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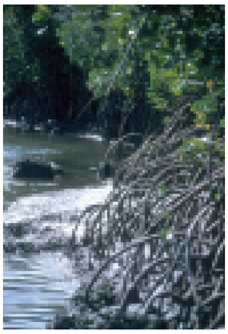

REVIEW OF THE LITERATURE ON THE LINKS BETWEEN BIODIVERSITY AND CLIMATE CHANGE

Impacts, Adaptation and Mitigation

Convention on
Biological Diversity

UNEP

CBD Technical Series No. 42

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Impacts, Adaptation and Mitigation

Acknowledgements

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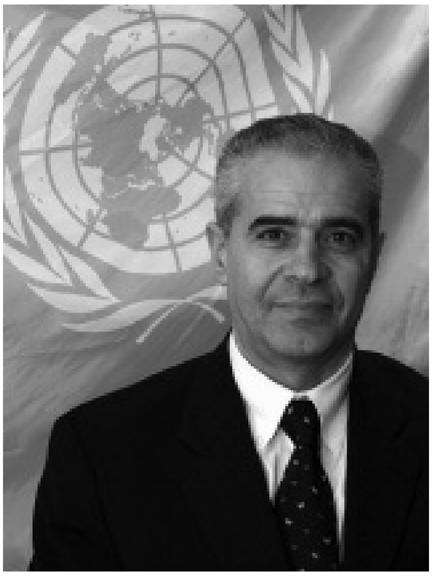
FOREWORD



The achievement of the three objectives of the Convention on Biological Diversity (CBD) – the conservation of biodiversity, the sustainable use of its components, and the fair and equitable sharing of the benefits from the use of genetic resources – is coming under threat from one of the world's major environmental, social and economic challenges: climate change.

Climate change adds to the global challenge of biodiversity conservation. There is ample scientific evidence that climate change affects biodiversity. It is threatening individual species as well as entire ecosystems, with negative consequences for human well-being. However, the links between biodiversity and climate change flow both ways. Biodiversity, through the ecosystem services it supports, makes an important contribution to both climate change mitigation and adaptation. The interlinkages between biodiversity, climate change, and sustainable development, have been recognized within both the Convention on Biological Diversity (CBD) and the United Nations Framework Convention on Climate Change (UNFCCC), as well as other international fora.

Healthy, intact ecosystems have long provided critical ecosystem services, providing people with food and shelter, protecting communities from drought and floods, and building the basis of much of our traditional knowledge, innovations and practices. As climate change threatens food security and increases exposure to natural disasters, these ecosystem services will become even more important and valued.

Where species and ecosystems are well protected and healthy, natural adaptation may take place, as long as the rate of change is not too rapid and the scale of change is not too great. However, where climate change stacks as an additional threat upon other stresses such as pollution, overuse or invasive alien species, natural adaptive capacity may be exceeded. It is important, therefore, to ensure that biodiversity conservation and management considers the interplay of all human activities, including climate change.

The report of the Second *Ad Hoc* Technical Expert Group (AHTEG) on Biodiversity and Climate Change, which has been published as CBD Technical Series No. 41, *Connecting Biodiversity and Climate Change Mitigation and Adaptation – Report of the Second* Ad Hoc *Technical Expert Group on Biodiversity and Climate Change*, is the outcome of scientific and technical deliberations conducted by experts from 23 countries as well as United Nations organizations, intergovernmental and non-governmental organizations, and representatives from indigenous communities. The present document provides a review of recent scientific literature on the links between biodiversity and climate change which was used for reference by the AHTEG.

I wish to thank the World Conservation Monitoring Centre of the United Nations Environment Programme (UNEP-WCMC) for preparing this important document.

Ahmed Djoghlaf
Executive Secretary
Convention on Biological Diversity

PREFACE

These three literature reviews on the 'Links between Biodiversity and Climate change: Impacts, Adaptation and Mitigation' were produced by UNEP-WCMC. They were commissioned by the UK Department for the Environment, Food and Rural Affairs (with additional support from the Ministry of Environment, Finland) to provide background material for the Convention on Biological Diversity (CBD) Second *Ad Hoc* Technical Expert Group (AHTEG) on Biodiversity and Climate Change. These reviews complement the CBD Technical Series No. 41 *Connecting Biodiversity and Climate Change Mitigation and Adaptation*. This work was reviewed at the meetings of the CBD second AHTEG on Biodiversity and Climate Change held in London in November 2008 and in Helsinki in April 2009. The reviews were subsequently peer reviewed.

The IPCC 4th Assessment Report (AR4; IPCC 2007) concluded that climate change will have significant impacts on many aspects of biological diversity; on ecosystems, species, 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. Adaptation strategies will be needed to respond to these impacts. Countries are starting to develop and implement adaptation policies. These adaptation strategies tend to focus on technological, structural, social, and economic developments, and the linkages between biodiversity and adaptation are often overlooked. Nevertheless, biodiversity is linked to climate change adaptation, in its role in adaptation strategies and the impacts of adaptation strategies on it. Biodiversity is also important with regards to mitigation policies. Indeed, the IPCC AR4 provided growing evidence of the importance of natural ecosystems in the carbon cycle and in mitigation policies. In addition, it was recognised that climate mitigation polices focussed on reducing carbon dioxide emissions can have impacts on biodiversity; both positive and negative.

Because of the importance of these impacts and of climate change itself, there has been a great deal of recent research on these three issues, though more for some than others. These three reviews focus on the scientific literature published after the AR4. The first part of this work reviews the literature on the impacts of climate change on biodiversity. The second section aims to provide a better understanding of the role of biodiversity in societal and in biodiversity conservation adaptation as well as the impacts of adaptation strategies on biodiversity. Finally the third section aims to highlight the developments in our understanding of the role of biodiversity in climate change mitigation, and the impacts of mitigation policies on biodiversity.

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SECTION 1

IMPACTS OF CLIMATE CHANGE ON BIODIVERSITY

A review of the recent scientific literature

Valerie Kapos, Jörn Scharlemann, Philip Bubb, Alison Campbell, Anna Chenery, Nathalie Doswald and Barney Dickson

CONTENTS

1.1 EXECUTIVE SUMMARY

The Intergovernmental Panel on Climate Change 4th Assessment Report (IPCC AR4) concluded that climate change will have significant impacts on many aspects of biological diversity; on ecosystems, species, 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 the 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, especially in non-equatorial 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 and more complex 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.

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 flow, and critically reduces the ability of many different ecosystems to sequester and/or retain carbon which can feedback to climate change. Modelling and experimental studies suggest that ecosystem function may change due to the combined effect of climate change and changes in ecosystem composition and structure. Furthermore, 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.

At the species level, recent observed evidence shows that climate change has already caused changes to the distribution of many plants and animals. Models of future climate change suggest that these distributional changes may lead to severe range contractions and the extinction of some species. Changes occurring for 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 per cent 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 further suggests that a greater percentage of species may be at risk from climate change.

Climate change is likely to affect ecological interactions, including competition, disease and host-parasite interactions, pollination, predator-prey interactions 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. Differences in phenological response to climate change between plants and pollinators may significantly affect their interactions and could lead to their extinction.

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 impacts on ecosystems and the species within 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 CO_2 to the atmosphere by 2100. One area of research that has expanded since the AR4 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 CO_2 emissions will be accelerated by up to 66 per cent due to feedbacks arising from global soil carbon loss and forest dieback in Amazonia as a consequence of climate change. 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, whereas it is thought that currently ecosystems are acting as a carbon sink sequestering 30 per cent of anthropogenic emissions.

On methodological questions, many of the studies reviewed here rely on experimental or modelled evidence of climate change impacts. Both of these simulate the biodiversity impacts of climate change in more or less realistic ways. Experimental studies are limited by the number of factors that can be manipulated simultaneously. Multi-factorial experiments may capture interactions more realistically. Most modelling studies are correlative and outputs from such studies are dependent on the choice of explanatory variables considered, with many studies only considering climatic variables. Given the importance of other factors such as land-use change in determining impacts, there is a strong case for building such factors into models in order to predict climate change impacts more accurately.

1.2 INTRODUCTION

1.2.1 Purpose and scope of the review

The Intergovernmental Panel on Climate Change 4th Assessment Report (IPCC AR4; IPCC 2007) concluded that climate change will have significant impacts on many aspects of biological diversity; on ecosystems, species, 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 report reviews the literature since the AR4 and before October 2008. It draws on recent research to summarise advances in our understanding of the impacts of climate change on biodiversity. However, it does not provide a full synthesis of the findings across different papers. Keyword searches in ISI Web of Knowledge, Scopus, and Google Scholar were carried out to obtain a broad coverage of the available literature.

The evidence for the impacts on biodiversity comes from three principal sources. First, from 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. Examples include observed phenological changes in bird arrival times and changes in distribution (Parmasan and Yohe 2003). Second, experimental studies using manipulations to elucidate responses to climate change. For example, examining the effect of addition of CO_2 on plant communities (Morgan *et al.* 2006; Bloor *et al.* 2008), or increases of temperature on plant phenology (Hovenden *et al*. 2008). Finally, and most widely, from 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; IPCC 2007), 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). Correlative modelling has been criticised for assumptions of equilibrium between species and current climate and an inability to account for variability within species or for population processes and migration. It is much improved by taking account of interactions between species, land cover and topographic variation (Heikkinen *et al.* 2007; Luoto and Heikkinen 2008). Species traits may have a strong impact on the performance of bioclimatic envelope models and some groups of traits are inherently difficult to model reliably (Pöyry *et al.* 2008). However, correlative and bioclimatic envelope modelling have been used successfully to simulate known species range shifts in the distant (Martinez-Meyer *et al.* 2004; Nogués-Bravo *et al.* 2008) 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 (Thuiller *et al* 2008).

This review attempts to distinguish among these classes of evidence in drawing attention to the most important recent findings on the impacts of climate change on biodiversity and their implications for human well-being.

1.2.2 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 0.7ºC over the last century and are predicted to continue rising. The IPCC (2007) projects that temperatures are likely to have risen by 1.1ºC to 6.4ºC by the end of the 21^{st} 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 (Fronzek and Carter 2007; Parra and 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 (Öhlemüller *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 carbon dioxide (CO_2) 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 CO_2 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 and 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.

1.3 ECOSYSTEMS

1.3.1 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. For example, there is some evidence of an upward shift of tree species (Beckage *et al.* 2008). There are likely to be future distribution changes in ecosystems from the tropics to the poles. The models project large impacts resulting from poleward shifts in boreal regions (Notaro *et al.* 2007; Alo and Wang 2008; Metzger *et al.* 2008; Roderfeld *et al.* 2008; Wolf *et al.* 2008a) and upwards shifts in montane systems, where lack of space at higher latitudes/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 and 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 of 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 and 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 (Levinsky *et al.* 2007; Buisson *et al.* 2008; Colwell *et al.* 2008; Trivedi *et al.* 2008b). Rising temperatures are a key factor in such turnover, but changing precipitation regimes are also important and rising CO_2 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 (e.g. Colwell *et al.* 2008; 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 regarding the role of climate change in facilitating the spread and establishment of invasive species, which can have major impacts on ecosystem composition (Hobbs *et al.* 2007b; Hellmann *et al.* 2008; Rahel and Olden 2008; Rahel *et al.* 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 and Schnitzer 2006; van der Heijden and Phillips 2008) and low rainfall (Swaine and Grace 2007). Model-based analyses of selected geographical areas indicate that climate-related changes in gross primary productivity (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 CO_2 concentrations (Morgan *et al.* 2007; Bloor *et al.* 2008) and which alters the availability of food for grass-eating herbivores. In other systems, trees may disappear as a result of drought (February *et al.* 2007; Foden *et al.* 2007; Badgley *et al.* 2008) and increase the probability of extinction for herbivores unable to digest C4 grasses, as well as the dispersal and dynamics of other plant species. Advances of the treeline also change the structure of montane systems (Beckage *et al.* 2008). Reduced calcification due to ocean acidification will change the structure of reef ecosystems even with relatively small increases in atmospheric CO_2 (Cao and Caldeira 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 per cent for northern European ecosystems (Olesen *et al.* 2007) as a result of longer growing seasons and higher CO_2 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 per cent. 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 or attribute 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, amongst other things, 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 CO_2 on tree growth (Lloyd and 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). 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 (Purse *et al.* 2005; e.g. 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 cynobacteria, resulting in increased threats to the ecological status of lakes and increased health risks (EEA *et al.* 2008).

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

1.3.2 Ecosystem types

1.3.2.1 Deserts and arid ecosystems

According to the IPCC AR4, deserts are likely to experience more episodic climate events, and inter-annual 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 droughts, but their impacts on productivity may be somewhat offset by the effects of increased atmospheric CO_2 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 increases 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.

1.3.2.1.1 Distribution

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 of a developing range shift in this species as a response to climate change (Foden *et al.* 2007). The data 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 likely to increase their vulnerability to climate change.

Modelling studies suggests that desert systems and their biodiversity are likely to be severely affected by climate change (Wu *et al.* 2007). 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 and that both generalist species with large geographic ranges and narrow-range endemics may be susceptible to climate change induced loss of potential range (Midgley and Thuiller 2007).

1.3.2.1.2 Composition

Recent climate change has already altered the composition of some desertic communities. A study of the composition of species in Chihuahuan Desert indicates a shift in composition since the late 1970s with an increase in woody shrubs and changes in small mammal species that coincides with recent climate change and cannot be attributed to other factors such as grazing (Brown, Valen and Curtin 1997). Changes in the precipitation regime may also lead to changes in plant composition (woody versus grass cover) potentially leading to competitive interactions and a decrease in biodiversity (Esler and Rundel 1999). 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). 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. At the other end of the spectrum, occasional catastrophic rainfall may cause dramatic flooding, which has been shown in at least one case (Thibault and 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.

1.3.2.1.3 Function

Water is the main limiting resource in arid ecosystems. Desert life is determined in part by the patterns of water availability, e.g. pulses of abundance and adaptation to drought (GEO 2006). Consequently changes in global or regional precipitation systems could severely affect these ecosystems (Brown, Valen and Curtin 1997). A higher incidence of drought may increase loss of bioproductivity, biodiversity and lead to erosion and deflation – leading to desertification (Le Houéro 1996; GEO 2006). Increasing future temperatures may increase evapotranspiration, which in combination with lower rainfall projected in some arid areas may decrease vegetation cover (GEO 2006). In contrast, it has been suggested that increased CO_2 levels may increase vegetation productivity (Lioubimtseva *et al.* 2004). Housman *et al.* (2006) found that productivity increased in Mojave Desert shrubs but only significantly during wet periods. Changes in productivity may also feedback changes in runoff, precipitation regimes and circulation (Lioubimtseva *et al.* 2004) as modelling suggests vegetation in arid ecosystems is important in the energy, moisture and carbon exchange and the atmosphere (Wang and Eltahir 2000).

1.3.2.1.4 Phenology

Housman *et al.* (2006) examined the effect of elevated CO_2 on production, photosynthesis and water retention of three Mojave Desert perennials over three years and found no change in the timing of peak canopy development. A three-year study conducted on desert plants in northern Oman shows that many aspects of these plants phenology (onset of growth, flowering, fruiting) are associated with rainfall (Gazanfar 1997). Changes in temperature extremes may also be important in characterizing differences in phenology between different sites (Elser and Rundel 1999). Changes in rainfall and temperature may therefore have an impact on desert phenology though there is a lack of literature in this area.

1.3.2.1.5 Ecosystem services

Changes to the composition, structure and function of deserts and arid regions have consequences dependent on what change occurs. Desertification leads to land degradation, loss of biodiversity and erosion which ultimately leads to loss of ecosystem services (GEO 2006). Increasing aridity and wind speed due to climate change may increase dust emissions from deserts. Although dust from deserts has some positive effect on the productivity of land and oceans, it can also be negative on some ecosystems and human health (GEO 2006). Deserts are not a large carbon sink (GEO 2006). However, if some areas are subject to increased precipitation, changes in vegetation cover may increase carbon sequestration in these areas (Lioubimtseva and Adams 2004).

1.3.2.2 Grasslands and 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 CO_2 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 CO_2 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 per cent; changing migration routes are a threat. Large reductions in species' range size have been projected.

1.3.2.2.1 Distribution

Recently, there have been further model-based predictions of northward shifts of savanna ecosystems into the Amazon in response to declining rainfall (Salazar *et al.* 2007; Cook and Vizy 2008), and the resulting increased incidence of fire in remaining forests is expected to increase 'savannization' still further (Barlow and 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).

1.3.2.2.2 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 Niño* 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 CO_2 concentrations are tending to reduce the importance of C4 grasses in some temperate grasslands (Soussana and Luscher 2007). Experimental doubling of CO_2 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 CO_2 concentration may be contributing to the shrubland expansions of the past 200 years and to invasions of grassland by woody plants.

1.3.2.2.3 Function

Water availability also affects grassland CO_2 dynamics, but its influence depends on microsite characteristics, e.g. in Yellowstone (Risch and Frank 2007). An understanding of this dependence needs to be incorporated into predictions of how changes in precipitation/soil moisture will affect CO_2 dynamics and how they may feed back to the global carbon 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 and Small 2007). Analysis of large numbers of data points for observed mean annual NPP, land cover class, precipitation, and temperature 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 (Del Grosso *et al.* 2008). 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 caused 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 increases 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 have been 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.

1.3.2.2.4 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 CO_2 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.

1.3.2.2.5 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.

1.3.2.3 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 CO_2 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.

1.3.2.3.1 Distribution

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 reductions in spring rainfall compared to other mountains in Europe (Bravo *et al.* 2008) and that these systems may decline dramatically (Metzger *et al.* 2008). Mediterranean tree species are modelled to replace sub-Mediterranean species in the Iberian Peninsula and to suffer less reduction in suitable habitat than other forest species (Garzón, Sánchez de Dios and Sainz Ollero 2008). However, other modelling work suggests that Mediterranean species may be more vulnerable due to their inability to expand in the projected suitable space (Schröter *et al.* 2005).

1.3.2.3.2 Composition

Bioclimatic modelling for European mammals has shown that potential mammalian species richness is predicted to reduce dramatically in the Mediterranean region (Levinsky *et al.* 2007). Ecophysiological studies of an endangered endemic tree in the Fynbos confirm its vulnerability to prolonged drought and the marked effect its loss would have on vegetation composition and structure (February *et al.* 2007). 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 (Sebastia, Kirwan and Connolly 2008).

1.3.2.3.3 Function

Modelling has suggested that Mediterranean systems will have the smallest increases, and in some cases decreases, in 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 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 per cent reduction and that the more humid ecosystems will become increasingly arid with attendant loss in productivity (Nunes *et al.* 2008).

1.3.2.3.4 Phenology

Advancement of spring and summer events since the 1970s has been shown for a sample of plants, insects and insectivorous birds (Gordo and 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.

1.3.2.3.5 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.

1.3.2.4 Forests and woodlands

According to IPCC AR4, modelling approaches predict that major changes in global forest cover are likely to occur 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.

1.3.2.4.1 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 150-200m (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 and 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 shift 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 and 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 analyses of 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 and 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 (Malhi *et al.* 2008; Li *et al.* 2008), and the role of sea surface temperatures in both the Atlantic and the Pacific in affecting rainfall in different seasons is now much better understood (Harris *et al.* 2008; Good *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 per cent (Salazar *et al.* 2007) to 70 per cent (Cook and Vizy 2008), and feedbacks from drying forest, increased fire incidence and fragmentation are expected to exacerbate these effects (Barlow and 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 and Power 2008) and ecophysiologists suggest that Amazon forest trees are capable of coping with increased temperatures (Lloyd and Farquhar 2008).

1.3.2.4.2 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 altitudes 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 and Peres 2008) and to include increasing abundance of lianas (Phillips *et al.* 2008).

1.3.2.4.3 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 a drying climate (Swaine and Grace 2007), but also considerable evidence that it is more immediately caused by canopy disturbance and increasing fragmentation of forest (Londre and Schnitzer 2006; van der Heijden and Phillips 2008).

1.3.2.4.4 Function

The evidence about the effects of climate change on such aspects of forest ecosystem function as CO_2 uptake, growth, and net primary production remains contradictory; warming and CO_2 fertilisation have largely positive effects (Luyssaert *et al.* 2007) while drying has negative effects on those aspects of ecosystem function. In temperate and boreal forests, increased warming and increased CO_2 concentrations are both likely to contribute to increased growth in deciduous species, but warming may reduce productivity in conifers (Ollinger *et al.* 2008). Increased respiration rates and changes in species composition seem likely to reduce carbon accumulation by temperate and boreal forests (Mohan *et al.* 2007; Kellomäki *et al.* 2008; Kurz *et al.* 2008b). The effects of elevated CO_2 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 and 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 and Innes 2008).

In Amazonian tropical forests, there is observational evidence of increasing productivity and related increased turnover, probably as a response to CO_2 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 per cent 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 (Bush *et al.* 2008; Barlow and Peres 2008).

1.3.2.4.5 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 that may be linked to changes in climatic variables (Wright and Calderon 2006).

1.3.2.4.6 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 (Nitschke and Innes 2006; Malhi *et al.* 2008).

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 than 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).

1.3.2.5 Tundra and 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 net carbon sinks to carbon sources, 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, are likely to experience large population declines and elevated extinction risks.

1.3.2.5.1 Distribution

There is some evidence that the area of tundra has decreased during the 20th century as forest tree species expand northwards, a shift associated with increasing temperatures (Juday *et al.* 2007). Recent modelling studies have further confirmed that likely encroachment from taiga vegetation into tundra is occuring 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). Specific tundra formations such as palsa mires in northern Europe are also modelled to disappear under most future climate change scenarios (Fronzek *et al.* 2008).

1.3.2.5.2 Composition

The changes in distribution outlined will also impact on the composition of arctic and tundra communities. Specific plant communities, such as those in snowbeds, may be invaded by species from adjacent communities (Björk and Molau 2007). Recent evidence indicates climate change has influenced and may influence trophic interactions in tundra ecosystems with changes in numbers of specialist predator species linked to changes in prey (Ines and Fuglei 2007). Recent work has also emphasised the importance of sea ice addition to the direct effects of climate itself in determining the survival and abundance of arctic species (Moore and 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).

1.3.2.5.3 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. However, carbon storage may increase as a result of tundra replacing polar deserts and by forests replacing tundra (Wolf *et al.* 2008a). Changes in permafrost could have effects on vegetation cover, soil hydrology and runoff (Fronzek *et al.* 2008). Melting of permafrost and other effects of warming on tundra (changes in soil moisture and vegetation cover) are also likely to increase methane emissions (Callaghan *et al.* 2007).

1.3.2.6 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. 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.

1.3.2.6.1 Distribution

Recently several new observational studies have confirmed glacial retreat on mountains around the world, including in China (Baker and Moseley 2007), the Alps (Cannone *et al.* 2008) and Colombia (Ruiz *et al.* 2008). There have also been observations of rapid colonization of the resulting deglaciated surfaces with more plastic species apparently playing a more important role in colonization (Cannone *et al.* 2008). Treelines have been observed moving up in altitude in both China (Baker and 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 and 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 modelling 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).

1.3.2.6.2 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 and Totland 2007).

1.3.2.6.3 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 under 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).

1.3.2.7 Inland waters

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 polewards 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 CO_2 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.

1.3.2.7.1 Distribution

There is new observational evidence of climatic impacts on inland waters. 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 popul-ations 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 on lakes especially in northern Arctic areas, increased and more rapid stratification, earlier and increased primary production, and decreased oxygenation at depth will possibly resultin 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 (Reist *et al.* 2006; Wrona *et al.* 2006a; Wrona *et al.* 2006b; Anisimov *et al.* 2007; Berry 2008). In monsoonal Asia, where ecological processes surrounding rivers are mediated by flow, disruptions in timing and velocity will have large environmental impacts (Dudgeon 2007). The interaction between climate change and land cover change is likely to lead to reduced discharge from many rivers that will in turn lead to significant loss of freshwater fish species (Xenopoulos *et al.* 2005; Xenopoulos and Lodge 2006)

1.3.2.7.2 Composition

There is new observational evidence of compositional change in fish communities in France over the last 15-25 years (Daufresne and 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 (Xenopoulos *et al.* 2005; Xenopoulos and Lodge 2006; 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 (Reist *et al.* 2006; Wrona *et al.* 2006a; Wrona *et al.* 2006b; Anisimov *et al.* 2007; Berry 2008). Temperature is a very important determinant of distribution and survival of aquatic macroinvertebrates at high latitudes, and changes in species composition have already been shown for boreal inland waters (Heino *et al.* 2009). It has been suggested that species characteristic of lentic systems may disperse more effectively than those of lotic systems (Hof *et al.* 2008), and therefore that lentic systems may show more rapid compositional change in response to changing climate (Heino *et al.* 2009). Models show that climate change will also affect wetland species composition through its effects on river flow, especially low water flows (Xenopoulos *et al.* 2005; Xenopoulos and Lodge 2006; 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 and Olden 2008; Rahel *et al.* 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 and Olden 2008; Rahel *et al.* 2008). Predictions done for Canada indicate that water temperature may change as much as 18° C by 2100, which would mean that a number of lakes will be newly vulnerable to invasion by smallmouth bass (Sharma *et al.* 2007).

Other factors that will interact with climate change in determining compositional change in inland waters include acidification (Conlan *et al.* 2007; Durance and Ormerod 2007), eutrophication (Heino *et al.* 2009) and land cover change, including change in composition of terrestrial systems (Chapin *et al.* 2005; Heino *et al.* 2009) and agricultural expansion (Heino *et al.* 2009).

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 and Chauhan 2006; Xenopoulos *et al.* 2005; Xenopoulos and Lodge 2006).

1.3.2.7.3 Ecosystem services

Significant impacts of climate change have been projected for both carbon storage and fisheries services 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 (Xenopoulos *et al.* 2005; Xenopoulos and Lodge 2006). A new concern is the effect of sea level rise on carbon storage in coastal wetlands, including 150,000 km^2 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 and 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 and Chauhan 2006). Ecosystem services provided by peatlands more broadly are also at risk, as temperature changes are expected to reduce their function as carbon sinks (Lloyd 2008).

1.3.2.8 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 CO_2 reduces the pH of surface waters and their concentrations of carbonate ions 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 compositions 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 suffer serious adverse effects from ocean acidification.

1.3.2.8.1 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 CO_2 released into the atmosphere by human activities over the last 200 years. As a result the ocean is the second largest sink for anthropogenic CO_2 after the atmosphere itself (Iglesias-Rodriguez *et al.* 2008). If emission levels were to continue at the same level to the year 2100, CO_2 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.* 2007).

Observational and experimental studies have shown that ocean acidification reduces the calcification rates of various calcifying species such as halimeda, benthic molluscs, forminifera and coccolithophores (Riebesell *et al.* 2007; Fabry 2008; Guinotte and Fabry 2008; 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 CO_2 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 lose 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 and Caldeira 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 CO_2 availability to increase their biomass (Guinotte and Fabry 2008).

1.3.2.8.2 Sea surface temperature

Since the AR4, further evidence has accumulated on the negative impacts of rising sea surface temperatures on species. The adult survival of the king penguin decreases with increasing temperature, 9 per cent decline per 0.26^{o} 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 (Hughes *et al.* 2007; Graham *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 the 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 and 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 3.6 m 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 and 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 and Roopnarine 2008).

Experimental studies suggest that the balance between native and introduced species of Eelgrass 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 with extreme weather events. In the Atlantic sea surface temperature is associated with around a 40 per cent increase in hurricane frequency and activity (Saunders and Lea 2008). Hurricanes can negatively impact coastal ecosystems such as coral reefs and mangroves, which can have both economic and social implications as both provide important ecosystem services.

1.3.2.8.3 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 and Heide-Jorgensen 2005; Simmonds and Isaac 2007; Laidre *et al.* 2008). 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 are 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 have shown 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.

1.3.2.8.4 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 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 a 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 per cent 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 sub-continent (Jagtap and Nagle 2007). However, some mangrove systems appear to be stable in the face of sea level rise (Sanders *et al.* 2008).

1.3.2.8.5 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).

1.4 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 have occurred in terrestrial and marine ecosystems; they include phenological changes (for example in leaf unfolding, flowering date, migration and time of reproduction), species distributions, community structure, species interactions, changes in ecosystem functioning and productivity, including shifts from cold-adapted to warm-adapted communities (e.g. Edwards and Richardson 2004; Rosenzweig *et al.* 2008). 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.

1.4.1 Changes in distribution

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 surveying and resurveying at high spatial resolution.

1.4.1.1 Poleward shifts

Meta-analyses of observed impacts on species found that there have been significant range shifts towards the poles in the recent past (Parmesan and Yohe 2003; Root *et al.* 2003). Recent observational evidence, including post AR4, for more species, including plants (Colwell *et al.* 2008), invertebrates (Hickling *et al.* 2006; Franco *et al.* 2006; Mitikka *et al.* 2008), and vertebrates (Gaston *et al.* 2005; Hickling *et al.* 2006; Hitch and Leberg 2007; Lemoine *et al.* 2007; Sorte and Thompson 2007; Schliebe *et al.* 2008) strengthen these findings of substantial latitudinal shifts of range boundaries, centres of occurrence and abundance.

Modelled future responses in distribution have further predicted poleward shifts for plants, insects, birds and mammals at various scales in the Northern Hemisphere, mainly in North America and Europe (Huntley *et al.* 2006; Harrison *et al.* 2006; Levinski *et al.* 2007; McKenney *et al.* 2007; Huntley *et al.* 2008a; Huntley *et al.* 2008b; Morin *et al.* 2008; Virkkala *et al.* 2008). The effects of climate change are expected to be more dramatic for specialist and range-restricted species; for example Huntley *et al.* (2008b) simulate these species to have little or no overlap between their present and potential future ranges. 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 differ among species and also among populations within species depending upon their biology and tolerances (Reist *et al.* 2006). Some species are projected to expand their ranges, which may affect species assemblages and, consequently, energy flow which may have unexpected results of distribution and abundance of species depending on the novel interactions that occur from the change (Reist *et al.* 2006). For instance, temperate marine species are projected to invade a warmer Arctic Ocean (Vermeiji and 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). Modelling done for sub-Saharan African plants (McClean *et al.* 2005) and breeding birds (Huntley *et al.* 2006) indicates a more complex pattern of distribution shift in response to climate change than for species in Europe due to more complex climatic patterns originating from large scale ocean-atmosphere circulations such as the *El Niño*/Southern Oscillation or the African Easterly Jet (Hulme *et al.* 2001; Nicholson 2001) and is species specific rather than general.

1.4.1.2 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 and Goulden 2008). Butterflies ranges have shifted upwards by over 200 m in 30 years in Spain, consistent with shifts in isotherms (Wilson *et al.* 2005; Wilson *et al.* 2007). Temperate mammal, South East Asian bird, and Madagascan amphibian and reptile ranges have shifted up-slope (Parmesan 2006; Peh 2007; Moritz *et al.* 2008; Raxworthy *et al.* 2008). Up-slope shifts have been modelled for tropical insects (Colwell *et al.* 2008) and, as tropical ecotherms 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 in other species (Lenoir *et al.* 2008).

1.4.1.3 Range contraction

With species distributions shifting polewards and up-slopes, the ranges of many species may contract, 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 for Scandinavian land birds for which the Arctic Ocean represents a natural barrier for northward movement (Virkkala *et al.* 2008). Jetz *et al.* (2007) have projected that 5 per cent of all land bird species will suffer range reductions of more than 50 per cent by 2050. This is particularly severe for species with limited dispersal abilities, e.g. reptiles and amphibians (Hickling *et al.* 2006), plants (Huntley 2007), species with 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 suffer significant range contractions (Xenopoulos *et al.* 2005; Xenopoulos and Lodge 2006), 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 (Parmesan 2006; Lenoir *et al.* 2008). There is some suggestion that novel biotic interactions could lead to decreased biodiversity in the future (Shuttle *et al.* 2007; Liow and Stenseth 2007) or engender more complex responses (Tylianakis *et al.* 2008) possibly dependent on co-factors such as dispersal ability (Brooker *et al.* 2007). Since the AR4, more modelling studies are endeavouring to incorporate biotic interactions into their species distribution models (e.g. Heikkinen *et al.* 2007).

1.4.2 Changes in population status

The AR4 stated that up to 30 per cent 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 (Wake 2007; Laidre *et al.* 2008; Wake and Vredenburg 2008).

Several modelling studies show that many species are likely to go extinct, e.g. 1-10 per cent of plant species in Europe will be lost by 2100, depending on climate scenarios and assumptions about migration potential of species (Bakkenes *et al.* 2006; van Vuuren *et al.* 2006); in the absence of migration, 10-50 per cent of plants are likely to disappear. The 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 and Middleton 2008).

According to Levinsky *et al.* (2007) up to 9 per cent of European mammals risk extinction, whereas 70-78 per cent may be severely threatened (losing over 30 per cent habitat) under one IPCC scenario, assuming no migration. Where unlimited migration is assumed such figures fall to 1 per cent and up to 46 per cent respectively. Some specialist species, such as the riverine rabbit, might lose most of their current habitat under projected 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 are expected at high latitudes, and 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 per cent 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 per cent 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 tree species modelled in the U.S. would lose at least 10 per cent of their suitable habitat, whereas 66 of 134 species would gain 10 per cent 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 or polar regions, are projected to be more at risk 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 may lead to yet greater species loss (van Vuuren *et al.* 2006; Jetz *et al.* 2007). 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.

1.4.2.1 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 lives in mountain streams and modelling of climate change impacts indicate a total population decline from 1612 to 146 individuals with 2.7°C temperature rise (Tseng and 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 (Araújo *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 (Araújo *et al.* 2006).

1.4.2.2 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 and Middleton 2008), and influence the metapopulation dynamics of desert bighorn sheep 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 and Middleton 2008).

1.4.2.3 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 and 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).

1.4.2.4 Competition/encroachment

The difference in response to climate change between different functional groups may potentially increase competition within ecosystems, e.g. grasslands (Cleland *et al.* 2006), which may impact on population status. 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 CO_2 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 and Totland 2007).

1.4.2.5 Pathogens, parasites and pests

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. 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 (Purse *et al.* 2005; e.g. Haines *et al.* 2006).

Pounds *et al.* (2006) found that amphibian declines have already been caused by climate change largely through increases in disease. This study is supported by a number of other studies (Alford *et al.* 2007; Fisher 2007; Laurance 2008; Muths *et al.* 2008). However, while Lips *et al.* (2008) found no evidence to support the hypothesis that the increase in amphibian disease epidemics is climate driven for Lower Central America and Andean South America, Bosch *et al.* (2007) showed significant association between change in local climatic variables and the occurrence of chytridiomycosis in Spain. The effects of climate, disease and other factors causing amphibian declines are not mutually exclusive. The largest study of global amphibian declines to date implicates climate change as a factor in amphibian decline, but stresses the importance of characteristics of the host, as well as other threats. 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).

Short-term, local experiments have demonstrated the impacts of predicted global change on plant health including a study showing that elevated atmospheric CO_2 concentration increases the risk of infection with rice blast (*Magnaporthe oryzae*) and the percentage of rice (*Oryza sativa*) plants affected by sheath blight (Jeger and Pautasso 2008); 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).

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, and (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 (Jeger and 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).

There is 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 CO_2 and global temperature that occurred 55 million years ago (Currano *et al.* 2008). Gordo and 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).

1.4.2.6 Food supply

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

Shifting species distributions, changes in numbers of individuals, or even changing environmental conditions could have a knock on effect for species relying on those species as food. For instance, changes in snowfall in the US have had an effect on the numbers of individuals and condition in a wolf, moose and fir system (Post *et al.* 1999). Changes in the abundance of prey species due to climate change can cause changes in predator numbers (Ims and Fuglei 2005; Carroll 2007). Schweiger *et al.* (2008) 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.

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

1.4.2.7 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. These phenological changes have the potential to affect species' populations directly or indirectly.

Reviews of reported climate change impacts on plant phenology support the IPCC AR4 conclusions of advanced leafing, flowering and fruiting (3-5 days per °C temperature increase) and delays in autumn events (Menzel *et al.* 2006; Cleland *et al.* 2007; Bertin 2008;

Fujimoto 2008). 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). A 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 CO_2 levels (Hovenden *et al.* 2008). Climate, particularly seasonality, is probably the primary driver of beta diversity among rainforest trees of the Western Ghats complex (Davidar *et al.* 2007).

There is ample evidence showing that the timing of reproduction of insects, birds and amphibians is influenced by spring temperatures (e.g. Gordo and Sanz 2005; Gaston *et al.* 2005; Both *et al.* 2006; Dolenec 2007; February *et al.* 2007; 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 (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). Gienapp, Hemerik and Visser (2005) created a statistical tool to predict laying dates of Great tits. However, this is one of a kind and in general it is uncertain how species will respond in terms of phenology when they reach temperature thresholds.

1.4.2.7.1 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; Mezquida *et al.* 2007; Kralj and Dolenec 2008; 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 and Ambrosini 2008) to meteorological conditions at the beginning of the breeding season. Changes in arrival times could have consequences on birds' fitness and reproductive success. Indeed, early arrival is beneficial for species fitness (as long as weather conditions are favourable) through less competition, access to more resources and opportunity to lay more clutches (Drent *et al.* 2003; Rubolini *et al.* 2005). Species arriving later or not changing could therefore face increased competition (Lemoine and Böhning-Gaese 2003) or be relinquished to sub-optimal habitat.

1.4.2.7.2 Resource availability

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 and 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 and 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 (Memmott *et al.* 2007).

1.4.2.8 Growth

There is some evidence that climate change may affect species growth which could therefore impact on populations. Growth of mature European Beech trees is lower at its southern range margin than further north and has also declined over the last 25 years (Jump, Hunt and Peñuelas 2006). Changes in precipitation and temperature have also resulted in changes in biomass of trees (Lapensis *et al.* 2005). Tropical tree growth rates are negatively correlated with mean annual temperature (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).

1.4.2.9 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 in Madagascar was over 65 per cent 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 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).

1.4.2.10 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. Under extreme regional climate change, by the mid-2080s all-male tuatara clutches would hatch at 100 per cent 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.

1.4.2.11 Dispersal

There is evidence that climate can directly affect the dispersal ability of species, thus facilitating or hindering range shifts and ultimately contributing to population status. Several species have increased their dispersal potential through phenotypic and evolutionary processes which are linked to climate (Thomas *et al.* 2001; Møller *et al.* 2006). Changes in dispersal may also be age dependent. Duckworth (2008) reports that older populations tend to be less dispersive than new ones. 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). Indeed there are similarities between meta-population dynamics and dispersal dynamics which suggest that 'colonising' species are more likely, to expand their ranges in response to climate change than less dispersive species or be faster at doing so (Duckworth 2008).

1.4.3 Characteristics and factors contributing to vulnerability or resilience

The IPCC AR4 estimated that 20-30 per cent 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. Preliminary analyses on the susceptibility of species to climate change according to their biological traits, suggest that for birds, amphibians and warm water corals as many as 35-70 per cent may be susceptible to 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 (Walther *et al.* 2005; Pompe *et al.* 2008; Sodhi *et al.* 2008), as are those with naturally fragmented or isolated populations; tropical montane species are also at particular risk (Wake and 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 and 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 and Innes 2006). Limited dispersal ability is also a key risk factor (Foden *et al.* 2008).

1.5 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 intra-specific responses to elevated CO_2 affect not only physiology and growth, but also higher order biotic interactions and lifetime fitness, ultimately leading to new ecosystem assemblages (Bradley and 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 per cent of the wild relatives of peanut, potato and cowpea may be threatened with extinction under climate change, most will lose over 50 per cent of their range size, and the distributions of many will become highly fragmented (Jarvis *et al.* 2008).

1.6 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, predator-prey interactions 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 and Totland 2007). Experimental work has also supported the potential role of CO_2 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 inter-specific 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 modelling studies project increases in economically important plant pathogens with warming, and experimental studies show similar patterns (Jeger and 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 per cent 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 and Sanz 2005).

There have been suggestions that climate change may affect predator-prey interactions. Changes relate from either direct effect of climate change on prey/predator numbers which then has knock-on effects on the other species (Ims and Fugelei 2005; Carroll 2007) or through conditions making prey more/less vulnerable to predators (Post *et al.* 1999; Schmitz *et al.* 2003).

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 CO_2 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 and 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 *et al.* 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 both species and ecosystems.

1.7 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 per cent 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 AR4.

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 CO_2 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 per cent 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.* 2008b; 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.* 2008a). 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 (Fenner *et al.* 2007; Breeuwer 2008; Garant *et al.* 2008; Heijmans *et al.* 2008). 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 150,000km^2 of low-lying peatlands may cause substantial emissions (Henman and Poulter 2008).

One area of research that has expanded since the AR4 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 *et al.* 2008). Indeed, it has been suggested that CO_2 emissions will be accelerated by up to 66 per cent due to feedbacks arising from global soil carbon loss and forest dieback in Amazonia as a consequence of climate change (Betts 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 *et al.* 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 its 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 per cent 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 0.38ºC or 8 per cent increase in warming compared to a model when feedback was not considered (Betts *et al.* 2008). Such modelling 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 and Galloway 2008); although it has been estimated that increased carbon sequestration may lead to an increase of N_2O emissions in grassland (Kammann *et al.* 2008).

Recent evidence supports the findings reported in the AR4 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).

1.8 METHODOLOGIES AND TOOLS

Many of the studies reviewed here rely on experimental or modelled evidence of climate change impacts. Both of these simulate the biodiversity impacts of climate change in more or less realistic ways.

Experimental studies are limited by the number of factors that can be manipulated simultaneously. Single-factor experiments (e.g. increasing CO_2 concentrations) represent least realistic simulations of future climate change, whereas multi-factorial experiments may capture interactions more realistically (e.g. manipulating CO_2 concentrations, temperature and nutrients simultaneously).

The limitations of modelling studies have been reviewed previously (Guisan and Thuiller 2005; IPCC 2007; Thuiller *et al.* 2007). Most modelling studies reviewed here are correlative, i.e. they derive functions or algorithms that relate the occurance of species and ecosystems to current climatic or other factors. These functions or algorithms are then used to project distributions under future climates, assuming that the observed correlations will hold in the future. The current distribution may not reflect the full fundamental climatic niche of a species, which may not be fully expressed. Where range limits are underestimates (likely for most field based studies) then the extinction risk will be over-estimated. Bioclimatic envelope studies generally assume no adaptation will occur due to the speed of climate change. Various studies have compared the available statistical modelling techniques, as well as the different ways of assessing the goodness of fit of such models (Jeschke and Strayer 2008).

Outputs from correlative studies are dependent on the choice of explanatory variables considered, with most studies only considering climatic variables (Dormann 2007), though some have incorporated land-use change into the models (Bomhard *et al.* 2005; Jetz *et al.* 2007). Further, the climatic variables included represent the mean over many years or decade, ignoring extreme events that are likely be more important in limiting distributions than average values. Downscaling of climate data is particularly poor for precipitation and extreme events. Similarly, most models only consider global climate drivers, ignoring local drivers, such as land-use change, urban effects or fire, as well as ignoring interactions among different impacts (Betts 2007; Midgley *et al.* 2007). Feedbacks from impacts to climate change, which often involve land ecosystem-atmosphere interactions, are often neglected. This can result in representations of global changes that are at best inconsistent and at worst completely misleading (Betts 2007).

Few modelling studies deal explicitly with interactions and dynamics among species, such as migration, dispersal and competition (Levinsky *et al.* 2007; Jeschke and Strayer 2008; Thuiller *et al.* 2008). Inclusion of interspecific interactions and dispersal ability in models can highlight the subtle balances of processes, non-linear dynamics and abrupt changes from species coexistence under climate change (Brooker *et al.* 2007). Further, feedbacks between several factors, such as climate change and land-use change impacts should be considered, to predict climate change impacts more accurately (Malhi *et al.* 2008).

Recently, the bioclimatic envelope approach has been questioned and Beale *et al.* (2008) argue that the species-climate associations from bioclimatic envelope methods are no better than chance for some European bird species investigated, suggesting that bird species are not sensitive to climate variables currently available but may be more affected by land cover or biotic interactions. However, other studies have validated climate envelope models by showing that historic population trends are driven by climate variables (Green *et al.* 2008). To achieve better predictions of impacts on species, possibly more complex models are needed such as those combining both niche model and individual-based approaches (Chamaille-Jammes *et al.* 2006; Keith *et al.* 2008). However, most importantly, the hypotheses generated by models need to be tested and validated, if possible using spatially or temporally independent data sets (Hijmans and Graham 2006; Grosbois *et al.* 2008). Currently validation of models is inadequate in most studies as lack of independent data sets endangers use of data partitioning methods. However, gathering such data sets for validation purposes would require long-term strategic observations and monitoring (Midgley *et al.* 2007), especially in areas where data are sparse, e.g. tropics (IPCC 2007). Lastly, research is needed and is currently ongoing to find direct evidence of how species respond to environmental variables (Gordo 2007).

1.9 CONCLUSION

The main lesson from recent research on the impacts of climate change on biodiversity is that many of the key findings at the time of the 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.

This review found compelling evidence from current trends that ecosystems are starting to respond in terms of their distribution, composition, structure and function to the changes in temperature, precipitation and increased CO_2 levels that are occurring. Experimental studies as well as modelling studies have given an indication of what changes may occur (or continue to occur) in the future. Together these studies indicate that some ecosystems may shift poleward or upwards in mountainous regions. Indeed there are already indications of uphill migration of some treelines. Changes in species composition and richness have been documented and are likely to become more widespread. Novel interactions between species may form or invasive species may become established. Such changes in species composition could lead to changes in the physical and trophic structure of ecosystems, with resulting effects on system function and composition. Furthermore, recent findings suggest that the impacts of climate change on ecosystems are likely to be amplified by positive feedbacks.

Nevertheless, more research is needed in certain areas. Experimental studies are extremely useful in determining the effect of climate change on aspects of ecosystem composition, structure and function. However, the time needed to evaluate responses in such experiments, as well as expense, limits the number of studies in this field. Furthermore, changes in greenhouse gas levels are occurring over time frames that are difficult to simulate in experimental studies. Dynamic vegetation modelling has helped to improve our understanding of potential effects of climate change on ecosystems though there is still progress to be made in these models (Prentice *et al.* 2007). More research is needed on desert ecosystems as there has been relatively little work since the AR4 on these ecosystems. Work on the impact of climate change on freshwater ecosystems is starting to emerge. For instance, linkages between climate impact on aquatic systems and invasive species are only starting to be analysed (Rahel and Olden 2008).

At the species level, there are numerous correlative modelling studies simulating the potential impact of climate change on their distribution. For the most part, evidence from observations corroborates the projected poleward expansion of many species. Changes at the trailing edge of species distributions are less well documented though some recent examples reviewed have indicated contraction at the southern margin of many northern hemisphere species. However, detailed effects of changes at the leading and trailing edge of species distributions is an area that needs to be further researched (Thuiller *et al.* 2008). To ensure more realistic projections of the impact of climate change on populations, metapopulation dynamics, species' dispersal ability and demography would need to be incorporated into classic static species distribution models. Scientific research is moving forward in this area with the use of combination or framework models (del Barrio *et al.* 2006; Keith *et al.* 2008; McRae *et al*. 2008) and mechanistic models (Kearny *et al.* 2008). Such models may also help improve our understanding of the effect of climate change on species abundance (Green *et al.* 2008; McRae *et al*. 2008). Land use is rarely incorporated in these modelling studies, yet is likely to impact species' responses to climate change and/or interact with climate change.

Phenological changes are already being observed and mismatches in phenological responses to climate change between species (e.g. plant-pollinator relationships) may significantly affect their interactions and potentially lead to their extinction. Climate change is also likely to affect ecological interactions, including competition, disease and host-parasite interactions, pollination, herbivory and predator-prey cycles. 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. However, there are fewer studies examining effects on predator-prey cycles or other ecological interactions. Species are likely to respond individually to climate change (Huntley *et al.* 2007), as they have in the past (Willis *et al.* 2007), and therefore novel interactions may occur with unknown effects.

Despite its importance, relatively little effort has yet been devoted to investigating the impacts of climate change on genetic diversity. 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.

The literature reviewed indicates that climate change is already impacting biodiversity from the ecosystem to the genetic level and feedbacks from ecosystem changes to the climate system could have serious implications, though there is some uncertainty in this area, such as turning current carbon sinks into sources. This suggests that adaptation and mitigation strategies will be increasingly important in the future.

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SECTION 2

THE LINKAGES BETWEEN BIODIVERSITY AND CLIMATE CHANGE ADAPTATION

A review of the recent scientific literature

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2.1 EXECUTIVE SUMMARY

The impacts of climate change are already being felt, and will continue to increase in magnitude. Countries are now starting to develop and implement adaptation policies to cope with these impacts. Adaptation strategies tend to focus on technological, structural, social, and economic developments, and the linkages between biodiversity and adaptation are often overlooked. Nevertheless, biodiversity is linked to climate change adaptation in three main ways; biodiversity can play a role in societal adaptation, biodiversity can be impacted by societal adaptation strategies, and biodiversity conservation is a sector that requires adaptation strategies in its own right.

Scientific literature on the **role of biodiversity in climate change adaptation** is scarce, but there is a growing body of evidence suggesting that ecosystem-based adaptation can be a cost-effective adaptation strategy across the major adaptation sectors. Adaptation strategies that aim to enhance the resilience of ecosystems to enable the continued provision of goods and services can be particularly important for poor people, who are often directly dependent upon their natural resources and have little access to technical measures.

Coastal adaptation: Coastal defences have traditionally relied upon 'hard defence' structures such as sea walls. However, evidence suggests that resilient coastal ecosystems, including mangroves, coral reefs, sand dunes and salt marsh can play an effective role in coastal protection. In addition, coastal ecosystems provide resources such as fish, and allow more flexibility to adapt to uncertain changes. They can also act as a buffer against extreme events. However, coastal ecosystems will not reduce impacts in all cases. Integration of 'hard defence' measures with proper land-use planning and ecosystem management is increasingly being promoted.

Adaptation in the water sector: Natural freshwater systems provide vital water regulation services, and can play a role in adaptation to water scarcity, as well as flooding. Actions to reduce degradation of watersheds, through reduced deforestation, afforestation, and soil conservation can lower vulnerability to drought; and the maintenance and restoration of the water regulating services of wetlands are important for flood control. As with coastal defence, the need for integration of improved watershed management with technological measures is receiving increasing attention, though not yet at the policy level.

Adaptation in agriculture: Diverse agricultural systems, incorporating new varieties of crops and crop diversification, are likely to be essential in maintaining food production under changing temperature and water conditions. Such agricultural systems are clearly dependent upon a range of crops, for which the maintenance of agro-biodiversity is critical. 'Good practice' natural resource management, including water and soil conservation, is also likely to play a major role in agricultural adaptation, particularly in drylands. Agroforestry, intercropping food crops with tree stands, has been identified as a promising option to improve resilience of agricultural systems to climate change.

Forest adaptation: Discussion of forests in relation to climate change tends to focus on their role in mitigation. However, forests provide a range of regulating services whilst providing important resources to those who depend on forests for their livelihoods, and can be particularly important during extreme events. Maintaining intact natural forests and selecting appropriate mixes of species for afforestation is likely to enhance their resilience to climate change, supporting their contribution to both mitigation and societal adaptation.

Adaptation in the urban environment: The incorporation of more green spaces, including the planting of trees, can play a role in urban adaptation by reducing heat stress and improving drainage during times of flood. Despite this, biodiversity is often overlooked in urban design and adaptation plans.

Health: Although the importance of biodiversity for health is recognised, few links have yet been made to the role of biodiversity in adaptation to the health impacts of climate change. This is an area for further research.

The contribution that biodiversity can make to societal adaptation will differ according to the circumstances, and in many cases technological solutions will be required. Analysis of the costs and benefits of adaptation options is uneven, and further research is required in this area. However, available evidence suggests that integrated management strategies, incorporating ecosystem management into broader cross-sectoral adaptation policies as a complement to structural and technological measures, are likely to result in more sustainable adaptation. This will require significant institutional support, which currently appears to be lacking.

The **impact of adaptation strategies on biodiversity** has been shown to be negative in many circumstances, particularly in the case of 'hard defences' constructed to prevent coastal and inland flooding. This can result in maladaptation in the long term if it removes natural flood regulation properties of coastal and freshwater ecosystems, for example. Conversely, adaptation strategies that incorporate natural resource management, such as improved agricultural practice, can be beneficial for biodiversity. The information available in this area is limited, as few adaptation strategies have been implemented.

There is an urgent need for **adaptation in the biodiversity conservation sector**, as the impacts of climate change on natural ecosystems are already being observed and are likely to increase in magnitude. This is required not just to achieve the conservation of biodiversity for its own sake, but to maintain the role of biodiversity in contributing to societal adaptation. The conservation sector is only recently beginning to develop adaptation measures, but strategies such as improved protected area design, maintaining habitat connectivity in the wider landscape, and reducing other anthropogenic pressures are likely to increase the resilience of biodiversity to climate change. Increasing the resilience of ecosystems to climate change also supports their role in climate change mitigation.

Ultimately, a broad perspective is required, focusing on how ecosystems can be managed and conserved in order to deliver ecosystem services in a changing climate, within the context of overall adaptation policy. There needs to be greater consideration of synergies and trade-offs in adaptation policy and planning, including improved understanding of the underpinning role of biodiversity, to avoid maladaptation and develop cost-effective responses to the impacts of climate change.

2.2 INTRODUCTION

2.2.1 Adaptation

The impacts of climate change are already being felt, and will continue to increase in magnitude. They include rising sea levels, increased drought and flooding, and impacts on agriculture. Until recently, efforts have been focused on the development of appropriate mitigation measures to reduce the scale of these impacts. However, the need to develop adaptation strategies to cope with the impacts to which we are already committed, or to which we are likely to be committed in the future, is becoming increasingly recognised (Goklany 2007; Pielke *et al.* 2007; Stern 2007).

According to the Intergovernmental Panel on Climate Change Fourth Assessment Report (IPCC AR4), adaptation can be defined as the '*adjustment in natural or human systems in response to actual or expected climatic stimuli or their effects, which moderates harm or exploits beneficial opportunities'* (IPCC 2007). Adaptation strategies aim to reduce the vulnerability or enhance resilience in response to these 'actual or expected changes' and associated extreme events, and will be required in both human and ecological systems (Adger *et al.* 2007). Currently, adaptive capacity is uneven both across sectors and within societies (Adger *et al.* 2007). The most vulnerable to the impacts of climate change are likely to be those in Least Developed Countries (LDCs), and Small Island Developing States (SIDS).

Adaptation is receiving increasing attention under the United Nations Framework Convention on Climate Change (UNFCCC). The Nairobi Work Programme on impacts, vulnerability and adaptation to climate change was established under the Subsidiary Body for Scientific and Technological Advice (SBSTA) in 2005. The five-year programme has the aim of assisting all Parties to the convention, especially developing countries, LDCs and SIDS, on matters regarding improvement of understanding and assessment of impacts, vulnerabilities and adaptation; and to make informed decision on practical adaptation actions and measures (UNFCCC 2008). Adaptation was also identified as one of the five key building blocks for a strengthened future response to climate change in the Bali Action Plan. Many LDCs have developed National Adaptation Plans of Action (NAPAs), which identify priority adaptation projects required to cope with the immediate impacts of climate change.

Although there are now a number of funds for adaptation, they are widely considered to be inadequate. Estimates of the sums needed to fund adaptation range from US$10-86 billion per year. These estimates are orders of magnitude higher than the sums generated under the existing funds (Ayers and Huq 2008; Harmeling and Bals 2008).

2.2.2 Biodiversity and adaptation

There is some recognition of the importance of ecosystems to adaptation in the text of the UNFCCC. Article 2 states that the ultimate objective of the convention is to stabilize greenhouse gases '*at a level that would prevent dangerous anthropogenic interference in the climate system'*. It then asserts that '*Such a level should be achieved within a time-frame sufficient to allow ecosystems to adapt naturally to climate change*'.

Moreover, Article 4 includes a commitment from all Parties that they shall: '*Cooperate in preparing for adaptation to the impacts of climate change; develop and elaborate appropriate and integrated plans for coastal zone management, water resources and agriculture, and for the protection and rehabilitation of areas, particularly in Africa, affected by drought and desertification, as well as floods.*' Consideration of the underlying ecosystems is crucial to successful adaptation in all of these sectors. More specifically, biodiversity is intimately connected to climate change adaptation in at least three ways:

1. Components of biodiversity can play a significant role in strategies for societal adaptation to climate change, and are particularly important for reducing the vulnerability of the poor and disadvantaged. This review will consider the role of biodiversity in the coastal, water resource, agricultural, forest, urban, and health adaptation sectors, including adaptation to extreme events.

2. Many of the strategies adopted for societal adaptation, especially those dependent on engineering and technology, can have significant negative impacts on biodiversity, and these will differ between sectors.

3. The components of biodiversity are themselves subject to considerable impacts from climate change, as established by Kapos *et al.* (2008) in the background documents for the first meeting of the Second AHTEG on Biodiversity and Climate Change. There is, therefore, a need for adaptation strategies within the conservation sector, both to conserve biodiversity for its own sake, and to maintain the role of biodiversity in societal adaptation.

This report reviews the literature published since the IPCC AR4 on the linkages between biodiversity and climate adaptation, focusing on these three topics in turn. This structure reflects the divisions in the literature on biodiversity and adaptation, and provides a useful way of organizing this literature review. Nevertheless, there is a risk that it can obscure some of the underlying connections between the three topics. This point will be taken up in the conclusion. Keyword searches in ISI Web of Knowledge and Google Scholar were carried out to obtain a broad coverage of the available literature. As the peer-reviewed literature in this area is limited, grey literature was also used.

2.3 THE ROLE OF BIODIVERSITY IN SOCIETAL ADAPTATION TO CLIMATE CHANGE

Strategies for societal adaptation to climate change are generally based on engineering structures, technological developments, and economic diversification. However, the evidence that adaptation strategies based on natural resources can play an important and cost effective role as part of integrated adaptation strategies is growing (Abramovitz *et al.* 2006; ProAct Network 2008). This evidence is grounded in the known links between ecosystems and human livelihoods (Abramovitz *et al.* 2006).

Ecosystems provide a number of services that play a significant role in maintaining human well-being. These include provisioning services, such as food, fuel and fibre, regulating services, such as carbon storage and water regulation, supporting services, and cultural services (MA 2005a). A recent study has estimated that welfare losses due to the loss of ecosystem services could be equivalent to 7 per cent of annual consumption by 2050 (European Commission 2008). Although the detailed linkages between biodiversity and ecosystem services are not always well understood, it is widely recognised that maintaining biodiversity promotes the continued provision of services under environmental change (Worm *et al.* 2006; Carpenter, Bennett and Peterson 2006; World Bank 2008; Palumbi *et al.* 2009).

The poor are often the most directly dependent on ecosystem services. It has been estimated that three quarters of the world's poorest people (those living on less than US$2 per day) depend on the environment for a significant part of their livelihoods (WRI 2008). In Africa, for example, more than 70 per cent of the population earn their living from agriculture, and most of the remaining population depend on exploitation of other natural resources through hunting, fishing, and use of forest products (Enow and Muhongo 2007).

It is for this reason that adaptation strategies that enhance the resilience of ecosystems, ensuring the continued provision of goods and services, can be particularly important for poor people (Adger, Arnell and Tompkins 2005a; Reid and Huq 2005; Thomas and Twyman 2005; AIACC 2007; Ravindranath 2007). Poor people with low adaptive capacity are vulnerable to the impacts of climate change, which will contribute to the loss of their natural resource base (Eriksen *et al.* 2007). Ecosystems, particularly those that have already been degraded, are likely to be severely impacted by climate change (Fischlin *et al.* 2007). A recent study has projected that annual losses to the Namibian economy due to the impacts of climate change on natural resources alone could be up to 5 per cent of GDP, and that this will affect the poorest members of society (Reid *et al.* 2008). It has been suggested that environmental degradation is lowering the resilience of people to climate change in the Niger Delta (Uyigue and Agho 2007), and in developing countries globally (Huq and Ayers 2007). Thus, the need to build resilience of ecosystems to maintain their productivity is often stressed in the development literature as a necessary part of adaptation strategies, particularly for vulnerable communities (Corfee-Morlot *et al.* 2003; Tompkins and Adger 2004a; Tompkins and Adger 2004b; Nkem *et al.* 2007; Reid *et al.* 2008; WRI 2008).

Similarly, in Small Island Developing States (SIDS), many people depend upon biodiversity resources that are already under stress (CICERO and UNEP/GRID-Arendal 2008). Adaptation strategies that involve the sustainable management and use of resources are likely to enable SIDS to become more resilient to climate change (Cherian 2007).

In addition, natural resource management strategies are more accessible to local communities than strategies based on infrastructure and engineering (Reid and Huq 2005; Hedger and Cacouris 2008), and community-based adaptation projects often involve the management of natural resources (Huq *et al.* 2005). Rehabilitating natural resources such as farm and grazing lands, forest, watersheds, and fisheries have become a central focus on a project-level scale across Asia and Africa (AIACC 2007).

Biodiversity is also included in many National Adaptation Plans of Action (NAPAs), which identify priority adaptation requirements in Least Developed Countries (LDCs), as these requirements are often linked to natural resource management (Shaw 2006). An analysis of the 30 NAPAs available in 2008 showed that 25 Parties identified adaptation projects related to biodiversity, eight of which were SIDS (Webbe 2008).

Although reflected on a project basis and in some NAPAs, the role of biodiversity in adaptation has received little attention at the scale of national and international adaptation policy (Nkem *et al.* 2007; Kalame *et al.* 2009). A small number of countries do identify natural resource management related actions in their adaptation plans (Webbe 2008), but it generally tends to be overlooked, particularly in developing countries. It has been argued that it is possible to incorporate biodiversity conservation into adaptation and mitigation planning to obtain 'win-win-win' strategies (Paterson *et al.* 2008). The remainder of Section 2 outlines the contribution that biodiversity can make to societal adaptation across the various adaptation sectors.

2.3.1 Coastal adaptation

2.3.1.1.Coastal defence

Adaptation in the coastal sector has received the most attention in the literature to date. This is largely due to the fact that coastal societies and ecosystems are particularly vulnerable to climate change, and impacts are already being felt. Even a one metre rise in sea level, the lowest expected this century, could displace nearly six million people across South Asia and 37 million people along the river deltas of East Asia (Dasgupta *et al.* 2007).

Protection strategies for sea level rise range from ‘hard’ defences, such as sea walls, dykes, and tidal barriers to ‘soft’ defences such as natural resource management (Adger *et al.* 2007). In most developed countries, ‘hard’ defences are preferred, particularly in built-up areas (Kirshen, Knee and Ruth 2008). These defences have often been built with little regard for the integrated nature of the coastal ecosystem, and can require costly repairs and upgrades (Duxbury and Dickinson 2007). The cost of infrastructure to prevent against storm surges and floods in the UK alone has been estimated at US$18-56 million annually (Mani 2007). More recent strategies include ‘managed realignment’ or ‘coastal retreat’, whereby infrastructure is moved inland to reduce the risk of impacts and allow the development of inter-tidal ecosystems, or ‘accommodation’ where planning restrictions prevent the development of infrastructure on floodplains or at certain distances from the shore (ProAct Network 2008; Glick, Staudt and Stein 2009). Managed realignment of hard protection structures, due to increased erosion rates and high costs of maintenance, is being trialled in the UK (ProAct Network 2008). Options for adaptation in the coastal zone are shown in Table 1.

TABLE 1: MAJOR ADAPTATION STRATEGIES FOR THE COASTAL ZONE (UNFCCC 2006)

Protection	Retreat	Accommodation
Hard structures: dykes, sea walls, tidal barriers	Establishing set-back zones	Early warning systems, hazard insurance
Soft structures: dunes or wetland restoration, beach nourishment	Relocating threatened buildings and hard protection structures	Land-use planning (building and agricultural practice)
Indigenous options: afforestation	Phasing out development in exposed areas	Improved drainage and desalination

Biodiversity can play a role in a number of coastal defence strategies. Soft engineering solutions incorporate activities such as planting of marsh vegetation in the intertidal zone and wetland restoration (Morris 2007). Coastal wetlands can absorb wave energy and reduce erosion through increased drag on water motion, a reduction in the direct wind effect, and directly absorbing wave energy (Day, Jr. *et al.* 2007). The accretion of sediments also maintain shallow depths that decrease wave strength (Koch *et al.* 2009).

Biodiversity based adaptation measures are receiving increasing attention in developing countries, particularly SIDS, where adaptive capacity is low and local communities depend upon their natural resource base (Cherian 2007). Mangroves, for example, can provide physical protection to coastal communities whilst providing provisioning goods and services such as productive fisheries; offering both physical protection and economic gain to the most vulnerable people (Adger *et al.* 2005a; McKinnon and Webber 2005; Reid and Huq 2005). It has been estimated that the value of mangroves for coastal defence in Malaysia is US$300,000 per km based on the cost of hard engineering that would otherwise be required (ProAct Network 2008). Nearly 12,000 hectares of mangroves planted in Viet Nam at a cost of US$1.1 million, saved an estimated US$7.3 million per year in dyke maintenance whilst providing protection against a typhoon that devastated neighbouring areas (Reid and Huq 2005).

Recent research has suggested that natural systems can actually be more effective at protecting coasts from erosion and flooding than hard defence structures (Costanza *et al.* 2008; Hanak and Moreno 2008), although this will not be the case in all situations, and modelling of societal responses to sea level rise in different areas rarely produces the same optimal response (Tol *et al.* 2006). Risk-based analyses have shown that generally it is advantageous to use expensive structural protection in highly developed areas, and ‘softer’ approaches such as land management in less developed areas (Hulme 2005; Kirshen *et al.* 2008). In addition to being more cost-effective, strategies focused on resource management tend to provide co-benefits such as biodiversity conservation, as well as in some cases contributing to mitigation through carbon sequestration (Trulio *et al.* 2007), and allow for more flexibility to adapt to uncertain changes in the future (Costanza *et al.* 2008; Kirshen *et al.* 2008; Luisetti, Turner and Bateman 2008; Koch *et al.* 2009).

Although coastal vegetation has significant potential in climate change adaptation, it requires a holistic management approach, with full participation from local authorities and communities (Tanaka 2009). Currently, ecosystem management initiatives for coastal protection tend to lack a scientific basis (Mascarenhas and Jayakumar 2008). Different species of mangrove, marsh plants, and seagrass have different wave attenuation capacities (Koch *et al.* 2009). In order to provide optimal coastal protection, mangrove belts need to be maintained at a certain width and thickness and planted vegetation needs to be given time to mature (ProAct Network 2008). The conditions in which coastal vegetation will offer protection are also not entirely known. There will be areas in which dunes play a better protective role, and others in which mangroves are more suitable (Danielsen *et al.* 2005). For example, conversion of coastal sand dunes to protective plantations might result in maladaptation as sand dunes can provide better protection (Bhalla 2007).

Integrated management of coastal ecosystems is required because of the interconnectivity of coastal systems. For example, mangrove protection against hurricane damage extends to increasing resilience of coral reefs (Grimsditch 2006; Gilman *et al.* 2008; Mumby and Hastings 2008; ProAct Network 2008). Waves approaching a coastal area travel across reefs and through seagrass beds before reaching mangroves, and the wave attenuation is not provided by one ecosystem alone (Koch *et al.* 2009). When planting vegetation for coastal defence, it is important to include species with tolerance for flooding and broad ranges within the intertidal zone (Morris 2007).

It is also important to reduce coastal ecosystem degradation. Many services provided by coastal and marine ecosystems are in decline (Leslie and Mcleod 2007). Although climate change could result in a 10-15 per cent loss of mangrove, the current rate of deforestation far exceeds this threat (Alongi 2008). This can reduce the resilience of coastal vegetation to climate change, and remove their capacity to act as a physical barrier (Gilman *et al.* 2008; ProAct Network 2008; Tornqvist and Meffert 2008), increasing vulnerability of coastal communities to extreme events (Danielsen *et al.* 2005). Sand extraction of dunes for construction increases vulnerability to storm surges (Sudmeier-Rieux 2006), and anthropogenic threats to reefs reduces their protective and fish provisioning services (Adger *et al.* 2005b; Kunkel, Hallberg and Oppenheimer 2006; Meadows and Brosnan 2008). For example, water flow may actually be accelerated through channels of fragmented reefs (Cochard *et al.* 2008). Environmental degradation can also reduce the potential for economic recovery due to loss of traditional livelihoods (Adger *et al.* 2005b). However, most communities have little experience of managing for resilience (Gibbs 2009), and the concept itself is not fully understood (Gilman *et al.* 2008). Capacity building in this area is likely to be required.

2.3.1.1.1 Integrated Coastal Zone Management

It is becoming increasingly recognised that integrated management of the entire coastal zone is required. Integrated Coastal Zone Management (ICZM) is being promoted due to the recognition that a combination of sustainable protective measures is required for the coastal zone (Duxbury and Dickinson 2007). Where hard structures are built without consideration for the impacts on buffering coastal ecosystems, they can actually reduce the adaptation potential of the coast, a process known as 'maladaptation' (Glick *et al.* 2009). Mangroves, for example, respond to sea level rise and coastal erosion by retreating inland (Alongi 2008), but may be significantly impacted where there is reduced area to move landward (Jagtap and Nagle 2007; Alongi 2008; Gilman *et al.* 2008). Land-use planning is therefore necessary to avoid this 'coastal squeeze' (Gilman *et al.* 2008). In Louisiana, the drainage of wetlands and starvation of natural sediments from the construction of canals and levees contributed to the land subsidence that lowered some areas below sea level (Glick *et al.* 2009).

Economic studies have suggested that considering integrated adaptation strategies can be beneficial, both for ecosystems and society (Costanza *et al.* 2008; Sugiyama, Nicholls and Vafeidis 2008). The management of coastal ecosystems can be combined with hard defence strategies and land-use planning (Jenkin 2005). For example, salt marshes can protect landward sea defences whilst providing a habitat for rare plants and migratory birds (Hulme 2005; Luisetti *et al.* 2008). This has been recognised in some developed countries. In the Netherlands, for example, flood prevention policy is shifting from dykes to realignment and ecosystem restoration, due to the difficulties of continuous dyke maintenance (Pasche *et al.* 2008; ProAct Network 2008). However, although similar ICZM activities are also being explored in the UK (de la Vega-Leinert and Nicholls 2008), coastal governance and the need to involve a variety of stakeholders means that progress is slow (Mcfadden 2008; Milligan *et al.* 2009). There are also trade-offs to be made. Managed retreat often only occurs on low value land and can be costly and difficult to implement, whereas accommodation through creation of new floodplain habitats is subject to the choice that this land can be lost to the sea (Hulme 2005; Richards *et al.* 2008).

Much of the literature surrounding the role of ecosystems in coastal protection is focused on reducing extreme event impacts, which will be discussed in section 2.3.1.3.

2.3.1.2 Fisheries

In addition to coastal protection, mangroves, coral reefs, and other coastal ecosystems play an important role in fisheries (FAO 2007; Glick *et al.* 2009). Recent studies in the Gulf of Mexico have estimated that mangrove fish and crab species account for 32 per cent of small-scale fisheries landings, and that coastal ecosystems contribute an estimated 77 per cent of the global ecosystem-services value calculated (Martinez *et al.* 2007). The resources provided by coral reefs are particularly important for SIDS (Walling and Creary-Chevannes 2004). In a study in the Philippines, 90 per cent of all fishers recognised the role of mangroves as a nursery site, in addition to their role in storm protection (Walton *et al.* 2006).

The vulnerability of fisheries to climate change and the implications for adaptation have not yet been considered on a large scale, but recent evidence suggests that impacts could be significant (Brander 2007; Coulthard 2008; Allison *et al.* 2009). The communities likely to be impacted most heavily include a number of less developed countries, where the most vulnerable groups rely on fisheries for 27 per cent of their protein (Allison *et al.* 2009). In Bangladesh, a community wetland management programme, which protected wetlands from degradation, has improved fish catches by an estimated 140 per cent and improved resilience of both the wetland and the community to environmental change. The success of this project led the government to include it in the fisheries strategy to reduce the siltation caused by forest clearance, wetland drainage and flood embankments (WRI 2008).

2.3.1.3 Reducing extreme event impacts

There is a growing wealth of literature linking disaster risk reduction (DRR) strategies with climate change adaptation. Although there is much uncertainty attached to the role of climate change in increased severity and incidence of extreme events, an increase in disasters such as flooding and hurricanes is predicted (Beniston *et al.* 2007; Francisco 2008). These disasters are likely to impact vulnerable areas such as SIDS and LDCs, particularly in Asia. It has been suggested that management of natural resources can contribute to DDR by reducing vulnerability to the event, and increasing adaptive capacity after the event (Sudmeier-Rieux 2006; Francisco 2008). The role of coastal ecosystems has received particular attention in this respect.

A number of studies carried out following coastal disasters such as tsunamis and hurricanes have documented an important role for wetlands, mangroves and coral reefs in coastal protection against extreme events and tropical storms (Danielsen *et al.* 2005; UNEP-WCMC 2006; Granek and Ruttenberg 2007; Olwig *et al.* 2007; Perez-Maqueo, Intralawan and Martinez 2007; IUCN 2008; Francisco 2008; Mattsson *et al.* 2009). Although tsunamis are not related to climate change, they provide an evidence base for the protective role against storm surges in general. Coastal ecosystems can provide a buffer against the wave impacts and also decrease the strength of the waves. Forest canopies in wetlands can diminish wind flow and reduce surface waves, whilst shallow water vegetation can limit wave build-up (Day, Jr. *et al.* 2007).

A Rapid Environmental Assessment by IUCN following the tsunami of 2004 found a clear correlation between damage of inland areas and human modifications to the coastline, with mature sand dunes especially effective in protection (Bambaradeniya *et al.* 2005). During Hurricane Katrina, levees fronted by extensive wetlands escaped substantial damage, suggesting that a well managed combination of hard and soft protection can play a role in climate change adaptation (Day, Jr. *et al.* 2007), and that the re-establishment of protective habitats could be important even for built up areas (Glick *et al.* 2009). Hydrological models and simulations have suggested that a 100m wide mangrove forest belt can reduce wave flow by 90 per cent (Alongi 2008), and that coral reefs offer protection against tsunamis (Kunkel *et al.* 2006) to add weight to observational reports. It has been estimated that coastal wetlands in the U.S. alone provide US$23.2 billion per year in hurricane protection (Costanza *et al.* 2008), and that the coastal protection value of mangroves exceeds their direct use value by over 97 per cent (Sanford 2009).

Despite the wide range of anecdotal reports and modelling exercises, there remains little empirical evidence on how much protection coastal ecosystems can provide against extreme events (Granek and Ruttenberg 2007; Feagin 2008), leading some to question the validity of diverting adaptation funds to coastal ecosystem management (Cochard *et al.* 2008). Reports that areas with mangrove and tree shelterbelts were significantly less damaged than other areas have been questioned due to the large number of caveats inherent in the studies (Dahdouh-Guebas and Koedam 2006), whereas other studies have found no such role for coastal ecosystems (Kerr *et al.* 2006).

Indeed, although a number of studies support the role of coastal ecosystems in coastal storm protection, they note that it has limitations (Kerr and Baird 2007). Furthermore, there is still a poor understanding of how much coastal ecosystems can attenuate extreme waves and hence provide protection (Barbier *et al.* 2008). The presence of sand dunes, mangroves, and coral reefs made little impact in the epicentre of the 2004 tsunami, although they reduced the power of the smaller waves in Sri Lanka (Adger *et al.* 2005b). Few studies take into account the variability between energy and speed of waves (Cochard *et al.* 2008). Ecosystem services are not linear across space and time. Wave attenuation may be higher in summer where biomass is highest for example, or when the tide is low, and it cannot be assumed that vegetation will automatically provide coastal protection (Koch *et al.* 2009). Protection by vegetation such as mangroves depends upon the stand size, density, species composition, and structure, and degraded ecosystems are less likely to function as buffers (Cochard *et al.* 2008; Koch *et al.* 2009; Tanaka 2009; Alongi 2008). A recent study using satellite imagery and field measurements found that survival rate of mangroves during extreme events increased with increasing stem diameter, but that the mangrove belt was mostly destroyed following inundation at depths greater than 6m (Yanagisawa *et al.* 2009).

It is important to recognise that ecosystems alone cannot reduce the impacts of storms, and that a balance of social capital and built defences are also needed (Perez-Maqueo *et al.* 2007). The limitations of ecosystems in coastal protection should be recognised for coastal planning, as should the ways in which protection by ecosystems can be enhanced (Tanaka 2009). This can be linked to the resilience of the ecosystem to environmental change (Sudmeier-Rieux 2006), which has been discussed in section 2.3.1.1.

2.3.2 Adaptation in the water sector

The impact on water resources is likely to be the major challenge posed by climate change. In some regions, too little water will lead to droughts and desertification, whereas in others too much water will lead to increased flooding (FAO 2007).

Desertification is considered to be one of the most threatening processes to livelihoods of the poor (MA 2005b) with more than 300 million Africans living in drought or drought-prone areas, a number likely to increase in Africa and on a global scale due to climate change (IPCC 2007). A new report projects that by 2030, 47 per cent of the world's population will be living in areas of high water stress, especially in Africa, with 24 to 700 million people expected to be displaced because of water scarcity (UNESCO 2009). Africa and Asia are expected to be the most impacted, with adaptation costs in the sub-Saharan urban water sector estimated at between 10 and 20 per cent of current overseas development assistance to the region (Muller 2007). Adaptation options for water shortage range from water use controls to the building of reservoirs and diversion of rivers into drought prone areas (Obersteiner 2006). Options for adaptation to flooding include structural defences similar to those used in coastal protection, watershed management, and flood planning. The major adaptation strategies for water related impacts are outlined in Table 2.

TABLE 2: MAJOR ADAPTATION STRATEGIES IN THE WATER SECTOR (OBERSTEINER 2006)

WATER STRESS	FLOODING
Desalination	Structural protection
Ground-water pumping	Watershed management
Water transfer	Land-use planning
Removal of invasive vegetation	Flood forecasting
Improved water efficiency (including demand-side management)	Relocation of populations
Soil moisture conservation	Insurance

Biodiversity can play a role in adaptation strategies to both drought and floods through watershed, wetland, forest, and agricultural management (Berry *et al.* 2008; Kundzewicz *et al.* 2008). Maintenance or restoration of forest and wetlands, for example, can reduce run-off in times of flood and also increase water retention during droughts (Krysanova *et al.* 2008).

2.3.2.1 Adaptation to water stress

Reduced vulnerability to drought, particularly in dryland regions, requires improved soil and water management (Falkenmark and Rockstrom 2008; Stringer 2008). The regulation of water flows in dryland regions has been strongly linked to the proportion of land covered by forest, grassland, and wetland, and maintaining vegetation cover can assist in adaptation to drought (Falkenmark and Rockstrom 2008). Upland watersheds play a vital role in water regulation. Run-off from mountainous areas in SIDS is often the major supply of water (Mata and Budhooram 2007), and in the Phillipines, watersheds are a critical part of the national economy (Lasco *et al.* 2008). Often these watersheds are degraded, and their rehabilitation is one adaptation option (MacKinnon 2007). Planting trees on slope fields, mini-terracing for soil and moisture conservation, and improved pasture management can also complement actions such as building of small-scale infrastructure in water resources management (World Bank 2008). Natural resource management has been included in the NAPA of the Niger, where water stress is the major issue, and the reduction of pressure on freshwater resources is receiving attention in Brazil where the use of pesticides has impacted water quality in many areas (Hedger and Cacouris 2008). Soil erosion measures such as conservation tillage can be coupled with rain water harvesting and are activities that can be undertaken by communities (Paavola 2008). Water management is cross-sectoral, and is particularly relevant to agricultural adaptation. It will be discussed in more detail in section 2.3.3.

2.3.2.2 Adaptation to flooding

In addition to water provisioning services, watersheds can reduce flooding and sedimentation whilst improving water quality downstream. A study of upland forests in a watershed in Madagascar has estimated their flood protection value at US$126,700, and peat bog in Sri Lanka that buffers floodwaters from rivers has an estimated annual value of more than US$5 million (Emerton and Bos 2004; Sudmeier-Rieux 2006). In the Morogoro region of Tanzania, reduced river flow and increased flooding has been attributed to deforestation in the mountains, and it has been suggested that effective governance of soil, forests and water resources are needed as adaptation measures, along with improved social capacity (Paavola 2008). Ecuador and Argentina have integrated forests and wetlands into their 'living with floods' strategies (World Bank 2008), and reforestation is recognised as an important option for adaptation in the watersheds of the Phillipines (Lasco *et al.* 2008). Viet Nam includes measures such as integrated management of watersheds in its disaster reduction planning, along with forest management, and soil and water conservation (Sudmeier-Rieux 2006). Large-scale afforestation projects in China have been carried out with the aim of reducing flooding and increasing water conservation, and countries of Central America are collaborating to protect watersheds and forest (Abramovitz *et al.* 2006).

Ecosystem management is also an effective adaptation strategy at the river basin scale and can be an alternative to the development of dams, which have a high environmental impact (Mata and Budhooram 2007). In developed countries, cost effective flood reduction strategies that allow re-growth of vegetation alongside rivers and establish vegetation buffers along streams, combined with the reduced development of infrastructure, are being promoted in some areas (Nelson *et al.* 2008). Some evidence that this can be an effective strategy has been provided in a modelling scenario exercise, which suggested that a combination of wetland restoration and hard defences provides optimal flood protection (Berry *et al.* 2008). Riparian floodplains can also help to reduce the levels of water pollution following extreme events (CCSP 2008). In Europe, the conservation or restoration of river floodplains has been included in a number of flood reduction strategies (Zaunberger, Agne and Miko 2009), although there are many new river management plans that do not include such measures (Krysanova *et al.* 2008).

2.3.2.3 Integrated watershed management

Water resource adaptation options need to be able to function under uncertain future climate change, but many adaptation measures do not sufficiently account for this (Krysanova *et al.* 2008). The building of dams and large-scale irrigation systems, for example, cannot completely protect against floods and can also damage the adaptation capacity of other sectors, an example of maladaptation (Fraiture *et al.* 2007; Palmer *et al.* 2008). Technical measures such as desalination, pumping of deep groundwater, and water treatment are very resource intensive (Krysanova *et al.* 2008). Increasingly, it is becoming recognised that water management requires an integrated approach, through 'integrated watershed management' which includes natural resource management along with social measures and infrastructure development (Galaz 2007; Bates *et al.* 2008; IUCN 2008; Kundzewicz *et al.* 2008). In principle, restoring and protecting freshwater habitats and watersheds and managing natural floodplains are key elements of such an approach (Glick *et al.* 2009). Reduction of pressure on freshwater resources would be beneficial regardless of the scale of the future impact (Kundzewicz *et al.* 2008), whereas activities such as river regulation, wetland drainage, intensive agriculture, and deforestation degrade freshwater habitats and lower adaptive capacity (Krysanova *et al.* 2008). The Government of India has initiated an Integrated Watershed Management Programme to restore degraded regions through rehabilitating and maintaining the natural resource base, which involves soil moisture conservation measures such as contours, afforestation, vegetating drainage lines, and engineering structures to collect rainwater (Bhandari, Suruchi and Ulka 2007).

These strategies recognise wetlands and river basins as an integral part of the hydrological regime (Harrison *et al.* 2008). It has been widely suggested that an ecosystem approach including wetland and floodplain management and restoration should not be viewed as an alternative to technological approaches such as reservoirs and irrigation systems, but as a complement to them (Mata and Budhooram 2007). However, there appears to be a lag between our understanding of interconnected freshwater resources and adaptation strategies implemented by policy makers (Galaz 2007).

Watershed management should be planned according to local conditions. For example, planting of some tree species could have negative impacts on water flow in some areas (Bhandari *et al.* 2007). In South Africa, 'Working for Water' programmes have been initiated to remove invasive tree species from water catchments where water-thirsty species have reduced the annual river flow by approximately 7 per cent (Mukheibir 2008). Similar impacts have been seen in China, where the monoculture tree species chosen for plantations were not suitable for the area (McVicar *et al.* 2007).

2.3.3 Adaptation in agriculture

The production of food crops is perhaps the most climate-dependent economic activity. Climate change is already affecting agriculture in developing countries negatively, and this situation is likely to worsen (IPCC 2007), with significant impacts on crop yields and on the productivity of grazing lands and livestock expected, through changes in temperature, precipitation, water availability, salinity, and the abundance of pollinators, pests and diseases (Rosenzweig and Tubiello 2007). Impacts will not be uniform, but will vary across regions and require a number of different adaptation strategies (Berry *et al.* 2008). Agricultural production is the main economic activity for rural communities of vulnerable regions such as Africa and India (Chatterjee, Chatterjee and Das 2005; Osbahr *et al.* 2008). In some countries in Africa, yields from rain-fed agriculture could be reduced by up to 50 per cent by 2020 (IPCC 2007). In Central and South Asia, crop yields could fall by as much as 30 per cent by 2050 as a result of climate change; India alone could lose 18 per cent of its rain-fed cereal production (Lobell *et al.* 2008). For agriculture in the world's drylands, the challenges are especially large due to predicted changes in hydrological cycles characterised by both increased droughts and increased risks of flooding (Falkenmark and Rockstrom 2008). Depending on the region and the available resources, options for adaptation range from relatively inexpensive changes, such as shifting planting dates or switching to an existing crop variety, to much more costly measures including the development of new crop varieties, increasing chemical and other inputs and irrigation systems (Rosenzweig and Tubiello 2007). Broadly speaking, the options for adaptation in agriculture include:

i) changes in the locations of cultivation (i.e. opening new areas for cultivation);
ii) changes to the crops cultivated, including substitution by new crops, new varieties and crop diversification; and
iii) changes to agricultural practice, including irrigation and soil management regimes and the use of agricultural inputs.

Biodiversity plays an especially strong role in supporting the latter two options.

2.3.3.1 Changes in location of cultivation

Climate change will lower the suitability of some areas for agriculture, and open up new suitable zones, particularly in northern latitudes. To adapt specific crops to changing environmental conditions, heat-tolerant wheat germplasm or cultivars better adapted to conservation agriculture are being sought or developed (Oritz *et al.* 2007). Alternatively, areas where crops are cultivated would have to change or expand. Indeed, areas of cultivation could shift geographically, following shifting climatic zonations (Rosenzweig and Tubiello 2007). For example suitability for wheat is projected to expand further north in both Eurasia and North America (Oritz *et al.* 2007). Moreover, livestock may also be moved to new zones (Berry *et al.* 2008).

2.3.3.2 Changes to crops cultivated

Within a given region, different crops are subject to different degrees of impact from current and anticipated climate changes (Lobell *et al.* 2008). One major avenue for adaptation is the substitution of different crops more suited to changing and new conditions. Rice, maize, and wheat contribute roughly half of the calories currently consumed by the world's poor (Lobell *et al.* 2008) the remainder of the world's food supply comes from a wide variety of other crops including sorghum, millet, sweet potato, cassava, groundnuts, sugar cane and many different beans. Adaptation in agriculture will include the adoption of many of these crops in areas and farms where they were not previously grown. For example, in a vulnerable community in India, growing new crops together with higher value crops for commercial sale was among the adaptation measures already being adopted to help cope with drought (Chatterjee *et al.* 2005). The most common adaptation strategies used by farmers in South Africa and Ethiopia include the use of different crop varieties (Bryan *et al.* 2009). Adopting new crops and varieties has also been an important aspect of recovery from extreme events in Zimbabwe (Chigwada 2005). Where salinisation is a problem due to rising sea levels or excessive water extraction, the introduction of salt tolerant crops and varieties can help to ensure continued agricultural production (Galvani 2007). Further, the use of currently under-utilised crops can help to maintain diverse and more stable agro-ecosystems (Bowe 2007). The use of indigenous and locally adapted plants can enhance the capacity of communities to cope with changing climatic conditions by providing alternative food and income sources that may be better suited to changing conditions (Eriksen 2005; FAO 2007). For example, the bambara groundnut, an ancient grain legume grown, cooked, processed and traded mainly by subsistence women farmers in sub-Saharan Africa, has great potential to provide continued production in the face of growing climate variability (Azam-Ali 2007).

Developing climate-tolerant crop and livestock varieties and genotypes, such as those tolerant to drought, heat stress, disease and saline conditions, is another avenue for increasing the adaptive capacity of farmers (Kesavan and Swaminathan 2006; Aggarwal 2008; Ortiz *et al.* 2008). Such selection will often depend on locally used varieties and crop wild relatives as sources of characteristics that contribute to drought or flood tolerance or the ability to withstand highly variable climate (Bailey-Serres and Voesenek 2008). However, changes in cultivars and livestock races can bring other climate-related risks. For example, adapting winter cereal production by using longer-maturing cultivars is dependent on there being enough precipitation over the extended growing season to sustain grain filling (Rosenzweig and Tubiello 2007). Increasingly, new crop varieties are being developed through genetic modification that can incorporate individual traits and does not depend on a long breeding programme. There is a danger that these costly (and in some cases environmentally risky approaches) may target environmental tolerances that are not appropriate to eventual real climatic changes in large areas of the world.

In addition to substituting new crops, races and cultivars with those currently in use, adaptation may involve crop diversification. Although empirical evidence is lacking, it is likely that farming practices can be more easily adapted to cope with changes in water availability or temperature if a larger number of crop varieties are available (Smale 2005; Weltzien *et al.* 2006; Bowe 2007; Reid, Simms and Johnson 2007; Thomas *et al.* 2007; Hedger and Cacouris 2008; Kouressy *et al.* 2008; Reidsma and Ewert 2008). Crop diversification and mixed cropping are currently being used in Brazil and Ghana, to increase the chances that at least one crop will survive and produce a harvest (Leavy and Lussier 2008). At its most successful, diversification also provides increased income by ensuring that there are several different income streams available (Leavy and Lussier 2008). Research on agro-ecosystems in China has suggested that diversification of agriculture is a promising poverty reduction strategy but requires efficient use of resources (Hengsdijk *et al.* 2007).

The adoption of new crops and development of new varieties and cultivars, whether through breeding or genetic modification, are clearly dependent on the availability of a range of crops; the maintenance of agrobiodiversity is therefore critical to such adaptation (Kotschi 2007; Fowler 2008). Ensuring the continued survival of crop wild relatives that provide additional genetic diversity for breeding and the development of new varieties is also crucial (Jarvis, Lane and Hijmans 2008).

2.3.3.3 Changes in agricultural practice

Changing many different aspects of farming practice, ranging from planting and harvest dates to water and soil management practices, will also be an important part of most agricultural adaptation strategies.

In many cases alterations to planting and/or harvest dates are helpful in dealing with climatic changes (Rosenzweig and Tubiello 2007; Bryan *et al.* 2009). Early sowing has been found to be helpful in some cases (e.g. in Australia; Luo *et al.* 2009), but it can be problematic if conditions are dry. Double cropping may even be possible in regions where the length of the growing season is increased due to climate change, as is projected for the Mediterranean climate regions in central Chile (Meza, Silva and Vigil 2008), but this is likely to increase the use of pesticides and fertilizers. The success of changes to cropping dates is also dependent on the availability of pollinators and therefore on changes to biodiversity within the surrounding landscape.

Globally, agriculture consumes more than 3000 litres of water per person per day to meet food demands (Molden 2007). 60 per cent of all agricultural production comes from rainfed land, while 40 per cent comes from irrigated areas (Fraiture *et al.* 2007). Changing precipitation regimes will likely alter this balance. Managing water supplies and demands will be vital to adaptation of agriculture worldwide, and especially in drylands (Falkenmark and Rockstrom 2008). Development of new irrigation systems is costly, improved capture storage and use of rainwater is less so (Shiferaw, Okello and Reddy 2007). Water conservation is particularly important in India as 68 per cent of the agriculture is rainfed, making watershed development through soil and water conservation vital for adaptation to climate change (Bhandari *et al.* 2007; Chatterjee *et al.* 2005). The use of water-efficient and/or perennial crops can reduce demand for water (Bell *et al*. 2008; Reid *et al.* 2007), and inexpensive measures to enhance water productivity of agricultural systems through soil and watershed conservation can improve rural incomes and diversify livelihood streams whilst increasing carbon sequestration (Castillo *et al.* 2007; Hartmann, Hediger and Peter 2007; Molden 2007; Noble 2007). In Senegal, where farmers have had to adapt to successive droughts and a drying climate, planting dense perennial hedges as windbreaks helps to improve the microclimate for crop growth (Seck, Abou Mamouda and Wade 2005).

In other areas, drainage or dyke building may be necessary to reduce flooding probabilities and the impacts of extreme events, and to make lowland areas usable for agriculture (Olesen 2006), but such measures can be costly and have negative impacts on biodiversity. Less expensive measures include raising beds and floating gardens, both of which are being adopted in flood-prone areas such as Bangladesh (Leavy and Lussier 2008).

Soil conservation and enhancement are also an important part of adaptation in agriculture. This can include structural methods such as terracing and stone bunding (Shiferaw *et al.* 2007), the use of chemical or organic fertilizers, changes to tillage practices, and agroforestry techniques. On the whole, good practice agriculture such as crop rotation, contour tiling, minimum tillage, the use of vegetation buffer strips, and agroforestry can all play major roles in adaptation (Berry *et al.* 2008). Conservation agriculture, which involves minimizing soil disturbance and maintaining cover through plantings or mulches, and organic agriculture (Huang 2008; Muller 2009) are promising options for adaptation in farming communities because they increase soil carbon and water retention, decreasing vulnerability to extreme weather events (FAO 2007; Lal *et al.* 2007; Thomas *et al.* 2007; WRI 2008). They also reduce the need for nutrient inputs and use of heavy machinery. In drylands, agricultural practices such as the use of shadow crops can enhance resilience by providing protection against extreme rainfall, and increasing infiltration into the soil (Blanco 2004). Vegetation litter, the use of nutrient enriching plants, reduced use of fertiliser, crop diversity, and maintenance of forest can also be utilised as adaptation strategies (Blanco 2004). Sand and dust storms can be reduced through the use of forest shelterbelts and improved cohesion of soil particles through practices such as mulching (Sivakumar 2005). Replanting of indigenous trees can reduce soil and wind erosion, as can ridging and mulching (Abramovitz *et al.* 2006). Farms using agro-ecological practices such as soil conservation have been found to be more resilient to hurricanes (Reid and Swiderska 2008).

Agroforestry, intercropping food crops with tree stands, can improve biophysical resilience and promote income diversification (Verchot *et al.* 2005) and is one of the most promising options for helping communities adapt and become resilient to the impacts of climate change. It provides permanent cover leading to soil conservation and microclimatic buffering, opportunities for diversification of the agricultural systems, and improved efficiency of water resources (Rao *et al.* 2007), and is especially important to smallholder farmers with significant biodiversity benefits (Verchot *et al.* 2007). Agroforestry and many other forms of agricultural good practice including reduced tillage, were originally designed as "best practice" management strategies, aimed at enhancing the long-term stability and resilience of cropping systems in the face of climate variability or of increased cultivation intensity (Rosenzweig and Tubiello 2007). They also serve an important role in climate change mitigation by enhancing carbon stocks within the agricultural landscape (Kandji *et al.* 2006). Further, they both increase and depend on biodiversity and ecosystem services.

The viability of the many different options available for adaptation in agriculture is dependent on the availability of financial, human and natural resources and on the willingness of farmers to consider the options (Reidsma 2007; Brondizio and Moran 2008). To date, there have been few examples of policy level decisions to promote adaptation in the agriculture sector (Ziervogel *et al.* 2008), and one of the challenges for adaptation researchers is to understand how best to address the information needs of policy-makers and report and communicate agronomic research results in a manner that will assist the development of food systems adapted to climate change (Gregory *et al.* 2008; Bryan *et al.* 2009). It has been suggested that adaptation strategies should invest in sustainable agriculture, promoting soil and water conservation and preserving biodiversity (Leavy and Lussier 2008), and should be part of a strategic governmental response (Bryan *et al.* 2009).

2.3.4 Forest adaptation

Much of the discussion related to forests and climate change has focused on mitigation, rather than adaptation (Guariguata *et al.* 2008). Although there is a wealth of literature on the ecosystem services provided by forest and the links to livelihoods, little is explicitly related to climate change adaptation. Much of the literature that does exist is related to management of temperate forest (Noss 2001; Millar *et al.* 2007; Ogden and Innes 2007; Locatelli *et al.* 2008; Guariguata *et al.* 2008; Kalame *et al.* 2009). However, the role of forests in societal adaptation is becoming increasingly recognised (Eliasch 2008), and has led to the development of initiatives such as the Congo Basin Forest and Climate Change Adaptation (COFCCA) project. Solidifying the links between forests and adaptation will be important to reduce damaging management practices that could lead to maladaptation in the longer term (Nkem *et al.* 2007).

2.3.4.1 Role of forests in adaptation

Forests can also contribute to adaptation as a component of biodiversity in three main ways; through structural defence against wind and soil erosion, through water regulation, and through the provision of timber and non-timber forest products (NTFPs) (Eriksen *et al.* 2006; Innes and Hickey 2006; McEvoy, Lindley and Handley 2006; Ogden and Innes 2007; UN 2008; WRI 2008; Paavola 2008), as has been discussed in previous sections. On a local scale, forests can provide shade and reduce exposure to heat; for example, a study in Kenya found that improved microclimate and catchment properties of a hilltop area were closely linked to good biodiversity status of the forest (Eriksen *et al.* 2006). Conversely, deforestation is a driving force for loss of ecosystem services and land degradation (Cangir and Boyraz 2008). Forest dwellers and those that rely on forest resources are often the poorest members of society and have low adaptive capacity (FAO 2007; Ravindranath 2007). Where access to NTFPs become marginalized, vulnerability of the poorest people increases (Eriksen, Brown and Kelly 2005; Paavola 2008). Both natural and plantation forests can provide 'safety nets' during periods of food shortage, and can provide an important contribution to food security (Nkem *et al.* 2007; Kalame *et al.* 2009). Community involvement in afforestation projects, for example, can diversify incomes and improve social capacity, reducing the vulnerability to climate change impacts (Spittlehouse 2005; Guariguata *et al.* 2008).

Forests can be particularly important during extreme events. In addition to the provision of 'safety nets', it has been suggested that forest cover can reduce landslide erosion by a factor of 4-5 compared with sites that lack substantial tree root strength, and reduce flooding (ISDR 2004; ProAct Network 2008). In a study of North Pakistan, it was estimated that 56 per cent of all landslides were due to land degradation from deforestation and grazing, and that protective forests would be a cost effective action to reduce disaster risk (Sudmeier-Rieux *et al.* 2007).

In the Amazon, forest has a major role in the regional hydrological regime (Correia, Alvala and Manzi 2008). Forest loss could push some subregions into a permanently drier climate regime, increasing vulnerability of societies to drought conditions (Betts 2007; Malhi *et al.* 2008). Recent research has suggested that there is the potential for large-scale die-back of the Amazon rainforest through a combination of degradation and drought (Nepstad *et al.* 2008; Phillips *et al.* 2008), although it is thought that intact forests will be more resilient to climate change impacts (Gullison *et al.* 2007; Bush *et al.* 2008; Malhi *et al.* 2008).

Forest management and conservation practices may help to decrease the vulnerability of those who depend on forest services for their livelihoods, while at the same time maintaining the mitigation capacity of forests (Guariguata *et al.* 2008; IUCN 2008). Adaptation in the forest sector (for both natural and plantation forest) can either enhance resistance and resilience of existing forests to climate change, or facilitate adaptation to new conditions (Locatelli *et al.* 2008). Other adaptation options include diversification of the forest economy and the forecasting of potential pest impacts (Ogden and Innes 2007; La Porta *et al.* 2008).

2.3.4.2 Role of biodiversity in forest adaptation

Climate change is rarely factored into forest planning (Nitschke and Innes 2008), possibly due to the uncertainties surrounding the vulnerability of forests to climate change (Chapin *et al.* 2007; Millar, Stephenson and Stephens 2007). A mixture of adaptation measures will be required, depending upon whether the goal is to manage for a specific ecosystem service, or for resilience in general (Locatelli *et al.* 2008).

2.3.4.2.1 Natural forest

Evidence suggests that intact forests, particularly old growth forests, will be more resilient to climate change (Betts, Malhi and Roberts 2008; Malhi *et al.* 2008). Strategies aimed at reducing emissions from deforestation and degradation (REDD) could therefore play a significant role in adaptation through maintenance of biodiversity and ecosystems services such as water cycling (Betts 2007; Betts, Sanderson and Woodward 2008; Malhi *et al.* 2008; Nepstad *et al.* 2008). Indeed, it has been suggested that REDD could be the most effective strategy for both adaptation and mitigation, as it is likely to reduce anthropogenic threats to forest (Berry *et al.* 2008). However, badly designed REDD strategies could increase vulnerability of local communities if they are denied access to important forest resources (Locatelli *et al.* 2008)

Many of the management activities required to enhance resilience in natural forest are similar to those required to maintain carbon stocks, such as reduced impact logging, forest conservation and sustainable forest management (Guariguata *et al.* 2008). However, there will also be trade-offs between adaptation and mitigation. For example, maintenance of the genetic diversity of forests is likely to play a large role in forest adaptation to climate change (Guariguata *et al.* 2008; Sevrin 2008; WRI 2008; Kalame *et al.* 2009), but is unlikely to be considered in mitigation strategies. Other strategies for forest adaptation can include the maintenance of different forest types across environmental gradients, expansion of the protected area network, the protection of climatic refuges, the reduction of fragmentation, and the maintenance of natural fire regimes (Noss 2001; Locatelli *et al.* 2008; Glick *et al.* 2009). These conservation strategies will be discussed in more detail in section 2.5.

2.3.4.2.2 Plantation forest

There is significant potential to adapt plantation forests to future conditions. Genetic diversity is likely to be important as different varieties may include phenotypes or genotypes that are adaptable to future climates. Good plantation practice would therefore ensure that a mix of species and range of age structures are incorporated into plantation forests (Berry *et al.* 2008; Guariguata *et al.* 2008). This will be beneficial for biodiversity in addition to improving adaptive capacity. As forest species are long-lived, adaptation measures undertaken now need to be planned according to likely future conditions and be flexible to change (Millar *et al.* 2007; Ravindranath 2007).

Although afforestation can stabilise soils in suitable areas and provide nutrient and water flow benefits, this needs to be considered in the context of current land use and can involve trade-offs, particularly with water usage (Berry *et al.* 2008; Ravindranath 2007). Indeed, reforestation may be beneficial in areas where former forests have been replaced by crops and therefore potentially restoring water quality (Plantinga and Wu 2003). On the other hand monocultures may have negative impacts (Calder 2007). Selecting appropriate species will include a consideration of the nutrient and water requirements of an area. An example can be given of the largest monoculture plantation in the American tropics in Venezuela, which suffered a large-scale tree mortality as a result of water stress during the 1997 *El Niño* (Guariguata *et al.* 2008). This is another example of a potential trade-off between adaptation and mitigation. Forest plantations for carbon sequestration have generally been established using genetically uniform stock with high growth rates, but low adaptive capacity, which will ultimately diminish their capacity in mitigation (Innes and Hickey 2006). Afforestation in unsuitable areas, using unsuitable crops and monocultures, can have significant impacts on biodiversity, soil erosion, nutrient cycling, and water regulation (Campbell *et al.* 2008).

The central role that forests can play in local adaptation has not been translated into broader adaptation policy (Nkem *et al.* 2007; UN 2008; Kalame *et al.* 2009). This is true both on national scales and under the UNFCCC (Locatelli *et al.* 2008). Forests are widely seen as carbon sinks for sequestration payments (Kalame *et al.* 2009), and there are significant socio-economic and political barriers to mainstreaming adaptation into sectoral forest policies (Kalame *et al.* 2009). Although forestry is generally not a priority in adaptation policy (Locatelli *et al.* 2008), a number of the NAPAs prepared by LDCs do have forest projects within their adaptation priorities (Guariguata *et al.* 2008). Developing countries that have identified forest adaptation priorities need further guidance to enhance the adaptive capacity of their forests (Guariguata *et al.* 2008).

2.3.5 Adaptation in the urban environment

The role of biodiversity in the urban environment is less intuitive than for other sectors. However, the urban environment is a large adaptation sector, and it should not be overlooked. The majority of the global population live in cities and will suffer impacts of climate change, mainly through overheating (with higher temperatures expected in cities than in rural areas), flash floods, and extreme weather events (Smith and Levermore 2008). ‘Structural’ adaptation measures in the urban environment can include improved building design (for increased ventilation, shading etc), increased use of air conditioning, and improved drainage through more permeable surfaces (McEvoy *et al.* 2006). Adaptation measures related to sea level rise in coastal areas and river basins have been discussed in section 2.3.1.

Biodiversity can play a role in urban planning through the expanse of green areas for cooling, improved use of natural areas for drainage and flood reduction, and urban tree planting for structural integrity and removal of pollutants (McEvoy *et al.* 2006; Berry *et al.* 2008). ‘Urban greening’ can improve the microclimate by modifying heat absorption (Smith and Levermore 2008), whereas paving over areas covered by vegetation and water reduces heat loss and increases vulnerability to flooding (Grimm *et al.* 2008). Increasing ‘blue space’ (e.g. lakes and canals) is also recommended for cooling and reduced risk of flooding (Grimm *et al.* 2008).

Clearly, structural measures are required for adaptation in the urban environment. However, a recent analysis of the built environment in Boston has suggested that a combination of both structural and 'green' adaptation measures is the optimal strategy to reduce the negative effects of climate change in the built environment, and that considering integration with land-use management and coordination amongst institutions is a necessary response to climate change (Berry *et al.* 2008; Kirshen, Ruth and Anderson 2008). Despite this, 'green space' is often overlooked in urban design and adaptation plans (McEvoy *et al.* 2006). A recognition of the ecosystem services and economic benefits that can be provided through incorporating ecology into urban design will be important for future sustainable city design (Grimm *et al.* 2008).

2.3.6 Health

Climate change is likely to have major impacts on health through heat exposure, extreme weather events, air pollution, malnutrition, reduced water quality and availability, water borne diseases, and spread of disease vectors (WHO 2008; Kjellstrom and Weaver 2009).

Although there is a growing body of literature supporting the importance of biodiversity for health (Chivian and Bernstein 2008), few links have yet been made to the role of biodiversity in adaptation to health impacts. Productive ecosystems are necessary for food production, freshwater production, fuel, and waste management (Corvalan, Hales and McMichael 2005) and the role of biodiversity in adaptation to extreme event impacts, heat exposure, water stress and food production has already been discussed. It has been reported that approximately one quarter of the global disease burden is due to modifiable environmental factors and that 42 per cent of incidences of malaria are associated with policies and practices related to land use, deforestation, and water resource management (Pruss-Ustun and Corvalan 2006). This would appear to be an area that requires increasing attention in the future.

2.3.7 Integration across sectors

This report has focused on separate adaptation sectors, as this is how the literature is generally organised. However, the need for integrated adaptation strategies across sectors to avoid maladaptation is becoming increasingly recognised (AIACC 2007). For example, there is a high level of interdependence between agriculture and water resources, where good watershed management can act synergistically to improve agricultural practice, whereas bad management can have a negative impact and vice versa (Lasco *et al.* 2008). Natural resource management in particular tends to run across a number of sectors.

The literature suggests that although integration of adaptation across sectors is preferable, including the integration of environmental measures, this will require significant institutional capacity (Agrawal 2008; Zaunberger *et al.* 2009). Integrating natural resource management into adaptation in particular requires considerable institutional support, and this is currently lacking (Tompkins and Adger 2004a; AIACC 2006; Eriksen *et al.* 2006; Locatelli *et al.* 2008; Kalame *et al.* 2009). Linkages are rarely made between adaptation policy and issues of governance and land tenure, which are key in developing adaptive capacity to manage resources (Agrawal 2008). One case study in the Philippines suggested that although there were significant synergies between adaptation options in the forest, agriculture, and water sectors, there were trade-offs involved at the institutional level due to tight budget constraints (Lasco *et al.* 2008).

Although such discussions are beyond the scope of this review, institutional networks to support the inclusion of biodiversity and the effective participation of local communities in adaptation strategies are likely to be a key determinant of the integration of biodiversity into adaptation (Adger *et al.* 2005a; Barbier 2006; FAO 2007; Resurreccion *et al.* 2008; Matthews and Quesne 2008; Bryan *et al.* 2009).

2.4 ADAPTATION STRATEGIES AND THEIR IMPACT ON BIODIVERSITY

There is very little literature surrounding the impacts of adaptation strategies on biodiversity, as few adaptation measures have actually been implemented (Adger *et al.* 2007; Paterson *et al.* 2008). However, potential impacts can be identified through our knowledge of likely adaptation measures and the environmental impacts of past management practices. Some countries, such as The Netherlands, England and France, have begun developing policy for climate change adaptation, in which the requirements to perform Environmental Impact Assessments (EIAs) and Strategic Environmental Assessments (SEAs) on adaptation projects have been recommended, as well as the need to consider ecosystem-based planning (Wilson and Piper 2008). This is based on the recognition that considering biodiversity in the design and operation of infrastructure projects can reduce environmental costs and increase the sustainability of the project (Quintero 2007). The environmental impacts of the commonly used/considered adaptation measures will be discussed in this section. When considering the impacts of adaptation strategies on biodiversity it is important to consider trade-offs, such as the implications for local incomes and adaptive capacity.

2.4.1 Coastal defence

2.4.1.1 Coastal barriers

Most of the literature available on this topic is related to the 'hard' structures constructed for defence against coastal erosion and sea level rise. Coastal protection, particularly in developed countries, has traditionally been in the form of dykes, seawalls, and tidal barriers, and construction in this area is likely to continue (IPCC 2007). It was recognised in the IPCC AR4 that structures such as seawalls and dams can alter sediment deposition, prevent inland migration of vegetation in response to changing sea levels, and impact upon salt marshes (IPCC 2007). This impact of hard defence structures is well documented, and there is evidence that this 'coastal squeeze' and altered sediment deposition is threatening mangrove ecosystems (Gilman, Ellison and Coleman 2007; Jagtap and Nagle 2007; Gilman *et al.* 2008), in addition to tidal flats, salt marshes, and dunes (Glick *et al.* 2009).

Few studies have considered the impact of hard defences on coastal ecology. Hard structures can result in changes in species composition, abundance and diversity, which have important consequences for the functioning of ecosystems (Airoldi *et al.* 2005). Recent research has shown that beaches protected by hard defences suffer reduced availability of habitat and macroinvertebrates due to the loss of upper intertidal zones, which has led to reduced species richness and abundance of shorebirds and seabirds (Dugan *et al.* 2008). One area that requires increasing attention is the impact of coastal structures on fish ecology. A recent study has suggested that species assemblages differ between natural and artificial reef structures, and that it is unclear whether artificial structures will be effective fish habitats (Clynick, Chapman and Underwood 2008). This will be particularly important in areas where defence structures impact upon mangrove and coral reef ecosystems that provide nursery grounds for fish. Sea wall construction has also been noted to have impacts on plant diversity at the upper borders of salt marshes (Bozek and Burdick 2005). There is also some evidence that habitats protected by hard coastal defence structures facilitate the invasion of invasive species, such as non-indigenous macroalgae (Bulleri, Abbiati and Airoldi 2006; Vaselli, Bulleri and Benedetti-Cecchi 2008).

However, habitat can also be created by engineered structures such as dykes and seawalls (Berry *et al.* 2008). A number of studies in Sydney Harbour, Australia, have found that intertidal molluscs (key species in rocky shore ecology) do occur on seawalls but with differing levels of abundance and diversity that has uncertain implications for intertidal biodiversity (Chapman 2006; Moreira 2006; Blockley 2007).

It is not just 'hard' protection measures that can impact upon biodiversity. Beach nourishment is a widely used 'soft protection' approach to deal with coastal erosion. Although there is much uncertainty, it is thought that beach nourishment can have significant biodiversity impacts through the dredging of habitats for sand material, which can bury shallow reefs, reduce fish habitats, reduce invertebrate densities, and impact upon turtle nesting (Bilodeau and Bourgeois 2004; Peterson and Bishop 2005; Speybroeck *et al.* 2006; Colosio, Abbiati and Airoldi 2007; Speybroeck *et al.* 2007; Fanini *et al.* 2009; Glick *et al.* 2009). However, it has been suggested that with proper planning beach nourishment would have a lower impact than the use of hard defences (Jones, Gladstone and Hacking 2007), and that a better understanding of the ecological impacts is required (Jones *et al.* 2008).

There can clearly be significant environmental impacts from hard defence construction which may vary under different spatial or temporal scales (Airoldi *et al.* 2005). However, trade-offs need to be considered where hard protection is necessary. It has been estimated that hard protection in Germany reduces US$300 billion of damage (Sterr 2008). Similarly, although flood control schemes in Bangladesh such as sluice gates reduce fish production and species richness, they can be beneficial for agriculture (Halls *et al.* 2008).

The impact on biodiversity from any adaptation measure may be large if it is not tailored to the coastal ecosystem (IPCC 2007). Indeed, although 'soft defences' generally have a more positive effect on biodiversity than hard defences (see section 2.3.1.1) an integrated coastal management strategy is likely to be more effective (Duxbury and Dickinson 2007).

2.4.1.2 Managed realignment and accommodation

The strategies of managed realignment and accommodation, which can involve the movement of infrastructure inland and improved land-use planning (Ellis 2008), can be beneficial for biodiversity as they are often combined with activities such as wetland restoration (Berry *et al.* 2008). Moving coastal defences inland can create new intertidal habitat (Hardaway *et al.* 2002; Berry *et al.* 2008), and can provide breeding and feeding grounds for water birds (Crowther 2007). It can also facilitate the inland migration of mangroves (Gilman *et al.* 2008), and can reduce coastal squeeze for wetland habitats more generally (Berry *et al.* 2008). Accommodation reduces building on coastal areas and can involve habitat restoration.

However, one study has suggested that whilst the realignment of embankments can reduce the requirement to constantly upgrade flood defences, there are clear incompatibilities between flood defence and habitat restoration objectives that need to be evaluated (French 2008). Movement of structures inland can facilitate the transition of salt marsh, the habitat thought to be most at risk from coastal defences, to mud flats (Gardiner *et al.* 2007). At realignment sites in the UK, biological monitoring has been poor, and although new habitats have been created they lack the biodiversity found in surrounding natural habitats (Atkinson 2004). In moving infrastructure there is also the potential for adverse impacts on biodiversity at the relocation sites (Berry *et al.* 2008), and careful land-use planning is clearly required.

Accommodating floods is likely to be necessary in some areas such as SIDS, and will lead to large-scale migration of environmental refugees, with unquantified environmental impacts.

2.4.2 Water management

River flood defence systems are similar to those used in coastal defence. River breakwaters, dykes, dams, levees, and floodgates are all large structures used to prevent flooding around rivers, and can have significant environmental impacts. Channelization, dams and other large-scale structures usually have the most negative impacts on biodiversity (Watkinson, Cornell and Tinch 2007). In addition to loss of natural vegetation along river banks, these structures can reduce connectivity between lakes, rivers, and riparian zones, and reduce sediment flows; contributing to the loss of wetlands (Huang *et al.* 2007; ProAct Network 2008). This can actually increase flooding and reduce water quality downstream *(*Abramovitz *et al*. 2006). They can also act as a barrier to the movement of aquatic species such as fish (Berry *et al.* 2008; Krysanova *et al.* 2008; Reid and Swiderska 2008). However, well planned adaptation measures, including some 'hard' defences where appropriate consideration for the environment is taken into account, can have no negative consequences or even beneficial consequences such as reclamation of flooded land (Watkinson *et al.* 2007; Hansson, Danielson and Ekenberg 2008).

In addition to defence from flooding, a number of engineering adaptation options are being employed to reduce water shortages, ranging from construction of dams and reservoirs to engineering to improve river flow, and diversion of rivers. Large infrastructure projects can have major environmental impacts, particularly diversion of rivers which requires extensive landscape planning (Larsen, Girvetz and Fremier 2007). Large-scale dams can cause deforestation, loss of habitats, impact on aquatic biodiversity, and reduce the services provided by downstream floodplains and wetlands (Mata and Budhooram 2007). Removing river vegetation to improve river flow can negatively impact biodiversity by disconnecting wetlands from water sources (Berry *et al*. 2008).

2.4.3 Agricultural practice

Many of the adaptation practices discussed for agriculture such as development of perennial wheat varieties, mixed cropping, agroforestry, and organic farming are all likely to be beneficial to biodiversity. This is because soil, water, and nutrient conservation are all vital for adaptation. However, as discussed in a review of agricultural mitigation strategies (Campbell *et al.* 2008), 'worst case' management practices will always have the potential to impact biodiversity (Berry *et al.* 2008), a review of which is beyond the scope of this report. Impacts will depend on local circumstances and conditions.

There are a number of specific adaptation strategies for agriculture that are likely to impact upon biodiversity. Draining wetlands to increase agricultural production during flooding, the use of dykes, and increases in irrigated agriculture can all have impacts on biodiversity through loss of habitat, soil erosion and eutrophication (Olesen 2006), as can increased use of pesticide to control increased pest outbreaks. These actions are also likely to have a negative impact on adaptation options in other sectors (Berry *et al.* 2008). The replacement of crop systems with monoculture crops selected for specific traits such as drought resistance could increase soil erosion and pesticide use whilst also lowering resilience to climate change (Abramovitz *et al*. 2006). Similarly, the use of genetically modified crops could have as yet unquantified environmental impacts, with risks of invasiveness and reductions in genetic fitness. However, these crops offer great potential for adaptation and trade-offs may be required (Berry *et al.* 2008).

Intensified agriculture, whilst providing gains in the short term, can degrade natural resources and lead to maladaptation in the long term, particularly for the most vulnerable groups (Paavola 2008). This highlights the need for integrated policy development across sectors (Berry *et al.* 2006).

2.4.4 Urban environment adaptation

Many of the strategies proposed in urban adaptation, including the increase of 'green' and 'blue' space, and urban tree planting (as discussed in section 2.3.5) will be beneficial to biodiversity (McEvoy *et al.* 2006; Berry *et al.* 2008). However, man-made streams and canals will not be substitutes for the loss of natural systems (Grimm *et al.* 2008), and where there is migration from rural areas due to climate change impacts, urbanisation will impact on biodiversity through habitat fragmentation and increased waste production (Grimm *et al.* 2008).

2.4.5 Health

There is very little information available on the links between the biodiversity impacts of adaptation and the health impacts of climate change. However, the increased spread of mosquitoes could be controlled by draining wetland breeding sites and introducing fish species to control mosquito larvae. This would likely have negative impacts on biodiversity, as would control through the use of chemicals (Berry *et al.* 2008).

2.5 ADAPTATION IN BIODIVERSITY CONSERVATION

Intact and resilient ecosystems can play a role in climate change adaptation, in many cases providing cost-effective options to reduce vulnerability to climate change impacts (see above). The range of current and potential impacts of climate change on biodiversity (Kapos *et al.* 2008) means that adaptation strategies are needed in the biodiversity conservation sector to address and minimise these impacts. Such strategies are needed not only to help achieve conservation goals, but also to ensure that biodiversity can continue to contribute to societal adaptation to climate change and to climate change mitigation.

The IPCC AR4 (Fischlin *et al.* 2007) outlined a number of potential adaptation strategies to reduce climate impacts on ecosystems, including the reduction of anthropogenic pressures, development of appropriate protected area networks, landscape management, controlled fire management, habitat restoration, captive breeding and assisted migration. A limited number of subsequent studies have identified possible adaptation strategies and frameworks for adaptation to maintain biological diversity and the capacity of species and ecosystems to accommodate and adapt to climate change (Huntley 2007; Mitchell *et al.* 2007; Berry *et al.* 2008; CCSP 2008; Gayton 2008; Heinz 2008; Ptato 2008; Glick *et al.* 2009). These include the protection of key ecosystem features or areas likely to act as 'refuges from climate change', maintaining representation and replication of species and ecosystems, and the restoration of damaged ecosystems (CCSP 2008). Recent research on adaptation to climate change in biodiversity conservation is reviewed here, organised into *autonomous* and *planned* adaptation.

2.5.1 Autonomous adaptation

The ultimate objective of the UNFCCC (Article 2; UNFCCC 1992) is to "achieve stabilization of greenhouse gas concentrations *[...] at a level that would prevent dangerous anthropogenic interference [...] within a timeframe that allows ecosystems to adapt naturally to climate change*". Current conservation practices, generally aimed at maintaining species diversity, can facilitate the variation that would allow ecosystems to 'adapt naturally' to environmental change (Berry *et al.* 2008). Indeed, there is some evidence that species have the capacity to adapt (e.g. Skelly *et al.* 2007), as can been seen by range shifts and phenological changes as responses to past climate change (Kapos *et al.* 2008). The full extent to which species will be able to adapt to climate change is largely unknown (Visser 2008), but there are likely limits to natural adaptation, particularly taking into account the scale of projected climate change.

Species may be able to adapt autonomously to climate change by:
i) dispersing to suitable habitats;
ii) changing their phenotype without a change in genotype via phenotypic plasticity; or
iii) adapting by genetic change over generations (evolutionary response).

The former two may occur rapidly, and have been observed more frequently than the latter as responses to recent climate change (Kapos *et al.* 2008). However, the main concern is whether species will be able to adapt fast enough to keep up with their changing environment with major biodiversity loss (Visser 2008). Some species will be more able to adapt than others, depending on generation times, ability to disperse, and dependency on other species, for example pollinators, hosts for parasites and symbionts (Baker *et al.* 2004; Best 2007). Potential further constraints to evolutionary responses to climate change include time lag between change and response, and erosion of genetic variation (Skelly *et al.* 2007). It is widely accepted that many species and ecosystems will not be able to adapt naturally to climate change under the timescales predicted, and that planned adaptation responses will be required.

2.5.2 Planned adaptation

Conservation management in the context of climate change faces several challenges, including resolving the tension between urgency of action (climate change is already having measurable impacts on biodiversity (Kapos *et al.* 2008)) and uncertainty about:
i) the nature and magnitude of climate change itself in any given location;
ii) the likely responses of species and ecosystems;
iii) the effect of the interaction of different responses; and
iv) the likely effect of management on responses.

There is still relatively little concrete scientific evidence on the effectiveness of different management strategies in relation to climate change, so much adaptation work is still based on ecological reasoning, rather than on extensive research and case studies (Heller and Zavaleta 2009). In the face of these uncertainties, there is a need for proactive management strategies that can quickly be adapted to new circumstances and changing conservation priorities (Heinz 2008; Lawler *et al.* 2009). These will require institutional coordination, incorporation of climate change scenarios into planning, and efforts to address multiple threats simultaneously (Heller and Zavaleta 2009).

2.5.2.1 Ecosystems

Planning conservation action with full consideration of climate change (and its associated uncertainties) could help to reduce the vulnerability of entire ecosystems (Ravindranath 2007). Species responses will ultimately determine the ability of ecosystems to adjust and persist under changed climates (Gayton 2008); changes in ecosystems will in turn promote further changes in species abundances, distributions and interactions, with the possible breakdown of traditional species relationships, such as pollinator/plant and predator/prey interactions (Backlund, Janetos and Schimel 2008). It has been suggested that ecosystems and communities themselves should not be the focus of conservation actions to adapt to climate change because differential responses among component species will mean certain changes in their composition and identities (Huntley 2007). However, the importance of maintaining ecosystem resilience (Kareiva *et al.* 2007) and its relationship to maintaining adequate extent and diversity of habitat to facilitate species adaptation has repeatedly been emphasised (Hopkins *et al.* 2007; Huntley 2007; Mitchell *et al.* 2007; Harley and Hodgson 2008). Therefore, many conservation interventions address management at the ecosystem scale, and aim at the continued existence of ecosystems and the provision of the services they provide.

Among the key needs that have been identified as driving conservation actions in the context of adaptation to climate change are:

- the need to maintain adequate populations of species and sufficiently large areas of ecosystems to ensure their resilience and ability to continue to maintain biodiversity and provide other ecosystem services;
- dispersing to suitable habitats; the need to ensure functional connectivity between populations and habitats so that species are able to shift their distributions in response to climate change;
- the need to reduce other stresses on ecosystems and species.

Adaptation to climate change in conservation management at the ecosystem scale, which aims to address these needs, therefore falls into three broad categories:

i) changes in the extent and design of protected area systems;
ii) changes in their management; and
iii) management of the wider landscape, including efforts to ensure functional connectivity.

2.5.2.1.1 Protected areas systems

Protected areas have long been used as an important tool to secure sites that are perceived as important in biodiversity conservation (Williams *et al.* 2005; Lee and Jetz 2008) and to reduce the pressures that affect the ecosystems and species within them. They are also potentially important tools for limiting the impacts of climate change on biodiversity (Hannah 2008; Heller and Zavaleta 2009). Extending and/or strengthening protected area networks is frequently emphasised as one of the fundamental options for adaptation to climate change in the conservation sector (Ravindranath 2007; Killeen and Solorzano 2008; MacKinnon 2008; Malhi *et al.* 2008), and is emphasised in a number of proposed adaptation frameworks (Mitchell *et al.* 2007; CCSP 2008; Heinz 2008; Mcclanahan *et al.* 2008).

However, current protected areas were established to conserve species and ecosystems in a stable climate; at best they were designed to conserve particular components of biodiversity as they were distributed at the time of the initial assessment and planning (Lemieux and Scott 2005; Huntley 2007), and at worst they were located in areas where conflicting demands for land were minimal (Pressey 1994; Mackey *et al.* 2008). As species' ranges shift in response to climate change, and ecosystem composition changes as a result, existing protected areas may play a limited role in facilitating biodiversity adaptation to climate change, though they are still likely to be important in conservation (Von Maltitz *et al.* 2006; Hannah 2008; Mackey *et al.* 2008; Rahel, Bierwagen and Taniguchi 2008; Heller and Zavaleta 2009). For example, vegetation modelling projects that 37-48 per cent of Canada's protected areas could experience a change in terrestrial biome type under doubled atmospheric carbon dioxide conditions (Lemieux and Scott 2005). Similarly, bioclimatic envelope models project a decline in north-west Europe in Natura 2000 sites of habitat suitable to support many of the species they currently protect (Vos *et al.* 2008). The first quantitative study of the exposure of the global protected area network to climate change has suggested that similar patterns are likely to hold true at a global scale (Lee and Jetz 2008).

Therefore, considerable effort will need to be devoted to expanding and redesigning protected areas systems to ensure that they include sufficient area to accommodate management practices that both facilitate change and maintain large populations of species of concern (Huntley 2007). Additional criteria and approaches for consideration in redesigning protected areas systems include:

i) that they should contain large enough core areas of ecosystems that will be relatively un-affected by climate change, which can serve as refugia from changing conditions (Julius and West 2007; Mitchell *et al.* 2007; CCSP 2008; Heinz 2008; Mackey *et al.* 2008; Vos *et al.* 2008);
ii) that they include stationary or displaced refugia of species of concern as projected by climate change (Araujo *et al.* 2004); and
iii) that habitat heterogeneity should be favored, including as far as feasible a full range of combinations of environmental conditions (Huntley 2007).

It has also been suggested that expanding reserves preferentially towards the poles and higher altitudes might provide greater scope for adaptation to climate change (Li, Krauchi and Gao 2006), but other authors have pointed out that in many regions, the options for doing this are severely limited by the availability of space and resources (Huntley 2007). Further, design of protected areas systems should consider questions of functional connectivity (see below), take advantage of 'buffer zones' to increase the effective size of reserves (Huntley 2007; Mitchell *et al.* 2007), link habitats in new suitable climate zones with existing relatively 'climate-proof' refugia and include diverse reserve management strategies (see below) (Williams *et al.* 2005; CCSP 2008; Vos *et al.* 2008).

One recent study emphasising the importance of connectivity has suggested that expanding protected area networks could delay loss of species representation under climate change until the middle of the century (Hannah 2008). Unfortunately, there is as yet little concrete evidence on how protected areas will perform in the face of climate change (Heinz 2008). The problem is still greater in the case of the marine protected areas (MPAs), where planning in the context of climate change is relatively recent (McLeod *et al.* 2009). One study found that existing no-take marine protected areas had no positive effect on the response of reef ecosystems to large-scale climate-related disturbance (Graham *et al.* 2008). Although there are expectations that MPAs will promote resilience and faster recovery from climate disturbance, site-specific studies suggest this may not be the case; the effectiveness of such management needs to be assessed across regional spatial scales (Graham *et al.* 2008). A further concern is that reserve expansion is a very expensive option (Von Maltitz *et al.* 2006).

The likely effects of climate change on protected areas systems raises the question of whether these networks should be regarded as fixed in space and time, or whether provision should be made for movement of protected areas boundaries (Pressey *et al.* 2007; Hannah 2008). Precedents exist in the form of areas that currently receive seasonal protection or where temporary restrictions on resource extraction (e.g. fisheries) are imposed. Movable protection is particularly relevant for marine systems where frontal zones and currents are likely to shift with climate change and where the areas involved are potentially enormous (Hannah 2008). The existing concept of 'adaptable' protected areas, whereby conservation status could be applied or removed as an area becomes more or less valuable as species habitat, could be relevant to climate change (Berry *et al.* 2008). The management objectives of individual protected areas and of whole systems will also need to be dynamic, changing as their composition changes over time (Huntley 2007; Mitchell *et al.* 2007), and even the concept of what constitutes a native species may need to be reconsidered (Huntley 2007). On the one hand, it may be appropriate to plan protected areas networks and their management in terms of 'potential native species' (Huntley 2007), and on the other, the arrival of some species that are in no traditional sense introduced but are better suited to new conditions than relict species from earlier conditions may lead to interactions and impacts not unlike those associated with invasive alien species (Dunlop and Brown 2008).

2.5.2.1.2 Protected area management

Ensuring the continued survival of ecosystems and species under changing climatic conditions requires not only adjustments to the extent and location of protected areas, but also changes in the ways in which they are managed. It is important that reserve management be adaptive and as well informed as possible by an understanding of the likely impacts of climate change (Brooker, Young and Watt 2007; Hopkins *et al.* 2007; Mitchell *et al.* 2007; Backlund *et al.* 2008; Killeen and Solorzano 2008; Ptato 2008). However, at least in some regions, reserve managers are not aware of likely climate change impacts on their reserves (Schliep *et al.* 2008). Management changes will be needed both to minimise the direct impacts of climate change on protected ecosystems and to reduce other threats not directly linked to climate change (Pearsal 2005; Noss 2008). Moreover, as species shift ranges, management of these areas may need to change to cater for these species.

Managing for reduced climate change impacts will include actions to preserve ecosystem processes such as regeneration and succession, which help to maintain seed sources and favourable microclimates for germination and establishment of new seedlings. In some cases it may also include active restoration of degraded habitats, which may also help to increase the effective size of the reserve (Julius and West 2007; Millar *et al.* 2007; Heinz 2008).

Reducing threats not linked to climate change also needs to be a key goal of protected area management (Fischlin *et al.* 2007; Huntley 2007; Mitchell *et al.* 2007; Dunlop and Brown 2008; Heinz 2008) so that resilience of populations can be maximised and ecosystem function can be maintained. Such threats include over-exploitation of resources, eutrophication and invasive alien species. Assessing and improving the effectiveness of protected area management (Hockings, Stolton and Dudley 2004) will be critical in dealing with these threats, as will integrating protected area management with management of the wider landscape, which may often play a major role in regulating the influence of such pressures (see below). Wildfires also need to be appropriately managed. Although in some cases they are a distinct threat to protected areas, some ecosystems depend on natural fire regimes (Berry *et al.* 2008). The incidence and intensity of wildfire and the impacts of invasive alien species are likely to be further exacerbated by climate change (Dunlop and Brown 2008), so the management objectives and practical management regimes of individual protected areas will need to be dynamic, and to change as the area's composition changes over time (Huntley 2007; Mitchell *et al.* 2007). Many authors emphasise the importance of adaptive management in maintaining the effectiveness of reserves under changing climatic conditions (Hopkins *et al.* 2007; Mitchell *et al.* 2007; Heinz 2008; Heller and Zavaleta 2009).

The management of buffer zones around protected areas is seen as an important tool for maintaining the integrity of protected areas and helping to ensure the continued functionality of their ecosystems and the delivery of ecosystem services, such as water yield regulation, that may be important in societal adaptation to climate change (Huntley 2007; Mitchell *et al.* 2007; Heller and Zavaleta 2009). However, it is important to recognise that their principle role is in increasing protected area effectiveness rather than in contributing directly to adaptation to climate change (Huntley 2007).

2.5.2.1.3 Functional connectivity

Numerous authors emphasise the importance of ensuring functional connectivity among natural areas in facilitating movement of species and their adaptation to climate change (Huntley 2007; Heinz 2008; Glick *et al.* 2009; Heller and Zavaleta 2009). For example, habitat connectivity has been identified as a particularly important adaptation strategy for many forest species (Roy and de Blois 2008), and could enhance the diversity and resilience of forest ecosystems to climate change (Chapin *et al.* 2007; Millar *et al.* 2007).

Upstream-downstream connectivity in rivers and water courses is also very important (Hopkins *et al.* 2007).

In many cases, improved connectivity is interpreted to mean the use of continuous habitat corridors, to reduce habitat fragmentation (Matisziw and Murray 2009), both in conjunction with protected areas and as part of broader habitat management (Hannah 2008). They are gaining increasing attention as a tool to facilitate the migration of species, as they could allow species to track environmental changes (Gayton 2008; Rahel *et al.* 2008; Roy and de Blois 2008; Glick *et al.* 2009). However, it is difficult to predict the utility of habitat corridors and the movements of individual species with confidence (Heinz 2008), particularly as the nature and utility of corridors varies greatly among species (Donald 2005; Donoghue 2008; Kettunen *et al.* 2009). In Costa Rica, researchers found that different bird species had different preferences for riverine forest corridors or hedgerows as avenues for movement among habitat patches (Gillies and Clair 2008). A recent review of work on hedgerows found that although some species use hedgerows as corridors, the benefits could not be adequately assessed even at the small scale, and the role of corridors at the landscape level for adaptation to climate change is even less understood (Davies and Pullin 2007). Some authors caution against the justification of large-scale corridors on grounds of climate change, since migration along corridors by standard dispersal mechanisms is unlikely to keep pace with projected change for many species (Pearson and Dawson 2005).

In theory, to be functional corridors would need to span environmental gradients and be a part of broader landscape planning to ensure that they are not threatened by planned infrastructure (Killeen and Solorzano 2008). Many authors sound cautionary notes about the feasibility of establishing such continuous habitat corridors in many situations (Huntley 2007; Hopkins *et al.* 2007; Mitchell *et al.* 2007), and point to the concept of stepping stones of natural and semi-natural areas and to management of the wider landscape to increase its 'permeability' to wildlife as being much more relevant (Von Maltitz *et al.* 2006; and see below). There is also more general concern that increasing connectivity should not be seen as a substitute for the conservation of large core areas of high quality habitat (Hulme 2005; Hodgson *et al.* 2009).

2.5.2.1.4 Management of the wider landscape

In addition to improving protected areas and their management, and enhancing connectivity among them, improved planning and management of the wider landscape is agreed to be fundamental to adaptation strategies in biodiversity conservation (Hopkins *et al.* 2007; Heinz 2008; Mackey *et al.* 2008; Wilson and Piper 2008; Heller and Zavaleta 2009). One goal of such improvements is to make the matrix around reserves more attractive to wildlife and therefore more permeable to species movements, which is expected to facilitate their dispersal (Chapman *et al.* 2003; Donald and Evans 2006; Hopkins *et al.* 2007; Mitchell *et al.* 2007). Many studies recommend 'softening' land-use practice in the matrix around reserves (Heller and Zavaleta 2009), but provide relatively little detail as to what such changes in practice might entail. In general more diversity friendly practices might include lower intensity farming with reducing agrochemical use (Berry *et al.* 2008), planting and restoration of hedgerows, management and restoration of ditches and ponds and maintenance of field margins and summer fallows (Donald and Evans 2006). In the tropics, agroforestry has great potential to increase the permeability of agricultural landscape (Villamore and Lasco 2008). In non-agricultural landscapes, reduced impact logging, restoration and fire management will be important to maintain forest integrity and increase landscape diversity (Guariguata *et al.* 2008). Management of upland streams to enhance resilience of freshwater bodies will also be important (Conlan *et al.* 2007).

As for protected areas management, management of the wider landscape will need to be done in adaptive fashion to enable it to take account of changes in climate and other conditions (Hopkins *et al.* 2007; Von Maltitz *et al.* 2007; Heinz 2008). Agri-environment schemes are one mechanism to promote such management, and they have the advantage that they are already in use and are adjusted regularly to take account of changing conditions and emerging needs.

These approaches provide multiple advantages in the context of climate change. They increase the amount of habitat available to species that can actually use the matrix, they increase the functional connectivity of landscapes for species that might need to disperse across them, and they reduce many threats not directly linked to climate change. Furthermore, in many cases they will increase the ability of the landscape to provide ecosystem services such as water yield, timber provision, pollination and pest control (Harris *et al.* 2006; Hannah 2008), and which could support societal adaptation (see section 2.3). In many cases they will also enhance carbon storage, providing a strong link between strategies for adaptation to climate change and those for mitigating it.

2.5.2.2 Species

Adaptation strategies for species are crucial as they represent the building blocks of ecosystems. Species responses will ultimately determine the ability of ecosystems to adjust and persist under changed climates (Gayton 2008). There are multiple adaptation intervention options available that are applicable to species. Excluding evolutionary adaptations (discussed under autonomous species adaptation above), species have been classified into four functional groups based on their response to climate change (Von Maltitz *et al.* 2007), which each require different adaptation strategies:

- ***persisters*** are tolerant to the new climate of their current location (dealt with in autonomous adaptation);

- ***obligatory dispersers*** physically move with the changing climate to track suitable climates as current range become unsuitable, either by dispersing autonomously (see above) or requiring assistance by human assisted translocation or dispersal into suitable habitats;
- ***range expanders*** continue to occupy parts of current range and expand into newly climatically suitable areas that are not currently available but to which the species are already well adapted; these species require either no intervention or if they become invasive need to be controlled
- ***no-hopers*** cannot do any of the above and will become prematurely extinct, although may persist under unsuitable climates for some time and might be maintained by *ex situ* conservation.

2.5.2.2.1 In situ *adaptation measures*

In situ conservation measures for species have not been well researched in the context of climate change adaptation, although *in situ* methods are a common conservation strategy. The approach is to increase the resilience of existing ecosystems and species in their current locations through site-based management, restoration and reduction of pressures from sources other than climate change. Removing non-climate pressures from species might give species more flexibility to evolve and adapt to climate change. Fischlin *et al.* (2007) and Heinz (2008) note that this may be the only practical large-scale adaptation policy for migratory species and marine systems. Habitat restoration might provide food and habitat for species, e.g. blocking drainage ditches on peatlands should raise water levels and reduce the vulnerability of cranefly populations to increased temperatures and summer desiccation, and therefore benefit a range of bird species (James Pearce Higgins pers.comm.). Food provision at feeding stations (e.g. urban bird feeders) might give species flexibility to adapt to climate-related pressures. Similarly, controlled fire management, reduction of fragmentation and other habitat managements might positively affect some species. However, there are many pressures affecting species and only a select few could be tackled with limited resources. Further complex species interactions need to be carefully considered before modifying habitats, providing food or changing fire regimes, to avoid negative consequences on other species.

2.5.2.2.2 Human-aided translocation

Translocation, also referred to as assisted dispersal, migration, or colonization, involves facilitating the movement of animals, plants and other organisms from sites that are becoming unsuitable due to global climate change to other sites where conditions are thought to be more favourable for their continued existence. Translocation has been suggested as one option to facilitate the movement of species into climatically suitable areas where the timescale or habitat fragmentation prevents their ability to move naturally (Glick *et al.* 2009) and has been recommended in US and UK adaptation strategies (Julius and West 2007; Mitchell *et al.* 2007).

Assisted migration can take a number of forms. Planting seedlings adapted to future climates is recognised as a key adaptation strategy in the forestry sector (O'Neill *et al.* 2008). It is also argued that commercial plant nurseries are a form of assisted migration e,g. in Europe, 73 per cent of native species investigated had commercial northern range limits that exceeded their natural range limits, which could provide a 'head start' on migration (Veken *et al.* 2008). More extreme forms of assisted migration involve the movement of species into areas that they had not previously inhabited, but that will now be climatically suitable. Species with small populations, fragmented ranges, low fecundity, or those suffering declines due to introduced pests or diseases could be candidates for facilitated migration (Aitken *et al.* 2008). Tested translocation techniques are available for many vertebrate species and some invertebrates though have mainly been undertaken for re-introduction (for examples see Heinz 2008).

However, translocations may have undesirable consequences and opinion is divided as to whether species should intentionally be moved out of their current range and into another area (Mueller and Hellmann 2008; Veken *et al.* 2008). The most controversial aspect is the potential impact on the ecosystem into which the species will be moved. One risk with translocation is that the species could turn invasive. The Monterey Pine which was confined to narrow sections of the California coast, was translocated to South America and has spread to Chile, New Zealand, Australia and South Africa (Fox 2007). By comparing past intracontinental and intercontinental invasions in the United States, Mueller and Hellman (2008) show that the risk of translocation to create novel invasive species is small, but translocated species that do become invasive could have severe effects, particularly fish and crustacean intracontinental invasions (Mueller and Hellmann 2008). Another issue is that translocations may fail, potentially resulting in extinctions. The lack of detailed knowledge about the species and limitations of existing models make it difficult to predict optimal future locations. Different models might give different projections, e.g. comparing static vs. dynamic models for carnivores in North America (Carroll 2007), climate vs. climate-habitat models for birds in Spain (Suarez-Seoane, Osborne and Rosema 2004), and even different populations within a species may respond differently (Tolimieri and Levin 2004). Furthermore, Beale *et al*. (2008) suggest that climate might not determine the distribution of all species. The characteristics of both the species and the translocation sites (see section on ecosystems above) need to be carefully considered (Hunter 2007), and many studies fail to adequately research the ecological requirements, community interactions, and genetic diversity of the species (McLachlan, Hellmann and Schwartz 2007).

In all such cases, the advantages and disadvantages of translocation need to be carefully assessed and decisions should consider the best option to minimise species loss under climate change as well as options to facilitate natural population spread, along with an awareness of unintended consequences (McLachlan *et al.* 2007; Hoegh-Guldberg *et al.* 2008). Hoegh-Guldberg *et al*. (2008) have developed a decision framework which can be used to outline potential actions for assisted colonization under a suite of possible future climate scenarios.

2.5.2.2.3 Ex situ *measures: captive breeding and germplasm banks*

Ex situ conservation measures would initiate captive maintenance programs for species that would otherwise become extinct due to climate change. They are not as preferable as the above options, but might be the last resort for the "no-hopers" (Von Maltitz *et al.* 2006). Captive rearing, husbandry and propagation methods have been described for many animals and plants; and zoos, aquaria, botanic gardens and seed banks are well established. Studies have investigated the potential for captive breeding of species and of gene banks, but few with explicit links to climate change.

For some species, captive breeding has been successful, e.g. Scimitar-horned oryx, now considered extinct in the wild, persists in large numbers in captivity (Iyengar *et al.* 2007). However, for other species, such as Arctic marine mammals, captive breeding as an adaptation option is largely unfeasible (Ragen, Huntington and Hovelsrud 2008) and should be seen as a last resort for populations approaching extinction, e.g. South African critically endangered riverine rabbit (*Bunolagus monticularis*) (Hughes *et al.* 2008). Previous non-climate related reviews of captive breeding have similarly suggested that this is a resource demanding and technically difficult activity, mainly restricted to vertebrates, and should be a last resort (Ayyad 2003), particularly given the low rates of success reported for many species, and the fact that it shifts attention away from *in situ* preservation of habitats (Hughes *et al.* 2008). Further, removal of all individuals into captivity would cause species to go extinct in the wild, with potentially severe consequences for the species' native ecosystem and their functioning. Another cause for concern is that captive breeding can reduce genetic diversity (Berry *et al.* 2008).

Genetic diversity might potentially be captured in banks storing germplasm, such as seeds, eggs and sperm. The maintenance of genetic seed banks could complement *in situ* measures to buffer against extinction threats from climate change and provide a source of germplasm for future restoration and research (Simpson and Wang 2007). Plants have commonly been stored in seed banks, and an objective of the Convention on Biological Diversity is that 60 per cent of threatened plant species should be held in accessible *ex situ* collections (Target 8, Global Strategy for Plant Conservation 2006). The UK Millennium Seed Bank Project aims to have banked seed from 10 per cent of the world's wild plant species by the end of the decade and Austria has maintained a network of genetic reserves distributed along environmental gradients (Geburek and Muller 2006). Maintaining genetic diversity of ecosystems is likely to be particularly important in maintaining resilience to climate change, and seed banks could contribute to this (Kleinschmit 2002). A recent study suggests that plant species endangered due to habitat destruction and climate change can be effectively and efficiently propagated *ex situ* (Millner *et al.* 2008), and the creation of gene banks has been suggested as an adaptation strategy for oaks and pines in Mexico (Gomez-Mendoza and Arriaga 2007) and plant species in China (Li and Xia 2004), as a complement to habitat conservation. The inclusion of crop wild relatives in seed banks could also contribute to agricultural adaptation (Jarvis *et al.* 2008).

Captive breeding and germplasm banks require large resources for their maintenance and are therefore unlikely long-term strategies for more than a few species. Further, the ecosystems might become so altered that reintroduction of species back into the wild becomes unfeasible, consigning these species to become "living fossils" (Heinz 2008).

2.5.2.3 Genes

High levels of genetic diversity within populations are desirable to ensure adaptability. As genetic diversity is correlated with population size and diversity, adaptation should strive to maintain or create large populations. Further gene flow between populations might be desirable, but mixing could also swamp local adaptation and result in homogenisation (Gregory *et al.* 2006). The adaptation strategies outlined above for ecosystems and species with their advantages and disadvantages are likely to apply similarly to genes. The causes and consequences of the maintenance and loss of climate-related genetic diversity within populations are currently poorly understood and require further research (Jump and Penuelas 2005).

2.5.3 Implementation considerations

Actions in the conservation sector should by definition have a positive impact on biodiversity. However, conservation following "business as usual" under future climate change can have negative impacts (Hannah *et al.* 2002). Conservation management in the context of climate change will need to identify in the short term effective actions that can improve the abilities of ecosystems and species to accommodate and adapt to climate change in the medium and longer term, despite the many uncertainties that still exist and the lack of concrete information about the effects of different management actions (Heller and Zavaleta 2008). As a whole, the sector is likely to draw on a wide range of options including expansion and alteration of protected areas systems and changes to their management, enhancing the functional connectivity between ecosystems through the use of corridors, stepping stones and wildlife friendly management of the wider landscape, reducing the impacts of pressures not linked to climate change, and managing species directly to enhance their ability to persist and to shift their ranges in response to climate change. While most of these adaptation measures will benefit some species if carefully implemented, there is a danger that some will have secondary consequences affecting biodiversity negatively, e.g. invasive species due to translocation. Therefore, careful consideration is required to minimise potential negative consequences before adaptation measures are implemented. Enhancing the resilience of biodiversity to the impacts of climate change is likely to be important both for societal adaptation and for mitigation.

2.6 SYNERGIES AND TRADE-OFFS BETWEEN ADAPTATION AND MITIGATION

Adaptation measures required will depend on the scale of the impacts. Adaptation is therefore closely related to mitigation, and some recognition of the synergies and trade-offs between adaptation and mitigation strategies is required (Ayers and Huq 2008). The IPCC AR4 reported that there was inadequate literature on the relationship between adaptation and mitigation policy (Adger *et al.* 2007) and this area is only recently beginning to be explored (Nyong, Adesina and Elasha 2007). In some cases, adaptation measures can contribute to mitigation, whereas in others they may run contrary to each other (Berry *et al.* 2008). It is important to recognise the areas in which trade-offs need to be made (Harper 2008), as well as to identify 'win-win' solutions. It has been suggested that natural resource management is one of the areas with the greatest potential for achieving the objectives of both adaptation and mitigation, due to the major role that ecosystems play in the carbon cycle (as reviewed in the background documents for the first meeting of the Second AHTEG on Biodiversity and Climate Change (Campbell *et al.* 2008)), and in underpinning adaptation strategies (Ravindranath 2007). Desertification, biodiversity and climate change are dealt with as separate issues under the international convention, when in fact they all interact (Eriksen *et al.* 2006; Cowie, Schneider and Montanarella 2007).

REDD is commonly identified in the literature as a strategy with the potential to support both adaptation and mitigation, whilst providing significant biodiversity benefits (Murdiyarso *et al.* 2005; Nabuurs *et al.* 2007; Ravindranath 2007; Righelato and Spracklen 2007; Eliasch 2008; Locatelli *et al.* 2008; Nepstad *et al.* 2008). Soil and water conservation through good agricultural practice and agroforestry can reduce carbon loss and enhance soil organic matter to reduce the vulnerability to drought and flooding (Nyong *et al.* 2007; Ravindranath 2007; Rosenzweig and Tubiello 2007; Verchot *et al.* 2007; Berry *et al.* 2008; Lal 2008). Planting species mixtures can stabilize soil, reduce flooding, and improve the adaptive capacity of forest plantations in the long term (Berry *et al.* 2008), and mangrove plantations can build resilience to coastal storms and also sequester carbon (Ayers and Huq 2008).

It is clear that there are significant areas of overlap between adaptation and mitigation. However, there are also trade-offs to be made. Water resources can be directly impacted by forestry mitigation activities where appropriate species are not used (Betts 2007). Adaptation options in the water sector can involve draining wetlands, turning them into a net source of emissions (Mata and Budhooram 2007). This area would appear to require further research. Any adaptation option that involves the loss and degradation of natural ecosystems can result in greenhouse gas emissions, and may result in maladaptation in the long term.

2.7 CONCLUSION

Adaptation to climate change is a relatively new field, and the literature available in this area is limited. Very few adaptation strategies have actually been implemented, but those that have tend to rely on technological and engineering measures. The limited evidence to date suggests that although technological and structural adaptation measures will be required, biodiversity will also play a vital role in adaptation to climate change.

The evidence presented here suggests that ecosystem-based adaptation can be a cost-effective strategy to address the impacts of climate change, particularly in vulnerable areas where adaptive capacity is low. Indeed, many of the examples to date have been linked to community-based adaptation, where local communities that rely directly on natural resources can increase their adaptive capacity through good management of their natural resource base (Huq *et al.* 2005). The lack of cost-benefit analyses of the different adaptation options make conclusions tentative, and most available evidence is anecdotal or based on case studies. However, it is clear that coastal ecosystems can play a role in coastal protection and buffer the impacts of storms while maintaining fish supplies; natural wetlands and rivers are vital in water adaptation; and forests play a role in water regulation and soil conservation whilst maintaining livelihood options. Crop diversity and good agricultural practice are likely to play a large role in agricultural adaptation. This is not just important for the poor, but for society as a whole.

The term 'ecosystem-based adaptation', although it has been used here, can give the impression that adaptation based on biodiversity is completely separate from other more structural measures; and that adaptation strategies are either ecosystem-based or structural. In fact, optimal adaptation strategies often involve the incorporation of biodiversity into wider adaptation planning as a complement to, rather than an alternative to, structural measures. Indeed the importance of adopting an integrated approach that incorporates adaptation measures that are based on biodiversity is highlighted throughout the literature.

Furthermore, climate change impacts can be exacerbated by management practices, such as the development of seawalls, flood management and fire management, that do not consider other sectors such as biodiversity conservation and water resource management; this results in maladaptation in the longer term (Hulme 2005; World Bank 2008). In addition, the use of technology and infrastructure can 'lock in adaptation' to a specific impact, whereas the incorporation of 'soft' adaptation measures, including land-use planning, natural resource management, and building social adaptive capacity, can allow for flexible responses (Kirshen *et al.* 2008; Matthews and Quesne 2008; Koch *et al.* 2009). Integration is required not just between biodiversity-based adaptation and technological measures, but also across different adaptation sectors, and will require significant institutional support.

Climate change is already having measurable impacts on ecosystems and on biodiversity more generally, and these are expected to grow. Adaptation in the conservation sector is required, not just to achieve the conservation of biodiversity for its own sake, but to maintain the role of biodiversity in contributing to societal adaptation. Adaptation strategies in the conservation sector are still in the early stages of development. They include factoring climate change into protected area design, managing the wider landscape to ensure functional connectivity between habitats, and reducing other pressures on ecosystems. Careful consideration of adaptation options such as assisted migration is required, as actions to improve the conservation status of one species might have wider impacts on biodiversity. More guidance is required on how to build resilience to climate change in ecosystems and species, particularly in developing countries where many people are directly reliant upon their natural resources.

Increasing the resilience of ecosystems to climate change also supports their role in climate change mitigation. The linkages between mitigation and adaptation are only beginning to be explored, but it is clear that natural resource management is one of the areas with the greatest potential for synergies. It is also an area in which trade-offs can exist. Managing the trade-offs and promoting the synergies between adaptation and mitigation in the land-use sector is likely to be important both in adaptation to climate change, and in limiting climate change to a level at which it is still possible to adapt.

Although we have separated this report into three sections, considering the role of biodiversity in societal adaptation, the impacts of adaptation strategies on biodiversity, and adaptation in the biodiversity conservation sector, it is clear that all three are interlinked. Ultimately, a broader perspective is required that focuses on how ecosystems can be managed and conserved in order to deliver ecosystem goods and services in a changing climate, within the context of overall adaptation policy.

The coverage of costs and benefit analyses across adaptation options is uneven, and further research is required in this area. There needs to be greater consideration of synergies and trade-offs in adaptation policy and planning, including improved understanding of the underpinning role of biodiversity, to avoid maladaptation and develop cost-effective responses to the impacts of climate change.

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SECTION 3

THE LINKAGES BETWEEN BIODIVERSITY AND CLIMATE CHANGE MITIGATION

A review of the recent scientific literature

Alison Campbell, Anna Chenery, Lauren Coad, Nathalie Doswald, Valerie Kapos, Francine Kershaw, Jörn Scharlemann and Barney Dickson

CONTENTS

3.1 EXECUTIVE SUMMARY

The Intergovernmental Panel on Climate Change Fourth Assessment Report (IPCC AR4) provided growing evidence of the importance of natural ecosystems in the carbon cycle and in mitigation policies. In addition, it was recognised that climate change mitigation policies focused on reducing carbon dioxide (CO_2) emissions can have impacts on biodiversity; both positive and negative.

Research since IPCC AR4 has served to strengthen the conclusion that biodiversity is important in mitigating climate change. This importance stems from the role of ecosystems in the carbon cycle. Ecosystems assimilate carbon dioxide from the atmosphere and then store it. Human-induced changes in those ecosystems can lead either to increased sequestration of carbon dioxide or to increased emissions of carbon dioxide and other greenhouse gases (GHGs). Promoting the former type of change and reducing the latter type of change can make a very significant contribution to climate change mitigation and promotes climate regulation services. The use of ecosystem-based mitigation policies can also contribute to sustaining a variety of ecosystem services including biodiversity conservation.

Ecosystem services include climate regulatory services such as carbon storage and sequestration, which form an important part of climate change mitigation. A number of these natural processes are being considered or used as part of mitigation policies. However, mitigation policies may also impact on certain ecosystem services including biodiversity.

There is considerable uncertainty about the volume of carbon stored in terrestrial and marine ecosystems. A recent study has estimated that over 2,000 gigatonnes carbon (GtC) is stored in terrestrial ecosystems, but this figure is likely to be an under-estimate. It has been estimated that terrestrial ecosystems sequester 2.1-3 GtC of atmospheric carbon annually, approximately 30 per cent of all anthropogenic CO_2 emissions. Marine ecosystems sequester large amounts of carbon through phytoplankton at the ocean surface, accounting for approximately 50 per cent of the global ecosystem uptake of CO_2.

The IPCC AR4 reported that 20 per cent of anthropogenic GHG emissions come from the loss of terrestrial ecosystem carbon stores through land-use change, primarily deforestation. This is equivalent to approximately 1.5 GtC per year. Uncertainty surrounding estimates of emissions from tropical forest deforestation remains high and the figure of 1.5-1.6 GtC per year remains the default value. It is widely agreed that estimating emissions from forest degradation is more difficult. Some estimate that forest damage from logging in the Amazon results in a 15 per cent reduction in carbon stocks, with increased susceptibility to fire damage releasing an additional 20 per cent of forest carbon.

Loss of carbon from soils due to land-use change is also difficult to assess, but is likely to be considerable. It has been estimated that soils lose carbon at the rate of approximately 1.6 GtC per year, almost identical to that lost through deforestation. Much of these soil-based emissions come from peat degradation. Human disturbances such as drainage for agriculture or forestry have transformed peatland from a sink to a source in large areas. Drainage and drying of peat also facilitates fires. In combination, these processes are estimated to result in the loss of 3 $GtCO_2$ to the atmosphere every year, or 10 per cent of global emissions.

The feedbacks from natural ecosystems due to a warming climate highlight the complex relationship between biodiversity and the carbon cycle. New observations on dampening of the carbon sink capacity are challenging the hypothesis that the carbon sequestration will be enhanced with climate change induced increases in net primary productivity.

The IPCC AR4 estimated that over the next century 345-1269 $GtCO_2$ equivalent could be abated through land-use based mitigation policies. This is about 15-40 per cent of total abatement requirements (to 445-490ppm) and could be realised through a combination of reduced loss of carbon stores, and sequestration policies. Since emissions from deforestation amount to 1.5 GtC per year, there appears to be high potential for cost-effective emissions reductions from a mechanism for Reduced Emissions from Deforestation and Degradation (REDD). This mechanism is currently in a demonstration phase in the United Nations Framework Convention on Climate Change (UNFCCC). It has been estimated that a well designed REDD mechanism could reduce deforestation rates by up to 75 per cent in 2030, and in combination with afforestation, reforestation and restoration, could make the forest sector carbon neutral. Economic modelling has suggested that REDD will be a competitive, low-cost abatement option. Moreover, a successful REDD mechanism has the potential to deliver significant additional benefits, contributing to biodiversity conservation at both the species and ecosystem level, whilst also supporting the maintenance of ecosystem services.

There is significant uncertainty attached to the level of carbon sequestration that can be achieved through afforestation and reforestation; and the potential for mitigation in this sector, particularly on decadal time scales, is often questioned. Whilst there is significant potential in increasing the capacity of the natural carbon sink, particularly in the tropics, there is a need for more integrated study of how land management changes may affect climate change. Sequestration schemes often require a trade-off; production forest results in higher carbon benefits but fewer biodiversity benefits, whereas multifunctional forest can have biodiversity benefits but is of lower sequestration value.

The role of improved soil management in climate change mitigation should be emphasised as it is the area with the highest potential outside of forest activities. Global soil organic carbon has a sequestration potential of 0.6-1.2 GtC with high levels of carbon stocks, much of which is contained under natural ecosystems rather than managed ecosystems. Whilst estimates of carbon storage in peat soil are still uncertain, largely due to lack of information on peat depth and density, advances are being made in this respect. A new estimate of 5 GtC stored in Indonesian peat utilises remote sensing technology supported by ground-based observations. The reduction in the rate of current peat degradation in Indonesia therefore has the potential to reduce emissions significantly, particularly as the degradation of peat soils is accelerating. Boreal regions have significant areas of peatland, acting as a large carbon sink. But there is peat degradation there too. Many peat bogs in Europe have been drained and are being restored and over 55 per cent of peatland area in Finland has been drained. Currently, there is very limited scope for inclusion of wetland or peatland in carbon accounting through the UNFCCC, and no direct mention in the text. The only option for inclusion in carbon accounting is where conversion of wetland areas is captured through management practices of other ecosystems, such as for forested peatland.

Geo-engineering techniques for mitigating climate change are not strictly ‘ecosystem-based’, but they do involve manipulation of the natural environment, particularly the marine environment, to increase the carbon storage and sequestration capacity, and this may have impacts on biodiversity. The technique with the most promise for mitigation is carbon capture and storage. This may involve the injection of CO_2 into the deepwater and this will alter ocean chemistry and could have significant consequences for marine organisms and ecosystems in the deep sea.

Renewable energy projects can also have impacts on biodiversity. Biofuel production has considerable impacts on biodiversity when it results in direct conversion of natural ecosystems and indirect displacement of agricultural land into natural ecosystems. Birds can be affected by wind turbines though collision with turbine blades, displacement from migration routes, and direct habitat loss. Mortality of birds as a result of wind turbines has been documented by a number of recent studies although some have argued that wind farm impact studies lack an evidence base and have minimal impacts on biodiversity. The biodiversity impacts of hydro-electric dams include habitat destruction, barriers to terrestrial migration, barriers to fish migration, reduced sedimentation and changes in flow altering downstream ecosystems, and fish mortality in turbines.

It is clear from the literature reviewed that climate change mitigation policy has the potential to impact biodiversity both positively and negatively. Currently, many renewable energy projects are being planned without consideration of biodiversity impacts; as are some land-based mitigation strategies such as monoculture plantations. There is also need to consider climate change adaptation strategies in which biodiversity also has an important role. However, due to the important role of ecosystems in the carbon cycle, it is clear that the potential exists to develop ‘win-win’ mitigation policies that are beneficial for both climate change mitigation and biodiversity. Moreover, such policies can be aligned with adaptation strategies to generate a ‘triple win’ policy.

3.2 INTRODUCTION

The overall objective of the United Nations Framework Convention on Climate Change (UNFCCC) is the *'stabilization of greenhouse gas concentrations in the atmosphere at a level that would prevent dangerous anthropogenic interference with the climate system'* (UNFCCC Article 2). In order to achieve this, global average temperatures should not increase more than 2ºC relative to pre-industrial levels; requiring a 60-80 per cent reduction in greenhouse gas (GHG) emissions by 2050 to stabilise atmospheric concentrations at 445-490ppm carbon dioxide equivalent (CO_2e; IPCC 2007). Therefore, mitigation efforts are required across all sectors, including through efforts to reduce emissions from land-use change and increase the capacity of the natural carbon sink, i.e. parts of the biosphere that can store carbon. Ecosystems, as an integral part of biodiversity, play an important role in the carbon cycle and therefore are important in the discussion on mitigation.

An increase in global average temperature of 0.7ºC has already been observed, with associated impacts on natural ecosystems and the services that they provide. Increasing temperatures are causing rising sea levels, melting sea ice, altered precipitation patterns and fire regimes, and are likely causing the altered frequency and severity of extreme events such as drought, heat waves, and hurricanes. Such impacts will have significant implications for human welfare (Stern 2007).

The Intergovernmental Panel on Climate Change (IPCC) Special Report on Emissions Scenarios (SRES) (Nakicenovic and Swart 2000) developed a number of narratives on how the world might develop in the 21st century, exploring impacts on global emissions if demographic, social, economic, technological, and environmental developments take specific directions at the global level. These scenarios, labelled A1, A2, B1, and B2, do not take into account the implementation of mitigation policies. Despite developments in population models since their development, the SRES projections are still considered to be representative of the range of likely outcomes (IPCC 2007). They include land-based GHG emissions as an important source throughout the century, through continued but reducing land-use change, and an increase in intensity of agricultural practices. There is, however, some evidence that emissions have been increasing at higher rates than those projected by SRES scenarios (Raupach *et al.* 2007).

In addition, feedbacks from ecosystems as a result of climate change and land-use change are significant, but are generally not incorporated into climate models because uncertainty is high (IPCC 2007). This in turn can lead to uncertainties in projections of future climate change, and therefore for the formulation of mitigation strategies (Strassmann *et al.* 2008). Despite this, the IPCC Fourth Assessment Report (AR4; IPCC 2007) provided growing evidence of the importance of natural ecosystems in the carbon cycle and therefore in mitigation policies. Furthermore, the Millennium Ecosystem Assessment (MA) carried out between 2001 and 2005 reported on the consequences of ecosystem change on human well-being. Their findings indicated that ecosystems have been degraded over the last 50 years due to anthropogenic activity and may continue to do so with significant cost to important ecosystem services (MA 2005). The MA also developed four scenarios to depict plausible future ecosystems and human well-being based on different assumptions of drivers of change; three of which take into account changes in policy (Carpenter *et al.* 2005).

In addition, it was recognised by the IPCC that climate change mitigation policies focused on reducing CO_2 emissions can have impacts on biodiversity; both positive and negative (IPCC 2007).

This report reviews the literature published after the IPCC AR4 and before October 2008 on the linkages between biodiversity and climate change mitigation policies. The aim is to highlight developments in our understanding of the role of biodiversity in climate change mitigation, and the impacts of mitigation policies on biodiversity. This review mainly focuses on the role of the carbon cycle in global climate change but it does recognise that other GHGs are important in climate change and in mitigation policies, though there is a lack of data in this sector. This review draws attention to the most important recent findings. However, it does not attempt a full synthesis of the findings across different papers. Keyword searches in ISI Web of Knowledge, Scopus, and Google Scholar were carried out to obtain a broad coverage of the available literature.

3.3 ROLE OF ECOSYSTEMS IN THE CARBON CYCLE

3.3.1 Carbon storage

Although it is known that both terrestrial and marine ecosystems constitute a significant carbon store, the exact figures are uncertain. Global estimates range from approximately 1500-2500 GtC (Cao and Woodward 1998; IPCC 2001). A recent study combining data for carbon stored in biomass (Ruesch and Gibbs 2008) with that of carbon stored in soil (IGBP-DIS 2000) has estimated that over 2,000 GtC is stored in terrestrial ecosystems (Campbell *et al.* 2008a).

A large amount of the terrestrial carbon is stored in forest systems (Eliasch 2008), but there are also significant stores in other ecosystems such as grasslands and wetlands (Table 1, *overleaf*). Carbon stored in soil accounts for a high percentage of the total terrestrial store.

Table 1: Global carbon stocks, GtC (IPCC 2001)

Biome	Vegetation	Soil	Total
Tropical forests	212	216	428
Temperate forests	59	100	159
Boreal forests	88	471	559
Tropical savannas	66	264	330
Temperate grasslands	9	295	304
Deserts and semi-deserts	8	191	199
Tundra	6	121	127
Wetlands	15	225	240
Croplands	3	128	131
Total	**466**	**2 011**	**2 477**

Carbon storage estimates to date undoubtedly underestimate the storage of carbon in soil (Table 1), particularly peat. A recent global assessment of peat has estimated that peatlands alone store 550 GtC, nearly 30 per cent of all global soil carbon, despite covering only 3 per cent of the land area (Parish *et al.* 2008), and are therefore the most important long-term terrestrial carbon store.

Tropical peatlands contain an estimated 16.5-68.5 GtC across only 8 per cent of the global peatland area, and sequester approximately 0.06-0.09 GtC per year (Page and Banks 2007). The carbon sink potential of tropical peatlands is 58 per cent of that of the temperate, boreal and subarctic peatlands combined, highlighting their importance in climate change mitigation (Lloyd 2008). Whilst estimates of carbon storage in peat soil are still uncertain, largely due to lack of information on peat depth and density, advances are being made in this respect (Jaenicke *et al.* 2008).

Boreal regions have significant areas of peatland, which act as large carbon sinks (Nilsson *et al.* 2008). The current estimates of carbon storage of boreal peats could also be an underestimate when taking into account the peat store in permafrost (Schuur *et al.* 2008). Although there is much uncertainty over the exact figure, particularly as peat depth estimates are still uncertain, this significantly increases previous estimates of the terrestrial carbon store. Recent studies have suggested that there is almost 100 GtC stored in North American Arctic soils alone (Ping *et al.* 2008); 66 per cent more than was recorded for previous estimates (Beer 2008). Indeed, Schuur *et al.* (2008) have estimated that 1672 GtC is stored in the northern circumpolar permafrost zone; equivalent to twice the atmospheric carbon pool and more than double the previous high-latitude inventory estimates. Such estimates have increased largely due to consideration of the carbon stored in peat.

As reported in the IPCC AR4, wetlands account for approximately 37 per cent of the terrestrial carbon pool (IPCC 2007). However, wetlands can act either as GHG sources or as sinks, emitting N_2O, CO_2 and CH_4 in addition to storing large amounts of carbon.

Comparatively, knowledge of carbon storage within marine environments is limited, and no equivalent literature exists. However, the total amount of carbon stored in the ocean has been estimated to be 50 times that of the atmosphere (IPCC 2001).

3.3.2 Carbon sequestration

Natural ecosystems are intrinsically linked to the carbon cycle. In addition to the historical carbon store in biomass and soils, ecosystems continually absorb atmospheric CO_2 through photosynthesis and store residual carbon in a process known as sequestration.

It has been estimated that terrestrial ecosystems sequester 2.1-3 GtC of atmospheric carbon annually (Luyssaert *et al.* 2007; Canadell and Raupach 2008), approximately 30 per cent of all anthropogenic CO_2 emissions. Much of this is realised by forest (Luyssaert *et al.* 2007); although over the past 10,000 years peatlands have sequestered an estimated 1.2 trillion tonnes of CO_2 (Parish *et al.* 2008). The Luyssaert *et al.* (2007) estimate for forest systems is based on a global database of flux observations, updated since the IPCC AR4.

Marine ecosystems sequester large amounts of carbon through phytoplankton at the ocean surface, a process that accounts for approximately 50 per cent of the global biological uptake of CO_2 (Arrigo 2007). Some of this carbon is pumped into the deepwater both through the food chain and through physical processes. The role of coastal margins is less well understood, although it is known that mangroves and seagrass sequester carbon (Yin *et al.* 2006).

3.3.3 Emissions from deforestation

The IPCC AR4 (IPCC 2007) reported that the loss of terrestrial ecosystem carbon stores through land-use change, primarily deforestation, accounts for 20 per cent of anthropogenic GHG emissions; equivalent to approximately 5.8 $GtCO_2e$ per year (or 1.5 GtC). This figure was gained from estimates for tropical deforestation in the 1990s (DeFries *et al.* 2002; Houghton 2003). Recognition of the importance of emissions from such land-use change has led to the commitment to include reducing emissions from deforestation and forest degradation in developing countries (REDD) in post-2012 commitments under the UNFCCC in the Bali Roadmap (Decision 1/CP.13; Decision 2/CP.13).

Uncertainty surrounding estimates of emissions from tropical forest deforestation remains (Achard *et al.* 2007; Olander *et al.* 2008), and the figure of 1.5-1.6 GtC per year remains the default value (Canadell and Raupach 2008). A third of these emissions come from the Amazon (Ramankutty *et al.* 2007). Recent studies have suggested that actual net tropical emissions were lower than these estimates for the 1990s (Stephens *et al.* 2007), particularly for the Brazilian Amazon, where a lower than average wood density and tree height (and therefore lower carbon stock) in the 'arc of deforestation' is not taken into account (Nogueira *et al.* 2007; Nogueira *et al.* 2008). Although this may be the case, the impact of deforestation on soil carbon is still largely unknown, with emissions dependent upon the land conversion and subsequent management practices (Murty *et al.* 2002). A disproportionate amount of deforestation in Southeast Asia, for example, takes place on peatland (Hoojier *et al.* 2006), and emissions from deforestation in this region are likely to be underestimates.

Despite the uncertainties over the exact figures, largely due to lack of data and differences in methodologies (Ramankutty *et al.* 2007), it is widely agreed that emissions from deforestation make a significant contribution to climate change (Laurance 2007; Eliasch 2008). In addition to releasing carbon stores in the atmosphere, deforestation removes the sequestration capacity of forest, reducing the ability of forest to act as a carbon sink (Stephens *et al.* 2007).

3.3.4 Emissions from forest degradation

It is widely agreed that estimating emissions from forest degradation will be more of a challenge due to the difficulties in measurement from satellite observations (Asner *et al.* 2005; DeFries *et al.* 2007). In addition, the definition of degradation is open to debate and can include unsustainable timber harvesting for commercial or subsistence use, in addition to other processes such as fire and drought; all of which lead to reductions in carbon stocks (Mollicone *et al.* 2007).

Despite these issues, the need to include degradation in the REDD mechanism is widely accepted, as was established at COP-13 of the UNFCCC in Bali. The area of degraded forest is comparable to that deforested (Asner *et al.* 2005; Feldpausch *et al.* 2005; Barreto *et al.* 2006; DeFries *et al.* 2007; Putz *et al.* 2008; Nepstad *et al.* 2008), with significant implications for carbon stocks. Asner *et al.* (2005) estimate that forest damage from logging in the Amazon results in a 15 per cent reduction in carbon stocks, and increased susceptibility to fire damage (Fearnside 2005a; Malhi *et al.* 2008) releases an additional 20 per cent of forest carbon. This estimate of 0.08 GtC lost annually from logging increases emissions estimates from deforestation in the Amazon (Defries *et al.* 2002) by 25 per cent. Indeed, it has recently been reported that clear-cut logging can release 40-60 per cent of carbon stored in vegetation (Sajwaj *et al.* 2008).

In a 'business as usual' deforestation scenario, it has been estimated that 24 per cent of the Amazon will be affected by drought and logging (Nepstad *et al.* 2008). Forest degradation can also be a precursor to deforestation (Asner *et al.* 2005; DeFries *et al.* 2007; Putz *et al.* 2008). At present, no Parties to the UNFCCC are required to report on degradation, unless forest management has been selected as an option under Article 3.4 of the Kyoto Protocol.

3.3.5 Emissions from general land-use change

Despite the current focus on emissions from deforestation and degradation, land-use changes across all ecosystems can release significant amounts of carbon into the atmosphere. Gross historical emissions from land-use change have been estimated at approximately 200 GtC (Canadell and Raupach 2008), higher than the loss estimate of 572 $GtCO_2$ reported in the IPCC AR4. Fires, of both natural and human origin, also contribute significantly to emissions, with the release of 1.7-4.1 GtC per year (Lavorel *et al.* 2007).

Although often not accurately accounted for in estimates of emissions from land-use change, a recent study has suggested that soils have lost 40-90 GtC to the atmosphere, and continue to lose carbon at rates of approximately 1.6 GtC per year (Asner *et al.* 2005; Smith 2008). Although this is similar to that lost through deforestation, there is a certain amount of overlap in the estimates, and the two figures cannot be considered additive. It has also been estimated that soil carbon accounted for 28 per cent of net loss from land-use change in the period 1850-1990 (Houghton 2005b).

Such estimates appear higher when taking peat degradation into account. Human disturbances such as drainage for agriculture or forestry have transformed wetlands and peatlands from a sink to a source in large areas (Parish *et al.* 2008; Lloyd 2008). Drainage and drying of peat also facilitates fires. In combination, these processes are estimated to result in the loss of 3 $GtCO_2$ to the atmosphere every year, or 10 per cent of global emissions. In *El Niño* years, increased fires can raise this figure (Parish *et al.* 2008). In Southeast Asia alone, emissions from peat drainage and fire average 2 $GtCO_2$ per year; equivalent to 8 per cent of global fossil fuel emissions from just 0.2 per cent of the land area (Hoojier *et al.* 2006), and almost twice the emissions from fossil fuel burning in Indonesia. Including emissions from peat in carbon accounting would raise Indonesia to third in the global emissions table, from 21st place (Hoojier *et al.* 2006; Uryu *et al.* 2008).

Conversion of natural ecosystems to agriculture can result in significant greenhouse gas emissions, through a combination of loss of stored carbon, and the large amounts of CH_4 and N_2O, as well as CO_2, emitted from agricultural practices (Berry *et al.* 2008; Lal 2008). For instance dry croplands and rice paddies accounted for nearly 30 per cent of global background N_2O emissions during the 1980-2000 period (Xu, Tian and Hui 2008). These emissions are expected to increase rapidly until the end of the century, as reported in IPCC AR4.

The knowledge base on carbon storage and emissions from soil and peatland is still small but developing, and is clearly an important area for further study. More generally, there is still a lack of information on GHG emissions from ecosystems such as grasslands and wetlands. The role of wetlands in particular is unclear (Lloyd 2008).

3.3.6 Ecosystems as ‘sinks’ or ‘sources’

As the amount of carbon sequestered by ecosystems is larger than that lost, global terrestrial ecosystems are acting as a net sink of approximately 1.5 GtC per year (the AR4 reported approximately 0.5-1.5 GtC). Sequestration at these levels would be equivalent to a 40-70ppm reduction of CO_2e in the atmosphere from anthropogenic emissions by 2100 (Canadell and Raupach 2008). Tropical forests account for a large proportion of this sink (Luyssaert *et al.* 2007), the absence of which would increase the current atmospheric CO_2 concentrations by 10 per cent (Betts *et al.* 2008a)

The exact processes involved with the oceanic carbon cycle are not well understood. However, it is clear that the ocean acts as a considerable sink; the AR4 reported that the size of the marine sink is approximately 1.8-2.6 GtC, and has increased by approximately 22 per cent from the 1980s to the 1990s. Subsequent modelling has supported this estimate of the oceanic carbon sink (Canadell *et al.* 2007b).

A recent study taking into account fluxes of the three major greenhouse gases (CO_2, CH_4 and N_2O) has strengthened these findings, suggesting a significant role of natural and relatively unmanaged ecosystems in slowing climate change through the provision of a net yearly sink of 3.55 $GtCO_2$; equating to roughly 0.5ppm atmospheric CO_2 per year (Dalal and Allen 2008). Natural ecosystems are acting as a sink for 55 per cent of anthropogenic GHG emissions (Canadell *et al.* 2007b).

Lal (2008) reports that the terrestrial sink is increasing at a net rate of 0.7 GtC per year, and is set to continue increasing due to increased CO_2 fertilisation. In addition, there is evidence that melting sea ice is increasing the sink capacity of the Arctic Ocean (Bates *et al.* 2006), and that increased CO_2 concentrations are increasing the capacity of oceanic sequestration (Luyssaert *et al.* 2007; Riebesell *et al.* 2007), although the impacts on biodiversity of an increased ocean sink have the potential to be significant through ocean acidification (Cao 2008).

However, this is not to say that *all* ecosystems are acting as carbon ‘sinks’. There is some evidence that emissions from land-use change are beginning to outweigh sequestration capacity, with the potential to reach a ‘tipping point’ whereby they will become net sources (Nepstad *et al.* 2008). Recent climate models have estimated that past land-use change, largely due to cropland and agricultural expansion, has eliminated potential future carbon sinks equivalent to emissions of 80–150 GtC over this century (Strassmann *et al.* 2008). There is evidence, for example, of a reduced sink in the Southern Ocean due to changes in circulation patterns as a result of increased temperature (Le Quere *et al.* 2007), and reductions of sinks in coastal margins through loss of vegetation (Duarte *et al.* 2005), but modelling results are still uncertain (Baker 2007). Over the past 100 years, anthropogenic impacts have turned peatlands from a net store to a source of carbon emissions (Parish *et al.* 2008).

Recent evidence of reduced sinks (Canadell *et al.* 2007a) suggests that on a global scale terrestrial ecosystems will provide a future positive feedback of uncertain magnitude, due to altered land-use practices and increasing temperatures (Heimann and Reichstein 2008).

3.3.7 Feedbacks to the climate system

Recent recognition of the scale of positive feedbacks to the climate system from land-use change and climate impacts has further increased the relevance of biodiversity to the UNFCCC objective of limiting climate change to a 2ºC rise. Although such feedbacks are not yet incorporated into global climate change projections and are still uncertain (Baker 2007), advances in this area are being made (Chapin *et al.* 2008).

3.3.7.1 Feedbacks from climate change

The feedbacks from natural ecosystems due to a warming climate highlight the complex relationship between biodiversity and the carbon cycle. New observations on dampening of the carbon sink capacity are challenging the hypothesis that carbon sequestration will be enhanced with climate change induced increases in net primary productivity (Canadell *et al.* 2007a). Carbon storage in temperate and boreal forests is mainly mediated by temperature, with greater carbon sequestration in warmer conditions (Delpierre *et al*. 2009). However, increased drought offsets these benefits (Granier *et al*. 2007; Vetter *et al*. 2008).

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 CO_2 to the atmosphere by 2100 (Canadell *et al.* 2007a). 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). Estimates for emissions from the thawing of permafrost, for example, have ranged from global increases of 100 GtC by 2100, to 40-100 GtC increases from Canada and Alaska by 2100. It has also been suggested that a 10 per cent thawing of the Siberian permafrost will release 40 GtC by 2050; an increase that will not be offset by the predicted advance of the tree line into the tundra (Ise *et al.* 2008; Schuur *et al.* 2008). Emissions on this scale would make reaching the target set of stabilization at a 2ºC rise difficult.

In addition to melting permafrost and soil respiration, peat emissions are linked to lowered water table levels, which are highly vulnerable to climate change (Ise *et al.* 2008), suggesting a need for water table management. One issue that has not received much coverage in the literature is that of potential impacts on sea level rise. It is not just increasing temperatures that can lead to such feedbacks. A study in California has suggested that inundation of the 150,000 km^2 of low-lying peatlands may cause substantial emissions (Henman and Poulter 2008).

The effect that wetlands will have on the climate will depend upon how the complex processes that govern GHG emissions as well as CO_2 sequestration depart from their historical steady-state. Northern wetlands are expected to release GHGs in a warming climate and suffer a reduction in area, whereas wetlands in the temperate, subtropical and tropical regions could increase in area (Lloyd 2008).

One area of research that has expanded since the AR4 is that of the projected Amazon drying and dieback. Although there is still considerable uncertainty, most models predict reduced precipitation in areas of the Amazon, which will lead to increased drying (Betts *et al.* 2008b). Amazon forest dieback may also exert feedbacks through changes in the local water cycle and increases in dust emissions. This is exacerbated by deforestation and degradation, which increases the vulnerability of forest and lowers resilience to climate change, which may decrease the Amazon forest's ability to absorb and sequester carbon (Malhi *et al.* 2008). Climate-ecosystem feedbacks have also been implicated in droughts in the Sahel and Western Australia (Chapin *et al.* 2008).

On a global scale, climate scenario modelling suggests that the terrestrial biosphere will become a carbon source by 2100, largely due to increased soil respiration and the dieback of the Amazon. Climate models incorporating these feedbacks led to a 0.38ºC or 8 per cent increase in warming compared to a model in which feedback was not considered (Betts *et al.* 2008b). Such modelling is, however, still uncertain (Chapin *et al.* 2008). The interaction of the carbon cycle with the nitrogen cycle is also not generally included in climate models (Gruber and Galloway 2008); although it has been estimated that increased carbon sequestration may lead to an increase of N_2O emissions in grassland (Kammann *et al.* 2008). Indeed it has been shown that at the ecosystem scale CO_2 and N_2O fluxes are correlated (Xu *et al.* 2008).

There are growing concerns that impacts of climate change will reduce the mitigation capacity of ecosystems and be exacerbated by land-use change and degradation. These combined effects could potentially lower resistance to climate change impacts in addition to increasing CO_2 emissions (Malhi *et al.* 2008).

3.3.7.2 Feedbacks from land-use change

As has been discussed previously, emissions from land-use change can be significant. These effects are likely to act in synergy with increasing temperature to exacerbate GHG emissions (Muller *et al.* 2007); particularly on the century time scale (Voldoire *et al.* 2007). This is not just true of the tropics. Humankind is ultimately controlling the carbon balance of temperate and boreal forests; either directly through forest management, or indirectly through nitrogen deposition (Magnani *et al.* 2007). Increased levels of pollution could impact on the carbon sink strength of ecosystems (Canadell *et al.* 2007a). There is evidence that increased nitrogen deposition causes carbon emissions from peat in Europe (Bragazza *et al.* 2006).

Impacts of land-use change do not just provide feedbacks through greenhouse gas emissions. Deforestation in Amazonia can exert a large influence on precipitation patterns (Correia *et al.* 2008). Between 25 and 50 per cent of rainfall is recycled from forest, forming one of the most important regional ecosystem services; and removal of 35-40 per cent of the Amazon could shift the Amazon into a permanently drier climate (Malhi *et al.* 2008). This combines with slash-and-burn, logging, and degradation to increase risk of fire (Aragao *et al.* 2008), and amplifies the climate-induced Amazon dieback described above (Betts *et al.* 2008b). Conversely, deforestation strongly increases precipitation during *El Niño* years (Da Silva *et al.* 2008). Current climate models do not incorporate these feedbacks from forest loss (Betts *et al.* 2008b; Malhi *et al.* 2008). Desertification and deforestation also play a large role in the monsoon and rainfall pattern in West Africa, increasing the monsoon flow over the Guinean region and reducing rainfall over the entire West African region (Abiodun *et al.* 2008).

The recent Large Scale Biosphere-Atmosphere program in Amazonia has provided mounting evidence that intact rainforests are more resilient to climate drying than current vegetation models suggest, but that a pattern of logging, degradation and fire could reduce this resilience (Bush *et al.* 2008; Malhi *et al.* 2008), potentially converting forest into 'brush' with low evapotranspiration and high albedo providing more feedback to the climate system (Nepstad *et al.* 2008). In addition, experimental evidence suggests that the forest will reach a drought threshold where resilience is lost, emphasising the need for combined mitigation and adaptation to climate change (Nepstad *et al.* 2008). This suggests that mitigation strategies aimed at protecting forest and reducing forest degradation could play a significant role in reducing the impacts of climate change on biodiversity and ecosystem services such as water cycling, particularly in the Amazon (Betts *et al.* 2008b).

Our understanding of the scale of feedbacks from land-use change is increasing, but still lacking, and it is important to better understand the role of natural ecosystems and management practices in the carbon cycle (Betts *et al.* 2008c; Bonan 2008; Chapin *et al.* 2008; Dalal and Allen 2008; Heimann and Reichstein 2008; Potter *et al.* 2008). For example, peatlands are not explicitly included in global climate models and therefore predictions of future climate change may be underestimates (Limpens *et al.* 2008). This emphasises the need to fully consider the role of biodiversity in mitigation policies.

3.4 ROLE OF BIODIVERSITY IN MITIGATION POLICIES

Terrestrial ecosystems clearly play a major role in the carbon cycle through the storage and net removal of carbon from the atmosphere. The role of the natural biosphere in climate change mitigation is recognised in the UNFCCC through Land Use, Land-Use Change and Forestry (LULUCF). Given the scale of biospheric carbon stores, losses and sequestration, and the potential to manage these processes, the inclusion of LULUCF in future international climate change agreements is of utmost importance (Cowie *et al.* 2007a; Mollicone *et al.* 2007; Schlamadinger *et al.* 2007; Henschel *et al.* 2008). The IPCC AR4 estimates that over the next century, 345-1269 $GtCO_2e$ could be abated through land-use based mitigation policies, 15-40 per cent of total abatement requirements, through a combination of carbon stock management (avoided deforestation and degradation) and carbon sequestration policies (Rokityanskiy *et al.* 2007). In addition, land-use based mitigation policies have the potential to deliver significant additional benefits for biodiversity.

3.4.1 Land-use activities under the UNFCCC

Annex I Parties, under Article 3.3 of the Kyoto Protocol, can use *"direct human-induced land-use change and forestry activities, limited to afforestation, reforestation and deforestation since 1990, measured as verifiable changes in carbon stocks,"* to meet emissions reductions targets. In addition, they can elect forest management, grassland management, cropland management, and revegetation for inclusion in the accounting process (Benndorf *et al.* 2007; Schlamadinger *et al.* 2007). There are calls by some to include all ecosystems, land-use types and mitigation activities in the LULUCF, rather than the narrow activities specified above (Cowie *et al.* 2007b; Mollicone *et al.* 2007). Whilst the lack of monitoring, reporting and verification methodologies may be an initial barrier to implementation, there is a notable omission of peatlands and wetland (Henschel *et al.* 2008), particularly as Annex I countries have large extents of these areas.

The rules for LULUCF were set after emissions reduction targets had been agreed. This has been viewed as a limitation, as in effect land-use activities 'offset' emissions in other sectors, rather than acting as an integral part of the mitigation portfolio (Benndorf *et al.* 2007). Issues still remain over the permanence of sequestration activities as management changes or natural disturbances can quickly release any carbon accumulated (Lal 2008).

The opportunities for Non Annex I countries to participate in such activities is also limited, and restricted to the Clean Development Mechanism (CDM); where Annex I countries can gain carbon credits through activities in Non Annex I countries. CDM activities are restricted to afforestation, reforestation and deforestation activities, and can make up only 1 per cent of the emissions reduction portfolio for Annex I countries (Dutschke 2007; Schlamadinger *et al.* 2007). A detailed discussion of the current structure of LULUCF and the potential for development in post-2012 agreements is beyond the scope of this paper, but can be found in the literature (Benndorf *et al.* 2007; Cowie *et al.* 2007a; Cowie *et al.* 2007b; Dutschke 2007; Mollicone *et al.* 2007; Rokityanskiy *et al.* 2007; Schlamadinger *et al.* 2007).

The Bali Action Plan, adopted by the UNFCCC at the thirteenth session of its Conference of the Parties (COP-13) held in Bali in December 2007, mandates Parties to negotiate a post-2012 instrument, including possible financial incentives for forest-based climate change mitigation actions in developing countries (Decision 1/CP.13). The Parties specified that the development of such an instrument should take into consideration 'the role of conservation, sustainable management of forests and enhancement of forest carbon stocks in developing countries.' COP-13 also adopted a decision on 'reducing emissions from deforestation in developing countries: approaches to stimulate action' (Decision 2/CP.13). This decision recognises both that reducing emissions from deforestation and forest degradation in developing countries (REDD) can promote co-benefits and may complement the aims and objectives of other relevant international conventions and agreements, and that the needs of local and indigenous communities should be addressed when action is taken to reduce emissions from deforestation and forest degradation (Eliasch 2008).

It is generally agreed that a post-2012 LULUCF agreement should aim to reduce emissions from land-use change (including REDD) and enhance carbon reservoirs, linked to adaptation strategies (Gibbs and Herold 2007; Schlamadinger and Bird 2007; Eliasch 2008). The following section will examine the potential for REDD and other LULUCF activities to contribute to climate change mitigation.

3.4.2 Issues with including ecosystems in mitigation policy

Despite the role of ecosystems and hence biodiversity in the carbon cycle, land-use based mitigation policy has been constrained by a number of issues, both methodological and practical. One such issue is that of uncertainties over the exact role of ecosystems in the carbon cycle, as detailed previously. Other methodological issues include the lack of accurate carbon accounting, difficulties in estimating emissions reductions, and separating the effect of natural disturbances from anthropogenic activities on carbon stocks (Cowie, Kirschbaum and Ward 2007b; Schlamadinger *et al.* 2007).

More specific concerns surrounding land-use based climate change mitigation include the practical issues of permanence, leakage, and additionality (Gibbs *et al.* 2007; Eliasch 2008). 'Permanence' refers to the issue that carbon locked up in biomass and soils may be released at a later date, either following human disturbance, or natural disturbance such as drought, fire, or pests (Eliasch 2008). 'Leakage' occurs when emissions reduced in one area, for example through protection of one section of forest, are simply displaced to deforestation nearby (Benndorf *et al.* 2007); and 'additionality' refers to a situation in which the emissions reductions or carbon savings would have occurred anyway in the absence of mitigation policy.

These issues have been a particular concern in project-based activities such as those currently allowed under the CDM, but are less so when emissions are reported through national level accounting, as is likely for a REDD mechanism (Eliasch 2008), which is discussed in section 3.4.3.1.

3.4.3 Potential for mitigation through forest activities

Mitigation strategies in the forest sector fall under two main areas, the maintenance of stored carbon through REDD, and the sequestration of carbon from the atmosphere through afforestation, reforestation, and restoration (ARR). It is not clear how the development of the REDD mechanism will interact with LULUCF, but it is possible that REDD will include afforestation and reforestation activities in an all encompassing mechanism. However, for the purposes of this report, afforestation and reforestation are treated separately from REDD in accordance with the current structure under LULUCF.

The IPCC AR4 reported that a combination of forestry activities would have the potential to achieve 0.4 GtC emissions reductions per year with a price of US$20 per tonne, and 1.3-4.2 $GtCO_2$ per year reductions in 2030 at costs up to US$100 US per tonne CO_2. Mitigation through reduced deforestation was considered to have greater potential than that offered by afforestation (IPCC 2007), and is therefore the focus of this section.

More recent analyses have suggested that including forests in the cap and trade system would reduce emissions by 2.6 $GtCO_2$ per year by 2030 (Eliasch 2008). Including ARR in this scenario adds another 0.9 $GtCO_2$ per year of emissions savings; with a total potential for 3.5 $GtCO_2$ emissions savings by 2030.

3.4.3.1 Reducing emissions from deforestation and forest degradation in developing countries (REDD)

In recognition of the importance of tropical forest in the global carbon cycle, and in the provision of biodiversity and ecosystem services, proposals for the development of a REDD mechanism are being rapidly developed (Canadell and Raupach 2008; Olander *et al.* 2008). As emissions from deforestation are currently approximately 1.5 GtC per year, there appears to be high potential for cost-effective emissions reductions from REDD (Canadell and Raupach 2008). Currently, the UNFCCC has no mechanism for reducing deforestation in developing countries (Gullison *et al.* 2007).

The exact form of REDD is still to be determined, but is likely to involve national-level accounting whereby reductions in emissions from deforestation are measured relative to a baseline, determined according to the circumstances and historical emissions of the country (Eliasch 2008). A national-level approach would reduce the risk of leakage. It is still unclear how Parties with low deforestation rates will be compensated, but there are various proposals considering how this might be achieved (Mollicone *et al.* 2007; Strassburg 2007; Strassburg *et al.* 2008; TCG 2008). Such proposals are detailed in the Eliasch Review (Eliasch 2008), which outlines a number of options including linking baselines to a global 'business as usual' emissions scenario in order to ensure that all forest stocks are incorporated. It is also unclear whether REDD will be financed through taxation, an international fund, or through the carbon market (Skutsch *et al.* 2007). The scale of emissions reduced, and particularly biodiversity benefits, will be determined by the design of the mechanism.

It is widely accepted that a successful mechanism for REDD will have to address the drivers of deforestation and will require effective targets, robust monitoring and measuring, appropriate financial mechanisms, and good governance (Eliasch 2008).

3.4.3.1.1 Mitigation potential

There remains significant deforestation pressure in the tropics. A recent study of the tropical humid biome has estimated that 27.2 Mha of forest, or 2.36 per cent of the total stock, was cleared between 2000 and 2005 (Hansen *et al.* 2008), with deforestation 'hotspots' in Brazil and Indonesia. It has further been estimated that current plans for infrastructure development in the Amazon will result in the release of approximately 32 GtC (Malhi *et al.* 2008), and 15-26 GtC in the next three decades in combination with fire, degradation and drought (Nepstad *et al.* 2008). However, it has also been suggested that effective enforcement of protected areas could avoid 17 GtC emissions by 2050 (Soares-Filho *et al.* 2006). Clearly, there is significant scope for reduction of deforestation, providing that the financial incentives are sufficient to cover the opportunity costs of land conversion (Nepstad *et al.* 2008). In addition to the scope for reducing deforestation, there appears to be considerable need. It has been estimated that the global economic cost of the climate change impacts of deforestation will rise to around US$1 trillion a year by 2100 in the absence of mitigation (Hope and Castilla-Rubio 2008).

Indeed, it has been estimated that an additional 87-130 GtC will be released by 2100 in the absence of policy measures (Houghton 2005a), whereas a 50 per cent reduction in deforestation rates by 2050 (and maintained until 2100, with a cessation in deforestation when only 50 per cent of the forest area remains) would avoid the direct release of up to 50 GtC, or nearly 12 per cent of total required reductions for stabilisation at 450ppm (Gullison *et al.* 2007). Eliasch (2008) suggests that a *well designed* REDD mechanism could reduce deforestation rates by up to 75 per cent in 2030; and in combination with ARR could make the forest sector carbon neutral. Absence of mitigation efforts through reducing emissions from the forest sector would increase atmospheric CO_2 levels by approximately 30ppm (Hope and Castilla-Rubio 2008). As the current CO_2 levels stand at 433ppm, and the stabilisation target is at 445-490ppm, the inclusion of REDD is critical to achieving reduction targets (Eliasch 2008).

There remains a scarcity of literature on the potential for reducing emissions from forest degradation specifically, although estimates of the scale of degradation suggest that the potential is high (section 3.3.4). A recent study has demonstrated that improved management of forest, e.g. through reduced impact logging, can reduce carbon emissions by approximately 30 per cent (Putz *et al.* 2008). Therefore, improved practices in tropical forest designated for logging would retain at least 0.16 GtC per year, (particularly in Asia), or 10 per cent of that obtainable through completely halting tropical deforestation (Putz *et al.* 2008). Recent evidence that many tree species with high carbon storage are preferred timber species (Kirby and Potvin 2007), suggests that species-level management will be important in reducing emissions through degradation.

The potential for REDD to contribute to emissions reductions through protecting carbon stores is clear, but there have been questions since the AR4 over the potential of old-growth forest to act as both a carbon store and a sink. Recent evidence suggests that old-growth and established forests can continue to accumulate carbon, contrary to the long-standing view that they are carbon neutral (Desai *et al.* 2005; Luyssaert *et al.* 2008; Phillips *et al.* 2008). In addition, old-growth forests can accumulate carbon in soils (Zhou *et al.* 2006), suggesting that REDD will contribute to emissions reductions through carbon sequestration in addition to maintenance of carbon stocks. Further, the carbon sink in old-growth Amazonian forest is comparable to the emissions from deforestation (Phillips *et al.* 2008), and it has been estimated that the atmospheric CO_2 concentration would be 10 per cent higher in the absence of the tropical forest sink (Betts *et al.* 2008a); although the potential future impact of climate change on this sink is uncertain (Heimann and Reichstein 2008).

3.4.3.1.1.1 Mitigation capacity in the face of climate change

Although it was noted in the IPCC AR4 that global change will impact upon carbon mitigation in the forest sector, the magnitude and direction of the change could not be predicted with confidence. This remains the case (Heimann and Reichstein 2008), although there is a growing body of literature in this area. Recently, it has been reported that 10 of 11 climate models project that tropical forests will continue to act as a net sink even in the face of climate change (Gullison *et al.* 2007), and evidence suggests that reduced deforestation and degradation can increase resilience of ecosystems to climate change impacts (Betts 2007; Malhi *et al.* 2008; Nepstad *et al.* 2008). There is, therefore, some evidence to support the claim that REDD could maintain the capacity of forests to resist climate change (below a certain threshold), and provide and assist with local adaptation to climate change (Betts *et al.* 2008c).

3.4.3.1.1.2 REDD and tropical peatland

As highlighted previously, peatland is not eligible for inclusion under any of the current carbon accounting mechanisms within the LULUCF. Whilst this could potentially remain the case for boreal peatland, tropical peatland has the potential to be covered under REDD; particularly if emissions are measured as the difference in both above and below ground carbon stocks between the original forest and the altered land use (Eliasch 2008). Forty-six per cent of deforestation in Southeast Asia occurs on peat (Hoojier *et al.* 2006), and accounts for substantial emissions of CO_2 to the atmosphere. However, to fully capture the carbon emissions from peatland it would be necessary to report carbon loss from soil below the current depth of 30cm specified by the IPCC (Miles and Kapos 2008).

3.4.3.1.1.3 Forest in Annex I countries

Although REDD has dominated recent forest discussions, it has been suggested that land use change and degradation in all areas (not just tropical) should be included in a future climate change agreement (Höhne *et al.* 2007; Mollicone *et al.* 2007; Eliasch 2008). Forests in temperate regions, particularly boreal forests, store large amounts of carbon; particularly in the soil (Ciais *et al.* 2008; Nabuurs *et al.* 2008), but current climate mitigation policies do not incentivise their conservation (Nabuurs *et al.* 2008).

3.4.3.1.1.4 Economic feasibility of REDD

Economic modelling has suggested that REDD will be a competitive, low-cost abatement option (Ebeling and Yasue 2008; Kindermann *et al.* 2008; Neeff 2008), as had been suggested in the IPCC AR4 (Fig. 1).

A 10 per cent reduction in deforestation from 2005 to 2030 could provide 0.3-0.6 GtC in emission reductions annually and would require US$0.4 billion to US$1.7 billion (Kindermann *et al.* 2008). It has been suggested that a 10 per cent annual reduction in deforestation would reduce deforestation emissions 75 per cent by 2020; and that a 50 per cent reduction of total deforestation could provide 1.5-2.7 GtC in emission reductions and would require US$17.2 billion to US$28.0 billion (Mollicone *et al.* 2007).

Indeed, it has been suggested that an investment in reducing deforestation on the same scale as that put into the Renewable Transport Fuel Obligation (RTFO) in the UK would result in avoided emissions 50 times greater than those currently achieved (Spracklen *et al.* 2008). The Eliasch Review (2008) similarly concluded that the cost of halving global emissions could be reduced by 50 per cent in 2030 through inclusion of the forest sector in a trading system. This would require finance of approximately US$17-33 billion per year, of which US$7 billion could be supplied by the carbon market, and US$11-19 billion would need to come from other funding sources. Opportunity costs of forest conservation have risen since estimates by Stern (2007) due to the rise in agricultural commodity prices, and now stand at US$7 billion (Eliasch 2008).

These levels of finance can be put in context when considering the costs of *not* reducing emissions from deforestation. Modelling for the Eliasch review has suggested that the net benefits (including ecosystem services) of a 50 per cent reduction in deforestation could amount to US$3.7 trillion over the long term, rising to US$6.3 trillion if 90 per cent deforestation is reduced (Braat and Brink 2008).

Whilst these global figures highlight the potential for REDD, it is at the national level that such finance will need to be realised. Nepstad *et al.* (2008) suggest that a 30 year programme costing US$8 billion (less than US$2 per tonne carbon) could result in the cessation of deforestation in the Amazon within 10 years. A study in Panama has estimated the total yearly cost of REDD at US$3.5 million (Potvin *et al.* 2008). Although clearly 'cost effective', these studies emphasise the need for significant financial investment and capacity building. The Eliasch Review (2008) estimates that such support for 40 forest nations could cost US$4 billion over five years. A number of countries are already receiving support through the World Bank's Forest Carbon Partnership Facility and the UN REDD programme in a demonstration phase.

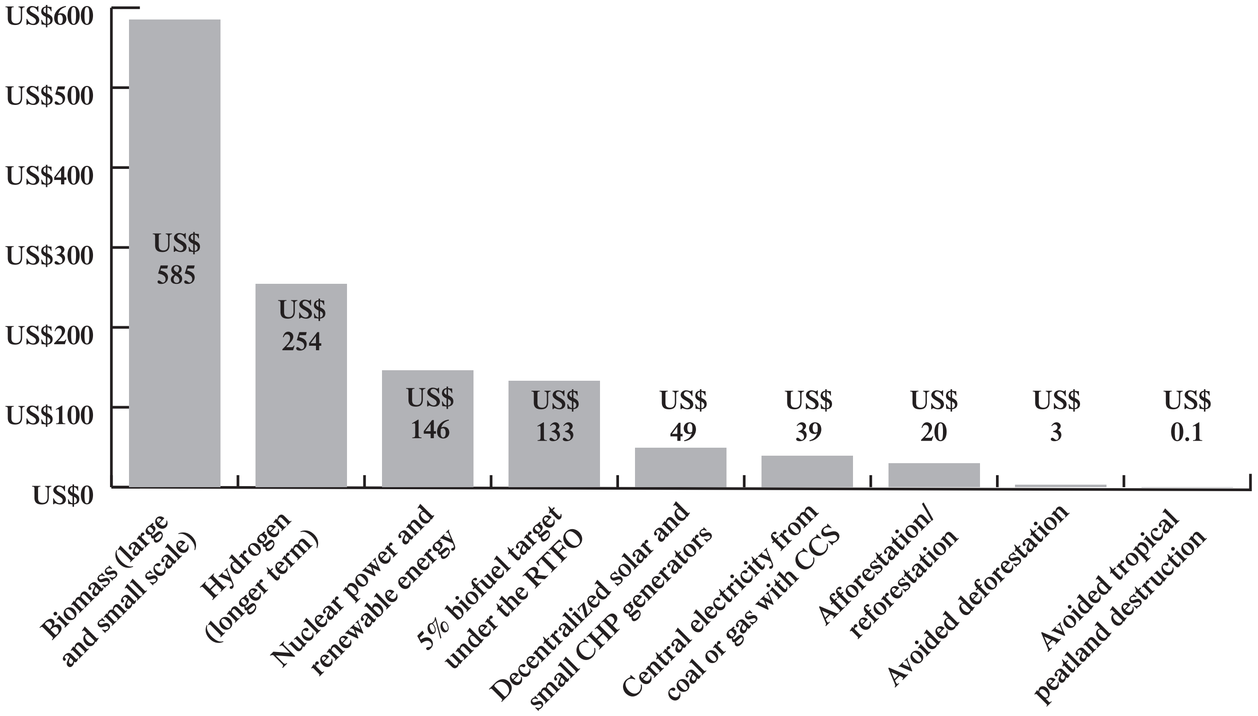
3.4.3.1.1.5 Methodological capabilities

The uncertainty in estimates of emissions from deforestation is largely due to data availability and methodological issues, and a significant body of research has gone into developing methods to resolve this (Gibbs *et al.* 2007; Herold and Johns 2007; Ramankutty *et al.* 2007; Olander *et al.* 2008). Although the exact form of a REDD mechanism is yet to be determined, Parties will be required to monitor carbon emissions from deforestation and degradation, which will require monitoring of forest area loss and proportion of biomass lost in degradation, in addition to knowledge of the biomass and the carbon content of each type of forest lost or degraded (Olander *et al.* 2008). Clearly, improvements in monitoring are required at a pan-tropical and national scale, and require commitments of capacity building and standardised protocols (Achard *et al.* 2007).

It is likely that reduced emissions will be measured against a baseline, probably established through historical rates. Establishment of baselines and monitoring of deforestation is likely to see a significant role for remote sensing data (Olander *et al.* 2008). Details of the available remote sensing options have been extensively reviewed (Herold and Johns 2007; Olander *et al.* 2008), with significant progress being made.

These estimates of deforestation need to be combined with

Figure 1: Cost comparison of carbon mitigation options (Spracklen *et al.* 2008). Original source IPCC AR4.



CHP: Combined heat and power CCS: Carbon capture and storage carbon stock estimates. There is no perfect method for estimating these stocks, but a range of options do exist, and it is generally agreed that technological constraints should not act as a barrier to the development of a REDD mechanism (Herold and Johns 2007). It is likely that methodologies for assessing and monitoring carbon stocks will be based on the current IPCC good practice guidelines (Olander *et al.* 2008).

Although there are clearly many technical and political issues to be resolved before REDD could be put in place, there is growing consensus that such issues can be overcome. General scientific opinion appears to suggest that tools in development for assessing and monitoring carbon stocks are accurate and feasible for use in a REDD mechanism (Eliasch 2008).

3.4.3.1.2 Biodiversity impacts

A successful REDD mechanism has the potential to deliver significant benefits, contributing to biodiversity conservation at both the species and ecosystem level, and to the maintenance of ecosystem services (Eliasch 2008). However, the design of REDD is still under discussion and will be the subject of negotiation (Skutsch *et al.* 2007). The different proposed versions of REDD are likely to have differing impacts on biodiversity (Strassburg 2007; Strassburg *et al.* 2008; TCG 2008), which will be influenced by the baselines adopted, and the financial mechanism employed (Mollicone *et al.* 2007; Eliasch 2008; TCG 2008). For example, whether or not incentives for REDD are directly connected to forest area (regardless of deforestation rates) will impact upon tropical forest conservation (Mollicone *et al.* 2007; Strassburg 2007; TCG 2008). It has been suggested that REDD should include an explicit means of rewarding actions that reduce emissions from deforestation and degradation in ways that deliver benefits for biodiversity and ecosystems, e.g. through premium credits for projects providing co-benefits (Eliasch 2008).

Although the biodiversity impacts and benefits of REDD will depend upon the exact mechanism decided upon, REDD is likely to have net biodiversity benefits through tropical forest conservation at the scale detailed above; as habitat conversion is the major cause of biodiversity loss (Ravindranath 2007). The potential for generation of finance at levels of US$1-10 billion is also at a scale not seen previously for forest conservation (Miles and Kapos 2008).

3.4.3.1.2.1 Biodiversity value of natural forest

Tropical forests have extremely high levels of biodiversity. The Amazon rainforest alone hosts about a quarter of the world's terrestrial species (Malhi *et al.* 2008). Deforestation continuing at projected rates in Southeast Asia could result in the loss of 79 per cent of Southeast Asian vertebrates by 2100, half of which are endemics (Brook *et al.* 2008). Mean total extinction rates of 20 per cent and 33 per cent of tree species in the Brazilian Amazon are projected under the optimistic and non-optimistic scenarios of habitat loss respectively (Hubbell *et al.* 2008). Forest degradation and logging increases access to bushmeat, further threatening many tropical forest vertebrates (Brook *et al.* 2008), and habitat fragmentation reduces the adaptive capacity of species to climate change (Brook *et al.* 2008), and influences species distributions (Escalante *et al.* 2007).

3.4.3.1.2.2 Possible biodiversity impacts of REDD

Despite the obvious biodiversity benefits of conserving tropical forest, an international REDD mechanism under UNFCCC will be focused on carbon storage, and may not explicitly support biodiversity and other forest ecosystem services. There may be risks of cross-ecosystem leakage under REDD, whereby protection of forest leads to additional pressure to convert or degrade other ecosystem types (Miles and Kapos 2008). This could have negative effects on the biodiversity of these other ecosystems, and should be considered in conservation planning through, for example, focus of funds on non-forest ecosystems and low-carbon forest (Miles and Kapos 2008).

Management practices, such as suppression of fires, may also impact biodiversity in the long term, as many forest processes rely on natural fire regimes (Berry *et al.* 2008). There is also no guarantee that representative forest types will be protected, with the representation of forests across environmental gradients beneficial for biodiversity conservation but not necessarily for reducing carbon emissions (Berry *et al.* 2008).

One aspect that may have an impact on biodiversity conservation is the definition of forest and forest degradation. It is difficult to agree appropriate universal definitions, and national definitions may be more applicable to the development of a REDD mechanism, but would have to be developed during the REDD preparation phase. Conversely, a definition clearly distinguishing between natural forest and plantations appears essential if afforestation and reforestation are to be included in the REDD mechanism. Under such a scenario, it is conceivable that deforestation could continue at present rates, provided the emissions were offset by the establishment of new plantations (Eliasch 2008). Afforestation does not always have positive biodiversity benefits, and can in fact have negative impacts when replacing natural ecosystems (section 3.4.3.2.2).

If REDD is to deliver benefits for biodiversity, it is important that feasibility studies and demonstration phases for REDD take into account the national pressures affecting biodiversity conservation, and assist in the development of tools to quantify and report methods for assessing and prioritising these benefits (Miles and Kapos 2008).

With regard to the impacts of REDD on local and indigenous communities, there are risks as well as opportunities, and there are issues of governance and tenure to be resolved (Peskett *et al.* 2008). It has been suggested that involving local communities in REDD is essential if it is to provide both biodiversity and carbon benefits (Singh 2008) and appears to be an aspect that requires further research. However, with the scale of finance that could be made available, REDD has a significant opportunity to provide financial benefits to local communities as well as maintain ecosystem services (Eliasch 2008).

3.4.3.2 Afforestation, reforestation and restoration

3.4.3.2.1 Mitigation potential

There is significant uncertainty attached to the level of carbon sequestration that can be achieved through afforestation and reforestation; and the potential for mitigation in this sector, particularly on decadal time scales, is often questioned (Canadell *et al.* 2007a). Whilst there is significant potential for increasing the capacity of the natural carbon sink, particularly in the tropics, it has been suggested that there is a need for more integrated study of how land management changes may affect climate change (Betts 2007; Chapin *et al.* 2008).

According to a range of cost estimates from US$20 to US$100, reforestation could sequester 0.16-1.1 GtC per year to 2100, with land requirements of up to 231 Mha (Canadell and Raupach 2008). Modelling for the Eliasch review has supported these figures, with an estimated mitigation potential of 0.9 $GtCO_2$ per year. A global analysis of land suitability for CDM-AR carbon 'sink' projects identified large amounts of land (749 Mha) as biophysically suitable and meeting the CDM-AR eligibility criteria, but much was on productive lands, grassland, or savanna. The implications of this would require consideration if the cap on CDM-AR were to be raised (Zorner *et al.* 2008).

Whilst in some regions afforestation has clearly had an impact (China has established 24 Mha of plantations to transform the forestry sector from a source to a sink, offsetting 21 per cent of their fossil fuel emissions), there is debate over the climate change mitigation benefits provided by afforestation thus far (Canadell *et al.* 2007a). Indeed, evidence questions the mitigation benefits of afforestation and reforestation; suggesting that although such activities are cost effective, the relative contribution of plantations to emissions reductions is relatively low (Strengers *et al.* 2008). Clearly, the previous land use goes a long way to determining the carbon benefits of afforestation. Expanding agroforests into areas currently under pasture could sequester significant amounts of carbon while providing biodiversity and livelihood benefits (Kirby and Potvin 2007; Jindal *et al.* 2008), whereas expanding into natural grassland or wetland can have negative impacts for both carbon and biodiversity (Berry *et al.* 2008).

The impact of afforestation on soil is an area identified by IPCC AR4 as requiring further research, and this appears to remain a priority. For example, a recent study in Africa has found that afforestation projects in savanna ecosystems had negative impacts on the carbon budget one year after plantation due to soil disturbances (Nouvellon *et al.* 2008), whereas afforestation of grassland has had a net positive impact in one region of China (Hu *et al.* 2008). Peatlands are used extensively for forestry in Canada and Scandinavia, in which carbon emissions from the draining of peat are likely to outweigh carbon sequestration (Parish *et al.* 2008). Effective forest carbon sequestration requires the management of all carbon pools, including traditionally managed pools such as bole wood and also harvest residues and soils (Gough *et al.* 2008). It is also important to consider local factors in reforestation policies (Clement and Amezaga 2008).

All evidence suggests that the greatest carbon benefits from afforestation and reforestation can be gained from the tropics (Bala *et al.* 2007). The climate change mitigation benefits of reforestation in boreal regions are less certain when taking albedo and evaporation into account (Bala *et al.* 2007; Bonan 2008); which has led to the conclusion that the best strategy for forest carbon management in temperate regions is to discourage land-use change and avoid large albedo changes (Bala *et al.* 2007; Canadell and Raupach 2008).

In addition, there are concerns over the response of plantation forest to climate change. It is thought that plantations have less natural resilience than natural forest to climatic perturbations, and climate induced changes in fire and insect outbreaks (Stephens *et al.* 2007), and it will be necessary to consider inter- and intra-species responses to climate change to optimise mitigation potential (O'Neill *et al.* 2008).

It should be emphasised that forests are valuable resources for many reasons unrelated to climate, but that this depends on the type of afforestation. Species selected for high carbon sequestration may have the greatest carbon mitigation benefits, but low biodiversity benefits. Mitigation strategies should not reduce the resilience of forest to climate change (Berry *et al.* 2008), and need to be planned with reference to potential future climatic conditions.

3.4.3.2.2 Biodiversity impacts

It is well publicised that sequestration schemes and policies often require a trade-off between biodiversity benefits and carbon benefits (Nelson *et al.* 2008). Production forest results in higher carbon benefits but fewer biodiversity benefits, whereas multifunctional forest can have biodiversity benefits but is of lower sequestration value (Garcia-Quijano *et al.* 2007a; Garcia-Quijano *et al.* 2007b). Furthermore, plantations often support lower levels of species diversity than natural ecosystems, and afforestation of natural ecosystems can have significant negative impacts on biodiversity (Cowie *et al.* 2007a; Berry *et al.* 2008).

Recent reviews of plantation forest and biodiversity have suggested that plantations can be beneficial, but only when planted on degraded land or on agricultural land where it can buffer edge effects and increase connectivity; whereas conversion of natural forest, and afforestation of non-natural forest land is detrimental (Brockerhoff *et al.* 2008). The conversion of grassland to plantation has been found to have negative impacts on ecosystem services in Ecuador (Farley 2007). Monoculture plantations are likely to have negative biodiversity impacts, and be less resilient to climate change, whereas promotion of heterogeneous plantations with native species and diverse gene pools reduces the biodiversity impact of plantations (Berry *et al.* 2008).

It has been suggested that plantation forestry can be the 'lesser of two evils' where land was earmarked for conversion (Brockerhoff *et al.* 2008). In addition, plantations can reduce degradation pressures on natural forest, but this requires landscape level land-use planning. In particular, the impacts of monocultures and biodiversity and ecosystem services should be assessed at a site level (Brockerhoff *et al.* 2008). The use of fast growing genetically modified or non-native trees could also have significant implications for biodiversity, particularly where the species have the potential to be invasive.

The evidence for negative impacts of afforestation/ reforestation CDM projects on the hydrological cycle have been well publicised, with evidence of reduced water flow following afforestation schemes (Jackson *et al.* 2005). Such impacts have been noted as a result of large scale afforestation in China (McVicar *et al.* 2007), largely in previously unforested and water stressed areas. However, afforestation of agricultural land in the tropics can increase the water infiltration capacity of soil (Ilstedt *et al.* 2007), and such impacts are largely dependent on the previous natural land cover. Although the current limit on afforestation projects through the CDM limits the scale of expansion, land use impacts and potential for conversion of natural ecosystems would need to be taken into account if afforestation efforts were to be increased (Jackson *et al.* 2005; Farley *et al.* 2005; Trabucco *et al.* 2008) through, for example, a REDD mechanism.

The biodiversity benefits of reforestation should be higher than those of afforestation because it is on naturally forested land. However, research has suggested that species utilisation of regrowth forest is variable, and landscape scale management is required to maximise biodiversity benefits through restoration (Bowen *et al.* 2007).

3.4.3.3 Forest management

3.4.3.3.1 Mitigation potential

In addition to reducing deforestation and improving sequestration capacity of forest, there are carbon benefits to be gained from managing existing forests to increase sequestration capacity. Although Annex I countries had the opportunity to include forest management in their carbon accounting, it is generally agreed that the system is limited and does not optimise the potential for sustainable use of forest in climate change mitigation (Nabuurs *et al.* 2008), particularly with reference to removal of forest products and substitution effects (Bottcher *et al.* 2008). Clear-cut harvesting and fire disturbance result in a lasting decrease in annual forest carbon storage in temperate forest (Gough *et al.* 2008).

Given recent questions surrounding the actual mitigation potential of temperate forest plantations, due to albedo effects (Bala 2007), there has been some agreement that reducing deforestation and sustainable forest management are the best options in these regions, and recent studies have identified 'hotspots' of European forest, where carbon storage and accumulation is high (Ciais *et al.* 2008; Nabuurs *et al.* 2008).

Improving forest management can significantly reduce carbon emissions (Putz *et al.* 2008), and most forest management strategies, such as control of fire by thinning and removal of undergrowth, have mitigation benefits (Berry *et al.* 2008). However, capacity for reduced emissions through forest management requires consideration of natural disturbances. In Canada, for example, forest has turned from sink to source following large scale insect outbreaks, resulting in the decision not to elect forest management as an accounting option (Kurz *et al.* 2008).

3.4.3.3.2 Biodiversity impacts

Whilst improved forest management practices can have significant biodiversity benefits, forest management specifically for climate change mitigation can have some negative impacts. Removal of woody debris for biomass and removal of undergrowth can have negative impacts on undergrowth-dwelling species and can alter ecosystem dynamics, as can control of fire regimes (Berry *et al.* 2008).

3.4.4 Potential for mitigation of other activities under the LULUCF

3.4.4.1 Improved cropland management

3.4.4.1.1 Mitigation potential

The IPCC AR4 estimated that agriculture accounted for 5.1-6.1 $GtCO_2e$ per year in 2005, or 10-12 per cent of global emissions, mostly through release of N_2O and CH_4, and is a global source of emissions. There is therefore significant potential for emissions reductions through agricultural management, mostly through reductions in loss of soil organic carbon (SOC). Agriculture is likely to remain a net source (Canadell *et al.* 2007a); particularly where cropland replaces natural ecosystems as cultivated soils generally contain 50-75 per cent less carbon than those in natural ecosystems (Lal 2008).

An in depth review of the full breadth of mitigation strategies within agriculture is beyond the scope of this report, but has been produced for Europe (Berry *et al.* 2008). Such mitigation strategies include improvements in livestock management, animal breeding and husbandry, grassland and grazing management, crop production, water management, reduced tillage, use of breed cultivars, use of nitrogen-fixing crops, and fertiliser management.

Enhanced carbon sequestration in soil is seen as the most important agricultural mitigation technique in Europe (Berry *et al.* 2008). Changing agricultural land use, in particular through agroforestry schemes, is one strategy to achieve this; as is the use of no-till agriculture. Agroforestry involves the planting of trees intermingled with crops and increases both standing biomass and soil sequestration, and has a high mitigation potential in the tropics (Verchot *et al.* 2007). Employing no-till agriculture minimises disturbance to soil carbon that can result in high levels of emissions to the atmosphere (Canadell *et al.* 2007a). Crop genetic diversity also aids the ecosystem to sequester carbon, and helps in preventing soil erosion (Hajjar *et al.* 2008), with a higher SOC content than soils under monoculture (Lal 2008).

It has been suggested that, in Europe at least, reductions in agricultural emissions do not occur through climate policy, but through improved management practices, which can provide both carbon and biodiversity benefits (Lal 2008). Agricultural management was specifically identified by the IPCC AR4 as a mitigation option with considerable potential co-benefits (IPCC 2007).

3.4.4.1.2 Biodiversity impacts

A review of the impacts of all potential agriculture mitigation techniques in Europe (Berry *et al.* 2008) has suggested that when 'worst practice' techniques are employed, no agricultural mitigation practice is beneficial in biodiversity terms. Those identified as detrimental to biodiversity, even under 'best practice' techniques included the improvement of species-rich grassland with specific varieties, the use of high sugar grasses, and displacement of food crops for biofuel.

Most 'best practice' techniques were considered to be either moderately or highly beneficial to biodiversity (Berry *et al.* 2008; Lal 2008), showing potential for considerable 'multiple benefits'. Although the IPCC suggests that agricultural improvements would in general be positive for biodiversity, it is acknowledged that this would require trade-offs. It is clear that the impacts will vary according to location, current biodiversity, and management techniques, making it difficult to apply top-down rules, and raising the question of what the impacts may be if agricultural management is dictated by climate policy (Berry *et al.* 2008). This issue was raised by the IPCC AR4, which noted the potential for reduced productivity of cropland to displace land-use change elsewhere, whereas increased productivity can 'spare' further land-use change.

In the tropics, it has been suggested that agroforestry can be beneficial for biodiversity, and might increase resilience of agriculture to climate change impacts (Kirby and Potvin 2007; Verchot *et al.* 2007; Bhagwat *et al.* 2008). The literature reviewed suggests that agricultural mitigation strategies in particular can have considerable overlap with adaptation strategies. For instance, soil and watershed conservation help to improve water productivity of agricultural systems and increase carbon sequestration (Castillano *et al.* 2007; Hartman *et al*. 2007; Molden 2007; Nobel 2007).

3.4.4.2 Grassland management

3.4.4.2.1 Mitigation potential

Grasslands can sequester large amounts of carbon, primarily in the soil (Table 1). Degradation of grasslands can therefore be a large source of carbon loss. A large body of literature in this area has focused on China, which has large areas of grassland with high stores of soil organic carbon (Yang *et al.* 2008). Degradation of these grasslands accounts for the biggest loss of carbon in China (Xie *et al.* 2007). Although the contribution to the carbon flux remains uncertain, land management practices have a large impact on uptake and release of CO_2 in grasslands (Cernusca *et al.* 2008).

Much of the literature focuses on management of grazing lands rather than unmanaged grasslands. Strategies for grassland management include enhancement of biomass production, the humification of biomass returned to the soil, facilitation of transfer of carbon deep into the subsoil by deep root system development, and the formation of organo-mineral complexes (Lal 2004). Good practice grassland management, including rotational grazing, nutrient management (Khan *et al.* 2007), and reduced burning, can increase soil carbon and reduce the loss of soil carbon through leaching (Lal 2004; Manson *et al.* 2007). Increased fungal biodiversity is also related to higher soil carbon storage in grasslands (Persiani *et al.* 2008). More recently, management of grasslands for species that are likely to increase NPP under conditions of increased CO_2 has been employed.

3.4.4.2.2 Biodiversity impacts

It has been suggested that improving degraded grassland would be a 'win-win' for climate change and human development (Neely and Bunning 2008), as better grasslands for livestock would provide better food security. Improvement of degraded grassland with native species can have positive biodiversity impacts. However, grassland management can also have negative biodiversity impacts on plant, vertebrate and invertebrate species; particularly where diverse grasslands are replaced by a limited number of specific varieties and high sugar grasses (Berry *et al.* 2008). Introduction of nitrogen-fixing species with the potential to become invasive can also have biodiversity impacts, as can increased use of fertiliser.

3.4.4.3 Re-vegetation

Re-vegetation is defined in the Kyoto Protocol as a direct human-induced activity to increase on-site carbon stocks through establishment of vegetation that does not meet the definitions of afforestation and reforestation (FCCC/CP/2001/13/Add. 1, page 58). Generally, the purpose is for erosion control on degraded lands (SCBD 2003). There is limited information available about the potential for re-vegetation to contribute to climate change mitigation, with the more recent studies quoting the figure reported in IPCC AR4 that vegetation regrowth and thickening in semi-arid regions and savannas accounts for 22-40 per cent of the carbon sink in the US (Canadell *et al.* 2007a).

As re-vegetation tends to be on degraded land, the effects are generally positive but will vary according to the methods used, and whether native or exotic species are utilised (SCBD 2003).

3.4.4.4 Improved soil management

Improved soil management is not an activity explicitly specified under LULUCF, as it is included under all of the activities described above. However, the role in climate change mitigation should be emphasised as it is often considered the area with the highest potential outside of forest activities. Global soil organic carbon has a sequestration potential 0.6-1.2 GtC (Lal *et al.* 2007) with high levels of carbon stocks, many of which is contained under natural ecosystems rather than managed ecosystems (Lal 2008). Emissions of approximately 78 $GtCO_2$ have been estimated from loss of soil carbon (Lal *et al.* 2007).

Degradation of soil also has biodiversity impacts through loss of biomass productivity and reduction in water quality. It has therefore been suggested that improvement in soil management under LULUCF is a 'win-win' strategy for biodiversity (Lal 2008). Wetland and peatland soils in particular are high in carbon and are currently being heavily degraded.

3.4.4.5 Wetland and peatland – options for inclusion in the LULUCF

Wetlands contribute to the global terrestrial carbon pool and act as sources or sinks as discussed earlier. However, uncertainty in how they will affect climate means that their role in current and future climate change mitigation is unclear (Lloyd 2008). However, northern wetlands are extremely susceptible to climate change as well as degradation. Many peat bogs in Europe have been drained and are being restored (Glatzel *et al.* 2008), and over 55 per cent of peatland area in Finland has been drained for agriculture and forestry between 1950 and 2000 accounting for approximately one third of national greenhouse gas emissions (Turunen 2002). Low carbon accumulation has been reported in peat bogs in Sweden as a result of high levels of nitrogen deposition, which alters the dominance of peat-forming vegetation (Gunnarsson *et al.* 2008).

The susceptibility of peatland to climate and land-use change has been emphasised (Ise *et al.* 2008), and although peatlands are currently acting as a carbon sink, they are likely to turn into a carbon source if current management strategies persist (Hoojier *et al.* 2006; Cagampan and Waddington 2008; Jaenicke *et al.* 2008; Neely and Bunning 2008; Parish *et al.* 2008; Uryu *et al.* 2008). Peatlands also support many specialised species and unique ecosystem types, and can provide a refuge for species that are expelled from non-peatland areas due to degradation and climate change (Parish *et al.* 2008).

Tropical peatlands are less susceptible to climate warming than those in the northern latitudes, but are currently threatened by land-use change (Lloyd 2008). The Indonesian peatlands, for example, have been identified as one of the largest stores of carbon in the terrestrial biosphere, but are threatened by anthropogenic threats such as biofuel production (Jaenicke *et al.* 2008; Uryu *et al.* 2008). As land-use change is the principal driver of tropical peatland loss, it has been suggested that they should be explicitly included in climate change mitigation policies (Parish *et al.* 2008). As discussed previously, reducing deforestation and degradation in tropical peatland may capture some of the tropical peat emissions (Hoojier *et al.* 2006), but there is no explicit mechanism for tropical peatland inclusion in the UNFCCC.

As current emissions from peatlands are largely down to anthropogenic degradation, and degradation increases susceptibility to climate change, it has been suggested that conservation of peatlands can be cost-effective mitigation measures (Parish *et al.* 2008; Spracklen *et al.* 2008). Restoration of peatlands is also an option. Restoration techniques generally involve raising the water table through water management and the reintroduction of peat forming vegetation (Cagampan and Waddington 2008; Limpens *et al.* 2008), which can rapidly reduce carbon loss whilst also reducing the vulnerability of peat to climate change (Kechavarzi *et al.* 2007; Limpens *et al.* 2008). However, restoration can be expensive and does not necessarily restore the carbon dynamics to the previous state (Cagampan and Waddington 2008). Conservation of peat through reduction of drainage and fires are therefore the highest priorities (Parish *et al.* 2008). Modification of agricultural practices in peatland is also important, as is the management of natural forest in peatland areas (Hoojier *et al.* 2006; Parish *et al.* 2008).

Currently, there is very limited scope for inclusion of wetland or peatland in carbon accounting through the UNFCCC. The only area of overlap is where the conversion of wetland areas is prevented due to carbon accounting in other ecosystems, such as for forested peatland (Henschel *et al.* 2008). However, recent increased estimates of the global peatland store have emphasised the need to manage peat, for resilience to increasing temperatures (Hoojier *et al.* 2006; Jaenicke *et al.* 2008; Parish *et al.* 2008; Uryu *et al.* 2008), through processes such as reduced degradation and maintenance of the water table (Ise *et al.* 2008).

Although there is growing literature in this area, the knowledge of the role of wetlands in the carbon cycle is still constrained and would appear to require further research.

3.4.5 Geo-engineering techniques

All of the mitigation policies discussed thus far have been biodiversity-based. This is not strictly the case for geo-engineering techniques, but they are included here as they involve manipulation of the natural environment, particularly the marine environment, to increase the carbon storage and sequestration capacity.

The IPCC AR4 reports that little is known of effectiveness, costs, or side effects of geo-engineering techniques such as carbon capture and storage and iron fertilisation (IPCC 2007).

3.4.5.1 Carbon capture and storage

3.4.5.1.1 Mitigation potential

Carbon capture and storage (CCS) is thought by many to be the best option for large-scale reductions in emissions from fossil fuel consumption, and involves the capture, liquefaction, and injection of CO_2 into geological formations or the ocean (Berry *et al.* 2008; Lal 2008). Technology for this process is available, and research and development in this field is increasing (Figueroa *et al.* 2008). Geological CCS can take the form of injection into coal seams, oil wells, stable rock strata, or saline aquifers (Lal 2008), whereas oceanic CCS involves injection of CO_2 into the deep sea, or into the seafloor of shallow seas (Huesemann 2006; Yamada *et al.* 2008).

The rationale behind oceanic CCS is that although oceans have the capacity to store several thousand GtC (Lal 2008), CO_2 is transferred into the deep ocean only at rates of 2 GtC per year (Huesemann 2008). Geological formations are also considered to have significant storage capacity, with the potential for 2,000 $GtCO_2$ reductions in emissions.

3.4.5.1.2 Biodiversity impacts

There has been more concern raised over the potential environmental impacts of ocean CCS than that of storage in geological structures, although there is the potential for leakage into aquifers and degradation of subsurface biodiversity, and issues of land-use change (Berry *et al.* 2008; Koornneef *et al.* 2008).

The risks of carbon capture and storage are not well known (Damen *et al.* 2006; Shepherd *et al.* 2007). Injection of CO_2 into the deepwater will alter ocean chemistry and could have significant consequences for marine organisms and ecosystems in the deep sea (Thistle *et al.* 2007; Lal 2008), with varying regional impacts (Watanabe *et al.* 2006). Deepwater fish have been shown experimentally to be more sensitive to environmental perturbations than shallow water species (Ishimatsu *et al.* 2006), and deep-sea injection into the seafloor could result in high rates of mortality for sediment dwelling organisms such as flagellates, amoebae, and nematodes (Barry *et al.* 2004). Potential impacts on bacteria have also been noted (Yamada *et al.* 2008). Leakage from carbon storage on the sea bed could also have significant impacts on communities in coastal and shelf seas (Widdicombe and Needham 2007). Increased acidification of oceans through leakage from volcanic vents has been shown to have large scale impacts on marine ecosystems (Hall-Spencer *et al.* 2008).

Conversely, if CCS has the potential to contribute to mitigation of climate change, it could have a positive overall impact for marine ecosystems such as through reduction of larger scale acidification impacts of global climate change (Magi 2008), through limited sea level rise, as well as migitating changes in sea surface temperatures with benefits to coral reef systems.

3.4.5.2 Ocean iron fertilisation

3.4.5.2.1 Mitigation potential

The option to increase the sequestration capacity of the oceans through iron fertilisation is based on the premise that adding trace amounts of iron will lead to phytoplankton blooms, higher productivity, and therefore increased sequestration (Smetacek and Naqvi 2008). This is receiving increasing attention, particularly through the private sector (Leinen 2008), but requires more extensive fieldwork and modelling before the mitigation potential could be adequately assessed (Lampitt *et al.* 2008; Buesseler *et al.* 2008).

Currently, the mitigation potential is uncertain (Buesseler and Boyd 2003; Gnanadesikan 2003), and it has been estimated that it would require fertilisation of an area the size of the entire Southern Ocean to sequester 3 per cent of current carbon emissions (Buesseler and Boyd 2003). It has also been suggested that the high sequestration efficiency determined in some pilot studies should not be taken as an indication that iron fertilisation will be efficient (Tollefson 2008). Indeed, as models have become more developed, the projected mitigation potential of iron fertilisation has dropped, and the likelihood that fertilisation will lead to the release of other GHGs such as N_2O has increased (Buesseler *et al.* 2008; Denman 2008; Jin *et al.* 2008; Krishnamurthy *et al.* 2008; Law 2008; Upson 2008). There is growing consensus that even the best case scenario for sustained fertilisation would have only a minor impact in sequestration terms (Zahariev *et al.* 2008). One study has found that approximately 75 per cent of iron added in fertilisation experiments is lost very rapidly, and each large-scale fertilisation has a persistence of only one year (De Baar *et al.* 2008). There is also a lack of an appropriate regulatory regime to monitor ocean fertilisation activities (Rayfuse *et al.* 2008).

It has also been suggested that although fertilisation schemes enhance uptake of CO_2 by phytoplankton, they do not facilitate the sinking of organic carbon into the deep ocean that is required for effective sequestration (Shepherd *et al.* 2007).

3.4.5.2.2 Biodiversity impacts

The potential environmental consequences of iron fertilisation are largely unknown but could be significant; partly because the scale would involve interference in natural productivity across immense expanses of ocean, which could have profound implications for marine ecosystems (De Baar *et al.* 2008; Cullen and Boyd 2008). Large-scale eutrophication from disruptions to nutrient cycling could cause deep ocean anoxia, shifting microbial community structure (Huesemann 2008). In addition, there could be serious implications on the food web structure and dynamics (Shepherd *et al.* 2007; Cullen and Boyd 2008). This could also have implications for fisheries (Parks 2008a; Parks 2008b). Some initial iron fertilisation studies have resulted in a trophic shift in the phytoplankton assemblage, favouring large diatoms (Denman *et al.* 2006; Henjes *et al.* 2007). In addition, carbon sequestered in the ocean has the potential to mineralize and increase ocean acidification (Matsumoto 2006), which has profound impacts on marine ecosystems (Cao 2008). This has led the International Maritime Organization to conclude that 'knowledge about the effectiveness and potential environmental impacts of ocean fertilisation is currently insufficient to justify large-scale operations' (Huesemann 2008).

Although most studies report negative impacts, it has been suggested that they have been based on worst case scenarios, and that iron fertilisation could boost krill populations and therefore the food supply of marine mammals such as whales (Smetacek and Naqvi 2008).

The IPCC AR4 reported that the potential for ocean fertilisation was largely unknown, but was not promising. Information published since the AR4 supports this claim by suggesting limited mitigation potential and likely large-scale impacts on oceanic food webs.

3.4.5.3 Nitrogen deposition

3.4.5.3.1 Mitigation potential

Increases in nitrogen deposition have been predicted to increase the size of terrestrial and marine carbon sinks (Karl and Letelier 2008), enhancing carbon uptake in forest ecosystems, with a lower impact on ocean sink strength. Combined, the land and ocean sinks may sequester an additional 10 per cent of anthropogenic carbon emissions by 2030 owing to increased nitrogen inputs, but a more conservative estimate of 1 to 2 per cent is more likely (Reay *et al.* 2008). Nitrogen fertilisation of the ocean suffers from the same issues as that of iron fertilisation, with limited potential for mitigation and high potential for biodiversity impacts (IPCC 2007). In addition, production of fertilizer requires fossil fuel use, further reducing emission reduction capacity (Glibert *et al.* 2008).

3.4.5.3.2 Biodiversity impacts

The biodiversity impacts of urea fertilisation are similar to those outlined for iron fertilisation. It is likely to change species composition, favouring those species that preferentially use urea as a nitrogen source, and could stimulate growth of toxic dinoflagellates, which could contribute to hypoxia (Glibert *et al.* 2008). Indeed, the impact of increasing nitrogen deposition on natural ecosystems is poorly understood (Dalal and Allen 2008), and the literature suggests that such impacts should be thoroughly assessed before employing these techniques for carbon management (Glibert *et al.* 2008).

3.5 RENEWABLE ENERGY

The renewable energy sector has been developed considerably in recent years in an effort to reduce reliance on fossil fuels. Although renewable energy projects are intended to reduce the impacts of climate change, they can also have impacts upon biodiversity (Paterson *et al.* 2008). Biofuels are included in this section as they are included in governmental renewable energy policies, although it is recognised that they are also a form of land management under the LULUCF.

3.5.1 Biofuel production

The production of liquid biofuels, namely ethanol and biodiesel, has been increasing rapidly in recent years (Gallagher 2008). The main crops used in ethanol production are maize and sugar cane, with rapeseed and oil palm used to produce biodiesel. Biofuels currently make up less than 1 per cent of the global energy supply (FAO 2008).

3.5.1.1 Mitigation potential

The production of liquid biofuels has been greatly incentivised over recent years as a tool for climate change mitigation and energy production. Biofuel plantations are intended to provide renewable energy for transport; reducing reliance on fossil fuels whilst sequestering carbon from the atmosphere. However, recent research has suggested that production of energy crops may do little to mitigate climate change; even increasing emissions by as much as 17-420 times that of fossil fuels (Righelato and Spracklen 2007; Fargione *et al.* 2008; Searchinger *et al.* 2008). This is largely due to the direct conversion of carbon rich natural ecosystems, such as tropical forest, into biofuel plantations, or the indirect conversion through displacement of agricultural activity into such lands (Gallagher 2008). In general, the carbon that can be sequestered through restoring forests is thought to be greater than the emissions avoided through production of liquid biofuel (Righelato and Spracklen 2007), especially when full life cycle analysis is undertaken (Scharlemann and Laurance 2008).

Clearly, the biofuel feedstock used and the location of plantations will determine the potential for climate change mitigation (Scharlemann and Laurance 2008). Gibbs *et al.* (2008) have shown that the replacement of carbon rich ecosystems by biofuel plantations results in carbon emissions over decades and centuries, whereas plantations on degraded land can have immediate carbon savings. However, there is still considerable disagreement on definitions for 'degraded land' (RSC 2008) and recent research has suggested that the global bioenergy potential for such land is less than 8 per cent (Campbell *et al.* 2008b) and 5 per cent (Field *et al.* 2008) of current energy demand globally.

The IPCC AR4 identified second generation biofuels as one of the key future technologies for mitigation, where non-feedstock crops are used for energy production in combination with processing technologies. However, the potential of 'second generation' biofuel for climate change mitigation is also doubtful (Gallagher 2008), particularly if it involves the large areas of land-use change projected (FAO 2008); and the technology is not yet available. The climate change mitigation potential of production of biofuel from micro-algae remains to be seen, but there is some optimism (Jenner 2008; Wang *et al.* 2008).

3.5.1.2 Biodiversity impacts

Biofuel production has considerable impacts on biodiversity when it results in direct conversion of natural ecosystems (RSC 2008) and indirect displacement of agricultural land into natural ecosystems (Gallagher 2008). Biofuel production is largely driven by government targets and subsidies, and future production is expected to increase by 10-15 per cent (Eickhout *et al.* 2008). Global land availability depends in part upon future technological advances, but pessimistic scenarios predict a 'land deficit' of approximately 200 Mha by 2020-30 (Roberts and Nilsson 2007; Gallagher 2008), even when not taking into account the land requirements of other climate change mitigation policies such as afforestation.

The global scale of conversion of natural ecosystems to biofuel plantations is unknown due to the difficulty of accounting for displaced agricultural conversion (Gallagher 2008), such as that of soybean into the Amazon (Martinelli and Filoso 2008). Biofuels can be produced with the greatest efficiency in the tropics, and the lack of economic incentives for the conservation of natural ecosystems leaves them vulnerable to replacement with biofuel crops (Doornbosch and Steenblik 2007). The expansion of oil palm in Indonesia and Malaysia, which account for 86 per cent of global oil palm production, is the most cited example of this. Although it is difficult to directly attribute forest loss to biofuel plantations, recent estimates have calculated that 55-59 per cent of oil palm expansion in Malaysia, and 56 per cent in Indonesia occurred at the expense of forest (Koh and Wilcove 2007b). Other studies have estimated that 27 per cent of forest loss has occurred as a result of oil palm plantations since 1982 (Uryu *et al.* 2008). Because Indonesia contains some three-quarters of Southeast Asia's remaining primary forests, the continuing loss of its primary forests would be disastrous for the region's biodiversity (Koh and Wilcove 2007a).

Issues of land conversion aside, the biodiversity impacts of biofuels are similar to those for plantation forest discussed previously. Plantation forest supports significantly lower levels of biodiversity than natural forest (Koh and Wilcove 2007b), and oil palm plantations support lower levels of biodiversity than other tree crops (Fitzherbert *et al.* 2008). Only 15 per cent of species recorded in primary forest are also found in oil plantations, and forest fragments between biofuel plantations supported less than half the species of continuous forest (Fitzherbert *et al.* 2008). Although oil palm is cited as one of the major threats to biodiversity, there is very little published research on this topic (Turner *et al.* 2008). Again, the biodiversity impacts of biofuel production will depend upon the previous land use and the crop used (RSC 2008; Scharlemann and Laurance 2008).

The potential biodiversity impacts of second generation biofuels are largely unknown, but recent reports have questioned their capacity to reduce biodiversity impacts as they do not produce beneficial co-products; and where all 'waste' biomass is removed for fuel this can impact on soil fertility (Eikhout *et al.* 2008, Gallagher 2008). However, recent research has suggested that native prairie species grown on degraded land can produce 238 per cent more energy than monocultures, whilst providing biodiversity benefits (Tilman *et al.* 2006); whilst woody biofuels on degraded land using native tree crops can provide ecosystem services (RSC 2008). However, monoculture non-food crops such as switchgrass have invasive traits and could have significant biodiversity impacts (RSC 2008).

3.5.2 Wind farms

The global growth rate of wind power in 2005 was 24 per cent, up 4 per cent from 2004 (Kikuchi 2008), and if this trend continues 120,000 MW is projected to be installed worldwide by 2010 (WWEA 2006). Germany is currently the highest user of wind energy (Huppop *et al.* 2006). In the UK, the government target is to have 7-8 per cent of its energy derived from wind power by 2010, which would require the installation of 2000 turbines onshore and 1500 offshore (Drewitt and Langston 2006).

Wind farms must be sited on open, exposed areas with high average wind speeds to be effective, which means that proposed sites are often upland, coastal and offshore areas (Huppop *et al.* 2006). Birds can be affected by wind turbines though collision with turbine blades, displacement from migration routes, and direct habitat loss (Allison *et al.* 2008; Fielding *et al.* 2006).

Mortality of birds as a result of wind turbines has been documented by a number of recent studies (Allison *et al.* 2008; Barclay *et al.* 2007; Barrios and Rodriguez 2004; Drewitt and Langston 2006; Everaert and Stienen 2007; Hüppop *et al.* 2006; Kikuchi 2008). However, rates vary greatly between studies. Some of the highest levels of raptor mortality were recorded at Altamont pass in California (Orloff and Flannery 1992), where annual rates of 75 golden eagles and 400 griffon vultures for the wind farm were recorded, and in Navarre, Spain, wind farms killed 7,150 birds in one year. Studies at other sites have not recorded such high rates (Fielding *et al.* 2006), and these high rates at Altamont and Navarre are thought to reflect site selection across known migration routes.

Fielding *et al.* (2006) suggest that site impact assessments at the planning stage could greatly reduce collision risk. In particular, wind farms should be located away from migration routes where possible. Altered bird behaviour to avoid wind turbines can produce a secondary form of habitat loss, with potential displacement away from migration routes, breeding grounds and feeding areas (Fox *et al.* 2006), and bird densities have been shown to decline rapidly with proximity to turbines (Fox *et al.* 2006; Larsen and Guillemette 2007). Habitat loss through the development of the turbine sites is low, and is estimated at 2-5 per cent of the total development area (Fox *et al.* 2006).

Bat fatalities from wind turbines have also been recorded (Arnett *et al.* 2008) and there is some evidence that turbine noise can modify anti-predator behaviour (Kikuchi 2008; Rabin *et al.* 2006), although have no impact on density (Lucas *et al.* 2008). In addition, noise from offshore wind farms can impact upon marine mammals (Carstensen *et al.* 2006; Madsen *et al.* 2006).

Recent evidence suggests that wind farm impact studies lack an evidence base (Stewart *et al.* 2007), and have minimal impacts on biodiversity (Devereux 2008). Although it is clear that environmental impact assessments should be conducted in land use planning, wind energy appears to have low impacts on biodiversity compared to other renewable energy options (Berry *et al.* 2008).

3.5.3 Nuclear power

Nuclear energy produces greenhouse gases through mining, enrichment, reactor construction, waste disposal and transport; with emissions higher than for wind and hydropower, and about the same as solar power (Lenzen 2008).

Environmental impacts of nuclear power can be extremely high in the event of leakage of nuclear material. Chernobyl reduced species richness, abundance, and population density of wildlife (Clouvas *et al.* 2007; Moller and Mousseau 2007; Moller *et al.* 2007), with mutations from radiation spread amongst the wider population. In addition, it has been reported that forests are acting as a 'sink' for radioactive isotopes from Chernobyl (Clouvas *et al.* 2007).

In the course of normal operations, there appears to be only minimal radioactive isotope release (Virbickas and Virbickas 2005; Jean-Baptiste *et al.* 2007; Eyrolle *et al.* 2008; Gauthier-Lafaye *et al.* 2008) although some changes in species diversity have been noted (Balciauskas 2005). However, uranium mine ponds can contaminate groundwater and soil, affecting ecosystems with either radioactivity or high levels of arsenic (Antunes *et al.* 2007; Carvalho and Oliveira 2007). As with other types of mining, degradation of habitat has negative impacts on biodiversity. Levels of radiation seen as safe for man have been found to be damaging for many other species (Fesenko *et al.* 2005).

Construction of reservoirs for water cooling can lead to changes in fish and bird diversity (Contador 2005), with fish mortality from cooling water intake costing approximately €0.5 million per year (Greenwood 2008). Release of heated water also reduces algal species diversity (Kim *et al.* 2008), alters fish species composition, and enhances water eutrophication (Contador 2005; Virbickas and Virbickas 2005).

3.5.4 Hydropower

Currently, hydropower provides about 20 per cent of the world's electricity supply and more than 40 per cent of the electricity used in developing countries (Bakis 2007). There are two main types of hydropower; large-scale (dam) hydropower, and small-scale 'run-of-river' projects. It is considered to be a sector with vast unexploited potential in developing countries. Hydropower does emit CO_2 through dam construction and algal build-up (Kaygusuz 2004; Ponseti and Lopez-Pujol 2006), and it has been suggested that dams in tropical areas actually cause more GHG emissions than savings (Fearnside 2005b).

The environmental issues involved with hydro-electric dams include habitat destruction, barriers to terrestrial migration, barriers to fish migration, reduced sedimentation and changes in flow altering downstream ecosystems, and fish mortality in turbines (Berry *et al.* 2008). Dams cause severe disruption to ecosystems through construction and flooding of large areas, which can completely alter the species composition of the area (New and Xie 2008). Large dams appear to be one of the most damaging renewable energy policies (Berry *et al.* 2008).

3.5.5 Solar power

Currently solar energy provides only 0.2 per cent of the world's energy, and production costs are still high and efficiency relatively low. However, with advances in technology, solar power is predicted to provide the world with large amounts of energy in the future (Fritsche *et al.* 2008). Despite this, there is little literature available on environmental impacts, which include: risk of water pollution through leaks of heat transfer fluid and coolants, disposal of toxic material, land requirements, and thermal pollution (Tsoutsos *et al.* 2005; Huesemann 2006; Mohr *et al.* 2007). Although land-use requirements are not large, large-scale plants can impact natural ecosystems through competition for land use on degraded or semi-natural lands (Berry *et al.* 2008).

3.5.6 Geothermal energy

Geothermal energy supplies 0.4 per cent of the global energy supply, and has the potential to increase its share as a relatively 'clean' and resilient energy source (Berry *et al.*

2008). Potential impacts on biodiversity include land subsidence, chemical pollution of waterways, construction impacts, soil erosion, and noise disturbance, but it is in general considered to have low biodiversity impacts in comparison with other renewable energy sources (Thórhallsdóttir 2007; Berry *et al.* 2008).

3.5.7 Tidal energy

Tidal energy is considered to have potential as a renewable energy source. It consists of either movement of water through turbines or tidal barrages. It therefore has the potential for a number of impacts on biodiversity through changes in flow, fish mortality, changes in salinity, altered sediment deposition, underwater noise, impacts on migration corridors, and physical disturbances (Berry *et al.* 2008; Boehlert 2008; Prater 2006). These can have both short- and long-term implications for biological communities, which are likely to depend upon their resilience to disturbance (Gill 2005; Cada *et al.* 2007). Modification of water circulation and currents is likely to have significant impacts on the benthic habitat (Boehlert 2008), and can change fish community structure. In addition, there are concerns that changed wave and noise patterns and physical structures will have an impact on cetaceans (Boehlert 2008), and birds (Clark 2006).

Due to potential impacts on coastal communities, tidal energy projects should involve substantial environmental assessments (Gill and Kimber 2005), and tidal barrages are likely to have the biggest impact (Berry *et al.* 2008). However, recent research for Scotland has suggested that net impacts for the environment could be positive (Allan *et al.* 2008).

3.6 OPTIONS FOR MULTIPLE BENEFITS THROUGH MITIGATION STRATEGIES

It is clear from the literature reviewed that climate change mitigation policy has the potential to impact biodiversity both positively and negatively. Currently, many renewable energy projects are being planned without consideration for biodiversity impacts; as are some land-based mitigation strategies such as monoculture plantations (Berry *et al.* 2008). However, due to the important role of biodiversity in the carbon cycle, it is clear that the potential exists to develop 'win-win' mitigation policies that are beneficial for both climate change mitigation and biodiversity (Paterson *et al.* 2008). In addition, it is important that mitigation strategies, such as the development of large dams or biofuel plantations, do not increase the vulnerability of ecosystems to climate change (Berry *et al.* 2008).

In recent literature, REDD has been cited as a mitigation strategy that can have benefits for climate change mitigation, biodiversity, and people (Ravindranath 2007; Eliasch 2008; Nepstad *et al.* 2008), as it promotes the conservation of high carbon and biodiversity rich forest (Righelato and Spracklen 2007). Good agricultural practices also have the potential to reduce biodiversity impacts and mitigate climate change, whilst also acting as an adaptation strategy to increase the resilience of agricultural land to climate change impacts (Rosenzweig and Tubiello 2007; Berry *et al.* 2008; Lal 2008); particularly through reducing degradation of soil (Lal 2008). In many cases, afforestation schemes involve monoculture plantations of fast growing trees (Bekessy and Wintle 2008), but where multispecies plantations are established on degraded land they can have biodiversity benefits, as can agroforestry schemes (Berry *et al.* 2008). This can act in support of both the Convention on Biological Diversity (CBD) and the UN Convention to Combat Desertification (UNCCD) (Cowie *et al.* 2007a).

Where mitigation strategies are beneficial to biodiversity, they can also reduce vulnerability to climate change impacts, maintain future capacity for climate change mitigation, and promote adaptation (Betts 2007). It is becoming clear that we are already committed to significant climate change impacts; and it is therefore important to identify areas in which synergies between biodiversity-based mitigation and adaptation strategies exist (Ravindranath 2007).

There is evidence that biodiversity-based mitigation strategies can also assist in adaptation to climate change through the maintenance of ecosystem services such as water, and flood control; and this was addressed in the previous section.

3.7 CONCLUSION

Research since the IPCC AR4 has served to strengthen the conclusion that biodiversity is an integral part of the carbon cycle, and important in mitigating climate change. A large amount of carbon is stored within ecosystems, although estimates are still uncertain and appear to underestimate the carbon stored in soils, particularly in peat. In addition, ecosystems are continually sequestering carbon from the atmosphere; acting as a net sink for anthropogenic greenhouse gas emissions. It has been estimated that terrestrial ecosystems sequester 2.1-3 Gt of atmospheric carbon annually, approximately 30 per cent of all anthropogenic CO_2 emissions. Marine ecosystems sequester large amounts of carbon through phytoplankton at the ocean surface.

Changes in land use, primarily through deforestation, are releasing significant amounts of the terrestrial carbon store to the atmosphere; accounting for 20 per cent of greenhouse gas emissions. Carbon loss from soil could be comparable to that lost from biomass through deforestation; and emissions from peat could account for 10 per cent of global emissions.

Recent evidence suggests that such damaging land-use practices, in combination with climate change, could reduce the capacity of the carbon sink over century timescales, providing a positive feedback loop to the climate system. Our understanding of the scale of feedbacks from land-use change is increasing, but still weak, and it is important to better understand the role of natural ecosystems and management practices in the carbon cycle. Recent research has highlighted the damaging feedback loops between climate change and land degradation in peatlands and the Amazon rainforest. Although the scale of this feedback is still uncertain, research suggests that the inclusion of natural ecosystems in climate policy is vital if we are to achieve the target specified in the UNFCCC objective of limiting climate change to a 2ºC rise in global average temperatures.

Improved land-use management practices can reduce the emissions from land-use change and increase the sequestration capacity of the biosphere; with the capacity to make a significant contribution to climate change mitigation. The IPCC AR4 estimated that over the next century, 15-40 per cent of total abatement requirements could be met through a combination of reduced loss of carbon stores, and sequestration policies. The use of ecosystem-based mitigation policies can also contribute to sustaining a variety of ecosystem services including biodiversity conservation.

All recent evidence suggests that there appears to be high potential for cost-effective emissions reductions from a mechanism for Reduced Emissions from Deforestation and Forest Degradation in developing countries (REDD). The exact mechanisms for REDD have still to be decided, but there is general agreement that a well-designed mechanism could reduce deforestation rates significantly. A halving of deforestation rates could account for up to 12 per cent of emissions reductions required by 2100, and economic modelling has suggested that REDD will be a competitive, low-cost abatement option. Moreover, a successful REDD mechanism has the potential to deliver significant additional benefits, contributing to biodiversity conservation at both the species and ecosystem level, as well as potentially helping to sustain livelihoods and reduce poverty. In addition, there is the potential for REDD to reduce vulnerability of forest to climate change impacts, and maintain the capacity of the sink.

There is significant uncertainty attached to the level of carbon sequestration that can be achieved through afforestation and reforestation; and the potential for mitigation in this sector, particularly on decadal time scales, is often questioned. Whilst there is significant potential in increasing the capacity of the natural carbon sink, particularly in the tropics, there is a need for more integrated study of how land management changes may affect climate change and biodiversity. Improved agricultural management has significant potential to be positive for both climate change mitigation and biodiversity if best practice management techniques are employed.

The role of improved soil management in climate change mitigation should be emphasised as it can be considered the area with the highest potential outside of forest activities. Currently, there is limited scope for inclusion of wetland or peatland in existing mechanisms, despite new evidence of their high carbon stores and contribution to global emissions. Evidence suggests that improved management of peatlands could substantially reduce emissions and reduce vulnerability to climate change impacts.

Geo-engineering techniques for mitigating climate change are not strictly ‘ecosystem-based’, but they do involve manipulation of the natural environment, particularly the marine environment, to increase the carbon storage and sequestration capacity; and could have substantial impacts on biodiversity. All evidence questions the capacity of ocean iron fertilisation, and highlights significant biodiversity impacts. Carbon capture and storage appears to have mitigation potential, but could have significant consequences for marine organisms and ecosystems in the deep sea. Renewable energy projects can also have impacts on biodiversity; particularly biofuel production and the construction of large dams.

It is clear from the literature reviewed that climate change mitigation policy has the potential to impact biodiversity both positively and negatively. Currently, many renewable energy projects are being planned without consideration for biodiversity impacts; as are some land-based mitigation strategies such as monoculture plantations. In particular, mitigation policies that reduce the capacity for adaptation to climate change should be avoided. However, due to the important role of ecosystems in the carbon cycle, it is clear that the potential exists to develop ‘win-win’ mitigation policies that are beneficial for both climate change mitigation and biodiversity; particularly through forest conservation, improved agricultural management, and land-use planning. Furthermore, in view of ecosystems contribution to adaptation strategies, policies can be found that in addition prove beneficial to climate change adaptation and effectively produce a ‘triple win’ policy.

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