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SUMMARY OF AVAILABLE SCIENTIFIC INFORMATION ON THE VULNERABILITY OF BIODIVERSITY TO THE IMPACTS OF CLIMATE CHANGE AND MITIGATION AND ADAPTATION ACTIVITIES

*Report Submitted by the World Conservation Monitoring Centre of the United Nations Environment
Programme*

Note by the Executive Secretary

1. Annex III of decision IX/16 outlines the terms of reference for the Ad Hoc Technical Expert Group (AHTEG) on Biodiversity and Climate Change including: identifying potential biodiversity-related impacts and benefits of adaptation activities, especially in the regions identified as being particularly vulnerable under the Nairobi work programme (developing countries, especially least developed countries and small island developing States).
2. In order to facilitate the consideration of this item by the AHTEG, the World Conservation Monitoring Centre of the United Nations Environment Programme (UNEP-WCMC) was contracted to prepare a review of the impacts of climate change biodiversity. This work was completed thanks to the financial support of the Government of the United Kingdom of Great Britain and Northern Ireland.
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Impacts of Climate Change on Biodiversity

A review of the recent scientific literature

October 2008

The United Nations Environment Programme World Conservation Monitoring Centre (UNEP-WCMC) is the biodiversity assessment and policy implementation arm of the United Nations Environment Programme (UNEP), the world's foremost intergovernmental environmental organization. The centre has been in operation since 1989, combining scientific research with practical policy advice.

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Prepared by

Valerie Kapos, Jörn Scharlemann, Alison Campbell, Anna Chenery, Barney Dickson



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Executive summary

The IPCC 4th Assessment Report (AR4) concluded that climate change will have significant impacts on many aspects of biological diversity. These impacts will include effects on ecosystems, on their component species and genetic diversity within species, and on ecological interactions. The implications of these impacts are significant for the long-term stability of the natural world and for the many benefits and services that humans derive from it.

Because of the importance of these impacts and of climate change itself, there has been a great deal of recent research, which has added to the evidence base. This review draws on recent research to summarise advances since the IPCC AR4 in our understanding of the impacts of climate change on biodiversity. The evidence for these impacts comes from three principal sources: direct observation; experimental studies; and modelling studies.

The main lesson from recent research is that many of the key findings at time of IPCC AR4 have been strengthened, with a greater range of evidence, including observational evidence, to support them. While there are some specific areas where new understanding has emerged or the balance of evidence has shifted, the larger scale picture is one of increased support for earlier findings.

The IPCC AR4 described the evidence for the effects of climate change on natural ecosystems. More recent observational, experimental and modelling work has pointed to several broad types of major changes to ecosystems as a result of climate change. Modelling studies combined with experimental evidence of species tolerances point to significant changes in the distribution of some ecosystems, principally due to increasing temperature and altered precipitation regimes. Likely distribution changes include poleward shifts in boreal regions and upwards shifts in montane systems, where lack of space at higher altitudes may cause some systems to disappear entirely. Ecosystem distribution changes are also expected to be large in the tropics, where the effects of rising temperatures and reduced precipitation are exacerbated by the effects of land use change. Drier conditions are expected to cause savanna ecosystems to move into equatorial regions now occupied by forests.

In addition to shifting their locations climate change will alter the composition of many ecosystems. Site level reductions in species richness are of concern because under changing environmental conditions, multiple species play a role in ensuring that ecosystem processes can continue. Processes potentially dependent on species richness include carbon storage. Climate change can also facilitate the spread and establishment of invasive species, which can have major impacts on ecosystem composition.

Changes in species composition can lead to changes in the physical and trophic structure of ecosystems, with resulting further effects on system function and composition. One such change is the invasion of temperate grasslands by woody plants. In other systems, trees may disappear as a result of drought. Coral reefs are especially subject to adverse impacts from climate change due to bleaching and diseases promoted by warmer temperatures and increasing pressures from ocean acidification. Many reef-building coral species are threatened with extinction. This has major implications for the large biological communities that coral reefs support.

Climate changes in combination with changes in ecosystem composition and structure have been shown both by modelling and experimentation to lead to changes in ecosystem function. Models suggest that global net primary production (NPP) has already increased in response to changes in temperature and precipitation during the 20th century. Regional modelling projects increases in NPP for some regions, but possible declines in others.

A key property of ecosystems that may be affected by climate change is the values and services they provide to people. These include provisioning services such as fisheries, which may improve in the short term in boreal regions and decline elsewhere, and timber production, where the response depends on population characteristics as well as local conditions and may include large production losses. The impacts on coral reefs threaten the vital ecosystem services these systems provide through fisheries, coastal protection and building materials. Climate change also affects the ability of terrestrial ecosystems to regulate water flows, and critically reduces the ability of many different ecosystems to sequester and/or retain carbon which can feedback to climate change.

At the species level recent observed and modelled climate change impacts show that climatic change has already caused changes to the distribution of many plants and animals, leading to severe range contractions and the extinction of some species. Changes of terrestrial species include shifts in spring events (for example, leaf unfolding, flowering date, migration and time of reproduction), species distributions and community structure. In marine-ecosystems changes have been demonstrated in functioning and productivity, including shifts from cold-adapted to warm-adapted communities, phenological changes and alterations in species interactions. Some species are unable to disperse or adapt fast enough to keep up with high rates of climate change and these species face increased extinction risk and, as a result, whole ecosystems, such as cloud forests and coral reefs, may cease to function in their current form. The IPCC AR4 estimated that 20-30% of species assessed would be at risk of extinction if climate change leads to global average temperature rises greater than 1.5 -2.5°C. Recent work suggests that for birds, amphibians and warm water corals as many as 35-70% may be at risk from climate change.

Despite its importance, relatively little effort has yet been devoted to investigating the impacts of climate change on genetic diversity. One clear impact is the fragmentation of populations when their habitats are fragmented by climate change. There may also be impacts on crop wild relatives, which are an important source of genetic diversity for crop diversity.

Climate change is likely to affect ecological interactions, including competition, disease and host-parasite interactions, pollination and herbivory. There is ample evidence that warming will alter the patterns of plant, animal and human diseases. Numerous modelling studies project increases in economically important plant pathogens with warming, and experimental studies show similar patterns. There is evidence that climate change may play a role in changing the distribution of diseases. Short-term, local experiments have demonstrated the impacts of predicted global change on plant health including rice.

Recent evidence suggests that mismatches in phenological responses to climate change between plants and pollinators may significantly affect their interactions. These patterns could lead to the extinction of pollinators and/or plants and disruption of their interactions.

Climate change impacts on ecosystems can exert significant positive feedbacks to the climate system. It is generally agreed that one of the main feedbacks to the climate system will be through the increase in soil respiration under increased temperature, particularly in the arctic with the potential to add 200ppm CO₂ to the atmosphere by 2100. One area of research that has expanded since the 4AR is that of the projected Amazon drying and dieback. Although there is still considerable uncertainty, most models predict reduced precipitation leading to increased drying of the Amazon rainforest. It has been suggested that CO₂ emissions will be accelerated by up to 66% due to feedbacks arising from global soil carbon loss and forest dieback in Amazonia as a consequence of climate change.

Although ecosystems are currently acting as a carbon sink to sequester 30% of anthropogenic emissions, global scale climate scenario modelling suggests that the terrestrial biosphere will become a carbon source by 2100, largely due to the increased soil respiration and the dieback of the Amazon.

1 Introduction

1.1 Purpose and scope of review

The IPCC fourth assessment report concluded that climate change will have significant impacts on many aspects of biological diversity. These impacts will include effects on ecosystems, on their component species and genetic diversity within species, and on ecological interactions. The implications of these impacts are significant for the long-term stability of the natural world and for the many benefits and services that humans derive from it.

Because of the importance of these impacts and of climate change itself, there has been a great deal of recent research, which has added to the evidence base. This review draws on recent research to summarise advances since the IPCC 4th Assessment Report (AR4) in our understanding of the impacts of climate change on biodiversity (IPCC 2007).

The evidence for these impacts comes from three principal sources: direct observation of changes in components of biodiversity in nature (either recently or in the distant past) that can be clearly related to changes in climatic variables; experimental studies where manipulations are used to elucidate responses; and modelling studies where our current understanding of the requirements and constraints on the distributions of species and ecosystems are combined with modelled changes in climatic variables to project the impacts of climate change and predict future distributions and changes in populations. Of the three main approaches to modelling (correlative, mechanistic and analogue; AR4), correlative modelling is by far the most common. It uses knowledge of the spatial distribution of species to derive functions that relate the probability of their occurrence to climatic and other factors (Botkin et al. 2007). Although correlative modelling has been criticised for assumptions of equilibrium between species and current climate, an inability to account for variability within species, for interactions between them or for population processes and migration, it has been used successfully to simulate known species range shifts in the distant (Martinez-Meyer et al., 2004) and recent (Araújo et al., 2005) past and has also recently been shown to be useful in simulating known population trends (Green et al. 2008) and provide a pragmatic tool for studying the potential impacts of future climate changes. Each of these sources and modelling approaches has advantages and disadvantages. This review attempts to distinguish among these classes of evidence and to address the implications for human well-being of the biodiversity impacts they project.

1.2 Dimensions of climate change

Climate change is a major global threat (Stern 2008) that has already had an observed impact on natural ecosystems. Global average temperatures have risen by 0.7°C over the last century and are predicted to continue rising. The IPCC (2007) projects that temperatures are likely to have risen 1.1°C to 6.4°C by the end of the 21st century relative to the 1980-1999 baseline. Although such projections do not account for mitigation policies, it is widely accepted that temperature rises are likely to surpass the lower bound, particularly as current models do not take into account climate-carbon cycle feedbacks.

Temperature rises are linked to changes in precipitation regimes, which can be predicted with less confidence as they are largely influenced by regional processes (Parra & Monahan 2008). Depending on the region, precipitation may be projected to increase, decrease and/or change in seasonal distribution. Increased incidence and severity of extreme events, such as hurricanes, tornadoes, catastrophic rainfall and drought, is also likely. Recent advances have improved understanding of the complex linkages between sea surface temperatures and precipitation regimes on land (Good et al. 2008; Harris et al. 2008) and helped to confirm that drought is indeed increasing in the Amazon (Li et al. 2008). Understanding

precipitation regimes and their influence is vital for projecting changes in many natural systems (Knapp et al. 2008).

It is also important to recognize that local climatic regimes comprising the full suite of climate variables are what influence the survival of species and ecosystems. With climate change, areas of rare climates are likely to shrink, and may result in the loss of rare endemic species (Ohlemuller et al. 2008).

Other consequences of climate change that are critically important for many natural systems include sea level rise and the melting of arctic sea ice, which have been observed globally and are projected to continue (IPCC, 2007).

Atmospheric concentrations of CO₂ can themselves have important direct influences on biological systems, which can reinforce or act counter to responses to climate variables and complicate projection of future responses. The direct effects of elevated atmospheric CO₂ are especially important in marine ecosystems and in terrestrial systems that are not water-limited.

Climate change is not the only pressure acting on natural systems and its effects are strongly dependent on **interactions** with these other pressures. Land use change and the attendant habitat loss are currently major threats to biodiversity worldwide. They reduce organisms' abilities to adjust their distributions in response to changing climate. They also produce local amplification of some climate change effects by causing fragmentation, degradation and drying of ecosystems (Barlow & Peres 2008). Climate change and land use change, including agricultural expansion, interact to increase the incidence of fire (Aragao et al. 2008), which is often raised still further during extreme climatic events like El Niño (Bush et al. 2008). Pollution is also likely to amplify many impacts of climate change, as is mining and oil and gas development (Fuller et al. 2008). Thus it is vital to consider the effects of climate change in the context of interacting pressures and the influence they may exert directly on natural systems and on those systems' abilities to respond to climate change.

2 Ecosystems

2.1 Types of impacts on ecosystems

The IPCC AR4 described the evidence for the effects of climate change on natural ecosystems. More recent observational, experimental and modelling work has pointed to several broad types of major changes to ecosystems as a result of climate change. Modelling studies combined with experimental evidence of species tolerances point to significant changes in the **distribution** of some ecosystems, principally due to increasing temperature and altered precipitation regimes. Such changes will happen first at present boundaries between ecosystem types (Thomas et al. 2008), and their actual occurrence is dependent on the ability of component species to migrate and to the availability of suitable substrates. Likely distribution changes include poleward shifts in boreal regions (Alo & Wang 2008; Metzger et al. 2008; Notaro et al. 2007; Roderfeld et al. 2008; Wolf et al. 2008a) and upwards shifts in montane systems, where lack of space at higher altitudes may cause some systems to disappear entirely. Ecosystem distribution changes are also expected to be large in the tropics, where the effects of rising temperatures and reduced precipitation are exacerbated by the effects of land use change (Lee & Jetz 2008). Drier conditions are expected to cause savanna ecosystems to move into equatorial regions now occupied by forests (Salazar et al. 2007), and altered precipitation regimes will also affect distribution of dryland ecosystems (Thomas et al. 2008). Projected changes in ecosystem distributions vary regionally (Metzger et al. 2008; Pompe et al. 2008). There is only very limited scope for changes in distribution of aquatic ecosystems, other than through the local disappearance some ecosystems (e.g. wetlands (McMenamin et al. 2008)) or change in physical type (e.g. river channels). Rising sea level will cause shifts in location of those coastal ecosystems that can move.

In addition to shifting their locations climate change will alter the **composition** of many ecosystems. Some observational studies have already documented species turnover and attendant changes in species richness within both terrestrial and aquatic ecosystems, especially at temperate latitudes (e.g. (Daufresne & Boet 2007; Lemoine et al. 2007; Moritz et al. 2008) but also in the tropics (Bunker et al. 2005; Bush et al. 2008; Phillips et al. 2008), as species less tolerant of new conditions are replaced by those with greater tolerance for warmer and drier conditions and increased fire occurrence. Modelling studies identify many more examples of likely species turnover (Buisson et al. 2008; Colwell et al. 2008; Levinsky et al. 2007; Trivedi et al. 2008b). Rising temperatures are a key factor in such turnover, but changing precipitation regimes are also important and rising CO₂ concentrations have important effects in the marine environment and where they favour C₃ plants such as trees over C₄ grasses. Their actual occurrence is dependent on the pool of available species and their migration rate, e.g. (Colwell et al. 2008). In some cases, the arrival of new species has been observed to lead to modest and probably transient increases in overall species richness (Buisson et al. 2008) in an ecosystem, but when species with the appropriate tolerances cannot reach a site, loss of intolerant species can lead to an overall impoverishment (Colwell et al. 2008; e.g. Deutsch et al. 2008; Huntley et al. 2008a). Site level reductions in species richness are of concern because under changing environmental conditions, multiple species play a role in ensuring that ecosystem processes can continue (Hobbs et al. 2007b). Processes potentially dependent on species richness include carbon storage (Bunker et al. 2005), so compositional changes may have important feedback effects on climate change. Regional losses in overall species richness can be exacerbated by land use changes (Higgins 2007). There is also increasing concern of the role of climate change in facilitating the spread and establishment of invasive species, which can have major impacts on ecosystem composition (Hellmann et al. 2008; Hobbs et al. 2007b; Rahel et al. 2008; Rahel & Olden 2008). Climate change has been recognised as one of several interacting factors that can enable native species to become invasive (van der Wal et al. 2008).

There is also rising concern that changes in species composition also lead to changes in the physical and trophic **structure** of ecosystems, with resulting further effects on system function and composition. Observed structural changes in forests include accelerating forest turnover and associated gap formation in the tropics (Phillips et al. 2008). An increased abundance of lianas has been observed in both tropical and temperate forest systems (Allen et al. 2007; Phillips et al. 2008) and related both to forest disturbance (Londre & Schnitzer 2006; van der Heijden & Phillips 2008) and low rainfall (Swaine & Grace 2007). Model-based analyses of selected geographical areas indicate that climate-related changes in GPP often lead to significant changes in ecosystem structure and dynamics. (Woodward et al. 2008). Another structural change that has been observed, induced in experimental manipulations and projected is the invasion of temperate grasslands by woody plants, which is facilitated by increasing CO₂ concentrations (Bloor et al. 2008; Morgan et al. 2007) and alters the availability of food for grass-eating herbivores. In other systems, trees may disappear as a result of drought (Badgley et al. 2008; February et al. 2007; Foden et al. 2007) and increase the probability of extinction for herbivores unable to digest C₄ grasses, as well as the dispersal and dynamics of other plant species. Reduced calcification due to ocean acidification will change the structure of reef ecosystems even with relatively small increases in atmospheric CO₂ (Cao 2008).

Climate changes in combination with changes in ecosystem composition and structure have been shown both by modelling and experimentation to lead to changes in ecosystem **function**. Models suggest that global net primary production (NPP) has already increased in response to changes in temperature and precipitation during the 20th century (Del Grosso et al. 2008). Regional modelling also projects increases in NPP for some regions, for example 35-54% for northern European ecosystems (Olesen et al. 2007) as a result of longer growing seasons and higher CO₂ concentrations. However, where water balance is more important, as in southern Europe, NPP is projected to decline or to increase only slightly relative to present day conditions. In contrast, experimental work using rain-out shelters has examined the response to reduced rainfall of ecosystems ranging from temperate grassland (De Boeck et al. 2008) to tropical rainforest (Brando et al. 2008) and has shown that soil drying has major and rapid effects on productivity, reducing biomass production by 10-30%. Changes in productivity will result in changes in litterfall and nutrient cycling. Where litterfall increases, it may contribute to increasing respiration and loss of soil carbon (Sayer et al. 2007).

Another aspect of ecosystem function that will certainly be affected by climate change is **phenology**. Many different approaches have been used to address this issue, but a fully coherent picture of likely responses has yet to emerge (Cleland et al. 2007). Long term observational data play a key role. They show that in warm temperate forests warming accelerates spring budburst and delays autumn leaf fall (Fujimoto 2008).

Changing climatic variables can have a profound influence on **successional processes and community dynamics**. Long term observational data on increasing rates of tree turnover in Amazonian forests (Phillips et al. 2008) are thought to reflect the effects of increased atmospheric CO₂ on tree growth (Lloyd & Farquhar 2008). Succession following glacial retreat in the alps involved different species and dynamics than had previously been observed (Cannone et al. 2008). Long-term observations showed that rainfall amounts and distributions were the key factors determining community dynamics and species dominance in a Californian grassland (Hobbs et al. 2007a).

A final, key property of ecosystems that may be affected by climate change is the **values and services** they provide to people. These include provisioning services such as fisheries, which may improve in the short term in boreal regions (Arnason 2007; Brander 2007) and decline elsewhere, and timber production for example by lodgepole pine, where the response depends on population characteristics as well as local conditions and may include large production losses (O'Neill et al. 2008). Climate change affects the ability of montane and other ecosystems to regulate water flows (Nunes et al. 2008; Ruiz et al. 2008), and

critically reduces the ability of many different ecosystems to sequester and/or retain carbon e.g. (Bunker et al. 2005; Morales et al. 2007; Wang et al. 2008), which can feedback to climate change.

The evidence that climate change can profoundly influence host–pathogen dynamics is growing, not only for plant diseases but also for animal and human diseases (e.g. Purse et al., 2005; Haines et al., 2006). The impacts of climate change on the range of the tick-borne disease Theileriosis (East Coast fever (ECF) in sub-Saharan Africa, the Northern Cape and Eastern Cape provinces of South Africa, Botswana, Malawi, Zambia and eastern DRC show increases in ECF suitability. Other areas in sub-Saharan Africa show different rates of range alteration.

Enhanced phytoplankton blooms favour cyanobacteria, resulting in increased threats to ecological status of Lakes and increased health risks (EEA 2008).

2.2 Ecosystem types

2.2.1 Deserts

According to the IPCC AR4, Deserts are likely to experience more episodic climate events, and interannual variability may increase in future, though there is substantial disagreement between projections and across different regions. Continental deserts are likely to experience more severe and persistent drought, but their impacts on productivity may be somewhat offset by the effects of increased atmospheric CO₂ concentrations, at least during wetter periods. Reduced biomass in deserts is likely to increase the fragility of soils and their vulnerability to erosion. Many desert species are vulnerable to increase in temperature and alteration of the rainfall regime will put at risk species that depend on rainfall events to initiate breeding. The Succulent Karoo Biome, a biodiversity hotspot, faces the loss of 2,800 plant species.

There is some recent evidence of expansion of desert ecosystems, through both shifting climatic regimes and anthropogenic degradation, at least in the Sahel, where an observed southward shift of the climate zones has been accompanied by the movement of species previously considered strictly Sahelian into the Sudanian zone (Wittig et al. 2007). Similarly, detailed census data on a long-lived Namib Desert tree, provide strong evidence that a developing range shift in this species as a response to climate change (Foden et al. 2007). They also show a marked lag between trailing edge population extinction and leading edge range expansion. Such a pattern is likely to apply to many of the sessile and poorly dispersed organisms characteristic of deserts and to increase their vulnerability to climate change.

Also important for changing distributions and populations of desert species is the occurrence of extreme climatic events such as severe droughts and catastrophic rainfall events. Severe droughts may cause die-off, which has severe implications for the population dynamics of long-lived desert plants and may therefore play an important role in determining future species composition (Miriti et al. 2007). Occasional catastrophic rainfall may cause dramatic flooding, which has been shown in at least one case (Thibault & Brown 2008) to cause significant species-specific mortality (of desert rodents) and thereby change dominance and interactions among species and alter long-term population and community trends.

The few experimental data available on elevated temperature and drought tolerance of desert species suggest susceptibility of leaf succulent species, but high drought tolerance of non-succulent shrubs (Musil et al. 2005). Such differential susceptibility is likely to lead to major changes in both composition and structure of desert ecosystems.

Some modelling work suggests that in China open shrub and desert steppe are among the ecosystems likely to be most severely affected by climate change (Wu et al. 2007).

The vulnerability of specific desert systems and their biodiversity has also been shown through bioclimatic modelling. Simple modelling approaches show strong reduction in spatial extent of bioclimates typical of Namaqualand, the heart of the Succulent Karoo, within the next five decades (Midgley & Thuiller 2007) and that both generalist species with large geographic ranges, and narrow-range endemics may be susceptible to climate change induced loss of potential range.

2.2.2 Grasslands & Savannas

According to the IPCC AR4, both tropical and temperate grasslands are sensitive to variability and changes in climate, which are likely to have strong effects on the balance between different life forms and functional types in these systems. The mixture of functional types (C3 and C4 photosynthetic systems) and their differential responses to climate variables and CO2 fertilisation mean that non-linear and rapid changes in ecosystem structure and carbon stocks are both likely and difficult to predict with any certainty. Rising temperatures are likely to increase the importance of C4 grasses, but CO2 fertilisation may promote C3 species and the expansion of trees into grasslands. The major climatic effect on the composition and function of grassland and savanna systems is likely to be through precipitation changes and associated changes in fire and disturbance regimes. Modelling has shown major reductions in rainfall as a result of large scale changes in savanna vegetation cover, suggesting positive feedbacks between human disturbance and climate change. The role of temperate grasslands in carbon storage is strongly dependent on rainfall. There are few studies on fauna. The proportion of threatened mammal species may increase by 10-40%; changing migration routes are a threat. Large reductions in species range size have been projected

2.2.2.1 Distribution

Recently, there have been further model-based predictions of northward shifts of savanna ecosystems into the Amazon in response to declining rainfall (Cook & Vizzy 2008; Salazar et al. 2007), and the resulting increased incidence of fire in remaining forests is expected to increase ‘savannization’ still further (Barlow & Peres 2008; Huntingford et al. 2008; Malhi et al. 2008). However in southern Africa there has been a suggestion that savanna areas may be ‘squeezed’ by expanding shrublands (Biggs et al. 2008).

2.2.2.2 Composition

Recently published observational studies suggest that changes in composition of grassland ecosystems are likely in response to climate change. Rainfall is critical in determining the community dynamics of temperate grasslands; a low abundance species became one of the dominant species in the grassland following a period of prolonged below-average rainfall and the abundance of an alien invasive grass was greatly increased following major El Nino events (Hobbs et al. 2007a). Mesic grassland ecosystems in the Pyrenees showed strong shifts in plant diversity and composition after a short period of warming and drought, as a consequence of acute vulnerability of some dominant grasses, losses of rare species, and aggregate and trigger effects of originally uncommon forb species (Sebastia et al. 2008). There are still few data on faunal changes in relation to climate in grasslands, but strong relationships between abundance and rainfall suggest that rainfall underpins the dynamics of African savanna ungulates, and that changes in rainfall due to global warming may markedly alter the abundance and diversity of these mammals (Ogutu et al. 2008b). Rising CO2 concentrations are tending to reduce the importance of C4 grasses in some temperate grasslands (Soussana & Luscher 2007). Experimental doubling of CO2 concentration over Colorado shortgrass steppe had little impact on plant species diversity, but it vastly increased above ground biomass and the abundance of a common shrub (Morgan et al. 2007). This and other experimental work (e.g. Bloor et al. 2008) provide evidence that rising atmospheric CO2 concentration may be contributing to the shrubland expansions of the past 200 years and to invasions of grassland by woody plants.

2.2.2.3 Function

Water availability also affects grassland CO₂ dynamics, but its influence depends on microsite characteristics, e.g. in Yellowstone (Risch & Frank 2007). . An understanding of this dependence needs to be incorporated into predictions of how changes in precipitation/soil moisture will affect CO₂ dynamics and how they may feed back to the global C cycle. Soil moisture distribution emerges as a key link between hydrologic and ecologic processes in semiarid grassland and shrubland in New Mexico, through its influence on evapotranspiration, respiration, and assimilation (Kurc & Small 2007). Analysis of large numbers of data points for observed mean annual NPP, land cover class, precipitation, and temperature. (Del Grosso et al. 2008) showed that precipitation was better correlated with NPP than temperature, and it explained much more of the variability in mean annual NPP for grass- or shrub-dominated systems than for tree-dominated systems. In the savannas of the Mara-Serengeti in East Africa, rising temperatures and declining rainfall throughout the 1990s and early 2000s combined with prolonged and strong ENSO episodes to cause progressive habitat desiccation and reduction in vegetation production in the ecosystem (Ogutu et al. 2008a). This exacerbated the debilitating effects of adverse weather on local plant and animal communities, resulting in high mortalities of ungulates. Experimental studies have also confirmed the importance of water regime in grassland ecosystem carbon dynamics. In subtropical savannas soil respiration is water-limited, and its sensitivity to soil moisture availability increased with increasing woody plant abundance (McCulley et al. 2007). However warming has also been shown to decrease above and below ground biomass production (Boeck et al. 2007; De Boeck et al. 2008) and decreases were seen in both species poor and more complex systems. These results suggest that warming and the associated soil drying could reduce primary production in many temperate grasslands, and that this will not necessarily be mitigated by efforts to maintain or increase species richness.

2.2.2.4 Phenology

There is experimental evidence that warming may lead to earlier grass flowering in temperate grasslands, but rainfall is likely to have an important influence on such responses (Cleland et al. 2006). Elevated CO₂ concentration may have the opposite effect on grass flowering but accelerate flowering in forbs (Cleland et al. 2006). The difference in phenological response between different functional groups may potentially increase competition within grassland ecosystems.

2.2.2.5 Ecosystem services

Changes to the composition, structure and function of grasslands have major implications for their effectiveness in storing carbon and in maintaining supplies of fodder and other resources important to local livelihoods. Invasion by woody plants will also alter their hydrological function, reducing water yield in many cases. Declines in populations and diversity of savanna mammals may have significant implications for potential revenues from nature-based tourism.

2.2.3 **Mediterranean systems**

According to the IPCC AR4, Mediterranean-type ecosystems are vulnerable to desertification and the expansion of adjacent arid and semi-arid systems expected under minor warming and drying scenarios. They may suffer some of the strongest impacts from global climate change, and these will be compounded by the effects of other pressures including land use, fire and fragmentation. The effects of increased CO₂ concentration are inconsistent and are tempered by the growth limitations imposed by increased drought. Desertification and expansion of arid ecosystems are likely to induce substantial range shifts at rates greater than migration capability for many endemic species. Loss of biodiversity is likely overall, including substantial changes to species richness as well as the extinction of some species.

There has been relatively little new work on Mediterranean-type ecosystems since the AR4. Evaluation of likely climate changes on mountains in the Mediterranean suggests that they will be subject to similar temperature rises and much greater decrease in spring rainfall compared to other mountains in Europe (Bravo et al. 2008) and that they may decline dramatically (Metzger et al. 2008).

2.2.3.1 Composition

Bioclimatic modeling for European mammals (Levinsky et al. 2007) has shown that potential mammalian species richness is predicted to reduce dramatically in the Mediterranean region. Ecophysiological studies of an endangered endemic tree in the Fynbos (February *et al.* 2007) confirm its vulnerability to prolonged drought and the marked effect its loss would have on vegetation composition and structure. Experimental studies of grassland turf from Mediterranean mountains show that its composition is sensitive to climate and becomes much more like lowland grassland after only short periods of warming.

2.2.3.2 Function

Modelling has suggested that Mediterranean systems will have the smallest increases, and in some cases decreases, in net primary production (NPP) in Europe (Morales et al. 2007), and that these systems will switch from sinks to sources of carbon by 2100, mainly as a result of deteriorating water balance. Watershed modelling shows that (Nunes et al. 2008) water runoff, particularly subsurface runoff, is highly sensitive to the increased temperature and reduced rainfall predicted for the region, showing as much as an 80% reduction and that the more humid ecosystems will become increasingly arid with attendant loss in productivity.

2.2.3.3 Phenology

Advancement of spring and summer events since the 1970s has been shown for a sample of plants, insects and insectivorous birds (Gordo & Sanz 2005). Insect phenology changed more than plant phenology, suggesting there may be decoupling of some plant-insect interactions, such as those between pollinators and flowers or herbivorous insects and their plant resources.

2.2.3.4 Ecosystem services

Climate-related changes to the composition, structure and function of Mediterranean-type ecosystems are likely to reduce their ability to house the high levels of biodiversity that are important in supporting nature-based tourism and various extractive activities. Their hydrological function is likely to be altered by increasing aridity.

2.2.4 Forests & Woodlands

According to IPCC AR4, modelling approaches predict that major changes in global forest cover are nearly certain at temperature rises over 3°C. Mostly they predict significant loss of forest towards the end of the century, particularly in boreal, mountain and tropical regions, but some climate-limited forests are expected to expand, particularly where water is not limited. Recent moderate climate changes have been linked to improved forest productivity, but these gains are expected to be offset by the effects of increasing drought, fire and insect outbreaks as a result of further warming. Estimates of the ability of tree species to migrate are uncertain, but northward shifts appear likely. Losses of species diversity have been projected, particularly in tropical forest diversity hotspots. Mountain forests appear particularly vulnerable. Extinctions of amphibian species in montane forests have already been attributed to climate change, and in most cases extinction risks are projected to increase.

2.2.4.1 Distribution

There are still relatively few observations of changes in forest ecosystem distribution because of the long time frames required for such shifts. However, long term monitoring in Scandinavia has shown uphill migration of treelines on the order of 150 – 200 m (Kullman 2007). Similarly, a long-term photographic record has been used to document the upward advance of alpine treeline through encroachment of woody vegetation into alpine meadows in Yunan, China (Baker & Moseley 2007). In both cases these changes have been linked to observed warming.

Many recent modelling studies reinforce the conclusions summarised in AR4 that the location of favourable conditions for some forest ecosystems is likely to shift appreciably and may do so faster than any possible migration rates. Overall, such studies have confirmed the likely northward expansion of boreal forests and the vulnerability of some forest zones on mountains and islands. In North America, modelling of tree species distributions in relation to climate models suggests a general movement of habitats towards the northeast (by as much as 800 km for the highest temperature changes) giving a northwards retreat of the spruce-fir zone and an advance of the southern oaks and pines but does not explicitly take into account the species' potential migration rates (Iverson *et al.* 2008). More generally, temperate and boreal forests are expected to expand northwards and upwards at the expense of tundra and alpine communities (Alo & Wang 2008; Wolf *et al.* 2008a). However, as noted in the AR4 there are likely to be major time lags involved in the northward expansion of boreal forest and further evidence of these lags comes from comparison of analysis treelines in relation to climatic conditions (MacDonald *et al.* 2008). In areas like British Columbia, where the potential for migration is limited by other factors, some of the most important conifer species are expected to lose a large portion of their suitable habitat and currently important sub-boreal and montane climate regions will rapidly disappear (Hamann & Wang 2006). In Europe the zones of forest-suitable climate are expected to shift towards the northeast (Casalegno *et al.* 2007), and in Southern Africa, a poleward expansion of the forest biome is also projected (Biggs *et al.* 2008).

Work since the AR4 has not fully resolved the controversy over predictions of Amazon rainforest dieback. As well as increasing temperatures, most models tend to demonstrate a reduction in dry season rainfall, particularly in eastern Amazonia (Li *et al.* 2008; Malhi *et al.* 2008), and the roles of sea surface temperatures in both the Atlantic and the Pacific in affecting rainfall in different seasons is now much better understood (Good *et al.* 2008; Harris *et al.* 2008). These changes are widely considered likely to lead to widespread dieback of the forest; projections for climate-related forest loss in Amazonia range from 18% (Salazar *et al.* 2007) to 70% (Cook & Vizzy 2008), and feedbacks from drying forest, increased fire incidence and fragmentation are expected to exacerbate these effects (Barlow & Peres 2008; Huntingford *et al.* 2008; Malhi *et al.* 2008). Nonetheless, some authors argue that the evidence is still far from conclusive: paleoecological investigations show little evidence of a change from forest to savanna during mid-Holocene dry phases, except at the margins of the forest zone (Mayle & Power 2008) and ecophysiologicalists suggest that Amazon forest trees are capable of coping with increased temperatures (Lloyd & Farquhar 2008).

2.2.4.2 Composition

Although it was not emphasised in the IPCC AR4, there is also substantial evidence based mostly on modelling that species composition of both temperate and tropical forests will also change as a result of climate change. This will result from differences in species tolerances to new conditions and in the rates at which they are able to move. For example, models showed that of 134 temperate tree species in eastern North America 66 would gain habitat under climate change and 54 would lose habitat, resulting in new patterns of species composition and increasing importance of southern oak and pine species (Iverson *et al.* 2008). The importance of deciduous species is expected to increase further north and at higher altitude in the Barents region (Wolf *et al.* 2008a) and in Scandinavia (Kellomäki *et al.* 2008), and regionally variable rates of treeline expansion in boreal zones like Siberia may lead to the development of transient forest communities with species abundances that differ from current patterns (MacDonald *et al.* 2008). The wide variation in the magnitude of optimum elevation shifts among forest plant species in Europe (Lenoir *et al.* 2008) is likely to lead to compositional change within forest assemblages and may result in the disruption of biotic interactions.

In tropical forests, compositional changes are expected to arise from the selective advantages of different species in dealing with the physiological demands of climate change (Malhi *et al.* 2008). Compositional changes are likely to be exacerbated by the impacts of forest fires (Adeney *et al.* 2006; Barlow & Peres 2008) and to include increasing abundance of lianas (Phillips *et al.* 2008).

2.2.4.3 Structure

Changes in tropical forest structure are associated with changes in composition (increased liana frequency) and increasing forest turnover and gap formation (Phillips et al. 2008). A similar pattern has recently been documented for floodplain forests in North America (Allen et al. 2007). There is some evidence that increased liana frequency may be a direct response to drying climate (Swaine & Grace 2007), but also considerable evidence that it is more immediately caused by canopy disturbance and increasing fragmentation of forest (Londre & Schnitzer 2006; van der Heijden & Phillips 2008).

2.2.4.4 Function

The evidence about the effects of climate change on such aspects of forest ecosystem function as CO₂ uptake, growth, and net primary production remains contradictory because of the conflicting effects of warming and CO₂ fertilisation (largely positive (Luyssaert et al. 2007) and drying (negative). In temperate and boreal forests, increased warming and increased CO₂ concentrations are both likely to contribute to increased growth in deciduous species, but warming may reduce productivity in conifers (Ollinger, V et al. 2008). Increased respiration rates and changes in species composition seem likely to reduce carbon accumulation by temperate and boreal forests (Kellomäki et al. 2008; Kurz et al. 2008b; Mohan et al. 2007). The effects of elevated CO₂ measured in experimental settings and implemented in models may overestimate actual field responses, because of many limiting factors such as pests, weeds, competition for resources, soil water, air quality, etc (Kirilenko & Sedjo 2007). The increased incidence of fire is also likely to have a major influence on the ability of these forest ecosystems to store carbon (Nitschke & Innes 2008).

In Amazonian tropical forests, there is observational evidence of increasing productivity and related increased turnover, probably as a response to CO₂ fertilisation (Phillips et al. 2008), but it is widely anticipated that this is a temporary response and that the warming, drying climate will reduce the mean net primary productivity, NPP, across Amazonia by approximately 30-50% by 2050 under a medium-high greenhouse gas emissions scenario (Harris et al. 2008). Climate change is likely to increase fire frequencies in these forests and new work shows how this is likely to reduce their productivity and ability to store carbon (Barlow & Peres 2008; Bush et al. 2008).

2.2.4.5 Phenology

As noted in the IPCC AR4, there is ample evidence of long-term changes in patterns of tropical forest tree and liana flowering or fruiting (Wright & Calderon 2006) that may be linked to changes in climatic variables.

2.2.4.6 Ecosystem services

The IPCC AR4 recognised the important role that forests play in delivering a wide range of ecosystem services, including the provision of timber, fuel and other non-timber forest products, carbon sequestration, regulation of hydrological processes and flows and retention of biodiversity. Both work summarised in the AR4 and more recent research suggests that warming and drying climates in combination with land use change, fire and other pressures are likely ultimately to reduce the capacity for carbon storage in the vital carbon reservoirs of both boreal and tropical forests (Malhi et al. 2008; Nitschke & Innes 2006).

A very important advance since the IPCC AR4 is the recognition that old growth forests continue to store carbon rather than being carbon-neutral (Luyssaert et al. 2008) and that they therefore play a vital role in offsetting carbon emissions.

Climate induced changes in boreal forest have the potential to affect their hydrological function and ultimately freshwater inputs to the Arctic Ocean and the formation of sea ice, but it is likely that

anthropogenic changes in this respect will be far greater climate induced changes to forest hydrology (Woo et al. 2008)

Work on modelling changes to production forest in Scandinavia showed significant changes to tree growth and species composition that would require adaptive changes in forest management (Kellomäki et al. 2008)

2.2.5 Tundra & Arctic

According to IPCC AR4, tundra and polar (Arctic and Antarctic) ecosystems are likely to be the most vulnerable to climate change, and may be turned from a net carbon sink to a carbon source, with significant feedbacks to climate through both carbon emission and changes to albedo. Tundra climates will shift rapidly polewards and vegetation change is likely to follow but with significant lags on tundra movement into polar desert and taiga encroachment on tundra due to slow growth and dispersal rates. Experiments have shown that changes in temperature alter species dominance and therefore species composition. Food availability may increase for some vertebrates in summer but decrease in winter. Endemic species such as the polar bear and arctic breeding birds likely to experience large population declines and elevated extinction risks.

2.2.5.1 Distribution

Recent modelling studies have confirmed that likely encroachment from taiga vegetation into tundra and suggest that it may not be balanced by northward movement of tundra vegetation in the Barents region, leading to a net reduction in area of tundra shrub ecosystems (Wolf et al. 2008a).

2.2.5.2 Composition

Recent work has emphasised the importance of sea ice addition to the direct effects of climate itself in determining the survival and abundance of arctic species (Moore & Huntington 2008) . Polar bear, seal and penguin populations are all affected by the abundance and dynamics of sea ice, which is being severely altered by climate change (Barber-Meyer et al. 2008; Schliebe et al. 2008; Wiig et al. 2008).

2.2.5.3 Ecosystem services

Recent work has suggested that carbon storage in arctic soils is much higher than previously thought (Beer 2008; Ping et al. 2008) and that climate warming in the region is therefore likely to release significant amounts of carbon that will affect the climate system.

2.2.6 Mountains

According to the IPCC AR4, Mountain regions have already experienced above average warming, and its impacts, including water shortages and reduced extent of glaciers, are likely to be exacerbated by other pressures causing ecosystem degradation, such as land use change, over-grazing and pollution. This has the impact of reduced glacier extent and water shortages. There is a disproportionately high risk of extinction for endemic mountain biota, partly because of their restricted geographic ranges and possibilities for migration, which can result in genetic isolation and stochastic extinctions. A reshuffling of species along altitudinal gradients is to be expected from their differential capacities to respond to change. Warming is expected to produce drying due to higher evapotranspiration in many mountain systems, and this will in itself reduce the feasibility of upward movement of treelines. Tropical montane cloud forests and their biota are particularly vulnerable to drying trends. Warming is already driving mass extinctions of highland amphibians, and many other species of mountain ecosystems are potentially subject to sharp declines.

2.2.6.1 Distribution

Recently several new observational studies have confirmed glacial retreat on mountains around the world, including in China (Baker & Moseley 2007), the Alps (Cannone *et al.* 2008), Colombia (Ruiz *et al.* 2008). There have also been observations of rapid colonization of the resulting deglaciated surfaces (Cannone *et al.* 2008) with more plastic species apparently playing a more important role in colonization. Treelines have been observed moving up in altitude in both China (Baker & Moseley 2007) and Scandinavia (Kullman 2007;Pauli 2007). Alpine species distributions have moved upwards over the last 100 years (Walther *et al.* 2005) and the rate at which they are doing so appears to be increasing. Further evidence of distributional shifts is provided by documented upwards changes in the distribution of plant species by 65 m in 30 years in mountains in southern California (Kelly & Goulden 2008) and by a new approach to analyzing shifts in forest plant species' optimal elevations in European mountains, which are shown to have moved upwards by 29 m per decade (Lenoir *et al.* 2008).

Modelling work also suggests that the distribution of mountain ecosystems may change appreciably. A regional climate modeling study in Costa Rica suggests that the future climate distribution for tropical montane cloud forests lies well outside their present-day distribution (Karmalkar *et al.* 2008).

2.2.6.2 Composition

There is a growing body of observational data on species declines in mountain ecosystems, principally in alpine systems (Pauli 2007), which combined with observations of distributional changes suggests that the composition of mountain ecosystems will change appreciably in response to climate change. High resolution modelling studies predict substantial species turnover in arctic alpine communities, even under scenarios of low climate change (Trivedi *et al.* 2008b). There is some suggestion that large scale, coarse resolution modelling studies may have overestimated montane species' ability to cope with increasing temperatures, thereby underestimating the potential impacts of climate change.(Trivedi *et al.* 2008a). Experimental work has suggested that climate warming may increase the importance of interspecific competition in governing species composition and alpine community structure (Klanderud & Totland 2007)

2.2.6.3 Ecosystem services

Some montane systems, particularly tropical montane cloud forest, high altitude bogs and some grasslands, such as those on the Tibetan Plateau (Wang *et al.* 2008) contain large amounts of carbon in their soils, which are vulnerable to release by climatic warming. Montane systems are also critical for regulating hydrological flows, and changing insolation and cloud cover and degradation of their structure will impede their ability to deliver these services (Ruiz *et al.* 2008).

2.2.7 **Freshwater systems**

According to the IPCC AR4, inland aquatic ecosystems are highly vulnerable to climate change, especially in Africa. Higher temperatures will cause water quality to deteriorate and will have negative impacts on micro-organisms and benthic invertebrates. Plankton communities and their associated food webs are likely to change in composition. Distributions of fish and other aquatic organisms are likely to shift pole-wards and some extinctions are likely. Changes in hydrology and abiotic processes induced by changes in precipitation as well as other anthropogenic pressures will have large impacts on aquatic ecosystems. Boreal peatlands will be affected most and suffer major changes in species composition. Many lakes will dry out. Increases in the variability of precipitation regimes will also have important impacts and may cause biodiversity loss in some wetlands. Seasonal migration patterns of wetland species will be disrupted. The impacts of increased CO2 will differ among wetland types, but may increase NPP in some systems and stimulate methane production in others. On the whole, ecosystem goods and services from aquatic systems are expected to deteriorate.

2.2.7.1 Distribution

There is new observational evidence of climatic impacts on inland wetlands. Recent research has shown that there has been a fourfold increase in permanently dry ponds in Yellowstone over the last 16 years and that this can be linked directly to dramatic declines in amphibian populations and diversity (McMenamin et al. 2008). Modelling work suggests that climatic warming in combination with other environmental changes may cause the nature of river channels to change in the Russian Arctic (Anisimov 2008). In the arctic, factors such as reduced ice-cover duration lakes especially in northern Arctic areas, increased and more rapid stratification, earlier and increased primary production, and decreased oxygenation at depth will possibly result in a reduction in the quality and quantity of habitat for species such as lake trout, and decreased water flow in summer is likely to decrease habitat availability and possibly deny or shift access for migrating fish (Anisimov et al. 2007; Berry 2008; Reist et al. 2006; Wrona et al. 2006a; Wrona et al. 2006b). In monsoonal Asia, where ecological processes surrounding rivers are mediated by flow, disruptions in timing and velocity will have large environmental impacts (Dudgeon 2007).

2.2.7.2 Composition

There is new observational evidence of compositional change in fish communities in France over the last 15-25 years (Daufresne & Boet 2007); species richness, proportions of warm water species and total abundance increased. Modelling has demonstrated negative impacts on the habitats of native fish species, including freshwater salmon (Battin et al. 2007), and especially at higher elevations and in headwater areas (Buisson 2008). In the Arctic, there is an expected decrease or local loss of native fish as southern Arctic and sub Arctic fish species migrate northwards. The broad whitefish, Arctic char complex, and the Arctic cisco are particularly vulnerable to displacement. Decreased water flow in summer is likely to decrease habitat availability and possibly deny or shift access for migrating fish (Anisimov et al. 2007; Berry 2008; Reist et al. 2006; Wrona et al. 2006a; Wrona et al. 2006b). Models show that climate change will also affect wetland species composition through its effects on river flow, especially low water flows (Harrison et al. 2008), though the interaction with socio-economic drivers of flow management is also very important.

There is considerable and growing concern about the linkages between climate change impacts on aquatic systems (including warmer water temperatures, shorter duration of ice cover, altered streamflow patterns, increased salinization, and increased demand for water storage and conveyance structures) and aquatic invasive species (Rahel et al. 2008; Rahel & Olden 2008). Climate change will influence invasive establishment by eliminating adverse winter conditions and will alter the distribution and ecological impacts of existing invasive species by enhancing their competitive and predatory effects on native species and by increasing the virulence of some diseases (Hellmann et al. 2008; Rahel et al. 2008; Rahel & Olden 2008). In Canada, predictions of water temperature changes of as much as 18 degrees C by 2100 mean that a number of lakes will be newly vulnerable to invasion by smallmouth bass (Sharma *et al.* 2007).

There is very little information on real or projected changes in aquatic ecosystems in the tropics, but it is clear that some major tropical wetlands are at risk from altered flows of freshwater (Gopal & Chauhan 2006).

2.2.7.3 Ecosystem services

Significant impacts of climate change have been projected for both carbon storage and fisheries service from inland waters. While this has been projected particularly strongly for the arctic (Wrona et al. 2006a; Wrona et al. 2006b), there is also a growing body of model-based evidence relating to other regions. A new concern is the effect of sea level rise on carbon storage in coastal wetlands, including 150,000 km² of freshwater peatlands worldwide below 5-m elevation and vulnerable to sea level rise, which are likely to emit significant amounts of carbon when they are inundated (Henman & Poulter

2008). The protection and biodiversity conservation roles of coastal wetlands are also at risk as in, for example, the case of the Sundarbans, the world's largest wetland, which is threatened by altered freshwater flows and sea level rise, which are both influenced by climate change (Gopal & Chauhan 2006).

2.2.8 Marine and coastal

According to the IPCC AR4, the most vulnerable marine ecosystems include warm water coral reefs, cold water corals, the Southern Ocean and sea-ice ecosystems. Ocean uptake of CO₂ reduces the PH of surface waters and their concentrations of carbonate ion and aragonite, which are vital to the formation of the shells and skeletons of many marine organisms including corals. Other impacts of climate change on marine ecosystems are through warming, increasing thermal stratification and reduced upwelling, which can alter nutrient fluxes and induce hypoxia, sea level rise, increase in wave height and storm surges and loss of sea ice. The productive sea ice biome is projected to contract substantially by 2050, with significant impacts on fish and krill populations and on their predators. Changes in planktonic, benthic and pelagic community composition have been observed and associated with climate change. Marine mammals, birds, cetaceans and pinnipeds are vulnerable to climate-related changes in prey populations. Melting ice sheets will reduce salinity, disrupt food webs and cause poleward shifts in community distributions. Both coral reefs and warm water corals will be suffer serious adverse effects from ocean acidification

2.2.8.1 Ocean Acidification

Since the AR4, concern about ocean acidification and its implications for biodiversity and ecosystem services has increased markedly. The oceans have absorbed around one-third of the total carbon dioxide (CO₂) released into the atmosphere by human activities over the last 200 years. As a result the ocean is the second largest sink for anthropogenic carbon dioxide after the atmosphere itself (Iglesias-Rodriguez et al. 2008). If emission levels were to continue at the same level to the year 2100, CO₂ concentration will rise by a factor of two relative to the present value and seawater pH will drop by a further 0.3pH units (Riebesell et al. 2008).

Observational and experimental studies have shown that ocean acidification reduces the calcification rates of various calcifying species such as halimeda, benthic mollusks, forminifera and coccolithophores (Fabry 2008;Guinotte & Fabry 2008;Riebesell et al. 2007;Riebesell 2008;Zeebe et al. 2008). Reduced calcification in shellfish such as oysters and mussels would impact worldwide commercial aquaculture production (Gazeau et al. 2007).

Ocean acidification can also result in ecosystem impacts, for example the long term impacts of permanent exposure to high CO₂ concentrations has been shown to result in substantial shifts in benthic community composition (Hall-Spencer et al. 2008). Coral reefs are negatively impacted by acidification and if future increases in seawater acidity affect calcification, then reefs could loose structural stability, which would indirectly have negative implications for reef communities and shore protection (Hoegh-Guldberg *et al.* 2007). Models show that continuing emissions could lead to potentially catastrophic levels of acidification and rescued calcification (Cao 2008)

However, the impacts of ocean acidification on marine species and habitats are not always negative. Although calcareous groups generally decline in abundance, photosynthetic groups such as brown algae and seagrasses utilise higher CO₂ availability to increase their biomass (Guinotte & Fabry 2008).

2.2.8.2 Sea surface temperature

Since the AR4, further evidence has accumulated of the negative impacts on species of rising sea surface temperatures. The adult survival of the king penguin (*Aptenodytes patagonicus*) decreases with

increasing temperature, 9% decline per 0.26°C of warming (Le Bohec *et al.* 2008). An inverse correlation has also been observed between mean annual sea surface temperature and reproductive frequency of the endangered loggerhead turtle (Chaloupka *et al.* 2008), critically endangered leatherback turtle (Saba *et al.* 2007) and Antarctic fur seals (Forcada *et al.* 2005).

The impacts of increasing water temperatures on individual species can have dramatic effects at the ecosystem level. For example many zooxanthellate reef-building coral species are threatened with extinction, and declines in their abundance are associated with bleaching and diseases driven by elevated sea surface (Carpenter *et al.* 2008; Lough 2008). This has major implications for the large biological communities that coral reefs support; for example, coral reef fishes will be negatively affected through effects on individual performance, trophic linkages, recruitment dynamics, population connectivity and other ecosystem processes (Graham *et al.* 2007; Hughes *et al.* 2007; Munday *et al.* 2008; Pratchett *et al.* 2008). The vital ecosystem services these systems provide through fisheries, coastal protection, building materials, new biochemical compounds, and tourism will also be affected (Hoegh-Guldberg *et al.* 2007). Eelgrass meadows and associated ecosystem services will also be negatively affected by predicted increases in summer temperature extremes (Ehlers *et al.* 2008).

Increase in sea surface temperature has also been observed to have some positive species level impacts. Sea surface temperature is positively correlated with the breeding success of endangered black-browed albatross (Rolland *et al.* 2008). The planktonic larvae of echinoderms and decapod crustaceans have increased in abundance in the North Sea, especially since the mid-1980s, as sea surface temperature increased (Kirby *et al.* 2008). However, such species level increase may be the basis for less positive major trophic restructuring of ecosystems.

Further evidence has also emerged for the role of increasing sea water temperatures in shifting in distributions of marine species. Northward movements have been observed for many species including warmer-water plankton in the north-east Atlantic, cetacean species in the western Ligurian Sea (Azzellino *et al.* 2008) and many fish species, including the silver john dory which has been estimated to have a northward movement of 50km/year (EEA *et al.* 2008). Temperature-related distributional shifts lower down the food chain have also facilitated poleward shifts among consumers, including the critically endangered leatherback turtle (McMahon & Hays 2006) and the Balearic shearwater (Wynn *et al.* 2007). Tropical fish species have also expanded polewards into warm temperate waters off South Africa (James *et al.* 2008). Increasing sea water temperatures will also change the vertical distribution of some species, for example the deepening of North Sea bottom dwelling fish by 3.6m per decade has been observed (Dulvy *et al.* 2008).

Changes and shifts in the distribution of commercial fish species could have serious implications for fisheries. Increased fisheries production is expected in high latitude regions (Brander 2007) and the GDP of Iceland is predicted to rise as a result (Arnason 2007). In other regions such as the North Sea, boreal fish production will decrease as warm water species become more abundant (Stenevik & Sundby 2007). Changes in the distribution and range changes of species resulting from climate change will impact fisheries management and in some cases quotas may need to be revised.

There is rising concern that oceanic warming may enhance the impact of invasive species and it is predicted that the Arctic Ocean will be subject to increased invasion from temperate species (Vermeij & Roopnarine 2008). Experimental studies suggest that the balance between native and introduced species of Eel grass along the California coast is likely to shift in response to rising sea temperatures (Shafer *et al.* 2008).

A further impact of rising sea surface temperatures is in altering the seasonal dynamics of upwelling zones and associated phytoplankton. There is some observational evidence suggesting that this is occurring off California and in the Gulf of Guinea (Barth *et al.* 2007; Wiafe *et al.* 2008).

Increased sea water temperature may be associated extreme weather events. In the Atlantic sea surface temperature is associated with around a 40% increase in hurricane frequency and activity (Saunders & Lea 2008). Hurricanes can negatively impact coastal ecosystems such as coral reefs and mangroves, this can have both economic and social implications as both provide important ecosystem services.

2.2.8.3 Sea Ice

Since AR4 there has been growing recognition of the importance for biodiversity of changes in the timing and extent of sea ice, which impose temporal asynchronies and spatial separations between energy requirements and food availability for many species at higher trophic levels (Moline *et al.* 2008). These mismatches lead to decreased reproductive success, lower abundances, and changes in distribution. In the Arctic the hooded seal, polar bear and narwhal are judged to be the three marine mammal species most sensitive to sea ice loss (Laidre *et al.* 2008; Laidre & Heide-Jorgensen 2005; Simmonds & Isaac 2007). For polar bears sea ice loss will result in large future reductions in subpopulations (Laidre *et al.* 2008), increased time on land (Schliebe *et al.* 2008) and reduced gene flow (Crompton *et al.* 2008) between subpopulations. In the Antarctic, alterations in winter sea ice dynamics are the changes most likely to have had a direct impact on the marine fauna (Clarke *et al.* 2007). Climate-mediated changes in ice dynamics affect krill, which is central to the Antarctic food web. Reduction in krill abundance will directly impact cetacean species (Nicol *et al.* 2008) and affect food chains all the way to top predators. Observational studies show that breeding success of macaroni penguins (Cresswell *et al.* 2008) and emperor penguins (Barber-Meyer *et al.* 2008) is correlated with sea ice extent, but that populations are still relatively stable.

2.2.8.4 Sea Level Rise and Coastal Ecosystems

As noted in AR4, sea level rise will have major impacts on some coastal ecosystems, although others are less susceptible because of their naturally dynamic nature and ability to accrete new sediments. Coastal marshes are susceptible to accelerated sea level rise because their vertical accretion rates are limited and they may drown. As marshes convert to open water, tidal exchange through inlets increases, which leads to sand sequestration in tidal deltas and erosion of adjacent barrier shorelines. The character of coastal wetlands in Estonia has been more unstable in the face of documented climatic changes than that of inland bogs (Kont *et al.* 2007). The most marked coastal changes in Estonia resulted from a combination of strong storms, high sea levels induced by storm surge, ice free seas and unfrozen sediments. A recent review of mangrove threats has suggested that climate change may lead to a global loss of 10-15% of mangrove forest (Alongi 2008), and sea level rise is one important component of that threat because sediment accretion is not keeping pace with it. This is especially problematic where there are limited options for landward migration, as in the Pacific islands (Gilman *et al.* 2006; Gilman *et al.* 2008) and parts of the Indian subcontinent (Jagtap & Nagle 2007). However, some mangrove systems appear to be stable in the face of sea level rise (Sanders *et al.* 2008).

2.2.8.5 Coral reefs

As noted in the AR4 coral reefs are especially subject to adverse impacts from climate change due to bleaching and diseases promoted by warmer temperatures and increasing pressures from acidification. There has been a recent and alarming rise in mass bleaching events (Graham *et al.* 2007; Lough 2008). Carbonate accretion is being affected by warming and ocean acidification to such an extent that coral will become rare on reef systems (Hoegh-Guldberg *et al.* 2007). These pressures are made more severe by interaction with other pressures from development and over-exploitation (Jackson 2008) but are not controlled by marine protected areas, so climate mediated disturbances need to be given high importance in conservation planning for coral reefs (Graham *et al.* 2008). Coral reef fish are negatively affected by coral bleaching, but will also suffer direct effects of climate change on individual performance, trophic linkages, recruitment dynamics, and population connectivity (Munday *et al.* 2008; Pratchett *et al.* 2008).

3 Species

Climatic change has already caused changes to the distribution of many plants and animals, leading to severe range contractions and the extinction of some species. The AR4 states, with very high confidence, that observational evidence from all continents and most oceans shows that species are being affected by regional climate changes, particularly temperature increases (Rosenzweig et al. 2008). Changes of terrestrial species include shifts in spring events (for example, leaf unfolding, flowering date, migration and time of reproduction), species distributions and community structure (Rosenzweig et al. 2008). In marine-ecosystems changes have been demonstrated in functioning and productivity, including shifts from cold-adapted to warm-adapted communities, phenological changes and alterations in species interactions. Most of these changes are in the direction expected with warming temperature (Rosenzweig et al. 2008). Some species are unable to disperse or adapt fast enough to keep up with high rates of climate change and these species face increased extinction risk (Menendez et al. 2006), and, as a result, whole ecosystems, such as cloud forests and coral reefs, may cease to function in their current form (Hoegh-Guldberg et al. 2008). Here we review recent observed and modelled climate change impacts on species, including changes in species distributions and population changes.

3.1 Changes in distributions

Climatic conditions, such as temperature and precipitation, determine suitable habitat for certain species. Rapid changes in climatic conditions are therefore likely to change the geographic extent of species distributions, resulting in latitudinal and/or altitudinal shifts and/or contractions of species' ranges. Documenting incipient range shifts requires intensive survey and resurvey at high spatial resolution.

3.1.1 Poleward shifts

Meta-analyses of observed impacts on species found that there have been significant range shifts towards the poles in the past (Parmesan & Yohe 2003; Root et al. 2003). Recent observational evidence for more species, including plants (Colwell et al. 2008), invertebrates (Franco et al. 2006; Hickling et al. 2006), and vertebrates (Gaston et al. 2005; Hickling et al. 2006; Hitch & Leberg 2007; Schliebe et al. 2008; Sorte & Thompson 2007) strength these findings of substantial latitudinal shifts of range boundaries, centres of occurrence and abundance.

Poleward shifts under future climate conditions have been projected for plants, insects, birds and mammals in North America and Europe (Fuller et al. 2008; Harrison et al. 2006; Huntley et al. 2006; Huntley et al. 2008a; Huntley et al. 2008b; Morin et al. 2008; Virkkala et al. 2008). Specialized and species with restricted ranges show little or no overlap between their present and potential future ranges, (Huntley et al. 2008b). The loss of climatic space and reduction in suitable habitat can lead to extinctions (Morin et al. 2008; Virkkala et al. 2008). Climate change will very likely affect the biodiversity of freshwater ecosystems across most of the Arctic including changes in habitat suitability and timing of availability (Wrona et al. 2006a). Projected shifts for Arctic fish populations will range from positive to negative in overall effect, differ among species and also among populations within species depending upon their biology and tolerances (Reist et al. 2006). Temperate marine species are projected to invade into a warmer Arctic Ocean (Vermeiji & Roopnarine 2008).

Shifts in distributions have mostly been studied in temperate zones, as changes are difficult to detect in the tropics because of minimal temperature gradients with distance (Colwell et al. 2008). Observed changes closely follow the predicted or modelled changes for European bird communities (Lemoine et al. 2007).

3.1.2 Altitudinal shifts

New observational evidence backs up findings that species tend to move upwards to higher elevations with increasing temperatures. Up-slope shifts have been observed for plants in Europe (Kullman 2007; Lenoir et al. 2008) and North America (Kelly & Goulден 2008). Butterflies ranges have shifted upwards by over 200m in 30 years in Spain, consistent with shifts in isotherms (Wilson et al. 2005; Wilson et al. 2007). Temperate mammal and South East Asian bird ranges have shifted upslope (Moritz et al. 2008; Parmesan 2006; Peh 2007). Upslope shifts have been modelled for tropical insects (Colwell et al. 2008) and, as tropical ectotherms already live near their thermal optimum, the impacts are likely to have most deleterious consequences (Deutsch et al. 2008). Observed altitudinal shifts for species on mountains and in grassy habitats were larger than other species (Lenoir et al. 2008).

3.1.3 Range contraction

With species distributions shifting polewards and up-slopes, the ranges of many species are contracting, if current and projected ranges do not overlap and species are unable to migrate. Interactions between climate change and landscape changes will impede range shifts, resulting in range contractions and potential extinctions (Carroll 2007). Range contractions have been observed for butterflies in Britain (Franco et al. 2006) and 5% of all land bird species are projected to suffer more than 50% range reduction by 2050 (Jetz et al. 2007). This is particularly severe for species with limited dispersal abilities, e.g. reptiles and amphibians (Hickling et al. 2006), plants (Huntley 2007), slower life history traits (Lenoir et al. 2008), and range restricted species such as polar and mountain top specialists, e.g. high elevation mammals as land area declines with increasing elevation (Moritz et al. 2008). Some stream fish are projected to be significantly impacted as their dispersal is limited to streams and rivers, whereas other cool- and warm-water fish are likely to colonise newly suitable sites, resulting in dramatic changes in species composition (Buisson et al. 2008).

In addition to direct impacts on species, distribution changes are likely to result in the disruption of biotic interactions and networks when interacting species have responded differently to warming, with important ecological and evolutionary consequences (Lenoir et al. 2008; Parmesan 2006).

3.2 Changes in abundance

The AR4 stated that up to a 30% of higher plant and animal species would be at high risk of extinction with a warming of 'only' 1.5-2.5°C over present temperatures. Many species have suffered population declines that have been attributed to the effects of climate change, acting through a range of mechanisms. However, other species have increased in both abundance and breadth of distribution.

Observational evidence indicates that populations of some birds in Europe have declined due to climate change, whereas others have increased (Gregory et al. 2008). Similarly some butterfly species have expanded their ranges in Germany and Britain (Menendez et al. 2006; Patrick et al. 2007). Tropical and polar species, and habitat specialists or restricted range species are at particular risk from climate change (Laidre et al. 2008; Wake 2007; Wake & Vredenburg 2008).

Several modelling studies show that many species are likely to go extinct, e.g. 1-10% of plant species in Europe will be lost by 2100, depending on climate scenarios and the assumption about migration potential of species (Bakkenes et al. 2006) (van Vuuren et al. 2006), in the absence of migration 10-50% of plants likely to disappear. Greatest extinction risks from global warming may occur in the tropics where biodiversity is also greatest (Deutsch et al. 2008; Tewksbury et al. 2008; Williams & Middleton 2008).

According to Levinsky et al. (2007) up to 9% of European mammals risk extinction, whereas 70-78% may be severely threatened (losing over 30% habitat) under one IPCC scenario, assuming no migration. Where unlimited migration is assumed such figures fall to 1% and up to 46% respectively. Some specialist species, such as the riverine rabbit, might lose most of their current habitat under project

climate change scenarios (Hughes et al. 2008). Endemic species were predicted to be most affected where no migration was assumed, and species richness was dramatically reduced in the Mediterranean region. Jetz et al. (2007) and Sekercioglu et al. (2008) evaluated exposure of all 8,750 land bird species to projected climate and land use change scenarios, both conclude that 400-900 species are projected to suffer dramatic range reductions by the year 2100. Significant impacts at high latitudes are expected, whereas endemic tropical species are most at risk largely due to land conversion. Worldwide, every degree of warming projected a nonlinear increase in bird extinctions of about 100-500 species (Sekercioglu et al. 2008). Only 21% of the species predicted to become extinct are currently considered threatened with extinction (Sekercioglu et al. 2008). More severe impacts are projected for the tropics, e.g. 74% of rainforest birds of north-eastern Australia are predicted to become threatened within the next 100 years (Shoo et al. 2005). Under climate change 54 of 134 species modelled in the U.S. would lose at least 10% of their suitable habitat, whereas 66 of 134 species would gain 10% of their suitable habitat (Iverson et al. 2008). For southern Africa projected modelled declines in the average population sizes of plants and vertebrates over the coming century are two to three times greater than the reductions that have occurred since circa 1700 (Biggs et al. 2008). Restricted range species or those in extreme locations, e.g. mountains, polar regions, are projected to be more at risks from climate change (Berry et al. 2006) (Laidre et al. 2008).

Climate change will severely affect biodiversity by 2100, however in the near future land-use change in may lead to yet greater species loss (van Vuuren et al. 2006). Interactions among species, as well as those between climate change and other pressures that may threaten species, such as habitat loss, need to be included in models (Carroll 2007).

Below we consider multiple mechanisms or causes impacting species in relation to climate change. Although considered separately below, these factors interact and impact on species.

3.2.1 Temperature

Some species are directly impacted by temperature. For example, temperatures exceeding 42°C killed over 3500 individuals of Australian flying-foxes in nine mixed-species colonies (Welbergen et al. 2008). Taiwan trout (*Oncorhynchus masou formosanus*) lives in mountain streams and modelling of climate change impacts indicate a total population decline from 1612 to 146 individuals with 2.7 degrees C temperature rise (Tseng & Chen 2008). The impact of increasing temperatures on amphibian and reptile species may be less deleterious than previously postulated; climate cooling might be more deleterious for the persistence of amphibian and reptile species than warming (Araujo et al. 2006). However, despite increasing temperatures being potentially advantageous to amphibians, these benefits might be offset by projected decreases in the availability of water (Araujo et al. 2006).

3.2.2 Precipitation

Precipitation and its seasonality and, in particular, droughts, have been shown to reduce populations of mammals and birds. Precipitation has been shown to explain spatial patterns of bird abundance in Australian tropical rainforest (Williams & Middleton 2008), influence the meta-population dynamics of desert bighorn sheep (*Ovis canadensis*) in the mountains of California (Epps et al. 2004) and abundance of swamp antechinus (Magnusdottir et al. 2008). Strong relationships between abundance and rainfall suggest that rainfall underpins the dynamics of African savanna ungulates, and that changes in rainfall due to global warming may markedly alter the abundance and diversity of these mammals (Ogutu et al. 2008b). Droughts can cause resource bottlenecks (i.e. lack of insects, nectar or fruit) to tropical birds in Australia (Williams & Middleton 2008).

3.2.3 Extreme events

Extreme temperature or precipitation events can have more significant impacts on species than gradual climatic changes. Extreme temperatures exceeding the physiological limits of species have caused

mortality in Australian flying-fox species (Welbergen et al. 2008). Floods have caused catastrophic, species-specific mortality in desert rodents resulting in rapid population and community-level changes (Thibault & Brown 2008). Interactions of extreme events with phenological changes can result in reduced fecundity. Over time with increasing temperatures, flowering is advancing at the Rocky Mountain Biological Laboratory, Colorado, USA, and therefore increasingly exposing buds to frost kills (Inouye 2008).

3.2.4 Phenological changes

Climate change impacts on the timing of many natural events have been documented for many species. Several hundred papers have been published during the past two years documenting phenological changes for plants and animals.

Reviews of reported climate change impacts on plant phenology support the IPCC AR4 conclusions of advanced leafing, flowering and fruiting (3-5 days per degree C temperature increase) and delays in autumn events (Bertin 2008; Cleland et al. 2007; Fujimoto 2008; Menzel et al. 2006). Most observations of phenological changes are from the northern hemisphere, including Europe (Menzel et al. 2006), Japan (Fujimoto 2008) and the Arctic (Hoye et al. 2007). As plants are finely tuned to the seasonality of their environment, shifts in timing of plant activity provide most compelling evidence that they are affected by climate change (Cleland et al. 2007). Large proportion of the observed variability in life cycle events can be attributed to climate change (Van Vliet 2008). Experimental results indicate that increased temperature is the driver of advancing first flowering in temperate grassland of Tasmania, and not elevated CO₂ levels (Hovenden et al. 2008). Climate, particularly seasonality, is probably the primary driver of beta diversity among rain forest trees of the Western Ghats complex (Davidar et al. 2007).

There is ample evidence showing that reproduction of insects, birds, and amphibians is influenced by spring temperatures (Both et al. 2006; Dolenec 2007; February et al. 2007; Gaston et al. 2005; e.g. Gordo & Sanz 2005; Parmesan 2007). Changes in phenology have been linked to population declines, potentially due to direct impacts, e.g. reduced number of eggs laid by the Helmeted Honeyeater (*Lichenostomus melanops cassidix*) (Chambers et al. 2008), or due to decoupling of species interactions (e.g. food, pollinators) causing high mortality in Pied Flycatchers nestlings (Both et al. 2006).

There are no studies projecting or modelling phenological changes, probably as it is uncertain how species will respond when they reach temperature thresholds.

3.2.5 Migration

A number of studies have reported variation in timing of migration among bird species, showing earlier spring arrivals for birds in both North America and Europe (Jonzen et al. 2006; Kralj & Dolenec 2008; Mezquida et al. 2007; Miller-Rushing et al. 2008). Analysis of year to year variations suggests that short-distance migrants in North America respond to changes in temperature, while mid-distance migrants responded particularly strongly to changes in the Southern Oscillation Index (Miller-Rushing et al. 2008). Birds may be able to adjust migration schedules phenotypically to tune their arrival dates optimally (Saino & Ambrosini 2008) to meteorological conditions at the beginning of the breeding season.

3.2.6 Food supply

There may be both direct effects of climate change on food supply for some species, and indirect effects such as through fire.

As phenology advances in response to climatic warming, there is potential for development of a mismatch between the peak of resource demands by reproducing animals and the peak of resource availability. Pied Flycatcher and Macaroni penguin breeding success is likely to be reduced with potential climate induced changes in prey availability (Both & te Marvelde 2007; Cresswell et al. 2008). For migratory herbivores,

such as caribou, development of a trophic mismatch is particularly likely because the timing of their seasonal migration to summer ranges, where calves are born, is cued by changes in day length, while onset of the plant-growing season on the same ranges is cued by local temperatures. As mean spring temperatures have risen by more than 4°C, caribou have not kept pace with advancement of the plant-growing season on their calving range. As a consequence, offspring mortality has risen and offspring production has dropped fourfold (Post & Forchhammer 2008). Phenological shifts have reduced the floral resources available to pollinators, resulting in a decreased diet breadth of the pollinators, and disruption of plant-pollinator interactions (Mommott et al. 2007).

Disturbance due to fires driven by an El-Niño event has resulted in a serious decline of fruit resources for sun bears (80% of sun bear fruit and 95% of another fruit species) and, due to the scale of fire damage, in a serious decline of prime sun bear habitat (44% reduction in species diversity) (Fredriksson et al. 2007).

3.2.7 Growth

Tropical tree growth rates are negatively correlated with mean annual temp (Clark et al. 2003), but might also be affected by drought (Feeley et al. 2007). Overall impacts of global change on tropical forest growth remain poorly understood (Feeley et al. 2007).

3.2.8 Fecundity and reproduction

There is some evidence that fecundity is affected by climatic variation and may therefore be affected by long-term climate change. The average fecundity of a threatened lemur (*Propithecus edwardsi*) in Madagascar was over 65% lower during El Niño years. While not as severe as deforestation or hunting, if El Niño events remain at the current high frequency there may be negative consequences for the population (Dunham et al. 2008). For the critically endangered Helmeted Honeyeater (*Lichenostomus melanops cassidix*) of central southern Victoria, Australia, climate also plays a role in the timing and success of breeding. During the period 1989 to 2006, the timing of laying became earlier and there was a possible reduction in the mean number of eggs laid per breeding season (Chambers et al. 2008). Similarly, Macaroni penguins breeding success was reduced (Cresswell et al. 2008).

Projections of range shifts often do not take species-ecosystem interactions into account. Schweiger et al. (2008) have modelled the relationship between a specialist butterfly species and its host plant in Europe for 2080 and found that there was a reduction in range overlap, potentially resulting in asynchrony between food sources and breeding causing starvation of young.

3.2.9 Sex ratios

In many egg-laying reptiles, the sex of offspring is determined by the temperature experienced during a critical period of embryonic development. Increasing air temperatures are likely to skew offspring sex ratios in the absence of evolutionary or plastic adaptation. Increasing air temperatures are likely to skew offspring sex ratios in cold climate reptiles such as the tuatara (*Sphenodon guntheri*). Under extreme regional climate change, by the mid-2080s all-male clutches would hatch at 100% of current nest sites (Mitchell et al. 2008). In theory tuatara could compensate behaviourally for the male-biasing effects of warmer air temperatures by nesting later in the season or selecting shaded nest sites, like other lizards do (Doody et al. 2006), however, many species are nesting earlier as the climate warms.

3.2.10 Dispersal

A long term study on the impact of temperature change on lizards showed juvenile dispersal declined dramatically over 16 years, correlated with a rise in spring temperature during development. This is likely to elevate the extinction risk of meta-populations (Massot et al. 2008).

3.2.11 Competition/encroachment

The difference in phenological response between different functional groups may potentially increase competition within ecosystems, e.g. grasslands (Cleland et al. 2006). Early successional species can germinate at higher soil temperatures and may thereby increase in importance within a habitat (Colwell et al. 2008). Experimental work has also supported the potential role of CO₂ enrichment in promoting woody plant invasion of grasslands through its effect on competitive interactions between grass and tree seedlings (Bloor et al. 2008). In alpine systems experimentally raised temperatures appeared to increase the negative relationship between resident species diversity and species establishment (Klanderud & Totland 2007).

3.2.12 Disease, parasites and herbivory

Climate change impacts on the complex interactions among host, pathogen and environment are poorly understood. However, there is some evidence that climate change is causing impacts on species by changing disease distributions and their severity, as species are stressed by increased temperatures.

Further research has added to the conclusion of Pounds et al. (2006) that amphibian declines have already been caused by climate change largely through increase in disease (Alford et al. 2007; Fisher 2007; Laurance 2008; Muths et al. 2008). A long-term observational study showed significant association between change in local climatic variables and the occurrence of chytridiomycosis in Spain (Bosch et al. 2007). The largest study of global amphibians to date implicates climate change as a factor in amphibian decline. In the 2,454 species that declined between 1980 and 2004, small range size, habitat loss, and extreme seasonality in precipitation contributed to the risk of decline (Sodhi et al. 2008). However, Lips et al. (2008) found no evidence to support the hypothesis that the increase in amphibian disease epidemics is climate driven. Environmental conditions impact the spread of amphibian disease as warming temperatures may facilitate the spread of diseases into previously un- or little-affected areas (i.e., higher elevations) (Muths et al. 2008).

Short-term, local experiments have demonstrated the impacts of predicted global change on plant health including a study showing that elevated atmospheric CO₂ concentration increases the risk of infection with rice blast (*Magnaporthe oryzae*) and the percentage of rice (*Oryza sativa*) plants affected by sheath blight (Kobayashi et al., 2006); experiments demonstrating species-specific responses to increased ozone concentrations of the susceptibility of young beech (*Fagus sylvatica*) and spruce (*Picea abies*) trees to *Phytophthora citricola* (Lüdemann et al., 2005); and a 12 year warming experiment with heaters suspended over plots in a mountain meadow in Colorado, USA in which there was a change in the prevalence of different species of plant pathogens (Roy et al., 2004). The evidence that climate change can profoundly influence host-pathogen dynamics is growing, not only for plant diseases but also for animal and human diseases (e.g. Purse et al., 2005; Haines et al., 2006).

Projected ranges under climate change of the tick-borne disease Theileriosis (East Coast fever) in sub-Saharan Africa show increases in suitability for some areas (Olwoch et al. 2008). Severity of plant diseases are projected to be correlated with climate change, e.g. *Plasmopara viticola* epidemics on grapes in an important wineproducing Italian region near Turin in 2030, 2050 and 2080 (Salinari et al., 2006), the range and severity of epidemics of *Leptosphaeria maculans* on oil seed rape (*Brassica napus*) in the UK for the 2020s and 2050s (Evans et al., in press), and (iii) the distribution and local impact of a range of forest pathogens (*Biscogniauxia mediterranea*, *Cryphonectria parasitica*, *Melampsora* spp., *Phytophthora cinnamomi* and *Sphaeropsis sapinea*) in France at the end of the 21st century (Desprez-Loustau et al., 2007) (Jeger & Pautasso 2008).

The lack of integrated long-term data on hosts, diseases and environment, especially for specific habitats, e.g. Arctic, limits our ability to predict the effects of climate change on diseases and interactions (Burek et al. 2008).

Some evidence from paleontological studies that pressures on plants may increase with climate change. Amount and diversity of insect damage to plants increased in association with an abrupt rise in atmospheric CO₂ and global temperature that occurred 55 million years ago (Curran et al. 2008). Gordo & Sanz (2005) suggest that insect phenology showed a steeper advance than plant phenology, resulting in a decoupling of some plant-insect interactions, including herbivorous insects and their plant resources. Damage to northern European birch forests caused by leaf-chewing and leaf-mining insects is projected to be at least double with expected climatic warming (Kozlov 2008). This increase in insect damage can change predictions of future forest composition (Wolf et al. 2008b). Predicted temperature increases are likely to enhance the potential insect impacts on vegetation in Europe, an issue generally neglected by vegetation models (Wolf et al. 2008b).

3.3 Characteristics and factors contributing to vulnerability or resilience

The IPCC AR4 estimated that 20-30% of species assessed would be at risk of extinction if climate change leads to global average temperature rises greater than 1.5 -2.5°C. Recent work suggests that for birds, amphibians and warm water corals as many as 35-70% may be at risk from climate change (Foden et al 2008), and has added to our understanding of the characteristics that contribute to species' risks of decline or extinction. Species with small ranges are at particular risk (Pompe et al. 2008; Sodhi et al. 2008; Walther et al 2005), as are those with naturally fragmented or isolated populations; tropical montane species are at particular risk (Wake & Vredenburg 2008). There is some suggestion that northern, cold-adapted species are at risk due to increased competition from species invading from warmer climates (Walther et al 2005; Berry et al. 2006). However, since AR4 greater concern has developed that in fact tropical species, especially ectotherms (reptiles, amphibians, insects), may be most at risk because of their low thermal tolerances (Deutsch et al. 2008). On the other hand, physiologists point out that tropical trees are capable of tolerating appreciable increase in temperature (Lloyd & Farquhar 2008). Species that require late-successional habitat in ecosystems that are intolerant of fire or drought are high risk from climate change and its interaction with fire (Nitschke & Innes 2006). Limited dispersal ability is also a key risk factor (Foden et al 2008).

4 Genetic diversity

Climate change impacts also affect genetic diversity and its maintenance. Genetic diversity is important both in its own right and in determining the resilience of species to the impacts of climate change and other pressures (Botkin et al. 2007). For example, experimental work has shown that eelgrass communities are much more resilient to increased temperature when they include high genetic diversity (Ehlers et al. 2008). In this example genetic diversity within a single species is crucially important for continued ecosystem function. Individual plant traits can also strongly influence the biogeochemical cycling of carbon, and differences in inter- and intraspecific responses to elevated CO₂ affect not only physiology and growth, but also higher order biotic interactions and lifetime fitness, ultimately leading to new ecosystem assemblages (Bradley & Pregitzer 2007).

Despite its importance, relatively little effort has yet been devoted to investigating the impacts of climate change on genetic diversity. One clear impact is in the fragmentation of populations when their habitats are fragmented by climate change, as in moist mountain ecosystems surrounded by drying lowlands or in sea ice dependent communities. There is evidence that reduction in sea ice cover is effectively reducing gene flow in polar bears (Crompton et al. 2008), and may have similar effects for other marine mammals (O'Corry-Crowe 2008).

The implications of climate change for genetic diversity also have potentially strong implications for human well-being. Crop wild relatives are an important source of genetic diversity for crop improvement. However, climate envelope modelling has shown recently that the survival of over 20% of the wild relatives of peanut, potato and cowpea may be threatened with extinction under climate change, most will lose over 50% of their range size, and the distributions of many will become highly fragmented (Jarvis et al. 2008).

Further research is needed to broaden and deepen our understanding of the role of genetic diversity in resilience to climate change and the degree to which that diversity is under threat from climate change and its interaction with other pressures.

5 Ecological Interactions

As is clearly illustrated above in the sections on ecosystems and species, climate change is likely to affect ecological interactions, including competition, disease and host-parasite interactions, pollination and herbivory. For example, it has been suggested that climate warming may increase the role of interspecific competition in determining alpine plant community structure and diversity (Klanderud & Totland 2007). Experimental work has also supported the potential role of CO₂ enrichment in promoting woody plant invasion of grasslands through its effect on competitive interactions between grass and tree seedlings (Bloor et al. 2008). Differences in phenological responses between different functional groups may potentially increase competition within grassland ecosystems (Cleland et al. 2006). The greater effect of warming in suppressing productivity in more species rich experimental communities (De Boeck et al. 2008)) has been attributed to negative impacts of intense interspecific competition for resources under conditions of high abiotic stress.

There is ample evidence that warming will alter the patterns of plant, animal and human diseases. A 12 year warming experiment in a mountain meadow in Colorado, USA showed a change in the prevalence of different species of plant pathogens (Roy et al. 2004). Numerous modeling studies project increases in economically important plant pathogens with warming, and experimental studies show similar patterns (Jeger & Pautasso 2008). There has been considerable recent concern over the role of climate change in promoting the emergence of new infectious diseases (Jones et al. 2008) and in changing the distribution of existing ones. Studies of the impacts of climate change on the range of the tick-borne disease Theileriosis (East Coast fever, ECF) in sub-Saharan Africa, the Northern Cape and Eastern Cape provinces of South Africa, Botswana, Malawi, Zambia and eastern DRC show increases in ECF suitability (Olwoch et al. 2008).

Recent evidence suggests that mismatches in phenological responses to climate change between plants and pollinators may significantly affect their interactions (Bertin 2008). Modelled phenological shifts in response to climate change reduced the floral resources available to 17-50% of all pollinator species. Reduced overlap between plants and pollinators also decreased diet breadth of the pollinators (Memmott et al. 2007). These patterns could lead to the extinction of pollinators and/or plants and disruption of their interactions. Long term and model-based studies of the Mediterranean basin showed a steeper advance in insect phenology than in plant phenology, suggesting the potential for increased decoupling of interactions between pollinators and flowers (Gordo & Sanz 2005).

Interactions between herbivores and plants are also likely to change as a result of climate influence. Recent observations of herbivore damage on plant fossils suggests that herbivore pressures on plants may increase with climate change; the amount and diversity of insect damage to plants increased in association with an abrupt rise in atmospheric CO₂ and global temperature that occurred 55 million years ago (Currano et al. 2008; DeLucia et al. 2008). On the other hand, as for pollination, phenological changes arising from climate change may cause decoupling between herbivores and their plant resources, as has been suggested for Mediterranean ecosystems (Gordo & Sanz 2005). Climate change has been blamed for the extreme severity of a recent mountain pine beetle outbreak in British Columbia, which has effectively turned the forest from carbon sink to carbon source (Kurz et al. 2008a). In northern Europe, damage of northern birch forests caused by leaf-chewing and leaf-mining insects is projected to be at least double with expected climatic warming (Kozlov 2008). This increase in insect damage can change predictions of future forest composition (Wolf et al. 2008b). Predicted temperature increases are likely to enhance the potential insect impacts on vegetation in Europe, an issue generally neglected by vegetation models (Wolf et al. 2008b). On the other hand, climate change can cause reductions in overlap between herbivores and their host plants (Schweiger, I 2008).

The effects of climate change on ecological interactions like these are a large part of the key to understanding the likely effects of climate change on species and ecosystems.

6 Feedbacks to climate

Natural ecosystems are an integral part of the carbon cycle. The relationship between climate and biodiversity is not linear; and climate change impacts on natural ecosystems can exert significant positive feedbacks to the climate system. Greenhouse gas emissions from land use change have been estimated to account for 20% of all anthropogenic emissions (IPCC 2007); an estimate that could be amplified by climate change. This feedback cycle is not incorporated into current climate models, but is an area of growing research; particularly following concerns over the continued climate change mitigation capacity of ecosystems such as forest reported in the IPCC 4AR.

It is generally agreed that one of the main feedbacks to the climate system will be through the increase in soil respiration under increased temperature, particularly in the arctic (Chapin *et al.* 2008), with the potential to add 200ppm CO₂ to the atmosphere by 2100 (Canadell *et al.* 2007). Although the exact dynamics are still unclear, recent research has suggested that feedbacks from the two major soil carbon stores, permafrost and peatland, could be considerable (Smith *et al.* 2008; Wang *et al.* 2008). Estimates for emissions from the thawing of permafrost, for example, have ranged from global increases of 100PgC by 2100, to 40-100Pg increases from Canada and Alaska alone by 2100. It has also been projected that a 10% thawing of the Siberian permafrost will release 40Pg by 2050; an increase that will not be offset by the predicted advance of the treeline into the tundra (Ise *et al.* 2008; Schuur *et al.* 2008). Loss of soil invertebrates in low diversity ecosystems can also contribute to carbon fluxes through altered ecosystem functioning. (Ayres *et al.* 2008; Barrett *et al.* 2008; Poage *et al.* 2008).

Peat emissions are linked to water table levels, which are highly vulnerable to climate change (Ise *et al.* 2008). In addition, experimental evidence has suggested that warming climate will alter the species composition of peat; with vascular plants dominating at the expense of peat forming species; reducing the capacity of peat to sequester carbon (Breeuwer 2008) (Garant *et al.* 2008) (Heijmans *et al.* 2008) (Fenner *et al.* 2007). It is not just increasing temperatures that can lead to such feedbacks. One issue that has not received much coverage in the literature is that of potential impacts on sea level rise. A study in California has suggested that inundation of the 150,000km² of low-lying peatlands may cause substantial emissions (Henman & Poulter 2008).

One area of research that has expanded since the 4AR is that of the projected Amazon drying and dieback (Huntingford *et al.* 2008). Although there is still considerable uncertainty, most models predict reduced precipitation leading to increased drying of the Amazon rainforest (Betts, Sanderson & Woodward 2008). Indeed, it has been suggested that CO₂ emissions will be accelerated by up to 66% due to feedbacks arising from global soil carbon loss and forest dieback in Amazonia as a consequence of climate change (Betts *et al.* 2006). Again, impacts are not solely down to increasing temperature; substantial loss of forest from the Amazon is likely to impact on both the global carbon cycle and the regional climate through altered precipitation and emissions of dust (Betts, Sanderson & Woodward 2008). This is exacerbated by deforestation and degradation, which increases the vulnerability of forest and lowers resilience for adaptation to climate change; therefore lowering the value of the Amazon in mitigation. Climate-ecosystem feedbacks have also been implicated in droughts in the Sahel and Western Australia (Chapin *et al.* 2008)

This is not just true of the tropics. Recent research has suggested that altered hydrology in boreal forest alters freshwater inputs to the Arctic Ocean, with subsequent impacts on sea ice, and feedbacks through changes in latent heat flux and albedo (Woo *et al.* 2008). Loss of vegetation can also influence the surface albedo, providing further feedbacks to climate. In addition, it has been suggested that impacts of climate change on temperate forest could reduce the capacity to act as a carbon sink (Gough *et al.* 2008); through processes such as increased severity of insect outbreaks (Kurz *et al.* 2008a).

Indeed, although ecosystems are currently acting as a carbon sink to sequester 30% of anthropogenic emissions, global scale climate scenario modelling suggests that the terrestrial biosphere will become a carbon source by 2100, largely due to the increased soil respiration and the dieback of the Amazon. Climate models incorporating these feedbacks led to a 0.38°C or 8% increase in warming compared to a model when feedback was not considered (Betts *et al.* 2008). Such modeling is, however, still uncertain (Chapin *et al.* 2008). The interaction of the carbon cycle with the nitrogen cycle is also not included in climate models (Gruber & Galloway 2008); although it has been estimated that increased carbon sequestration may lead to an increase of N₂O emissions in grassland (Kammann *et al.* 2008).

Recent evidence supports the findings reported in the 4AR that impacts of climate change on ecosystems are likely to be amplified by positive feedbacks. Further research incorporating such feedbacks into climate models is required (Chapin *et al.* 2008).

7 Conclusion

The main lesson from recent research on the impacts of climate change on biodiversity is that many of the key findings at time of IPCC AR4 have been strengthened, with a greater range of evidence, including observational evidence, to support them. While there are some specific areas where new understanding has emerged or the balance of evidence has shifted, the larger scale picture is one of increased support for earlier findings.

The IPCC AR4 described the evidence for the effects of climate change on natural ecosystems. More recent observational, experimental and modelling work has pointed to several broad types of major changes to ecosystems as a result of climate change. Modelling studies combined with experimental evidence of species tolerances point to significant changes in the distribution of some ecosystems, principally due to increasing temperature and altered precipitation regimes. Likely distribution changes include poleward shifts in boreal regions and upwards shifts in montane systems, where lack of space at higher altitudes may cause some systems to disappear entirely. Ecosystem distribution changes are also expected to be large in the tropics, where the effects of rising temperatures and reduced precipitation are exacerbated by the effects of land use change. Drier conditions are expected to cause savannah ecosystems to move into equatorial regions now occupied by forests.

In addition to shifting their locations climate change will alter the composition of many ecosystems. Site level reductions in species richness are of concern because under changing environmental conditions, multiple species play a role in ensuring that ecosystem processes can continue. Processes potentially dependent on species richness include carbon storage. Climate change may play a role in facilitating the spread and establishment of invasive species, which can have major impacts on ecosystem composition.

Changes in species composition can lead to changes in the physical and trophic structure of ecosystems, with resulting further effects on system function and composition. One such change is the invasion of temperate grasslands by woody plants. In other systems, trees may disappear as a result of drought. Coral reefs are especially subject to adverse impacts due to bleaching and diseases promoted by warmer temperatures and increasing pressures from ocean acidification. Many reef-building coral species are threatened with extinction. This has major implications for the large biological communities that coral reefs support.

Climate changes in combination with changes in ecosystem composition and structure have been shown both by modelling and experimentation to lead to changes in ecosystem function. Models suggest that global net primary production (NPP) has already increased in response to changes in temperature and precipitation during the 20th century. Regional modelling projects increases in NPP for some regions, but possible declines in others.

A key property of ecosystems that may be affected by climate change is the values and services they provide to people. These include provisioning services such as fisheries, which may improve in the short term in boreal regions and decline elsewhere, and timber production, where the response depends on population characteristics as well as local conditions and may include large production losses. The impacts on coral reefs threaten the vital ecosystem services these systems provide through fisheries, coastal protection, building materials, new biochemical compounds, and tourism. Climate change also affects the ability of terrestrial ecosystems to regulate water flows, and critically reduces the ability of many different ecosystems to sequester and/or retain carbon which can feedback to climate change.

At the species level the recent observed and modelled climate change impacts show that climatic change has already caused changes to the distribution of many plants and animals, leading to severe range contractions and the extinction of some species. Changes of terrestrial species include shifts in spring

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events (for example, leaf unfolding, flowering date, migration and time of reproduction), species distributions and community structure. In marine-ecosystems changes have been demonstrated in functioning and productivity, including shifts from cold-adapted to warm-adapted communities, phenological changes and alterations in species interactions. Some species are unable to disperse or adapt fast enough to keep up with high rates of climate change and these species face increased extinction risk and, as a result, whole ecosystems, such as cloud forests and coral reefs, may cease to function in their current form. The IPCC AR4 estimated that 20-30% of species assessed would be at risk of extinction if climate change leads to global average temperature rises greater than 1.5 -2.5°C. Recent work suggests that for birds, amphibians and warm water corals as many as 35-70% may be at risk from climate change.

Despite its importance, relatively little effort has yet been devoted to investigating the impacts of climate change on genetic diversity. One clear impact is the fragmentation of populations when their habitats are fragmented by climate change. There may also be impacts on crop wild relatives, which are an important source of genetic diversity for crop diversity.

Climate change is likely to affect ecological interactions, including competition, disease and host-parasite interactions, pollination and herbivory. There is ample evidence that warming will alter the patterns of plant, animal and human diseases. Numerous modelling studies project increases in economically important plant pathogens with warming, and experimental studies show similar patterns. There has been considerable recent concern over the role of climate change in changing the distribution of diseases. Studies of the impacts of climate change on the range of the tick-borne disease Theileriosis (East Coast fever, ECF) show increases in areas of ECF suitability in Africa. Short-term, local experiments have demonstrated the impacts of predicted global change on plant health including rice.

Recent evidence suggests that mismatches in phenological responses to climate change between plants and pollinators may significantly affect their interactions. These patterns could lead to the extinction of pollinators and/or plants and disruption of their interactions.

Climate change impacts on ecosystems can exert significant positive feedbacks to the climate system. It is generally agreed that one of the main feedbacks to the climate system will be through the increase in soil respiration under increased temperature, particularly in the arctic with the potential to add 200ppm CO₂ to the atmosphere by 2100. One area of research that has expanded since the 4AR is that of the projected Amazon drying and dieback. Although there is still considerable uncertainty, most models predict reduced precipitation leading to increased drying of the Amazon rainforest. It has been suggested that CO₂ emissions will be accelerated by up to 66% due to feedbacks arising from global soil carbon loss and forest dieback in Amazonia as a consequence of climate change.

Although ecosystems are currently acting as a carbon sink to sequester 30% of anthropogenic emissions, global scale climate scenario modelling suggests that the terrestrial biosphere will become a carbon source by 2100, largely due to the increased soil respiration and the dieback of the Amazon.

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