Chapter 4. Terrestrial and Inland Water Systems

Coordinating Lead Authors

Robert Scholes (South Africa), Josef Settele (Germany)

Lead Authors

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

Contributing Authors

Duan Biggs (Australia), Paulo Brando (Brazil), Wendy Foden (South Africa), Scott Goetz (USA), Nicola Golding (UK), Patrick Gonzales (USA), Thomas Hickler (Germany), Melodie McGeoch (Australia), Camille Parmesan (USA), Beatriz Rodriguez-Labajos (Spain), Klement Tockner (Germany), Marten Winter (Germany)

Review Editors

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

Volunteer Chapter Scientist

Marten Winter (Germany)

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Executive Summary

The effects of climate change on, and the adaptations by ecosystems are expressed at two nested levels: the individualistic behaviour of constituent organisms; and the response of the ecosystem as a whole.

Individual plant and animal species have moved their ranges and altered their abundance and seasonal activities in response to climate change in the past. There is *high certainty* that they are doing so now and will continue to do so in the future. No past climate changes are precise analogs to the current and projected environmental changes, so responses inferred from the past only give indications, especially at the local scale. [4.2.2] The broad patterns of poleward and upward movement in response to a warming climate are well-established for the distant and recent past and are forecast by a wide range of models. [4.2.4.6, 4.3.2, 4.3.2.5] Mountain-top endemic species with limited dispersal capability will be particularly vulnerable as their habitat contracts in response to rising temperature. [4.3.2.5] Species responses to climate change in the modern era are constrained by the simultaneous presence of other stresses, including but not limited to harvest pressure, habitat fragmentation and loss, competition with alien species, exposure to novel pests and diseases, nitrogen loading and increasing carbon dioxide and tropospheric ozone. [Figure 4-1, 4.2.4-4.2.4.6, 4.3.3-4.3.3.5] Dams and other barriers in river systems constrain dispersal of fully aquatic species. [4.3.3.3] Landuse change and water resource development are likely to continue to dominate the threats to freshwater ecosystems and many terrestrial ecosystems, with climate change becoming an increasing stress later in the century. Climate change exacerbates the other threats. In some systems, such as high altitude and latitude freshwater and terrestrial ecosystems will with *high certainty* lead to major changes in species distributions and ecosystem function. [4.3.2.5, 4.3.3.1, 4.3.3.3, 4.4.1.1] In freshwater systems, adaptation responses to counter increased variability of water supply for urban and agricultural use will compound this effect. [4.3.3.3, 4.3.4.5] Species movement into areas where they were not present historically is facilitated both by climate change and by increased dispersal opportunities associated with human activities. Climate niche models, which have well-documented weaknesses, suggest that under mid-range warming scenarios for the end of the 21st century, the climatically-preferred distribution for a tenth to a third of plant and vertebrate species will lie entirely outside the footprint of the current distribution. For high-warming scenarios this rises to a fifth to a half. It is inferred that this will increase the risk of premature extinction for these species substantially, but the duration of the period until actual extinction occurs is unknown. [Figure 4-9, 4.3.2.5]

Changes in the ecosystem disturbance regime (e.g., the frequency and intensity of events such as fires, pest outbreaks, wind-storms, episodes of low or high river flows and droughts on land) are apparent in many parts of the world. Such changes, beyond the range of natural variability, will with *high certainty* become pervasive during the 21st century, in large but not exclusively due to climate change. Theory and observation both suggest that changes in ecosystem composition, structure and function will more frequently be manifested as relatively abrupt and spatially-patchy transitions following such disturbances, than as a gradual and uniform shift in the abundance and distribution of individual species. [Box 4-2, 4.2.4.4, Table 4-3, 4.2.4.6, 4.3.1, 4.3.2.5, Figure 4-12, 4.3.3-4.3.3.5]

There is widespread and well-established evidence of terrestrial and freshwater ecosystems changing abruptly (and irreversibly, in practical management terms) both in the distant past and more contemporary times. Such changes have occurred in response to climate changes of comparable magnitude to those projected in the coming century, acting alone or in combination with other drivers. There are plausible mechanisms, supported by experimental evidence and model results, for the existence of such ecosystem ‘tipping points’ in both tundra permafrost systems and the rainforests of the Amazon basin, among others. [Boxes 4-4 and 4-6] In both the arctic and Amazonian cases, the consequences for the regional and global climate system of large-scale ecosystem regime shifts would be substantially negative, given the large changes carbon pools, methane fluxes, albedo and transpiration potentially involved. [4.2., 4.3.3-4.3.3.4, Figures 4-10 and 4-11]

The terrestrial biosphere pools in which carbon is currently being stored are vulnerable to climate change, changes in disturbance regime and other ecosystem stressors and changes, including land use change. Terrestrial and freshwater ecosystems have been responsible for the uptake of about a quarter of all anthropogenic CO_2 emissions in the past half century. [4.3.2.3] The net fluxes out of the atmosphere and into plant biomass and soils show large interannual variability. No conclusion can be drawn regarding observed trends in the net fluxes into or out of terrestrial ecosystems at the global scale. [4.2.4.1, 4.2.4-4.2.4.4, Table 4-3, 4.3.2-4.3.2.4, 4.3.3-4.3.3.4, Box 4-4]

There is widespread evidence for increases in tree mortality in recent decades, sometimes accompanied by increased growth rates and/or tree recruitment, leading to more rapid turnover of forest biomass stocks. The quantitative attribution of this observed trend between the several possible causes is as yet uncertain. The consequences for the provision of timber from particular forests are likely to be variable between regions and products. [4.3.2, 4.3.3.1, 4.3.3.4, Boxes 4-4 and 4-6]

The capacity for autonomous adaptation within ecosystems and their constituent organisms is substantial, but in many ecosystems, with *medium certainty*, is insufficient to cope (without significant loss of species and ecosystem services) with the rate and magnitude of climate change projected under moderate or high climate change scenarios for this century, unless aided by deliberate adaptation strategies. Under high-magnitude climate change scenarios, substantially loss-free adaptation may be unattainable, even if assisted by human interventions (*medium certainty*). [4.4.1] The capacity for ecosystems to adapt to climate change can be increased by reducing the other stresses operating on them; reducing the rate and magnitude of change; reducing habitat fragmentation and increasing connectivity; maintaining a large pool of genetic diversity and functional evolutionary processes; assisted translocation of slow moving or blocked organisms along with the species on which they depend; and manipulation of disturbance regimes to keep them within desired ranges. [4.4.1-4.4.1.2, 4.4.3]

4.1. Past Assessments

The topics assessed in this chapter were last assessed by the IPCC in 2007, principally in the Working Group II report chapters 3 (Freshwater resources and their management; Kundzewicz *et al.*, 2007) and 4 (Ecosystems, their properties, goods and services; Fischlin *et al.*, 2007). Together they found that ‘Observational evidence from all

continents and most oceans shows that many natural systems are being affected by regional climate changes, particularly temperature increases'. Although circumstantial evidence was offered that anthropogenic climate change was a cause of the observed changes, it was not possible to attribute a relative proportion of the changes to this cause with any confidence. An important finding was that 20-30% of the plant and animal species that had been assessed to that time were considered likely to be at increased risk of extinction if the global average temperature increases exceeded 2-3°C. Fischlin *et al.*, 2007) also stated that substantial changes in structure and functioning of ecosystems (terrestrial, marine and other aquatic) are very likely under a warming of more than 2-3°C above pre-industrial levels. No timescale was associated with these findings.
The report noted that vulnerability to climate change could be exacerbated by the presence of other stresses, such as pollution.

4.2. A Changing View of Ecosystem Change: Thresholds and Tipping Points

Ecologists no longer view ecosystems as inherently unchanging unless perturbed by an externally originating disturbance (Hastings, 2004). A growing body of both theory and observation (Scheffer, 2009) supports the emerging view that most ecosystems vary substantively over time in the relative magnitude of their components and fluxes, even under a relatively constant environment. Furthermore, attempts to restrict this intrinsic variation - or that resulting from imposed disturbances - are frequently futile, and may damage the capacity of the ecosystem to adapt to changing environments (Folke *et al.*, 2004).

A related conceptual shift has been to view ecosystems as containing humans, rather than existing in a natural state that is altered by humans without any feedback to the human system (Gunderson and Holling, 2001). The emerging view is of relatively tightly coupled social-ecological systems, which means that their joint dynamics are governed not only by biophysical processes such as energy flows, material cycles, competition and predation, but also by social processes such as economics, politics, culture and individual preferences (Walker and Salt, 2006).

4.2.1. Ecosystems as Dynamic and Adaptive Entities

This concept of ecosystems as dynamic rather than constant entities, along with the realisation that due to the reach and intensity of human activities few ecosystems can any longer be considered to be in an anthropogenically-unperturbed state, has led to increased focus on how ecosystems behave in the vicinity of their environmental limits, rather than solely on their behaviour near the middle of their domain of their inferred preferred state, (Scheffer *et al.*, 2009). The notion of thresholds has become a prominent ecological and political concern (Lenton *et al.*, 2008; Scheffer *et al.*, 2009; Leadley *et al.*, 2010). Some of the thresholds that have been discussed reflect a human preference that the ecosystem stays within certain bounds. Others reflect fundamental biophysical properties, such as the melting point of ice, which determines, for instance, the occurrence of permafrost-based ecosystems. A third category of threshold reflects system dynamics: the point at which the net effect of all the positive and negative feedback effects that characterize ecosystems is positive, causing a small perturbation to become accentuated and lead to a change in ecosystem state, from which it will not spontaneously return in the short term. The new state will typically exhibit different dynamics, mean composition, sensitivity to environmental drivers and flows of ecosystem services relative to the prior state. An ecosystem regime shift is said to have occurred (see glossary).

The early detection and prediction of ecosystem thresholds, particularly of the third type, is an area of active research. There are indications (Scheffer, 2009) that an increase in ecosystem variability signals the impending approach of a threshold, but in practice, such signals may not be detectable against background noise and uncertainty until it is too late to avoid the threshold (Biggs *et al.*, 2009).

The term 'resilience' has attracted a range of meanings in different disciplines (see gloassary). In ecology it is used both in the sense of 'a measure of the ability of these systems to absorb changes of state variables, driving variables and parameters, and still persist' (Holling, 1973), but also in a broader sense of being able to adapt to change, without necessarily remaining within the same dynamical range (Walker *et al.*, 2004). One source of such adaptability is the process of evolution, which generates variability and then selects for those organisms better able to survive and reproduce in the given environment. Organisms also typically exhibit a degree of physiological or behavioural adaptability (phenological adaptation) that does not require genetic change. A less well-understood source of adaptation is the configuration and functional form of interactions within an ecosystem, that leads to new stable states (www.regimeshifts.org). Third occurs when the ecosystem contains human agents sufficiently powerful to alter the ecosystem regime, and particularly where learning and technological development are involved.

[INSERT FIGURE 4-1 HERE
Figure 4-1: Some of the main mechanisms by which ecosystems are affected by a changing global environment. Several aspects of contemporary global change (left-hand columns), including those that affect the climate, combine and interact through a large number of mechanisms (middle columns) to influence ecosystem outcomes (right-hand column), and ultimately human wellbeing. The signs of the interactions are in some cases always negative, in others always positive, and in yet others either positive or negative, depending on circumstances. As a result, and because of the complexity of the interactions, a range of outcomes - from beneficial to detrimental- is plausible for given scenarios of change and locations. In some cases the outcome is not currently predictable, even if the climate futures were known precisely.]

_____ START BOX 4-1 HERE_____

Box 4-1. Historical and Projected Biome Shifts

Biome Shifts in the Past

A ‘biome shift’ is the collective change in a community of organisms, such that the ecosystem represented at a location can be said to have changed from one biome to another, typically because a previously-dominant functional type is now absent or insignificant. Analysis of the molecular phylogenies of 11 000 vascular plant species from seven southern hemisphere biomes suggests that clusters of related species (clades) have an inherently limited capacity to change the biome in which they occur (Crisp *et al.*, 2009). Field observations over the past century in numerous locations in boreal, temperate and tropical ecosystems have detected movements of sets of plant species upwards in elevation and to higher latitudes (Table 4-1, Figure 4-2), frequently attributed to climate change. 20th century biome changes occurred after extended periods in which temperature or precipitation departed one-half to two standard deviations from historical mean values. The biome change cases include upslope shifts of boreal conifer forest into alpine grassland in Scandinavia (Kullman and Öberg, 2009); temperate broadleaf forest into boreal conifer forest in the U.S. (Beckage *et al.*, 2008); temperate broadleaf forest into montane heathland in Spain (Peñuelas and Boada, 2003); and a southward shift of the Sahel (moist savanna replaced by dry bushland), Sudan (savannas moving into woodland), and Guinea (savannas moving into tropical deciduous forest) zones in Africa (Gonzalez *et al.*, 2012).

[INSERT TABLE 4-1 HERE
Table 4-1: Biome shifts in the past from published field research that examined trends in periods > 30 y of biomes in areas where climate, not land-use change or other factors, predominantly influenced vegetation. Derived from meta-analysis in Gonzalez et al. (2010). Pre-AR4 publications are included to provide a comprehensive review. Shift type: elevational (E), latitudinal (L), none detected (N). Biomes (and abbreviations), from poles to equator: tundra and alpine (UA), boreal conifer forest (BC), temperate conifer forest (TC), temperate broadleaf forest (TB), temperate shrubland (TS), tropical grassland (RG), tropical woodland (RW). Rate of change in temperature and fractional rate of change in precipitation are derived from linear least squares regression of 1901-2002 data (Mitchell and Jones, 2005; Gonzalez et al., 2010). Climate trends indicate general regional changes at 50 km spatial resolution because the references do not give uniform site-specific climate data to compare across locations; * rate significant at $P \leq 0.05$.]

[INSERT FIGURE 4-2 HERE
Figure 4-2: Biome shifts in the past at sites in Table 4-1, derived from meta-analysis in Gonzalez *et al.* (2010). Temperature change is the rate from linear least squares regression of 1901-2002 temperatures (Mitchell and Jones, 2005; Gonzalez *et al.*, 2010). Arrows indicate general direction of shifts.]

Many documented cases of biome shifts consist of movement of mountain and polar treelines into alpine grassland or tundra. Use of tree rings to reconstruct tree and shrub age classes back to the 18th century has detected biome shifts in the Canadian Rockies (Luckman and Kavanagh, 2000), European Alps (Leonelli *et al.*, 2011) the North American tundra (Payette, 2007; Suarez *et al.*, 1999), Siberia (Kirdyanov *et al.*, 2012), and the Urals (Devi *et al.*, 2008). In a meta-analysis of 20th century treeline studies (Harsch *et al.*, 2009), half the cases showed treeline advance consistent with the observed change in climate and only 1% showed recession, with advances more common with treelines of diffuse form (as opposed to krummholz form). Of 23 published cases that used field data to examine trends in biome location over periods > 30 y, 19 detected shifts consistent with climate change and 16 of these occurred in areas of significant 20th century temperature or precipitation change. A detection and attribution analysis in the African Sahel attributed the biome shift to anthropogenic climate change, rather than local deforestation or soil factors (Gonzalez *et al.*, 2012).

Projected Biome Shifts

Projections of potential future vegetation indicate substantial susceptibility of ecosystem to biome shifts (Table 4-2, Figure 4-3). Five dynamic global vegetation models (DGVMs) and one equilibrium climate model project biome changes on 5-30% of global land from ~1990 to 2100 for a range of the CMIP3 GCM runs of the IPCC SRES emissions scenarios (Scholze *et al.*, 2006; Alo and Wang, 2008; Sitch *et al.*, 2008; Gonzalez *et al.*, 2010; Bergengren *et al.*, 2011). A 5-30% change of biome is in a similar order of magnitude to the 25% change in global land cover from ‘natural’ to ‘cultivated’ over the period 1700 to 2000 (Ellis *et al.*, 2010). Time lags between greenhouse gas emissions, changes in climate and vegetation response commit ecosystems to change long before responses become manifest (Rosenzweig *et al.*, 2008; Jones *et al.*, 2009).

Projections generally agree on extensive poleward shifts of vegetation and degradation of tropical biomes, although the exact spatial distributions of various biomes differ substantially due to differences in GCMs, emissions scenarios, and vegetation models (Figure 4-3). Tundra, alpine, and boreal conifer biomes show the highest vulnerability to biome shifts, due to high exposure and high sensitivity to warm temperatures. Temperate mixed forest shows high vulnerability due to projected loss of coniferous species and thus conversion to temperate broadleaf forest. In contrast, tropical evergreen broadleaf forest (with the exception of the Amazon –see Box 4-4) generally shows low vulnerability to biome shifts. The resilience of other rainforests derives from high temperature tolerances and mitigation of water stress by increases in equatorial precipitation (Malhi *et al.*, 2008) as well as the wide latitudinal ranges of woody plant species (Weiser *et al.*, 2007). Regional vegetation modeling efforts (Table 4-2; Jones *et al.*, 2009; Scheiter and Higgins, 2009; Shuman *et al.*, 2011; Hickler *et al.*, 2012) confirm the above global findings.

[INSERT TABLE 4-2 HERE
Table 4-2: Projected biome shifts from spatial analyses published since AR4 of potential climate change impacts on vegetation biogeography. The principal AR4 biome projection analysis (Scholze et al., 2006) is included for comparison. Gonzalez et al. (2010) conducted analyses on equal-area spatial data; others did not analyze pixels of equal area. Bergengren et al. (2011), Gonzalez et al. (2010), and Scheiter and Higgins (2009) directly reported biome change as fraction of land area; other biome change fractions are estimates from authors’ maps. Analyses give changes from ~1990 to 2100, except for Heubes et al. (2011), who projected changes from 2000 to 2050.]

[INSERT FIGURE 4-3 HERE
Figure 4-3: Projected biome changes from analyses listed in Table 4-2. Dashed lines connect maxima and minima of ranges of estimates of a projection.]

_____ END BOX 4-1 HERE _____

4.2.2. What the Paleoecological Record can and cannot Tell Us

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

Rapid, regional warming before and after the Younger Dryas cooling event (11.7-12.9 ka) provides a relatively recent analogy for climate change of a speed that approaches that projected for the 21st century (Alley *et al.*, 2003; Steffensen *et al.*, 2008). Ecosystems and species responded rapidly during this period by shifting distributions and abundances, and there were some notable large animal extinctions, probably exacerbated by humans activities (Gill *et al.*, 2009; Dawson *et al.*, 2011). In some regions, species got locally or regionally extinct, but there is no evidence for climate-driven large-scale extinctions during this period of rapid climate change (Botkin *et al.*, 2007; Willis *et al.*, 2010c). However, these climate excursions differ from those projected for the future because they were regional rather than global, may have only regionally exceeded rates of warming projected for the future, and started from very cold baseline (Alley *et al.*, 2003). The mid-Holocene around ca. 6 ka provides a very recent example of the effects of modest climate change, because regional warming during this period (ca. 0.5-1.5°C above pre-industrial temperatures in some regions) was the same order of magnitude as the warming the Earth has experienced over the last century. Ecological effects were small compared to periods with larger climate excursions, but even this small degree of warming was characterized by frequent fires in a drier Amazon (Mayle and Power, 2008), development of lush vegetation and lakes in a wetter Sahara (Watrin *et al.*, 2009), temperate deciduous forests in Europe expanding further north and up to higher elevations (Prentice *et al.*, 1996), and large-scale migration of Boreal forest into a warmer tundra (Jackson and Overpeck, 2000). However, there are no really exact analogs for future climate change: none of the well-studied past periods of large climate change involved simultaneously the rates, magnitude and spatial scale of climate change projected for the next century and beyond (Jansen *et al.*, 2007; Schulte *et al.*, 2010; IPCC AR5 Chapter 5). Direct analogy with the paleoecological record is also unwarranted because future climate change will interact with other global changes such as land use change, invasive species, pollution and overexploitation of natural resources (Pereira *et al.*, 2010). There is high confidence that these interactions will to be important: the paleoecological record illustrates that exploitation by humans helped drive many large mammal species to extinction during periods of climate change in past (medium confidence; Lorenzen *et al.*, 2011).

It has been demonstrated that state-of-the-art vegetation models are able to simulate much of the biome-level equilibrium response of terrestrial to large paleoclimate change (Prentice *et al.*, 1996; Salzmann *et al.*, 2008; Prentice *et al.*, 2011). These same models predict large shifts in biomes and species when applied to 21st century climate change (Sitch *et al.*, 2008, Cheaib *et al.*, 2012). Thus, the paleoecological record, and models that have been tested against that record provide a coherent message that biomes will move, species mixtures will change, and novel new plant communities will emerge in response to changing and often novel future climates (Williams and Jackson, 2007; Williams *et al.*, 2007b; MacDonald, 2010; Prentice *et al.*, 2011). As such, the paleoecological record and models indicate that it will be difficult 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 (high confidence; Jackson and Hobbs, 2009; Hickler *et al.*, 2012).

There remains much to be learned from the paleoecological record, since much of the complex, time-dependent change at regional scales has not 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.*, 2009; Harrison and Goni, 2010), and this same record thus provides a critical model evaluation opportunity that should be more thoroughly exploited to gain confidence in time-dependent simulations of future change, particularly given the complex role that interacting climate change and vegetation disturbance has played in the past (e.g., Marlon *et al.*, 2009; Jackson *et al.*, 2009; Williams *et al.*, 2009; Daniau *et al.*, 2010; Dawson *et al.*, 2011).

The paleoeclimatic record also reveals that past radiative climate forcing change was slower than that anticipated for the 21st century (IPCC AR5 Chapter 12), but even these slower changes often drove surprisingly abrupt, or non-linear, regional change in terrestrial and aquatic systems (e.g., Harrison and Goni, 2010; Williams *et al.*, 2011), as did even slower orbitally-driven change during the most recent Holocene interglacial (e.g., Booth *et al.*, 2005; Kropelin *et al.*, 2008; Williams *et al.*, 2010; 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 larger, more rapid changes. State-of-the-art climate and Earth system models are unable to simulate the full range of abrupt change observed in the past, and thus there will be unanticipated abrupt changes in climate and associated ecosystem impacts in the future (medium confidence; Leadley *et al.*, 2010; Valdes, 2011; IPCC AR5, Chapter 5).

4.2.3. Landscapes and Social-Ecological Systems

Ecosystems do not exist in isolation from one another or from human systems. The terrestrial surface can be visualised as being seamlessly covered by landscapes, each of which consists of a set of interacting ecosystems (such as forests, grasslands, croplands, rivers, lakes and human settlements). All landscapes experience some degree of interaction with people: they may live within them, use products and ecosystem services derived from them, manage them, disturb their processes, or alter their environments. The intensity, pattern and type of engagement with ecosystems by people varies according to many factors, including population density, affluence, culture, markets, policies and the ecological features of the landscape, among others. The responses of terrestrial and freshwater ecosystems to climate change are therefore mediated by human activities at a range of spatial and temporal scales, and to varying degrees. Together, the social and ecological elements form a linked system (a ‘social-ecological system, SES) in which the degree of coupling can range from very close to slight (Berkes *et al.*, 2003). It is the SES which is the unit of response to climate change.

4.2.4. Multiple Stressors and the Role of Climate Change

While isolated effects of the main drivers of global change (e.g. climate change, habitat loss, chemical pollution and biological invasions) are increasingly well documented (Millennium Ecosystem Assessment, 2005b; Settele *et al.*, 2010a) there is much less knowledge about their consequences when acting in combination. Multiple stressors can act in a non-additive way (Settele *et al.*, 2010b), potentially invalidating findings and interventions based on single-factor analysis.

Climatic and non-climatic drivers of ecosystem change must be distinguished if the joint and separate attribution of changes to their causes is to be performed – of paramount importance if the processes of change are to be understood and future changes are to be predicted. Within this chapter we elaborate on drivers and stressors which often act in concert with climate change, ranging from a high degree of relatedness to climate change (rising CO_2; tropospheric ozone; total, diffuse and UV radiation), over intermediate (land use change) to low degrees (invasive species; chemicals in land use and N deposition).

4.2.4.1. Land Use and Cover Change (LUC)

Land use continues to change worldwide, as a result of agriculture, bioenergy production, changing technologies, policies, markets, urbanization and lifestyles, among other factors. Land use change, leads to cover change, along with changes in climate and other environmental drivers such as nutrient loading, pollution and disturbance regimes. Land cover change, acting through habitat loss and fragmentation, remains the largest cause of contemporary terrestrial and freshwater biodiversity loss (Millennium Ecosystem Assessment, 2005a). Many of the CMIP5 Earth System Model projections include scenarios of anthropogenic land cover change, which are the dominant cause of large-scale vegetation change in the RCP scenarios. At the global scale, the effects of climate change on the global distribution of vegetation cover is secondary to land use change in these projections.

Conversion of forests and woodlands to annual and perennial agriculture, grazing pastures and commercial plantations are the most frequent land use changes in tropical and subtropical areas of Asia, Africa, Oceania and South America (Table 4-3). Afforestation at the expenses of agricultural lands is more frequent in Europe (Rounsevell and Reay, 2009; Schwaiger and Bird, 2010)

[INSERT TABLE 4-3 HERE
Table 4-3: Summary of climatic impacts of land use/land cover changes across continents.]

The relationships between climate change, land use change and land cover change is difficult to tease apart, since all three generally occur at the same time and interactively. Climate change affects the productivity of land, which leads to land-use change. Land cover change contributes to climate change through biophysical effects and changes in the sources or sinks of greenhouse gases, land use change through greenhouse gases emission changes. Conversion of forests to cropland, pasture or degraded land is usually suggested to cause warming, due to the reduction in evapotranspiration, but this need not be the case. Four out of five global climate models suggested local summer cooling effects following forest conversion to croplands and pastures (Pitman *et al.*, 2009).

Equilibrium experiments with the Geophysical Fluid Dynamics Laboratory's climate model (GFDL; Findell *et al.*, 2007) concluded that non-GHG mediated climatic impacts resulting from land use change were relatively minor overall, but significant in few regions (Eastern Europe, Northern India, and Eastern China). More substantial effects were projected by van der Molen *et al.* (2011) who found a strong equator to pole gradient of modeled climate response to land cover change, from almost null in the tropics, increasing towards the Northern Hemisphere mid-latitudes (Figure 4-4). This gradient results from damping feedbacks in the tropics due to a regional decrease in total cloud cover, rather from polar amplification. The damping feedback in the tropics is due. This gradient results from damping feedbacks in the tropics due to a regional decrease in total cloud cover, rather from polar amplification. A decrease in radioactive forcing was suggested in areas that switched to cropland in the south western United States of America, due to increased snow albedo (Mishra *et al.* (2010b), also increasing runoff and baseflow by 8 and 6 mm respectively and decreasing evapotranspiration by 15 mm. The same processes (a snow-vegetation-albedo feedback) operates in reverse in high latitudes leading to warming due to vegetation density increases (Levis, 2010).

[INSERT FIGURE 4-4 HERE
Figure 4-4: Spatial distribution of the sensitivity of mean temperature response changes in albedo resulting from land cover change. Albedo change and temperature response were computed on a monthly basis and displayed as annual mean. Areas where the albedo change is less than 0.002 are masked out. b) Zonal average climate response over land: the blue line represents the zonal average of the numbers displayed in (a), and the red line represents the ratio of the zonal average change in temperature to the zonal average change in albedo. (van der Molen *et al.*, 2011).]

The effects of land cover change on precipitation are variable. Many studies report rainfall decreases, soil moisture decreases and changes in the rainfall pattern (e.g. McAlpine *et al.*, 2009, Mendelsohn and Dinar, 2009; Eliseev and Mokhov, 2011). Numerical experiments showed land use-driven radioactive forcing change or about 0.11 W m^{-2} to be result from conversion to agriculture, causing warming and rainfall decreases in subtropics (Eurasia, North America, Amazonia, Central Africa) but rainfall increase in east China (Eliseev and Mokhov, 2011).

In a recent metanalysis, Don *et al.* (2011) found soil organic carbon losses from 9 to 30% in tropical regions because of forest conversion in croplands, grasslands and secondary forests. Afforestation or reforestation is often recommended to promote carbon sequestration. However, this does not necessarily result in climate cooling. A slight warming effect over the very long term (250 years) is predicted by Schwaiger and Bird (2010) because the warming effect of albedo changes neutralize cooling effects due to carbon sequestration. Reforestation may initially increase GHG emissions under some circumstances (Mendelsohn and Dinar, 2009).

Decreasing trends in potential evaporation by agriculture were accelerated in China from 1956 to 2005 for stations in regions with significant agricultural influences (Han *et al.*, 2012). In the arid and semi-arid regions of China the decreasing trend was mainly due to the aerodynamic term in the potential evaporation equation (which includes temperature, humidity, wind speed and stomatal conductance) while in the humid and sub-humid regions, it was mainly due to decreased net radiation. In both regions agriculture conversion caused a significant increase in wind speed.

Changes in albedo, nor any other biophysical effects and some indirect greenhouse effects (such as changes in soil erosion), are not currently included in the formal rules for accounting for the climate effects of land use activities (Schwaiger and Bird, 2010; Kirschbaum *et al.*, 2012).

_____ START BOX 4-2 HERE _____

Box 4-2. Uncertainties in Future Land Use Pathways

Growth in demand for food, feed, fibre, and fuel has accelerated in recent years, placing new pressures on terrestrial and freshwater ecosystems and outpacing the growth in supply. The principal driver of this acceleration in demand is the rate at which per capita consumption is growing in emerging economies. Policy shifts in OECD countries favoring the expansion of biofuel production, sometimes at the expense of food crop production, have contributed to this acceleration (Lapola *et al.*, 2010; Searchinger *et al.*, 2008). Growth in demand that outpaces growth in supply, climate-related crop failure (Lobell *et al.*, 2011), declines in the growth of agricultural productivity (from 3% per year in the period 1960-1990 to to 1% per year 1990-2010) and possibly a shortage of arable land that is not already under cultivation, especially in the temperate zone (Lambin and Meyfroidt, 2011) are all contributing to a rise in commodity prices that is likely to persist through 2020 (OECD/FAO, 2010) and growing pressure on remaining native ecosystems on soils that are potentially suitable for cultivation.

Emerging economies start from a relatively low per capita consumption base. The growth of these economies will therefore play an important role in future land use trends. Deforestation in developing countries is now correlated with the export of agricultural commodities (DeFries *et al.*, 2010), providing an early symptom of the growing role of agricultural trade with emerging economies as a driver of land-use change. The trajectory of future growth in demand for land- and freshwater-based production remains uncertain, since it depends on economic trends and policies, which are themselves dependent upon complex political and social processes. One of the uncertainties about future land use trends is climate policy. Deforestation rates in the Brazilian Amazon declined 68% (as of 2011) below its ten-year average (INPE, 2012), the result of policy and market signals (Soares-Filho *et al.*, 2010); this single trend represents a 1.5% reduction in global anthropogenic carbon emissions.

Future increases in the supply of land- and freshwater-based products may be achieved through increases in yield and through increases in the area devoted to production. The former is often associated with increases in water use, the application of fertilizers and other agro-chemicals, and in nitrogen loading associated with livestock confinement. The latter implies conversion of natural and semi-natural ecosystems to cropland, grazing land, and tree plantations, or intensified management of natural ecosystems. Suitable land for agricultural expansion is concentrated today in South America, Africa, and Eastern Europe (Lambin and Meyfroidt, 2011), and much of this land currently supports natural ecosystems. Most of the expansion of agricultural, livestock, and tree-based production by 2030 is likely to take place in South America (led by Brazil) and Africa (Lambin and Meyfroidt, 2011), and could drive the conversion of forests and savannas to agriculture and forestry, and changes in the fire regime.

In the CMIP3 climate projections assessed in AR4, only two climate models included the biophysical effects of anthropogenic land cover change as a climate forcing (Meehl *et al.*, 2007). In AR5, however, Working Group 1 assesses projections from models following the CMIP5 protocol, which includes scenarios of land use and land cover consistent with the scenarios of greenhouse gas emissions under the Representative Concentration Pathways (Moss *et al.*, 2010; Hurtt *et al.*, 2011).

The 4 main RCPs used for future climate projections are named according to the global mean radiative forcing projected by the end of the 21st Century, and the Integrated Assessment Model (IAM) used to generate them: RCP8.5-MESSAGE (Riahi et al., 2011); RCP6.0-AIM (Masui et al., 2011); RCP4.5-GCAM (Thomson et al., 2010); and RCP2.6-IMAGE (Van Vuuren et al., 2011b). Each IAM generated spatially-explicit future land use scenarios consistent with the emissions scenario and the associated socio-economic scenario (Table 4-4). These were further processed to an internally-consistent spatial resolution and land cover classification by (Hurtt et al., 2011).

[INSERT TABLE 4-4 HERE
Table 4-4: Summary of drivers and outcomes of land use / cover scenarios associated with Representative Concentration Pathways (Hurtt *et al.*, 2011).]

In the RCPs, land cover change is driven by a number of socioeconomic drivers, some of which are related directly to climate policy although many are not. In scenarios that included climate policy, land cover change was influenced by such policy – however the nature of climate policy effects on land use differs between RCPs and/or IAMs, with some climate mitigation policy scenarios including conservation of primary vegetation as carbon sinks while others included replacement of primary vegetation with biomass and / or biofuel plantations.

The extent to which primary vegetation is replaced by secondary vegetation, crops or pasture varied between the RCPs, but there is no simple linear relationship between the extent of vegetation change and the level of radiative forcing (Figure 4-5). Larger reductions in primary vegetation cover were seen in RCP8.5 (the highest emissions scenario) due to a general absence of pro-active measures to control land cover change, but large reduction were also seen in RCP2.6 (an aggressive mitigation scenario with low emissions) due to widespread conversion of land to biomass and biofuel crops coupled with carbon capture and storage. Smaller reductions were seen in RCP6.0 and RCP4.5, with the latter involving conservation of primary forest or regrowth of secondary forest as a mitigation measure. Hence both the highest and lowest emissions scenarios were associated with greater extents of land cover change.

[INSERT FIGURE 4-5 HERE
Figure 4-5: 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 (KleinGoldewijk, 2001) and for scenarios of future land use / cover consistent with the Representative Concentration Pathways RCP8.5-MESSAGE. RCP6.0-AIM, RCP4.5 GCAM and RCP2.6-IMAGE out to 2100. Reconstructions and future scenarios were harmonized for internal consistency (Hurtt et al., 2011).]

Spatial patterns of the future land cover change in all RCPs were largely similar to patterns of historical change (Figure 4-6) as most future change is projected to be a continuation of past change. By definition, primary vegetation cover cannot increase on the timescales considered here, so in scenarios where cropland and pasture are projected to decrease, these are replaced with secondary vegetation. Both tropical and boreal forest regions are projected to undergo declining primary forest cover in all 4 RCPs, but in RCP6.0 total forest area remains approximately constant.

[INSERT FIGURE 4-6 HERE
Figure 4-6: Fractional cover of primary vegetation at 1850, 2005 based on satellite data and historical reconstructions (KleinGoldewijk, 2001) and at 2100 for scenarios of future land use / cover consistent with the Representative Concentration Pathways RCP8.5-MESSAGE ( ), RCP6.0-AIM, RCP4.5 GCAM and RCP2.6-IMAGE. Reconstructions and scenarios were harmonized for internal consistency (Hurtt et al., 2011).]

_____ END BOX 4-2 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, Galloway *et al.*, 2008). Human activities currently transform 160 TgN/year of nitrogen gas in the atmosphere into reactive forms of N that can be readily used by plants and microorganisms in land and in the ocean. This is close to the pre-industrial global flux of about 255 TgN/year from all natural sources, and increasing. The human-caused flow from land to the coastal oceans in rivers is about 50 TgN/y, additional to the estimated natural flux of 30 TgN/year. Many of the sources of additional reactive nitrogen share root causes with changes in the carbon cycle - such as increased use of fossil fuels and expansion and intensification of global agriculture - so N deposition, rising CO_2 concentrations and warming are increasing together at global scales (Steffen *et al.*, 2011). This large addition of reactive N into the biosphere can reinforce global warming (e.g., though production of N_2O associated with fertilizer manufacture and application), or diminish warming (e.g., by increasing C storage in many terrestrial ecosystems). The balance of these effects is difficult to estimate (Erisman *et al.*, 2011; WG1 chapter 6). Regional trends in N fluxs differ substantially: N fertilizer use and N deposition are stable or declining in some regions, such as Western Europe; but N deposition and its impacts on biodiversity and ecosystem functioning are projected to increase substantially over the next several decades in other regions, especially in regions dominated by rapidly-growing emerging economies (Galloway *et al.*, 2008).

Experiments and observations, most of which are in temperate and boreal Europe and North America, show a consistent pattern of increase in the dominance of a few nitrogen-loving plant species and loss of overall plant species richness at N deposition loads exceeding between 5 and 20 kgN/ha/year (Bobbink *et al.*, 2010; but see Stevens *et al.*, 2010). N deposition is currently above these limits in much of Europe, eastern North America, and Southern Asia (Galloway *et al.*, 2008), including in many protected areas (Bleeker *et al.*, 2011). Experiments examining the interactive impacts of N deposition and elevated CO_2 or climate change on natural systems are rare (Rustad 2008), but recent studies show that interactions are common, strong and complex (Thompson *et al.*, 2008; Langley and Megonigal, 2010; Gaudnik *et al.*, 2011; Hoover *et al.*, 2012; Eisenhauer *et al.*, 2012; but see Zavaleta *et al.*, 2003 for evidence of additive effects). For example, combinations of treatments mimicking future N deposition, elevated CO_2 concentrations and warming had larger negative impacts on pollinator populations than could be predicted from individual treatments (Hoover *et al.*, 2012). Analyses using the multi-factor biodiversity change model GLOBIO3 suggest that N deposition will continue to be a significant contributing factor to terrestrial biodiversity loss over the next several decades, but will be a smaller factor than climate change and much smaller 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 also foresee that N deposition impacts will be important, but climate change effects will begin to dominate over the next several decades (Belyazid *et al.*, 2011).

Since the AR4 report there has been a large increase in the number of studies using models, observations and experiments to understand and predict the interactive effects of N deposition, climate change and CO_2 on ecosystem function since. Broadly this research shows that ecosystem function is mediated by complex interactions between these factors, such that many ecosystem responses remain difficult to understand and predict (Norby and Zak, 2011; Churkina *et al.*, 2010). For example, experiments, observations and models suggest that the observed increase in productivity and carbon storage in forests in many parts of the world is due to combinations of N deposition, climate change, fertilization effects of rising CO_2, and forest management (Magnani *et al.*, 2007; Huang *et al.*, 2007; Pan *et al.*, 2009; Churkina *et al.*, 2010; Bontemps *et al.*, 2011; Bellassen *et al.*, 2011; de Vries and Posch, 2011; Eastaugh *et al.*, 2011; Norby and Zak, 2011; Shanin *et al.*, 2011; Lu *et al.*, 2012). Despite difficulties in quantifying the relative contributions of these factors and their interactions, it appears that the effects of N deposition and rising CO_2 have generally dominated in much of the Northern hemisphere. However, climate change is predicted to rival or supersede N and CO_2 as the key driver over the coming decades. Experiments in grasslands 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 (Ma *et al.*, 2011; Menge and Field, 2007), in part due to shifts in plant species composition (Langley and Megonigal, 2010) and the complex dynamics of coupled C,N andP cycles (Menge and Field, 2007; Niboyet *et al.*, 2011).

4.2.4.3. Tropospheric Ozone

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

A meta-analysis of over 300 articles addressing the effect of ozone on tree growth (Wittig *et al.*, 2009) - largely focussed on northern-hemisphere temperate and boreal species - concluded that current levels of tropospheric ozone are suppressing growth by 7% relative to pre-industrial levels. Based on experimental data and scenarios of future tropospheric ozone levels, further decreases of 11% and 17% relative to current growth rates are projected for 2050 and 2100. Modelling studies extrapolating measured dose-response functions suggest a 5 to 30% contemporary reduction in NPP worldwide (Sitch *et al.*, 2007) and 1-16% in temperate forests (Ainsworth *et al.*, 2012).

The mechanisms by which ozone affects plant growth are now better known (Hayes *et al.*, 2007; Ainsworth *et al.*, 2012) but remain largely based on a few crop species grown in temperate zones. Chronic exposure to ozone at levels above about 40 ppb reduces stomatal conductance and impairs the activity of photosynthetic enzymes (The Royal Society, 2008). Carbon assimilation rates are reduced and respiration increases, leaf area is reduced and leaf senescence accelerated - all leading to a reduction in NPP and biomass accumulation. Gymnosperms and grasses are less sensitive than angiosperms. Lower stomatal conductance should theoretically increase river runoff, although studies that measured runoff are contradictory on this issue (Wittig *et al.*, 2007, McLaughlin *et al.*, 2007; Mills *et al.*, 2009; Huntingford *et al.*, 2011a).

The negative effects of rising O_3 on NPP are large enough to largely cancel the projected increase in NPP due to elevated CO_2 but the possible interactive effects between CO_2 and O_3 are poorly understood. Reduced stomatal conductance widely observed under elevated CO_2 should help protect plants from ozone damage. Some chamber experiments (Bernacchi *et al.*, 2006) and model studies (Klingberg *et al.*, 2011) suggest this to be the case. The one plot-scale FACE study of CO_2 and O_3 interactions in a temperate forest (Karnosky *et al.*, 2005; Hofmockel *et al.*, 2011) suggests that the effects of O_3 and CO_2 are not independent and may partly compensate for one another. There is genotopic variation in sensitivity to O_3 (Ainsworth *et al.*, 2012). Other than changing cultivars or species, there is not believed to be much scope for management actions promoting adaptation to higher levels of O_3 (Teixiera *et al.*, 2011).

4.2.4.4. Rising CO_2

The issue of the effects of rising CO_2 on the global carbon cycle are discussed in chapter 7 of the WG1 report. The discussion here focussed on impacts on terrestrial ecosystems and inland water systems.

Since AR4, investigation of elevated CO_2 effects on plants and ecosystems has focussed mainly on Free Air CO_2 Enrichment (FACE) techniques, which have now been in use for approximately 20 years (Leakey *et al.*, 2009) and are considered more realistic than earlier approaches using enclosed chambers because the interaction with the atmosphere is more like that of natural systems. C3 plant species, which includes all tree species, show a general increase in photosynthesis under elevated CO_2, although the response varies between species. Although FACE experiments indicate lower responses of crop yields to elevated CO2 than chamber studies, FACE experiments suggest greater biomass production of trees (Leakey *et al.*, 2009). Despite a certain amount of acclimation ("down-regulation") under long-term exposure, stimulation of photosynthetic carbon assimilation due to elevated CO_2

persists at a level of 19 to 46% for 600 ppm CO_2 relative to 370 ppm. Palaeo records also indicate increased growth under higher CO_2 (Prentice and Harrison, 2009). Nitrogen use efficiency and dark respiration are also both increased under higher CO_2. Transpiration is decreased due to reduced opening of stomatal apertures which does not acclimate to higher CO_2, leading to greater water use efficiency – this is corroborated by additional evidence from studies of stable carbon isotopes (Barbosa *et al.*, 2010; Koehler *et al.*, 2010; Silva *et al.*, 2010; Maseyk *et al.*, 2011). C4 plant species, which includes some tropical grasses and some crops, undergo photosynthesis via a different biochemical pathway which is not directly affected by elevated CO_2. However, CO_2 rise generally increases water use efficiency of C4 plants which can exert indirect effects on growth. The influence of rising CO_2 on water use efficiency has greater impacts on water-stressed ecosystems.

Elevated CO_2 can assist in maintaining resilience of individual species against other stresses, such as high temperature and drought. It is also suggested that faster growth rates under higher CO2 can allow woody plants to become mature enough to withstand ground fires, and hence alter the fire/vegetation regime in rangelands (Bond and Midgley, 2000). It is also suggested that differential species responses to elevated CO_2 may be altering competition, increasing the likelihood of faster-growing species such as lianas out-competing slower-growing species such as trees (Lewis *et al.*, 2009a).

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 and as yet poorly predictable (Potvin *et al.*, 2007). For example, in one ten-year temperate grassland experiment in Minnesota, elevated CO2 halved the loss of species richness due to nitrogen addition (Reich, 2009), whereas no such change was reported for a similar experiment in annual grassland on serpentine soils in California (Zavaleta *et al.*, 2003) or an alpine grassland in France (Bloor *et al.*, 2010). Some of the mechanisms of community and production changes under multi-factor forcing may be mediated through changes in plant litter quality, although in the temperate grassland experiment described above the changes in litter decomposition rates were less marked than the changes in the chemical composition of tissues (Knops *et al.*, 2007). Hyvönen *et al.* (2007) review the combined effects of elevated CO_2, temperature, nitrogen deposition and management on carbon sequestration in temperate and boreal forests and conclude that findings extrapolated from experiments that only consider one of these factors in isolation can be misleading. Since FACE experiments either examine CO_2 effects in isolation, or in conjunction with a very limited set of imposed climate changes such as elevated temperature, the extent to which CO_2 responses are affected by a range of temperature changes or other climatic or environmental factors remains poorly-studied.

Inclusion of CO_2 effects can be a first-order influence on model projections of ecosystem and hydrological responses to anthropogenic climate change. For example, most CMIP3 climate models project a drying, warming trend in Amazonia, and this leads to widespread forest loss being simulated under most climate projections in a model that ignores CO_2 effects (Salazar *et al.*, 2007). However, when CO_2 effects are included, the same model projects forest loss under fewer climate projections (Lapola *et al.*, 2009). Nevertheless, a very severe regional climate scenario still leads to a simulated Amazon forest die-back even with CO_2 effects included (Betts *et al.*, 2004).

Impacts of CO_2 physiological forcing may extend beyond the organisms themselves – increased water use efficiency may significantly influence hydrological impacts. Since the influence of rising CO_2 on water use efficiency has greater impacts on water-stressed ecosystems, impacts of CO_2 on water use efficiency may become more important if drought becomes more prevalent. 21st-Century continental-scale runoff is projected by some models to either increase more or decrease less when CO_2-induced increases in water use efficiency are included in addition to climate change (Betts *et al.*, 2007; Wiltshire *et al.*, submitted), potentially reducing an increase in water-stress due to rising population or climate change (Wiltshire *et al.*, submitted), although other models project a smaller response Cao *et al.*, 2009. There are conflicting views on whether this effect is becoming significant already (Gedney *et al.*, 2006; Gerten *et al.*, 2008). Reduced transpiration under rising CO_2 may also affect future regional climate change itself (Boucher *et al.*, 2009) and may enhance the contrast between the land and ocean surface warming (Joshi *et al.*, 2008).

The current generation of dynamic global vegetation models (DGVMs; Sitch *et al.*, 2008), which includes those used within the CMIP5 Earth System Models, uses formulations based on experimental work that pre-dates FACE experiments (e.g., Farquhar *et al.*, 1980; Collatz *et al.*, 1992). However, the DGVMS simulate present-day global carbon budgets that agree with those inferred from observations such as the atmospheric CO2 record (Sitch *et al.*, 2008).

The effect of CO_2 on plant physiology independent of its role as a greenhouse gas has implications for the definition of "dangerous climate change" in terms of levels of global mean temperature rise, and for the use of "CO_2 equivalent" metrics for comparing difference greenhouse gases. Aside from uncertainties in the response to a given change in CO_2 concentration, the relative importance of radiative forcing and "CO_2 physiological forcing" depend on uncertainties in the response of global climate to radiative forcing and also on the relative proportions of CO_2 and other greenhouse gases. The impacts of, for example, a 2°C rise in global mean temperature may have a very different net impact on ecosystems depending on the change in CO_2 concentration accompanying this rise. 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 (Figure 4-7). Similarly, inertia in the climate system means that warming would be expected to continue for many decades after stabilisation of GHG concentrations, so the relative contribution of CO_2 physiological effects would diminish over time.

[INSERT FIGURE 4-7 HERE
Figure 4-7: Illustration of uncertainty in the CO_2 concentration associated with 2°C global warming. Frequency distribution of the mean atmospheric CO_2 concentration associated with a simulated 2°C global warming of above pre-industrial in an ensemble of 58 variants of the HadCM3C Earth System Model, for the RCP2.6 scenario of global emissions peaking and declining with the aim of limiting warming to near 2 K. "Frequency" refers the number of model variants first reaching 2 K global warming at the same time that their simulated atmospheric CO_2 concentration reaches the values shown, after smoothing of temperature variability (Betts *et al.*, in press).

Comparison of the effects of CO2 with other GHGs in terms of the current radiative forcing-based metrics may therefore not be representative of the relative effects of different GHGs on ecosystems and hydrology Huntingford *et al.*, 2011b.

There are, however, still important limitations on understanding of CO_2 responses. Large-scale FACE experiments have still only been conducted for up to two locations for any particular ecosystem type, and there is still no tropical FACE experiment. The scale of controlled experiments limited to approximately $100m^2$, and extrapolation to larger scales may not be appropriate as other constraints may be important. For example, a tree-ring study Girardin *et al.*, 2011 indicate that doubling CO_2 leads to a maximum of 14% increase in growth in Canadian boreal forests due to CO_2 fertilization. Similarly, the limited size of FACE experiments means that influences on catchment-scale hydrology have not yet been studied experimentally. Hyvönen *et al.*, 2007 review the combined effects of elevated CO_2, temperature, nitrogen deposition and management on carbon sequestration in temperate and boreal forests and conclude that findings extrapolated from experiments that only consider one of these factors in isolation can be misleading. The majority of FACE experiments examine CO_2 concentrations approximately double pre-industrial (approximately 600ppmv), which has been projected to be reached by the end of the 21st Century under a number of scenarios including RCP6.0-AIM van Vuuren *et al.*, 2011a, whereas higher concentrations are projected under higher emissions scenarios; RCP8.5-MESSAGE reaches a CO_2 concentration of approximately 900ppmv by 2100 van Vuuren *et al.*, 2011a. The extent to which photosynthesis and nitrogen and water use efficiency continue to increase with ongoing CO_2 rise therefore remains poorly-constrained.

4.2.4.5. Diffuse versus Direct Radiation

The quantity and size distribution of aerosols in the atmosphere alters both the amount of solar radiation reaching the Earth's surface and its distribution between direct radiation and diffuse radiation. There are observed trends in both quantities in many parts of the world, usually in the direction of overall 'dimming' of around 30 W m^{-2}, with an accompanying increase in diffuse radiation of up to 20 W m^{-2} (Kvalevåg and Myhre, 2007). For a constant total radiation, an increased fraction received as diffuse radiation theoretically increases net photosynthesis, since more leaves are illuminated at levels below which saturation sets in leaves exposed to direct solar radiation (Knohl and Baldocchi, 2008; Kanniah *et al.*, 2012). In a DGVM which included this process, an increase in diffuse fraction of solar radiation due to volcanic and anthropogenic aerosols and cloud cover was simulated to lead to approximately a 25% increase in the strength of the global land carbon sink between 1960 and 19999 (Mercado *et al.*, 2009). However, under a future scenario of climate change and decreased anthropogenic aerosol concentration, this enhancement of the land carbon sink was simulated to decline to near zero by the end of the 21st Century (Mercado *et al.*, 2009). This influence on plant growth and the land carbon budget is a potentially important unintended consequence of solar radiation management schemes that involve the injection of aerosols into the stratosphere to reduce radiant forcing (Boucher *et al.*, in preparation).

4.2.4.6. Alien Species

Invasive alien species are having increasingly negative impacts on human wellbeing, production systems and biodiversity (high confidence; Brook, 2008; Burton *et al.*, 2010; McGeoch *et al.*, 2010). Climate change will exacerbate some invasion impacts and ameliorate others (Peterson *et al.*, 2008; Bradley *et al.*, 2009; Britton *et al.*, 2010). In most cases climate change is increasing the likelihood of the establishment, growth, spread and survival of invasive species populations (Daehler, 2003; Hellmann *et al.*, 2008; Dukes *et al.*, 2009; Walther *et al.*, 2009; Bradley *et al.*, 2010; Huang *et al.*, 2011; Chown *et al.*, 2012). Warming has enabled species to extend their distributions into areas in which they could previously not survive and reproduce, including alien and invasive plants, invertebrates, fish, birds and other taxa (high confidence; Walther *et al.*, 2009; Kleinbauer *et al.*, 2010). Since the IPCC AR4, the number of observations for the spread and/or establishment of alien species attributed to climate change has increased for particular taxa, as well as for particular areas, including polar regions (e.g., true bugs in Japan: Musolin, 2007; aquatic organisms: Rahel and Olden, 2008; different plant and animal taxa globally: Walther *et al.*, 2009; Smith *et al.*, 2012; Masters and Norgrove, 2010; different insect taxa globally Robinet and Roques, 2010; vascular plants on mountain tops: McDougall *et al.*, 2011, vascular plants in the Antarctic: Chown *et al.*, 2012). The risk of alien species establishment is high and increasing substantially in ice-free areas of Antarctica, particularly those areas of the continent that are experiencing warming (high confidence; Chown *et al.*, 2012).

Studies of future climate projections suggest spread and/or establishment of many alien species (e.g. forest insects and pathogens in North America: Dukes *et al.*, 2009, different plant and animal taxa in Norway: Gjershaug *et al.*, 2009) but also show range contractions of alien plants (Bradley *et al.*, 2009).

Positive interactive effects between climate change and invasive species will be experienced in some cases, such as increases in resources for threatened species (Caldow *et al.*, 2007), forest structural recovery (Bolte and Degen, 2010) and available biomass for timber and fuel (O'Donnell *et al.*, 2012). The effect of invasions on net changes in carbon stocks are situation specific, and may be either positive or negative (Williams *et al.*, 2007a). Rising CO2 levels will increase the growth rates of most invasive species (Mainka and Howard, 2010), but the effectiveness of invasive species management for sequestering carbon is uncertain and context specific (Peltzer *et al.*, 2010). Longer term, indirect effects of invasive species are likely to be more important than direct, short-term effects, as a result of changes in soil carbon stocks and tree community composition (low-medium confidence; Peltzer *et al.*, 2010). The expansion of alien species in some areas and contraction in others will contribute to community re-organisation and the formation of novel communities and interactions in both terrestrial and freshwater habitats (high confidence; e.g., Peterson *et al.*, 2008; Rahel and Olden, 2008; Britton *et al.*, 2010; see also chapter 4.3.2.5.).

Invasive species are more likely than native species to have traits that favour their survival and reproduction under changing climates; they tend to have faster growth rates and are particularly likely to be favoured in non-resource limited environments (medium to high confidence; Daehler, 2003; Chown *et al.*, 2007; Leishman *et al.*, 2007; Colautti *et al.*, 2010; Buswell *et al.*, 2011; Clements and Ditommaso, 2011; Davidson *et al.*, 2011; Zerebecki and Sorte, 2011; Willis *et al.*, 2010a). Invasive plants tend to use more water than native species at some scales, and on average have higher overall metabolic rates, foliar nitrogen concentrations and photosynthetic rates (Leishman *et al.*, 2007). Higher evapotranspiration rates and carbon sequestration are particularly evident in invasive species-dominated systems in warmer, wetter climates where invasive species change the dominant growth form from herbaceous to woody (low confidence; Cavaleri and Sack, 2010). Water use will be greater in dry climates depending on the individual traits of the invasive plant species concerned (low confidence; Farley *et al.*, 2005; Cavaleri and Sack, 2010).

Threat syndromes are formed by synergistic interactions between climate change and invasive species, along with landscape change, habitat disturbance and human-facilitated breakdown of dispersal barriers (Angeler and Goedkoop, 2010; Bradley *et al.*, 2010; Brook, 2008; Winder *et al.*, 2011). Climate change and invasive species are highly likely to change the risk and properties of fire and the interaction is being reported more frequently. This is a direct result of increased fire risk under higher temperatures and increased plant biomass as a consequence of plant invasion (high confidence; Brook, 2008; 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 water temperature, water properties and water demand (Rahel and Olden, 2008; Britton *et al.*, 2010), with riparian habitats being particularly vulnerable (medium confidence; Schnitzler *et al.*, 2007).

Generalist, domesticated alien pollinators are likely to be favoured over native specialist pollinators, are likely in some instances be able to compensate functionally for the loss of native pollinators (net effect uncertain; Potts *et al.*, 2010; Schweiger *et al.*, 2010). The number of alien weeds, insects and pathogens in production systems is increasing (medium confidence). A range of climate change-related variables (extreme events, changes in precipitation, temperature and CO2) are likely to exacerbate the establishment and spread of pests, vectors and pathogens and negatively impact on crop production (Potts *et al.*, 2010; Robinet and Roques, 2010; Ziska *et al.*, 2011; Clements and Ditommaso, 2011). Warming has led to the spread of many invasive insect species such as the mountain pine bark beetle and resulted in forest destruction (high confidence; Kurz *et al.*, 2008). Management strategies will become less effective as a consequence of decoupling of biocontrol relationships and less effective mechanical control as biomass of invasive species increases (low-medium confidence; Hellmann *et al.*, 2008).

4.3. Vulnerability of Terrestrial and Freshwater Ecosystems to Climate Changes

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

4.3.1. The Importance of Changes in Disturbance Regime

The species composition at a given location is determined by three factors: the physiological tolerance of the species in relation to the range of conditions experienced at the site; the interactions with other species, including competition, predation, pests and mutualists; and whether the species have been able to reach the location in evolutionary or recent times . The average environmental conditions are often less of a determinant of species ranges than the extremes, such as the occurrence of exceptionally cold or hot days, or droughts exceeding a certain duration . The changes in probability of such extremes is typically disproportionately larger than the relative change in the mean (see IPCC, 2012, but also (Diffenbaugh *et al.*, 2005). Competitive exclusion would tend to lead to a small set of dominant species if it were not for occasional disturbances that relieve the competitive pressure and create opportunities for marginalized species. Fire, floods and strong winds are all examples of biodiversity-sustaining events, provided that their frequency and intensity does not deviate greatly from the regime that prevailed at the location in evolutionary time. Although non-endemic species are more easily transported to new locations for which their physiology is adapted in the contemporary, interconnected world than in the past, their establishment usually requires a disturbance that weakens competition from pre-established species.

It is thought that widespread compositional shifts resulting from the combined effects of global change forcings will be relatively abrupt and associated with changes in the disturbance regime, rather than reflecting a gradual and continuous expansion of ranges and changes in abundance. A ‘disturbance regime’ refers to the totality of different types of disturbance events in a system, each characterised by their probability-intensity function and other relevant attributes, such as their seasonal distribution.

4.3.2. Evidence of Change in Ecosystems

[*Text will be added here to i) explain the importance of the various indices of change and why they were chosen, ii) relationships between the various indices and iii) highlight some of the major uncertainties, primarily by referring to the Detection and Attribution box*]

_____ START BOX 4-3 HERE _____

Box 4-3. Detection of Change in Ecosystems and Attribution to its Causes

[INSERT FIGURE 4-8 HERE

Figure 4-8: Confidence in Detection and Attribution of observed responses of terrestrial ecosystems to climate change. [Draft - confidence levels have been derived based on expert judgment of the available literature following the IPCC uncertainty guidance (Mastrandrea *et al.*, 2010). This analysis will be revised based on broader input from the scientific community. [*The points in the figure represent global and cross-taxa assessments; the positioning might still be changed especially for specific taxa as well as for specific regions]*.]

Confidence in detection of change is considered to be very high when there is high agreement between many independent sources of evidence (e.g., between ground observations and remote sensing, models and observations, experiments and observations, etc.) and where there is robust evidence that the changes are outside of their natural range of variation (see WGII, Chapter 18). Confidence in attribution to climate change is very high when changes correspond to a sound mechanistic understanding of responses to climate change; where time series of observations are sufficiently long to detect trends correlated with climate change; and where confounding factors can be accounted for or are of limited importance. It is also important to account for the spatial distribution of climate change trends: for example, warming trends are very high in the Arctic and small or negative in parts of South America (Burrows *et al.*, 2011). Statements of confidence for detection and attribution are given without references, as detailed traceability is provided in the sub-sections that follow.

This analysis of detection and attribution does not cover projected future changes in ecosystems. The absence of observable changes thus far does not necessarily alter confidence in projections of large change in the future because 21st century climate change is projected to substantially exceed changes experienced over the last century, and because ecosystem responses to climate change may be non-linear and may appear after considerable time lags.

Phenology (see 4.3.2.1) - There is high confidence in the detection of shifts in phenology, i.e., the timing of key life history events such as flowering in plants or nesting for birds, and in the attribution of these shifts to climate change. Since the AR4 report there has been a significant increase in the spatial, temporal and taxonomic coverage through ground-based phenological observation networks, data mining and remote sensing. These observations show that phenology for many, but not all species has shifted over the last decades to centuries. Confidence in the attribution of these temporal shifts in phenology to climate change is high. Simple attribution of temporal shifts in phenology to qualitative expectations due to global warming (e.g., Parmesan and Yohe, 2003) has given way to more refined analyses and models that account for regional differences in warming trends, urban heat island effects, confounding effects of other global change drivers, non-linear responses of phenology to warming, etc. These improvements in understanding since the AR4 report notwithstanding, there remains a substantial fraction of species that have not responded in a predictable manner to climate change and a wide range of confounding factors that limit confidence in attribution of phenological shifts to climate change.

Primary Productivity (see 4.3.2.2) & Biomass and C stocks (see 4.3.2.3) - Evidence has accumulated from a wide range of sources including eddy flux towers, inversion of atmospheric CO_2 concentrations, remote sensing and models that, once the effects of deforestation are accounted for, terrestrial ecosystems are currently net sinks for carbon over much of the Northern hemisphere and parts of the Southern hemisphere. Measurements of increased tree growth over the last several decades, a large sink for carbon, are coherent with this. Confounding factors such as N deposition, afforestation and land management make attribution of these trends to climate change difficult. Most studies speculate that rising CO_2 concentrations are contributing to this trend, but there is no clear, consistent signal of a climate change contribution.

Transpiration (see 4.3.2.4) - Transpiration is now being monitored at a variety of scales ranging from the individual plants to large regions. A variety of studies suggest that transpiration increased over the period 1970-1990, but there is substantial uncertainty in this estimate due to low agreement across studies. This change has been attributed to global warming, and the absence of a continued trend in from the 1990's onward has been attributed to increased water stress. However, confidence in attribution is low because of the large number of confounding factors that influence transpiration including N deposition, land management, irrigation, etc.

Species extinctions (see 4.3.2.5) - There is high confidence that species extinctions are at or above the highest rates of species extinction in the fossil record. However, only a small fraction of observed species extinctions have been attributed to climate change — most have been ascribed to invasive species, overexploitation or habitat loss or modification. For those species where climate change has been invoked as a causal factor in extinction, there is little agreement among investigators concerning the importance of climate variation in driving extinction and even less agreement that extinctions were caused by global warming. Therefore, confidence in the attribution of extinctions to climate change is very low.

Species distributions (see 4.3.2.5) - Depending on the species and species group, there is low to high confidence in the detection of species range shifts and that these range shifts can be attributed to climate change. The number of species studied has considerably increased since the AR4 report. Meta-analyses show that many arthropods have moved large and statistically significant distances towards the poles (many 10's of km) over the last several decades. In contrast, range shifts in other species groups such as plants are only beginning to be detectable. There is a wide range of evidence from models, field studies, and quantitative comparisons between observed and expected range shifts that some species and species groups, especially arthropods, have moved due to climate change. However, range shifts in many other species and species groups are more difficult to attribute to climate change because the climate change 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.

Tree mortality (see 4.3.3.1) - Tree mortality appears to be increasing globally, but there is low confidence in detection due to a lack of temporal and spatial coverage of tree mortality. In particular, the increase in reporting of mortality may reflect greater observation effort rather than a real increase in mortality. Tree mortality can clearly be linked directly to climate impacts, especially heat and drought. However, attribution to directional climate change is difficult due to lack of time series measurements for a large range of forested systems and because areas were tree mortality is occurring do not necessarily correspond to regions with high rates of recent warming (Burrows *et al.*, 2011).

Tundra regime shift (see 4.3.3.4) - Detection - Both field and satellite measurements indicate an increase in shrub growth and permafrost melting in many areas of the Arctic tundra. Lack of temporal coverage makes it difficult to know if this is a directional signal leading to medium confidence in detection of a trend. It is not clear to what extent this presages boreal forest encroachment into tundra, which would have even larger impacts on ecosystem structure and function. Experiments, models and paleo responses of tundra to past warming all indicate that the increase in shrub coverage and permafrost melting are predictable responses to climate warming trends. High magnitudes of climate change over the last 50 years for most of the Arctic facilitate attribution (Burrows *et al.*, 2011). However, the lack of long-term monitoring data makes it difficult to attribute these changes with high confidence to climate change.

Boreal forest regime shift (see 4.3.3.1.1) - An increase in tree mortality is observed in many boreal forests, with particularly widespread dieback related to insect infestations 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 of a clear temporal trend in morality at the global scale. Models and paleobotanical data indicate that regression of boreal forest at its southern edge is a predictable response to warming. High magnitudes of climate change over most of boreal forest also facilitate attribution to global warming (Burrows *et al.*, 2011). In some cases, tree mortality can be linked directly or indirectly (e.g., due to changes in insect and pathogen ranges) to fluctuations in climate. However, the lack of long-term data sets and high spatial heterogeneity means that attribution to climate change of low confidence.

Amazon forest regime shift (see 4.3.3.1.3 and Box 4-4) - Long-term increases in tree turnover (increased mortality and growth) in the humid tropical forests of the Amazon basin has been documented with moderate confidence. In addition, two episodes of extreme drought over the last decade have lead to increased mortality of large trees. There is low confidence that this represents a trend towards degradation of humid tropical forest, since structural changes in intact forests have been minor. Observed changes in tree turnover have been ascribed to a variety of mechanisms including rising CO_2 effects on vines, recovery from past disturbance and changing climate: the attribution to these various drivers is difficult due to limited temporal and spatial coverage. Models and experiments lend credence to a contribution of climate to increased tree mortality and growth. However, confidence in attribution of observations to climate change is very low is because warming trends over the last 50 years are weak in this region (Burrows *et al.*, 2011

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

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

_____ END BOX 4-3 HERE _____.

4.3.2.1. Phenology

A large number of site-specific studies, and more recently satellite imagery, allow monitoring of phenology. The advance in spring events (breeding, bud burst, breaking hibernation, flowering, migrating,) has been reported for many regions, especially from the northern hemisphere (e.g.: Europe, North America, Arctic) but fewer in the southern hemisphere (e.g.: Africa, Antarctica) and for many plant and animal taxa. Since the AR4 many new studies showing phenological adaptations of species as response to climate change (e.g., Amphibians: Kusano and Inoue, 2008; birds: reviewed in e.g. Pulido, 2007; Thorup *et al.*, 2007; Møller *et al.*, 2010; mammals: Adamík and Král, 2008; plants: Cleland *et al.*, 2007; Gordo and Sanz, 2009; plants and animal taxa: Høye *et al.*, 2007; Primack *et al.*, 2009) and meta analyses were published (observed data: Parmesan, 2006; Parmesan, 2007; Cook *et al.*, 2012b; Ma and Zhou, 2012; Wolkovich *et al.*, 2012; experimental warming experiments: Cleland *et al.*, 2012; Wolkovich *et al.*, 2012).

Generally, there is a great interspecific variability in phenological responses to changing climatic factors. Uncertainties and biases are introduced in research that compares different taxonomic groups or geographic regions by often incomplete or non-overlapping time series, averaged geographical scales, lack of consideration of effects of local climatic variability (e.g. wind speed, climatic conditions at stop-over places) and mostly unknown pressures in winter ranges for migratory species (Hudson and Keatley, 2010). With too short time series long term trends in phenological changes cannot be detected, although responses to annual climate variability can often be characterized. Cross taxa observations show high variation in species- and location specific responses to increasing temperatures in both direction and magnitude (e.g. Parmesan, 2007; Primack *et al.*, 2009).

A variety of environmental drivers could affect behavioral changes in species. Visser *et al.* (2009) analysed experimentally-manipulated temperature effects on laying dates of the great tit (*Parus major*) and showed that higher temperatures lead to significantly earlier egg laying dates in 5 out of 6 years. However, first arrival dates (FAD) and first laying dates, commonly used metrics, can be biased and not accurately represent overall population trends. Miller-Rushing *et al.* (2008) showed that declining sizes of migration cohorts or of populations were likely to account for a large amount of the variation in previously documented changes in migration times. After accounting for changes in migration cohort size, they found that migration distance, climatic variables, and migration date explained portions of the variation in migratory changes over time. Changes in migration phenology cannot only be attributed to changes in temperature changes but also to differing variability of changed feathers during molting times, food availability at stop-over places and differing health conditions of individual species (Gordo, 2007; Møller *et al.*, 2010).

Experiments are an important source of information on physiological and phenological responses to changing climate, but caution has to be exercised when translating their results into future projections using models. Wolkovich *et al.* (2012), in a large meta-analysis, showed that warming experiments substantially underpredict advances in the timing of flowering and leafing of plants. They also showed that experimentally-derived temperature sensitivities of species didn’t matched the observations in the wild.

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. Species unable to adjust their behavior, i.e. advancement of spring events to keep track with changing climates are likely to be negatively affected. The degree, direction and strength of the asynchrony due to changing climatic variables depends on differences in the phenology of the interacting species (van Asch and Visser, 2007; Møller *et al.*, 2010). Increasing temperatures may either bring species more into synch or more out of synch, depending on their respective starting positions (Singer and Parmesan, 2010). Changes in (a)synchrony of interacting species is assumed to effect dynamical features of ecological communities such as trophic cascades, competitive hierarchies, and species coexistence (Nakazawa and Doi, 2012). Ahola *et al.* (2007) could show that the competition between the resident great tit (*Parus major*) and the migratory pied flycatcher (*Ficedula hypoleuca*) is increasing when the onset of breeding of both is becoming closer to each other and the abundances of both species are increasing. The short-distant migrant and resident hosts of the common cuckoo (*Cuculus canorus*) a brood parasite, advanced their phenology as response to increasing spring temperatures more than the cuckoo and the long-distant hosts (Møller *et al.*, 2011). This pattern resulted in a relative host change to more long-distant hosts. van Asch and Visser (2007) report increasing asynchrony of the winter moth (*Operophtera brumata*) and its feeding host oak tree (*Quercus robur*) due to increasing spring temperatures and not changing winter temperatures. In some analyzed years 90 % of the moth eggs hatched before the first oak buds opened (van Asch and Visser, 2007). The edible dormouse (*Glis glis*) a nest predator advanced its hibernation termination significantly with -8 days per decade in the Czech Republic due to increasing annual spring air temperatures, leading to increased nest predation in three out of four bird surveyed species (Adamík and Král, 2008).

[*Placeholder for a more thorough discussion and evaluation of the cited studies with a critical evaluation of the details of quality. It is planned to discuss most recent literature on increase in pollen load and increasing frequency of early spring extreme events*]

Changes in breeding phenology are reported from various regions and different taxa (insects, birds, mammals, amphibians; e.g. reviewed in Parmesan, 2006; Parmesan, 2007; Møller *et al.*, 2010). Various factors can be attributed to changes on breeding phenology, e.g. food availability, increasing spring temperatures. In the northern hemisphere several studies show advancements of egg laying dates in birds (e.g. estimated from Møller *et al.*, 2010: Ø -0.13 days/year -0.8 - + 0.51; n= 52 species; Parmesan, 2007: -3.70 days/decade ± 0.7, n= 41 species). In mammals, Réale *et al.*, 2003 found for the female North American squirrel (*Tamiascurus hudsonicus*) an advancement of 18 days of the mean parturition day in 10 years in the Yukon area, Canada, coinciding with increasing abundance of white spruce cones, the major food source. For the southern hemisphere the opposite pattern, a significant delay of the mean breeding date for two of nine seabirds in the Eastern Antarctic 2.8 to 3.7 days for 1950-2004 (Barbraud and Weimerskirch, 2006). Parmesan (2007) found taxonomic groups to be advancing at signficantly different rates, with amphibian breeding date advancing by eight times as much as other groups, birds and butterflies were not significantly different from each other, but both showed spring advancement three times stronger than for herbs and grasses. Since most butterfly species eat herbs and grasses as host plants, this suggests an increasing asynchrony between these two interacting groups.

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

In habitats dominated by snow, snow cover is a more important factor than temperature. Interactions between temperature precipitation determining snowmelt changes, which is reported to lead to earlier flowering and appearances of plants and arthropods in Greenland (Høye *et al.*, 2007) and lead to earlier flowering in an alpine plant (Lambert *et al.*, 2010). Earlier snow melts are reported to decrease floral resources and can hence effect survival of dependent insects (Boggs and Inouye, 2012).

Large-scale changes and variations in vegetation have been monitored by satellite since 1982, by quantifying the differences in visible and near-infrared wavebands of outgoing radiation (Normalised Difference Vegetation Index, NDVI) which is related to the quantity of photosynthetically-active vegetation. Measurements taken every 15 days by the Advanced Very High Resolution Radiometer (AVHRR) on a polar orbiting satellite are used to determine the seasonal cycle of NDVI, and changes in this are quantified by defining the ‘start’ and ‘end’ of the growing season as the times at which the rate of change of NDVI is at its maximum and minimum (Jeong *et al.*, 2011). Other instruments such as the Moderate Resolution Imaging Spectrometer (MODIS) have also been used more recently, along with other metrics such as Leaf Area Index (the ratio of leaf area to ground area; Ahl *et al.*, 2006). Ma and Zhou (2012) combined species-level observations, meta-analysis, NDVI measurements and phenology modeling to reveal geographically different advancements in spring phenology (start-of-season) of Chinese plant communities in different Ecoregions and Biomes.

Several studies used NDVI to quantify large-scale phonological changes from 1982 up to various dates in the 1990s and 2000s, and all studies agree on a general increase in growing season length of the order of 1-10 days per decade across the northern hemisphere and at continental scales, due to a combination of an earlier start date and later end date. Jeong *et al.* (2011) extended this to 2008, and found that on average at the hemispheric, the start of the growing season advanced by 5.4 days and the end was delayed by 6.6 days. Rates of change varied over time and at smaller scales. Changes were found to be strongly related to temperature trends, although changes in precipitation may also be key drivers, especially in more arid regions (McPherson, 2007).

However, at local scales, the areas that are identified as having become more or less green do not correspond in different studies. The disagreements may be due to methodological differences, such as the use of different satellite sensors, vegetation indices, and trend detection methods. The relatively short duration of satellite observations makes trend detection particularly sensitive to the choice of analysis period. Nevertheless, the general emerging picture of an increase in growing season length is consistent with ground-based phenology studies.

[INSERT TABLE 4-5 HERE
Table 4-5: Changing timing of phenological events based on observations (since AR4). + delay in days per decade, - advancement in days per decade. [More studies will be added.]]

4.3.2.2. Primary Productivity

Primary production is the process of plant growth that underpins energy flow through all ecosystems, enabling services such as crop, forest, livestock and fisheries harvest. It is also fundamental to the regulation of global carbon cycle (see 4.3.2.3. below). Trends in the amount, timing, location and type of primary productivity are therefore important indicators of ecosystem function. Well-established theory, experimentation and observation all agree that primary production is directly and indirectly sensitive to almost all aspects of climate change (see Figure 4-1) as well as to many of the other changes simultaneously taking place in the world. The complexity of the interactions and their frequently non-linear form means that at a given location the net outcome can be an increase in productivity, no change or a decrease.

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

It is increasingly possible to estimate the net uptake on land at subglobal scales, using inversion modelling techniques and the growing network of precision atmospheric observations. There is broad agreement that the net land uptake in untransformed terrestrial ecosystems is globally distributed, almost equally between forested and non-forested ecosystems, but is offset in the tropics by a large carbon emission flux resulting from land use change, principally deforestation (Pan *et al.*, 2011).

Remote sensing from space can in principle be used to monitor trends in vegetation cover and leaf area and thus infer primary productivity. A commonly-used metric is the Normalised Difference Vegetation Index (NDVI), for which nearly 30 years of records are available, covering the entire world. The observed trends are discussed under the various ecosystem-specific discussions below. In some cases the trends are sufficiently strong and consistent to support a confident statement about the underlying phenomenon, but in many cases they are not, and may reflect inadequacies in the indicator, method of analysis and length of the record. The AR4 reported a trend of increasing seasonally-accumulated NDVI (“greening”) at high northern latitudes (Fischlin *et al.*, 2007; based on Sitch *et al.*, 2007), but later observations show a lower rate and no geographical uniformity (Goetz *et al.*, 2007). More than 25% of North American forest areas, excluding areas recently disturbed by fire, showed a decline in greenness and no systematic change in growing season length, particularly after 2000 . NDVI trend analysis studies in rangelands show varying patterns around the world, with substantial areas of disagreement between studies (Millennium Ecosystem Assessment, 2005b; Bai *et al.*, 2008; Beck *et al.*, 2011a; Fensholt *et al.*, 2012). There is general agreement that the Sahel has shown widespread greening over the period 2000-2010, along with an increase in rainfall, but no consensus on whether the detected signal represents productivity by palatable grasses, trees or unpalatable forbs and to what degree it reveals the effects of land management efforts versus responses to climate (Anyamba and Tucker, 2005; Hickler *et al.*, 2005; Hellden and Tottrup, 2008; Prince *et al.*, 2007).

Tree rings record changes in tree growth over approximately the past millennium. Ring width and wood density indicate annual tree productivity and isotopes within the rings reveal changes in the environment the tree was exposed to. Variations in tree rings arise from a variety of factors, including temperature, moisture stress, CO_2 fertilization and ozone damage. The most recent rings in most tree ring chronologies are before the 1990s (Gedalof and Berg, 2010) so tree ring-based conclusions for more recent dates are based on a smaller body of evidence. Tree ring studies may not be representative of forests in general, as most studies were specifically designed to examine growth in response to environmental changes (Gedalof and Berg, 2010) - sites which are not sensitive to environmental changes may therefore be under-represented. Direct repeated measurements of tree girth increment in forest monitoring plots (discussed in 4.3.2.3) are an alternate data source for recent decades.

Many tree ring records show a tendency of accelerated tree growth during much of the 20th century (Briffa *et al.*, 2008), which often correlates with rising temperature. Direct CO_2 effects, inferred from the remaining increased growth once the effects of drought and temperature have been accounted for, have been identified in approximately 20% of the sites in the International Tree Ring Data Base (Gedalof and Berg, 2010) and studied in detail at some sites (Koutavas, 2008). Since the 1980s a number of tree ring records show a decline in tree growth (Wilson *et al.*, 2007). Several possible causes have been suggested for this, including increasing water stress and ozone damage

4.3.2.3. Biomass and Carbon Stocks

Forests around the world are routinely monitored for management and research purposes. Such inventories typically record the species composition and stem size distribution in hundreds of small plots. In conjunction with data on the forest area, usually derived from remote sensing, the forest inventory data allows forest carbon stocks to be estimated. Repeated inventories in principle allow the growth rate and change in carbon stock to be estimated as well. Forest inventory systems are well-developed for northern hemisphere temperate and boreal forests (e.g. Europe: Nabuurs *et al.*, 2010; USA: Ryan *et al.*, 2010; China: Wang *et al.*, 2010). Such data exist for forests and woodlands in tropical countries as well (e.g. Gabon: Maniatis *et al.*, 2011) but are typically less available and comprehensive there (Romijn *et al.*, 2012). The REDD+ initiative and advances in remote sensing (e.g. Baccini *et al.*, 2012) are likely to greatly increase data coverage, quality and availability in the future. Historically, forest inventories were designed to track timber volumes. Their use to infer total biomass and ecosystem carbon stocks requires further information and assumptions, which reduce the certainty in the absolute values, but have a lesser effect on trend detection.

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

4.3.2.4. Transpiration and its Role in the Terrestrial Water Cycle

Transpiration is the uptake of soil moisture by plant roots and its subsequent evaporation to the atmosphere via stomata, and is a critical component of plant physiology. It is also a key component of the water cycle, comprising approximately 50% of the total land-atmosphere moisture flux (Lawrence *et al.*, 2007). As well as affecting ecosystems themselves, changes in water removal by transpiration may therefore also affect soil moisture and runoff, with implications for fresh water availability in the context of a changing climate (Betts *et al.*, 2007). Transpiration also influences surface temperature through the partitioning of energy fluxes between sensible and latent heat (Long *et al.*, 2006). Transpiration is affected by meteorological and hydrological conditions such as air temperature, humidity, soil moisture and incident solar radiation. It is also affected by atmospheric composition, with higher concentrations of CO_2 generally being associated with decreased transpiration through reduced opening of stomatal apertures.

Total evapotranspiration (ET: transpiration through plant stomata plus evaporation from the ground surface and leaf surfaces) is monitored with flux towers, but since most records are only of 10-15 years duration FLUXNET, 2012 there is insufficient data to calculate large-scale, long-term trends. ET can also be estimated from meteorological observations using the Penman-Monteith equation (Monteith, 1965; Penman, 1948) or simulated with models constrained by observations. Estimates of ET from 1120 globally distributed stations indicate that global land mean ET increased by approximately 2.2% between 1982 and 2002, a rate of increase of 0.5mm yr^{-2} (W. *et al.*, 2010), while other studies using constrained models in combination with other methods indicated trends of between 0.25mm yr^{-2} to 1.1 mm yr^{-2} during the 1980s and 1990s but with a plateau or even decline since approximately 2000, attributed to reduced net radiation (Vinukollu *et al.*, 2011; Zeng *et al.*, 2012). A multi-model study using a data-driven empirical model and process-based land models also suggests that global evapotranspiration rose from the 1980s to the late 1990s but this trend then ceased, with increasing soil moisture limitations being cited as the cause.

Experimental studies demonstrate that increasing CO_2 concentrations leads to a decrease in transpiration and increase in intrinsic water use efficiency (iWUE, the ratio of photosynthesis to stomatal conductance, Leakey *et al.*, 2009), so this may be acting to offset climate-driven increases in ET to some extent. Historical changes in iWUE can be inferred from records of stable carbon isotopes in woody plants (Peñuelas *et al.*, 2011) and even the bodies of herbivores (Barbosa *et al.*, 2010) since iWUE has been shown to be related to the change in the carbon isotope discrimination $\Delta^{13}C$ (Duquesnay *et al.*, 1998). 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 (Gagen *et al.*, 2010; Andreu-Hayles *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.*, 2010).

Detection of ecosystem influences on land hydrology depend critically on the hydrological observations. AR4 reported work by Gedney *et al.*, 2006 which appeared to have detected the effects of CO_2 physiological forcing of vegetation (decreased transpiration) in contributing to increased large-scale runoff in the reconstructions of Labat *et al.*, 2006. However, a more recent datasets Dai *et al.*, 2009 showed different runoff trends in some areas. Moreover, there is considerable variation between models in the importance of this for historical river flows relative to other drivers such as land use and precipitation Gedney *et al.*, 2006; Gerten *et al.*, 2008. Comparison of direct anthropogenic effects such as CO_2 rise and land use change with anthropogenic climate effects is difficult since the anthropogenic component of the precipitation contribution has not yet been established. Moreover, other direct anthropogenic contributions to runoff also need to be considered Dai *et al.*, 2009 so a full-system attribution of anthropogenic CO_2 emissions (and other greenhouse gases and other anthropogenic effects) has yet to be provided.

A key influence on the significance of increased iWUE for large-scale transpiration may be whether overall leaf area of primary vegetation has remained approximately constant Gedney *et al.*, 2006 or increased Gerten *et al.*, 2008. While results vary considerably between sites, tree ring studies overall appear to suggest that tree growth did not increase globally since the 1970s Peñuelas *et al.*, 2011 Andreu-Hayles *et al.*, 2011. However, basal area measurements at over 200 plots across the tropics suggest that biomass and growth rates in intact tropical forests have increased in recent decades (Lewis *et al.*, 2009a). The net impact of CO_2 on global scale transpiration therefore remains poorly constrained.

4.3.2.5. Changes in Species Range, Abundance, and Extinction

Species respond to climate change by adapting, by moving out of unfavorable and into favorable climates, or by going globally extinct (Dawson *et al.*, 2011, Bellard *et al.*, 2012; section 4.2.2). Substantial evidence has accumulated since the AR4 reinforcing their conclusion that terrestrial plant and animal species' ranges have moved in response to warming over the last several decades and this movement will accelerate over the coming decades. There is growing evidence that range shifts in response to climate change has and will disrupt many species interdependencies (see section 4.3.2.1.). 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 favorable to them and vice versa. In contrast, high uncertainties concerning global species extinctions attributed to climate change have become more apparent since the AR4 report.

There is a strong correlation between terrestrial species range shifts and climate warming over the last several decades and the number of studies and range of species studied has greatly increased since the AR4 report (e.g., Chen *et al.*, 2011). The most recent synthesis of range shifts indicates that terrestrial species have moved poleward about 17 km per decade (sites in Europe, North America and Chile) and 11 m per decade in altitude up mountains (sites in Europe, North America, Malaysia, and Marion Island), which corresponds to predicted range shifts due to warming (Chen *et al.*, 2011). The "uphill and poleward" view of species range shifts in response to warming is an simplification of species response to changing climate, since response to climate change is also conditioned by changes in precipitation, interactions with land use, and possibly many other factors. This can lead to responses that are not predictable from warming alone (Rowe *et al.*, 2010; Crimmins *et al.*, 2011; Hockey *et al.*, 2011). Shifts in species distributions in response to recent warming are consistent with range shifts in response to climate variation observed in the paleontological record (section 4.2.2). Detailed investigations of the mechanisms underlying observed range shifts show that there are many confounding factors (e.g., Crimmins *et al.*, 2011; Hockey *et al.*, 2011), but the increase in the number of studies and variety of species examined since the AR4 report has substantially increased our ability to detect range shifts and attribute them to changes in climate (Parmesan and Yohe, 2003; Parmesan, 2006; Chen *et al.*, 2011). There is, however, tremendous variation in range shifts between species and species groups. Much of this variation can be explained by large differences in regional patterns of temperature trends over the last several decades (Burrows *et al.*, 2011) and by differences in species dispersal capacity, life history and behavior (Lenoir *et al.*, 2008; Devictor *et al.*, 2012). On average, species appear to be tracking recent changes in climate (Chen *et al.*, 2011), but there is also evidence that many species are lagging far behind (Lenoir *et al.*, 2008; Devictor *et al.*, 2012). Species with short life cycles and high dispersal capacity - such as butterflies or herbaceous plants - are generally tracking climate more closely than longer-lived species or those with more limited dispersal such as birds and trees (Lenoir *et al.*, 2008; Devictor *et al.*, 2012).

Models indicate that range shifts for terrestrial species will accelerate over the coming century. Much of the work since AR4 relies on species distribution models that predict future ranges based on current relationships between climate and species distribution (a.k.a., "niche" or "bioclimatic envelope" models; Peterson *et al.*, 2011), applied to projected future climates. A variety of mechanistic species distribution models are also being developed and more widely used (e.g., Dawson *et al.*, 2011; Cheaib *et al.*, 2012). Recent model comparisons suggest that niche models often predict larger range shifts that more mechanistic models (Morin and Thuiller, 2009; Kearney *et al.*, 2010; Cheaib *et al.*, 2012). There has been progress in model validation, such as the use of hindcasting of observed recent or paleontological range shifts, but validation needs to be reinforced using a wide array of data types (Araujo *et al.*, 2005; Pearman *et al.*, 2008; Nogues-Bravo, 2009; Dawson *et al.*, 2011). Most models do not realistically account for species migration rates, so they generally indicate changes in areas of favorable and unfavorable climate from which shifts in species distribution are inferred (but see Midgley *et al.*, 2006; Hein *et al.*, 2011 and Meier *et al.*, 2012 for examples of models that include migration). Major findings of niche modeling studies can be summarized as follows. In regions with weak climate gradients (e.g., little altitudinal relief), most species would need to migrate many 10's to 100's of km by the end of the century to keep pace with climate change (Leadley *et al.*, 2010). Species that cannot migrate will see their favorable climate space diminish or disappear, but migration that keeps pace with climate change would allow some species to increase their range size (Thomas *et al.*, 2006; Wilson and Maclean, 2011). Models that account for migration mechanisms indicate that many species will be unable to keep pace with future climate change due to dispersal and establishment limitations (Nathan *et al.*, 2011; Meier *et al.*, 2012; Renwick *et al.*, 2012). Biotic interactions such as pollination or predator-prey networks can be disrupted due to decoupling of range overlaps or phenological mismatches, and this may cause much greater impacts on biodiversity than generally predicted (Schweiger *et al.*, 2008; Bellard *et al.*, 2012; Nakazawa and Doi, 2012). Climate change mitigation would substantially reduce the distance that species would need to migrate to track favorable climates (Thuiller *et al.*, 2005; Wilson and Maclean, 2011).

Mountains will provide an extremely important climate refuge for many species (Thuiller *et al.*, 2005; Engler *et al.*, 2011; Gottfried *et al.*, 2012; Pauli *et al.*, 2012) except for species already at the tops of mountains – they are among the most threatened by climate change because they cannot move upwards (Thuiller *et al.*, 2005; ;Sauer *et al.*, 2011; Engler *et al.*, 2011). Pauli *et al.* (2012) reported an increase in species richness from plant communities of European mountain tops in the boreal zone due to increasing temperatures and a decrease on the Mediterranean tops, very likely due to a decrease in the water availability in Southern Europe. Additionally, plant communities of mountain tops already show an increasing number of warm adapted species which could be attributed to increasing temperatures (Gottfried *et al.*, 2012). However, changes in treelines can be difficult to interpet. Treelines in Sweden have responded strongly to the warming since the beginning of the century (Kullman and Öberg, 2009), while land use changes have been a more important driver of recent tree line dynamics in the central European Alps (Gehrig-Fasel *et al.*, 2007).

Observed changes in species abundance are difficult to relate to climate change, because of the complex set of factors mediating population dynamics in non-managed populations. 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 (Myers-Smith *et al.*, 2011). In the Antarctic, two native vascular plants, Antarctic pearlwort and Antarctic hair grass have become more prolific over recent decades, perhaps because they benefit more from warming of soils than do mosses (Hill *et al.*, 2011). Changes in species abundance in these regions are also the most straightforward to attribute to anthropogenic climate change, because high latitude regions have generally exhibited the clearest signals of anthropogenic climate change. Penguin populations have declined in several areas of the Antarctic, including a recent extirpation of an Emperor penguin population, and some researchers have attributed these to regional changes in climate (Trathan *et al.*, 2011). The attribution to regional climate change appears reasonable, but the link to global warming is tenuous (Barbraud and Weimerskirch, 2006). A variety of other changes in abundance in plant and animal populations appear to correlate with climate change, but confident attribution to climate change is difficult because of the importance of confounding factors such as disease, land use change, etc. (e.g., shifts in community structure - Parmesan and Yohe, 2003; increasing shrub abundance in a wide range of ecosystems - Naito and Cairns, 2011; declining amphibian populations - Kiesecker, 2011; population changes in birds - Thaxter *et al.*, 2010).

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

Species distribution models do not predict population changes, but the shifts in predicted species distributions can be used to infer areas where species populations are likely to decline or increase. Projected areas of local extinction from these types of models ranging from near 0% of the current range to more than 95% (Settele *et al.*, 2008; Bellard *et al.*, 2012), and projected local colonization rates are equally variable. Forest models that include population dynamics generally indicate that tree population dynamics tend to slow the response of tree range changes to climate change (Nathan *et al.*, 2011; Hickler *et al.*, 2012). There has been progress in coupling species distribution and species abundance models for a wide range of organisms (Keith *et al.*, 2008; Midgley *et al.*, 2010; Iverson *et al.*, 2011; Schippers *et al.*, 2011; Renwick *et al.*, 2012); however, it is too early to draw strong conclusions.

Global species extinctions, many of them caused by human activities, are now at the very upper limits of observed natural rates of extinction in the fossil record (Barnosky *et al.*, 2011). Most extinctions over the last several centuries can be attributed to habitat loss, overexploitation or invasive species (Millennium Ecosystem Assessment, 2005b). Mollusks, especially freshwater mollusks, have by far the highest rate of documented extinctions of all species groups (Barnosky *et al.*, 2011). Mollusks 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 Kappes and Haase, 2012). In contrast, changes in climate have been identified as one of the key drivers of extinctions of amphibians (Pounds *et al.*, 2006), one of the most threatened groups globally (Secretariat of the Convention on Biological Diversity, 2010). There have been more than 160 likely extinctions of amphibians over last two decades, many of them in Central America (Pounds *et al.*, 2006; Kiesecker, 2011). One of the most notable cases has been the extinction of the golden toad and Monteverde harlequin frog of Central America, which has been ascribed with "very high confidence" to changes in climate (Pounds *et al.*, 2006). 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 invasive fungal infection and land use change, with regional differences in climate as a key contributing factor and ii) these changes in regional climate cannot be clearly attributed to anthropogenic climate change (Sodhi *et al.*, 2008; Collins, 2010; Anchukaitis and Evans, 2010; Hof *et al.*, 2011;Kiesecker, 2011). While this case highlights difficulties in attribution of extinctions to climate change, it also points to a growing consensus that the interaction of climate change with other global change pressures that poses the greatest threat to species (Pereira *et al.*, 2010; Hof *et al.*, 2011).

Model projections of future species extinctions due to climate change have received considerable attention since the AR4 report. Studies using a wide range of methods to estimate future extinction risk generally project large increases in extinction rates compared to current rates and very large increases compared to the paleontological record (Pereira *et al.*, 2010; Bellard *et al.*, 2012). However, the projected extinction risk by the end of the 21st century due to climate change ranges from below 1% to above 50% of species in the species groups that have been studied (Pereira *et al.*, 2010; Bellard *et al.*, 2012). In addition, methodological challenges, large variation in the climate sensitivity of species groups and unknowns in how extinction risk will be translated into realized extinctions over the next century mean that there is very high uncertainty concerning the fraction of species that are likely to go extinct over the next century (Willis and Bhagwat, 2009; Kuussaari *et al.*, 2009; Dawson *et al.*, 2011; McMahon *et al.*, 2011). Evidence from the paleontological record and a lack of confidence in the models used have led to concern that forecasts of very high extinction rates due to climate change (>15%, such as those indicated in the AR4 report), may be overestimated (Botkin *et al.*, 2007; Willis and Bhagwat, 2009; Pereira *et al.*, 2010; Dawson *et al.*, 2011;Hof *et al.*, 2011; Bellard *et al.*, 2012). On the other hand, potential tipping points in terrestrial ecosystems, species interactions and other mechanisms that are not accounted for in most models may mean that future extinction risks have been substantially underestimated (Leadley *et al.*, 2010; Bellard *et al.*, 2012; Urban *et al.*, 2012). There is broad agreement that climate change will result in shifts in species ranges and species abundance, and that in the context of habitat fragmentation and other global change pressures this will contribute substantially to increased extinction risk of terrestrial species over the coming century (Pereira *et al.*, 2010). There is no clear scientific consensus concerning the magnitude of direct impact of climate change on extinction risk, but most experts agree that it is significant.

[INSERT FIGURE 4-9 HERE
Figure 4-9: Historic and future projections of species extinctions. The extinction rate is a function of the duration of the period over which you calculate it. The long-term historical rate, over millions of years, is about one species per million species per year. Observed rates over the past century (coloured bars and dots) are higher than this, but are near or only a small amount above the upper limit of the 'natural range' for century-scale extinctions, outlined in dotted blue lines. Most of the projections for the late 21st century are very substantially outside the natural range.The data sources are depicted as followed : (i) the Cenozoic fossil records (dashed blue lines outline the envelope of observed fossil extinction rates; the solid blue line indicates the mean) ; (ii) historical records (I -= red line - mainland mammal extinctions, redrawn from Barnosky et al. (2011) ; II = red dot - island extinctions of birds ; III = red dot - island extinctions of mammals, both redrawn from Loehle and Eschenbach (2012) ; IV = global bird extinctions ; Pimm et al., 2006 ; all three dots are based on values for extinctions within a 500 year interval) ; and (iii) projected extinctions for the year 2100 (redrawn from Pereira et al., 2010). The six projections of future extinctions include climate change (yellow), land use (green) or both (black) as drivers (1 = Sekercioglu et al., 2008 - birds, 2 = van Vuuren et al., 2006- plants, 3 = Jetz et al., 2007- birds , 4 = Malcolm et al., 2006– plants , 5 = Malcolm et al., 2006– plants and vertebrates , 6 = Sinervo et al., 2010– lizards). Future rates are based on estimates of species 'committed to extinction'. The timing of these extinctions and the fraction of these species that will actually go extinct are subject to very high uncertainty. Note all future rates are centered on 100 year time interval and that projection 5 was extrapolated from 2050 to 2100 (see Pereira et al., 2010).]

4.3.3. Impacts on Major Systems

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

Figure 4-10 provides an overview of two perspectives on estimating ecosystem vulnerability to climate change at the global scale. Figure 4-10a examines projected future climate change in realationship to current inter-annual variability (Beaumont *et al.*, 2011, and see similar analysis by Williams *et al.*, 2007b). This analysis highlights those areas where interannual variability is low compared to projected changes in climate, i.e., where future climate is likely to be outside of temperature fluctuations already experienced by ecosystems (Beaumont *et al.*, 2011). In this analysis, it is the biomes and ecoregions around the equator that are projected to be the most exposed to extreme climates. High latitude systems do not appear as vulnerable in this analysis because they already experience high interannual variability. Figure 4-10b compares future and preindustrial climate based on differences in minimum, mean and maximum temperatures and annual precipitation. This analysis focuses on areas where the absolute differences between future and current climates are high. For example, high northern latitude systems are identified as vulnerable because of the very large temperature differences projected for these regions by the end of the century (often exceeding 6°C, Figure 4-10b1). The biomes (Figure 4-10b2) projected to be the most heavily impacted are tundra (Tu); tropical and subtropical coniferous forests (TSC), especially of Central America; and mangroves (Ma), especially in the Indo-Malay. Thus, some ecosystems appear to be vulnerable because climate change will be large in an absolute sense (e.g., tundra, boreal systems), others because climate change will be large compared to the climate variability that they currently experience (e.g., Amazon and Congo basin humid tropical forests), and some because of both (e.g., in Central America, Andes, Indo-Malay, Himalayas). This analysis highlights why there is a particularly strong focus on climate change impacts on equatorial, sub-equatorial, mountain and high latitude systems in the following sub-chapters.

[INSERT FIGURE 4-10 HERE
Figure 4-10: Two views of the vulnerability of terrestrial biomes to future climate change, both analyses are based on a moderate, A1b, greenhouse gas emissions scenario and multiple climate models. A) Analysis of the Earth's 200 ecoregions of exceptional biodiversity. This estimation of vulnerability is based on the difference between future climate and the current interannual variability of annual mean temperature. Large scores indicate that future climate will be well outside the range of current interannual variability for many months per year (> 2 standard deviations based on current inter-annual variability). B.1) Analysis of all of the Earth's ecoregions. This estimation of vulnerability is based on a muli-variate analysis of absolute differences in minimum, mean and max temperatures and annual precipitation (= climate difference). Large values indicate that it will be much hotter and wetter/drier than preindustrial climate. B.2) Climate differences for the Earth's major biomes. Biomes are as follows: tropical and subtropical moist broadleaf forests, TSM; tropical and subtropical dry broadleaf forests, TSD; tropical and subtropical coniferous forests, TSC; temperate broadleaf and mixed forests, TeB; temperate coniferous forests, TeC; boreal forests/taiga, BT; Mediterranean forests, woodlands, and scrub, Me; tropical and subtropical grasslands, savannas, and shrublands, TSG; temperate grasslands, savannas, and shrublands, TeG; flooded grasslands and savannas, FG; montane grasslands and shrublands, MG; tundra, T; deserts and xeric shrublands, DX; and mangroves, Ma. [*This figure may be completed with an analysis of RCP climate projections and modified to ease comparisons (e.g., map projections).*]]

An issue that emerged in the AR4 report and has grown substantially in prominence since then is that of thresholds or 'tipping points' (Lenton *et al.*, 2008; Scheffer *et al.*, 2009; Leadley *et al.*, 2010). Figure 4-11 provides an map of regional tipping points that could have significant consequences in terms of impacts on global climate or biodiversity (Lenton *et al.*, 2008; Leadley *et al.*, 2010). Tipping points occur when positive feedback processes dominate over negative feedback loops, and the system begins to change as a result of both its internal dynamics and the external forcings, leading to a change in state that is difficult or slow to reverse (Lenton *et al.*, 2008; Scheffer, 2009; Leadley *et al.*, 2010). Many examples of such thresholds have now been identified (Scheffer 2009). Most are at the local scale, but the potential existence of ecological thresholds at a sufficient scale to have regional or global consequences is high. Posited examples include a drought and temperature threshold in the Amazon beyond which forest dieback and fires could contribute to large C fluxes to the atmosphere, or thawing of Arctic permafrost leading to emissions of CH_4 and CO_2 in sufficient quantities to further warm the Earth and amplify the permafrost warming (Lenton *et al.*, 2008; Leadley *et al.*, 2010). Large-scale tipping points have not occurred in the recent past, but there is good evidence for tipping points in the distant past (section 4.2.2) and the sub-sections below highlight reasons for concern that they could occur in the near future. A thorough analysis of all the highlighted tipping points in Figure 4-11 is not possible in this section, so we have highlighted some of the most important and well-studied tipping points (i.e., tundra, boreal forest, Amazon tipping points). Others are discussed in Lenton *et al.* (2008) and Leadley *et al.* (2010).

[INSERT FIGURE 4-11 HERE
Figure 4-11: A global map of tipping points in terrestrial ecosystems that have climate change as an important driver. This map focuses on regional tipping points identified by Lenton *et al.* (2008) or Leadley *et al.* (2010) as having very large impacts on global biogeochemical cycles or biodiversity. While some of these tipping points are primarily driven by climate change (e.g., tundra and boreal systems), most are driven by a combination of biophysical, ecosystem and socio-economic tipping mechanisms.]

4.3.3.1. Forests and Woodlands

Forests and woodlands, are the principal providers of timber and pulp, biofuel and a regulated flow of good-quality water worldwide. They also supply other services, including food and, cultural services such as recreation. Forests are the habitat of a large fraction of the earth's terrestrial biodiversity, especially in tropical regions (Mittermeier *et al.*, 2003). Climate change and forests interact strongly. Climate and atmospheric CO_2 concentrations are major drivers of forest productivity and forest dynamics. At the same time forests play an important role in controlling climate through the large amounts of carbon they can store or release, and through direct effects on the climate such as the absorbtion or reflection of solar radiation (albedo), cooling through evapotranspiration and the production of cloud-forming aerosols (Arneth *et al.*, 2010;Pan *et al.*, 2011; Pielke *et al.*, 2011).

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

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

The future of the interaction between climate and forests is unclear. The carbon stored by intact and regrowing forests appears have stabilized compared to the 1990's, after having increased in the 1970's and 1980's (Canadell *et al.*, 2007; Pan *et al.*, 2011). The drivers behind the forest carbon sink vary greatly across regions. They include forest regrowth and simulation of C sequestration by climate change, rising atmospheric CO_2 concentrations and N deposition (Pan *et al.*, 2011). Most models suggest that rising temperatures, drought and fires will lead to forests becoming a weaker sink or a net carbon source before the end of the century (Sitch *et al.*, 2008; Bowman *et al.*, 2009). Climate change-mediated disease and insect outbreaks could precipitate this transition (Kurz *et al.*, 2008). The greatest risks for large feedbacks from forests to climate arise from widespread tree mortality and fire in tropical forests and low latitude areas of boreal forests, as well as northward expansion of boreal forests into arctic tundra (Lenton *et al.*, 2008; Kriegler *et al.*, 2009; Good *et al.*, 2011).

Recent evidence suggests that the stimulatory effects of global warming and rising CO_2 concentrations on tree growth may have already peaked in many regions (Norby *et al.*, 2010) 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). Detection of a coherent global signal is hindered by the lack of long-term observations in many regions, and attribution to climate change is difficult due to the multiplicity of mechanisms mediating mortality (Allen *et al.*, 2010a).

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

[INSERT FIGURE 4-12 HERE
Figure 4-12: Forest Fire Danger Index (FFDI) and its changes simulated with the HadGEM2-ES Earth System Model (Jones *et al.*, 2011a). Left column: 30-year mean FFDI (MacArthur, 1973; Noble *et al.*, 1980) for 1980-2010 (top) and 2070-2100 under RCP2.6 (middle) and RCP8.5 (bottom). Right column: differences in 30-year mean FFDI relative to 1980-2010 for 2070-2100 under RCP2.6 (middle) and RCP8.5 (bottom). Maps show 30-year means of FFDI values calculated using daily model data, following the methodology of (Golding and Betts, 2008). FFDI is not shown where the simulated vegetation cover is less than 10% for the respective time period (coloured grey).]

4.3.3.1.1. Impacts of climate change on boreal forests

Projections of climate-driven changes in boreal forests over the next few centuries remain divergent on some issues, partly as a result of different processes of change being considered in different models. In particular, the inclusion or exclusion of fire makes a big difference.

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). If fire is considered, some models suggest a boreal forest die-back due to increased fire activity with climate change. (DGVM Intercomparison paper, Sitch *et al.*, 2008**).** These shifts put at risk the boreal carbon sink, estimated at 0.5 Pg year^{-1} in 2000-2007 (Pan *et al.*, 2011).

A number of the Earth System models being run for AR5 include dynamic vegetation, These include immediate feedbacks from vegetation change to local climate (e.g. via surface albedo) so are arguably more consistent than models in which the vegetation is fixed, or changed periodically outside of the model and then fed back in. However, the dynamic vegetation models they are also vulnerable to any inherent biases in climate models.

Whereas boreal forest productivity is widely expected to increase as a result of warming (Hari and Kulmata, 2008; Zhao and Running, 2010), and early analyses of satellite observations confirmed this trend in the 1980s, more recent and longer-term assessments show many areas of boreal forest have instead experienced productivity declines (Goetz *et al.*, 2007; Parent and Verbyla, 2010; Beck *et al.*, 2011b; de Jong *et al.*, 2011; Parent and Verbyla, 2010). The reasons for these “browning” trends have been debated but they are not directly related to fire disturbance (because fires produce random rather than systematic trends in time series). The best evidence to date indicates that warming-induced drought, specifically the greater drying power of air (vapor pressure deficit), induces photosynthetic down-regulation of boreal tree species, particularly conifer species, most of which are not adapted to the warmer conditions (Welp *et al.*, 2007; Bonan, 2008) Satellite evidence for warming-induced productivity declines has been corroborated by tree ring studies (Barber *et al.*, 2000; Hogg *et al.*, 2008; Beck *et al.*, 2011b; Porter and Pisaric, 2011; Griesbauer and Green, 2012) and long-term tree demography plots in more continental and densely forested areas (Peng *et al.*, 2011; Ma *et al.*, 2012). Conversely, productivity has increased at the boreal-tundra ecotone where more mesic conditions support an expected warming-induced growth response (Rupp *et al.*, 2001; McGuire *et al.*, 2007; Goldblum and Rigg, 2010; Beck *et al.*, 2011b).

Warming and drying, coupled with productivity declines, insect disturbance and associated tree mortality, also favor greater fire disturbance. The boreal biome fire regime has intensified in recent decades, exemplified by increases in the extent of area burned but also a longer fire season and more episodic fires that burn with greater intensity (Girardin and Mudelsee, 2008; Macias Fauria and Johnson, 2008; Kasischke *et al.*, 2010; Turetsky *et al.*, 2011). The latter is particularly important because more severe burning consumes soil organic matter to greater depth often to mineral soil, providing conditions that favor recruitment of deciduous species that, in the North American boreal forest, replacing what was previously evergreen conifer forest (Johnstone *et al.*, 2010; Bernhardt *et al.*, 2011). Fire mediated composition changes in post-fire succession influence a host of ecosystem feedbacks to climate, including changes net ecosystem carbon balance (Bond-Lamberty *et al.*, 2007; Goetz *et al.*, 2007; Welp *et al.*, 2007; Euskirchen *et al.*, 2009) as well as albedo and energy balance (Randerson *et al.*, 2006; Jin *et al.*, 2012; O'Halloran *et al.*, 2012). The most comprehensive study to date (Randerson *et al.*, 2006) focused on a set of flux tower measurements across a composition gradient, but was necessarily limited to a few sites in central Alaska. The extent to which the net effect of these feedbacks will exacerbate or mitigate additional warming is not well known over the larger geographic domain of the boreal biome, except via modeling studies that are relatively poorly constrained due to sparse *in situ* observations.

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

The vulnerability of permafrost to thawing and degradation with climate warming is critical not only for determining the rate of an arctic – boreal biome shift, and its associated net feedback to climate, but also for predicting the degree to which the mobilization of very large carbon stores frozen for centuries will exacerbate additional warming (Schuur *et al.*, 2008; Schuur *et al.*, 2009; Tarnocai *et al.*, 2009; Romanovsky *et al.*, 2010; Schaefer *et al.*, 2011). 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 substrate. The degree of thermokarsting associated with different substrates, and associated topographic relief, is tremendously variable because soils with greater rock and gravel content tend to have ice formations that, when melted, produce greater surface slumping and displacement than areas that areas comprised of loamy or organic soils (Jorgenson *et al.*, 2010). 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 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.

[INSERT FIGURE 4-13 HERE
Figure 4-13: Forest mortality related to climatic stress; white dots: localities with documentation; background map: potential environmental limits to net primary production (from Allen *et al.*, 2010a). [*Figure will be modified/replaced for the SOD incorporating one picture per continent to illustrate forests with high mortality rates (= diebacks).*]]

4.3.3.1.2. Impacts of climate change on temperate forests

The largest areas of temperate forest are found in eastern North America, Europe and eastern Asia. The overall trend for forests in these regions has until recently been an increase in growth rates of trees and in total carbon stocks. This has been attributed to a combination of increasing growing season length, rising atmospheric CO_2 concentrations, nitrogen deposition and forest management – specifically regrowth following formerly more intensive harvesting regimes (Ciais *et al.*, 2008b). The relative contributions of these factors has been the subject of substantial and unresolved debate (Boisvenue and Running, 2006).

The world's temperate forests act as an important carbon sink, 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 decline in tree growth rates; increasing tree mortality (Allen *et al.*, 2010a) and changes in fire regimes, insect outbreaks and pathogen attacks. These trends threaten the substantial role of temperate forests as net carbon sinks. 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. Severe forest fires have occurred in several regions over the last decade, for example 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 has been a large increase in the number of modeling studies of climate change impacts on temperate forests at the species and biome levels since the AR4 report. As was the case in the AR4 report, nearly all models project poleward and upward movement of temperate tree species and forests over the next several decades at their equatorial or low altitude range limits, and increased growth and potential for colonization in newly-favorable climates at their poleward or high altitude limits. The most signficant advances since the AR4 report have been the increase in multi-model studies of climate change impacts; greatly improved coverage of eastern Asia; an increase in the variety of modeling approaches used; and modest improvements in model validation.

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 as a result of gains in tree cover in eastern North America and Europe; or decrease due to reductions in total carbon storage per hectare and loss of tree cover (Sitch *et al.*, 2008). Projections for eastern Asia are less variable: temperate forests remain carbon sinks over the coming century, with carbon storage generally peaking by mid-century and then declining (He *et al.*, 2007; Sitch *et al.*, 2008; Ni, 2011; Peng *et al.*, 2009). However, regional vegetation models for China predict a substantial northward shift of temperate forest (Weng and Zhou, 2006; Ni, 2011). There is little indication from either models or observations that the responses of temperate forests to climate change are characterized by tipping points. The feedback mechanisms operating in boreal and tropical forests which lead to tipping point behaviour appear to be less dominant in temperate forests (Bonan, 2008).

At the species level, all models predict that in all regions of the world overstory and understory species will shift their distributions poleward and up in altitude in response to climate change. Projected range shifts vary from several km per decade to hundreds of km per decade. Multi-model comparisons for temperate forests illustrate that there are very large differences in species response and that models differ greatly in the severity of projected climate change impacts on species range (Kearney *et al.*, 2010; Morin and Thuiller, 2009; Cheaib *et al.*, 2012). Tree growth models generally project increased tree growth at the poleward and high altitudinal range limits over most of the next century (Ni, 2011). New approaches to modeling 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 already moving poleward and up in altitude, possibly due to climate warming at the end of the 20th century (Lenoir *et al.*, 2008).

There is overwhelming evidence from both observations and models that climate change has resulted in ranges shifts of temperate forests and their constituent overstory and understory species. These shifts will alter biodiversity and ecosystem services from temperate forests (Dale *et al.*, 2010). Major uncertainties are whether these climate change impacts will be characterized by gradual replacement of species, continued stimulation of tree growth and forest carbon storage over the coming century, or by large-scale dieback at the equatorial and lower altitude range margins, greatly increased disturbance rates, reduced tree growth, all leading to the conversion of temperature forests from net carbon sinks to carbon sources (Dale *et al.*, 2010).

4.3.3.1.3. Impacts of climate change on tropical wet and dry forests

The responses of tropical forests to variability and change in climate and atmospheric CO2 concentration are superimposed upon the direct influences of human activities. These include the harvest of individual plants or animals, management interventions to favour some species or structural features over others, conversion of forests to cropland or grazing land, and changes in ignition sources and fire regimes. Outcomes are difficult to attribute to any single cause, and future changes can be expected to be a consequence of interactions between several causes.

Much of our understanding of primary tropical forest responses to climate variation and the changes in atmospheric composition that have already taken place comes from forest plots that have been measured at multiple-year intervals for the last 2-4 decades. Over 200 plots of typical size ranging from 0.5 to 50 ha have been monitored in the humid tropics (Lewis *et al.*, 2009a). Coverage varies between regions, with over 120 in Amazonia (Chave *et al.*, 2008; Phillips *et al.*, 2009), 80 in Africa (Lewis *et al.*, 2009b; Chave *et al.*, 2008) but only approximately 10 in Asia (Phillips *et al.*, 1998; Chave *et al.*, 2008).

In most of these plots, the forest is undergoing significant changes in tree composition, with slow-growing, high wood-density species declining and fast-growing, light-wooded species increasing. Most plots are accumulating biomass while some have declining biomass. Approximately 70% of the plots in Amazonia and Africa show an increase in overall biomass, with the net change in biomass across 79 plots in an African network being 0.63 (0.22 – 0.94) Mg C ha^{-1} a^{-1} between 1968 and 2007 (Lewis *et al.*, 2009b) and 0.24 (0.07-0.39) Mg C ha^{-1} a^{-1} across 10 plots in a separate pan-tropical network (Chave *et al.*, 2008). In 123 plots in Amazonia, biomass increased by 0.45 (0.33 – 0.56) Mg C ha-1 a-1 between 1975 and 2005. Plots in Panama and Malaysia indicate decelerating relative growth rates between the early 1980s and early 2001 (Feeley *et al.*, 2007), and annual measurements of 6 species in a 216 ha plot of upland forest in Costa Rica showed a general decline in growth between 1984 and 2000 (Clark *et al.*, 2003). Earth observation studies may also be consistent with large-scale, long-term changes in tropical forest (Weishampel *et al.*, 2001; Nemani *et al.*, 2003; Cao *et al.*, 2004) although long-term Earth Observation studies remain limited.

The widespread changes in forest composition, tree growth, mortality and biomass are consistent with atmospheric CO2 fertilization responses or increased water use efficiency (Lewis, 2006; Lloyd and Farquhar, 2008), although changes in solar radiation and the ratio of diffuse to direct radiation may also be significant influences (Mercado *et al.*, 2009; Lewis *et al.*, 2009a; Brando *et al.*, 2010). Recovery from past disturbance has also been suggested as a contributing factor in Africa (Muller-Landau, 2009) and Amazonia, although Lewis *et al.* (2009a) note that this would be expected to lead to decreased stem density and turnover rather than the increases that are generally observed. Sampling bias towards post-disturbance plots has been estimated to be too small to explain the biomass increases in Amazonia (Gloor *et al.*, 2009 but see Fisher *et al.*, 2008). The plot network has sparse coverage in large portions of the Amazon, especially in the eastern and southeastern region where seasonal drought is most pronounced (Fisher *et al.*, 2008). Localised studies with annual monitoring find relationships with local climate variability (Clark *et al.*, 2010) although a general increase in biomass across the tropics appears to be inconsistent with the observed rise in temperatures and decreasing precipitation (Lewis *et al.*, 2009a).

Since AR4, there is new experimental and observational evidence of moist tropical forest thresholds of drought tolerance, and that these thresholds are exceeded within the current climate regime. Two large-scale forest manipulation experiments in which rainfall was partially excluded from forest plots found that tree mortality rose abruptly once a soil moisture deficit threshold was reached that is within the current moist tropical forest climate envelope. Drought-induced mortality was highest for large canopy trees, which means that the effects of severe drought episodes on forest structure and carbon stocks are long-lasting (Nepstad *et al.*, 2007; Brando *et al.*, 2008; Fisher *et al.*, 2008; da Costa *et al.*, 2010). Measurements of forest plots are consistent with these field experiment results. A threshold of drought tolerance appears to have been surpassed in 2005 during a severe drought episode in the Amazon when trees containing 1.2-1.6 Pg of carbon were killed above background levels of mortality (Phillips *et al.*, 2009). A drought in 2010 was more extensive and severe which may have caused even greater impacts on the aboveground carbon stocks of Amazonian forests, although field measurements have not been conducted yet (Lewis *et al.*, 2011). This evidence of forest sensitivity to reductions in rainfall has been interpreted, by some, to be inconsistent with MODIS satellite observations of increased canopy “greenness” during the 2005 drought (Saleska *et al.*, 2007; Samanta *et al.*, 2010). Subsequent studies concluded that elevated dry season greenness is likely to be an artefact of dry season leaf flushing and cloud cover effects (Asner and Alencar, 2010; Brando *et al.*, 2010).

The effect of drought-induced tree mortality on moist tropical forests depends in part upon its interactions with the forest fire regime. If drought is accompanied by fire, tree mortality can increase sharply and forest flammability can remain elevated for years to come (Holdsworth and Uhl, 1997). In the Amazon region, there is a threshold of drought beyond which forests become flammable. This threshold is lower for forests of low stature or low canopy density (leaf area index; Ray *et al.*, 2005). During years of average rainfall, tall moist tropical forests in the Amazon, SE Asia, Mexico, and Central Africa maintain low susceptibility to fire because of the high moist content of the fine fuel layer on the forest floor (Uhl and Kauffman, 1990; Ray *et al.*, 2005; Curran *et al.*, 2004). Severe drought can greatly increase forest susceptibility to fire by provoking leaf-shedding and tree mortality, decreasing canopy density and allowing greater radiation to reach the forest understory (Ray *et al.*, 2005; Brando *et al.*, 2008). These drought-stressed forests do not necessarily catch fire, however, unless they are exposed to ignition sources, such as the fire ignited by land managers. If they do catch fire, the mortality of adult trees can range from 6 to 50% (Cochrane and Schulze, 1999; Balch *et al.*, 2009) and is higher under drier conditions (Brando *et al.*, 2010). If grasses or other highly-flammable vegetation becomes established in the high-light conditions of the burned forest, forest flammability be further elevated (Balch *et al.*, 2010).

Beyond its influence on ignition sources, land use can affect tropical forest vulnerability to climate change through the structural impacts of selective logging and through the regional climatic effects of large-scale forest conversion to pasture and cropland. Logging operations selectively extract canopy trees and damage or topple non-target trees, elevating forest susceptibility to fire by increasing the number of gaps in the forest canopy and the amount of fuel on the forest floor (Uhl and Kauffman, 1990; Curran *et al.*, 2004). Forest clearing and conversion to crops and pastures generally reduces evapotranspiration and increases albedo. Most regional landcover and climate studies predict that current and future landcover change will reduce rainfall and increase air temperatures (Costa *et al.*, 2007; Coe *et al.*, 2009). In the Amazon, the smoke produced by land management fires can also inhibit rainfall under some circumstances (Andreae *et al.*, 2004). In sum, the impacts of climate change on tropical forests will depend upon its effects on rainfall and air temperature and the contributions of these changes to interacting effects of periodic droughts and land uses that are already increasing the frequency and severity of forest fires in the Amazon, Indonesia, and Meso-america (Nepstad *et al.*, 2008; see Box 4-4).

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

The complex interactions between drought, fire, and land-use are not fully represented in coupled models of climate and vegetation, but important progress has been made in capturing some of these interactions since AR4. Modeled simulations of future climate demonstrate that temperature is confidently projected to increase. However, precipitation change is highly uncertain and varies considerably between climate models (WG1 Annex A) Projected future impacts of climate change on tropical forests vary according to both the climate model (Scholze *et al.*, 2006; Zelazowski *et al.*, 2011) and vegetation model (Sitch *et al.*, 2008) used. Defining the potential climatological niche for humid tropical forests (HTF) in terms of annual precipitation and maximum cumulative water deficit, related to the strength of the dry season, Zelazowski *et al.*, 2011) systematically compared projected changes in the HTF niche from 17 of the CMIP3 climate models for global warming of 2°C and 4°C relative to preindustrial (Figure 4-14). Cases with and without increased water use efficiency (WUE) due to CO_2 were examined, but CO_2 fertilization of photosynthesis was not considered. Some climate projections implied increases in forest cover while others implied decreases – in Amazonia and South East Asia, without increased WUE, the balance was more towards decreased forest cover whilst that in Africa was more towards an increase. With increased WUE the balance was more towards increased niche area. One climate model projection implied nearly an 80% reduction in the area of the HTF niche in Amazonia for 4°C global warming without increased WUE, and 50% decrease even with increased WUE. However nearly all projections implied some level of increased niche area up to 40% with increased WUE. In Africa, projected niche changes ranged from a 40% decrease up a 120% increase with and without increased WUE, and in SE Asia changes ranged from a 20% decrease to a 40% increase. A broadly similar range of changes in forest cover were simulated with a Dynamic Global Vegetation Model (DGVM) driven by a different set the CMIP3 models (Scholze *et al.*, 2006), and also by 4 DGVMs driven by climate patterns from a single climate model (Sitch *et al.*, 2008).

[INSERT FIGURE 4-14 HERE

Figure 4-14: Changes in the potential climatological niche for humid tropical forests at 2°C (top) and 4°C (bottom) global warming relative to pre-industrial, with climate changes simulated by 17 CMIP3 GCMs. Decrease in ecosystem water demand due to rising CO_2 is considered. Contours show number of models for which the regional climate change indicates retraction (red) or expansion (green) of the niche. Blue shows the temperature limits for tropical forest. Histograms show, for each region, the number of models indicating different percentage changes in expansion and contraction of the niche area (Zelazowski *et al.*, 2011).

A key uncertainty in future impacts on tropical forests is the strength of direct CO2 effects on photosynthesis and transpiration (see section 4.3.2.4). Lloyd and Farquhar, 2008 present physiological arguments that increased photosynthesis due to CO_2 fertilization would be expected to offset detrimental effects of temperature rise. Korner (2009) notes that this does not necessarily translate into an increase in overall forest biomass. DGVMs and the current generation of Earth System Models, including those used within CMIP5 (e.g., Jones *et al.*, 2011a) generally use formulations for CO2 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 optimised in their behaviour in the global carbon cycle on the basis of more recent data (Jones *et al.*, 2011a). Betts *et al.*, in press point out that, irrespective of uncertainties in the strength and impacts of CO_2 fertilization itself, the impacts relative to warming will also depend on the relative rates of future CO_2 and temperature rise which is affected by uncertainties in both climate sensitivity and the future greenhouse gas mix.

As described above, changes in climate are manifested in moist tropical forests as changes in forest susceptibility to fire, and fire intensity. Fuel moisture is a key influence, and increased drying due to enhanced evaporation would therefore increase fire danger (Zelazowski *et al.*, 2011). In Amazonia, the regions projected by models to undergo increased meteorological fire danger overlap with those projected to undergo ongoing deforestation (Golding and Betts, 2008), so both climate change and deforestation may enhance each other through increased fire activity.

While there is potential for some humid tropical forest loss as a consequence of climate change later in the 21st Century, direct human-induced deforestation and its interactions with episodic drought can be expected to impact the forest much sooner if current trends continue. Climate change may therefore be only one influence on tropical forests, with interactions with land use, especially through fire, also being critical. While some DGVMS include consideration of fire, many do not, so model projections of future impacts of climate change on tropical forests generally do not allow for synergistic impacts with land use.

Given the potential for land use and climate change to each increase the impacts of the other, reduced deforestation activity and the re-establishment of forest or tree plantations on land that has been cleared of forest could be viewed as a climate change adaptation measure since it could reduce the risk of fire occurrence during drought. In the Brazilian Amazon, a 68% decline in deforestation as of 2011 (INPE, 2012) demonstrates that radical reductions in forest clearing are feasible.

_____ START BOX 4-4 HERE _____

Box 4-4. A Possible Amazon Basin Tipping Point

The structure, composition, and tree mortality patterns of Amazon forests are being altered through severe drought episodes, deforestation, logging, and fire. These disturbances are self-reinforcing through inter-connected climatic, ecological, and land-use processes (Davidson *et al.*, 2012). There is evidence that (1) air surface temperatures increase and regional precipitation decreases as deforestation converts additional forests to cattle pasture and cropland (moderate confidence) (da Costa *et al.*,); (b) forest conversion to pastures and crops increases fire ignition points on the landscape (high confidence) (Silvestrini *et al.*, 2011); (c) tree mortality caused by drought, logging or fire increases forest susceptibility to fire (high confidence) (Uhl and Kauffman, 1990; Ray *et al.*, 2005; Balch *et al.*, 2009; Meir *et al.*, 2009); and (d) thresholds of drought beyond which tree mortality and forest flammability increase were exceeded in 1997/98 (fire), 2005, and 2010 and demonstrated in rainfall exclusion experiments (high confidence) (Nepstad *et al.*, 2007; Alencar et al. 2012; Brando *et al.*, 2008; Phillips *et al.*, 2009; da Costa *et al.*, 2010; Lewis *et al.*, 2011; da Costa *et al.*, 2010).

These aspects of the Amazon forest-climate-land use system create the potential for large-scale, near-term (by 2030) degradation of the region's forests and a shift towards savanna-like ecosystems that are far shorter in stature, periodically burned, and in some cases invaded by flammable grasses and herbaceous vegetation (Veldman and Putz, 2011); this regional forest degradation could affect more than half of the forests of the Amazon Basin through clearing, logging, fire, or drought, and could release 20±10 Pg of carbon to the atmosphere (Nepstad *et al.*, 2008) This process can be interpreted as moving beyond an Amazon tipping point (moderate confidence) because it would be self-reinforcing. Recent downward trends in annual deforestation rates could postpone regional forest degradation in the Amazon. As of 2011, annual deforestation in the Brazilian Amazon was 68% lower than the ten-year average ending in 2005 (INPE, 2012; Soares-Filho *et al.*, 2010).

Climate change could exacerbate this process in the near term. For example, increased air temperature resulting from higher atmospheric CO_2 concentration is expected to increase the frequency of severe droughts (Shiogama *et al.*, 2011). These droughts could be similar in intensity to the ones that have affected the Amazon region in recent decades, which were associated with the El Niño Southern Oscillation (1997/98) and the North Atlantic Tropical Anomaly (2005, 2010; Lewis *et al.*, 2011; Trenberth, 2012). The increase in atmospheric CO_2 and increased tree water use efficiency that should result from this rise is consistent with the observed compositional shifts in Amazon forest plots towards faster-growing tree species and with net biomass accumulation (Lewis *et al.*, 2011), but these changes are too subtle to influence/counteract regional forest degradation.

In the long term (by 2100), climate change could drive the replacement of large portions of the Amazon forest by other vegetation types that are more adapted to drier, warmer climatic conditions (Malhi *et al.*, 2008). In some scenarios, these climatic conditions could force the replacement of wet forests by ecosystems that are more similar in structure to dry, transitional forests (more likely) or even savannas (less likely), especially in the Southeast portion of the Amazon Basin (Scholze *et al.*, 2006; Sitch *et al.*, 2008; Malhi *et al.*, 2009; Poulter *et al.*, 2010; Rammig *et al.*, 2010)These new ecosystems, however, are expected to be less diverse, more dynamic, and shorter in stature than current savannas or transitional forests.

Predictions that Amazon forests will be replaced by other ecosystems are mostly based on results from dynamic global vegetation models (DGVMs; Sitch *et al.*, 2008) driven by General Circulation Models of climate (GCMs). In the multiple GCM ensemble used in AR4, the ensemble mean suggested wetting across most of South America, but observational constraints indicate a higher probability of drying in the Amazon (Shiogama *et al.*, 2011).

DGVMs tend to represent physiological processes to be highly sensitive to increases in air temperature (e.g., respiration), but not always to reductions in soil water moisture (Galbraith *et al.*, 2010), and therefore do not necessarily reflect field results from the rainfall exclusion experiments. This weaker effect of soil moisture on the replacement of Amazon forests is partially expected, because most DGVM do not represent explicitly drought-induced tree mortality (Brando *et al.*, 2008). If mortality is represented as a function of soil water stress, the effects of drier, warmer climatic conditions could be stronger (da Costa *et al.*, 2010). DGVMs also simulate increased water use efficiency and photosynthesis due to rising CO_2 concentrations, which is a key influence on the extent and magnitude of dieback simulated in future projections (Lapola *et al.*, 2009). It is argued that CO_2 physiological effects may ameliorate drought or temperature stress to some extent (Lloyd and Farquhar, 2008; Zelazowski *et al.*, 2011), but while this process may have had a notable influence in the past as CO_2 has risen from previously lower levels (Lewis *et al.*, 2009a), its future importance at higher CO_2 is contested. The absence of a Free Air CO_2 Enrichment experiment in tropical forests is a major limitation on understanding of the extent to which CO_2 effects may counter warming or drying of the climate.

While climate change alone could replace part of wet Amazonian forests, land use change could accelerate this process by killing trees (Barlow and Peres, 2008; Balch *et al.*, 2009; Brando *et al.*, 2012), increasing sources of ignition to fire (Nepstad *et al.*, 1999; Alencar *et al.*, 2006; Aragão *et al.*, 2008), increasing surface air temperature (Costa *et al.*, 2007), reducing local precipitation (Costa *et al.*, 2007), and providing seed source for exotic grasses, which can invade forest edges and alter the regime (Balch *et al.*, 2009; Veldman and Putz, 2011) (diagram). Thus, the likelihood of a forest dieback could be reduced via reductions in both CO_2 emissions and deforestation.

[INSERT FIGURE 4-15 HERE
Figure 4-15: 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 increases the likelihood of subsequent burning, especially if burning is followed by forest invasion by grasses or flammable herbs. Deforestation inhibits rainfall in the region and provides ignition sources to flammable forests, contributing to this dieback.

_____ END BOX 4-4 HERE _____

4.3.3.2. Rangelands and Drylands, including Mediterranean-type Systems

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

4.3.3.2.1. Savannas

Savannas are, by definition, coexistences of trees and grasses. Therefore all the global change considerations that apply separately to trees and grasses also apply in savannas, but with the additional consideration of how their differential effects on the tree and grass functional types will alter the structure (i.e. fractional tree cover) of the savanna. Structure in turn strongly influences the herbaceous species composition, number and abundance of herbivores and carnivores, frequency, type and intensity of fires, carbon storage and supply of ecosystem services such as grazing or fuelwood. Savannas are characterised by annual to decadal relatively low-intensity fires, which are not only a key factor in maintaining the tree-grass proportions, but constitute a major global source of pyrogenic emissions from land to atmosphere. Since savannas in the limited sense of neo-tropical vegetation with grasses and more than 10% but less than 60% tree cover occupy about 12% of the global land surface, small changes in tree cover can have globally-significant effects on carbon storage and the emissions of radiatively-active gases and particles. In a looser sense, mixed tree-grass systems throughout the world, and of natural or human-created origin, may occupy over a third of the land surface.

There is evidence from many parts of the world that the tree cover and biomass in savannas has increased over the past century (Scholes and Archer, 1997). In the process the mean carbon density (carbon stored per unit land area, in the soil and tree biomass) increases, , but the grazing potential decreases. The increasing tree cover in savannas has largely been attributed to changes in land management, such as continuous high grazing livestock pressure and reductions in the frequency and intensity of fires. It has been suggested that rising CO_2, which stimulates the growth of trees (with a C3 photosynthetic system) more than that of grasses (predominantly C4 in the tropics), may also be a cause or predisposing factor, possibly acting to relieve a demographic bottleneck that keeps trees trapped within the flame zone of the grasses (Bond and Midgley, 2012). The rising global mean air temperature, changes in precipitation and other changes that parallel rising CO_2 must be considered as potential causes as well.

Elevated CO_2 experiments with savanna saplings (Kgope *et al.*, 2010) support the hypothesis that the competition between trees ad grasses may be altered in the favour of trees under doubling of the atmospheric CO2 concentration above the preindustrial level. The spatial distribution of grasslands above savannas on elevational gradients suggests that a warming also favours trees. The ubiquitous observation of strong contrasts in the degree of savanna thickening between land parcels with the same climate but different land use histories further implies that land management plays a significant role. As yet there are no definitive experiments that can attribute the causes of savanna tree cover changes to climate change or management, or between rising CO2 and temperature.

4.3.3.2.2. Grasslands and shrublands

A large body of work from aaround the world shows that dry grasslands and shrublands are responsive in terms of primary production, species composition and carbon balance to changes in water balance (precipitation and evaporative demand) within the range of projected climate changes (e.g., Sala *et al.*, 1988; Snyman and Fouché, 1993; Fay *et al.*, 2003). Similarly, many grasslands and shrublands are in either very cool or very warm areas, and are expected to respond to warming, but in opposite ways. Grasslands are exposed to rising CO_2, which is postulated to stimulate plant production, but more so for temperate grasses, herbs and woody plants with a C3 photosynthetic system than for C4 tropical grasses (see section 4.2.4.4). Some grasslands are simultaneously exposed to elevated levels of nitrogen deposition, which alters species composition, increases primary production up to a point and decreases it thereafter (see section 4.2.4.2, Bobbink *et al.*, 2010; Cleland and Harpole, 2010). Most rangelands are subject to grazing and fire. All these factors may have interactive effects, and species composition changes may overhelm or alter the direct climate effects (Suttle *et al.*, 2007).

Drawing on relevant published studies, Ravi *et al.* (2010) found that changing climate and land use have resulted in increased in aridity and higher frequency of droughts in drylands around the world. They predict increasing dominance of abiotic controls of land degradation (in contrast to direct human- or herbivore-driven degradation). In particular they higlight changes in hydrology and the erosion of soil by wind.

Rainfall amount and timing have large effects on a wide range of biological processes in grasslands and shrublands, including seed germination, seedling establishment, plant growth, community composition, population and community dynamics production, decomposition and respiration. In a controlled experiment with communities of tallgrass prairie species in Kansas, Fay *et al.* (2008) showed that changes in the interval between rainfall events and the rainfall delivered by individual storms altered carbon assimilation and respiration rates independently of changes in the total amount of water supplied. A similar experiment on a Mediterranean semiarid grassland community (Miranda *et al.*, 2009) found that delayed watering led to decreases in plant community productivity and to delays in flowering time. Species diversity was not affected by delayed onset of rain; however, it was reduced by changes in the frequency, amount and seasonality of wetting. In the Mongolian steppe (Shinoda *et al.*, 2010), drought drastically reduced above ground plant mass but did not substantially affect the root mass. A subset of species did not recover to pre-drought levels once precipitation returned to normal levels. A water and temperature manipulation experiment in China indicated that changes in water balance had a stronger effect on grassland microbial processes leading to carbon storage than changes in temperature (Liu *et al.*, 2009). Engler *et al.* (2011) emphasized that precipitation changes were as important for mountain flora in Europe as temperature changes, and the greatest composition changes are likely where decreased precipitation accompanies warming.

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

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

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

Relatively little is known regarding the combined effect of climate change, other global change factors and increasing use pressure on the large mammal herbivores in rangelands and the pastoral livelihoods that depend on them (Thornton *et al.*, 2009).

4.3.3.2.3. Deserts

The deserts of the world, defined as land areas with an arid or hyperarid climate regime, occupy 35 % of the global land surface. Deserts are sparsely populated, but the people who do live there are amongst the poorest in the world (Millennium Ecosystem Assessment, 2005b). There is medium agreement but limited evidence that the present extent of deserts will increase in the coming decades, despite the overall projected increase in global rainfall, as a result of the strengthening of the Hadley circulation, which determines the location of the circum-tropical desert lands approximately 15-30° N and S, and results in decreased rainfall (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 (Alkama *et al.*, 2012), which differs in sign between deserts closer to the equator than 20° and those closer to the poles. In model simulations, extension of the ‘warm deserts’ causes warming, while extension of the ‘cold deserts’ causes cooling, in both cases largely through albedo-mediated effects. Two special circumstances of desert areas are important with respect to climate change. Deserts are expected to become warmer and drier at faster rates than other terrestrial regions (Stahlschmidt *et al.*, 2011). Most deserts are already extremely hot, and therefore further warming is likely to be physiologically injurious rather than beneficial. Secondly, the ecological dynamics in deserts are rainfall event-driven (Holmgren *et al.*, 2006), often involving the concatenation of a number of quasi-independent events. According to a conceptual model elaborated by Stahlschmidt *et al.* (2011) some desert tolerance mechanisms (e.g., biological adaptations by long-lived taxa) may be outpaced by global climate change.

4.3.3.3. Rivers, Wetlands, and Peatlands

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

Globally, glaciers are shrinking and annual snowpacks persist for less time due to climate 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 a first phase, stream and river flow will generally increase due to intensified melting. During this phase, 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. In a second phase, a threshold is crossed when snowfields melt so early and glaciers have shrunken to the point that late-summer stream flow is reduced. During this phase, reduced stream flow is projected to have much broader negative impacts on freshwater species. Figure 4-16 indicates that a critical threshold is crossed a roughly 50% of loss of glacial cover, after which species diversity of freshwater invertebrates rapidly declines.

[INSERT FIGURE 4-16 HERE
Figure 4-16: Accumulated loss of regional species richness (gamma diversity) as a function of glacial cover. Obligate glacial river macroinvertebrates begin to disappear from assemblages when glacial cover in the catchment drops below approximately 50%. Each data point represents a river site and lines are Lowess fits (taken from Jacobsen *et al.*, 2012).]

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) and one third of the world's soil carbon stock (Limpens *et al.*, 2008; Page *et al.*, 2011). About 14-20% of the world's peatlands are currently used for agriculture (Oleszczuk *et al.*, 2008) and many, particularly peat swamp forests in Southeast Asia, are still undergoing major transformations through drainage and burning in preparation for oil palm and other crops or through unintentional burning (Limpens *et al.*, 2008; Hooijer *et al.*, 2010). Deforestation, drainage, and burning in Indonesian peat swamp forests can release 1,400 Mg CO_2 ha^{-1} (Murdiyarso *et al.*, 2010), contributing significantly to global GHG emissions, especially during periods of intense drought associated with ENSO when burning is more common (Page *et al.*, 2002). Anthropogenic disturbance has changed peatlands from being a weak global carbon sink to a source (Frolking *et al.*, 2011) with large inter-annual variability.

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

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

Rising temperatures resulting in early onset and increased duration and stability of the thermocline in temperate lakes during summer (Winder and Schindler, 2004) are likely to favour a shift in dominance to smaller phytoplankton (Parker *et al.*, 2008; Yvon-Durocher *et al.*, 2011) and cyanobacteria (Johnk *et al.*, 2008; Paerl *et al.*, 2011; Wiedner *et al.*, 2007), especially in those experiencing high anthropogenic loading of nutrients (Wagner and Adrian, 2009); with impacts to water quality, food webs and productivity (Gyllström *et al.*, 2005; O'Reilly *et al.*, 2003; Parker *et al.*, 2008; Shimoda *et al.*, 2011; Verburg *et al.*, 2003). Long-term shifts in macroinvertebrate communities have also been observed in European lakes where temperatures have increased (Burgmer *et al.*, 2007), noting that warming may increase species richness in smaller temperate waterbodies, especially those at high altitude (Rosset *et al.*, 2010). While less studied, it has been proposed that tropical ectotherms will be particularly vulnerable because they will approach critical maximum temperatures proportionately faster than species in high latitude environments, despite lower rates of warming (Deutsch *et al.*, 2008; Hamilton, 2010; Laurance *et al.*, 2011).

There is growing evidence that climate induced changes in precipitation will significant alter ecologically important attributes of hydrologic regimes in rivers and wetlands, and exacerbate impacts from human water use in developed river basins (Aldous *et al.*, 2011; Xenopoulos *et al.*, 2005). 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 (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 likely to impact on freshwater ecosystems, through changes in physical habitat and water quality (Bryant, 2009). Declining rainfall and increased inter-annual variability is likely to increase low-flow and dry-spell duration in dryland regions, leading to reduced water quality in remnant pools (Dahm *et al.*, 2003), reduction in floodplain egg- and seed-banks (Capon, 2007; Jenkins and Boulton, 2007), the loss of permanent aquatic refugia for fully aquatic species and water birds (Bond *et al.*, 2008; Johnson *et al.*, 2005; Sheldon *et al.*, 2010), and terrestrialisation of wetlands (Davis *et al.*, 2010b).

Climate induced changes in precipitation is also likely to be an important factor altering peatland vegetation in temperate and boreal regions, with decreasing wetness during the growing season generally associated with a shift from a *Sphagnum* dominated to vascular plant dominated vegetation type and a general decline of C sequestration in the longer term (Limpens *et al.*, 2008). Projected declines in precipitation and longer dry seasons in major tropical peatland areas in southeast Asia, are likely to result in lower water tables more often and for longer periods, with an increased risk of fire (Li *et al.*, 2007; Rieley *et al.*, 2008; Frolking *et al.*, 2011).

Sea level rise will lead to direct losses of coastal wetlands with associated impacts on waterbirds and other wildlife species dependent on fresh water (BMT WBM, 2010; Pearlstine *et al.*, 2010; Traill *et al.*, 2010) but the impact is likely to 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).

_____ START BOX 4-5 HERE _____

Box 4-5. Snowpack-fed Catchments

Higher winter temperatures have led to seasonal changes in the hydrology of rivers draining catchments with significant snow storage. More winter precipitation is falling as rain instead of snow, and springtime melt is occurring earlier (Trenberth *et al.*, 2007; FOD-WG2 Chapter 3.2.2). In some systems, this has already resulted in discharge peaks much earlier in the year than previously (Figure 4-17; see also Barnett *et al.*, 2005). Despite considerable evidence that stream and river organisms are strongly influenced by their flow regimes (Bunn and Arthington, 2002), thus far few biotic changes have been detected that can be unequivocally linked to the above flow alterations (Poff and Zimmerman, 2010). However, higher winter peak flows are implicated in the projected decline in salmonid populations in the Pacific northwest of the US, due to scouring of the streambed during egg incubation periods (Battin *et al.*, 2007). Less snowpack also results in lower flows in summer and fall, reducing the amount of available spawning habitat and further increasing water temperatures. Almost everywhere in western North America, a 10%– 50% decrease in the spring–summer streamflow fractions will accentuate the typical seasonal summer drought with important consequences for warm-season water supply, aquatic ecosystems, and wildfire risks (Stewart *et al.*, 2005). Reductions in summer base flows will increase the competition between consumptive users of water and the environment. 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 4-17 HERE
Figure 4-17: Daily average discharge in the Carson River, California, U.S. in 1952 and 1996 provides an example of changes in stream discharge in response to earlier onset of snowmelt. Discharge spikes occur in February in 1996 rather than later in spring (from Palmer *et al.*, 2009, redrawn from Stewart *et al.*, 2005).]

_____ END BOX 4-5 HERE _____

4.3.3.4. Tundra, Alpine, Permafrost Systems

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

Continued climate change is projected to cause terrestrial vegetation and lake systems of the Arctic to change substantially in the future, with an ongoing expansion in woody vegetation cover projected in tundra regions over the 21st Century in the CMIP5 Earth System Models (Ciais *et al.*, in preparation) and by dynamic global vegetation models driven by other climate model projections. Changes may be complex and in some cases involve non-linear and threshold responses to warming and other climatic change (Hinzman *et al.*, 2005; Müller *et al.*, 2009; Bonfils *et al.*, 2012). Due to long response times of vegetation to both warming and increased CO_2 (Ciais *et al.*, in preparation; Falloon *et al.*, in preparation) Earth System Models project Arctic vegetation change to continue long after any stabilization of global mean temperature. In some regions, reduced surface albedo due to increased vegetation cover is projected to cause further local warming even in scenarios of stabilised global radiative forcing due to greenhouse gases (Falloon *et al.*, in preparation).

7 of 19 sub-populations of polar bears are declining in number, while 4 are stable, 1 is increasing and the remaining 7 have insufficient data to identify a trend (Vongraven and Richardson, 2011). Declines of two of the sub-

populations are linked to reductions in sea ice (Vongraven and Richardson, 2011). Polar bear populations are projected to decline greatly in response to continued Arctic warming (Hunter *et al.*, 2010), and it is expected that the populations of other Arctic animals (e.g., fox and caribou) will be affected dramatically (Post *et al.*, 2009; Sharma *et al.*, 2009). Simple niche-based, or climatic envelope models have difficulty in capturing the full complexity of these future changes (MacDonald, 2010).

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

The arctic tundra biome is also recently experiencing shifts in two additional phenomena; fire disturbance and permafrost degradation. Both of these processes facilitate conditions for woody species establishment in tundra areas, either through incremental migration or via more rapid “leap-frogging” to areas reinitialized by burning (Epstein *et al.*, 2007; Goetz *et al.*, 2011). When already present at the boreal-tundra ecotone, even sparsely, shrub and tree species show increased productivity with warmer conditions (Devi *et al.*, 2008; Andreu-Hayles *et al.*, 2011; 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) and produce thermokarsting associated with melting of ice layers and wedges in what were formerly permanently frozen soils (permafrost). There is tremendous variability in the degree of thermokarsting associated with fire disturbance, depending upon the substrate and ice content (Jorgenson *et al.*, 2010) but the combination of thermokarsting and seasonal cryoturbation create conditions that alter vegetation succession (Racine *et al.*, 2004; Lantz *et al.*, 2009; Higuera *et al.*, 2011).

The area of permafrost is projected to continue to decline over the first half of the 21st Century in all emissions scenarios (Figure 4-18). In the RCP2.6 scenario of an early stabilization of CO_2 concentrations, permafrost area is projected to stabilize at a level approximately 20% below the 20th Century area, and then begin a slight recovering trend. In RCP4.5, in which CO_2 concentration is stabilized at approximately 550ppmv by the mid-21st Century, permafrost continues to decline for at least another 250 years. In the RCP8.5 scenario of ongoing CO_2 rise, permafrost area is simulated to approach zero by the middle of the 22nd Century (Figure 4-18).

[INSERT FIGURE 4-18 HERE
Figure 4-18: Top panel: Scenarios of atmospheric CO_2 concentration under the Representative Concentration Pathways (RCPs). Bottom panel: Simulations of past and future northern hemisphere permafrost area with a maximum thaw depth less than 3m deep (Caesar *et al.*, in preparation using the HadGEM2-ES Earth System Model (Jones *et al.*, 2011b). Note different x-axis scales in the two panels, and also the change in scale after 2100 in the bottom panel.]

Frozen soils and permafrost currently hold more than twice the carbon than the atmosphere and thus represent a particularly large vulnerability to climate change (i.e., warming). Although the Arctic is currently a net carbon sink, continued warming will likely act to soon turn the Arctic to a net carbon source, that will in turn create a potentially strong positive feedback to accelerate Arctic (and global) warming with additional releases of CO_2, CH_4, and perhaps N_2O, from the terrestrial biosphere into the atmosphere (Schuur *et al.*, 2008; Schuur *et al.*, 2009; Maslin *et al.*, 2010; McGuire *et al.*, 2010; O'Connor *et al.*, 2010; Schaefer *et al.*, 2011). Moreover, this feedback is already accelerating due to climate-induced increases in fire (McGuire *et al.*, 2010; O'Donnell *et al.*, 2011). The rapid retreat of snowcover, and resulting spread of shrubs and trees into areas currently dominated by tundra has also already begun, and will continue to serve as a positive feedback accelerating high latitude warming (Chapin *et al.*, 2005; Bonfils *et al.*, 2012).

Alpine systems are already showing a high sensitivity to on-going climate change and will be highly vulnerable to change in the future. In western North America, warming, glacier retreat, snowpack decline and drying of soils is already causing a large increase in mountain forest mortality, wildfire and other ecosystem impacts (e.g., Westerling *et al.*, 2006; Crimmins *et al.*, 2009; van Mantgem *et al.*, 2009; Pederson *et al.*, 2010; ; Muhlfeld *et al.*, 2011), and disturbance will continue to be an important agent of climate-induced change in this region (Littell *et al.*, 2010). Alpine ecosystems already appear to already be changing in response to climate change in Africa, Tibet, the Alps, and elsewhere (Cannone *et al.*, 2008; Cui and Graf, 2009; Allen *et al.*, 2010a; Eggermont *et al.*, 2010). For example, in a study of permanent plots from 1994 to 2004 in the Austrian high Alps, a range contraction of subnival to nival vascular plant species was indicated at the downslope edge, and an expansion of alpine pioneer species at the upslope edge (Pauli *et al.*, 2007). As with the Arctic, permafrost thawing in alpine systems could provide a strong positive feedback in those systems (e.g., Tibet; Cui and Graf, 2009).

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Box 4-6. 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 and biomes. Thus far these changes appear to be not so much threshold responses as gradual and systematic shifts across temperature and moisture gradients. Responses are expressed through gross and net primary production, microbial respiration, fire and insect disturbance, vegetation composition, species range expansion and contraction, surface energy balance and hydrology, active layer depth and permafrost thaw, and a range of other inter-related variables. Because the high northern latitudes are warming more rapidly than other parts of the earth, due at least in part to arctic amplification (Serreze and Francis, 2006), the rate of change in these ecological processes are sufficiently rapid that they can be documented *in situ* (Hinzman *et al.*, 2005; Post *et al.*, 2009; Peng *et al.*, 2011; Elmendorf *et al.*, 2012) as well as from satellite observations (Goetz *et al.*, 2007) 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, are facilitated by more rapid shifts associated with fire disturbance. Each of these interacting processes, as well as insect disturbance and associated tree mortality, are tightly coupled with warming-induced drought. Similarly, gradual productivity increases at the boreal-tundra ecotone are facilitated by leap-frog dispersal into areas disturbed by tundra fire and thermokarsting. In North America these coupled interactions set the stage for changes in ecological processes, already documented, consistent with a biome shift characterized by increased deciduous composition in the interior boreal forest and evergreen conifer migration into tundra areas that are, at the same time, experiencing increased shrub densification. The net feedback of these ecological changes to climate is multi-faceted, complex, and not yet well known across large regions except via modeling studies, which are often poorly constrained by observations.

[INSERT FIGURE 4-19 HERE
Figure 4-19: Tundra Biome shift. Earth system models predict a northward shift of Arctic vegetation with climate warming, as the boreal biome migrates northward 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.]

_____ END BOX 4-6 HERE _____

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

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

4.3.3.5.1. Plantation forestry

While a majority of general aspects on forests are already dealt with in section 4.3.3.1, here we focus on issues that are specifically relevant to plantation forests. Forest plantations currently cover only a minor fraction of the global forest area, estimated at 3% to 7%, but this area is growing rapidly by a few million ha per year (FAO, 2010; Kirilenko and Sedjo, 2007). Most new plantations are established in the tropics and subtropics (Kirilenko and Sedjo, 2007), where concerns have been raised about unsustainable forest practices (Shearman *et al.*, 2012). Nevertheless, the shift from natural forests to plantations is projected to accelerate in the future, partly driven be increasing demand for bioenergy (Kirilenko and Sedjo, 2007).

In most areas with extensive forest plantations, forest growth rates have generally increased during the last decades, but the variability is large, and in some areas production has decreased. 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, as many environmental drivers and changes in forest management interact (Boisvenue and Running, 2006; Kirilenko and Sedjo, 2007; Hemery, 2008).

According to forest growth models, climate change is, at least under moderate climate change, likely to increase forest yields globally, in particular in temperature-limited northern forests and when new forest plantations are established in areas that become suitable as a result of climate change (Briceno-Elizondo *et al.*, 2006; Kirilenko and Sedjo, 2007; Lindner *et al.*, 2010). However, projected regional variability is large, many models may overestimate potential positive effects of elevated CO_2 (Kirilenko and Sedjo, 2007; see section 4.2.4.4), and the effects of disturbances, such as wildfires, forest pests and pathogens, and extreme events, such as windstorms are either poorly or not at all represented in most models (Kirilenko and Sedjo, 2007). As these effects are major drivers of forest dynamics (Köhl and San-Miguel-Ayanz, 2011), model projections are likely too optimistic. Using a model that accounts for fire effects and insect damage, Kurz *et al.* (2008) showed that recent insect outbreaks might have caused a transition in the Canadian forest sector from a sink to a source of carbon.

Many plantation forests are monocultures or mixtures of few species. Low species (and often also genetic) diversity compared with natural stands might render plantation forests particularly vulnerable to climate change (e.g. Hemery, 2008). Choosing provenances that are likely well adapted to current and future climates is extremely difficult because of uncertainties in climate projections and because saplings are particularly sensitive to climatic extremes (Broadmeadow *et al.*, 2005). Furthermore, it is highly uncertain how forest pests and pathogens will spread as a result of climate change and trade, new pathogen-tree interactions might occur (Brasier and Webber, 2010), and trees that are not well adapted to future climates are more susceptible to pests and pathogens (Schlyter *et al.*, 2006; Bolte *et al.*, 2009). Therefore, risk spreading by promoting multi-species mixed stands and natural regeneration, which can increase genetic diversity (Kramer *et al.*, 2010), has been advocated as an plausible adaptation strategy (Bolte *et al.*, 2009). At least at the southern border of cold-adapted species, such as Norway spruce (*Picea abies*) in Europe, climate change will very likely lead to a shift in the main tree species used for forest plantations (Iverson *et al.*, 2008; Bolte *et al.*, 2009).

4.3.3.5.2. Bioenergy systems

Bioenergy sources include traditional forms such as wood and charcoal and more modern forms such as the industrial burning of biomass wastes, the production of ethanol and biodiesel. While traditional biofuels have been in general decline as users switch to fossil fuels or electricity, they remain dominant energy sources in many less-developed parts of the world, such as Africa, and retain a niche in developed countries. The production of modern bioenergy is growing rapidly throughout the world in response to climate mitigation policies that incentivise their use, or as a strategy to decrease oil dependence and thus increase energy security (Kirilenko and Sedjo, 2007). The WG III chapter on energy addresses their potential as a climate mitigation strategy, while the sensitivity of biofuel crops to climate change should be quite similar to those previously mentioned for plantation forestry (which to a large extend are grown for bioenergy purposes) and/or agricultural systems (WGII, chapter 7). In a review on climate change impacts on biofuel yields in temperate environments (Oliver *et al.*, 2009a) it was concluded that elevated CO_2 is likely to increase drought tolerance of bioenergy crops (as it is paralleled by improved plant water use), which may lead to constantly high yields. Generally potentials of bioenergy production under climate change might be high, but are very uncertain (Haberl *et al.*, 2011).

An important part to deal with here is the ecosystem impacts of large-scale land use changes related to the growing of bioenergy biofuels. Policy shifts in OECD countries favor the expansion of biofuel production, sometimes at the expense of food crop production, and placing new pressures on terrestrial and freshwater ecosystems (Searchinger *et al.*, 2008; Lapola *et al.*, 2010). It is e.g. unclear if the global trend to reduced forest loss will continue - there are substantial pressures to deforest for the production of food and biofuels (Wise *et al.*, 2009; Meyfroidt and Lambin, 2011). Under the assumption of doubling the growth rate of demand for bioenergy until 2030, a scenario by Buongiorno *et al.* (2011) would lead to severe consequences for the global forest sector with a global reduction of forest stocks of 2 % or a 4% reduction for Asia.

Bioenergy potentials are strongly influenced by human food requirements (incl. feed required for livestock), thus integrated approaches to optimize food and bioenergy are needed (Haberl *et al.*, 2011). Such considerations ignore areas for the conservation of biodiversity and ecosystems. These are more explicitly dealt with by Field *et al.* (2008) and Tilman *et al.* (2009), while in both papers the usage of abandoned land, which was previously under agricultural use, is regarded as an option for biomass production which reduces net warming. However, this ignores that e.g. such habitats may be core elements in cultural landscapes of high conservation value (many species rich grasslands in Europe have been croplands before and later abandoned), and that the productivity of such areas could be too limited for biofuels (Spangenberg and Settele, 2009). As an alternative, Fargione (2011) discusses biofuel crop yield increases on existing cropland, with the aim to avoid expansion of agriculture.

4.3.3.5.3. Cultural landscapes

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

In such landscapes, conservation efforts - as all across the globe - often go for the conservation of ecosystems that contain endangered biotic communities. However, in such cultural landscapes this aim might be hard to achieve due to the very dynamic nature of systems.

Keeping species in a favorable conservation status in cultural landscapes (one aim e.g. of European policies; EU Council, 1992, can normally only be achieved through appropriate management, as the vast majority of endangered species in the wider countryside depend on certain types of land use for their survival. This requires profound knowledge of the systems and species involved, and as this is rarely the case, conservation success was limited (see Thomas *et al.*, 2009 for a notable exception).

As in many other cases, population and niche models are available and partly already validated (Thomas *et al.*, 2009). This shows where future challenges are to be found: particularly in the quantification of relative importance of climate change in comparison with the habitat and its management (Settele and Kühn, 2009). So far the majority of changes can be attributed to land use as the most obvious driver (Nowicki *et al.*, 2007), while the impact of climate change can be readily detected in few examples (Devictor *et al.*, 2012), combined effects seem to better explain the overall picture (Schweiger *et al.*, 2008; Schweiger *et al.*, 2012), where attribution to climate change seems low to medium.

As a consequence, preliminary ideas have been put forward to adjust the European Natura 2000 protected area network to take into account changing climatic conditions and to enable migration or dispersal of species across the landscape (Bertzky *et al.*, 2010; Vohland *et al.*, 2010).

4.3.3.5.4. Urban ecosystems

Over half of humanity lives in urban areas with a yearly increase of ca. 67 million people (Pickett *et al.*, 2011). Although urban areas cover only <3% of Earth's terrestrial surface they harbor a large variety of species (McKinney, 2008). Urban areas are themselves drivers of climatic change as they are accounted for 80% of greenhouse gas emissions and also heavily impacted by increased minimum and sometimes decreased maximum air temperatures, reduced or increased precipitation and altered biogeochemical cycles (Grimm *et al.*, 2008). An important threat to cities comes from extreme events. Many cities are located at coasts or rivers. However since there is medium confidence (based on physical conditions) that projected increases in heavy rainfall events would contribute to increases in local flooding (IPCC, 2012) there is limited evidence that urban areas will be over-proportionally affected by floods in the future. Despite rainfall events, it is very likely that sea level will contribute to rise in future and hence affect coastal urban areas (IPCC, 2012). For the future it is virtually certain that the frequency and magnitude of warm days and nights (heat waves) will increase globally (IPCC, 2012). However McCarthy *et al.* (2010) projected with the integration of an urban land-surface model in the HadAM3 Global Climate Model a significant higher increase in the frequency of hot nights in urban areas compared to rural areas. Hence also the social aspects in urban areas (health status, outdoor activities) and urban infrastructures will be increasingly affected (IPCC, 2012). There is high agreement among scientists that urban climate effects (e.g., increased temperatures) nowadays are similar to projected changes of climatic variables at a local scale. Similarly to all other ecosystems, ecological impacts of changing climatic factors in urban ecosystems will change species compositions as well as compositions of traits. Knapp *et al.* (2008) found that traits compositions of urban plant communities are changing during urbanization towards adaptive characteristics of dry and warm environments. With increasing temperatures, this effect is likely to continue also in the future and is likely to be exacerbated compared to non-urban surroundings. Urban areas are one of the main starting points for the establishment and dispersal of alien species (e.g. for plants through urban gardening; Knapp *et al.*, 2012). With increasing air temperatures especially in winter time and the warming effect of urban areas, it is very likely that the number of established alien plants in urban areas will increase and their spread into rural areas (see also 4.2.4.6.). With increasing numbers of alien species, it is likely that also the BVOC (Biogenic volatile organic compounds) emissions from gardened alien plants will increase (Niinemets and Penuelas, 2008; see also 4.2.4.6.).

4.3.4. Impacts on Key Services

Ecosystem Services are the benefits which people derive from ecosystems [see glossary]. The classification system proposed by the Millennium Ecosystem Assessment (Millennium Ecosystem Assessment, 2003) is widely used. It recognises *provisioning services* such as food, fibre and water (also known as 'goods' in the economic literature, and which have their own chapters in this assessment); *regulating services* such as climate regulation, pollination, pest and disease control and flood control; *supporting services* which are used by people indirectly, through other services, and include items such as primary production and nutrient cycling; and *cultural services* which include recreation, aesthetic and spiritual benefits. The list of ecosystem services is long and growing; and almost all are potentially vulnerable to climate change. This chapter discusses only a few regulating and cultural services where the link to climate change has been examined.

4.3.4.1. Habitat for Biodiversity

Climate change can alter habitat for species by inducing i) shifts in habitat distribution that are not followed by species, ii) shifts in species distributions that move them outside of their preferred habitats and iii) changes in habitat quality (Urban *et al.*, 2012). There is some evidence that these climate change impacts have already occurred. For example, loss of sea ice is altering habitats for polar bears and may partially explain declines in polar bear populations in some area. However, this is not yet a widespread phenomena. Models of future shifts in biome, vegetation type and species distributions, on the other hand, suggest that many species could be outside of their preferred habitats with the next few decades due to climate change (Urban *et al.*, 2012).

Hole *et al.* (2009) found that the majority of African birds are projected to move large distances over the next 60 years resulting in substantial turnover of species within protected areas (>50% turnover in more than 40% of Important Bird Areas of Africa) and migration across unfavorable habitats. Many birds may find suitable climate in the large network of protected areas, but will be forced to cope with new habitat constraints. Similarly, Araujo *et al.*, 2011) found that approximately 60% of plants and vertebrate species would no longer have favorable climates within European protected areas, often pushing them into unsuitable or less preferred habitats. Wiens *et al.*, 2011) project similar effects in the western US, but also find that climate change may open up new opportunities for protecting species in areas where climate is currently unsuitable. Indeed, in some changes climate change may allow some species to move into of areas of lower current or future land use pressure (Bomhard *et al.*, 2005). These and other studies strongly argue for a rethinking of protected areas networks and of the importance of the habitat matrix outside of protected areas as a key to migration and long-term survival of species.

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

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

4.3.4.2. Pollination, Pest and Disease Regulation

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

Among the regulating services most strongly related to biodiversity, pollination and biocontrol of pests have to be highlighted. Climate change tends to increase the abundance of pest species particularly in previously cooler climates, but assessments of changes in impacts are hard to make (Scherm, 2004). Insect pests are directly influenced, e.g. through the quality of food plants (Pascal, 2006) or via the effects on their natural enemies (predators and parasitoids). Direct impacts are via the relatively high temperature optima of insects, which lead to increased vitality and reproduction (Allen *et al.*, 2010a). Mild winters in temperate areas promote frost susceptible pests. For the vast majority of indirect effects, e.g. spread of insect borne diseases, information is scarce (for further assessments on climate change effects on pest and disease dynamics see WGII, chapters 7 and 11).

Pollination

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

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

It can be expected that the impacts of climate change on the main pollinator groups (bees and syrphid flies) show similar patterns like e.g. butterflies (Roberts *et al.*, 2011; Settele *et al.*, 2008). For butterflies it has been shown that drastic climate change effects can be expected under a set of future scenarios (Spangenberg *et al.*, 2011) and that only for few species rather positive effects can be assumed (while connectivity of the landscape and the mobility of species are important and widely unknown factors, which might drastically change the expected future impacts). While for the majority of species the climate space for the insect is limiting, there are species with particular dependence on certain plants which might suffer because of the different movements of future climatically suitable space of plants and butterflies (Schweiger *et al.*, 2008; Schweiger *et al.*, 2010). As in butterflies the plants are largely independent from the insect in their development, one might expect more severe developments in tighter interacting pollination systems (absence of pollinator could mean absence of pollination dependent plant and vice versa; compare results of Biesmeijer *et al.* (2006).

4.3.4.3. Climate Regulation Services

Ecosystems moderate the climate locally through a range of mechanisms, including reducing the near-ground wind velocity, cooling the air through shading and the evaporation of water and ameliorating low temperatures through releasing heat absorbed during the day or summer. This service is widely recognized and values, for instance in the establishment of windbreaks, gardens and urban parks. The focus of this section is on processes operating at much larger scales – the region to the whole globe. Terrestrial ecosystems affect climate at large scales through their influence on the physical properties of the land surface and on the composition of the atmosphere with respect to radiatively-active gases and particles. Approximately a quarter of the CO_2 emitted to the atmosphere by human activities in the past decade was taken up by terrestrial ecosystems, reducing the rate of climate change proportionately (WG 1 Chapter 6).

However, the overall effect of tropical afforestation on global temperature could be 1.75 times the effect expected on the basis of changes in carbon storage alone, due to physical processes such as changes in evapotranspiration (Arora and Montenegro, 2011). Observations and model evidence indicates that, broadly speaking, forests make warm areas cooler and cold areas warmer, through the competing effects of evapotranspiration (cooling) and surface albedo (in dark-canopied forests, generally warming). Model simulations suggest that if more than 40% of the pre-1700 extent of the Amazon forest were to be cleared, rainfall in the region would be reduced (Sampaio *et al.*, 2007). According to satellite observations, the effect of conversion of the Brazilian savannas (*cerrado*) to pasture is to induce a local warming, (Loarie *et al.*, 2011), which is partly offset when the pasture is converted to sugarcane. It has been suggested (Ridgwell *et al.*, 2009) that planting large areas of crop varieties with high-albedo leaves could help regional cooling. Model analysis indicates this strategy could be marginally effective at high latitudes, but have undesirable climate consequences at low latitudes, and measurements show that the current range of leaf albedo in major crops is insufficient to make a meaningful difference (Doughty *et al.*, 2011).

4.3.4.4. The Sensitivity of Nature-based Tourism

There are both direct and indirect impacts of global climate change on tourism (Simpson *et al.*, 2008; United Nations World Tourism Organization (UNWTO), 2010). Climate is an important determinant of the suitability of locations for tourist activities, drives seasonal demand in many destinations, and effects the costs of tourism operators (e.g. snowmaking, heating and cooling) (Gössling *et al.*, 2012; Simpson *et al.*, 2008; Hoffmann *et al.*, 2009). The indirect impacts of climate change includes issues such as water availability, the loss of biodiversity, production options in agrotourism industries such as food and wine, and the spread of diseases and disease vectors such as malaria, whose presence discourages tourism (Simpson *et al.*, 2008). Sea level rise, storm surges, and other extreme weather events may damage infrastructure that tourism depends on and affect tourist choice. Furthermore, climate change mitigation policies, such as aviation carbon taxes that increase the cost of travel, may also reduce demand for tourism services. Climate change induced socio-economic problems in host countries may lead to security concerns, and in source countries to a reduction in disposable income among potential tourists. Critically, decisions to travel are based on perceptions and expectations formed by the media, marketing, the internet and other sources, and may be disconnected from the actual conditions at a destination (Uyarra *et al.*, 2009; Biggs, 2011).

4.3.4.5. Water, including Quality and Groundwater

Rivers and streams provide a range of ecosystem services that provide benefits to society and are likely to be affected by climate change (see Table 4-1; Palmer and Filoso, 2009). Many of the implications of climate change on the provisioning services of water for food production and domestic supply are discussed in detail in WGII Chapter 3. Over 80% of the world’s population is currently exposed to high levels of threat to water security because of land use change, pollution and water resource use (Vörösmarty *et al.*, 2010). Under climate change, reliable surface water supply is likely to decrease due to increased temporal variations of river flow that are caused by increased variability in precipitation and decreased snow/ice storage (see WGII Chapter 3). In some Mediterranean regions, e.g. southwest of Western Australia, significant reductions in surface flows have already forced governments to pursue alternative water sources (desalination, stormwater harvesting, recycled wastewater). In snow dominated regions, warming will mean that less winter precipitation falls as snow and melting of winter snow occurs earlier in spring (high confidence; Barnett *et al.*, 2005). This means a shift in peak discharge to winter and early spring and away from summer and autumn when human demand is highest. In some locations, water storages may not be sufficient to capture winter runoff, particularly if they are also used to safeguard downstream communities from flooding (Barnett *et al.*, 2005).

Climate change not only poses risks to the quantity of water available for human society but also the quality. Warming is implicated in the increasing frequency and intensity of harmful cyanobacterial blooms (Paerl and Paul, 2012). Reduced summer flows in streams and rivers, increases the risk of low dissolved oxygen events and associated poor water quality.

4.4. Adaptation

4.4.1. Autonomous Adaptation by Ecosystems

Autonomous adaptation refers to the adjustments which social-ecological systems (SES) make, of their own accord, in response to a changing environment (Smit *et al.*, 2000); also called ‘spontaneous adaptation’ (Smit *et al.*, 2007). In the context of human systems it is sometimes referred to as ‘coping capacity’. Here we focus on adaptation to a changing climate in all its manifestations, recognising that climate change in the narrow sense is necessarily associated with changing CO_2 and ozone, and in practice is usually accompanied by changing land use, abundance of competing or mutualistic organisms and other environmental stressors, such as pollution.

All systems have some capacity for autonomous adaptation, or they would not exist at all. Ecosystems or SES 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 it has experienced and survived. The capacity for autonomous adaptation is necessary for *resilience* but is not synonymous with that term as it is now used (Walker *et al.*, 2004). Adaptability to one set of challenges does not automatically confer greater adaptive capacity to a different set of challenges, especially if those challenges are novel (i.e. outside the range of experience of the system). Furthermore, even in relation to environmental changes which are familiar and within the magnitude range of previous adaptation, an environmental change that is more rapid than in the past or is accompanied by other stresses may exceed the demonstrated adaptive capacity of the system. It is reasonably speculated, but not firmly and generally established, that insulating systems from environmental variability erodes their adaptive capacity over time.

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 underlying genetic code (through evolution – the specific meaning of ‘adaptation’ within biology); and change in the composition of the community, either through in- or out-migration, or local extinction.

4.4.1.1. Phenological and Physiological

The ability to project future impacts of climate change on ecosystems is complicated by the potential for adaptations by species, which may alter apparent relationships between climate variables and species / ecosystem viability. Adaptation by individual species may increase their ability to survive and flourish under different climatic conditions, possibly leading to lower risks of extinction than expected from observationally-based relationships with climate (Botkin *et al.*, 2007), but may also affect their interdependence with other species leading to disruption of community interactions (Visser and Both, 2005).

Phenology is typically highly adapted to the climate seasonality of the environment in which the organism evolved. The phenology of any species also needs to be keyed to the phenology of other species with which it interacts, such as competitors, food species and pollinators. Thus change in phenology is both a sensitive indicator of adaptation (Menzel *et al.*, 2006), but also a potential mechanism of ecosystem disruption if adaptation is insufficiently rapid or coordinated between interdependent species, or is cued to environmental signals that are not affected by climate change, such as day-length (Bradley *et al.*, 1999; Both and Visser, 2001; Parmesan, 2006). The environmental cues for phenological events are complex and multi-layered (Körner and Basler, 2010; Singer and Parmesan, 2010). For instance, late-succession temperate trees require a chilling period in winter, followed by a threshold in daylength, and only then are sensitive to temperature. As a result, projecting current phenological trends forward is risky, since the relative importance of cues can change (Cook *et al.*, 2012b). The results are sometimes counterintuitive, such as the increased sensitivity of flowering in high-altitude perennial herbs in the Rocky mountains to frost as a result of earlier snowmelt (Inouye, 2008).

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

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

The physiological processes in organisms can either adapt through plasticity or genetically (i.e., through evolution). The former is generally regarded as rapid but limited, and the latter as less constrained, but slow (see 4.4.1.3) although long-term studies of a few organisms indicate rapid genetic adaptation to a changing climate (Bradshaw and Holzapfel, 2006). Acclimation to higher temperatures and/or higher CO2 concentrations is seen in many organisms, but to very widely varying extents and general understanding remains limited (Hofmann and Todgham, 2010). A key physiological adaptation with large potential consequences for the global climate system is the sensitivity of the relationship between organism temperature and respiration rate, particularly in decomposer organisms (Jones *et al.*, 2003). The shape of the relationship varies between locations with different climates and shows apparent acclimation within months or years to a changing climate (Giardina and Ryan, 2000; Luo *et al.*, 2001; Rustad, 2001). The relationship is also dependent on C substrate type and the presence of other respiration-controlling environmental factors, complicating the determination of the inherent rate and its adaptation to a changing climate. Several competing hypotheses can explain the observed reduction in respiration rates in soil warming experiments after an initial peak – physiological adaptation; depletion of readily available substrate (Kirschbaum, 2004); and varying temperature sensitivities in different soil carbon pools (Knorr *et al.*, 2005).

4.4.1.2. Evolutionary and Genetic

There is increasing evidence that species have adapted to recent climate warming or to climatic extremes through rapid evolution (e.g., Franks and Weis, 2008; Chown *et al.*, 2010; Ozgul *et al.*, 2010; Hill *et al.*, 2011; Hoffmann and Sgro, 2011; Karell *et al.*, 2011) Rapid evolution can occur through many mechanisms including selection of existing resistant genotypes within populations, mutation and selection of new genes and perhaps even through epigenetic processes such as DNA methylation (Lavergne *et al.*, 2010, Paun *et al.*, 2010). Most evidence for rapid evolution comes from species with rapid life cycles; however, even long-lived species such as trees may adapt to some extent over the coming century through selection of resistant genotypes (Jump *et al.*, 2008; Karell *et al.*, 2011; Kramer *et al.*, 2010). For example Karell *et al.* (2011) found increasing numbers of brown genotypes of the tawny owl (*Strix aluco*) in Finland in 28 years and attributed it to milder, fewer snow-rich winters, which is strong selection pressure against the heritable trait white genotype. Rapid evolution could save species from local and global extinction, i.e., "evolutionary rescue", but at the price of substantial loss in genetic variability and reductions in populations size if the impact of climate on populations is severe (Urban *et al.*, 2008, Bell and Gonzalez, 2009). Whether species adapt to new environmental conditions can also be inferred from the degree to which environmental niches are conserved when environment is changed. Petitpierre *et al.* (2012) found little evidence for niche shifts in 50 terrestrial plant invaders in their new ranges.

It has been argued that rapid evolutionary responses are of little importance in the face of climate change (Parmesan, 2006). Evidence from the fossil record clearly shows that range shifts and other non-evolutionary responses dominate the long-term response of species to climate change, suggesting that adapation is of minor importance (Schoener, 2011). However, evolutionary reponses may have buffered such range shifts (Davis *et al.*, 2005; Warren *et al.*, 2011) and, most importantly, rapid adaptation may prevent species extinctions (Bell and Gonzalez, 2009).

There is low confidence whether observed changes e.g., in phenological behavior or range shifts attributed to higher temperatures are caused by environmental induced phenotypic plasticity or local evolutionary adaptation. Levels of phenotypic plasticity and/or evolutionary adaptation are highly variable and could be different at different locations within the same species (Lyon *et al.*, 2008). Phillimore *et al.* (2010) showed for the common frog across Britain, that population differences in earlier spawning due to increasing spring temperatures could be attributed largely to local genetic adaptation. Under given climate change projections for Britain locally adapted frog populations need to expand their adaptation range towards earlier spawning to track changes in increasing spring temperatures, since phenotypic plasticity is not sufficient enough to compensate projected changes (Phillimore *et al.*, 2010).

Rapid advances in quantitative genetics, genomics and phylogenetics have provided valuable insights into the functional importance of genetic variability within populations and among species (Davis *et al.*, 2010a; Salamin *et al.*, 2010; Hoffmann and Sgro, 2011). For example, the timing of budburst in trees has been shown to be under partial genetic control and to be one of the important adaptive responses of trees to climate (Vitasse *et al.*, 2009). Studies of extent genetic variability across species ranges and with models that couple gene flow with spatially-explicit population dynamics suggest that populations are equally sensitive to climate change in ways that are counterintuitive. In some cases, too much or too little gene flow to populations at range margins may have created fragile, maladapted populations, which is in contrast to the current wisdom that populations at the range margins may be best adapted to global warming (Bridle *et al.*, 2010; Hill *et al.*, 2011). Conversely, there is also evidence from experiments, models and observations that populations in the center of species ranges may in some cases be more sensitive to environmental change than those at range boundaries (Bell and Gonzalez, 2009). Generalization will be complicated by the complex interactions between local adaptation, gene flow and population dynamics (Bridle *et al.*, 2010).

4.4.2. Human-Assisted Adaptation

Assisted adaptation means a deliberate, external intervention with the intent of increasing the capacity of the target organism, ecosystem or SES to survive and function at an acceptable level, in the presence of climate change; also known as ‘planned adaptation’ (Smit *et al.*, 2007), although SES may have autonomously-planned adaptation as well. This chapter focuses less on the adaptation of people, human communities and infrastructure, since they are the topics of WGII chapters 8 to 17, and more on non-human organisms and the ecosystems they form. ‘Assistance’ in this context means a range of actions, from ensuring the presence of suitable habitat and dispersal pathways and the reduction or removal of other stressors, to physically moving organisms, storing them and establishing them in new places.

4.4.2.1. Reduction of Non-Climate Stresses

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.

4.4.2.2. Protected Areas: Amount, Location, and Layout

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 current protected area footprint. It is noted that the current geography of protected areas is far from optimal under the current climate, and that it is cheaper to proactively acquire land in the areas of likely 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). Hickler *et al.* (2012) analysed the layout of protected areas in Europe and concluded that under projected 21st century climate change and found that a third to a half of them would be occupied by different potential vegetation than they currently represent. The amount of new area that needs to be added to the existing protected area network to ensure future representativity 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 purely local scale (Heller and Zavaleta, 2009); and better integration with a biodiversity-hospitable landscape outside the protected area is more effective than treating the protected areas as islands.

4.4.2.3. Landscape and Watershed Management

Human assisted adaptation implies the intervention of humans on ecosystems, in order to mitigate and/or adapt to climatic change impacts. The general principles for management adaptations in United States were summarized from a major literature review by West *et al.*, 2009. The prevailing idea is that, until now, adaptation to climate change has been mainly focused on strategies for improving the resilience of ecosystems to persist in their current states. They suggest that in the context of climate change a successful management of natural resources will require to cycle 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. A common perception is that management agencies aim at maintaining public lands and waters unchanged. Barriers and opportunities pointed out by the authors were divided into four categories: (1) legislation and regulations, (2) management policies and procedures, (3) human and financial capital, and (4) information and science. At the national level, changes can be divided into several categories: (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 expectation of ecosystem change.

Human-mediated migration (assisted migration)of species was proposed as a solution when fragmentation of habitats limits migration potential of many species or when natural migration rates are outstripped by the pace of climate change, (Chmura *et al.*, 2011; Vitt *et al.*, 2010). In the northwestern U.S., warmer temperatures and changed precipitation predispose forests to disturbance by wildfire, insects, and disease; and ultimately change forest structure and composition at the landscape scale. In this area, effective approaches to climate adaptation will likely include assisted migration of species and populations, and density management (Chmura *et al.*, 2011;). A prioritization of species for seed banking, both for restoration purposes and for potential assisted migration in the future, is an example of grassland ecosystem used by the Dixon National Tallgrass Prairie Seed Bank (Vitt *et al.*, 2010). A genome scan approach identified four potentially adaptive loci in important grassland species *Arrhenatherum elatium.* Knowledge on adaptive loci might in the long run also help to adapt ecosystems to adverse climate change effects through assisted migration of ecotypes rather than introduction of new species (Michalski *et al.*, 2010).

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

4.4.2.4. Assisted Migration and Restoration

Assisted migration and habitat restoration are receiving growing attention as a potentially necessary measure to conserve species in the face of climate change (Hoegh-Guldberg *et al.*, 2008; Loss *et al.*, 2011). There is low agreement in the scientific community whether it is best to increase the resilience of ecosystems to climate change, thus help to preserve existing communities, or to enhance the capacity of ecosystems to transform in the face of overwhelming forces of species migrations and modifications of ecosystem function by climate change. The options for management can be summarized in three main categories: i) try to maintain / improve existing habitat so that species don't have to move, ii) maintain or improve migration corridors, or iii) intervene heavily by translocating species (Hoegh-Guldberg *et al.*, 2008, Loss *et al.*, 2011).

There is high agreement among the scientific and conservation community that maintaining or improving migration corridors or ecological networks , in large part because the measures required for reducing climate change impacts on species are 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, therefore, is a more ecosystemic approach that assisted species migration. Some caution should be excercised because observational and modeling studies show that increases in habitat connectivity do not always improve population dynamics of target species, may decrease species diversity, and facilitate spread of invasive species (Cadotte, 2006; Brisson *et al.*, 2010; Matthiessen *et al.*, 2010).

There is medium agreement whether the concept of assisted migration is one possible mitigation option. t overcoming migration limitations to species response to climate change (Hoegh-Guldberg *et al.*, 2008; Loss *et al.*, 2011; Vitt *et al.*, 2009; Willis and Bhagwat, 2009; Hewitt *et al.*, 2011). The speed of 21st century climate change and substantial habitat fragmentation in many areas of the world mean that many species will be unable to migrate or adapt fast enough to keep pace with climate change. If this results in significant reductions in range size this is likely pose problems for long-term survival of the species. Moving species may help overcome migration limitations, but raises serious issues about introducing species into ecosystems where they do not currently occur. Some ecologists believe that careful selection of species would minimize the risk of undesirable impacts on existing communities or ecosytem function (Minteer and Collins, 2010), but others argue that the history of intentional species introductions shows that the outcomes are often unpredictable and in many cases have had disasterous impacts (Ricciardi and Simberloff, 2009). Moreover, the degree and magnitude of phenotypic responses or genetic adaptation to climate change are very variable among species, making decisions which species might be translocated more complex. An additional issue is the number of species that could potentially require assisted migration that could easily overwhelm funding capacity, implying a debatable prioritization process (Minteer and Collins, 2010).

4.4.2.5. Ex Situ Conservation

Conservation of plant and animal genetic resources outside of their natural environment, in gardens, zoos, breeding programmes, seed-banks or gene-banks has been widely advocated as an ‘insurance’ against both climate change and other sources of biodiversity loss and impoverishment (Khoury *et al.*, 2010). There are many examples of existing efforts of this type, some very large and with global scope (Millennium Seed Bank, Svalbard vault etc). Several issues remain largely unresolved (Li and Pritchard, 2009): the physiological, institutional and economic sustainability of such efforts into the indefinite future; the fraction of the intra-specific variation that is preserved (and how much needs to be preserved for future viability); and whether it is possible to reintroduce such species (especially animals) successfully into the wild after generations of ex-situ conservation.

4.4.3. Barriers, Limits, and Incentives to Adaptation

The autonomous adaptation of wild organisms to climate change relies heavily on the capacity to migrate to areas of suitable climate. Even for those organisms theoretically capable of doing so at the required rate, physical barriers often prevent their movement. These can be topographic (e.g., valleys, mountain ranges and water bodies), but are increasingly human-created, such as fences, roads and areas of unsuitable habitat (such as croplands or settled areas). Increasing habitat fragmentation reduces the pathways that a migrating organism can take.

Organisms can also adapt physiologically or evolutionarily. There are some limits to either the rate or the magnitude of such adaptations. There are apparently upper temperature limits to the setting of seed in many grasses.

4.4.4. Consequences and Costs of Inaction and Benefits of Action

Failure to act plausibly leads to ecological, social and economic *damages* due to ecosystem change. The necessary actions to cope with unavoidable damages generate *adaptation costs*, while *mitigation costs* are associated with actions to tackle undesired future damages. Examples of these costs, based on recent literature, are shown in Table 4-6. The timing of action also has cost implications: increasing costs due to delay must be weighed against the risks associated with premature measures (Szlavik and Csete, 2012). In addition to the direct financial costs of action, further costs may appear through trade-offs between services: e.g. afforestation for climate mitigation is costly in terms water provision (Chisholm, 2010). Traditional agriculture with low fossil fuel use preserves soil carbon sinks and supports on-sitebiodiversity (Martinez-Alier, 2011), but due to the lower per hectare yields, requires a larger area to be dedicated to agriculture.

[INSERT TABLE 4-6 HERE
Table 4-6: Examples of the costs of action and inaction with respect to climate change and ecosystems. Types of cost: damage (D), adaptation (A), mitigation (M).]

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

Economic calculations are appropriate when at least one component of the ecosystem services is traded in markets (like for biofuels; Spangenberg and Settele, 2009). If climate regulation services are translated into a tradable item, through carbon markets or payments for avoiding deforestation, then their value can be expressed via market prices (Shaw *et al.*, 2011).

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

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

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

Several of the alternatives to fossil fuel require extensive use of the land surface. Bio-energy requires land to either be taken from food production or from natural ecosystems. Many renewables require significant land areas. Hydroelectricity usually involves the impoundment of large bodies of water behind dams, leading to flooding of pre-existing ecosystems. Solar energy effectively involves decreasing the albedo of areas of ground surface. Large-scale wind energy involves the location of arrays of turbines across the landscape. As an illustration, the “aggressive mitigation” scenario RPC2.6 relies heavily on both bio-energy and renewables as major components of the energy mix (Figure 4-20), so there is clear potential for unintended consequences for terrestrial and inland water systems.

[INSERT FIGURE 4-20 HERE
Figure 4-20: Mix of energy sources for the RCP2.6 “aggressive mitigation” scenario (Vuuren *et al.*, 2011). Bio-energy, either with or without carbon capture and storage (CCS), and many renewables have implications for the terrestrial landscape and hence may carry risks of unintended consequences to terrestrial and inland water systems.]

In RCP2.6, although food production remains dominant, the scenario still involves some land use / land cover change (Box 4-2) in order to facilitate the use of bio-energy. By 2100, bio-energy crops occupy approximately 4 billion hectares, approximately 7% of global cultivated land. Modification of the landscape and the fragmentation of habitats are major influences on extinction risk (Fischer and Lindenmayer, 2007), especially if native vegetation cover is reduced or degraded, human land use is intensive and “natural” areas become disconnected. Hence, additional extensification of cultivated areas for energy crops may pose extinction risks.

RCP2.6 also includes a substantial increase in renewables above current levels of deployment, either as hydropower, solar or wind. Damming of river systems for hydropower can cause fragmentation of the inland water habitat with implictions for fish species, and monitoring studies indicate that flooding of ecosystems behind the dams can lead to declining populations, e.g., of amphibians (Brandão and Araújo, 2007). Large dams may also result in CH_4 emissions due to decay of flooded vegetation in anaerobic conditions and the subsequent release of CH4 from the water surface and especially from water passing through turbines and spillways (Fearnside, 2005; Lima *et al.*, 2008), so dams may act as sources of greenhouse gas emissions.

Concern is often raised over wind turbines posing a danger to birds and bats, but estimating mortality rates is complex and difficult (Smallwood, 2007). It has been estimated that wind farms in Europe and the USA cause between 0.3 and 0.4 wildlife fatalities per gigawatt-hour (GWh) of electricity, and while a similar fatality rate is estimate for nuclear power stations, fossil-fuel power stations are estimated to cause approximately 5.2 fatalities per GWh (Sovacool, 2009), although this quantification has been criticized as having key methodological flaws (Willis *et al.*, 2010b).

Solar energy systems, by design, aim to maximise the absorption of solar energy and hence widespread deployment of photovoltaics (PV) has the potential to exert a positive radiative forcing on climate through decreased surface albedo. Generally such a forcing is 30 times smaller than the avoided radiative forcing arising from the use of PV to substitute fossil fuels, so overall PV still exerts a net reduction of climate warming (Nemet, 2009). Nevertheless, at the local scale there are some plausible circumstances in which the decreased surface albedo substantially reduces the effectiveness of PV as a negative climate forcing (Nemet, 2009).

Adaption measures may also result in unintended consequences. Relocation of agricultural areas as a climate change adaptation measure could pose risks of habitat fragmentation and loss similar to those discussed above in the context of mitigation through bio-energy. Deliberate relocation of vulnerable and important species – assisted migration – may also be a potential conservation measure in the context of facilitating adaptation to climate change (Maclachlan *et al.*, 2007. However this may directly conflict with other conservation priorities, for example by actually facilitating the introduction of invasive species (Maclachlan *et al.*, 2007).

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

4.5. Emerging Issues and Key Uncertainties

The likely presence of thresholds in ecosystem response to climate change, and specifically the type of thresholds characterised as a ‘tipping point’ emerged in the Fourth Assessment Report and has grown substantially in prominence since then, but remains a major source of uncertainty. In general (Field *et al.*, 2007), negative feedbacks currently dominate the climate-ecosystem interaction, but in several areas, such as the boreal ecosystems positive feedbacks could dominate under moderate warming. For positive feedbacks to propagate into a ‘runaway’ process leading to a new equilibrium state, the strength of the feedback has to exceed that of the initial perturbation. This has not been demonstrated for any plausible and immanent large-scale ecological process as yet, but for most processes increasing degrees of warming tip the balance towards positive rather than negative feedbacks (Field *et al.*, 2007).

A significant source of uncertainty in ecological responses 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 understanding the systems better, but some will remain irreducible because of the failure of predictive models when faced with mathematical bifurcations – a problem that is well-known in climate science. Probabilistic statements about possible outcomes are theoretically possible in this context, but ecosystem science is far from being able to conduct such analyses routinely.

The issue of biophysical feedbacks between ecosystem state and the climate, over and above the effects mediated through greenhouse gases, is emerging as significant in many areas. Such effects include those caused by changes in surface reflectivity (albedo) or the partitioning of energy between latent energy and sensible heat.

The combined effects of elevated carbon dioxide and tropospheric ozone on plant productivity has critical consequences for the suture sink strength of the biosphere, since they of similar magnitude but opposite sign.

The costs of adaptating of ecosystems, biodiversity and ecosystem services to a changing climates is poorly known, as are the costs of failing to do so.

Frequently Asked Questions

FAQ 4.1: How does climate change contribute to species extinction?

Climate change increases the risk of extinction because some species are neither able to adapt to the new environment, nor move to habitat which is more favourable. Changes in climate may have already contributed to the extinction of a small number of species, such as frogs in Central America. In the future, many more species will be at risk, especially when climate change combines with other pressures such as habitat destruction or pollution. There is consensus that minimizing climate change will help protect species from extinctions. Under moderate rates and amounts of climate change and with the assistance of conservation actions, the majority of species should be able to adapt to the new climates, or respond by moving to higher latitudes or altitudes.

FAQ 4.2: Why does it matter that ecosystems are altered by climate change?

Ecosystems are the machinery that delivers things essential for all people, everywhere in the world – including obvious items like food, clean water and timber, but also less-obvious ones such as control of pests and diseases, regulation of the climate and pollination of flowers. When ecosystems change, their capacity to supply these services changes as well, for better or worse. Change in ecosystems includes what species they contain (and in what proportions), how they appear (tall or short, open or dense) and how they work (e.g. productive or unproductive). Climate change, among other factors, has an effect on all these aspects of ecosystems, as well as on their location and extent. Ecosystem change has knock-on effects on many sectors, including human health through altering where diseases are found, agriculture through grazing supply and pollinators, infrastructure through changing risks of flooding. Although in the long-term not all ecosystem changes are detrimental to all people, a rapidly-changing set of ecosystem services from any given location will require adaptation actions by the people who are affected by them.

FAQ 4.3: What are the non-greenhouse gas effects of rising carbon dioxide?

Some greenhouse gases also exert further influences on the environment in addition to their impact on radiative forcing. The concentration of carbon dioxide affects photosynthesis and transpiration in plants, with photosynthesis generally being enhanced under higher CO_2 concentrations while transpiration is generally decreased. The growth of plants and/or the efficiency with which they use water can therefore be increased. The response varies considerably between species. Increased atmospheric CO2 concentration also increases the acidity of ocean water, with potential consequences for organisms that rely on the the production of calcium carbonate since the rate of this depends on pH.

FAQ 4.4: What costs are caused by changes in ecosystem services due to climate change?

Climate change might reduce service provision options of ecosystems. Coral reefs are at risk due to mass coral bleaching induced by temperature rise. They have been calculated to supply services of several 1000 $/ha especially for moderation of extrme events and for opportunities for recreation and tourism. If pollination services are at risk, e.g. due to climate change related asynchrony of flowering time of plants and activity periods of pollinators, a service worth 153 billion € per year might have to be compensated by other means (e.g. manual pollination), which would be much more expensive than the services provided for free by pollinating insects and other animals.

FAQ 4.5: What are the opportunities for better managing ecosystems under climate change?

Management of ecosystems can contribute to mitigate climate change effects. Tropical forest for example deliver a wide set of services, among which climate regulation (including carbon capture and storage) is a major one. The average value of all supporting services derived from these forests is several hundred dollars per ha annually. One should aim at large CO_2 removals through accelerated carbon capture, which can be achieved by restoring natural ecosystems (especially forests).

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

Land use change often leads to the occurrence of warming driven by increased radioactive forcing. This is caused by increased GHG emissions and “CO_2-fertilization” effects. Changes in precipitation are more variable, although many works report rainfall decreases (drying), desertification, and changes in rainfall pattern. Increases in surface albedo by land decovering are another climate change driver. Direct and indirect consequences in the local climate have to be expected in many regions (i.e. Eastern Europe, Northern India, and Eastern China) after albedo decreases by deforestation. A regional decrease in total cloud cover at the top of the atmosphere after deforestation may eliminate the climate response in the tropics, thus causing a strong meridional (equator to pole) gradient of climate response to land cover change.

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Table 4-1: Biome shifts in the past.

	Location	Reference	Plots	Time Period	Shift type	Retracting biome	Expanding biome	Temperature change (°C century^{-1})	Precipitation change (century^{-1})
1.	Alaska Range, Alaska, USA	Lloyd and Fastie, 2003	18	1800-2000	L	UA	BC	1.1*	0.03
2.	Baltic Coast, Sweden	Walther *et al.*, 2005	7	1944-2003	L	TC	TB	0.6*	0.08
3.	Becca di Viou, Italy	Leonelli *et al.*, 2011	1	1700-2008	E	UA	BC	0.9*	-0.06
4.	Garibaldi, British Columbia, Canada	Brink, 1959	1	1860-1959	E	UA	BC	0.7*	0.16*
5.	Goulet Sector, Québec, Canada	Payette and Filion, 1985	2	1880-1980	E	UA	BC	1.4*	0.19*
6.	Green Mountains, Vermont, USA	Beckage *et al.*, 2008	33	1962-2005	E	BC	TB	1.6*	0.06
7.	Jasper, Alberta, Canada	Luckman and Kavanagh, 2000	1	1700-1994	E	UA	BC	0.6	0.21*
8.	Kenai Mountains, Alaska, USA	Dial *et al.*, 2007	3	1951-1996	E	UA	BC	0.7	0.06
9.	Kluane Range, Yukon, Canada	Danby and Hik, 2007	2	1800-2000	E	UA	BC	0.7	0.05
10.	Low Peninsula, Québec, Canada	Payette and Filion, 1985	1	1750-1980	N	-	-	1.4*	0.19*
11.	Mackenzie Mountains, Northwest Territories, Canada	Szeicz and Macdonald, 1995	13	1700-1990	N	-	-	1.4*	0.03
12.	Montseny Mountains, Catalonia, Spain	Peñuelas and Boada, 2003	50	1945-2001	E	UA	TB	1.2*	-0.03
13.	Napaktok Bay, Labrador, Canada	Payette, 2007	2	1750-2000	L	UA	BC	1.1*	0.05
14.	Noatak, Alaska, USA	Suarez *et al.*, 1999	18	1700-1990	L	UA	BC	0.6	0.19*
15.	Putorana Mountains, Russia	Kirdyanov *et al.*, 2012	10	1500-2000	E	UA	BC	0.3	0.10
16.	Rahu Saddle, New Zealand	Cullen *et al.*, 2001	7	1700-2000	N	-	-	0.6*	0.03
17.	Rai-Iz, Urals, Russia	Devi *et al.*, 2008	144	1700-2002	E	UA	BC	0.3	0.35*
18.	Sahel, Sudan, Guinea zones, Senegal	Gonzalez, 2001	135	1945-1993	L	RW	RG	0.4*	-0.48*

	Location	Reference	Plots	Time Period	Shift type	Retracting biome	Expanding biome	Temperature change (°C century^{-1})	Precipitation change (century^{-1})
19.	Sahel, Burkina Faso, Chad, Mali, Mauritania, Niger	Gonzalez *et al.*, 2012	14	1960-2000	L	RW	RG	-0.01*-0.8*	-0.31*-0.09
20.	Scandes, Sweden	Kullman and Öberg, 2009	123	1915-2007	E	UA	BC	0.8*	0.25*
21.	Sierra Nevada, California, USA	Millar *et al.*, 2004	10	1880-2002	E	UA	TC	-0.1	0.21*
22.	South Island, New Zealand	Wardle and Coleman, 1992	22	1980-1990	E	TS	TB	0.6*	0.03
23.	Yambarran, Northern Territory, Australia	Sharp and Bowman, 2004	33	1948-2000	N	-	-	-0.06	0.35*

Table 4-2: Projected biome shifts.

Area	ΔTemperature, A.D. (°C)	Emissions scenario	Number of GCMs	Vegetation model	Number of biomes	Spatial resolution (km)	Biome change, fraction of area (%)	Criterion	Reference
Dynamic global vegetation models (DGVMs)									
World	1.5	+ <2° C	16	LPJ	2	~150	~5	P >0.80	Scholze *et al.*, 2006
World	2.4	B1	3	MC1	13	50	10	confidence >0.8	Gonzalez *et al.*, 2010
World	2.5	+2-3° C	16	LPJ	2	~150	~5	P >0.80	Scholze *et al.*, 2006
World	3.4	A1B	3	MC1	13	50	13	confidence >0.8	Gonzalez *et al.*, 2010
World	3.5	+ >3° C	16	LPJ	2	~150	~5	P >0.80	Scholze *et al.*, 2006
World	4	A2	3	MC1	13	50	16	confidence >0.8	Gonzalez *et al.*, 2010

Area	ΔTemperature, A.D. (°C)	Emissions scenario	Number of GCMs	Vegetation model	Number of biomes	Spatial resolution (km)	Biome change, fraction of area (%)	Criterion	Reference
World	3.1-4.7	historical climate and B1, A1B, A2	3	MC1	13	50	12	confidence >0.8	Gonzalez *et al.*, 2010
World	~3.5-5.5	A1B	8	CLM	5	~280	~10-30	range of GCMs	Alo and Wang, 2008
World	4.6	A1FI	1	HyLand	2	~250-375	~10	|change| >50% of area	Sitch *et al.*, 2008
World	4.6	A1FI	1	LPJ	2	~250-375	~20	|change| >50% of area	Sitch *et al.*, 2008
World	4.6	A1FI	1	ORCHIDEE	2	~250 x 375	~10	|change| >50% of area	Sitch *et al.*, 2008
World	4.6	A1FI	1	TRIFFID	2	~250 x 375	~15	|change| >50% of area	Sitch *et al.*, 2008
Africa	Un-reported	A1B	1	aDGVM	5	~30	~26	change in one GCM	Scheiter and Higgins, 2009
Amazon	2	A2	1	HadCM3LC	2	~250 x 375	~30	change in one GCM	Jones *et al.*, 2009
Europe	2.9-4.9	A2	2	LPJ-GUESS	13	~12 x 18	~30-40	change in one GCM	Hickler *et al.*, 2012
Siberia	2	+2.6° C after 130 y	none	FAREAST	2	372 sites	~5	|change| >50% of area	Shuman *et al.*, 2011
Equilibrium models									
World	2–4	A1B	10	EVE	5	~100	37	average of GCMs	Bergengren *et al.*, 2011
tropical forests	2	+2° C	16	MWCD	2	~100	<5	P >0.80	Zelazowski *et al.*, 2011
tropical forests	4	+4° C	16	MWCD	2	~100	~5	P >0.80	Zelazowski *et al.*, 2011
South America	Un-reported	A2	14	CPTEC-PVM2	13	~170	~5-40	confidence >0.75	Lapola *et al.*, 2009
West Africa	Un-reported	A2	17	GAM	5	~10	~50	weighted average of GCMs	Heubes *et al.*, 2011

Table 4-3: Summary of climatic impacts of land use/land cover changes across continents.

Region and author/s	Study type	Land use and/or land cover change	Attributed climate change	Soil, hydrological and biodiversity effects	References
a) Africa					
Africa	model evaluation by scenario analysis	deforestation to agriculture	increased climate change		Alcamo *et al.*, 2011
Soudan and Sahel	satellite image analysis: NOAA–AVHRR 8 km-resolution vegetation index (NDVI)	replacement of native vegetation by crops	explained by rainfall variability and land use change		Bégué *et al.*, 2011
b) Asia					
China	model simulations	deforestation	rainfall decreases and modulation of Enso impacts		Zhang *et al.*, 2009
Northwest China	sensitivity analysis, model, calibration and verification,	deforestation to grassland	rainfall increase and warming	runoff increase affected by climate, but modulated by land use change	SuFen *et al.*, 2008
Tibetan Plateau	review	deforestation, grassland degradation induced by population demands	flooding and droughts	snow melting at permafrost,	XueFeng and Graf, 2009
Tibetan Plateau	remote Sensing, Geographic Information, System, and Global Position System	Farmland increases by 2-3%, Unused woodland and abandoned land (74%)	warming by unused land at 4000 m	water consumption increased by cultivation	Jin *et al.*, 2010

Region and author/s	Study type	Land use and/or land cover change	Attributed climate change	Soil, hydrological and biodiversity effects	References
Indian Himalayan mountains	experimental data and soil organic carbon estimations	deforestation to croplands, and abandoned land	increase in diurnal temperatures, glacial retreat and changes in rainfall pattern	At higher altitudes (>1700m a.s.l.), influence of climate on SOC was more predominant than that of vegetation type and landform, than at lower altitudes (900–1700m a.s.l).	Martin *et al.*, 2010
Loess Plateau, China	model analysis of historical data	shrubland to grassland	rainfall decreases and warming	decreased runoff and EVT	Li *et al.*, 2009
Xinjiang, China	weather station data	switch to barren land/population demand	warming in the last 50 yr		Wu *et al.*, 2010
China	use of d-bases	quadruplication of paddy rice area	warming (+ 2ºC)		Gao and Liu, 2011
China	spectral data	switch to paddy rice	warming, CH_4 emissions and radiactive forcing increases		Gao and Liu, 2011
Tibet	use of d-bases	overgrazing	warming	SOC decreases	Dai *et al.*, 2011
Inner Mongolia, China	RS, GIS and GPS technology,	land covering waving by population demands	rainfall decreases and warming		BaoQuan *et al.*, 2009
NE Thailand	SWAT model	double population demand	warming (+4%)	runoff (+ 3-5%)	Graiprab *et al.*, 2010
c) Europe					

Region and author/s	Study type	Land use and/or land cover change	Attributed climate change	Soil, hydrological and biodiversity effects	References
Southern Europe	Model studies and carbon accounting tool	afforestation/reforestation	warming by albedo effects neutralize cooling effect of carbon sequestration	carbon sequestration	Schwaiger and Bird, 2010
Southern Europe	Bird data and community inidcators	deforestation to land abandonment, fire impacts and urbanization	warming	Bird communities in forest habitats had colder-dwelling bird species with more northern distributions than farmland, burnt or urban areas	Clavero *et al.*, 2011
United Kingdom	review	afforestation/reforestation	decreased GHG emissions. Net sink of GHG		Rounsevell and Reay, 2009
d) North America					
Mid-west USA	meteorological forcing data and AR4 GCMs	switch to cropland	reduction of radioactive forcing by snow albedo	runoff and baseflow increased by 8 and 6 mm, EVT decreased by 15 mm	Mishra *et al.*, 2010
conterminous United States (CONUS)	observation minus reanalysis (OMR) approach	deforestation	warming		Fall *et al.*, 2010
Great Lakes River Basin, USA			warming of river water	nutrient discharge in river	Wiley *et al.*, 2010
e) Oceania					
Australia	review	land clearing for agriculture	increased GHG emissions; more frequent and severe droughts; exacerbation of El Niño effects		McAlpine *et al.*, 2009

Region and author/s	Study type	Land use and/or land cover change	Attributed climate change	Soil, hydrological and biodiversity effects	References
f) South America					
Argentina	d-bases	deforestation and roturation of grasslands	increase in GHG emissions	SOC decreases and pollution by pesticides	Viglizzo *et al.*, 2011
Atlantic Forest, Brazil	experimental data	deforestation to pasture	warming	Soil temperature increases and water filled soil pore space decreases. Decreases in CH4 capture and N-cycle impoverishment	Carmo *et al.*, 2012
Amazonia, Brazil		deforestation by market driven land use change	rainfall decreases, droughts, warming and fires.		Cochrane and Barber, 2009
Mato Grosso, Brazil	GIS and NOAA/AVHRR data	deforestation (soybean and grasslands)			Yoshikawa and Sanga-Ngoie, 2011
Uruguay river (Uruguay and Argentina)	**d-bases and hydrology model**	**agricultural expansion**	**rainfall increase**	**streamflow increases by climate, runoff increases by land use change**	**Saurral *et al.*, 2008**

Table 4-4: Summary of drivers and outcomes of land use / cover scenarios associated with Representative Concentration Pathways.

RCP	Key assumptions / drivers	Land use / cover outcomes
8.5-MESSAGE	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 ha from 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	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-GCM	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	**Overall trends in land use and land cover mainly determined by demand, trade and production of agricultural products and bio-energy**	**Much agriculture relocates from high income to low income regions** **Increase in bio-energy production, new area for bioenergy crops near current agricultural areas.**

Table 4-5: Changing timing of phenological events based on observations (since AR4).

Location	Period	Species/Indicator	Observed changes (days/decade)	References
Powdermill Nature Reserve, Pennsylvania, USA	1991-2006	78 songbirds, Spring migration	+0.8	Van Buskirk *et al.*, 2009
Northern Europe	1960-2007 (with differences among sites)	43 Long distant migrant birds, Spring arrival	-0.73	Saino *et al.*, 2011
Northern Europe	1960-2007 (with differences among sites)	74 Short distant migrant birds, Spring arrival	-2.14	Saino *et al.*, 2011
Antarctica; Dumont d'Urville Station, Ade´lie Land	1950-2004	4 of 9 Seabirds, Spring arrival	+0.95 to +5.56	Barbraud and Weimerskirch, 2006
Antarctica; Dumont d'Urville Station, Ade´lie Land	1950-2005	2 of 9 Seabirds, Egg laying date	+0.51 to +0.67	Barbraud and Weimerskirch, 2006
20 European sites	1947-2007 (with differences among sites, Ø 37 years)	Common Cuckoo (*Cuculus canorus*), Spring arrival	-1.4	Møller *et al.*, 2011
20 European sites	1947-2007 (with differences among sites, Ø 37 years)	16 Short-distant migrants, Spring arrival	-3.95	Møller *et al.*, 2011
20 European sites	1947-2007 (with differences among sites, Ø 37 years)	26 Long-distant migrants, Spring arrival	-1.62	Møller *et al.*, 2011
Dlouhá Loucka, Czech Republic	1980-2005	Edible dormouse (*Glis glis*)	-8	Adamík and Král, 2008
Dlouhá Loucka, Czech Republic	1980-2006	1 (Collared flycatcher, *Ficedula albicollis*) of 4 Songbirds	-3,5	Adamík and Král, 2008
Kluane Lake, Yukon, Canada	1992-2002	mean lifetime parturition date	-18	Réale *et al.*, 2003
Tokyo, Japan	12-32 year periods within 1976-2007	3 Amphibians, spawning date	-1.2 to -3.9	Kusano and Inoue, 2008
China	1980's-2000's	72 vascular plant species, spring phenology	-0.63 to -11.63	Ma and Zhou, 2012
Rocky Mountains, Colorado, USA	1975-2008	Glacier lily (*Erythronium grandiflorum*), 1st, peak and last flowering date	-3.2	Lambert *et al.*, 2010

Table 4-6: Examples of the costs of action and inaction with respect to climate change and ecosystems.
[Types of cost: damage (D), adaptation (A), mitigation (M)]

Category of ecosystem service	Ecosystem Services	Examples of cost	Type of assessment	Ecosystem	Type of cost			Source
					D	A	M	
Provisioning	Food	**Losses in fish provision:** 30–50% decline in clupeid catch since the late 1970 in Lake Tanganyika, provider of 25–40% of animal protein supply for the populations of the surrounding countries. Attributed to decreased primary production (by about 20% over the past 80 years) probably due to alteration of nutrient supply, as an effect of rising surface temperature and falling wind speed.	Observed	Freshwater	•			O'Reilly *et al.*, 2003
		Managing delays in action to reduce fish harvest come at a financial cost. However, delays, along with poor environmental conditions due to climate change, significantly increase the probability of fish population collapse. Data of multispecies lake fisheries and European eel (*Anguilla anguilla*), among others, were used to develop the simulation model.	Modelled	Freshwater / coastal		•		Brown *et al.*, 2012
	Fresh water	The expansion of *Pinus radiata* plantations in the South African Fynbos is consistent with a climate mitigation strategy. While increasing carbon sequestration and timber production, **streamflow estimates tend to decline**. Values of carbon and water are crucial to assess economic viability of plantations under different scenarios.	Modelled	Terrestrial			•	Chisholm, 2010
	Fibre and fuel	**Declining forage provision** In California, according to most projections. The choice of valuing forage using livestock profits or the substitute fodder prices makes a large difference, with losses ranging between $14 million and $570 million.	Modelled	Terrestrial	•			Shaw *et al.*, 2011
Regulating	Climate regulation	Models for California terrestrial ecosystems show dissimilar effects. Increase in **carbon sequestering capabilities** could generate additional value of over to $300 million annually in the near future and as much as $22 billion annually by 2070. Other models predicting sharp loss in carbon storage capacity derive in social costs of –$650 million to more than –$5 billion annually for the period 2005–2034, to high as –$62 billion annually by the period 2070–2099.	Modelled	Terrestrial	•			Shaw *et al.*, 2011
	Erosion protection	Reallocation of Alaska Native villages at **risk from flooding and erosion** (184 of 213), represents a cost of USD100 million as the average cost per village (lower figure). This means 1.25% of annual federal spending in Alaska.		Terrestrial / Coastal		•		Huntington *et al.*, 2012
	Natural hazards	Change in **forest fire regulation**: Increased forest risk by 20%-30% in the southern edge of Amazonia and central Brazil	Modelled	Terrestrial	•			Cook *et al.*, 2012

Category of ecosystem service	Ecosystem Services	Examples of cost	Type of assessment	Ecosystem	Type of cost			Source
					D	A	M	
		Change in **forest fire regulation**: Increase in the number of large fire years in Alaska (decrease fire return interval, increase in fire severity). Impacts of fire in landscape	Observed	Terrestrial	•			Huntington *et al.*, 2012
		Adaptation to forest fire: In Alaska, maintaining the existing fire regime and landscape characteristics requires greater expenditure on fire prevention and fire fighting	Argued	Terrestrial		•		Huntington *et al.*, 2012
Cultural	Cultural diversity	Altered composition of **heritage** biotas. Changes in microbes and lower plants responding to environmental variability, contribute to either biodeterioration (disintegration of heritage structures) or bioprotection. Biodeterioration risk is argued in the case of some World Heritage sites (e.g. rock art site of Twyfelfontein in Namibia, the medieval stone architecture at Mapungubwe, South Africa and Petra in Jordan stone tombs at Koguryo, North Korea, rock carvings on sandstone at Alta in Norway and the Mogao caves in China).	Hypothesized / Modelled	Terrestrial	•			Viles and Cutler, 2012
Supporting	Biodiversity	In Alaska, changes in the fire regime lead to changes in the forest ecosystem structure. The **habitat** for moose increases while that of caribou declines. Both have a role as a subsistence food resource for indigenous people	Observed	Terrestrial	•			Huntington *et al.*, 2012
		Gene pool protection. Higher temperatures assumed to lead to decreases in population size of the endangered Taiwan trout (*Onocorhynchus masou formosanus*). The WTP to avoid this outcome was 0 to 33.6 US$/cap*yr.	Stated	Freshwater	•			Tseng and Chen, 2008
	Nutrient cycling	Higher ecosystem maintenance cost and reduced productivity of the forest ecosystems in the southern edge of Amazonia and central Brazil	Modelled	Terrestrial	•			Cook *et al.*, 2012
	Pollination	Based on world market prices and FAO statistics, the global value share of pollination of crops for 2005 has been calculated as 153 billion Euro . Climate change of intermediate relevance as impacting factor	Observed	Terrestrial	•			Gallai *et al.*, 2009; Kuldna *et al.*, 2009

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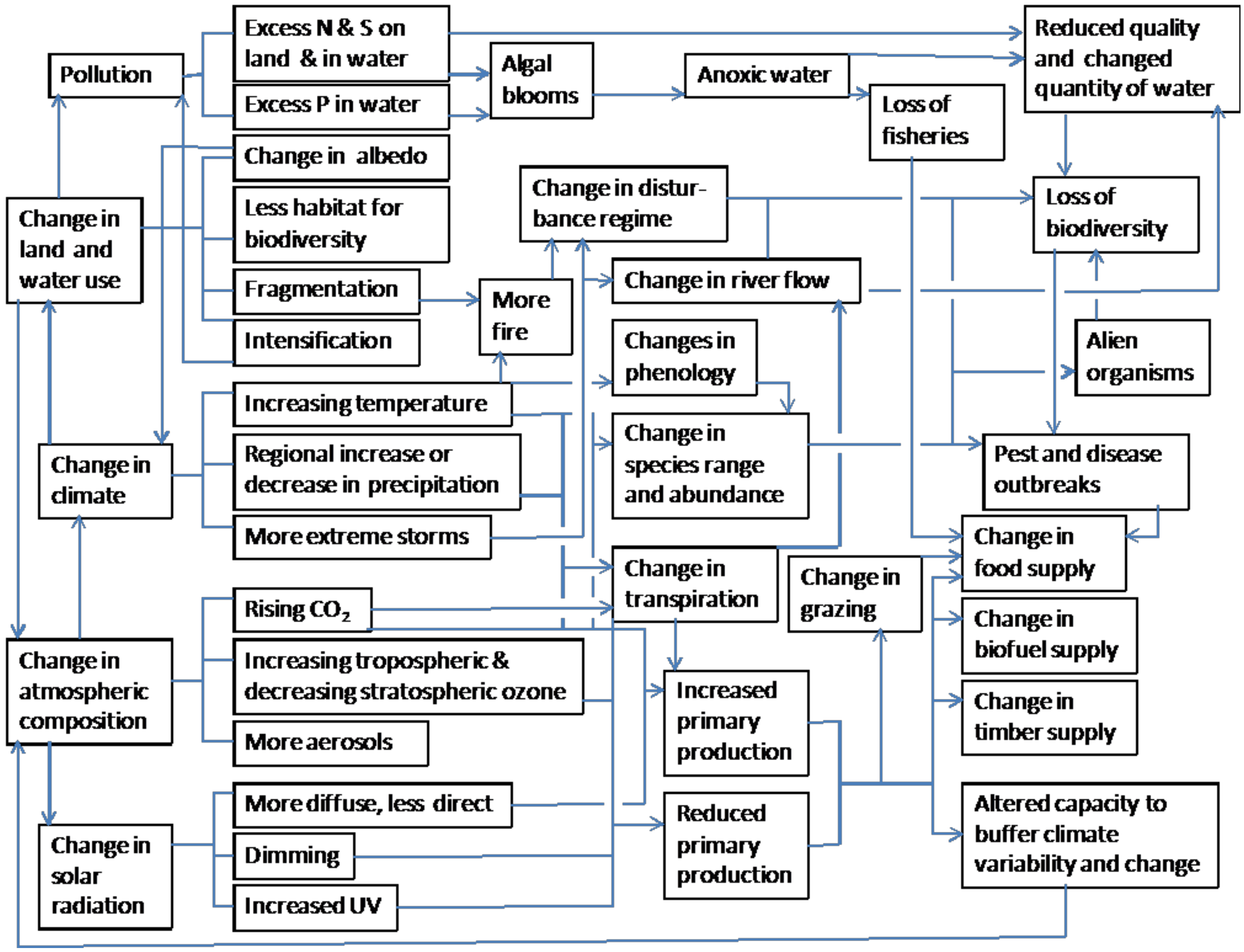


Figure 4-1: How the many aspects of climate change affect ecosystems.

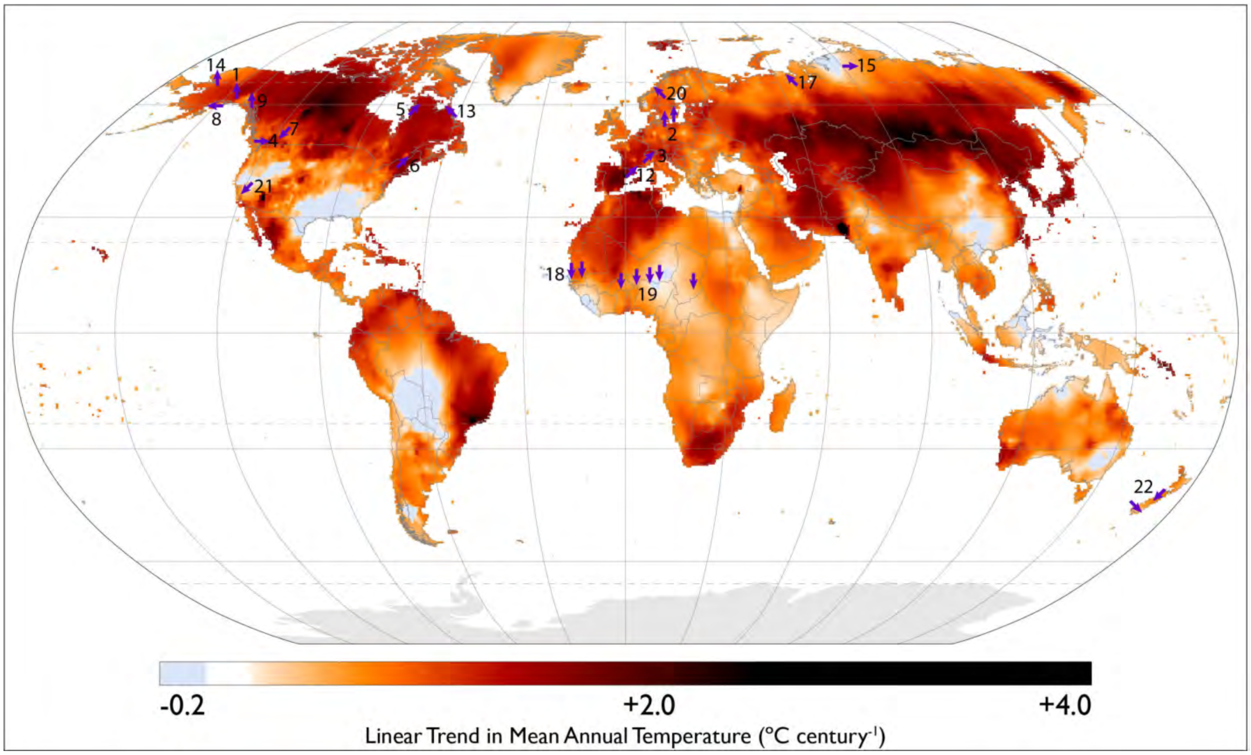


Figure 4-2: Biome shifts in the past.

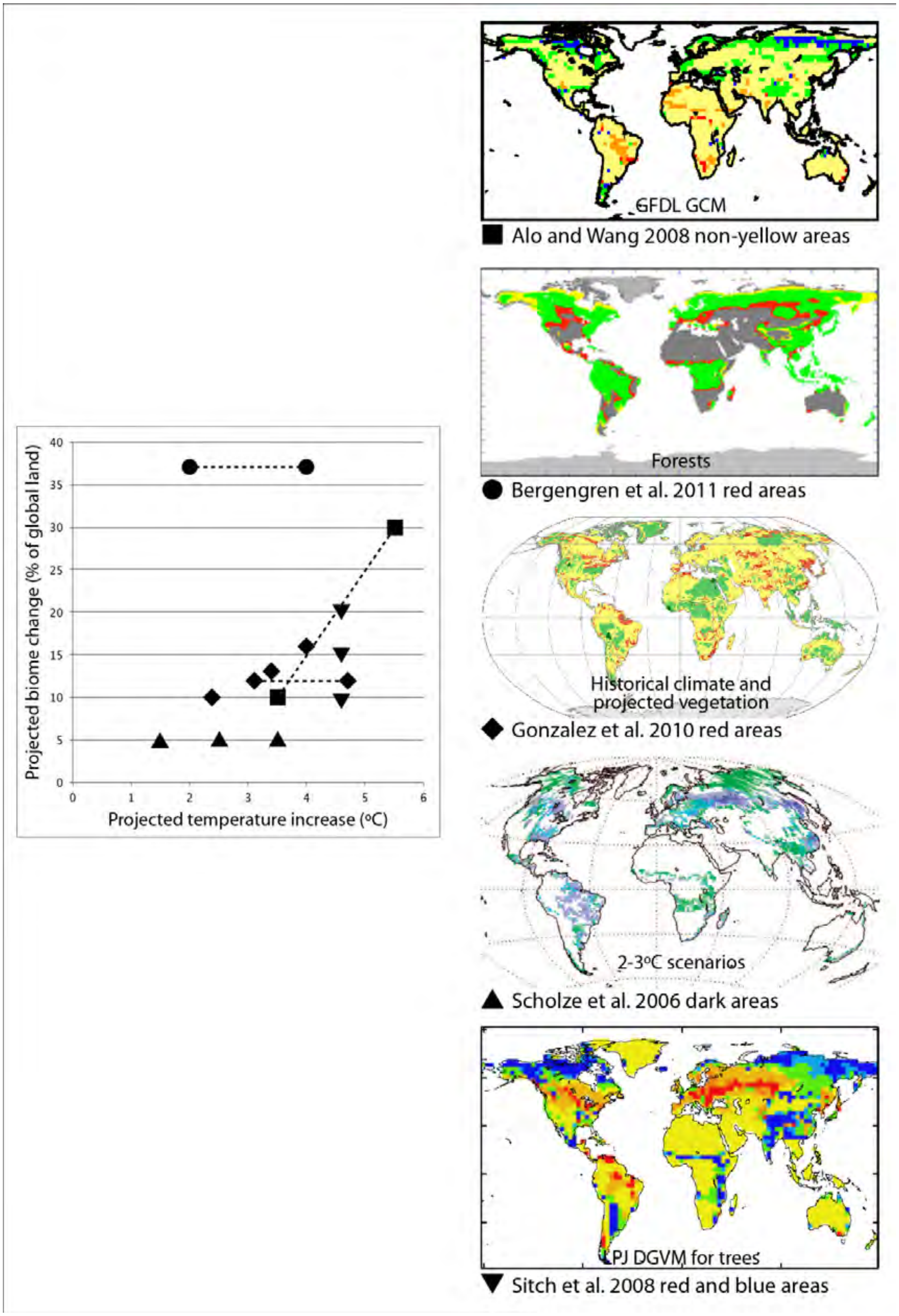


Figure 4-3: Projected biome changes.

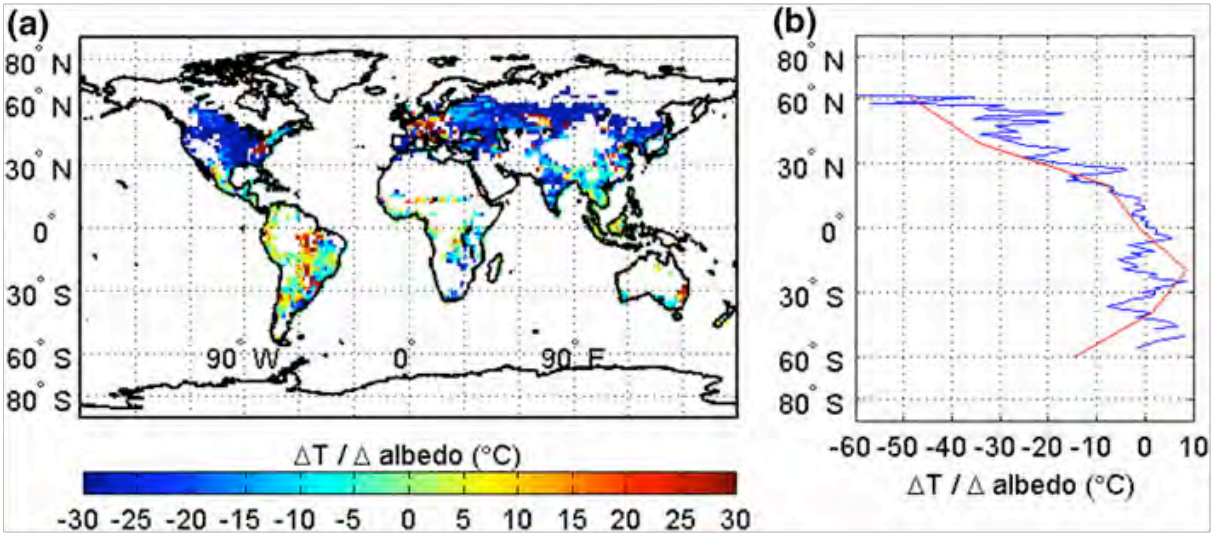


Figure 4-4: Spatial distribution of the sensitivity of mean temperature response changes in albedo resulting from land cover change.

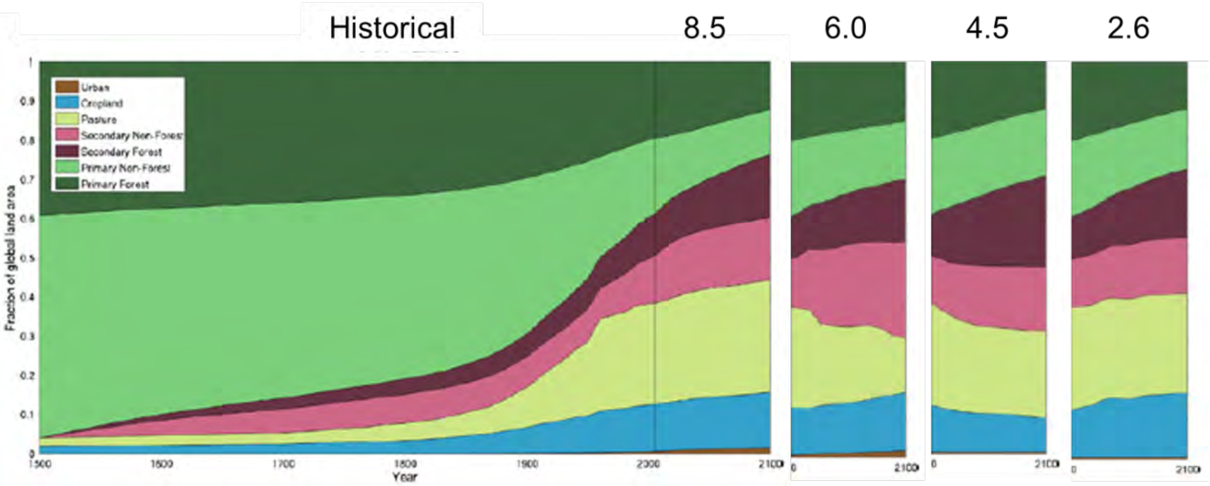


Figure 4-5: Proportion of global land cover occupied by primary and secondary vegetation (forest and non-forest), cropland, pasture and urban land, from satellite data and historical reconstructions up to 2005.

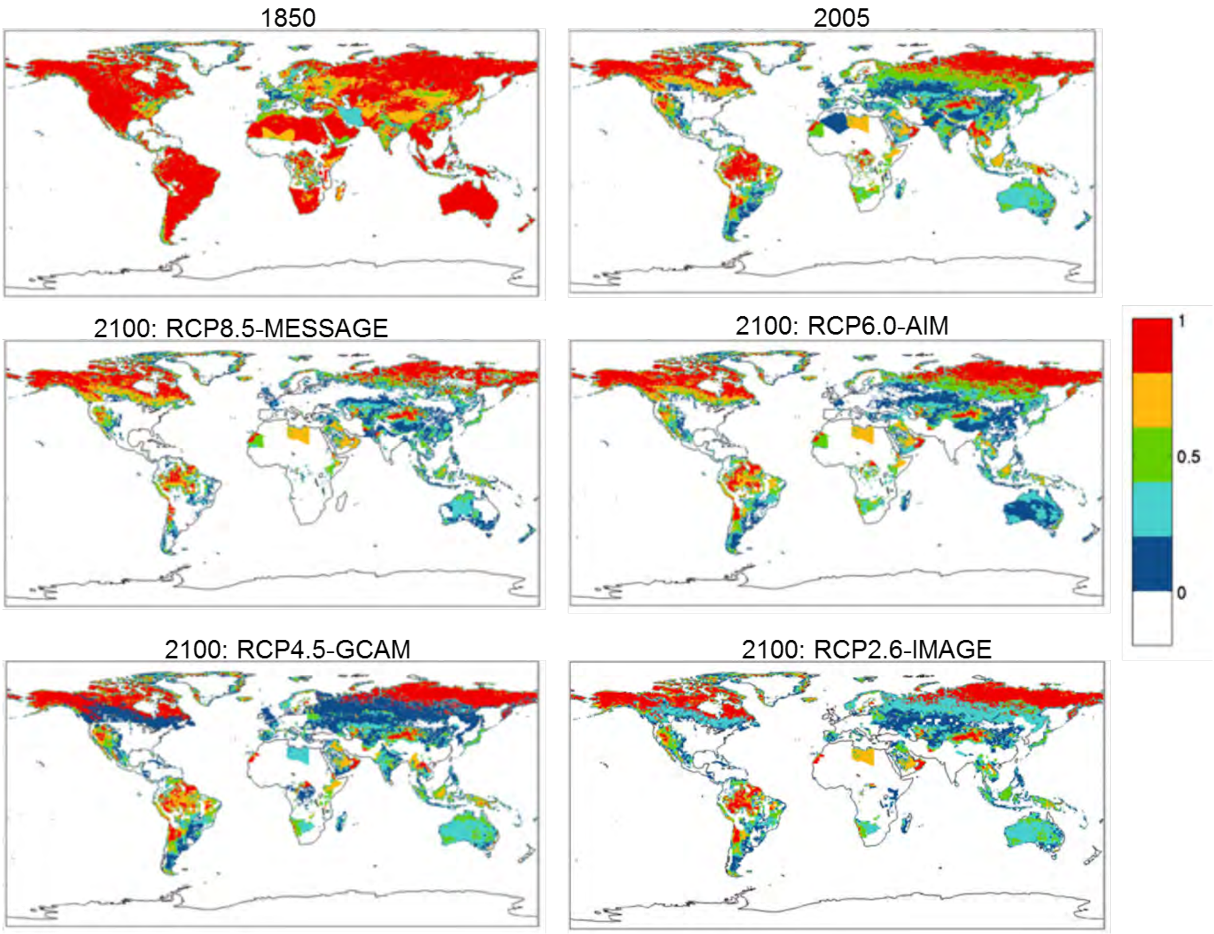


Figure 4-6: Fractional cover of primary vegetation at 1850, 2005 based on satellite data and historical reconstructions.

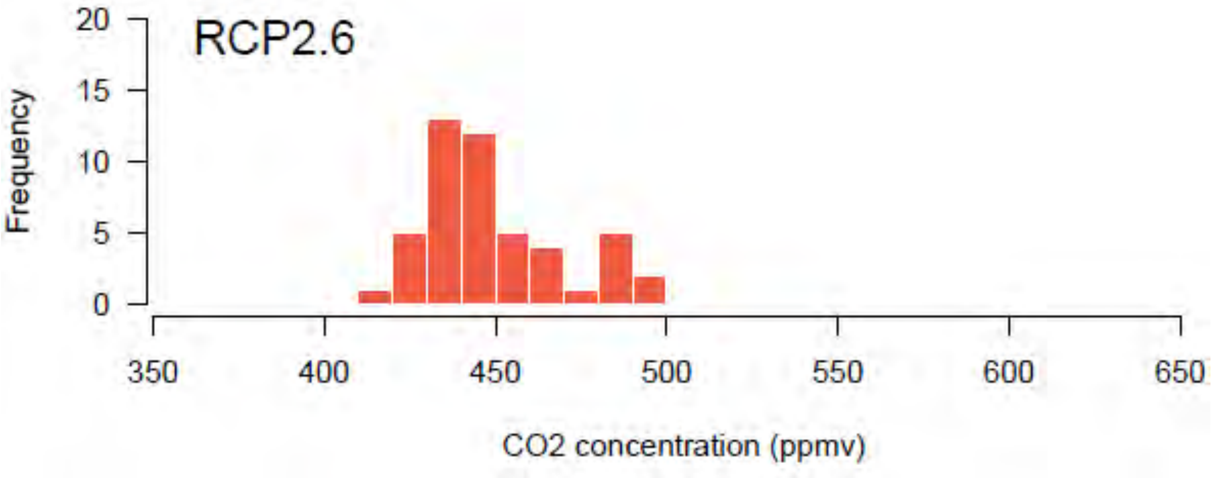


Figure 4-7: Illustration of uncertainty in the CO_2 concentration associated with 2°C global warming.

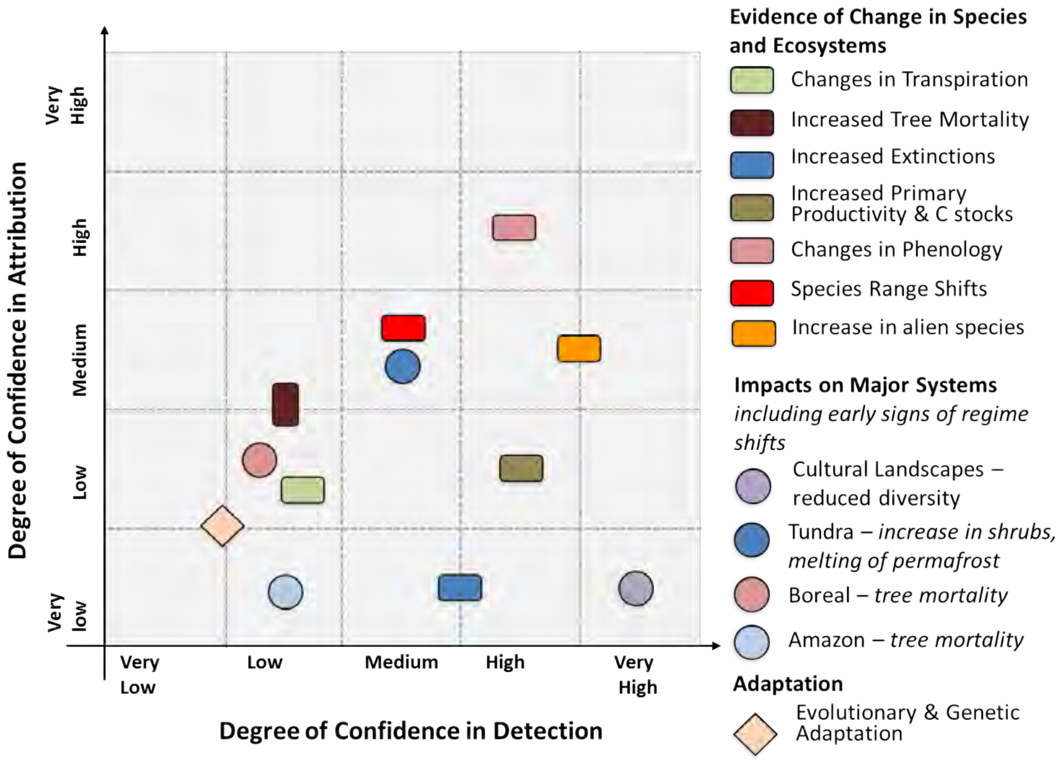


Figure 4-8: Confidence in Detection and Attribution of observed responses of terrestrial ecosystems to climate change.

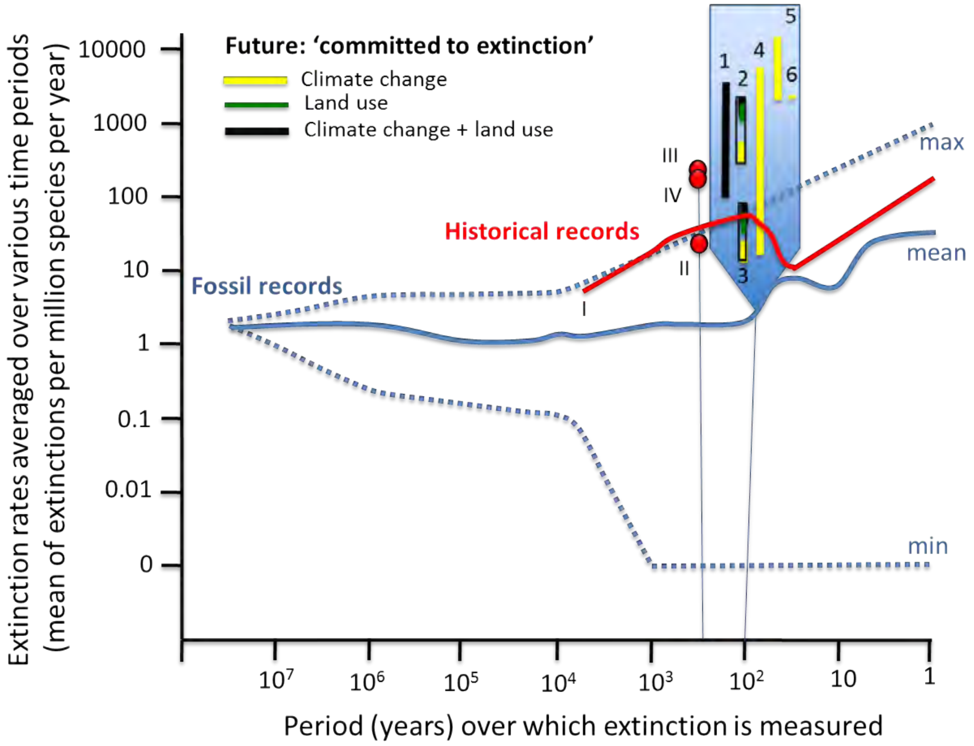
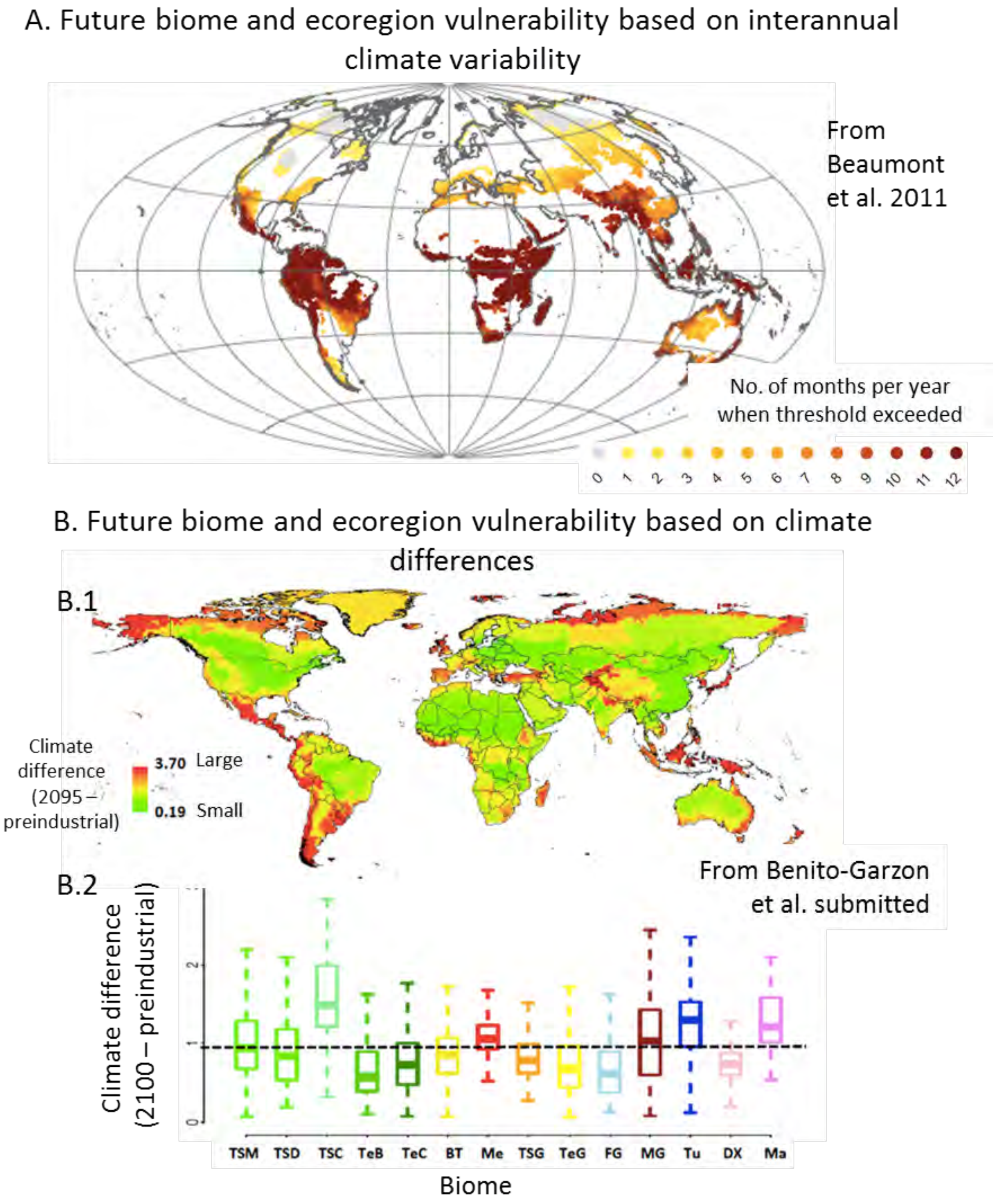


Figure 4-9: Historic and future projections of species extinctions.



A. Future biome and ecoregion vulnerability based on interannual climate variability

From Beaumont et al. 2011

B. Future biome and ecoregion vulnerability based on climate differences

From Benito-Garzon et al. submitted

Figure 4-10: Two views of the vulnerability of terrestrial biomes to future climate change.

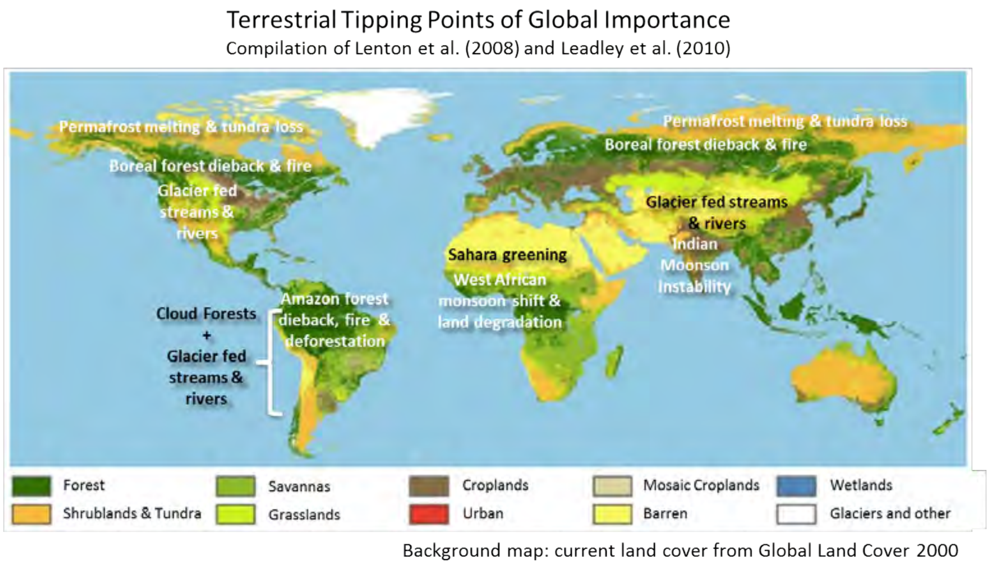


Figure 4-11: Map of tipping points.

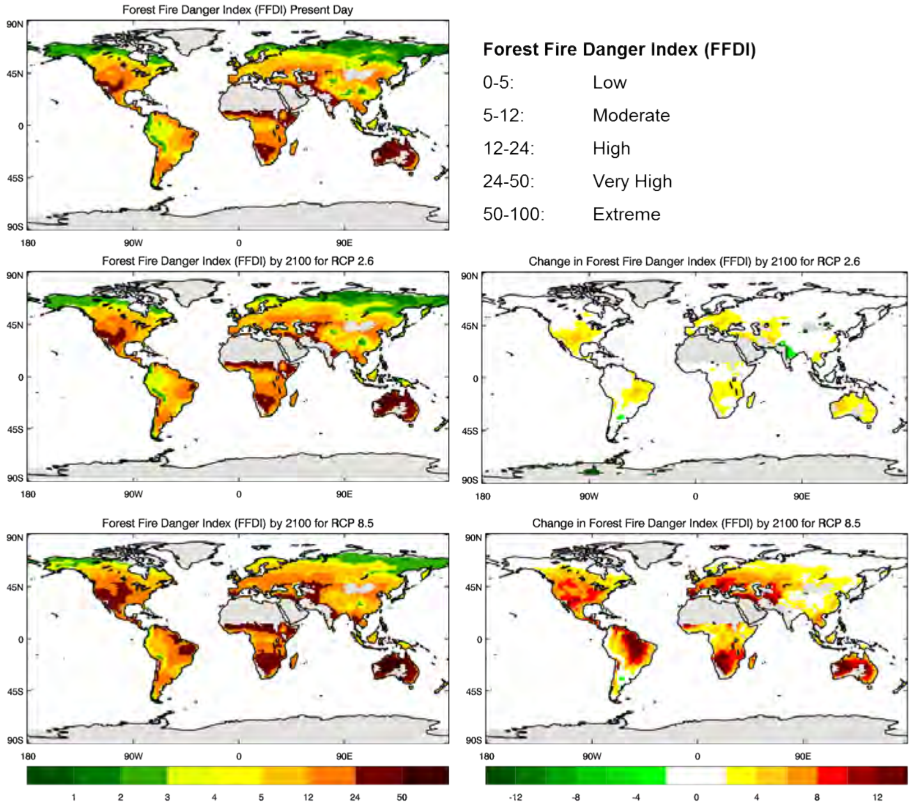


Figure 4-12: Forest Fire Danger Index simulated with the HadGEM2-ES Earth System Model.

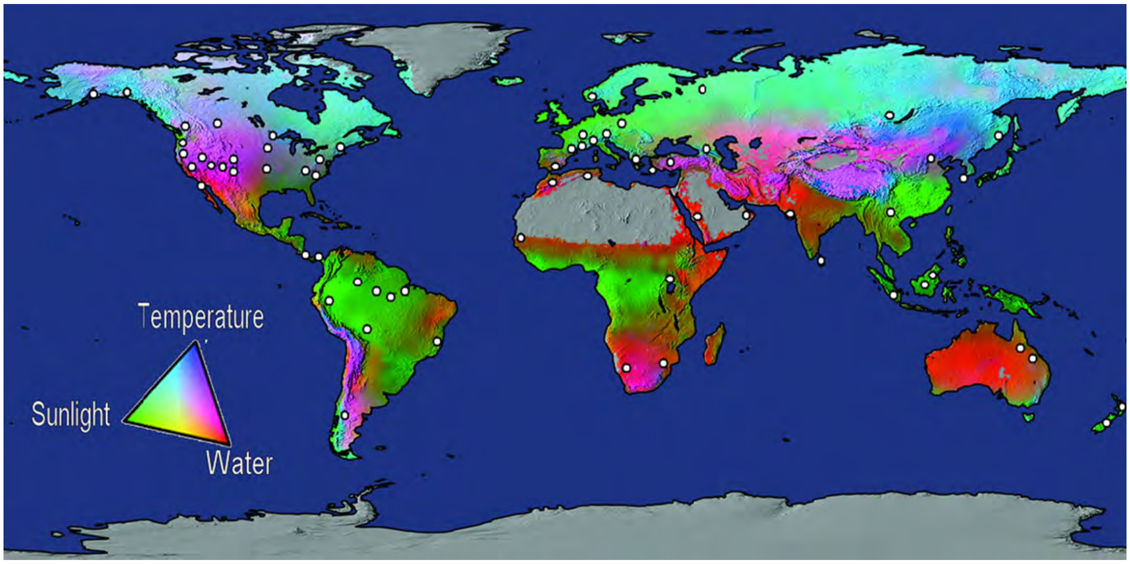


Figure 4-13: Forest mortality related to climatic stress.

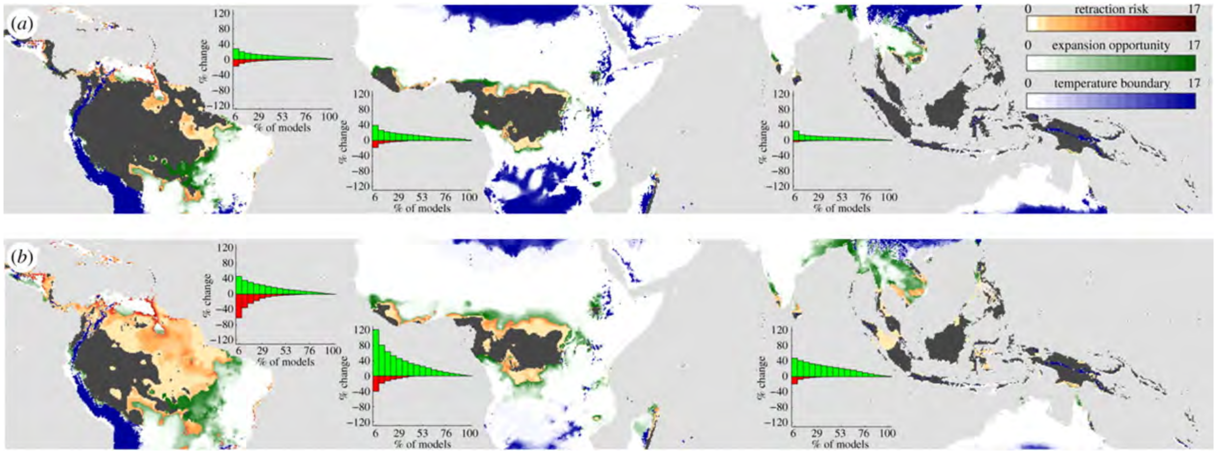


Figure 4-14: Changes in the potential climatological niche for humid tropical forests.

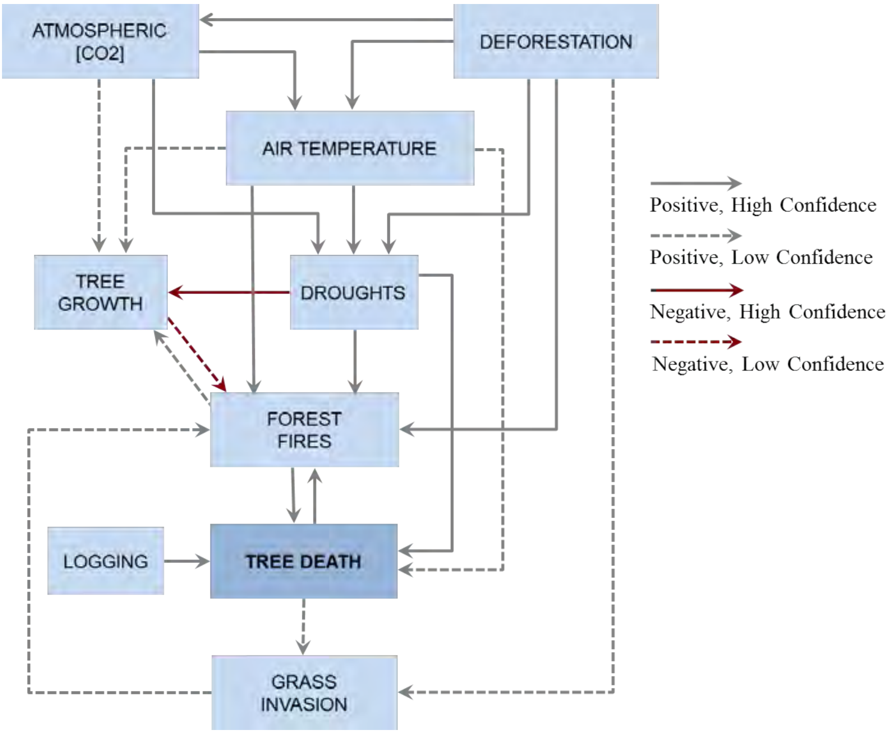


Figure 4-15: Amazon die back.

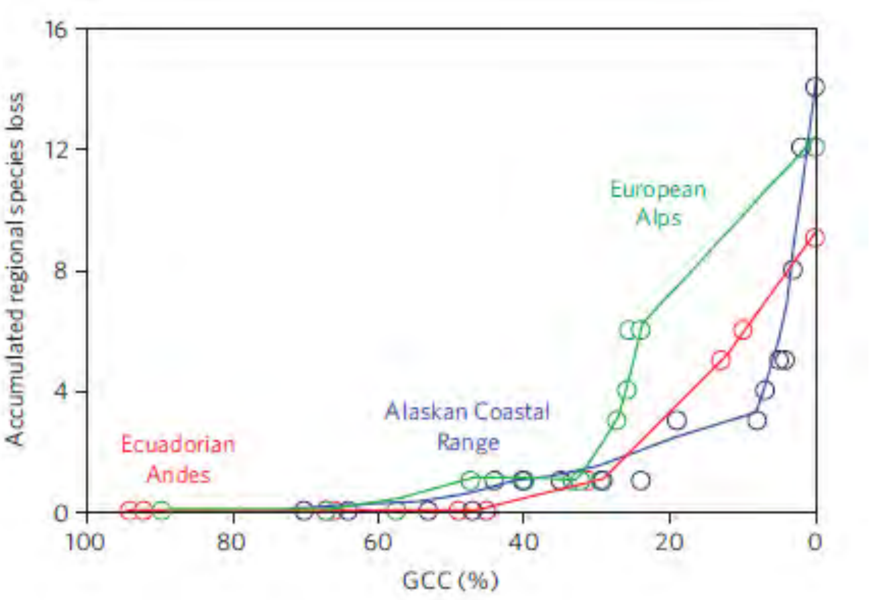


Figure 4-16: Accumulated loss of regional species richness (gamma diversity) as a function of glacial cover.

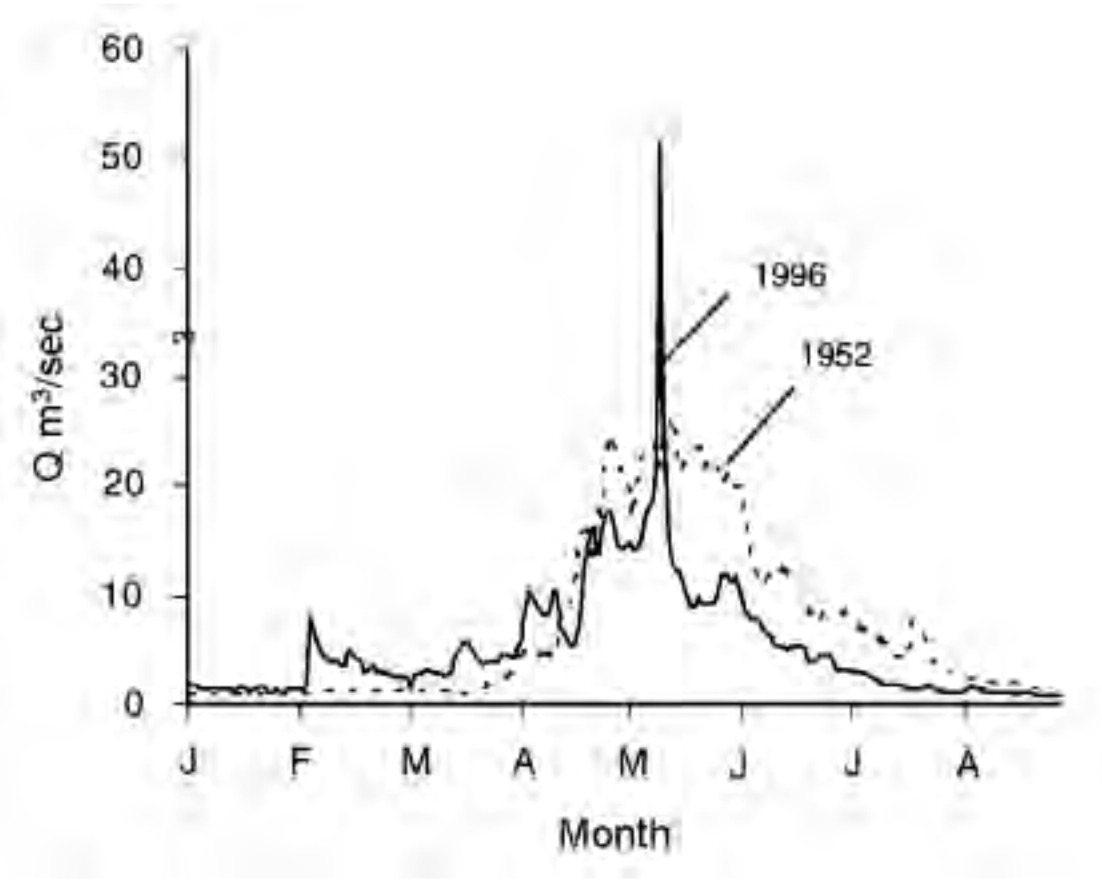


Figure 4-17: Daily average discharge in the Carson River.

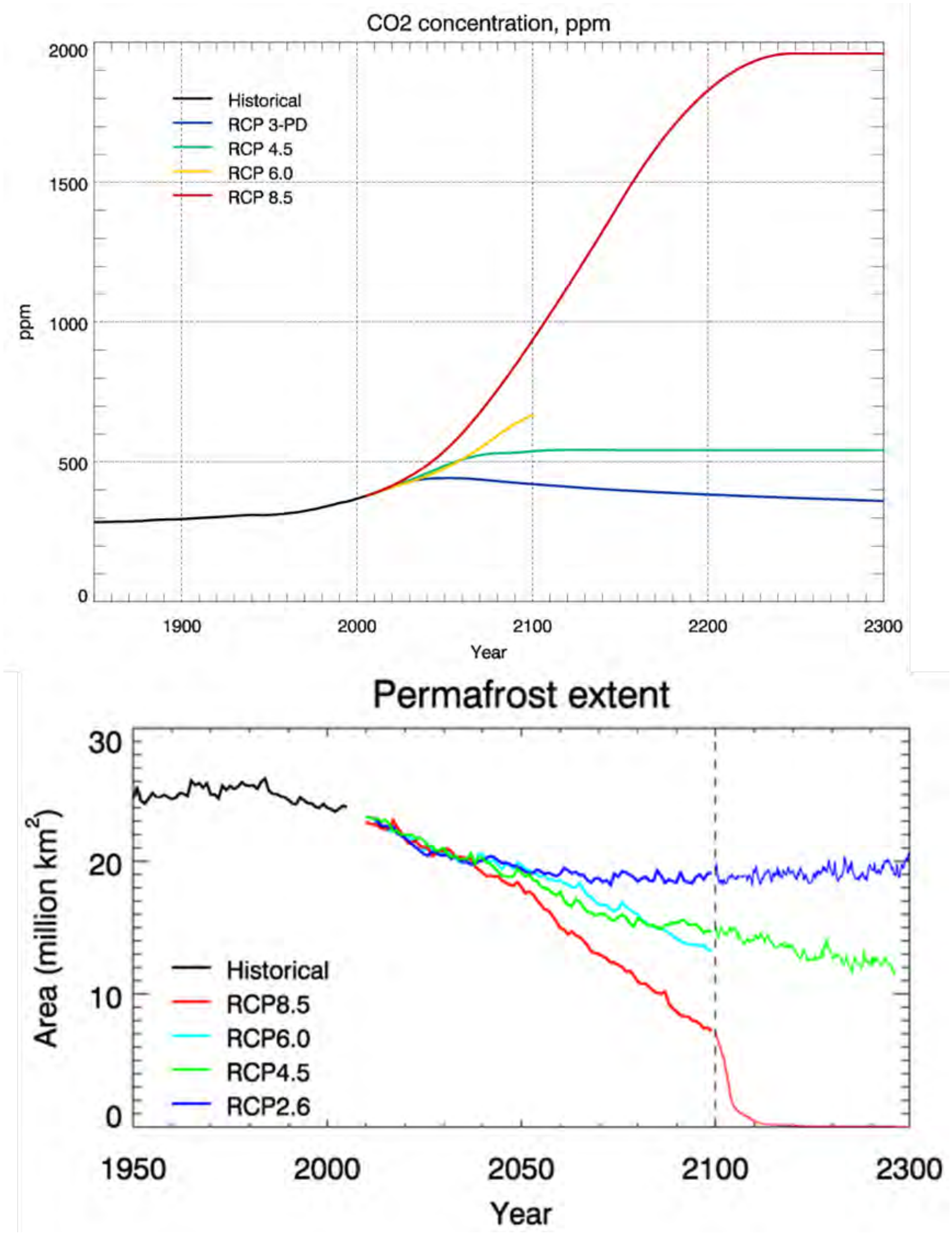


Figure 4-18: Top panel: Scenarios of atmospheric CO_2 concentration under the Representative Concentration Pathways (RCPs). Bottom panel: Simulations of past and future northern hemisphere permafrost area with a maximum thaw depth less than 3m deep.

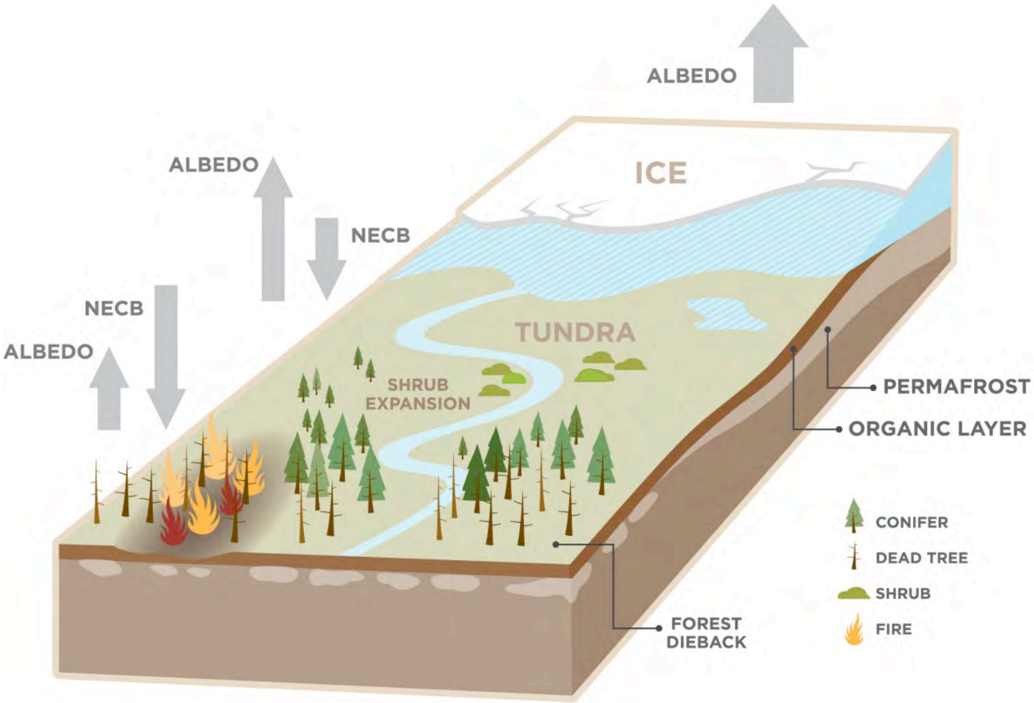


Figure 4-19: Tundra biome shift.

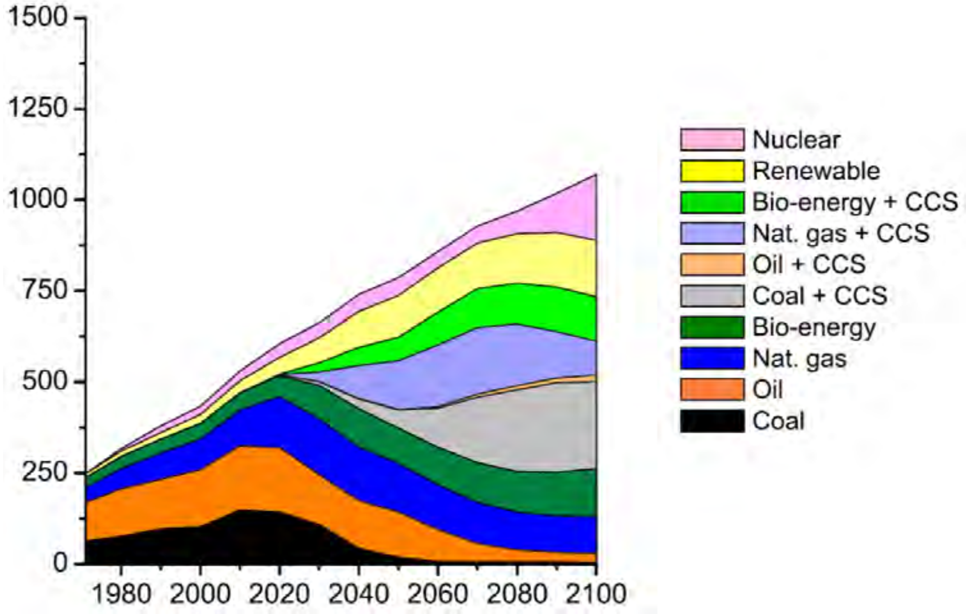


Figure 4-20: Mix of energy sources for the RCP2.6 “aggressive mitigation” scenario.