada, 2003	50	1945-2001	E	UA	TB	1.2*	-3
13.	Napaktok Bay, Labrador, Canada	Payette, 2007	2	1750-2000	L	UA	BC	1.1*	5
14.	Noatak, Alaska, USA	Suarez *et al.*, 1999	18	1700-1990	L	UA	BC	0.6	19*
15.	Putorana Mountains, Russia	Kirdyanov *et al.*, 2012	10	1500-2000	E	UA	BC	0.3	10
16.	Rahu Saddle, New Zealand	Cullen *et al.*, 2001	7	1700-2000	N	-	-	0.6*	3
17.	Rai-Iz, Urals, Russia	Devi *et al.*, 2008	144	1700-2002	E	UA	BC	0.3	35*
18.	Sahel, Sudan, Guinea zones; Senegal	Gonzalez, 2001	135	1945-1993	L	RW	RG	0.4*	-48*
19.	Sahel, Burkina Faso, Chad, Mali, Mauritania, Niger	Gonzalez *et al.*, 2012	14	1960-2000	L	RW	RG	-0.01*- to 0.8*	-31*-to 9
20.	Scandes, Sweden	Kullman and Öberg, 2009	123	1915-2007	E	UA	BC	0.8*	25*
21.	Sierra Nevada, California, USA	Millar *et al.*, 2004	10	1880-2002	E	UA	TC	-0.1	21*
22.	South Island, New Zealand	Wardle and Coleman, 1992	22	1980-1990	E	TS	TB	0.6*	3
23.	Yambarran, Northern Territory, Australia	Sharp and Bowman, 2004	33	1948-2000	N	-	-	-0.06	35*

Table 4-2: Summary of drivers and outcomes of LUCC scenarios associated with Representative Concentration Pathways (Hurtt *et al.*, 2011). RCPs are identified with the radiative forcing by 2100 (8.5, 6.0, 4.5 and 2.6 Wm^{-2}) and by the name of the model used to generate the associated land use/cover scenarios (MESSAGE, AIM, GCAM and IMAGE; see Hurtt *et al.* (2011) for further details).

RCP	Model and references	Key assumptions / drivers	Land use / cover outcomes
8.5	MESSAGE, Riahi et al. (2007)	No climate change mitigation actions; radiative forcing still rising at 2100 Strong increase in agricultural resource use driven by the increasing population (rises to 12 billion people by 2100) Yield improvements and intensification assumed to account for most of production increases	Increase in cultivated land by about 305 million hafrom 2000 to 2100 Forest cover declines by 450 million ha from 2000 to 2100 Arable land use in developed countries slightly decreased - all of the net increases occur in developing countries.
6.0	AIM, Fujino *et al.* (2006); Hijioka *et al.* (2008)	Mitigation actions taken late in the century to stabilize radiative forcing at 6 Wm^{-2} after 2100 Population growth and economic growth Increasing food demand drives cropland expansion	Urban land-use increases Cropland area expands Grassland area declines Total forested area extent remains constant
4.5	GCAM, Smith and Wigley (2006); Wise *et al.* (2009)	Mitigation stabilizes radiative forcing at 4.5 Wm^{-2} before 2100 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 Food demand met through crop yield improvements, dietary shifts, production efficiency and international trade.	Preservation of large stocks of terrestrial carbon in forests Overall expansion in forested area Agricultural land declines slightly due to afforestation,
2.6	IMAGE Van Vuuren *et al.*, (2007); van Vuuren *et al.* (2006)	Overall trends in land use and land cover mainly determined by demand, trade and production of agricultural products and bio-energy. Expansion of croplands largely due to bioenergy production. Production of animal products met through shift from extensive to more intensive animal husbandry	Much agriculture relocates from high income to low-income regions Increase in bio-energy production, new area for bioenergy crops near current agricultural areas. Pasture largely constant

Table 4-3: Key risks for terrestrial and freshwater ecosystems from climate change and the potential for reducing risk through mitigation and adaptation. Key risks are identified based on assessment of the literature and expert judgments by chapter authors, with evaluation of evidence and agreement in supporting chapter sections. Each key risk is characterized as very low to very high. Risk levels are presented in three timeframes: the present, near-term (here, assessed over 2030-2040), and longer-term (here, assessed over 2080-2100). For the near-term era of committed climate change, projected levels of global mean temperature increase do not diverge substantially across emission scenarios. For the longer-term era of climate options, risk levels are presented for global mean temperature increase of 2°C and 4°C above preindustrial levels. For each timeframe, risk levels are estimated for a continuation of current adaptation and for a hypothetical highly adapted state. Relevant climate variables are indicated by icons. For a given key risk, change in risk level through time and across magnitudes of climate change is illustrated, but because the assessment considers potential impacts on different physical, biological, and human systems, risk levels should not necessarily be used to evaluate relative risk across key risks, sectors, or regions.



Key risk	Adaptation issues and prospects	Climatic drivers	Supporting ch. sections	Timeframe	Risk for current and high adaptation
Reduction in terrestrial carbon sink: Carbon stored in terrestrial Medium high ecosystems is vulnerable to loss back into the atmosphere. Key mechanisms include an increase in fire frequency due to climate change and the sensitivity of ecosystem respiration to rising temperatures (*medium confidence*)	Adaptation prospects include managing land-use (including deforestation), fire and other disturbances and non-climatic stressors.	🌡 🌡! ☀	4.2.4 4.3.2 4.3.3	Present Near-term (2030-2040) Long-term (2080-2100) 2°C 4°C	Very low Medium Very high
Boreal tipping point: Arctic ecosystems are vulnerable to abrupt change related to the thawing of permafrost and spread of shrubs in tundra and increase in pests and fires in boreal forests (*medium confidence*)	There are few adaptation options in the Arctic.	🌡 🌡!	4.3.3.1.1 Box 4-4	Present Near-term (2030-2040) Long-term (2080-2100) 2°C 4°C	Very low Medium Very high
Amazon tipping point: Moist Amazon forests could change abruptly to less carbon-dense drought and fire-adapted ecosystems (*medium confidence*)	Policy and market measures to reduce deforestation and fire.	🌡 🌡! ☀	4.3.3.1.3 Box 4-3	Present Near-term (2030-2040) Long-term (2080-2100) 2°C 4°C	Very low Medium Very high
Tree mortality and forest loss: Tree mortality has been observed to have increased in many places and has been attributed in some cases to direct climate effects and indirect effects due to pests and diseases. The dead trees increase the risk of forest fires (*medium confidence*)	Adaption options include more effective management of fire, pests and pathogens.	🌡 🌡! ☀	4.3.3.1 Box 4-2	Present Near-term (2030-2040) Long-term (2080-2100) 2°C 4°C	Very low Medium Very high
Increased risk of species extinction: A large fraction of the species that have been assessed are vulnerable to extinction as a result of climate change, often in interaction with other threats. Species with an intrinsically low dispersal rate, especially when occupying flat landscapes where the projected climate velocity is high, and species in isolated habitats such as mountain tops, islands or small protected areas are especially at risk. Cascading effects through organism interactions, and especially those vulnerable to timing (phenological) changes amplifies the risk (*high confidence*)	Adaptation options include reducing habitat modification, habitat fragmentation, pollution, over-exploitation and invasive species; protected area expansion, assisted dispersal, ex situ conservation.	🌡 🌡! ☀	4.3.2.5 4.3.3.3 4.3.2.1 4.4.2	Present Near-term (2030-2040) Long-term (2080-2100) 2°C 4°C	Very low Medium Very high
Invasion by non-native species: Disruptions of species interactions and the increase in physiological stress as a result of being near the edge or outside of the historical climate niche increases the vulnerability of ecosystems to invasion by non-native (alien) species, especially in the presence of increased long-distance dispersal opportunities. In the extreme this can result in biome shifts, with consequent changes in the spectrum of ecosystem services provided (*high confidence*)	Climate is one driver among many. Adaptation options are limited, largely based on reducing other stresses and measures to slow the unintended arrival of aliens. Intensive direct intervention in controlling emergent invasive species is an option, but could be overwhelmed by the rapidly rising number is cases.	🌡 ☀ 🌧	4.2.4.6	Present Near-term (2030-2040) Long-term (2080-2100) 2°C 4°C	Very low Medium Very high

Climatic drivers of impacts

🌡 Warming trend	🌡! Extreme temperature	☀ Drying trend	🌧 Precipitation

Risk & potential for adaptation

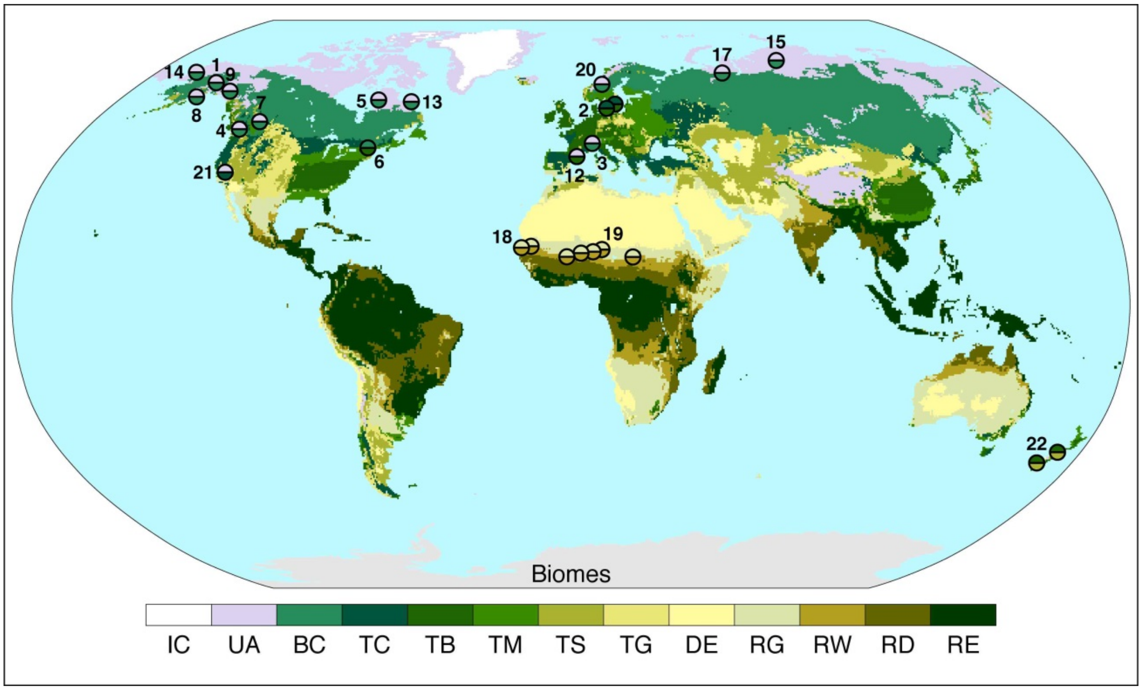


Figure 4-1: Locations of observed biome shifts during the 20th century, listed in Table 4-1, derived from Gonzalez *et al.* (2010). The color of each semi-circle indicates the retracting biome (top for North America, Europe, Asia; bottom for Africa and New Zealand) and the expanding biome (bottom for North America, Europe, Asia; top for Africa and New Zealand), according to published field observations. Biomes, from poles to equator: ice (IC), tundra and alpine (UA), boreal conifer forest (BC), temperate conifer forest (TC), temperate broadleaf forest (TB), temperate mixed forest (TM), temperate shrubland (TS), temperate grassland (TG), desert (DE), tropical grassland (RG), tropical woodland (RW), tropical deciduous broadleaf forest (RD), tropical evergreen broadleaf forest (RE). The background is the potential biome according to the MC1 dynamic global vegetation model under the 1961-1990 climate. **[Illustration to be redrawn to conform to IPCC publication specifications.]**

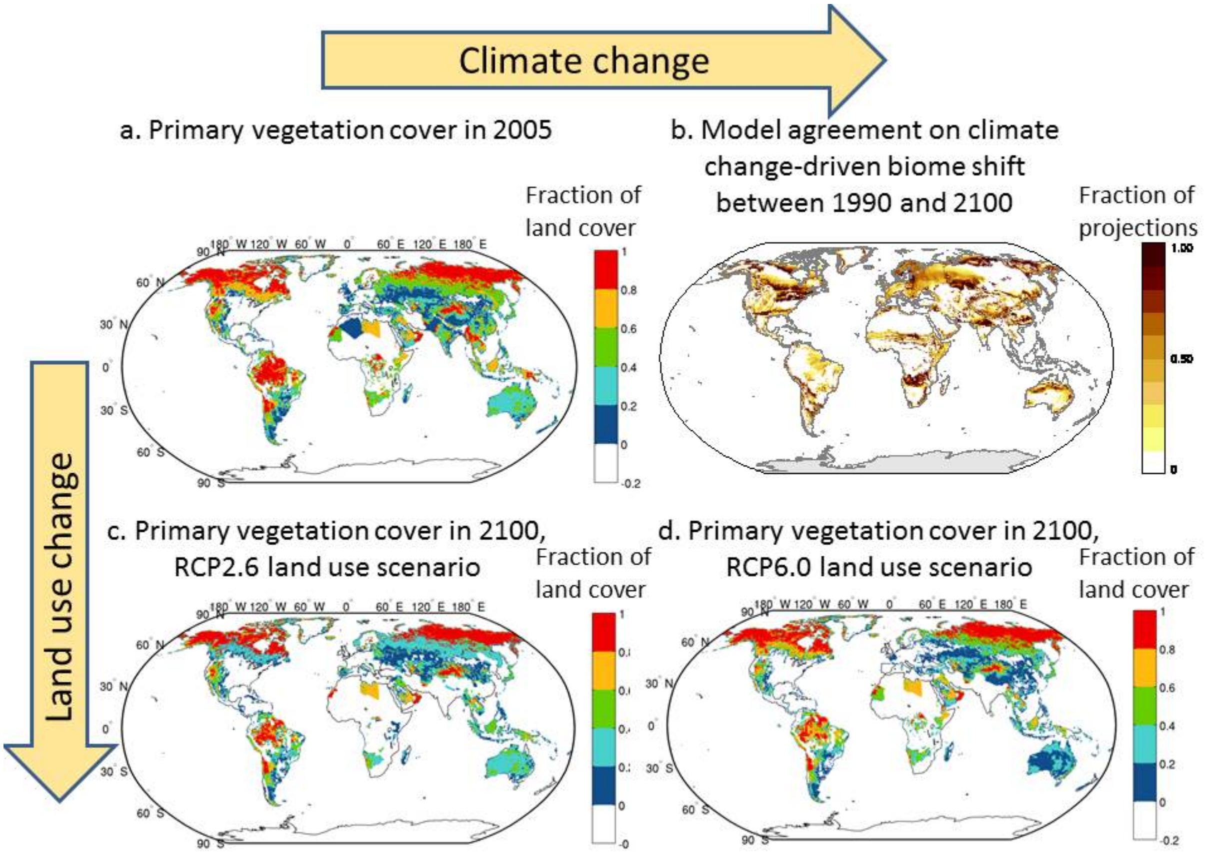


Figure 4-2: Implications of climate change and land use change for biome shifts. (a) Fraction of land covered by primary vegetation in 2005 (Hurtt *et al.*, 2011); (b) Fraction of simulations showing climate change-driven biome shift for any level of global warming between 1990 and 2100, with no direct anthropogenic land use change, using the MC1 vegetation model under 9 CMIP3 climate projections (3 GCMs each forced by the SRES A2, A1B and B1 scenarios; Gonzalez *et al.*, 2010); (c) Fraction of land covered by primary vegetation in 2100 under the RCP2.6 land use scenario with the IMAGE model, with no effect of climate change (Hurtt *et al.*, 2011); (d) Fraction of land covered by primary vegetation in 2100 under the RCP8.5 land use scenario with the MESSAGE model, with no effect of climate change (Hurtt *et al.*, 2011). Comparison of coloured areas in (b) with those in (a) shows where climate- driven biome shifts would occur in current areas of primary vegetation. Comparison of (b) and (d) shows where climate-driven biome shifts would occur in areas of primary vegetation projected under a land use scenario associated with RCP6.0. Comparison of (c) and (a) illustrates a scenario of land use change associated with RCP2.6, in which global climate change is projected to be smaller than that driving the biome shifts in (b) as a result of mitigation measures, some of which involve land use. Further details of the RCP land use / cover scenarios are given in Box 4-1, Figure 4-3 and Table 4-2.

[Illustration to be redrawn to conform to IPCC publication specifications.]

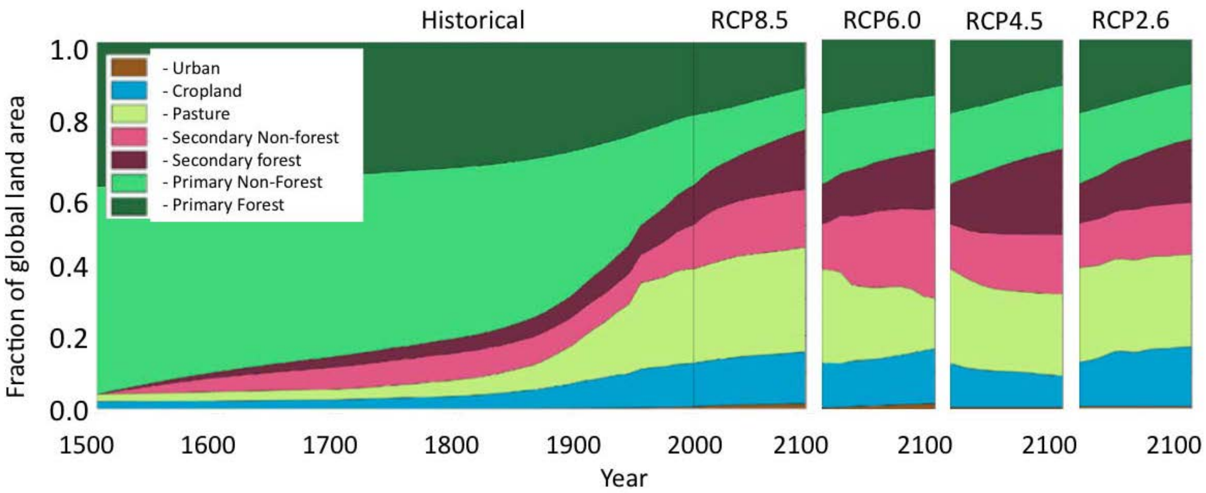


Figure 4-3: Proportion of global land cover occupied by primary and secondary vegetation (forest and non-forest), cropland, pasture and urban land, from satellite data and historical reconstructions up to 2005 (Klein Goldewijk *et al.*, 2010; Klein Goldewijk *et al.*, 2011), and from scenarios associated with the RCPs from 2005 to 2100 (Hurtt *et al.*, 2011)**. [Illustration to be redrawn to conform to IPCC publication specifications.]**

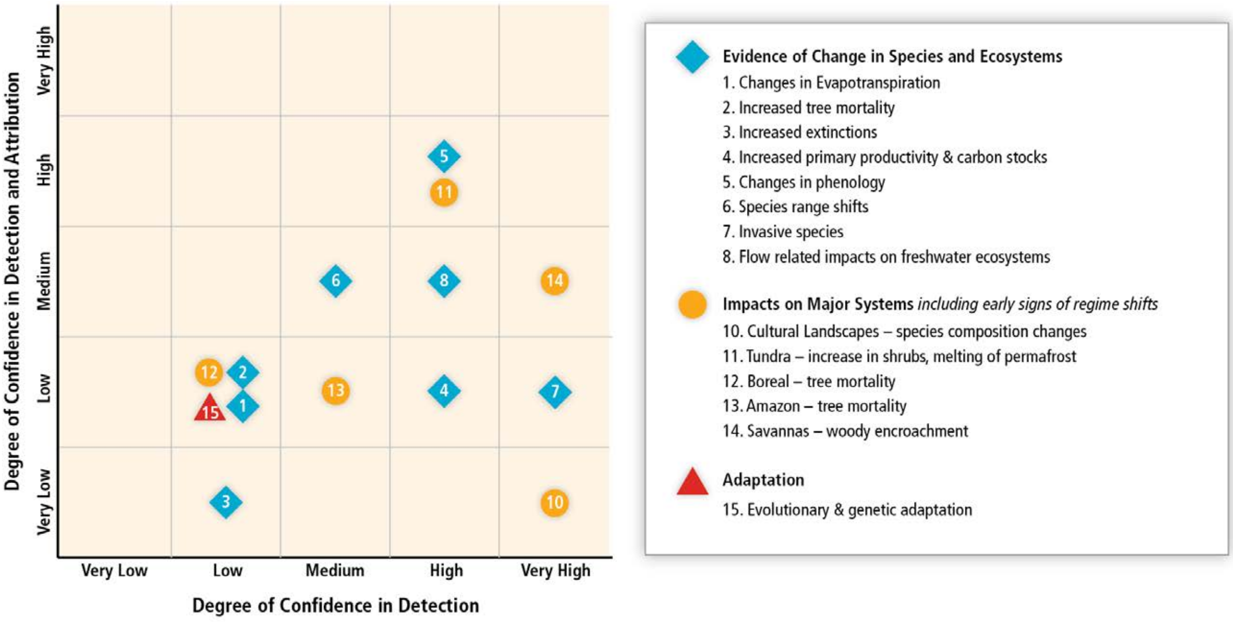


Figure 4-4: Confidence in detection of change and attribution of observed responses of terrestrial ecosystems to climate change. Confidence levels are based on expert judgment of the available literature following the IPCC uncertainty guidance (Mastrandrea *et al.*, 2010), attribution criteria outlined in Chapter 18 and detection criteria defined in the text. The symbols in the figure represent global and cross-taxon assessments; the positioning may be different for specific taxa or regions. The following sections provide the details of the assessments that were used in positioning each of the points Phenology, Section 4.3.2.1; Primary Productivity, Section 4.3.2.2; Biomass and C stocks, Section 4.3.2.3; Evapotranspiration, Section 4.3.2.4; Species distributions, Section 4.3.2.5**;** Global species extinctions, Section 4.3.2.5**;** Invasive and alien species**,** Section 4.2.4.6**;** Tree mortality, Section 4.3.3.1, Box 4-2; Boreal forest regime shift, Section 4.3.3.1.1, Box 4-4; Amazon forest regime shift, Section 4.3.3.1.3, Box 4-3; Tundra regime shift, Section 4.3.3.4, Box 4-4; Woody encroachment, Section 4.3.3.2.2; Cultural landscapes, Section 4.3.3.5.3; Evolutionary and genetic adaptation, Section 4.4.1.2.

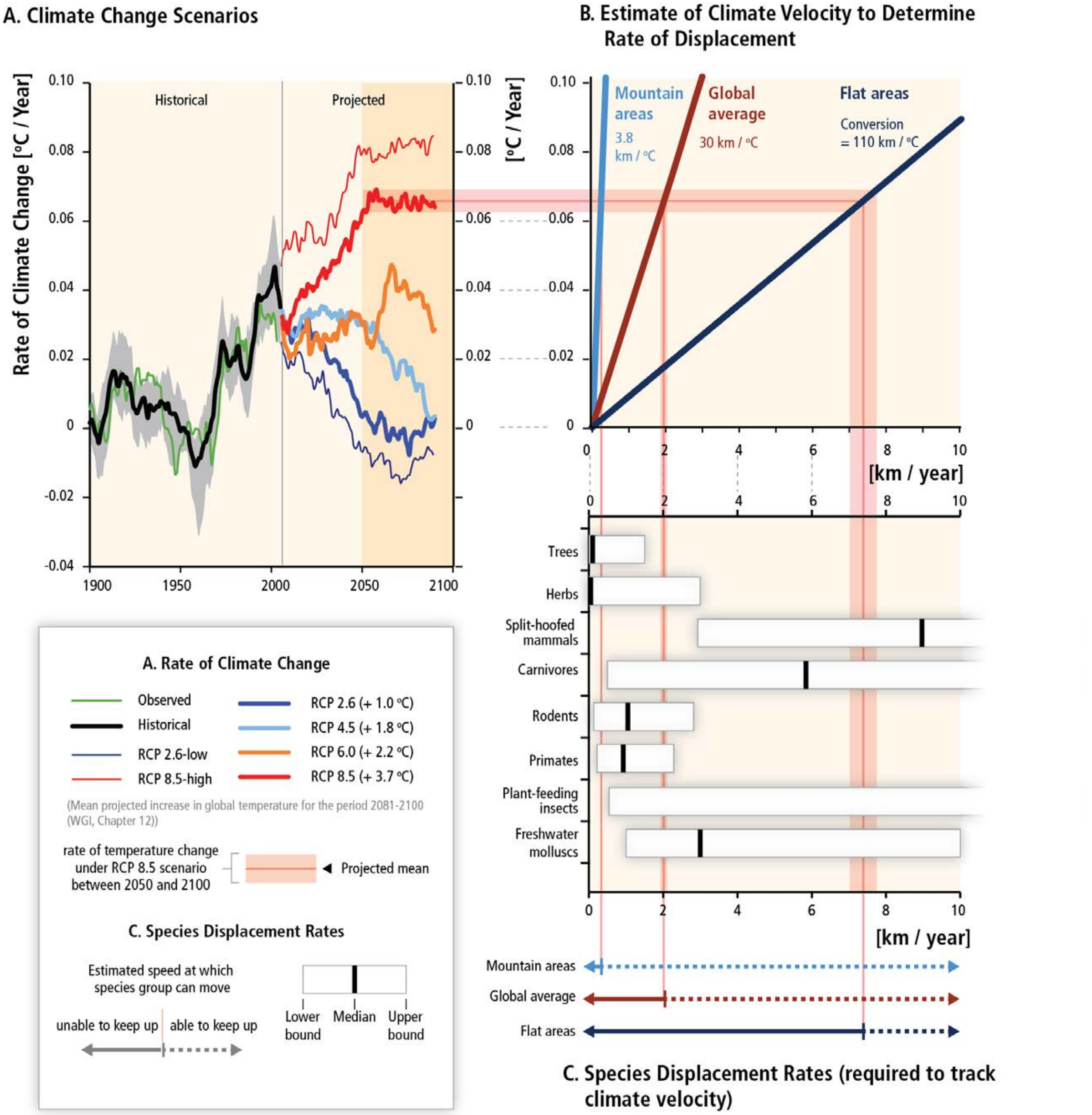


Figure 4-5: (A) Rates of climate change, (B) corresponding climate velocities and (C) rates of displacement of several terrestrial and freshwater species groups in the absence of human intervention. Horizontal and vertical pink bands illustrate the interpretation of this figure. Climate velocities for a given range of rates of climate change are determined by tracing a band from the range of rates in panel A to the points of intersection with the three climate velocity scalars in panel B. Comparisons with species displacement rates are made by tracing vertical bands from the points of intersection on the climate velocity scalars down to the species displacement rates in panel C. Species groups with displacement rates below the band are projected to be unable to track climate in the absence of human intervention. (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 CMIP5 climate model ensembles for the historical period (grey shading indicates model uncertainty) and for the future based on the four RCP emissions scenarios. Data were smoothed using a 20-year sliding window, and rates are means of between 17 and 30 models using one

member per model. Global average temperatures at the end of the 21st century for the four RCP scenarios are from WGI AR5 Chapter 12. (B) Estimates of climate velocity for temperature were synthesized from historical and projected future relationships between rates of temperature change and climate velocity (historical: Burrows *et al.*, 2011; Chen *et al.*, 2011; Dobrowski *et al.*, 2013; projected future: Loarie *et al.*, 2009; Sandel *et al.*, 2011; Feeley and Rehm, 2012). The three scalars are climate velocities that are representative of mountainous areas (left), averaged across global land areas (centre), and large flat regions (right). (C) Rates of displacement are given with an estimate of the median (black bars) and range (boxes = ca. 95% of observations or models for herbaceous plants, trees and plant-feeding insects or median ± 1.5 inter-quartile range for mammals). Displacement rates for herbaceous plants were derived from paleobotanical records, modern plant invasion rates and genetic analyses (Kinlan and Gaines, 2003). 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 modelled 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). Displacement rates for mammals were based on modelled dispersal rates of a wide range of mammal species (mean of Schloss *et al.*, 2012 for Western Hemisphere mammals and rates calculated from global assessments of dispersal distance by Santini *et al.*, 2013 and generation length by Pacifici *et al.*, 2013). Displacement rates for phytophagous insects are based on observed dispersal distances and genetic analyses (Peterson and Denno, 1998; Kinlan and Gaines, 2003; Schneider, 2003; Berg *et al.*, 2010; Chen *et al.*, 2011). The estimate of median displacement rate for this group exceeds the highest rates on the axis. These displacement rates do not take into account limitations imposed by host plants. Displacement estimates for freshwater molluscs correspond to the range of passive plus active dispersal rates for upstream movement (Kappes and Haase, 2012).

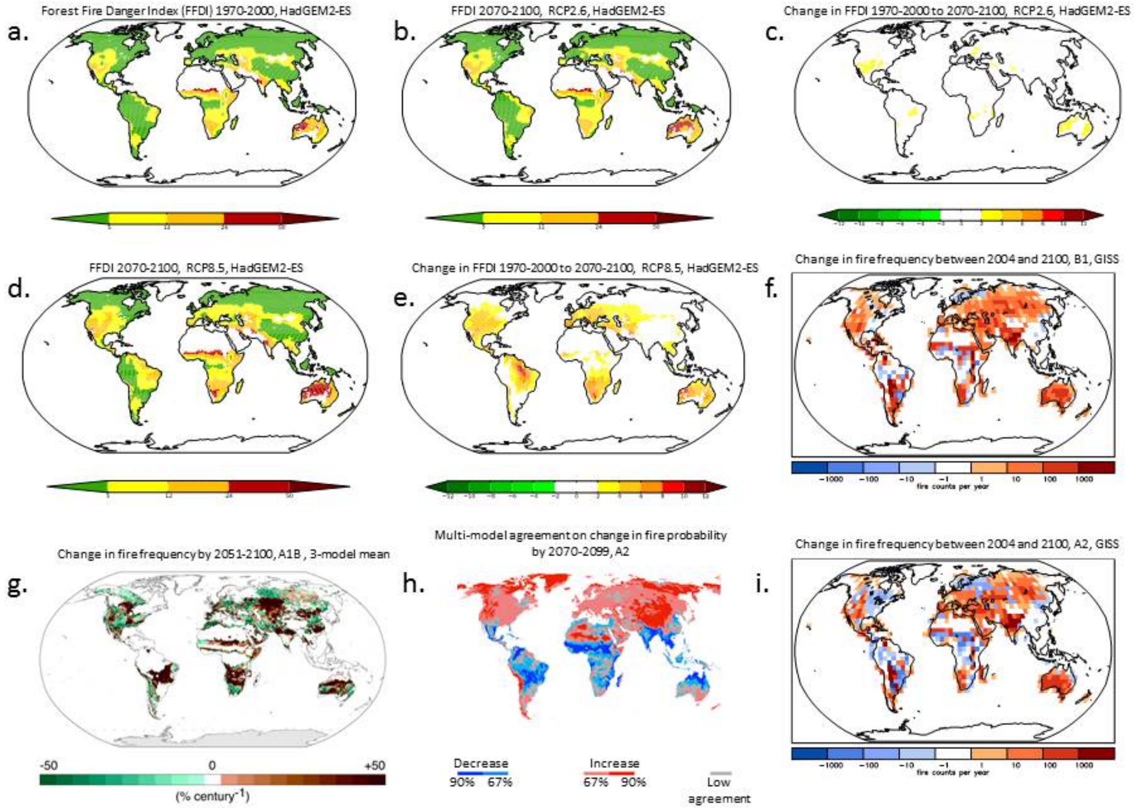


Figure 4-6: Projected changes in meteorological fire danger, fire probability and fire frequency with different methods and climate models. (a)-(e) 30-year annual mean MacArthur Forest Fire Danger Index (FFDI) and change simulated with the HadGEM2-ES Earth System Model, with areas of no vegetation excluded (Betts *et al.*, 2013; (a) FFDI 1970-2000; (b) FFDI 2070-2100, RCP2.6; (c) change in FFDI by 2070-2100 relative to 1970-2000, RCP2.6;(d) FFDI 2070-2100; RCP8.5 (e) change in FFDI by 2070-2100 relative to 1970-2000, RCP8.5. (f) Change in fire frequency by 2100 relative to 2004, SRES B1, simulated using climate and land cover projections from the GISS GCM and IMAGE IAM (Pechony and Shindell, 2010). (g) Change in fire frequency by 2051-2100 relative to 1951-2000, SRES A1B, simulated with the MC1 vegetation model driven by 3 GCMs (CSIRO-Mk3.0, HadCM3, MIROC 3.2medres; mean over 3 simulations; Gonzalez et al., 2010). (h) Agreement on changes in fire probability simulated with a statistical model using climate projections from 16 CMIP3 GCMs, SRES A2 (i) Change in fire frequency by 2100 relative to 2004, SRES A2, simulated using climate and land cover projections from the GISS GCM (AR4 version) and IMAGE IAM (Pechony and Shindell, 2010). Changes in FFDI (a)-(e) and fire probability (h) arise entirely from changes in meteorological quantities, whereas changes in fire frequency (f) (g) (i) depend on both meteorological quantities and vegetation.

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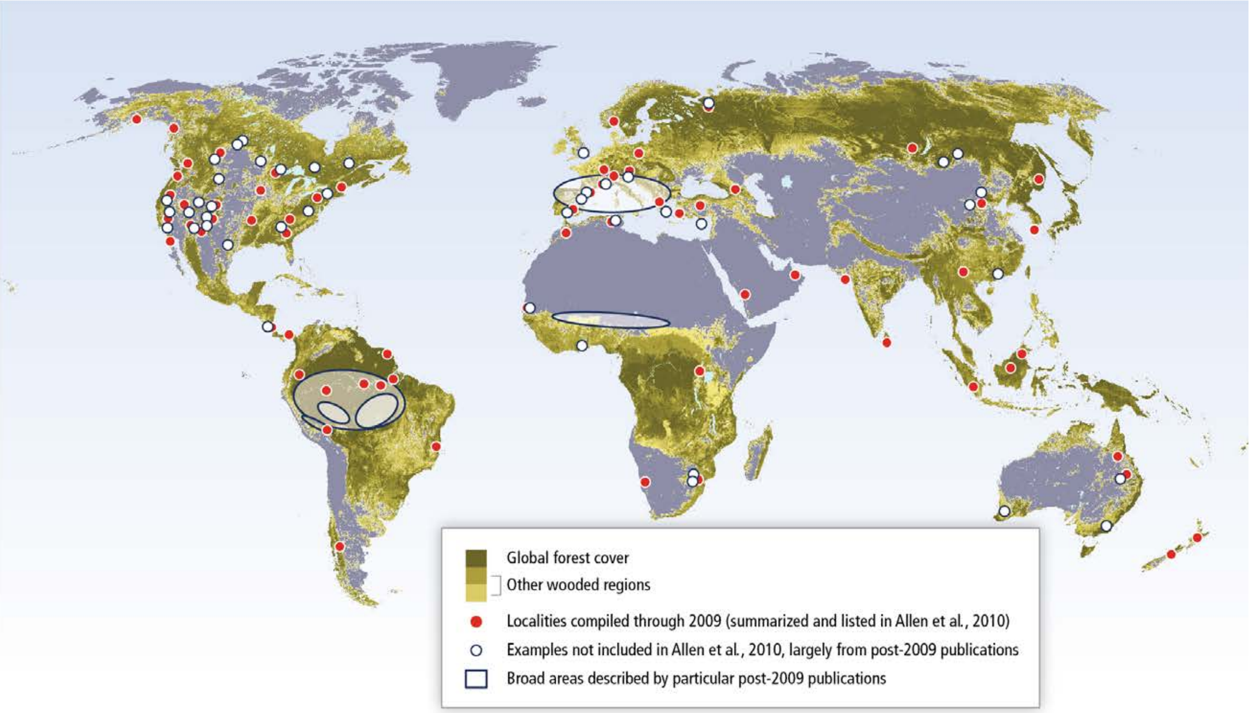

Figure 4-7: Locations of substantial drought- and heat-induced tree mortality around the globe since 1970 (global forest cover and other wooded regions based on FAO, 2005). Studies compiled through 2009 (red dots) are summarized and listed in Allen *et al.* (2010a). Localities and measurement networks not included in Allen *et al.* (2010a), which are largely from post-2009 publications, have been added to this map (white dots and shapes). New locality references by region: Africa – Mehl *et al.*, 2010, van der Linde *et al.*, 2011; Fauset *et al.*, 2012; Gonzalez *et al.*, 2012; Kherchouche *et al.*, 2012; Asia – Dulamsuren *et al.*, 2009; Kharuk *et al.*, 2013; Liu *et al.*, 2013; Zhou *et al.*, 2013; Australasia – Brouwers *et al.*, 2012; Fensham *et al.*, 2012; Keith *et al.*, 2012; Matusick *et al.*, 2012; Brouwers *et al.*, 2013; Matusick *et al.*, 2013; Europe – Innes, 1992; Peterken and Mountford, 1996; Linares *et al.*, 2009; Galiano *et al.*, 2010; Vennetier and Ripert, 2010; Aakala *et al.*, 2011; Carnicer *et al.*, 2011; Linares *et al.*, 2011; Sarris *et al.*, 2011; Marini *et al.*, 2012; Cailleret *et al.*, 2013; Vilà-Cabrera *et al.*, 2013; North America – Fahey, 1998; Minnich, 2007; Klos *et al.*, 2009; Ganey and Vojta, 2011; Michaelian *et al.*, 2011; Peng *et al.*, 2011; DeRose and Long, 2012; Fellows and Goulden, 2012; Kaiser *et al.*, 2012; Millar *et al.*, 2012; Garrity *et al.*, 2013; Kukowski *et al.*, 2013; Williams *et al.*, 2013; Worrall *et al.*, 2013; South America – Enquist and Enquist, 2011; Lewis *et al.*, 2011; Saatchi *et al.*, 2013.

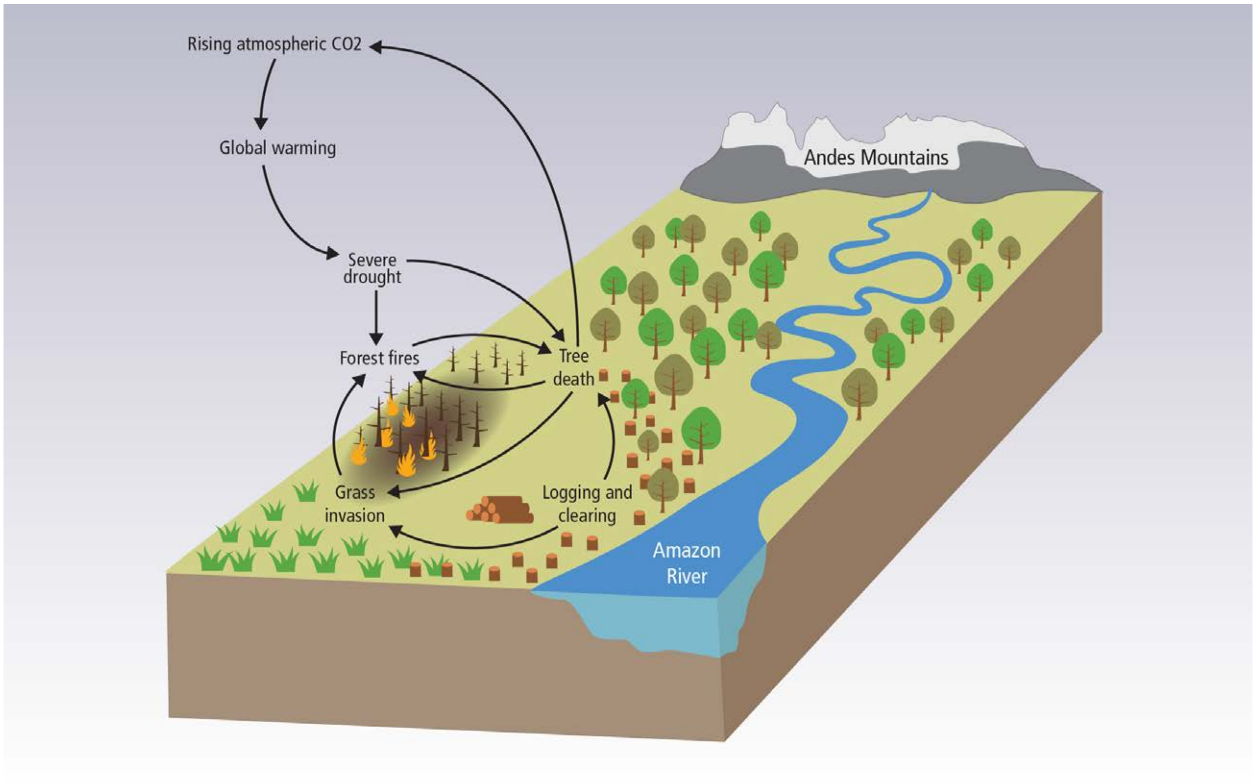

Figure 4-8: The forests of the Amazon Basin are being altered through severe droughts, land use (deforestation, logging), and increased frequencies of forest fire. Some of these processes are self-reinforcing through positive feedbacks, and create the potential for a large-scale tipping point. For example, forest fire kills trees, increasing the likelihood of subsequent burning. This effect is magnified when tree death allows forests to be invaded by flammable grasses. Deforestation provides ignition sources to flammable forests, contributing to this dieback. Climate change contributes to this tipping point by increasing drought severity, reducing rainfall and raising air temperatures, particularly in the eastern Amazon Basin (*medium confidence; medium evidence, medium agreement).*]

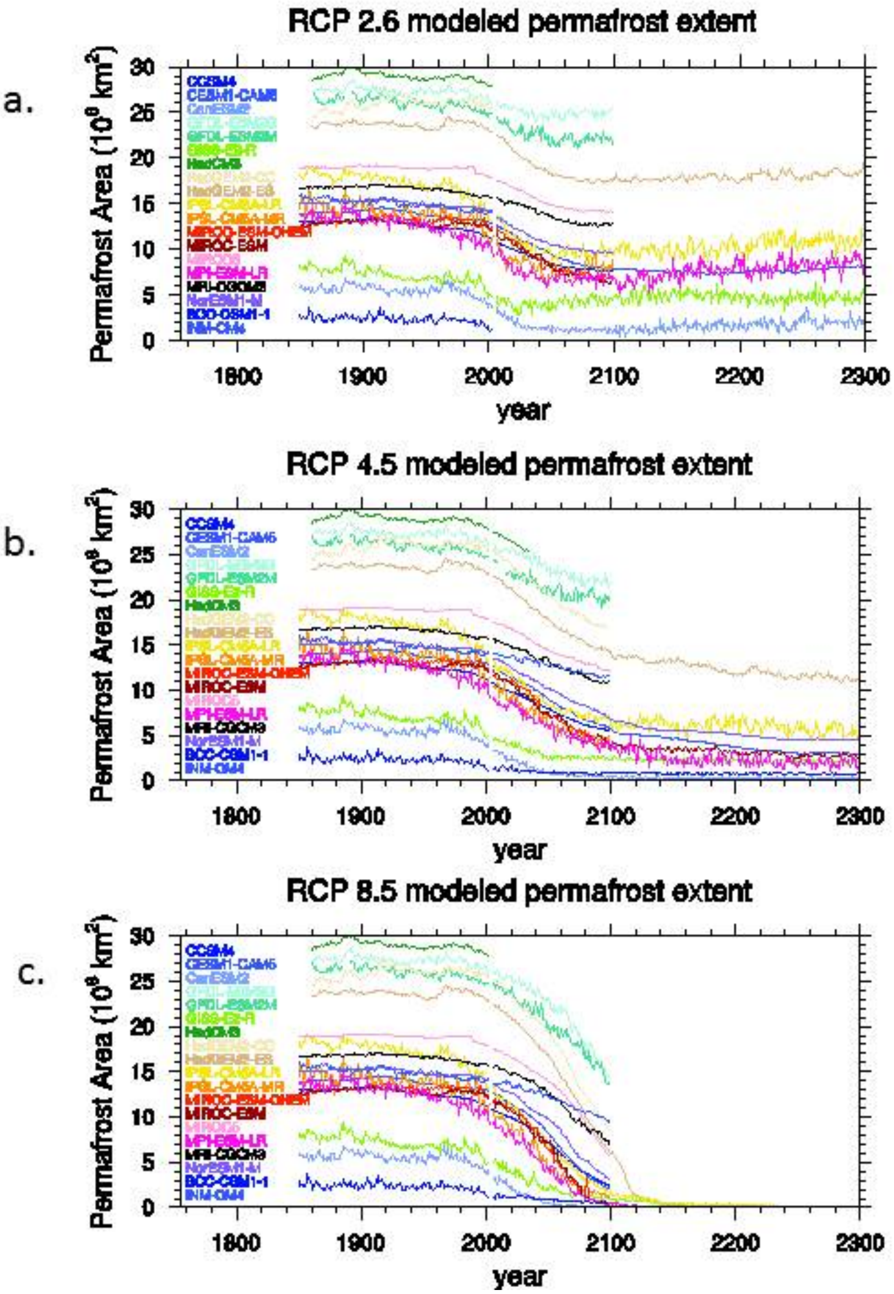

Figure 4-9: CMIP5 multi-model simulated area of Northern Hemisphere permafrost in the upper 3m of soil, from 1850 to 2100 or 2300 depending on extent of individual simulations. Each panel shows historical (1850-2005) and projected (2005 to 2100 or 2300) simulations for (a) RCP2.6, (b) RCP4.5, and (c) RCP8.5. The observed current permafrost extent is 15×10^6 km^2. (Based on Koven *et al.*, 2013, with analysis extended to 2300 following Caesar *et al.*, 2013). **[Illustration to be redrawn to conform to IPCC publication specifications.]**

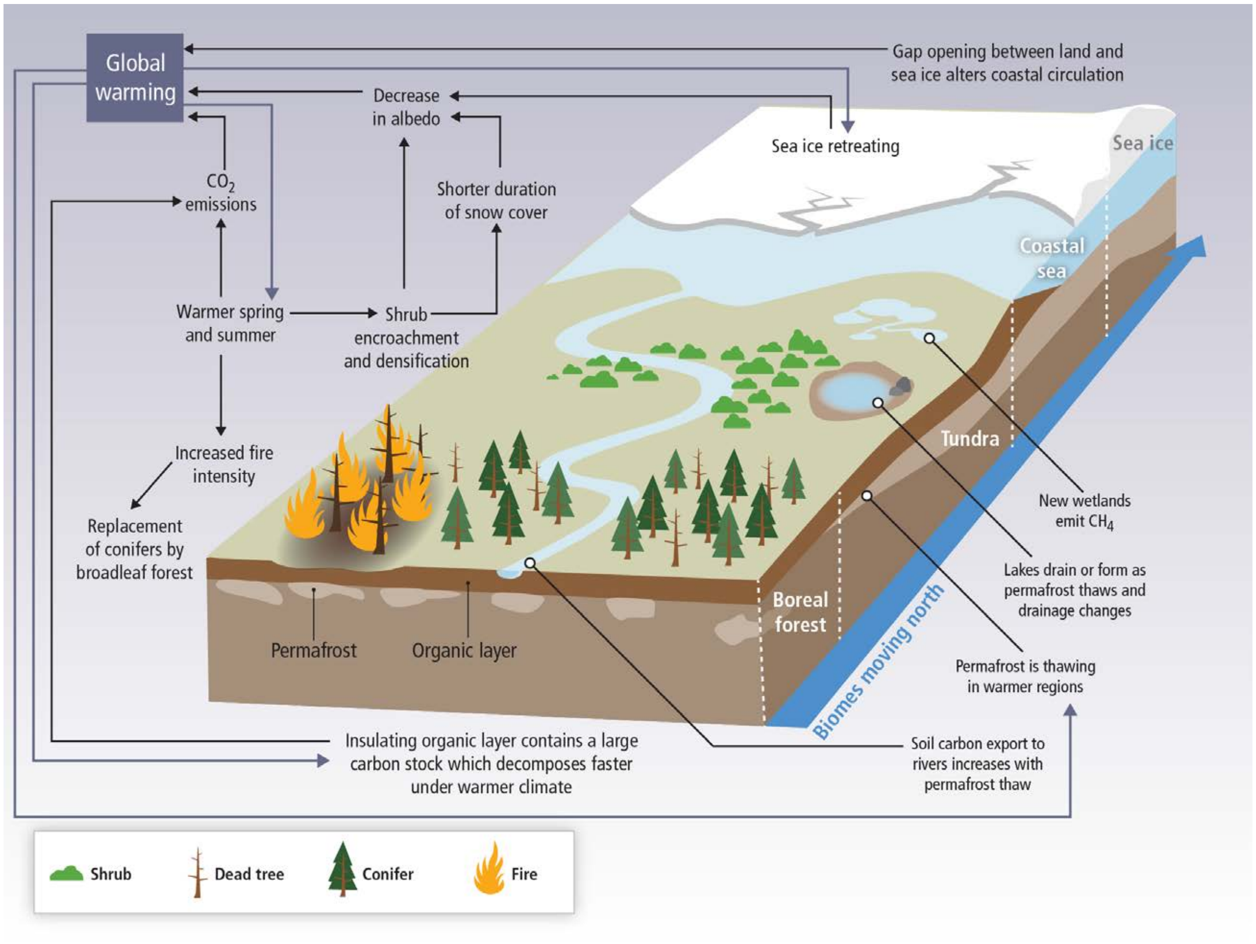

Figure 4-10: Tundra-Boreal Biome Shift. Earth system models predict a northward shift of Arctic vegetation with climate warming, as the boreal biome migrates into what is currently tundra. Observations of shrub expansion in tundra, increased tree growth at the tundra-forest transition, and tree mortality 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, permafrost thawing and methane emissions, with net feedbacks to additional climate change.

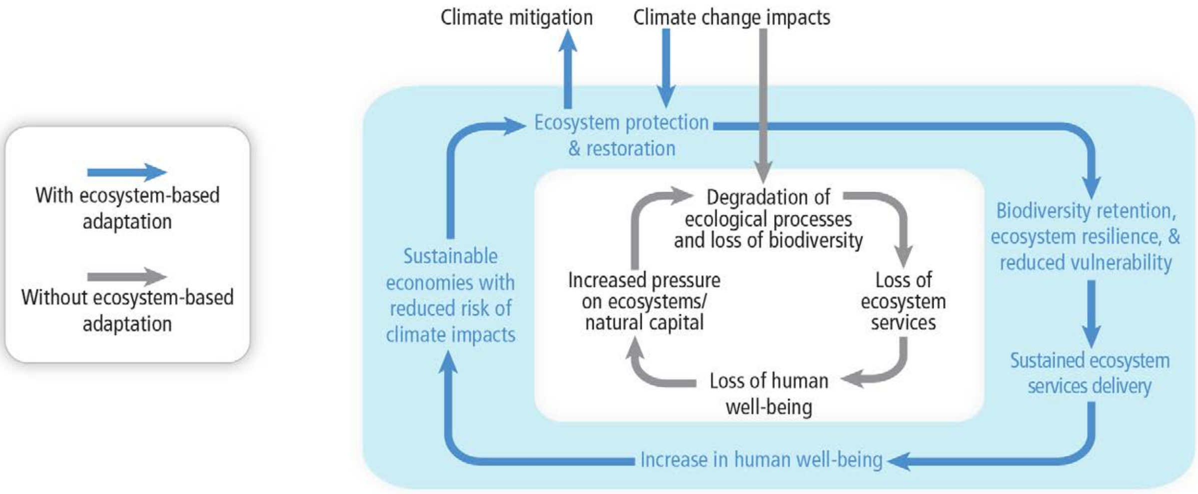

Figure EA-1: Adapted from Munang et al. (2013). Ecosystem based adaptation (EBA) uses the capacity of nature to buffer human systems from the adverse impacts of climate change. Without EBA, climate change may cause degradation of ecological processes (central white panel) leading to losses in human well-being. Implementing EBA (outer blue panel) may reduce or offset these adverse impacts resulting in a virtuous cycle that reduces climate-related risks to human communities, and may provide mitigation benefits.

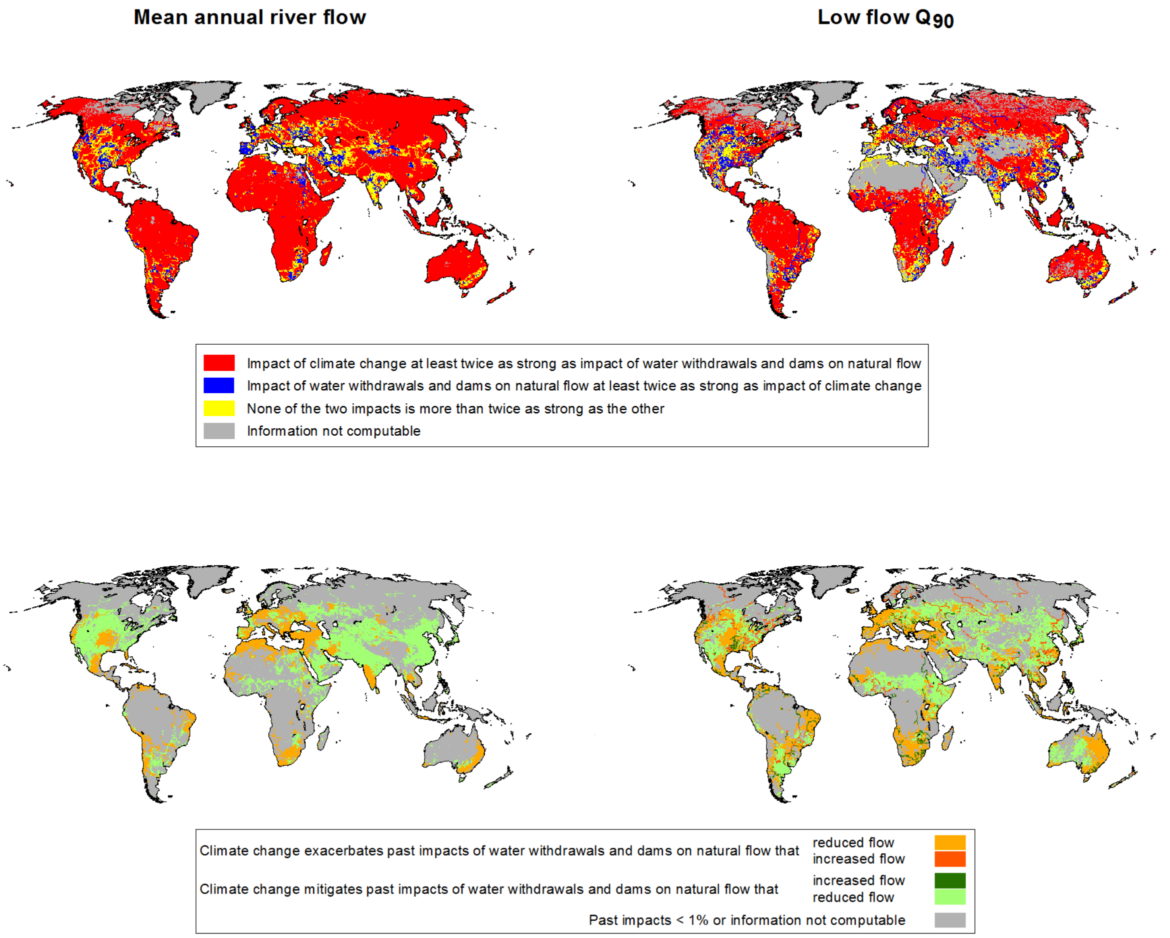

Figure RF-1: Impact of climate change relative to the impact of water withdrawals and dams on natural flows for two ecologically relevant river flow characteristics (mean annual river flow and monthly low flow Q_{90}), computed by a global water model (Döll and Zhang, 2010). Monthly Q_{90} was defined as the flow that is exceeded in 9 out 10 months. 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. Please note that the figure does not reflect spatial differences in the magnitude of change.
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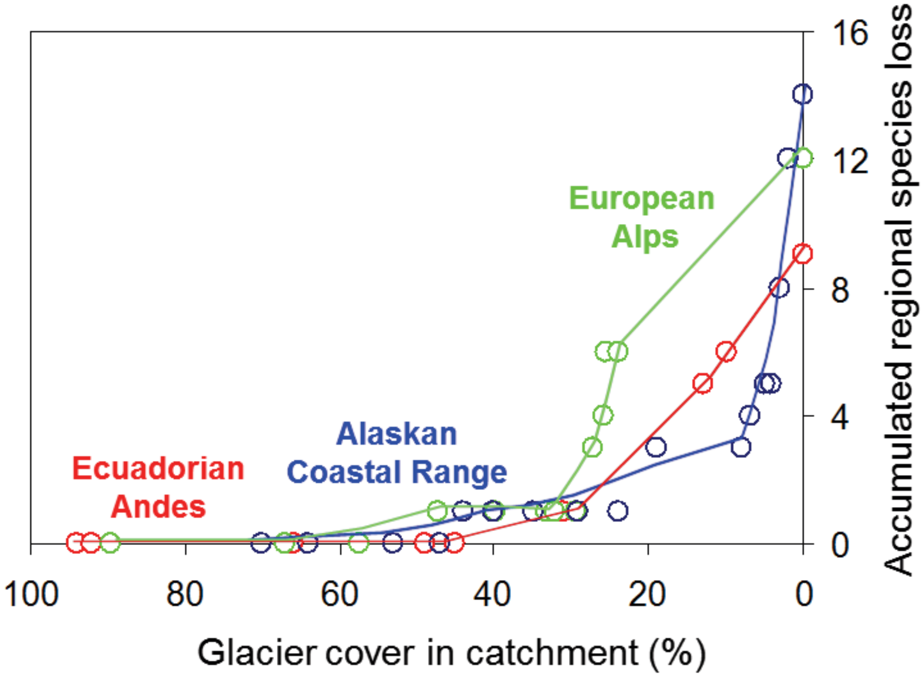

Figure RF-2: Accumulated loss of regional species richness (gamma diversity) of macroinvertebrates as a function of glacial cover in catchment. Obligate glacial river macroinvertebrates begin to disappear from assemblages when glacial cover in the catchment drops below approximately 50%, and 9-14 species are predicted to be lost with the complete disappearance of glaciers in each region, corresponding to 11, 16 and 38% of the total species richness in the three study regions in Ecuador, Europe and Alaska. Data are derived from multiple river sites from the Ecuadorian Andes and Swiss and Italian Alps, and a temporal study of a river in the Coastal Range Mountains of southeast Alaska over nearly three decades of glacial shrinkage. Each data point represents a river site or date (Alaska), and lines are Lowess fits. Adapted by permission from Macmillan Publishers Ltd: *Nature Climate Change*, Jacobsen *et al.*, 2012, © 2012.

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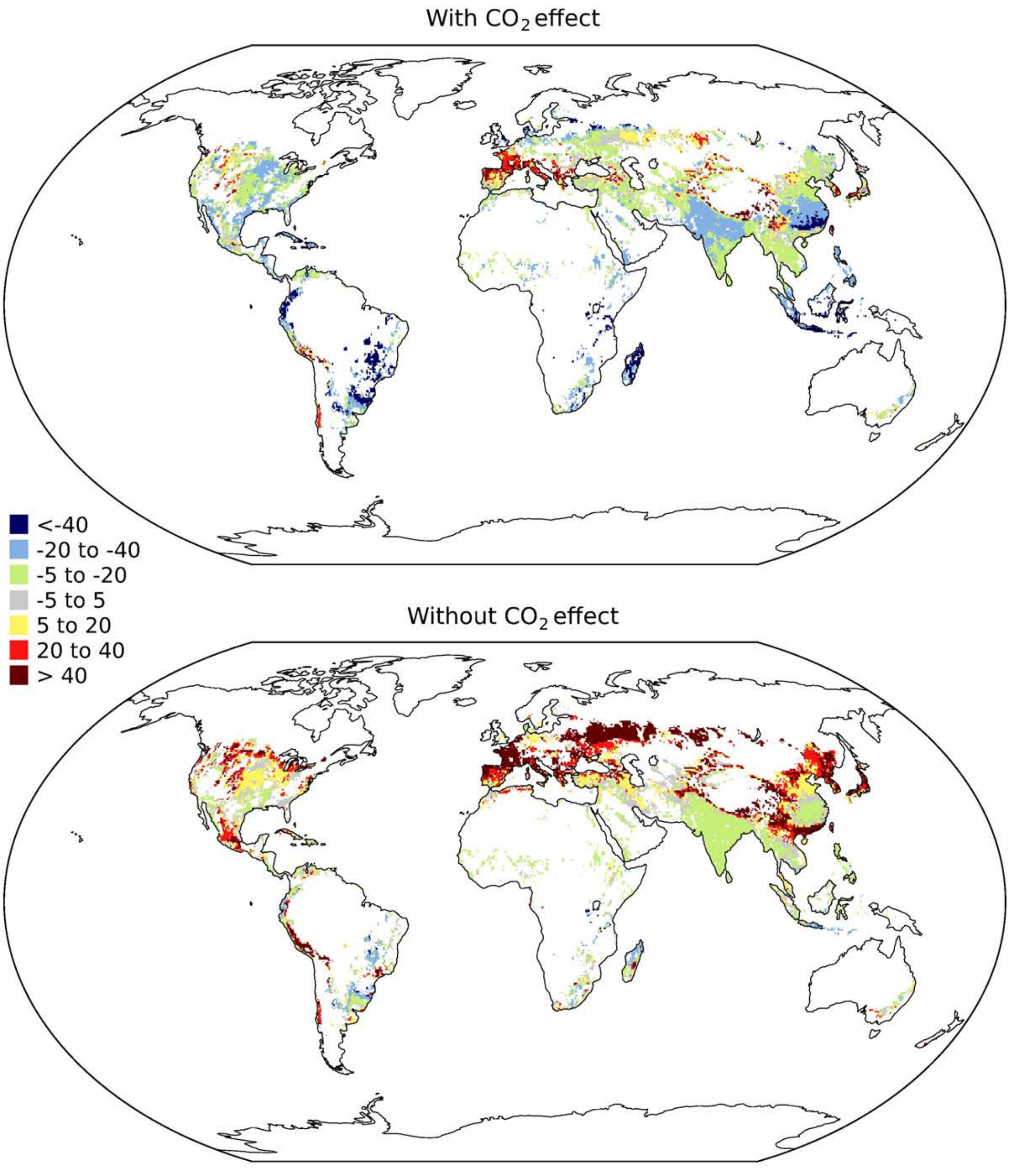

Figure VW-1: Percentage change in net irrigation requirements of 11 major crops from 1971–2000 to 2070–2099 on areas currently equipped for irrigation, assuming current management practices. Top: impact of climate change including physiological and structural crop responses to increased atmospheric CO_2 concentration (maximum effect in the absence of co-limitation by nutrients). Bottom: impact of climate change only. Shown is the median change derived from climate change projections by 19 GCMs (based on the SRES A2 emissions scenario) used to force a vegetation and hydrology model. Modified after Konzmann *et al.* (2013).
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