IPCC WGII Fourth Assessment Report – Draft for Expert Review

Chapter 4: Ecosystems, their Properties, Goods, and Services

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Executive Summary

- TAR findings that climate change is affecting ecosystems can be confirmed, but with evidence of impacts on the structure, function and species composition of ecosystems generally confined to local scales: Causal mechanisms are better understood, and models and observations have been matched in a small number of cases (medium confidence).
- Observed climate change impacts seem in some cases to be more rapid than previously projected by ecological models. However, it is not yet clear if this implies a higher sensitivity of ecosystems than previously assumed. This higher sensitivity might merely be the result from previously poorly represented changes in climate variability or may be caused by other, additional drivers/stresses, which interact synergistically with climate changes to induce ecosystem responses (low confidence).
- Observations of species range responses globally (“fingerprints of climate change”) seem consistent with projections, although inadequate representation of time-lag effects may confound projections of increasing risk of species extinctions (high confidence).
- Species range modeling using mainly IPCC IS92a, SRES A and B family climate change scenarios projects broad scale changes, mainly reductions, in species range sizes by 2050. In some cases, species persistence requires species migration rates much higher than is theoretically possible or the adaptive capacity of the species may be exceeded. Consequently, range size reduction will lead to a substantially increased risk of extinction for between 15% and 40% of indigenous biota during this century (medium confidence).
- Many modelling approaches of species’ and ecosystem responses to climate change suffer from determinism, which generally prevents clear estimates of uncertainty.
- Fragmentation of natural habitats and other human pressures significantly reduces the potential for species migration in response to changing climate, and is therefore likely to limit natural adaptation processes (high confidence). Moreover, conservation efforts are often poorly prepared for climate change especially because the resilience of species and ecosystems in many protected areas may be exceeded under most projected climate change scenarios (high confidence).
- The role of fire in controlling ecosystem structure and function at a global scale, previously poorly recognised, and the associated dependence of fire regime on climate, indicates the potential for major rapid changes in ecosystem structure, function and species composition, and feedbacks to the climate system that have yet to be fully quantified (high confidence).
- The increased susceptibility of ecosystems to alien invasive species under climate change scenarios is a major uncertainty for both biodiversity and ecosystem functioning (medium confidence).
- Climate change may foster some pests, diseases, or pathogens and may reduce the occurrence of others. The associates risks or possibilities of gains are largely not well understood. The associated uncertainty is considerable and with little prospect for improvement unless costly and extensive research is conducted (medium confidence).
- The ecological impacts of climate change mitigation such as aforestation, marine carbon sequestration, marine iron fertilization are a key uncertainty with respect to their impacts on biodiversity and ecosystem functioning (high confidence).

Current sensitivities, vulnerabilities and trends

There is ever-increasing evidence of changes in terrestrial species behaviour, geographic range and ecosystem function, attributable to the early signs of climate change. Evidence suggests that climate change may induce the extinction of local populations (e.g. of butterflies, frogs and several other species). Many insect species, on the other hand, have the ability to respond quickly to changes in climate through geographic range shifts. Marine food web systems are undergoing significant and fundamental changes, triggering major reorganization of warm temperate marine ecosystems. In contrast, the diversity of colder temperate, sub-Arctic and Arctic marine species has decreased. Coral reefs, the most diverse marine ecosystems, have become one of the clearest indicators of climate change's ecological impacts, while on land, shifts in fire frequency provide clear evidence of changing ecosystem function.

Assumptions about future climate

Changes in the long-term mean state of climate is likely to impose important constraints on numerous ecosystems, but the most significant impacts of climatic change may be driven by shifts in the intensity and the frequency of extreme weather events. Drought plays an important role in forest dynamics, and has been responsible for pulses of tree mortality in the Argentinian Andes. Conversely, in the many other parts of the world, particularly continental mid- and high-latitude regions, wind, snow and frost are the main causes of forest damage. Recent heat-wave events in Europe have both short-term and long-term implications for vegetation; combined heat-stress and drought can damage vegetation, with sometimes irreversible consequences. Desertification processes advance at higher rates during periods of extreme events, such as in drought years. Our ability to model the impacts of climate extremes is incomplete, and is complicated by species-specific responses.

Future impacts and key vulnerabilities

There is a strong correlation between areas of high human activity and areas of high species richness, with profound negative consequences for biodiversity. This implies that future synergistic impacts of direct human-induced stresses and climate change will induce significant biodiversity loss. Fragmentation of habitat, for example, reduces the ability of natural species to adapt to climate change, and loss of habitat due to climate extremes has been shown to lead to greater conflict between indigenous species and humans. Synergies between climate change and other human-induced changes may not be well estimated by simply summing the impacts of the individual stresses, but our predictive capabilities are currently limited in this regard. Many marine ecosystems and species are at risk due to current trends towards non-sustainable harvesting. Stocks of the large top predators are declining, leading to a rate of decrease in freshwater diversity that is even greater than in many terrestrial ecosystems. Greatest effects have been observed in fresh waters within tropical regions, particularly South and South East Asia, and in dryland areas.

Species with large area requirements are under greatest threat of extinction when habitats become fragmented, as are species with highly fluctuating populations and low rates of population growth. The most vulnerable species may be found at the interfaces (ecotones) between ecosystems. Species that reproduce slowly and disperse poorly, and those which are isolated or are highly specialized, are likely to be highly sensitive to seemingly minor stresses. At the ecosystem function level, it is likely that systems close to thermal or water availability limits will respond most readily and rapidly to climate change. Satellite observed changes in vegetation green-up, for example, indicate a rapid response to a warming-induced longer growing season.

Disturbance by fire is increasingly recognized as a globally important process potentially exacerbating climate change effects in terrestrial ecosystems. The drivers of fire are all influenced by components of climate change. Exclusion of fire by suppression strategies seems relatively ineffective in reducing fire hazard, and therefore human systems need to adapt to fire rather than expend resources mitigating fire occurrence. Human introduction of fire into ecosystems has rapidly transformed forests to flammable shrublands and grasslands with massive biodiversity implications. Some communities have evolved to be flammable, and geographic shifts in key species will cause the entire community to shift in response – one of the few powerful exceptions to the rule of individualistic species response to climate change. Invasive alien species (IAS) and climate change are likely to act synergistically to the detriment of natural ecosystems and indigenous biota, but theory predicting outcomes is poorly developed. Climate change may well provide additional triggers to accelerate invasions, by creating disturbances within the ecosystem that quiescent IAS can exploit. Changes within native communities caused by disturbance to ecosystem integrity by climate change may lead to a general increase in the invasive potential of alien species.

Adaptive responses

Monitoring climate change ecosystem response is important to facilitate adjustments in management strategies. A primary management adaptation is to reduce as many ancillary stresses on the resource as possible. Maintaining widely dispersed and viable populations of individual species minimizes the probability that localized catastrophic events will cause extinction. Theory on conservation strategies under changing climate is developing, and proposes consideration of protected area placement in relation to possible shifts in species and ecosystem distributions. In highly fragmented habitats, the placement of conservation areas on a north-south axis may enhance movements of habitats and wildlife by providing migration corridors or stepping stones. Managers may use prescribed fires and other techniques to reduce fuel load and the potential for catastrophic fires. For selected highly threatened species, captive breeding and translocation, possibly also combined with habitat restoration, may be appropriate strategies, but are likely to present significant cost implications, especially for plants. Translocation of wildlife between areas within their natural range might also be used to mitigate the effects of small to moderate climate change.

4.1 Introduction

4.1.1 Scope

4.1.1.1 Ecosystems

The ecosystem concept defines interacting groups of organisms and their environment as functioning units, allowing the study of their interdependences. The ecosystem concept is scale-free, and can comprise the entire globe (addressed for example by the field of global biogeochemistry) to small, well-circumscribed systems such as a single vernal pool, or even the group of micro-organisms colonizing a piece of detritus at the bottom of that pool. Ecosystems are now recognized as providing a fundamental support system for human well-being. Human society benefits directly from the goods provided by ecosystems, and in the past few decades, we have realized that ecosystems provide a range of services that also benefit human well-being.

This chapter discusses the general properties, goods and services of all ecosystems in the context of a changing climate, but with the emphasis on natural ecosystems (including pristine, and little or extensively managed ecosystems). Intensively managed ecosystems such as croplands or other agroecosystems are treated in chapter 5 (food, fibre). Inasmuch as recent climate change impacts on ecosystems are discussed in chapter 1, this chapter focuses on projections of future impacts as based on scenarios from WG I and chapter 2. Certain ecosystem goods and services are treated in detail in other sectoral chapters: chapters 3 and 6 (water), and 8 (health). Key findings from this chapter will be picked up by chapters 17 to 20. Region-specific aspects of ecosystems are also discussed in the regional chapters 9 to 16; this chapter focuses on the global to local aspects of ecosystems as grouped by commonly accepted natural functional classifications. Often ecosystems are delineated geographically, but this chapter will also consider functional and structural properties of ecosystems at a hierarchy of scales, namely biomes and communities, populations, and underlying ecophysiological processes.

The following ecosystems are discussed within this chapter (Table 4-1):

- Terrestrial ecosystems
  - Desert biomes
  - Grasslands and savanna biomes
  - Mediterranean system biomes
  - Forests and woodland biomes[1]
  - Tundra and other arctic biomes
  - Mountain Ecosystems
- Aquatic freshwater ecosystems
  - Freshwater lakes and rivers
  - Inland wetlands[2]
- Oceans, shallow seas and their ecosystems[3]

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[1] Deforestation was coordinated with chapter 5 and chapter 9 WG III

[2] Costal wetlands are covered in chapter 6

[3] Coastal systems and low-lying areas as well as coral reefs and atoll island systems are all covered in chapter 6

Table 4-1: *The major terrestrial biomes of the world addressed in this report, together with their NPP and carbon stores (extracted from Sabine et al., 2004).*

Biome	Area (km^2)	NPP ($PgC\ y^{-1}$)	Plant Carbon (PcC)	Soil Carbon (Pg C)
Tropical forest	17.5	20.1	340	692
Temperate forest	10.4	7.4	139	262
Boreal forest	13.7	2.4	57	150
Arctic tundra	5.6	0.5	2	144
Mediterranean shrubland	2.8	1.3	17	124
Tropical grassland/savanna	27.6	13.7	79	345
Temperate grassland	15.0	5.1	6	172
Desert	27.7	3.2	10	208
Total	120.3	53.7	650	2097

4.1.1.2 Biomes, ecosystems, population systems, and ecophysiology

This chapter follows a hierarchical approach in assessing impacts on ecosystems, noting however that the classification of levels in the hierarchy is somewhat arbitrary and constrained by the levels at which experimental work and modelling has been undertaken. Thus ecosystems may be defined at large spatial scales and long-term temporal scales (decades to millennia), i.e. the biosphere (ecosphere) at the global scale (centuries, millennia), biomes at the continental scale (decades to millennia), and large assemblages of organisms forming communities at the regional scale (years to centuries). Population systems on the other hand are functionally defined, often by key-stone species and facilitate the assessment of individual species responses to climate change (months to centuries). Threatened species as impacted by climate change will be discussed at this level together with possible consequences for overall biodiversity. Finally, fast ecophysiological responses (seconds, hours, days, months) of microorganisms, plants, and animals operate at small (microscopic) scales from a leave or organ to the cellular level, and are assessed where these impacts scale up to make a significant impact at higher spatial scales, or where the mechanistic understanding assists in assessing key thresholds in higher level responses.

Biomes are defined at the largest spatial scale and are typically delineated by broad geographic areas on the earth’s surface. Typical examples are forests or grasslands. A biome is characterized by a vegetation of a similar structural type and the animals living within that geographic area. There are many different classification schemes in use. In this assessment we distinguish 14 major biomes.

Biomes are structures within the biosphere, sometimes recognizable through remote sensing and usually are the result of long-term, self-organizing processes such as the formation of soils, a balanced nutrient capital, land forms and often typical vegetation forms. Biomes often consist of several ecosystem types and can be considered to be near equilibrium with respect to their biophysical and chemical environment given a relatively stable climate on the time scale of a few millennia and in the absence of major, external disturbance events.
Early geographers attempted to explain the distribution of biomes only in terms of climate control (Schimper, 1903), but several recent findings corroborate that disturbance regimes such as fire or insects, can influence vegetation structure and function sufficiently to negate the sole direct control by climate in many parts of the world. It has been found that globally, many ecosystems do not attain their potential biomass expected from rainfall and temperature

conditions, but that fire regime maintains a much lower biomass and vegetation structure (Bond and Van Wilgen, 1996; Bond *et al.*, 2005). Therefore, at a coarse scale, climate is the potential dominant controller of biome distribution, mainly through the effects of temperature and water balance (Whittaker, 1975), but that potential is significantly modified in as much as 30% of the vegetated surface of the earth by fire regime (Bond *et al.*, 2005 see also Fig. 4.2).

Biome borders are generally fuzzy and are connected by extensive intermediate forms, often termed ecotones, which are generally found at the interfaces between vegetation of different structural or compositional characteristics, such as forest edges. In these zones, ecosystems may be exposed to a combination of the factors that influence the surrounding. Because of this, some ecotonal areas have been suggested as useful early indicators of climate change impacts, notably the northern limits of the circumpolar boreal forest, since climate models project strongest warming at high latitudes.

Biomes are likely to be differentially sensitive to climatic change (e.g. Kirschbaum and Fischlin, 1996; Sala *et al.*, 2000b; Gitay *et al.*, 2001), with temperature-limited biomes prone to impacts of warming, and water-limited biomes prone to increasing levels of drought. Fire dependent biomes are often in a meta-stable state that can result in fairly rapid switches in vegetation structure under climate and other environmental changes (Bond *et al.*, 2005).

Communities are found at a level below biomes and are characterized by a particular species assemblage generally loosely associated under a given environmental condition. Exposed to climate change communities undergo changes which are called secondary succession, which may encompass changes in the dominance of particular species, e.g. species typically dominant at late successional phases may be replaced by species typical for early successional phases.

Communities are often characterized by a specific species composition and thus a specific diversity. A recent review of studies which link diversity and ecosystem functioning, and their implications to changes in ecosystem services (Millennium Ecosystem Assessment, 2005b p43-46) proposes that there are three kind of biotic changes that are key to understanding how an ecosystem's biota affects its response to environmental change: First, changes in community structure which imply a loss of key predators or mutualists such as pollinators, or other influential species, may alter the ecosystem functioning. Second, the loss of ecologically equivalent or redundant species can decrease the reliability or predictability of a given community. Thirdly, the loss of species that are ecological complements to one another can reduce the range of conditions over which the community can maintain the same or a similar functioning. Human-induced changes due to fragmentation, degradation, or conversion of habitats to managed habitats generally reduce taxonomic diversity. It is concluded, though speculatively, that theory and data suggest that the stability of ecosystem functioning, and hence the stability of the services we derive from ecosystems, are likely to be more sensitive to perturbations and more unpredictable if the diversity of a given community is reduced.

Land use change alters communities typically by affecting the locally most abundant species, which again changes the range of characteristics present in the community, the interactions among species, the number of species present in the regional species pool (including gene pools) and the spatial structure of the landscape (Millennium Ecosystem Assessment, 2005b p. 3).

Population systems are found at an intermediate level. They are typically characterized by a particular species and can in this respect be seen below the level of communities. However, geographically population systems may well exceed the confines of a community.

Distinguishing populations from other structures such as biomes and communities shifts the focus away from the primary producers, i.e. the plants, to that of higher trophic levels, i.e. consumers such as animals. Typically, populations of many animal species are highly vagile and may easily disperse or migrate from one ecosystem to another and tend to go beyond the level of a single ecosystem or even the biome. E.g. many migratory birds inhabit forests and grasslands, depending on the season and visit wetlands on their migratory routes.

Ecophysiological responses to climatic changes act continuously and on a small scale, and represent the primary response of individual organisms to environmental variability. Ecophysiological mechanisms underpin individual tolerance to environmental stress, and comprise a broad range of responses. In plants, enzymatic and biochemical responses facilitate the production of protective compounds that limit photo-oxidative stress, thermal stress and even screen solar radiation; stomatal behaviour facilitates responses to air and soil dryness and leaf temperature regulation; hormonal control regulates the balance between root and shoot growth, and resource allocation to reproduction, for example, is adjusted according to external cues or physiological status. In animals, respiration and other vital functions are controlled by ecophysiological responses often mediated by biochemical control, and reproductive responses, such as foetal abortion, may respond to environmental cues through biochemical regulation.

Thus ecophysiological responses define the absolute tolerance limits of individuals to environmental conditions, and may scale up to control species geographic range limits, and even ecosystem functions at a regional and global level. Possibly the best example of this is the role of stomata on the surfaces of plant leaves, whose behaviour in regulating water flux scales from individual leaf, to impacts through canopy to regional and even higher levels of ecosystem organization (Hetherington and Woodward, 2003).

4.1.1.3 Goods and services

Ecosystems provide many goods and services (Fig. 4.1). Many are of vital importance for the functioning of the biosphere as a life-support system, and provide the basis for the delivery of tangible benefits to human society. Some of these, such as water purification, are well known, but others, such as carbon sequestration, are less obvious and often hardly recognized, yet they are vital and contribute directly and indirectly to human welfare by providing society not only with food, energy, and a multitude of materials, but also by maintaining the physical and chemical balance of our environment. In the context of climate change, the carbon sequestration response of ecosystems has become a key ecosystem service that is formally recognised as a mitigation strategy for climate change. It is now recognised that terrestrial ecosystems in all likelihood absorb and sequester on average about 25% of anthropogenically emitted CO_2. Some ecosystems services are not of direct use for humans, yet are a necessary prerequisite for all goods and services of interest for humans, thus serving humans indirectly. These are termed supporting services, and comprise the self-organizing processes resulting in soil formation, local climate modifications, nutrient cycling, all needed for primary and secondary etc. productions by ecosystems (Fig. 4-1). Critically also, ecosystems maintain biodiversity, an increasingly valuable resource that underpins many of the goods and services humans enjoy from ecosystems.

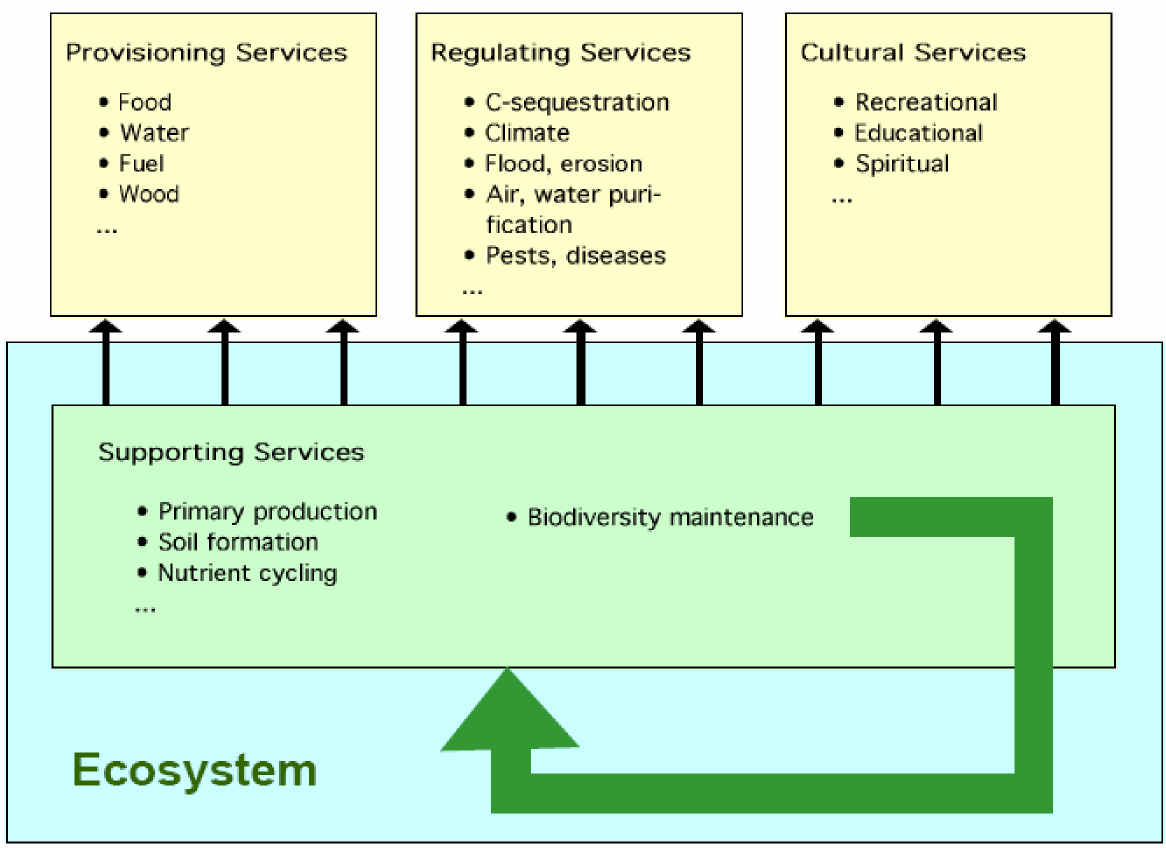

Fig. 4.1: *Human well-being depends inextricably on ecosystem services and provided goods. However, ecosystems are subject to ongoing changes and represent highly dynamic and responsive functional entities which persist only if the supporting services are also available to the ecosystem itself (dark green arrow). In this context biodiversity is often seen as a given prerequisite (e.g. Millennium Ecosystem Assessment, 2005b). Yet this product of two to three billion years of evolutionary processes is threatened if ecosystems are perturbed to a degree, that exceeds their resilience. The fundamental self-dependent biodiversity maintenance of ecosystems can therefore be seen as a core service, by which the ecosystem supports its own existence. All other provisioning, regulating, and cultural services, which are the ones of prime human interest, depend exclusively on the minimal functioning of the basic supporting services, including the self-supporting biodiversity maintenance (listed services are incomplete and serve only illustrative purposes).*

Not all of the goods provided by ecosystems are treated in detail in this chapter. In particular food, fibre, and wood production is dealt with mainly elsewhere, i.e. in chapter 5. This chapter discusses the following provisioning services and products, which may be obtained from ecosystems:

- Primary production (as the basis for fuel wood, fiber, and fodder production)
- Non-fibre products (e.g. fruits, nuts, spices, aromatic plants, and herbs, edible mushrooms, game, rattan, medicinal and cosmetic products, resins etc.).

Regulating services which at a global scale are of paramount relevance for the human existence encompass:

- carbon sequestration (as mentioned above)
- climate and water regulation

- disease, pest, and pathogen regulation
- water and air purification.

Given our mandate we do hardly address the multitude of non-material benefits or cultural services.

4.1.2 Key issues

According to Article 2 of the UNFCCC the ultimate goal of the climate convention is to stabilize atmospheric greenhouse gas concentrations at a level which will allow ecosystems to adapt naturally. Thus ecosystems occupy a central position in assessing climate change impacts and defining “safe limits” to climate change. In the recent and more distant past, ecosystems have experienced and responded to drastic climate changes, encompassing large changes in temperature, atmospheric CO_2 and other environmental variables (Harrison and Prentice, 2003). This might suggests that ecosystems and their component species should be able to tolerate future climate change. Two main facts do not support such a view: Firstly, during periods of past rapid change, ecosystems were not subject to human induced pressures, such as extractive use of goods, and increasing fragmentation of natural habitats. (e.g. Bush *et al.*, 2004). Current ecosystems are certainly expected to tolerate some level of future climate change and in some form or another will continue to exist on this globe (e.g. Kirschbaum and Fischlin, 1996; Gitay *et al.*, 2001). However, strong evidence suggests that current ecosystems will undergo considerable changes above key thresholds, and it is questionable whether these changes can be considered as "natural adaptation". Secondly, in the context of global atmospheric conditions over the past few millennia at least, projected future conditions are unprecedented (WG I, chapter 2, and e.g. Petit *et al.*, 1999) and therefore expected transformations may include irreversible changes such as species extinctions.

Ecosystems are subject to and interact continuously with, changes in the physical environment, in particular at the ecophysiological level, including all climate changes observed up to the present. This results in ecosystem responses, which e.g. can be assessed also quantitatively by various means such as impact models. A key issue is the question whether ecosystems will be impacted by future, i.e. not yet experienced climate changes in a manner, which exceeds their resilience, i.e. their capability to adapt naturally. In this context it is of less importance whether the magnitude, or rate of climate change, or a combination of both exceeds the adaptive capacity of ecosystems. What matters is whether this capacity will be exceeded or not, since once an ecosystems has crossed a critical threshold, its non-linear response may lead to completely unexpected states, which are particularly difficult to project and add a critical, but poorly quantified uncertainty.

In this chapter we will address this issue by assessing the current understanding of the resilience of ecosystems to climate change, hereby distinguishing between natural adaptation not complemented or supported by human intervention from that where human assistance becomes significant for the ecosystem's adaptation to the changing climate. Such human assistance may e.g. consist of planting better adapted species or transporting species from one location to another.

There remains a difficult, specific issue for this chapter due to the term "natural adaptation" as introduced by Art. 2 of the UNFCCC. It means basically the so-called autonomous adaptation, which is to be distinguished from human planned adaptation (see glossary). Natural adaptation is by definition the same as the ecosystem response, since ecosystems do not have any goal-oriented adaptive capability. Thus we will not use the exactly same concept as other chapters while identifying vulnerabilities, which define vulnerability as the insufficient adaptive capacity for those ecosystems, which are sensitive and which are subject to large exposures to climatic changes. Instead we will use the term resilience and will make efforts to identify key vulnerabilities by looking at thresholds (or ranges) beyond which the ecosystem's resilience is exceeded.

4.1.3 Conclusions from the TAR

In the TAR climate change was treated as being one of the many pressures to which ecosystems are exposed (Gitay *et al.*, 2001). The TAR reported for the first time on a substantial set of observations from a large number of studies (Root *et al.*, 2003), which demonstrated that not only the well known ecophysiological responses to increased ambient CO2 concentrations, N deposition, temperatures, and to changes in precipitation regimes, but also changes in structures of ecosystems, communities, and distribution ranges of populations had been observed.

The report also emphasized the complexity of ecosystem responses, highlighting the spectrum of factors and effects which need to be considered in the context of climate change. E.g. the previously expected generally beneficial productivity increases due to CO2 fertilization effects had to be treated in a more differentiated manner, since such effects were found much less often under field conditions than in experimental setups. These findings were not only of particular relevance for the agricultural sector (the TAR treated agroecosystems together with less or little managed ecosystems, while this report treats agricultural systems in the separate chapter 5), but had poorly constrained implications for the structure and function of natural ecosystems as well. For the first time the report also emphasized impacts on trophic levels above that of the primary producers, i.e. the plants, in a separate wildlife section. Moreover, given that many species' range of distribution depend strongly on climate, exhibited changes over the past few decades for the bulk of species were found to be consistent with the expected responses to the local warming observed during those periods. However, these findings have also been criticized on the basis of using a biased data set, since biologists would tend to do study regions where impacts of climate change are expected and regions with no effect would be unlikely to be published. Others argued that high confidence was unwarranted because the analysis simply showed a correlation, not a real cause and effect (Jensen, 2003).

Observable responses to ongoing climate changes were also revealed in a 150-year trend in 26 lakes and rivers in the northern hemisphere, which showed later freezing and earlier ice break-up from a 1.8°C increase in air temperature over continents. Consistent findings were also reported for arctic and alpine regions, where the then observed warming had turned tundra from a net carbon sink to a source of up to 0.7 Gt C yr-1. Similar effects were reported for inland wetlands, notably some peat lands storing significant amounts of carbon.

While the TAR reported that losses in forest cover appeared to have slowed relative to 1980–1995, nonetheless fragmentation, non-sustainable land use such as logging of mature forests, or overgrazing of rangelands, ongoing desertification, and general ecosystem degradation had all led to losses of biomass over significant areas in developing and developed countries.
Finally the TAR assessed threats to biodiversity for the first time, and estimated that 25% of the world’s mammals and 12% of birds to be at significant risk of global extinction due primarily to anthropogenic climate change.

4.1.4 Conceptual framework

Ecosystems are subject and interact continuously with changes in the physical environment including that of climate change. This results in ecosystem responses, which e.g. can be assessed quantitatively by means of models, given they allow to project such responses into the future. These impact models can simulate ecosystem responses at various spatial and temporal scales. Relevant progress has been made in recent years at all scales, i.e. the global, regional, and local scale (e.g. Cramer *et al.*, 2001; Sarmiento *et al.*, 2004; Thomas *et al.*, 2004b; Ainsworth and Long, 2005a; Bond *et al.*, 2005).

Natural adaptation (as "introduced" by Art. 2 of the UNFCCC) is called autonomous adaptation and is distinguished from planned adaptation. However, ecosystems as treated in this chapter are at best partly managed and thus, natural adaptation is by definition the same as the ecosystem response. Thus we can't use the usual concept of identifying vulnerability, i.e. by looking at insufficient adaptative capacity for sensitive ecosystems with a large exposure to climate change. Instead we will use terms like resilience and making efforts to identify key vulnerabilities by looking at thresholds (or ranges) beyond which the resilience "breaks down".
Vulnerability of a system in general is normally considered to be a function of the sensitivity, the adaptive capacity, and the exposure of the system to some changes in its environment (see glossary). Due to our need to distinguish autonomous from planned adaptation of ecosystems, we assume in this chapter that the vulnerability of an ecosystem to climatic changes is determined by its sensitivity, its resilience, and its exposure to climatic changes.

This chapter is based on the findings since the TAR and uses results from other chapters (notably 1 and 2) to assess future impacts of climate change on ecosystems and their goods and services at the biome/community, population, and ecophysiological level.

4.2 Current sensitivities, vulnerabilities, and acclimation

The atmospheric conditions of today are atypical for at least the past 750'000 years (Augustin *et al.*, 2004), and are imposed on a 100 ppmv CO_2 increase and a global warming of about 6ºC since the LGM only 18'000 to 21'000 years ago. Ecosystems have not only responded to strong past climatic changes such as this relatively recent warming , but they also respond to the still moderate ongoing climatic changes (chapter 1, Moberg *et al.*, 2005). Ecosystems of the distant past which existed under conditions of warmer climates similar to the predicted levels of global warming were essentially different from the modern ecosystems both at the regional, zonal, and local levels (e.g. Velichko *et al.*, 2002), and in terms of species and life form composition. For example, grassland ecosystems became a dominant feature of the planet only about 8 million years ago (de Menocal, 2004). There are, however, certain limitations which must be taken into consideration when estimating the environmental changes expected within the 21st century.

Most importantly, many ecosystems may take several centuries and for some ecosystems where soil formation is involved even millennia to reach a quasi-equilibrium with changed climate (e.g. Lischke *et al.*, 2002; Harrison and Prentice, 2003). Thus transient responses and the functioning of ecosystems under continuously changing conditions need to be understood.

4.2.1 Climate variability, past climates, and paleoclimates

The links between climate variability and natural ecosystems has improved in recent years as a recent progress in the understanding of the behavior of decadal-scale climatic oscillations and their impacts that include ENSO (El Niño/Southern Oscillation) and the NAO (North Atlantic Oscillation). These low-frequency phenomena indirectly determine the response of vegetation, notably through the shifts in major controls on vegetation (temperature, precipitation, snow cover) that they imply. For example, the European Alps experience changes in regional climates that can be attributed in part to shifting behaviour of the NAO (Serreze *et al.*, 1997; Wanner *et al.*, 1997; Hurrell and van Loon, 1997; Beniston and Jungo, 2002). In particular, lack of snow in the Alps in the late 1980s and early 1990s, related to low precipitation levels, can to a large degree be attributed to the high positive values of the North Atlantic Oscillation index during these periods, as shown by Beniston (2003). Reversals of precipitation regimes in the Pacific region and beyond during El Niño events can disrupt vegetation through drought, heat stress, spread of parasites and disease, and enhanced occurrence of fire (e.g., Diaz and Markgraf, 1993). In some cases, the compounded effects of ENSO and global warming seem to have triggered changes in certain ecosystems, such as in the Parana river basin and the River Plate estuary in Argentina.

Latest findings for the past 750 000 years (Augustin *et al.*, 2004) have shown that expected anthropogenic climate change has little precedence under these past conditions and to which modern ecosystems which have evolved since the Pliocene for the most part. Moreover, this holds for CO_2 concentrations never as high for even tens of millions of years and temperatures never that warm for even millions of years. Modern biota have evolved under the relatively cool low CO_2 conditions for most of the Pleistocene. This may predispose them to negative effects of excessive anthropogenic warming (e.g. Alverson *et al.*, 2001).

Much evidence demonstrates that past climate changes have impacted the earth’s biomes repeatedly and significantly over the course of geological history, with biomes approximating those currently present arising after the Cretaceous age (Beerling and Woodward, 2001).

Dropping atmospheric CO_2 concentrations and the establishment of seasonal climates resulted in the establishment of a modern assemblage of terrestrial biomes and their associated fauna by the Pleistocene (de Menocal, 2004). Pleistocene climate oscillations between warm glacial and cool interglacial conditions (the latter the predominant state for this era, Augustin *et al.*, 2004), appear to have caused substantial spatial shifts in major biomes, with northern Hemisphere glaciation causing repeated suppression and even elimination of biota at high latitudes. Fewer extremes may have led to lower levels of impact in southern Hemisphere ecosystems, and thus a greater accumulation of diversity (Jansson, 2003). The principal response to these changes appears to have been geographic shifts in species ranges, usually idiosyncratic and individualistic, such that current communities and species assemblages may not have existed in the past (Graham and Grimm, 1990). However, adaptive responses through selection has resulted in some level of species variation due to past climate change (Davis and Shaw, 2001).

4.2.2 Current sensitivities

Chapter 1 provides in depth discussion of recent evidence on the sensitivity of ecosystems to recent climate changes at various scales. While earlier IPCC reports described several ecosystems to be resilient versus warming of up to 1° C (e.g. Kirschbaum and Fischlin, 1996) recent studies resulted in a new, more differentiated view of the sensitivity of ecosystems (e.g.

Walther *et al.*, 2002). Moreover, many pressures on ecosystems exerted by human activities as described earlier (e.g. Gitay *et al.*, 2001) have in many cases intensified. Among these are pollution (e.g. Lelieveld *et al.*, 2002), unsustainable utilization (e.g. Vitousek *et al.*, 1997b), the growing human mobility (e.g. Mack *et al.*, 2000), and notably land-use changes, to which climate change merely adds.

Land use and land cover change, including both conversions (complete replacement of one land cover type with another, e.g. forest to cultivated land) and modifications (more subtle changes in cover or management practices, e.g. intensification of agricultural land), in association with climate change, are recognized as primary drivers of global change (Guo, 2000; Hansen *et al.*, 2001; Korner, 2003; Lambin *et al.*, 2003; Lepers *et al.*, 2005). Indeed, some authors have proposed that land use change is likely to be a more important driver than climate change during this century (Sala *et al.*, 2000b; Slaymaker, 2001).

Ecosystem changes associated with land use and land cover change are complex, involving a number of feedbacks (Lepers *et al.*, 2005; Millennium Ecosystem Assessment, 2005b). For example, conversion of forest to agricultural land works as a driver of climate change, by representing a major contributor to greenhouse gas release to the atmosphere via losses of biomass and soil carbon (Gitz, 2003; Canadell *et al.*, 2004; Levy *et al.*, 2004), and additionally through a ‘land use amplifier effect’ (Gitz, 2003), where the overall global carbon sequestration (‘sink’) capacity (see Chapter 1) is decreased. In contrast, reforestation, and other land use or land management changes such as modifications to agricultural practices can work to mitigate climate change through carbon sequestration (Lal, 2003; Jones and Donnelly, 2004; King *et al.*, 2004a; Lal, 2004; Wang *et al.*, 2004a; de Koning *et al.*, 2005). Land use change also drives other ecosystem changes in combination, or above and beyond climate change, such as biodiversity changes, soil erosion, and land and water pollution (Lambin *et al.*, 2003). Hansen *et al*. (2001) provide some examples of feedbacks between climate, land use and biodiversity, emphasizing the importance of not treating these factors in isolation.

4.2.3 Recent trends

There have been a number of developments in scenarios and their application since the TAR (see Chapter 2), often based on the global SRES storylines (Nakicenovic *et al.*, 2000), or alternative scenario frameworks (e.g. the Millennium Ecosystem Assessment scenarios, see chapter 2). Other developments have been largely carried out on regional or local scales, especially in Europe, either representing down-scaled versions or modifications of the global SRES scenarios (Holman *et al.*, 2004a; Holman *et al.*, 2004b; Ewert *et al.*, 2005; Rounsevell *et al.*, 2005; van Meijl *et al.*, 2005; Abildtrup *et al.*, 2007; Verburg *et al.*, 2007).

Worldwide, most regional and global scenarios indicate an expansion of agricultural land over the next decades due to the tradeoff between food supply and demand as moderated by international trade, with the biggest changes occurring in the tropics (Alcamo *et al.*, 2005). These scenarios generally assume high population growth rates together with low but steady economic growth, which stimulates food demand (Alcamo *et al.*, 2005). Many scenarios also show trend reversing at some point in mid-future their direction from increasing to decreasing agricultural land uses. For example, many African scenarios suggest an eventual slowing of population growth and technological “catch-up” which accelerates improvements in crop yield and causes a shift from expanding to contracting agricultural land. Such trends would relieve some of the pressure on existing unmanaged natural land and have positive consequences for biodiversity (Alcamo *et al.*, 2005). In contrast to other world regions, most European scenarios show agricultural land abandonment (Ewert *et al.*, 2005; Rounsevell *et al.*, 2005; Verburg *et al.*, 2007). Land abandonment scenarios tend to assume that increases in the supply of agricultural goods due to the effect of technological development on productivity will offset changes in food demand (Ewert *et al.*, 2005; Rounsevell *et al.*, 2005; Verburg *et al.*, 2007). Scenarios for which agricultural land abandonment does not occur are often characterized by yield reductions arising from extensification of the agricultural production system. It is likely that both agricultural land abandonment and extensification would create opportunities for ecosystems and conservation (Rounsevell *et al.*, 2005), although the loss of mountain pastures to regrowing forests is regarded by several authors as a negative impact in terms of biodiversity (Dirnbock *et al.*, 2003; Giupponi *et al.*, 2007).

Forest scenarios tend to mirror agricultural scenarios in that forested areas are often merely indirectly determined from the assumed expansion or contraction of agricultural land (Alcamo *et al*., 2005). The majority of scenarios indicate a continued rapid deforestation in many parts of Africa and Latin America over the next decade (Alcamo *et al*., 2005). However, slowing of agricultural land expansion would also lead to the eventual slowing of deforestation, which has important implications for carbon dioxide fluxes and other ecosystem processes (Alcamo *et al*., 2005; Smith *et al*., 2005) This is exemplified in Europe, which again diverges from the global trends with widespread agricultural abandonment creating the conditions for potential reforestation (Kankaanpää and Carter, 2004; Rounsevell *et al*., 2007b).

Global data on population trends are rare and generally confined to large body species, particularly mammals, well known groups such as birds, or economically valuable species such as fish (Millennium Ecosystem Assessment, 2005b p 29). *[IUCN red lists provide extensive information – which we will reference later]*.

Many fish populations are at risk due to current trends towards non-sustainable harvesting, i.e. overfishing (Jenkins, 2003). Particularly stocks of the large top predators are declining, leading to a rate of decrease in freshwater diversity, which is greater than that in many terrestrial or apart from coral reefs most marine ecosystems (Jenkins, 2003). Further stresses of freshwater systems are caused by pollution, siltation, water extraction, dam construction, large scale hydro-engineering projects, and the introduction of exotic species (see also Box 4-1). The greatest effects have been observed in fresh waters within tropical regions, particularly South and South East Asia, and in dryland areas.

Box 4.1: Changes in plankton, fish distribution and production in the NE Atlantic

The principal climate indicator for the North Atlantic, the NAO, has been rising over the past 30 years and the surface waters of the North Atlantic have been warming. This has caused extensive changes in the planktonic ecosystem. Although the precise mechanisms are not fully understood, we can detect consequences for plankton production, biodiversity, species distribution, and fisheries production.

Phytoplankton abundance in the NE Atlantic increased in cooler regions (north of 55^oN) and decreased in warmer regions (south of 50^oN). The effects propagate up through herbivores to carnivores in the plankton food web (bottom-up control), because of tight trophic coupling. Similar effects may be expected for other mid-latitude pelagic ecosystems, because the proposed mechanisms are general and the results for the NE Atlantic are consistent and based on very large scale, long-term sampling (Richardson and Schoeman, 2004).

In the North Sea the population of the previously dominant copepod species, *Calanus finmarchicus* declined and was replaced by southern species (Beare *et al.*, 2002). The seasonal timing of plankton production also altered in response to recent climate changes. This has consequences for plankton predator species, including fish, whose life cycles are timed in order to make use of seasonal production of particular prey species (Edwards and Richardson, 2004). The survival of young cod in the North Sea appears to depend on the abundance, seasonal timing and size composition of their prey. Changes in all of these since 1958 resulted in increased survival and good recruitment of cod throughout the 1960's and 70's and then a progressive decline over the past thirty years (Beaugrand *et al.*, 2003).

The decline of the European cod stocks due to overfishing has been exacerbated by climate induced changes in plankton production and these stocks are no longer able to provide as much surplus for the fishery as in the 1960's and 70's. As the stocks declined they have become more sensitive to the effects of the climate indicator (the NAO), due to shrinkage of the age distribution and geographic extent (Brander, 2005). This interaction between fishing and climate change effects has important implications for management policies.

To some extent the adverse effects of warming on fisheries production of the traditional "northern" species, such as cod, may be offset by increases in "southern" species, such as red mullet. There has been a northward shift in the distribution of many plankton and fish species by more than 10^o latitude over the past thirty years (Beaugrand *et al.*, 2002; Brander *et al.*, 2003). This shift is particularly associated with the shelf edge current running north along the European continental margin and the northward shift does not apply across the whole Atlantic, because warming is not uniform across the whole basin.

Future warming is likely to alter the spatial distribution of primary and secondary pelagic production, affecting ecosystem services such as oxygen production, carbon sequestration and biogeochemical cycling and placing additional stress on already-depleted fish and mammal populations.

Myneni *et al*. (1997) have reanalyzed the satellite data to detect if their were indications of responses to the observed widespread warming signal over land in the Northern hemisphere. From their NDVI (i.e. a greenness index, indicating active plant growth) data for 1981 to 1991 they found a surprisingly large response. They found an earlier greening of vegetation in spring of up to 10 days and a later decline of a few days in autumn over large parts of the Northern hemisphere. Although the analysis was confirmed later by Los (Los, 1998), many have criticized this kind of analysis and blamed it on drift in the satellite instruments.

Lucht *et al*. (2002) have used a global vegetation model and an independent high-resolution database of observed climate. They simulated a similar trend over the eighties and a marked setback in this trend after the 1991 volcanic eruption of Mount Pinatubo, which caused a temporary cooling. The observed trend toward earlier spring budburst and increased maximum leaf area in the 80s is reproduced by the model, but restored the two subsequent years after the eruption. They also simulate a small increase in carbon uptake due to this cooling (decomposition was reduced more than primary productivity), which can well explain the slower increase of atmospheric CO_2 in those years.

Observed satellite data indicate a longer growing season to which vegetation immediately responds. Such phenomena have also been observed elsewhere. Several studies, for example, report a polewards or upwards shift of the treeline border between trees and tundra (e.g. Lavoie and Payette, 1994; Walsh, 1995). Increases in the width of tree rings (Villalba *et al.*, 1994; Villalba *et al.*, 2003) have also been detected near treeline. Similar changes in mountainous areas have been observed on other continents. In the Olympic Mountains of Washington USA, sub-alpine forest has invaded higher-elevation alpine meadows, partly in response to warmer temperatures (Peterson *et al.*, 1994). In Alaska comparison of photographs taken in 1948-50 to those taken in 1999-2000 of the area between the Brooks Range and the Arctic coast show an increase in shrub abundance in tundra areas, and an increase in the extent and density of spruce forest along the treeline (Sturm *et al.*, 2001). On the Argentine Islands the populations of two native Antarctic flowering plants increased rapidly between 1964 and 1990, coincident with the strong regional warming over the Antarctic Peninsula. The Antarctic pearlwort population increased 5-fold while the Antarctic hairgrass increased 25-fold. The unusually rapid increases are attributed to warmer summer temperatures and/or a longer growing season, which enhance the plant's ability to reproduce (Fowbert and Smith, 1994).

Advances in remote sensing based techniques have made it possible to assess impacts of recent changes in climatic conditions (Roerink *et al.*, 2003), who report thatdriest areas are the most sensitive.

All these examples show that biomes respond in many different ways to trends in climate. When these warming trend continue all biomes will change (Leemans and Eichhout, 2004). However, it is difficult to interpret the changes as a specific response at the biome level. Responses are generally made at the individual species level, which cumulatively manifest themselves at the ecosystem, landscape and biome levels.

There is evidence that climate change will hasten extinction of local populations. This has been shown for butterflies, frogs and several other species (Walther *et al.*, 2002; Parmesan and Galbraith, 2004).

Plants: Over the past decades a northward extension of various plant species has been observed many regions which is likely to be attributable to increases in temperatures (Ad Hoc Technical Expert Group on Forest Biological Diversity, 2002). The density of trees and shrubs increased in Arctic and alpine tundra communities (Molau, 1997). In the Alps, for example, evidence exists that climate warming over the past 60 years may have encouraged spruce and pine species in the sub-alpine region and sub-alpine shrubs to grow on the summits (Theurillat and Guisan, 2001). The net effect is an increase in species richness in 21 out of 30 summits in the Alps compared with 50 to 100 years ago. In western Europe, thermophilic (i.e. warmth demanding) plant species have become more abundant compared with 30 years ago (van Oene *et al.*, 2001). For example, 41 new plant species have been added to the latest version of the Dutch flora. These species are nowadays found at more than three locations and have survived three generations. Despite the increase in abundance of thermophilic plants, a remarkably small decline in the presence of traditionally cold-tolerant species is observed (van Oene *et al.*, 2001). Similar trends can be observed in North American communities (Parmesan and Galbraith, 2004).

Lichen and moss flora: It is expected that species with a high migration capacity have the ability to quickly change their geographic distribution. Recent changes in the Dutch lichen flora as well as changes in the moss flora are very clear examples. Since the end of the 1980's particularly warm temperate species with a (sub-) Atlantic or Mediterranean distribution pattern have been increasing. Species with a boreo-montane distribution have been decreasing (van Herk *et al.*, 2002).

Fungi: Just like lichens, fungi can quickly expand their distribution range if the conditions are favourable. *Plicaturopsis crispa*, a small fungi that lives on dead branches and trunks of various deciduous trees, had a typical southern distribution in Europe and was until recently completely absent in the lowlands of western Europe. At the end of the 1980’s the first observations were recorded (RIVM *et al.*, 2003). Since then, the population expanded strongly and became abundant in many locations far north and west of the original range. This explosive increase can only be explained by the lack of severe winters during the last decades.

Birds: Compared to plant studies, relatively few studies have been focused on changes in the distribution area of birds. Comparison of breeding distributions of birds in the UK for two time periods (1968-1972 and 1988 - 1991) showed that the northern margins for many species had moved approximately 19 km northwards. The range shift occurred during a period when central England’s temperature warmed by about 0.5°C) over the last century, and the 10-year period 1988-1997 was the warmest such period in the record (Thomas and Lennon, 1999) . Penguin communities are also considered to be sensitive to changes in climate. The reduction of ice also has severe consequences for several species of penguins in Antarctica (Barbraud and Weimerskirch, 2001; Croxall *et al.*, 2002). Recent changes in these seabird populations reflect direct and indirect responses to regional climate change. Increased snowfall reduces hatching success. Additionally, the melting of the sea ice and the consequent decrease in extent changes krill abundance and distribution (Loeb *et al.*, 1997) on which penguins forage. Adequate food availability has strongly declined regionally. Many large penguins’ colonies have been rapidly declining over the last decade and are cause of extreme concern.

Amphibians: Amphibian populations are declining around the world (Alford and Richards, 1999). Climate change is only one of the many factors that is believed to have contributed to the decline. Other factors are radiation, predation, habitat modification, environmental acidity and toxicants, diseases, and interactions among these factors. The changes have been quite dramatic. In Costa Rica, for example, twenty of 50 species of frogs and toads in a 30-km^2 study area, including the locally endemic Golden toad (*Bufo periglenes*), disappeared following synchronous population crashes in 1987 (Pounds, 2001). Their results indicate that these crashes probably belong to a constellation of demographic changes that have altered communities of birds, reptiles and amphibians in the area and are linked to recent warming. The changes are all associated with patterns of dry-season mist frequency, which is negatively correlated with sea surface temperatures in the equatorial Pacific and has declined dramatically since the mid-1970s.

Also in Puerto Rico three frog species are presumed to be extinct and eight populations of six different species of endemic frogs are significantly declining at elevations above 400 m. Burrowes *et al*. (2004) monitored 11 populations of eight frog species for several decades in Puerto Rico. They focused on climate change and diseases as the potential cause for the decline. Their analysis of weather data indicates a significant warming trend. They found a strong correlation between years with extended drought periods and frog decline. The 1970's and 1990's, which represent the extinction periods, were significantly drier than average.

Mammals: The larger mammals often have a position at the top of the food chain. Therefore, they are very vulnerable to changes in the trophic levels below them. Furthermore, they often require large nature areas with a large amount of suitable space to maintain a viable population. They are also sensitive to a large number of indirect impacts of climate change.

Seals will also potentially be negatively affected by increasing temperatures. Stirling *et al*. (Stirling and Smith, 2004) concluded that if the climate continues to warm in the Arctic, it is likely that rain will be more widespread during early spring. This removes the protection by birth lairs and exposes young ringed seal pups to predation. This negatively affects populations of seals and the polar bears that depend on them for food (see also Box 4-5).

Marine species: The distribution area of species is changing rapidly in marine ecosystems. RIVM (2003), for example, presents monitoring data of a population increase of the Scaldfish (*Arnoglossus laterna*) and the Lesser weever fish (*Echiichthys vipera*) along the Dutch coast. These species occur from the Mediterranean Sea till the south of Scandinavia but were rare in the North Sea. Beare *et al*. (Beare *et al.*, 2004) analyzed trawl data from Scottish research vessels over the last 75 years. They clearly showed that catches of the warm water pelagic species, Anchovy (*Engraulis encrasicholus*) and Sardine (*Sardina pilchardus*), increased suddenly after 1995. All these increases correlate well with the increase in temperature since the end of the 1980's.

Rappé (2003) described some remarkable observations on marine coastal organisms including autochthonous crabs and shrimps, molluscs and aliens of Belgium. Populations fluctuate during the last two decades following severe or mild weather conditions. They disappear after harsh winters and come back more rapidly than in the past. Some offshore species are stranded or extirpated more often. Species new to the area turn up and establish. These observations are believed to be mainly triggered by changes in the oceanographic/climatic conditions, luring or forcing southern species into the North Sea and its coastal waters. There are similar examples along the American coasts as well (Parmesan and Galbraith, 2004).

Increasing evidence is found that every part of the whole food web in marine systems is undergoing significant changes. An important basis of the food chain is zooplankton. Some zooplankton species have shown a northward shift of up to 1000 km, in combination with a major reorganization of marine ecosystems. These shifts have taken place southwest of the British Isles since the early 1980s and, from the mid 1980s, in the North Sea (Beaugrand *et al.*, 2002). In contrast, the diversity of colder temperate, sub-Arctic and Arctic species has decreased. Furthermore, a northward extension of the ranges of many warm-water fish species in the same region has occurred, indicating a shift of marine ecosystems towards a warmer northeastern Atlantic. An invasion of warm-water species into the temperate areas of the northeast Atlantic has also been observed. For example, the cold-temperate *Calanus finmarchicus* copepods are now rapidly replaced by the warm-temperate *Calanus helgolandicus*. Most of the warm-temperate and temperate species have migrated northward by about 250 km per decade, which is much faster than the migration rates expected in terrestrial ecosystems (Parmesan and Yohe, 2003).

Insects: Many invertebrate species have been observed for very many years in a row. A large study of changes in the distribution area of 35 butterfly species in Europe by Parmesan *et al*. (1999) concluded that 22 species have shifted their ranges northwards by 35-241 km. This is consistent with a 0.78°C warming over the past century. In addition, other Lepidoptera the Microlepidoptera species also have undergone significant changes in distribution area that can be attributed to climate change (Ellis *et al.*, 1997).

That insects have the ability to quickly respond to changes in climate is illustrated by the enormous northward expansion of the Mountain pine beetle (*Dendroctonus ponderosae*) in

Canada in the latter half of the 20th century (Logan *et al.*, 2003). This species will further expand its distribution in response to increases in temperature. Data from the Canadian Forestry Center show a significant increase in the number of infestations occurring in areas that were historically climatically unsuitable for the beetle. The mountain pine beetle population has doubled yearly in the last several years. It caused mortality of pine trees across about two million hectares of forest in British Colombia in 2002 alone. The beetle's range has been limited mainly to the southern half of British Colombia by the occurrence of cold winter temperatures and summers too cool for the beetles to complete their development in a single year. These large scale pest infestations have large economic impacts on the forestry sector.

Carbon uptake and release of ecosystems directly respond to changes in temperature and precipitation. This can clearly be seen from the seasonal amplitude of atmospheric CO_2 concentrations, which follows seasonal patterns and is much larger in the northern hemisphere than in the southern hemisphere. This difference reflects the amount of land ecosystems. Myneni *et al*. (1997) illustrated that the leafing-out of plants, which is part of their phenological response, rapidly follows warming over large regions. They used the NDVI, a greenness-index derived from radiation data obtained by satellite, and showed unambiguously showed that the growing season started 10 days earlier at the end of the eighties compared to early eighties.

Their result was controversial because few believed that responses would be so widespread and immediate. Lucht *et al*. (2002)simulated this trend independently using an observed climatology and global dynamic vegetation models. Their models simulated the observed greening trend and additionally showed changes in seasonal uptake and release of carbon, which helped to explain interannual differences in changes in atmospheric CO_2. They showed that events, such as the outbreak of Pinatubo and El Niño influenced carbon uptake in vegetation due to the slight changes in weather patterns.

4.2.4 Extreme events

While changes in the long-term mean state of climate are likely to impose important constraints on numerous ecosystems, many significant impacts of climatic change may emerge through shifts in the intensity and the frequency of extreme weather events. Drought in particular plays an important role in forest dynamics, and has been responsible for pulses of tree mortality in the Argentinian Andes, according to Villalba (1997), and in the eastern Mediterranean (Körner *et al*., 2005). In the Canadian Rockies, Luckman (1994) has identified times of extreme cold as a cause of tree death, with a sustained period of cold summers in 1696–1701 apparently being responsible for extensive tree mortality. Conversely, in the many other parts of the world, particularly continental mid- and high-latitude regions, wind, snow and frost are the main causes of forest damage. Heat waves such as the recent 2003 event in Europe (Beniston, 2004a; Schär *et al.*, 2004 see also Box 4-2) have both short-term and long-term implications for vegetation, particularly if accompanied by drought conditions. The long-term response of vegetation to a particular climatic extreme varies according to specie; some plants may in the year following a major dessication or heat-stress event enhance their biological productivity in order to ensure the plant's reproduction and perpetuation in the face of adversity. Wind-storms can have particularly damaging consequences for forests; according to WSL/SAEFL (Eidg. Forschungsanstalt WSL, 2001), the December 1999 "storm-of-the-century" that affected western and central Europe, including the Alps, destroyed trees at a rate up to 10 times that of the annual felling rate (in parts of France and Switzerland).

Box 4-2: Ecological Impacts by the European Heat Wave 2003

Anomalous hot and dry conditions between June and mid-August of 2003 (Luterbacher *et al.*, 2004; Schär *et al.*, 2004) affected Europe as a whole, and particularly the southern Mediterranean regions (Rebetez, 2004). These heat wave conditions have been attributed mostly to anthropogenic causes with a high level of confidence (Schar and Jendritzky, 2004). While the event itself can not be directly attributed to human influence, the risk of the occurrence of such an event has at least doubled by human influence (high level of confidence, Stott *et al.*, 2004).

The major impact of this anomalous event on vegetation and ecosystems appears to have been through drought stress (Gobron *et al.*, 2005), and a high prevalence of wildfires. More than 25'000 wildfires were recorded during this period in Portugal, Spain, Italy, France, Austria, Finland, Denmark and Ireland, immolating almost 650'000 ha of forest vegetation, 390'000 ha of which were in Portugal (~5% of forest cover in that country, with a financial impact exceeding 1 billion Euro De Bono *et al.*, 2004). Importantly, however, vegetation appears to have recovered generally from the effects of the drought conditions by 2004 (Gobron *et al.*, 2005) which conforms with the findings of modeling studies (Fischlin *et al.*, 2007).

Fire suppression encourages carbon sequestration, especially in humus and soil (e.g. Wardle *et al.*, 2003), providing an important sink for carbon globally. In Siberia, 22 million ha of forest burned in the summer of 2003, releasing 250 million tons of CO_2 (Schiermeier, 2005), with important implications for regional and even global carbon budgets. At a larger scale, emissions of methane and CO_2 during the 1998 El Nino conditions amounted to 2.1 Pg, or roughly 66% of the anomaly in CO_2 growth rate of that year (van der Werf *et al.*, 2004). The suppression of natural fires is problematic, since it tends to accumulate fuel with the concomitant risk of large fires, but fire management is a complex problem that is often obscured by over-simplification of the use of fire as a management tool, and by human economic concerns (Pyne, 2001).

The risk of summers as warm as 2003 may increase by two orders of magnitude in the next 40 years (Stott *et al.*, 2004). The atmospheric conditions that result in similar heatwave conditions are likely to increase in frequency (Meehl and Tebaldi, 2004) and may approach the norm by 2080 in an SRES A2 world (Beniston, 2004a; Schär *et al.*, 2004). Such climatic conditions may place significant stress on natural vegetation both through drought impacts (Fischlin *et al.*, 2007), and an accelerated fire regime that may reduce the success of trees, and favour fire-tolerant species and vegetation types. This conversion of vegetation structure on a large enough scale may even cause an accelerated change in climate (Cox *et al.*, 2000). Emissions of other compounds such as carbonyl sulphide by wildfires feedback on regional climates through their effects in the stratosphere (Notholt *et al.*, 2003), and have even been shown to disrupt cloud formation processes (Koren *et al.*, 2004).

Projections of the impacts of climate change on species richness in southern Europe suggest significant species richness losses even under mean climate change conditions (Thuiller *et al.*, 2005), l*et al*one for climate extreme events such as heatwaves. However, the recovery noted in European vegetation to the summer 2003 suggests substantial resilience in some ecosystems to this enhanced stress if such events remain rare (Gobron *et al.*, 2005).

Extreme events can cause mass mortality of individuals and contribute significantly to determining which species occur in ecosystems (Parmesan *et al.*, 2000). Hurricanes can lead to direct mortality, and their aftermath may cause declines due to loss of resources required for foraging and breeding (Wiley and Wunderle, 1994). Loss of habitat due to hurricanes can also lead to greater conflict with humans; for example fruit bats (*Pteropus* spp.) have declined recently on American Samoa due to a combination of direct mortality events and increased hunting success by local peoples in the altered habitats (Craig *et al.*, 1994). Many of these extreme climatic events are cyclic in nature, such as the sea surface temperature changes that are associated with the El Nino – Southern Oscillation (ENSO). Sea temperature increases associated with ENSO events have been implicated in reproductive failure in seabirds (Wingfield *et al.*, 1999), reduced survival and reduced size in iguanas (Wikelski and Thom, 2000), and major shifts in island food webs (Stapp *et al.*, 1999). Extreme climate events may also trigger disturbances such as fire.

4.2.5 Disturbance regimes

Fire: Fire influences community structures such that it discriminates strongly against those species that are not tolerant of or adapted to fire. Some communities have evolved to be flammable, and geographic shifts in key species will cause the entire community to shift in response – one of the few exceptions to the rule of individualistic species response to climate change (Brooks *et al.*, 2004b).

Fire-prone vegetation types (grasslands, savannas, mediterranean shrublands and boreal forests Bond and Van Wilgen, 1996; Goldammer, 1999) cover a total of 40% of the world's land surface (Chapin *et al.*, 2002). Fires always reduce plant biomass and, depending on their frequency and severity, can also replace trees with shrublands or grasslands, which can cause significant and rapid shifts in ecosystem structure and function (Bond *et al.*, 2005).

The implications of the importance of fire globally are manifold (Fig. 4-2). Firstly, exclusion of fire by suppression strategies seems relatively ineffective in reducing fire hazard (Keeley, 2002; Schoennagel *et al.*, 2004; Van Wilgen *et al.*, 2004), and therefore human systems need to adapt to fire rather than expend resources mitigating fire occurrence. Secondly, human introduction of fire into ecosystems, e.g. to island ecosystems, have transformed forests to flammable shrublands and grasslands (Ogden *et al.*, 1998). Thirdly, the drivers of fire, in terms of ecosystem productivity, fuel accumulation, and environmental fire risk conditions, are all influenced by components of climate change (Williams *et al.*, 2001).

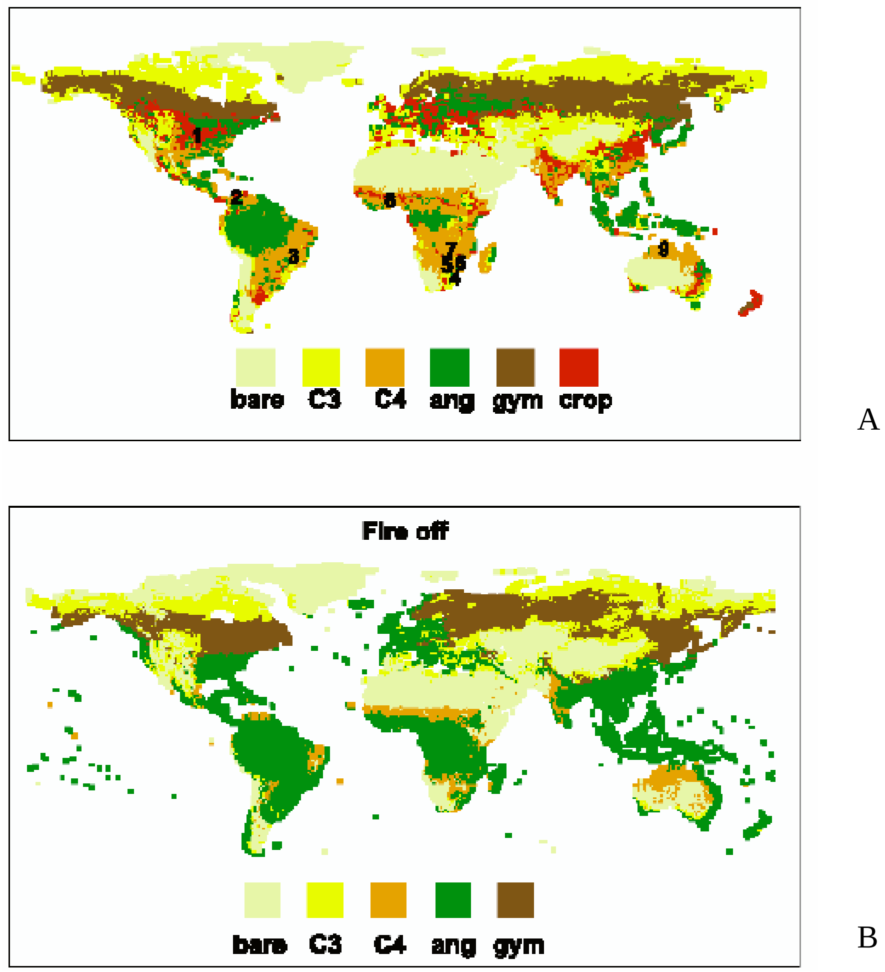

Fig. 4.2: *Global biome distribution with and without fire.* ***A****) ISLSCP Landcover according to dominant functional types. See Table 1 for conversion of landcover classes to dominant functional types. Squares indicate the location of long-term fire exclusion studies listed in Table 2. Source: ftp://daac.gsfc.nasa.gov/data/inter_disc/biosphere/land_cover.* ***B****) Distribution of dominant functional types measured by cover and simulated with 'fire off.*

Fire-prone biomes are found in tropical and subtropical regions with preponderance in the southern Hemisphere (Bond *et al.*, 2005). However, significant fire-susceptible biomes such as boreal forest are also found at high latitudes in the northern Hemisphere (e.g. Kasischke *et al.*, 1995; Peng and Apps, 1999). Unfortunately, a global view of fire is lacking because this process is not well modelled at broad scales, though progress is now being made (Fosberg *et al.*, 1999). Nevertheless, it is possible to derive some insights into the potential implications of climate change for fire regime.

Results from Australia show that increasing temperatures will stimulate increased frequency of fire (Williams *et al.*, 2001) and results from Canada show that show recent increases in fire frequency can be confidently ascribed to anthropogenic climate change. Fire frequency has not been incorporated into projections of species diversity changes under climate change, but is embedded in DGVMs that project changes in vegetation structure as fire regime changes (Bachelet *et al.*, 2003; Lenihan *et al.*, 2003). Climate change-induced increases in fire frequency and expansion of fire prone regions due to both climate change and introduced species suggest that this disturbance will increase in significance as a factor in threshold type responses under climate change.

Invasive alien species (IAS): Invasive alien species (IAS) rank alongside climate change as a primary threat to biodiversity in ecosystems (Sala *et al.*, 2000a). IAS are now so pervasive that they can be considered a significant component of global environmental change themselves (Vitousek *et al.*, 1997a), and intense international effort has been directed at the exclusion and control of IAS as an autonomous problem. Biological invasions have been regarded as too complex to allow for prediction beyond fairly simple generalisations (Crawley, 1989).

However, it has become increasingly obvious that the identity and impact of invasive species may interact with other components of environmental change, particularly climate change, introducing further complexity (Dukes and Mooney, 1999b). Although the potential threats to communities from IAS and climate change have been considered in detail separately, there are relatively few studies that consider their interactive impacts on indigenous biota and ecosystems.

Climate change and IAS are likely to act synergistically to the detriment of natural ecosystems and indigenous biota (Hattenschwiler and Korner, 2003; Macdonald, 2004; Richardson and van Wilgen, 2004), but theory predicting outcomes is poorly developed. Natural scientists are only just beginning to conceptualize how to integrate the combined impacts of such environmental threats, and depart from reductionist approaches to addressing threats individually. Complex interactions may cause unexpected outcomes; for example, following eradication of an invasive rabbit species from the Kerguelen Islands in the southern Ocean, recovery of native vegetation was slowed by recent warming and drying thus allowing exotic vegetation to increase in dominance (Chapuis *et al.*, 2004).

Invasions by alien organisms generally have two distinct phases: a quiescent phase, during which populations are maintained without high growth rates, followed by an active phase, during which explosive expansion of geographic range is triggered. The lag time between the phases can range from decades to a century. A novel biotic and/or abiotic disturbance regime is a primary key to the success of IAS, and communities may become more invasible when an extreme event disrupts the their integrity (Smith and Knapp, 1999). Comparisons across areas of Britain reveal that human-disturbed habitats tend to be rich in alien species (Crawley *et al.*, 1996); similarly, disturbance facilitates IAS success in southern African ecosystems (Le Maitre *et al.*, 2004). IAS may set into motion a self-reinforcing cycle by either introducing or enhancing a disturbance regime, such as fire (Brooks *et al.*, 2004a).

Climate change may well provide additional triggers for IAS expansion, by creating disturbances within the ecosystem that quiescent IAS can exploit. For example, climate change has been predicted to lead to greatly increased rates of species turnover (greater than 40% of the species) in local communities in Mexico, for example (Peterson *et al.*, 2002b). Such species turnover will undoubtedly lead to severe ecological perturbation (disruption/disturbance) of these communities. This emphasizes a general point, that changes within native communities caused by disturbance to ecosystem integrity by climate change may lead to a general increase in the invasive potential of alien species (Hattenschwiler and Korner, 2003; Macdonald, 2004; Richardson and van Wilgen, 2004).

However, disturbance is not a prerequisite for invasion to occur and many invasions do take place without disturbance (Lodge, 1993). Invasive alien species have dispersal and other life history traits that are generally favoured by the predicted changes to climate, unlike the majority of species within native ecosystems. For example, rising atmospheric CO_2 differentially favours plant species with high growth rates (Poorter, 1993), and woody plants (Ainsworth and Long, 2005b), with poorly known but likely significant implications for plant IAS. The few available experiments on IAS suggest strong positive effects of elevated CO_2 (Smith *et al.*, 2000b; Polley *et al.*, 2002), but it is difficult to tease out species-specific effects on native species *vs* IAS. For example, Hättenschwiler (2003) show that two indigenous European temperate forest species had a muted response to elevated CO_2, while indigenous ivy, an indigenous deciduous species and the IAS *Prunus laurocerasus* showed significant responses. In contrast, Nagel (2004) show a clear CO_2 stimulation of a grass IAS, and lack of response in a co-occurring native species.

Combined with a higher reproductive output (Nagel *et al.*, 2004) and possibly greater seedling survivorship (Polley *et al.*, 2002), elevated CO_2 and the negative impacts of climate change on indigenous species (Thomas *et al.*, 2004a) may well provide significant advantages to fast-growing IAS, especially for woody plants, in the context of the invaded ecosystem.

4.3 Assumptions about future trends

[Reminder: Assumptions need still to be coordinated with other chapters

- *For instance with chapter 3 to ensure consistency among the assumptions about water availability and quality. Disturbance regimes such as frequencies of heavy precipitation events leading to erosion, floods, avalanches, mud slides, but also rock fall, fires, insect calamities, diseases (chapter Health)*
- *High impact, low probability events (WG I, chapter 2)*
- *Assumptions about emission scenarios and climate change scenarios (mostly given to us from others, on behalf of writing this chapter, WG I, chapter 2) to further coordinated, comparable assessment.*
- *Pressures from human use and needs, e.g. effects from LUC (various chapters, WG III)]*

Climate change impacting on ecosystems means not only warming or changes in precipitation. To assess accurately and meaningfully impacts of climate change on ecosystems we need scenarios which include in addition to changes in means also changes in the variance and the frequency of rare or extreme events affecting ecosystems. This includes not only changes in means and extremes of air temperature and precipitation, but also sea level rise and sea temperatures (some also covered in chapter 6) plus elements of global change, which are not directly related to climate change.

Albeit it is widely recognized that climate change forms part of global change, interrelationships between the two are often not treated in a fully integrated manner. Despite the recognized importance of land use change in driving global change it is rarely included in current global climate models (Hansen *et al.*, 2001; Holman *et al.*, 2004a; Levy *et al.*, 2004; Zebisch *et al.*, 2004). This limitation is attributed to an insufficient understanding of the underlying causes and prospective consequences of land use and land cover change at the global scale (Hansen *et al.*, 2001; Lambin *et al.*, 2001; Lambin *et al.*, 2003). However, much progress has been made in this area, both in terms of a better understanding of the underlying processes as well as in the development of more sophisticated models (Lambin *et al.*, 2003). Several studies incorporating land use change have been undertaken at local and regional scales, and are discussed in further detail in the following section. The explicit inclusion of land use change in analyses of climate change could therefore lead to some unexpected outcomes (Hansen *et al.*, 2001). Consequently, many impact studies of climate change may represent conservative estimates of projected changes.

4.3.1 Assumption about future climate

Ecosystems are often studied at scales which are considerably finer than those of the climate models. Yet climate models are the best available tools to study impacts on ecosystems. The mismatch in scales calls for downscaling (e.g. Gyalistras and Fischlin, 1999), to obtain physically consistent bioclimatic scenarios.

At the regional scale, RCM simulations undertaken in the EU-PRUDENCE project framework (Christensen *et al.*, 2002) lead to the conclusion that, on average, much of Europe is likely to experience a rise in average temperatures by about 4°C in the period 2071-2100 compared to the reference period 1961-1990, when using the IPCC Scenario A-2 for greenhouse gas emissions; this scenario represents an upper range of possible futures developed by the IPCC (Nakicenovic *et al.*, 2000). The distribution of changes in summer maximum temperatures suggests a general northward migration of climatic zones, with warming exceeding 6°C from the Iberian Peninsula to SW France. Many model simulations show that there is an asymmetric increase in the upper extremes of summer temperature, i.e., the shifts in the upper extremes of summer temperatures are stronger than the changes in means, and can exceed 8-10°C in some parts of western and southern Europe. Such changes can have significant repercussions for hydrology and ecosystems, whereby extreme temperatures are likely to exert stronger controls on evaporation or desiccation, heat and water stress on plants than mean temperatures.

Future precipitation trends, while more problematic to simulate in climate models than temperature, nevertheless exhibits a dual trend in many model simulations, i.e., a general reduction in *average* annual precipitation, and a simultaneous increase in *extreme* precipitation events. This has already been shown to be the probable case for much of Europe by Christensen (2003), where as a result of much warmer summer temperatures, precipitation tends to decrease on average. On the other hand, the energy supplied by the higher temperatures is capable of triggering short-lived but very severe convective rainfall events, which in some parts of the Alps could increase by as much as 30% for a 2°C warming (Frei *et al.*, 1998). The tendency of a dual simultaneous increase of both low and high precipitation extremes in the mid-latitudes in a warmer climate has also been reported elsewhere, notably for North America (e.g., Trenberth, 1999; e.g., Houghton *et al.*, 2001b).

Very preliminary results from modeling studies of extreme winter storms, based on model studies of events such as the 1999 *Lothar* storm (Goyette *et al.*, 2003), suggest an increase in the frequency of strong winds originating in the Atlantic at the expense of föhn-type storms related to southerly flow across the Alps. It should be borne in mind that the sensitivity of forests to wind-stress increases markedly if the interval between wind-storms is reduce, thereby hindering recovery of forests and forest ecosystems following a major storm.

A particular challenge for climate scenario generation are complex terrains such as mountainous regions, where resolution errors may exceed the climate change signal (e.g. Gyalistras *et al.*, 1994). Few model simulations have attempted to directly address issues related specifically future climatic change in mountain regions, primarily because the current spatial resolution of general circulation models (GCM) and even regional climate models (RCM) is generally too crude to adequately represent the topographic detail of most mountain regions and other climate-relevant features such land-cover that are important determinants in modulating climate in the mountains (Beniston, 2003). Recent simulations have incorporated mountain regions within larger domains of integration (e.g., the Alps or the Scandes in Europe), thereby enabling some measure of climatic change in mountains. High-resolution RCM simulations (5-km and 1-km scales) are used for specific investigations of processes such as surface runoff, infiltration, and evaporation (e.g., Arnell, 1999; Bergström *et al.*, 2001), extreme events such as precipitation (Frei *et al.*, 1998), and damaging wind storms (Goyette *et al.*, 2003), but these simulations are too costly to operate in a "climate mode".

Projections of changes in precipitation patterns in mountains are tenuous in most climate models because the controls of topography on precipitation are not adequately represented. In addition, it is now recognized that the superimposed effects of natural modes of climatic variability such as El Nino/Southern Oscillation (ENSO) or the North Atlantic Oscillation (NAO) can perturb mean precipitation patterns on time scales ranging from seasons to decades (Beniston and Jungo, 2002). Even though there has been progress in reproducing some of these mechanisms in coupled ocean-atmosphere models (Osborne *et al.*, 1999), they are still not well predicted by climate models.

4.3.2 Assumption behind species distribution models

In the past few years, numerous projections of climate change impacts on species geographic ranges and even risk of extinction have been made based on empirical knowledge of their current geographic distribution. While the approach is expedient, and may provide a first-cut assessment of impacts, it is limited by many assumptions and uncertainties (Pearson and Dawson, 2003). Such bioclimatic models, or more better termed Species Distribution Models (SDMs) are empirical models relating field observations to environmental predictor variables, based on statistically or theoretically derived response surfaces (Guisan and Thuiller, 2005).

Species data can be simple presence, presence–absence or abundance observations based on random or stratified field sampling, or observations obtained opportunistically, such as those in natural history collections (Graham *et al.*, 2004). Environmental predictors are optimally chosen to reflect the three main types of influences on the species: (i) limiting factors, defined as factors controlling species eco-physiology (e.g. temperature, water, soil composition); (ii) disturbances, and (iii) resources, defined as all compounds that can be assimilated by organisms (e.g. energy and water). These relationships between species and their overall environment can cause different spatial patterns to be observed at different scales, often in a hierarchical manner (Pearson *et al.*, 2004).

Equilibrium postulate: As both species and environmental data are usually sampled during a limited period of time and/or space, models fitted using these can only reflect a snapshot view of the expected relationship. A convenient working postulate is to assume that the modelled species is in pseudo-equilibrium with its environment (Guisan and Theurillat, 2000). Although this is a required assumption for projecting the model in space or time, surprisingly few critical considerations have been raised in the recent literature on how close a given modelled system really is to an equilibrium (but see Araújo and Pearson, 2005 Svenning, 2004, limited filling of), and how long it would take to reach a new equilibrium after an environmental change (but see Davis *et al.*, 1998). Nonetheless, a study comparing Pleistocene and Holocene range projections for 48 species in North America showed a high degree of "niche conservatism" and predictive power of this technique, supporting to some extent its value as a predictive tool (Martinez-Meyer *et al.*, 2004)

Niche concept: SDMs strongly rely on the niche concept (Guisan and Theurillat, 2000; Guisan and Thuiller, 2005). Leibold (1995) opposes two concepts of the niche as either driven (i) by the environmental requirements of species, or (ii) by the impact that the species can have on its environment. Hence, the former is more embedded within an autecological and physiological approach to the niche (environmental niche), whereas the latter is related more to trophic levels and food web theory (trophic niche). As they seem to apply to different spatial scales only the requirement concept and environmental niche are usually considered in SDMs. Within these, Pulliam (2000) further distinguishes between the view that species occupy all of their suitable habitats – the fundamental niche – and the alternative view that species are excluded from a part of their fundamental niche by biotic interactions, resulting in the realized niche that is actually observed in nature (Silvertown, 2004).

Competition: Competition is an important theoretical mechanism that is absent from most examples of SDM research. Recent analyses showed that the inclusion of additional predictor variables representing the presence–absence of known competitors can significantly increase the predictive power of models (Leathwick and Austin, 2001). Such findings suggest that even at relatively coarse resolution and regional extent, the presence or absence of a given competitor might influence the distribution of another species, but they do not provide a formal proof that the observed competition effect really occurs in nature. Including other species as predictors might simply provide information about physical conditions that are not accounted for by those environmental descriptors included in the model.

According to these basic assumptions, several analyses have now assessed the impacts of climate change on species distributions, from insects (Sutherst *et al.*, 2000), hertpiles (Araújo *et al.*, 2006), birds (Peterson, 2003), mammals (Burns *et al.*, 2003), to plants (Thuiller *et al.*, 2005).

4.3.3 Uncertainty in predicting future potential species distributions

Model uncertainty: Recent studies have demonstrated significant variability in species distribution predictions of species range shifts (Thuiller *et al.*, 2004a; Araújo *et al.*, 2005), and shown that uncertainty provided by the combination of different analyses, spatial resolutions, scales, modeling techniques and evaluation methods was greater than the variability of using different climate change scenarios (Thuiller, 2004). Such a challenge demonstrates that different analyses using different models and resolutions are not comparable, because the way in which models are constructed varies and this strongly influences the model outputs (Thuiller, 2004). A new recent study Araújo *et al*. (2005) showed that significant improvements could be achieved if ensembles of range shift projections were produced and appropriately analysed see also (see also Thuiller, 2004; Thuiller *et al.*, 2005), but there remains a need to investigate and test different approaches for deriving and combining different forecasts.

Species migration: In most projections, species migration is inappropriately taken into consideration, relying either on a “no migration”, an “unlimited migration” scenarios, or both (e.g. Peterson *et al.*, 2002a; Thomas *et al.*, 2004b; Thuiller *et al.*, 2005). With “no migration”, a species can only loose habitat as climate changes, whereas in the “unlimited migration”, all habitats that become suitable can be colonized, two rather unlikely extreme situations. As migration capabilities of organisms depend on both dispersal characteristics and fecundity, they must be accounted for each species individually when deriving projections. Although recent tentative for integrating species migration in SDMs (Iverson *et al.*, 2004; Williams *et al.*, 2005) seem successful, further developments are here required to better integrate the migration ability of species into SDMs.

Biotic interactions: Because SDMs are based on the realized rather than the fundamental species niche, projections into future climate, where biotic interactions may have changed, are likely to generate mistakes (Davis *et al.*, 1998). The degree of prediction errors should be related in some way to a species capacity to occupy its full fundamental niche in the current and future climate. For most species, it is unknown how much its fundamental niche is represented by its realized niche, although it is expected it should relate to its competitive and dispersal abilities.

[Mechanistic models of ecosystem response:
A section on assumptions and uncertainties inherent in dynamic global vegetation models (DGVMs) and other approaches, such as patch models will be developed here]

4.4 Key future impacts and vulnerabilities

4.2.1 Impacts across multiple hierarchical scales – a global perspective

To project responses of ecosystems to projected climatic changes, many assumptions can and are being made. They give raise to basically three approaches, which can be grouped in the following manner: (i) correlative, (ii) mechanistic, and (iii) anolog approaches. For the correlative (i) and mechanistic (ii) approach, studies and insights from the presence form the basis to assume that the very same mechanisms will continue to exist in the future and that the same set of causes will be responsible for the ecosystem's response under future climatic conditions. Both approaches often deploy models, which are either based on correlative, or causal relationships between aspects of ecosystems structure and function, or a mixture of both. They capture the assumptions and the current understanding of the driving mechanisms and are used to project observed sensitivities and trends quantitatively into the future; Evidence from past climate changes (iii) is used to extrapolate for analogous future situations. All three approaches have their merits and drawbacks.

[Ron Neilson (new CA) will provide some text here for the SOD using dynamic global vegetation model assessments of changes in biome structure and function (e.g. Cramer, 2001; Cramer, 1999; Woodward, 2001; Woodward, 1998; Wagner, 1998; Marland, 2003; Hansen, 2001 and many others].

Ecophysiological responses to climatic changes, including increased ambient CO2 concentrations, can be measured in the present. This allows to asses also the future ecophysiologcial responses of ecosystems such as forests to global climate change (e.g. Buchmann, 2002) inasmuch they do not depend on functions and structures given at the population or ecosystem scale, which may limit the correctness of such projections.

[Still to be written:
Interaction of climate change impacts (mainly temperature and precipitation) and the direct effects of CO_2 and nitrogen deposition
CC impacts on net biosphere productivity, NBP (net biome prod.), NEP
CC impacts on NPP, GPP, respiration

CC impacts on landscapes
CC impacts on ecosystems and interference with management intensification and pollution
CC impacts on ecosystems open new niches for invasive alien species
CC impacts on ecosystems exceeding certain thresholds causing nonlinear responses: minimal productivity, drought, flooding, eutrophication, biodiversity
Management intensification and pollution and CC impacts, e.g. water salinisation and acidification (see also chapter 3 and 6)
Threshold effects and key vulnerabilities]

4.4.2 Grasslands and savannas

Grassland systems are bioclimatically diverse, being found in tropical regions (C_4-dominated grasslands) and temperate cool regions (C_3-grass and herb-dominated systems). Grasslands are generally disturbance-controlled, by both fire and often intense grazing (or mowing as for example in intensively managed European montane grasslands). Savanna systems are characterized by a dynamic mix of grasses and trees (Scholes and Archer, 1997), and are common in tropical regions. Fire regime is now known to exert an important control on the tree-grass balance in these systems (Bond *et al.*, 2005), but herbivore grazing is also a key disturbance that may determine vegetation structure (Scholes and Archer, 1997; Fuhlendorf *et al.*, 2001). A key feature of all of these systems is the sensitivity of composition and structure to disturbance regimes, which can be strongly human-mediated. For the purposes of this section, we group C_4 and C_3/C_4 mixed grassland types with savannas, and deal with temperate grasslands (C_3 dominated) separately.

Tropical savannas and grasslands cover roughly 28 million km^2 of the globe, almost three times that of temperate forests. Temperate grasslands cover an area of about 15 million km^2, slightly more than that of boreal forests. These systems are in many areas strongly influenced by human mediated control, including a range of grazing and other physical disturbance regimes, most notably fire. Tropical savanna NPP is second only to that of tropical forests, yet these systems maintain only one quarter of the above-ground biomass of tropical forests. This is due to fire, which reduces the biomass that these systems could potentially maintain given ambient rainfall (Bond *et al.*, 2005). As such, these systems are a potential sink for carbon contingent on the prevailing fire regime. For example, Australian savanna systems are currently a net carbon sink of between 1 and 3 t C ha^{-1} y^{-1}, depending on the frequency and extent of fire (Williams *et al.*, 2004). Exclusion of fire has the potential to transform many savannas to forest ecosystems, and if it could be achieved, would result in an increase of closed forest cover from its current 27%, to 56% of the vegetated surface of the world (Bond *et al.*, 2005).

The impacts of human pressure on land-cover characteristics and ecosystem services in these systems have long been a focus of biophysical and socio-economic studies. Much research now reveals that the impacts of savanna land-use on vegetation cover may feedback to affect regional climatic conditions. In four out of five savanna systems studied globally, modeled conversion of savanna to grassland resulted in a 10% rainfall reduction, suggesting a positive feedback between anthropogenic impacts and changing climate (Hoffmann and Jackson, 2000). This result is supported by the impacts of modeled removal of savanna systems from global vegetation cover, which has larger effects on global precipitation than for any other biome (Snyder *et al.*, 2004). At the continental scale, modeled conversion of tropical forest to a savanna structure causes reductions in rainfall in tropical regions, but increased rainfall in central southern Africa, due to complex teleconnections in the atmosphere (Semazzi and Song, 2001).

Climate change may affect savanna ecosystem function and species composition in different ways depending on the level of warming and rainfall change, and the physiological effects of CO_2 levels on plants. Under moderate warming, i.e. less than 4ºC, a general expansion of woody vegetation and positive carbon sequestration is seen in North American. However, with more than 4°C warming, forests may be replaced by savanna types, partly due to fire effects,and savanna systems may increase their cover by up to 50% (Bachelet *et al.*, 2001). In an illuminating study of transient system responses to a range of emissions scenarios (White *et al.*, 2000a), it was found that IS92a emissions scenarios could ultimately reduce mid-latitude carbon sinks (partly in savanna systems) that offset increasing carbon sinks at high latitudes, thus leading to a net global carbon source from vegetation. This effect was muted and delayed by more than a century in a scenario of CO_2 stabilization at 550 ppmv.

Impacts of drying, warming and increased wind speed under IS92a and SRES emissions scenarios, resulting in temperature increase of 2.5ºC to 3.5ºC by 2100, have been simulated to cause a reduction in vegetation cover and possible remobilization of Kalahari dune systems in southern Africa as early as 2040 (Thomas *et al.*, 2005). Such sensitivity of land cover to climate change mirrors that found in Sahelian savannas, and underscores the potential vulnerability of these systems to climate change. By contrast, modeling suggests that current and higher atmospheric CO_2 levels may reduce the historic vulnerability of Sahelian systems to drought, by facilitating increased primary productivity and water balance through the physiological fertilization effect (Wang and Eltahir, 2002).

Empirical experimental investigation of impacts of warming, rainfall change and atmospheric CO_2 level are virtually absent in savannas, but more common in C_3/C_4 mixed and C_4 dominated grasslands. Suggestions that rising atmospheric CO_2 would differentially favour C_3 types in mixed C_3/C_4 systems has not been conclusively supported by field experiments. In short-grass prairie, field CO_2 fumigation revealed a 26-47% increase in production under 720 ppmv CO_2 and 2.6ºC warming, regardless of photosynthetic type (Morgan *et al.*, 2001a), and there were no impacts on ecosystem-level emission rates of all measured trace gases (Mosier *et al.*, 2002). In a tropical South African grassland, a four to five decades long CO_2 enrichment also revealed no substantial increase in C_3 success in a C_4-dominated system, and muted impacts on aspects of nitrogen cycling (Stock *et al.*, 2005) . However, both field studies showed increased soil water content at elevated CO_2 (see also Ferretti *et al.*, 2001). Interestingly, regional climate modeling indicates that the biological impacts of CO_2 fertilization on grasslands may scale up to affect regional climate (Eastman *et al.*, 2001).

Few experimental approaches have assessed ecosystem responses to multifactorial treatments, as opposed to single species responses (Norby and Luo, 2004). It is difficult to manipulate and control interactive treatments, especially if they include temperature modification under field conditions. Consequently there are few field ecosystem warming studies (Rustad *et al.*, 2001), and these are typically confined to grasslands, which are easier to control. A critical finding from these approaches has been the apparent acclimatization of soil respiration to warming of 2ºC, suggesting a low feedback potential of ecosystem respiration to atmospheric CO_2 composition (Luo *et al.*, 2001). Warming of a mixed C_3/C_4 New Zealand grassland to simulate extreme heat wave events caused increased representation of C_4 species within a single growing season, but a reduction of productivity by over 60% in the absence of C_4 plants (White *et al.*, 2000b).

Drought appears to exert significant impacts on savanna and C_4 grassland ecosystems. Notable reductions in both plant cover and productivity were simulated along an aridity gradient in southern African savanna in response to the substantial drying trend of ~8 mm yr^{-1} since 1970 (MAR ranged from 299 to 918 mm yr-1 in 2000 Woodward and Lomas, 2004). Increased rainfall variability (50% increased dry spell duration) caused a 10% reduction in productivity of a tall-grass prairie C_4 grassland (Fay *et al.*, 2003).

At the individual level, some African savanna tree phenologies appear sensitive to seasonal air temperatures (Chidumayo, 2001). However very little work appears to have been done on the potential responses of individual savanna plant species to climate change. Using mechanistic modeling approaches it has been shown that the tree-grass balance in savannas may shift towards trees. This is explained bypast and continuously rising atmospheric CO_2 concentrations,and the consequent diminishing ability of grasses to suppress more rapidly growing tree saplings in grass fires (Bond and Midgley, 2000; Bond *et al.*, 2003). However, only one field experiment has been established worldwide to explore these interactions in savanna systems under elevated atmospheric CO_2 conditions (http://savanna.ntu.edu.au/publications/savanna_links17/co2_study.html).

For an African arid savanna raptor species, general reductions in population success have been simulated under climate change scenarios, not only for reduced precipitation scenarios, but also for scenarios of increased inter-annual variation in precipitation with no mean change (Wichmann *et al.*, 2003). Similarly, substantial reductions (by 4-98%) in species ranges for almost 80% out of 179 modeled species have been projected using IS92a-driven regionally interpolated climate scenarios. Affected were South African bird and animal species, including many savanna species, with particular impacts on extinctions of species currently found in South Africa's Kruger National Park wildlife reserve (Erasmus *et al.*, 2002). Observations of population declines in three ungulate species in this reserve suggest that a trend of reducing summer rainfall, possibly related to regional warming trends, are likely to result in their extirpation if warming continues, especially as boundary fencing inhibits their potential migration (Ogutu and Owen-Smith, 2003).

[Temperate grasslands text not complete]

4.4.3 Forests and woodlands

Relatively dense canopy forests cover 43 Mkm^2 (~30% of all land) with 17.6 (tropical rain, 41%), 7.5 (tropical season, 17.4%), 10.4 (temperate, 24.2%), and15.1 (boreal, 35.5%) out of 149 Mkm^2 and they store above and belowground about 1'199 Pg C (e.g. Kasischke and Stocks, 2000), which is about 160% of all C in the atmosphere. The circumpolar boreal forest of the northern hemisphere alone stores over 30% (70.2 Pg Smith *et al.*, 2004) of all C contained in the biosphere. Forests share the presence of trees with savannas covering another 19 Mkm^2 (~13% of all land, see section 4.4.2) and another 8.5 Mkm^2 are wood or shrub land (~5.7% of all land).

In the context of climate change it is relevant that forests can only persist in a certain range of temperatures (-7 to 37°C annual mean) and precipitation (~ >400 mm annual total). In a warming world mean precipitation is expected to increase globally with temperature due to the Clausius-Clapeyron equation. Those forests, which are limited by the minimum requirements, may in some cases profit from these changes (e.g. Bachelet *et al.*, 2001). However, precipitation is distributed worldwide in a highly uneven manner and thus the actual conditions forests will experience in future climate change is associated with major uncertainties. While the upper bound for precipitation is of minor concern, since only persistent flooding results in tree mortality, the upper bound for temperature is more critical, since it is linked to precipitation in a complex, non-linear manner, which may lead to highly significant impacts not only on the forests themselves, but also to the climate system due to the fact that forests store large quantitites of C (e.g. White *et al.*, 2000a; Cox *et al.*, 2004). Especially the latter may also be temporarily critical for forests which may profit on the long run, yet may undergo stressful periods showing die backs an the concomitant release of C to the atmosphere during the transition phase (e.g. Fischlin *et al.*, 2007).

It is important to consider that the actual impacts of climate change on forests depends also on the seasonal and diurnal rainfall and temperature pattern and may be moderated by the hydrological regime as influenced by the hydrologically relevant surroundings of a forest stand (e.g. Zierl and Bugmann, 2005). Since past as well as projected future increases in mean temperature and changes in precipitation have and might affect more winter precipitation (e.g. Gyalistras, 2003; Xoplaki *et al.*, 2003) while summer temperatures might go up and precipitation down, the potential for drought conditions during the critical growing phase is given. In addition, such conditions may cause changes in relevant disturbance regimes such as fire (Flannigan *et al.*, 2000) or insects (Logan *et al.*, 2003).

Rising atmospheric CO_2 conditions have not only an effect onto forests by climate, but ambient CO_2 concentrations affect also directly and immediately tree growth and net ecosystem productivity (NEP), which is in the case of forests particularly relevant for climate change due to the potential biotic feedbacks. Moreover, exact ecophysiological responses of trees under drought conditions are of particular relevance, since they are not only affecting the fate of the forests, but may also feedback to local climate (Leuzinger *et al.*, 2005).

Considerable progress with respect to understanding these processes, such as the role of fire (Kasischke and Stocks, 2000; Kasischke and Penner, 2004) – a climate change affected disturbance regime - was made in the last years, albeit considerable gaps remain. Major and still ungoing efforts were and are made to quantify NEP usingnetworks of flux towers (e.g. Enquist *et al.*, 2003). However, in contrast to short lived plants or tree seedlings, experiments attempting to simulate future enhanced CO_2 conditions for adult trees under field conditions are still rare. At least partly some long overdue concerns about inadequate experimental setups (Körner, 1995) could be improved (e.g. Korner, 2004). Yet, the considerable reverberation, often subtle, that the change in the chemical composition of the atmosphere may have are still poorly understood (Korner, 2003) and this research is in this respect still in its infancy.

Apart from drought conditions increasing temperatures are of concern, and may affect the northern or altitudinal boundaries, which are temperature determined. Globally important processes may result, e.g. large quantities of C may be released from the boreal region, which were previously immobilized in the cold and frozen soils (Harden *et al.*, 2000). This is of particular concern, since the region has been a long term C sink since early Holocene (Woodwell *et al.*, 1998) as corroborated recently from ice core studies (Smith *et al.*, 2004) resulting in the storage of the largest, biome-specific fraction of the biospheric C. In addition, disturbances such as fire or insects are expected to have significant impacts in the boreal region. These expectations, also emerged earlier from GCM-based studies (Stocks *et al.*, 1998), are now confirmed by more recent observations, showing a gradual increase of the fire frequency in the North American boreal forest over the last 3 decades (Kasischke and Stocks, 2000). The question arises whether fire renders the boreal forest, previously a long-term sink, into a source offsetting the beneficial effects from CO_2 fertilization and climate change? Furthermore, the well known asymmetry between the rate by which boreal forests are lost at the southern boundary due to drought and fire versus the rate by which boreal forests can encroach into the tundra (e.g. Kurz and Apps, 1999) present themselves relative to these findings in a more critical manner than that was previously understood (e.g. Woodwell *et al.*, 1998).

[To be completed migration rates: In contrast to the tree lines in mountainous areas (relatively fast migration within slopes of valleys, unless warming exceeds altitude) boreal forest's northern limit expansion of trees is governed in a complex manner by precise climatic conditions (T,P), soil moisture, plus genetic factors (e.g. seeds from trees at one latitude cannot be successfully grown at other latitudes, more southern or more northern). The latitude-specific ecotype/variety is well adapted (by selection) to break dormancy and germinate at specific photoperiods. Picea glauca migration rate was estimated to be confined by such mechanisms to 300 m/a.]

[To be completed for other types of forests such impacts in tropical forests e.g. (Korner, 2004) and of course most recent quantitative assessments using recent models (GCMs with improved "biotic components", DGVM, and forest models at scales below that).]

4.4.4 Deserts

Deserts are part of dryland ecosystems where annual precipitation is less than two thirds of potential evaporation (Chapter 1, Millennium Ecosystem Assessment, 2005a). Using the index of aridity, defined by the ratio of annual mean precipitation (P) over potential evapo-transpiration (PE), deserts fall within $P/PET < 0.03$ for hyper-arid zone to $0.03 < 0.2$ for arid zones with mean annual precipitation ranges of less than 25mm to about 200mm respectively. Deserts have been classified into cold temperate, hot subtropical and foggy cool coastal deserts depending on the role of temperature as a limiting factor in ecosystems functioning. Foggy deserts are subjected to maritime effects resulting in stable temperature regime over different seasons and between day and night. For cold deserts temperature is important in ecosystems processes while for hot deserts rainfall is the main factor.

Seasonality of rainfall has been used to characterise deserts. Some deserts have a distinct winter or summer, for instance, the Kalahari desert in southern Africa and the Chihuahuan Desert in southern New Mexico. While other deserts are areas of transition with other cases having irregular and complex cycle of rainfall. The seasonality characteristics of deserts has profound effects on the ecosystems, accounting for the different biomes and also the fire characteristics associated with them (Nicholson, 2002). Interannual variability of rainfall is another factor used to differentiate between different deserts and is an important element in determining the sensitivity of the region to large-scale climatic change.

However, in addition to temperature and rainfall desert systems are more likely than other ecosystems to be subjected to salt stress (Wickensa, 1998). Studies on desert-margin species in the semi-desert environment of the Negev have noted that there is a likelihood that increasing the atmospheric CO2 concentration may reduce the impact of salinity on plant growth. This could favour greater productivity in desert-marginal areas and may increase productivity and biomass of natural desert vegetation and soil organic matter (Poorter and Perez-Soba, 2001).
The results of GCMs in relation to deserts are complex and contradictory. There are large differences between results of models in terms of the magnitude and distribution of warming and change in precipitation over deserts (Lenihan *et al.*, 2003; Lioubimtseva and Adams, 2004). Despite the variations in models the general trend is that changes in rainfall averages will be season-specific and inter-annual variation in rainfall amount is likely to increase (Schwinning

and Sala, 2004). GCMs also forecast an increase in evapo-transpiration greater than precipitation in mid-continent regions with a potential for more severe, longer-lasting droughts (Lioubimtseva and Adams, 2004). It is well established that arid and semiarid systems are among the most sensitive systems to precipitation changes, because of the central role that soil moisture plays in governing almost all ecosystem processes including plant functional diversity (Leemans and Kleidon, 2002; Nicholson, 2002; Schwinning and Sala, 2004).

Some of the factors to note on future climate in deserts is that important vegetation–climate feedbacks which might affect these regions may originate from adjacent forested and wooded areas. E.g. it has been indicated that, an initial increase in forest vegetation in West Africa could result in a substantial increase in rainfall over the Sahel and Sahara due to the monsoon moving further north. Future climate impacts prediction models need to take into account current evidence to the effect that internally generated natural climate fluctuations (e.g., El Niño events) are amplified by vegetation feedbacks under the present climate in desert-marginal regions, as shown from results of GCMs that explain climate switches in the Sahara (Claussen *et al.*, 1999; Schneider, 2003). Further observations have been made to the effect that rising atmospheric CO_2 may alter atmospheric humidity, through widespread stomatal and transpirational responses, resulting in atmospheric feedbacks that will tend to lessen currently observed vegetation responses to CO_2 in experimental study conditions (Morgan *et al.*, 2004b).
The role of dust blown into the atmosphere from arid lands is another factor that has received limited attention in GCM simulations. Increased dust flux may increase aridity and also suppress rainfall out-side the desert areas themselves while if there is increase in rainfall over arid areas this will reduce dust flux and sustain more wet conditions (Lioubimtseva and Adams, 2004).

From the unstable history of the Sahara and other parts of the northern monsoon system it is likely that future dramatic sudden changes in deserts may occur. In fact desert regions may have a greater sensitivity to CO_2-induced climate change than GCMs predict (Lioubimtseva and Adams, 2004). But recent assessments point out that there are still major uncertainties on the potential effects of increasing concentrations of CO_2 and future climate change in arid ecosystems. For southern Africa, it has been noted that decrease in rainfall combined with increased land use pressure may lead to expansion of desert areas through the desertification processes (Millennium Ecosystem Assessment, 2005a). On the other hand analysis of results of climate models show, for some models that deserts of the southwest USA will shrink by up to 60% as precipitation increases (Bachelet *et al.*, 2001; Hardy, 2003).

Using past deserts response to global temperature change during the Holocene, especially the last interglacial era (130,000 years ago) it has been noted that it is likely that these systems might act as a globally significant sink of carbon into soils and vegetation also under future climate change (Lioubimtseva and Adams, 2004). But such conclusions are contradicted by suggestions that the current observed changes are of a magnitude not witnessed in history of ecosystems and occur concurrently with continued human pressure on the land surface area (Millennium Ecosystem Assessment, 2005a).

Assessments show variations within models used to predict change in vegetation type and distribution in response to the predicted climate changes and that all models are imperfect in predicting vegetation under current climate (Lioubimtseva and Adams, 2004). Investigations on impacts of climate change in desert ecosystems need to assume a more integrated approach and consider among others, (i)vertical heterogeneity in soil moisture for instance as outlined under the Walter’s two-layer hypothesis but recognising differences in rooting system of woody species and (ii) horizontal heterogeneity in soil moisture which can be enhanced by land use effects giving rise to differences between canopy and intercanopy patches in interception, runoff, evaporation, and plant water use (Breshears and Barnes, 1999).

The spatial and temporal coverage of studies on likely response pattern of desert ecosystems to climate change are limited. Few countries outside the United States covered by desert ecosystems have resources to conduct in-depth experiments on the potential effects of climate change on these systems and ongoing experiments are of short duration. Studies with the longest duration fall between 5 to 8 years, which is too short a period to capture longer-term processes such as plant adaptation and soil biogeochemistry which may result in significant system feedbacks (Morgan *et al.*, 2004b). Further, field based experiments are limited (Morgan *et al.*, 2001b; Milchunas *et al.*, 2004; Morgan *et al.*, 2004a). Results based on experiments conducted in environmental growth chambers do not capture complex field conditions that include for instance, competition for resources between plant communities, differences in species response patterns to elevated CO_2 as well as changes in climate and microclimatic feedbacks (Morgan *et al.*, 2004a). As a result findings of chamber experiments present potential trends because for E.g. they may involved cases where CO_2 is instantaneously doubled for the duration of the experiment while in reality the effect of CO_2 will likely be realized differently in different years and environments, and will evolve over time with exception of cases of extreme events (Morgan *et al.*, 2004a). These limitations constrain comparison across deserts in different regions and limits ability to say with any confidence how even the most intensively studied desert shrub communities of the southwestern United States will respond to rising CO_2 in the long run (Lioubimtseva and Adams, 2004).

Nevertheless understanding of the response of arid to semi-arid land ecosystem to CO_2 enriched environment have improved. It is generally agreed that in the herbaceous layer C_3 photosynthesis is limited by present-day ambient CO_2 concentrations while C_4 photosynthesis is nearly CO_2 saturated (Hamerlynck *et al.*, 2002; Morgan *et al.*, 2004a; Morgan *et al.*, 2004b). However, recent evidence shows that the differences between the two functional groups is not as great when based on their photosynthetic pathway rather evidence shows that in water-limited systems, soil water savings and higher water use efficiency from CO_2-induced stomatal closure is the dominant force behind growth enhancements under elevated CO_2 (Morgan *et al.*, 2001b; LeCain *et al.*, 2003; Nelson *et al.*, 2003; King *et al.*, 2004b). The CO_2 enriched environment stomato closure results in reduced transpiration per unit leaf area and higher soil water content which promotes higher plant biomass yields and water use efficiency (Anderson *et al.*, 2001). It is likely that the arid environments offer more opportunity for CO_2-related improvements in plant water relations than in more wet systems but this needs further investigation.

What is yet to be established is the degree to which results of the CO_2 enrichment experiments produce CO_2 photosynthesis-driven responses vs. indirect, water-driven responses, which are tightly coupled to climatic co-variables such as temperature and humidity(Morgan *et al.*, 2004b). Also there are indications that certain base level of water availability is necessary to detect significant CO_2-induced soil water and plant production responses such that below very dry conditions the CO_2 is not detected E.g. in the Mojave Desert the threshold is believed to lie within rainfall ranges of 150 – 250 mm yr- (Morgan *et al.*, 2004b).

Experimental results in South western deserts of USA have shown that desert shrubs and wet-season semi-desert plants responded strongly to increased CO_2 (of up to 550ppmv, the estimated level for 2050) over wet conditions similar to El Niño events (Bassiri *et al.*, 1998; Smith *et al.*, 2000a). The potential for shrubs to be favoured under higher CO_2 has been noted elsewhere, for instance in Southern Africa (Bond and Archibald, 2003). The results from the US showed greater year-to-year variation in production cycles at elevated CO_2, suggesting that deserts may become even more episodic in future (Smith *et al.*, 2000a). This is to some extent in line with the predicted increase in extreme events for much of the desert regions of Southern Africa (Scholes and Biggs, 2004). Deserts together with semi arid areas have been classified under the current climate as "pulse-reserve" systems in which a pulse such as an extremely wet period triggers active growth among stagnant biological reserves such as seeds and dormant plants and animals until the effects of this phase is reduced (Ogle and Reynolds, 2004; Schwinning and Sala, 2004). Current assessments seem to indicate that climate change is likely to enhance this pattern although others have noted the need for the "pulse-reserve" paradigm to consider the role of soil moisture storage to plant growth (Reynolds *et al.*, 2004).

However, while other growth chamber experiments show that increase in CO_2 will significantly affect desert shrubs, others as well as field experiments suggest no major change with exception to certain individual species (Poorter and Perez-Soba, 2001). Recent evidence showed that stimulation due to elevated CO_2 was strongest (high ratio of standing biomass) during the period of maximum soil moisture content, and lowest during the periods of low soil moisture, E.g. in the Mojave Desert, large increases in new shoot biomass for shrubs at elevated CO_2 occurred over a wet year (Smith *et al.*, 2000a; Hamerlynck *et al.*, 2002; Morgan *et al.*, 2004b). Variations with herbaceous plants have also been observed, for instance introduced annual grass species showed greater response to elevated CO_2, but for some species the quality of the seeds produced was lower indicating the need for further long term assessments (Dukes and Mooney, 1999a; Smith *et al.*, 2000a). These results concur with findings in semi-arid grasslands where for 5 years duration experiments with elevated chamber CO_2 treatment (720+ 20 mmol/mol); the hypothesis of equal species response to elevated CO_2 among C_3 species was rejected, more opportunist species that increase under disturbance but are more susceptible to drought conditions exhibited higher aboveground phytomass, and these were the least digestible grass species due to lower N concentration, a factor reducing forage quality (Morgan *et al.*, 2004a). For the C_3 species, that showed greater response to CO_2 this appeared to be related to increased seed recruitment (Morgan *et al.*, 2004b). The strong responses noted over the five period also showed that the likelihood that the effect of atmospheric CO_2 concentrations in arid to semi-arid ecosystems will be strongly featured over the 2050 to 2070 period.

Indications from studies investigating likely trends in adaptation to seasonal and inter-annual variability in rainfall in arid regions are that increased occurrence of summer drought in future could favour the most deep-rooted shrub species (Schwinning and Sala, 2004). Investigations of drought effects on the hydrologic interactions between plants and soil on three perennial species representing different rooting patterns and responsiveness to precipitation pulses in the Colorado Plateau Ecosystem with average precipitation of 215 mm showed that summer drought affected all species more negatively than winter drought. Results also showed that all three species took up deeper soil water under drought conditions and shallow soil water after a large rainfall event in summer. But after a particularly dry summer only the deepest-rooted species which included the *Ceratoides lanata* shrub continued to take up soil water (Schwinning and Sala, 2004). These indications compare with observations made for in semi-arid grasslands that an increase in deep soil water due to elevated CO_2 environment is likely to favour deep-rooted species (Nelson *et al.*, 2003).

More attention is required to understand for instance, how shrubland invasion will change ET and the physical mechanisms involved in these changes. Current evidence points to the fact that ET is the primary cause of loss of water from soil in areas where shrub invasion has occurred. While shrubs have lower transpiration rates as a result of low leaf area index (LAI), rapid evaporation occurs from exposed surfaces. When surface soil moisture is limited, shrubs through deeper roots permit transpiration and in contrast to C_4 grasses, C_3 shrubs can transpire all year (Kurc and Small, 2004). Consequently the linkages between surface soil, ET and plant growth under climate change are poorly understood and significantly limit our capability to assess impacts of climate change.

In one study in Central Mexico desert where infiltration only wets up to 10 cm depth of the dry soil on 80% of rainy days, it was shown that the surface soil layer (0–5 cm) is often the primary source of water for ET during the summer monsoon season (Kurc and Small, 2004). It was concluded from this study that daily ET does not increase with higher root zone soil moisture. But the surface soil layer is the primary source of ET and direct evaporation from bare soil in areas of shrub invasion is likely to be a large component of the total flux (Kurc and Small, 2004). In addition to complex interactions that may result from shrub invasion another influential factor, i.e. dew, needs to be incorporated in models of climate change impacts in desert systems. Dew is an important source of moisture for plants, biological crusts, insects and small animals (Jacobs *et al.*, 2000).

A non-native invasive grass in the US known to increase the frequency of intense fires from a 75–100-year cycle to a 4–7-year cycle, has been shown from experimental results, to be far more productive than native plants during wet years under high CO_2 (Smith *et al.*, 2000a). Earlier studies predicted an increase in annual primary production of above 50% in response to doubling CO_2 in desert regions (Lioubimtseva and Adams, 2004). Such changes could have major effect on desert water cycles, wildlife habitat and socio-economic activities (Hamerlynck *et al.*, 2000). However, these results require further investigation given that deserts are both water- and nutrient-limited systems and current studies have not fully shown how interactions with increased levels of CO_2 will change these limitations (Smith *et al.*, 2000a).

On the other hand data on effects of elevated atmospheric CO_2 on root dynamics in arid native plant communities is insufficient to make generalizations partly due to the diversity of techniques used, and inconsistencies in responses (Norby and Jackson, 2000). Earlier studies on the effects of CO_2 on root production in the herbaceous layer in particular indicated greater increases in belowground production with elevated CO_2 than for aboveground. However, subsequent work observed very large increases in root production with elevated CO_2 during years of intermediate levels of precipitation, with smaller effects in a very wet year and no effect in a very dry year(Milchunas *et al.*, 2004). It has been noted that very wet years or very dry years may overwhelm the relatively small effects of CO_2 on water conservation and therefore these extreme events periods may not be conducive to the water-use-efficiency response to CO_2 for the belowground biomass(Naumburg *et al.*, 2003; Morgan *et al.*, 2004a). Systems with high aboveground to belowground biomass are likely to be more susceptible to grazing and fire and have higher potential for instability during drought (Dukes and Mooney, 1999a; Milchunas *et al.*, 2004).

In terms of biodiversity warming is likely to favour mobile species and eliminate more the rare, isolated sedentary ones (Dukes and Mooney, 1999a; Hardy, 2003). In Chihuahuan desert it is estimated that the habitat of about half the species will disappear (Peterson *et al.*, 2002a; Lenihan *et al.*, 2003).

Desertification

The UN Convention on Combating Desertification (UNCCD) defines desertification as land degradation in arid, semi-arid and dry sub-humid lands resulting from various factors including climatic variation and human activities. Land degradation is defined as a loss of productivity. To facilitate a way to measure degradation, the Millennium Ecosystem Assessment refined the UNCCD definition and defined land degradation as “a persistent reduction in the capacity of ecosystems to supply services" (Millennium Ecosystem Assessment, 2005a).

Numerous studies have shown that arid to semi-arid ecosystems found over 40% of the land surface area of the globe to be highly vulnerable to desertification (Box 4-3). The question is to which extent climate change will make drylands more vulnerable to desertification? If so how will this manifest itself?

Box 4-3: Is desertification in the Sahel related to climate change?

Biophysical characteristics: The Sahel falls in the tropical arid to semi-arid zone marked by a steppe climate with annual rainfall of 150-600 mm. In the north and south the transition into the Sahara desert and the Sudanian savanna respectively are gradual and are known to have fluctuated historically (Hiernaux and Turner, 2002). Regular seasonal monsoonal systems bring rain to the Sahel over a 2-5 month period, peaking in August, but spatial and temporal rainfall distribution is irregular and highly unpredictable (coefficient of variation > 20-30%, Hulme, 2001).

Rains fall in the form of high intensity convective storms during the warm summer, resulting in high potential evapotranspiration. This contributes to runoff and higher soil crusting than would be expected given the sandy soils and limited relief (Hiernaux and Turner, 2002; Nicholson, 2002). Soils contain little organic matter, are weakly structured, and have acidic top soils poor in soluble nitrogen and phosphorus. Dust is commonly formed during the dry season.

The short wet season supports lower biomass compared to arid lands with comparable total annual rainfall such as the Kalahari Desert (Nicholson, 2002). Except for transition zones, the Sahel vegetation is characterized by a lack of succulent, perennial grass and shrub forms typical of similar environments. Instead, due to the rainfall regime, herbaceous species dominate, mainly in the form of an annual grass layer, with scattered trees and shrubs.

The few woody plant species have a diverse phenology but have shown drought induced mass mortality, and regeneration capacity in wet periods. The woody and herbaceous layers compete for water and nutrients in the topsoil (1m) where most of the root system is developed, with a few tree taproots that access deep water sources. This creates a dynamic equilibrium between herbaceous and woody plants, which is thought to prevent the bush encroachment trends seen in comparable systems elsewhere (with exception of areas of slightly greater moisture and nutrient concentrations, Hiernaux and Turner, 2002).

Drought in the Sahel: Unlike many other arid lands, the Sahel experiences intense multiyear drought persistence. Mean rainfall decreased by 25–40% between 1931–1960 and 1968–1997 and since 1970 almost every year has been anomalously dry (Nicholson, 2000; Hulme, 2001). This has resulted in significant socio-economic challenges and has attracted diverse international interest groups focused on the causes of desiccation (Nicholson, 2000; Batterbury and Warren, 2001; Foley *et al.*, 2003).

Although persistent drought was attributed initially to desertification caused by land use pressure, current understanding increasingly rejects this hypothesis (Nicholson, 2000; Xue and

Fennessy, 2002). Desertification does not spread southwards into the savanna systems, rather, vegetation cover responds dramatically to inter-annual rainfall fluctuations (Tucker and Nicholson, 1999; Foley *et al.*, 2003). There is little evidence of large-scale soil denudation, increase in surface albedo, or reduction of the productivity of the land, with the exception of spatially limited degradation processes due to species composition change (Nicholson, 2000). This system may be vulnerable to greater drought frequency, as seedbanks of most Sahelian species are transient, and few species produce long-lived seed (Hiernaux and Turner, 2002). Persistently increasing dust load is thought to be linked to vegetation cover decline, due both to climatic desiccation and human land use (Hiernaux and Turner, 2002). This dust may affect very remote areas, with as a potential link even to Caribbean rainfall patterns (Prospero and Lamb, 2003). While the radiative role of the dust has been demonstrated, the direct link with climate in the Sahel is yet to be confirmed (Nicholson, 2000; Foley *et al.*, 2003).

The Sahara and Sahel have in the past been characterized by sudden, abrupt climatic changes, many of which occurred with no apparent warming (Claussen *et al.*, 1999; Foley *et al.*, 2003; Lioubimtseva and Adams, 2004). For example, The Sahel was more vegetated with grass and shrub cover during the middle Holocene (~6'000 B.P.), (Hoelzmann *et al.*, 1998; Prentice and Jolly, 2000). The Saharo-Sahelian boundary had shifted to 22–23°N, 500 km north of its present location (Foley *et al.*, 1998). However, 5'000 to 6'000 years ago the Sahara and Sahel region switched to much more arid conditions (Demenocal *et al.*, 2000; Foley *et al.*, 2003).

Modeling studies coupling vegetation and atmosphere also simulate an abrupt transition from wet to dry conditions (Claussen *et al.*, 1999). While these results require further validation, they suggest that about 5'500 years B.P. a critical threshold was reached at which gradual reductions in rainfall due to slow changes in the Earth's orbit were amplified by land-surface feedback mechanisms (Claussen *et al.*, 1999). Oceans appear to have played a limited role (Foley *et al.*, 2003).

According to this theory of regime shifts in the Sahel, the most recent regime shift occurred around 1969, resulting in the subsequently experienced droughts. Various studies indicate that the recent, three decade long drought in the Sahel could be a result of complex interactions among the atmosphere, land, and ocean (Foley *et al.*, 2003). This is based on the following evidence:

- The interannual variability of oceans SSTs is linked to inland precipitation, but not so at the decadal scale (Foley *et al.*, 2003).
- Studies using simple models of the atmosphere, ocean, and terrestrial vegetation, suggest that vegetation feedbacks amplify the interdecadal variability of the Sahel precipitation while reducing its interannual variability. Therefore vegetation interactions influence the persistence of drought at the decadal scale (Zeng *et al*., 1999; Wang and Eltahir, 2000).
- Sensitivity studies using GCMs with their highly unrealistic, idealized scenarios of land-cover change greatly exaggerated land degradation (Taylor *et al.*, 2002; Foley *et al.*, 2003).
- The 1969 drought may have been initiated by changes in oceans sea surface temperature (SST) patterns, but the magnitude and multi decadal persistence of Sahel droughts is driven by nonlinear feedbacks between the natural vegetation and the atmosphere (Zeng *et al.*, 1999).

This evidence supports the hypotheses that changing SSTs act as a trigger for climate change in the Sahel, while the vegetation-atmosphere feedbacks reinforce that trigger during the transition until a new equilibrium, again of limited stability, is reached (Foley *et al.*, 2003).

Although more work is required to understand the process, desertification has become less controversial and literature shows greater consensus that it is a product of interactive human, meteorological and ecological processes acting on each other at different magnitude spatially and over time (Dube and Kwerepe, 2000; Reynolds and Stafford-Smith, 2002). There is also consensus, although not fully implemented, that assessment of desertification should consider the interactive processes of climate, fire and land use activities such as herbivory (Reynolds and Stafford-Smith, 2002). Global data bases for modelling desertification are maturing, for instance an attempt has been made to use IMAGE 2.2 to model the vulnerability of land to water erosion under current conditions assuming lowest sensitivity in areas of natural vegetation (Leemans and Kleidon, 2002). However, there remains a limited attempt to integrate biophysical and socio-economic factors, with most studies putting more emphasis on biophysical factors.

There is growing consensus that arid to semi-arid lands are sensitive to desertification because they are episodic systems marked by highly variable rainfall (Holmgren *et al.*, 2001; Leemans and Kleidon, 2002; Nicholson, 2002). Work is ongoing to understand the long term interactive processes operating within the "pulse-reserve" envelope characteristic of arid lands, and the implication of this on desertification and feed back on climate processes. Evidence has shown that desertification processes advance at higher rates during periods of extreme events, for instance in drought years when demand for ecosystems goods and services far exceeds supply (Dube and Pickup, 2001; Holmgren *et al.*, 2001).

Desertification is also influenced by ecological aspects of an area such as soil type, landscape characteristic and vegetation in addition to human and climatic factors. For instance, silty soils have been found to be the most erodible while soils rich in clay were classed to be least erodible (Leemans and Kleidon, 2002). Land degradation processes in arid lands that combine changes in vegetation with changes in soil in the form of for instance scalding, erosion and compaction will have an impact on water and nutrient cycles with consequences on primary productivity (Breshears and Barnes, 1999; Ash *et al.*, 2002). What is critical for arid lands is the slow recolonization of soil surfaces by different algae components binding together the grains of the surface layer. Once eroded underlying soils are exposed to both wind and water erosion and this could be up to at least 20 years after disturbance. In arid lands of southern Utaht soils may take 5000–10,000 years to form as a result desertification processes leading to degradation of soil loss may be considered irreversible (Lioubimtseva and Adams, 2004).

The strong link between drought and shrub recruitment in areas of high land use pressure is also well established (Holmgren and Scheffer, 2001) although more recent evidence points to the need to consider also human influences such as CO2 fertilisation on woody plants due to the already experienced raise to 360 ppmv from pre-industrial levels (Bond and Archibald, 2003). Other evidence has shown that the productivity of shrubs and annual plants which so far are associated with land degradation processes, more than doubled under the elevated CO2 levels estimated for 2050 (Smith *et al.*, 2000a).

In the prairie and forestland of western US and other parts of the world drought following wet events tends to increase human induced fire risk. The risk of fire is higher in areas invaded by annual weeds and as a result, fire perpetuates desertification as prospects for recruitment by perennial species become more remote (Smith *et al.*, 2000a).

Evidence indicates that desertification process that change soil texture and structure and hence impact on soil moisture retention have a potential to influence meteorological processes. Model and station data have linked local precipitation variability with time scale of soil moisture in

West Africa (Nicholson, 2002). More studies are required to demonstrate the occurrence and magnitude of these combined processes in the short to long-term period. Other model results have shown that low latent heat flux resulting from depletion of vegetation which is ultimately linked to changes in albedo as indicated in earlier studies, has a role in weak continental water cycle and altered energy balance in arid landscapes.

Field studies have noted the potential role of woody plants in providing a microclimate that is conducive for regeneration and growth of some of the original perennial species that may be disappearing through processes such desertification (Holmgren and Scheffer, 2001). In southeastern Arizona areas under the canopy of mesquite trees were associated with improved soil physical conditions, enriched nutrient status, reduced soil temperature, higher soil moisture and reduced net radiation and it was concluded that this environment was the reason for the greater cover of perennials such as the Arizona cotton top found under these trees (Tiedemann and Klemmedson, 2004).

However, other findings do not agree with the facilitating theory and its spatial significancy in retaining the original vegetation of an area. It has been noted that trees can intercept a large proportion of rain, for instance a desert shrub was found to intercept about 35% of incoming rain while in Kenya where rain was less than 20mm *Acacia tortilis* intercepted 50 to 100% of the rain (Robinson, 2004). Further, areas invaded by shrubs have been shown to have large inter canopy space leaving soil exposed.

Degraded areas with no shrub recruitment and are dominated by short-living pioneers species with shallower root system will be more susceptible to drought (Snyman, 2005). These areas are exposed to heat and other erosion agents which lead to degradation of soil, loss of moisture retention capacity and depletion of soil carbon. According to the Millennium Assessment report desertification accounts for a billion tons of carbon loss every year and a fraction of this carbon enters the global atmosphere (Millennium Ecosystem Assessment, 2005a)
Despite the large volume of literature linking desertification with changes in primary production the coupling of change in primary production with change in secondary production, for instance livestock production is variable over space and time (Ash *et al.*, 2002). It has proved difficult to demonstrate the loss in secondary production due to land degradation although in most work this loss is assumed to be occurring. The role of landscape heterogeneity in arid lands in the productivity of herbivores has been noted as one factor, which contribute to the apparent lack of coupling of primary and secondary production to help assess the role of land degradation (Ash *et al.*, 2002). In other cases the effect of degradation is masked by external interventions such as supplementary feeding and changes in species composition for instance in Southern Africa there are ongoing efforts to switch from cattle ranching to game farming.

4.4.5 Tundra

[LA had an accident, needs to be completed]

4.4.6 Mediterranean ecosystems

Mediterranean-climate systems are globally unique because of their wet winters and dry summers. They are located in the mid-latitudes of all continents, mainly on western coastal regions (Mediterranean basin, south-west coast of North America, South Africa, and Australia). Although of limited spatial extent, these systems are important because of their high levels of biodiversity (Cowling *et al.*, 1996) and the considerable services they provide for the human societies that exist in these regions., Their biodiversity is at present threatened by intense human pressures (Myers *et al.*, 2000),. These systems were not explicitly reviewed in previous IPCC assessments, but it was suggested that they are generally threatened by desertification as they are quite sharply separated from adjacent semi-arid and arid systems, and only relatively minor warming and drying may be sufficient to shift this border.

Mediterranean type systems are shrub-dominated and fire-prone, due to their dry summers, and much research here has focused on this aspect (Montenegro *et al.*, 2004), in addition to the general mechanisms of plant species response to drought and nutrient limitation. These systems have evolved since the Miocene as flammable systems and have possibly experienced relatively minor changes in climate during the Pleistocene, allowing the diversification of endemic and near-endemic families. A long history of human impacts in the Mediterranean basin itself is reflected in a strongly disturbance-tolerant flora (Lavorel, 1999), but land abandonment, mainly in the Mediterranean Basin, underlies increasing ongoing forest reestablishment (Mouillot *et al.*, 2003).

Climate change-induced impacts on vegetation in these systems are likely to be mediated strongly by fires, which may increase in frequency and favour shrubs over trees (Mouillot *et al.*, 2002), and sprouting plants over those that regenerate by seeds (Bond and Midgley, 2003). A double CO_2 scenario suggests a 40-50% increase in fire escapes in California (Fried *et al.*, 2004). Indeed, increased fire frequencies have been observed in some parts of the Mediterranean basin (Pausas and Abdel Malak, 2004), but not in others (Mouillot *et al.*, 2003). Increased fire frequencies have been simulated for the Mediterranean Basin, accompanied by an increased dominance of fire-tolerant shrubs (Mouillot *et al.*, 2002). In California, notable switches in dominant vegetation types such as from needle-leaved species to broad leaved trees, and changes in the competitive balance between trees and grasses due to increased fire frequencies have been projected together with the concomitant implications for vegetation productivity and carbon sequestration (Lenihan *et al.*, 2003). With increasing precipitation, however, an increased success of forest types, and greater carbon storage has been projected in California (Bachelet *et al.*, 2001). In the Mediterranean region itself, vegetation structural response to a doubled CO_2 climate is projected to be strongly sensitive to rainfall, with possibly increased forest success in the eastern Mediterranean and Turkey, and increased shrub cover in northern Africa if rainfall does not decrease (Cheddadi *et al.*, 2001).

Significant warming trends have been noted in the Mediterranean Basin (Rebetez, 2004) and in the Cape Floristic Kingdom (Warburton *et al.*, 2005). Warming has lengthened the growing season by 16 days in the Mediterranean Basin (Penuelas and Filella, 2001a). Experimental warming has been shown to cause shifts in species relative dominance in Mediterranean Basin communities (Wessel *et al.*, 2004).

The impacts of rising atmospheric CO_2 on vegetation may counteract some effects of drying trends on plant primary productivity (Drake *et al.*, 1997; Ainsworth and Long, 2005b), and may already have counteracted negative impacts of climate trends during the 20th century (Osborne *et al.*, 2000). Experiments have provided mixed results, with many species showing only marginal benefits (Dukes *et al.*, 2005), and limited increases in above-ground productivity (e.g. Blaschke *et al.*, 2001). Ecosystem carbon storage may increase due to reductions in litter decomposition rate induced by elevated CO_2 (De Angelis *et al.*, 2000).

Reductions in rainfall frequency have been projected for some Mediterranean regions (e.g. Cheddadi *et al.*, 2001), which is likely to exacerbate drought stress conditions. Soil water content exerts direct control over ecosystem water and CO_2 flux in Mediterranean Basin system below key water content thresholds (Rambal *et al.*, 2003), and may substantially reduce ecosystem carbon and water flux (Reichstein *et al.*, 2002). The anomalous 2003 summer drought event in Europe exerted a major physiological impact of the on Mediterranean vegetation and ecosystems, but most showed a full recovery by 2004 (Gobron *et al.*, 2005).

Experiments at the individual plant level confirm high levels of drought resistance in established Pinus halepensis (Borghetti *et al.*, 1998). However, Ponderosa pine forests showed marked reductions in productivity and water flux at key levels of water stress encountered during a 1997 heatwave, which did not recover for the rest of the season, indicating the existence of threshold responses to extreme events (Goldstein *et al.*, 2000). Mediterranean Basin pines (Martinez-Vilalta and Pinol, 2002) and other woody species (Penuelas *et al.*, 2001), showed species-specific tolerances to drought stress in the dry summers of 1994 and 1998.

Experimental drying differentially reduced the productivity of Mediterranean basin shrub species (Llorens *et al.*, 2003; Llorens *et al.*, 2004; Ogaya and Penuelas, 2004) and tree species (Ogaya and Penuelas, 2003), and even the flowering phenology of Mediterranean basin shrub species (Llorens and Peñuelas, 2005), which suggest complex changes in the relative success of species in this region under drying scenarios. Species-specific tolerance to drought in woody plants can be explained by their xylem hydraulic characteristics (Martinez-Vilalta *et al.*, 2002). Drought may also act indirectly on plants by reducing the availability of soil phosphorus (Sardans and Penuelas, 2004). A mediterranean desert ecosystem showed net losses of carbon in dry years (Hastings *et al.*, 2005), suggesting increasing carbon efflux under a drying scenario.

Some bioclimatic niche-based modeling approaches suggest a substantial threat to species persistence and species richness, with significant range reductions projected for the majority of species modeled in the Cape Floristic Region (Midgley *et al.*, 2002; Midgley *et al.*, 2003). Tree and shrub geographic ranges are projected to shift unpredictably, and fragment, under IS92a scenarios (Shafer *et al.*, 2001). In southern Europe, species turnover rates are projected to be high under a range of scenarios (Thuiller *et al.*, 2005). Reductions in range size threaten an increase in the likelihood of species extinctions, with as many as 30-40% committed to extinction by 2050 (Thomas *et al.*, 2004b). Species of lowland plains appear to be at higher risk than those associated with mountain ranges both in California (Peterson, 2003) and the Cape Floristic Region (Midgley *et al.*, 2003), although in the Mediterranean Basin, mountain species show a high risk (Thuiller *et al.*, 2005).

4.4.7 Wetlands, Freshwater lakes and rivers

Several detailed accounts of the global distribution, functions and values of inland aquatic ecosystems, i.e. streams, rivers, lakes, reservoirs and wetlands, have been published in recent years (Mitsch and Gosselink, 2000; Gitay *et al.*, 2001; Wetzel, 2001). Rivers transport not only water, but also nutrients, carbon and various other chemical compounds from the land to the oceans. Wetlands play a significant role in the transformation of materials passing through them from land to rivers and lakes. The endorheic lakes and northern peatlands serve as a sink for sediments and carbon, respectively (Schlesinger, 1997; Mitsch and Gosselink, 2000). However, it is important to note that inland aquatic ecosystems are not evenly distributed: Some of the mighty rivers drain the tropical and subtropical lands, several of them pass through highly arid regions. The vast majority of natural freshwater lakes is located in the higher latitudes and the largest proportion of wetlands (peatlands) is in the boreal region. The Amazon transports several times more water to the oceans but only a fraction of the sediments than that carried by Rivers Ganga and Brahmaputra together. The delta of Ganga-Brahmaputra supports high biodiversity and is highly productive.

Inland aquatic ecosystems are closely associated with the global water cycle. Their nature and extent depend directly upon the hydrological processes which are determined by temperature regimes. The structure and functioning of the aquatic ecosystems, and hence their goods and services, are affected by both components of global climate change- the rise in temperature and the change in precipitation regimes. However, the aquatic ecosystems closely interact with their terrestrial watersheds and are already under such great stress from a variety of anthropogenic pressures that it is often difficult to separate the impacts of climate change and other human activities. Storage, extraction and diversion of water in the rivers alter the hydrological regimes, both up- and downstream, with serious consequences for the biota, water quality and all ecosystem processes. The wetlands are drained and reclaimed. Both domestic and industrial wastewaters are discharged often without treatment into rivers, lakes and wetlands. Various human activities in the catchments result in increasing sediment and pollutant load into aquatic ecosystems.

The projections for future climates (e.g. Cubasch *et al.*, 2001) show that the snow cover and extent of sea ice will decrease, the glaciers will retreat and the global average precipitation will increase. These changes, which will however not be uniformly distributed, will directly affect the aquatic ecosystems through changes in their hydrological regimes.

The projected changes in temperature and precipitation will have several direct and indirect impacts on the availability of water resources, the water quality (see also Chapter 3) and across the entire scale of biological organization - from species to communities to ecosystems (Fig. 4.3).

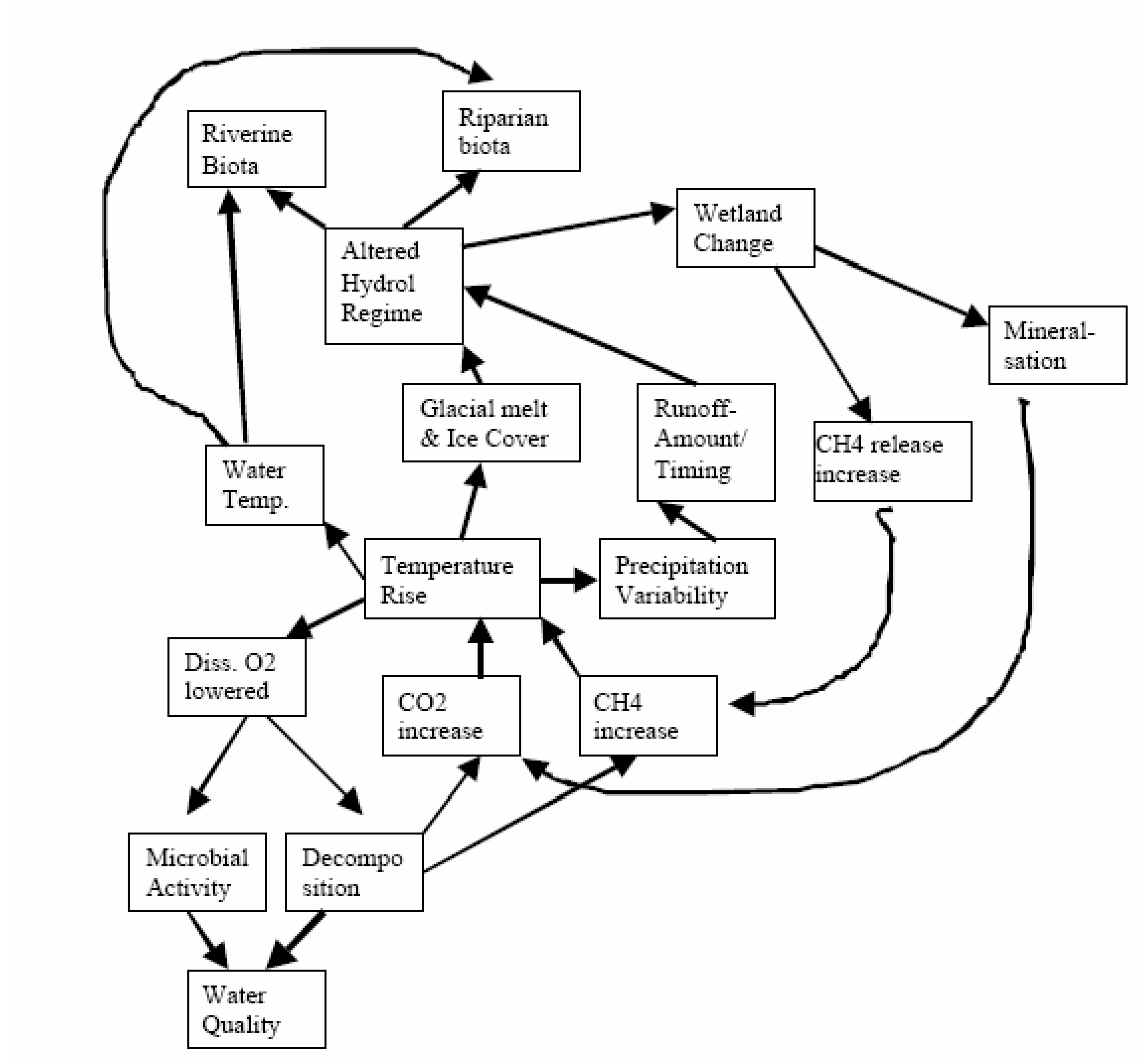

***Fig. 4.3:** Impacts of climate change on inland aquatic ecosystems.*

The rise in water temperature will lower the oxygen concentrations, particularly in the hypolimnion (Verburg *et al.*, 2003). Lower oxygen levels and/or its total lack in the deeper layers will affect the microorganisms and benthic invertebrates, and hence the rates of decomposition, and in turn the water quality. So called “dead zones”, which are totally anoxic and hence lifeless, as observed in Lake Erie in 2001, are expected to become more frequent as a consequence from a warmer climate (Kling *et al.*, 2003).

Growth and distribution of many aquatic and wetland species is determined by temperature. Therefore, an increase in water temperature is bound to shift their distribution range and affect their abundance, and consequently the species composition of communities (Lake *et al.*, 2000). Fish have been of particular interest for their direct economic value. The impacts of climate change on fish diversity and production in aquaculture were discussed in SAR (Everett *et al.*, 1996) and TAR 9 (Gitay *et al.*, 2001). According to (Poff *et al.*, 2002), the coldwater salmon and trout are likely to disappear from a large part of their current range in the USA. Even a 1° C rise in temperature is expected to reduce the trout habitats in streams by 7-16% and a 3° C rise may reduce them by up to 54% (Rahel *et al.*, 1996). In a recent study, Jackson and Mandrak (Jackson and Mandrak, 2002) show that species such as smallmouth bass (*Micropterus dolomieu*) whose northern limit of distribution is restricted by temperature related growth and survival characteristics, is likely to expand into more lakes in Ontario (Canada) and eventually eliminate the cyprinid species such as *Phaxinus eos, Phaxinus neogaeus, Pimephales promelas* and *Margariscus margarita*.

Small increase in temperature has been shown to cause local extinctions of aquatic invertebrates that require prolonged periods of near freezing winter temperature (Lehmkuhl, 1974). Consequently it is expected that climate change may affect the diversity and productivity of the aquatic invertebrates adversely, particularly in northern latitudes (Hogg and Williams, 1996; Poff *et al.*, 2002; Kling *et al.*, 2003).

Another recent analysis of the impacts of climate change in the Great Lakes region (Kling *et al.*, 2003) shows that reduced ice cover and lower water levels in lakes would result in an initial increase in phytoplankton production but later productivity would gradually decline from lacking nutrient regeneration. Changes will also occur in the species composition of algal booms and the timing of their occurrence. Algal blooms of inedible nuisance blue-green algae are expected to dominate the algal community. Consequently all other organisms in the food chain will be affected. An analysis of 40-yr data of Lake Washington by Winder and Schindler (Winder and Schindler, 2004) suggests that the climate affects the physical and biological processes differently such that the energy flow among trophic levels can be interrupted. They observed that Pacific decadal oscillation and ENSO extended the duration of the lake stratification by 25 days , which lead to the development of spring algal blooms. While the rotifer (*Keratella*) peak also increased, the other zooplankton exhibited little or no change. Thus, the altered timing of the phyto- and zooplankton peaks had implications for the food chains, and in turn the water quality. In general zooplankton species are known to differ greatly in their response to thermal regimes (Gerten and Adrian, 2002). Thus especially the seasonal pattern of of warm periods is considered critical, and therefore increasing variability in climate would cause different responses in different parts of the Earth.

The potential impacts of global warming on lakes ecosystems can be gauged by a recent study of 15 Swedish lakes along an altitudinal gradient of 270 to 1140 m, with an air temperature difference of 6° C (Karlsson *et al.*, 2005). The study showed that the productivity varied by an order of magnitude and it was related to the duration of the ice free period and the influence of organic carbon and nutrient inputs from the terrestrial vegetation in their catchments.

From within the tropics, O’Reilly *et al*. (2003) have presented evidence that climate change may have large impacts on aquatic ecosystem functions and services. From the historical data on water temperature, carbon isotope reecords in sediments and fish yields in Lake Tanganyika, they observed that a rise in surface water temperature since the beginning of the 20th century has increased the stability of water column, reduced mixing and decreased deep water nutrient upwelling. Consequently, the data suggest that the primary productivity may have decreased by about 20% and the fish yield have declined by 30%.

River and stream ecosystems will be affected by changes in flow regimes – the amount, timing and the peak of water flows. Thus channel morphology and the interaction of the rivers with their floodplains will be impacted. Though warmer water temperatures favour primary productivity, overall primary production will decline due to the subsequent decrease in flow. Significant changes will occur in the microbial and animal communities because of their narrow thermal requirements and the reduction in flow. Higher atmospheric CO2 concentrations are likely to change the nutritional quality of the terrestrial litter entering streams (Lindroth, 2001; Tuchman, 2002; Tuchman, 2003b; Tuchman, 2003a). Such changes are expected to greatly affect the food web relationships as the utilization of litter by various invertebrates and then microbes is directly related to its chemical quality (Ostrofsky, 1997).

Wetlands are transitional areas (ecotones) between terrestrial and deep open water systems and are consequently sensitive to even small changes in their hydrological regime. Both inland coastal wetlands will thus be the most affected. Reduced flow and lower water levels mean that large areas of wetlands will dry up. While some riparian and lake littoral wetlands may shift towards the water, their total area will depend upon the seasonal water level changes. Lowering of the water table following reduced recharge of the groundwater will further adversely affect many wetlands that depend upon them. The increased variability in precipitation regimes will be a significant factor in affecting the wetlands because the timing, duration and depth of water directly influence the wetland plants and animals at all stages of their life cycle. Gopal and Chauhan (Gopal and Chauhan, 2001) showed that such variability in monsoonal regions risks to decrease wetland biodiversity. In seasonal wetlands, prolonged dry periods will cause a change in the species composition of plant communities as the terrestrial species invade the wetland areas; for example, in Keoladeo National Park (India), the reduced availability of water to the Park has caused terrestrialization with colonization by species such as *Desmostachya, Saccharum* and *Prosopis juliflora* (Chauhan and Gopal, 2001).

Waterfowl and other avian populations that use wetlands are of special interest. Large populations of them seasonally migrate long distances, often between continents. The migrations are influenced by large seasonal changes in temperature, availability of suitable habitats and food in different regions at different times of the year. Global warming together with the change in timing of precipitation in waterfowl habitats will affect considerably the migration routes as reduce the survival of many species up to the risk of extinction. However, changes in wetland hydrology and vegetation are known to affect not only migratory, but also the resident bird populations.

Biodiversity in wetlands has to be seen in a broader context as it is defined to include all those organisms that are directly or indirectly dependent upon wetland habitats at any stage during their life cycle (Gopal and Junk, 2000). Many insect species, which can otherwise be considered terrestrial, depend entirely upon wetland trees, e.g. in the Amazonian floodplains, and their survival will be threatened with the loss of those tree species (Erwin, 1988). In many wetlands such as the ephemeral pools in drylands, the biodiversity is known the least. However, the organisms in these wetlands are tightly linked to the temperature and precipitation for emergence from their resting stages to complete their life cycle. Alterations in the precipitation regimes with increased variability is likely to cause loss of biodiversity (see Graham, 2003).

Coastal wetlands will still be affected more because the reduction in freshwater flows will be accompanied by intrusion of salinity from rising sea levels. Submergence under higher levels of seawater will mean that the coastal wetlands will shift landwards depending upon the coastal morphology and ability of the biotic communities to colonise new areas. The changes in erosional and depositional processes will be another major factor in determining the extent of wetlands. It is estimated in a study of coastal salt marshes that *Spartina* peat will have to accrete at a rate six times higher that at present to counter the effects of erosion and high water levels (Hartig *et al.*, 2000). Less sediments will be transported by rivers whereas the frequent storm surges may cause more erosion. Mangrove wetlands such as Sunderban in the Ganga-Brahmaputra delta will be severely affected. Coastal lagoonal wetlands that are estuarine in character, will turn saline affecting the biotic communities and their productivity. In Lake Chilika, on the east coast of India, the fish and prawn catch increased several fold after the seawater was allowed to enter the lagoon by making a passage through the sand bar. Based on a study of responses of zooplankton to salinity levels, Schallenberg *et al*. (2001) have demonstrated that climate change will alter the biodiversity and structure of zooplankton communities of coastal wetlands and that these changes will have a large bearing on the food chains and other ecosystem functions. More recently, similar changes have been reported also for the phytoplankton communities (Flöder and Burns, 2004).

The effect of warming and increased winter precipitation in northern latitudes will greatly affect the peatlands which are there a dominant component of the landscape. Recent studies confirm that even short-term changes in climate can alter the peatland carbon and nitrogen mineralisation (Keller *et al.*, 2004) and the composition of plant communities (Weltzin *et al.*, 2000; Weltzin *et al.*, 2001; Weltzin *et al.*, 2003; Keller *et al.*, 2004). Whereas these studies relate changes in CO2 and CH4 emission to temperature and water levels, Ziska *et al*. (Ziska *et al.*, 1998) had demonstrated stimulation of methane flux from tropical paddy fields by elevated concentrations of carbon dioxide.

Further impacts on inland aquatic ecosystems will occur through changes in the human use of water and land resources. The decreased supply and increasing demand for water will threaten the aquatic ecosystems directly, and the land use changes will cause stresses through pollution. The future of the aquatic ecosystems will therefore depend greatly upon the adaptive response of the humans in their water management and conservation of wetlands. Integrated water management strategies that adapt to changing availability of water resources will be required. The need for greater storage and diversion of water (including interbasin transfers) can cause more disruption in the flow regimes as water is stored during peak flow periods and released during the dry low flow periods.

Water management can directly influence the rate and trend of climate change itself. For example, the fluxes of methane and nitrous oxides from rice paddies vary considerably from the flooded phase to draining phase (Frolking *et al.*, 2004).

4.4.8 Mountains

Mountain regions occupy roughly 20% of the continental land masses, but because of their great altitudinal range they exhibit, within short horizontal distances, climatic regimes that correspond to those of widely-separated latitudinal belts (Beniston, 2004b). As a consequence, mountain ecosystems feature high biodiversity. Vegetation changes at well-defined ecotones and ecoclines are governed *inter alia* by shifts in climate, and there is thus a strong incentive to monitor and investigate such changes as one possible indicator of abrupt climatic change. There is increasing interest in analysing different regional ecotones for evidence of climatic change, as shown by Diaz (1997). For example, the distribution of potential vegetation types can be used to create regional climate indices, in order to evaluate changes in different climate parameters through time. Another approach consists in identifying regional climatic indices for areas which represent ecotonal boundaries or extreme conditions for current climate. It then becomes possible to determine whether these types of regions undergo changes in time, which may suggest expansion or contraction of such regions, and either amelioration or worsening of conditions in hitherto extreme environments.

Treelines represent one such major ecological discontinuity as trees reach their limit of climatic tolerance; in mountain regions, this occurs at the interface between subalpine vegetation and low-growing alpine vegetation on mountains, and is primarily controlled by the carbon balance (Körner, 1999). As the tree limit is approached, factors other than competition for solar energy to maintain a positive carbon balance assume greater significance; the response of trees is thus to reduce the amount of carbon allocated to wood production and to reduce growth. Photosynthesis at high elevations is thus one of the keys to explain the rapid transition of ecosystems at the treeline (Smith and Knapp, 1990; Körner, 1998). In some mountain regions, such as the European Alps, the treeline is located below its potential limit because of century-old grazing practices; in other regions such as the Himalayas, deforestation has in past decades transformed much of the forest environment and has fragmented many ecosystems (Becker and Bugmann, 2001). Even though the timberline is not a perfect ecocline in many regions, it is an example of a visible ecological boundary that may be subject to change in coming decades. This change could either take place in response to a warmer climate, or as a result of recolonization of pastures that have been cleared in the past for pastoral activities.

McNeely (1990) has suggested that the most vulnerable species at the interface between two ecosystems will be those that are genetically poorly adapted to rapid environmental change. Those that reproduce slowly and disperse poorly, and those which are isolated or are highly specialized, will therefore be highly sensitive to seemingly minor stresses. Not all boundaries are sensitive to climatic change (Körner, 1998; Bugmann and Pfister, 2000), while others may respond with very long lag times (Davis, 1989). There are instances where ecotones are the result of disturbance rather than climate.

Because temperature decreases with altitude by 5-10°C/km, a first-order approximation regarding the response of vegetation to climate change is that species will migrate upwards to find climatic conditions in tomorrow’s climate which are similar to today (e.g., MacArthur, 1972; Beniston, 2000). According to this paradigm, the expected impacts of climate change in mountainous nature reserves would include the loss of the coolest climatic zones at the peaks of the mountains and the linear shift of all remaining vegetation belts upslope. Because mountain tops are smaller than bases, the present belts at high elevations would occupy smaller and smaller areas, and the corresponding species may thus become more vulnerable to genetic and environmental pressure (Peters and Darling, 1985; Bortenschlager, 1993). However, the migration hypothesis may not always be applicable because of the different climatic tolerance of species involved, including genetic variability between species, different longevities and survival rates, and the competition by invading species (Dukes and Mooney, 1999a).
In regions where climatic change may lead to warmer and drier conditions, mountain vegetation could suffer as a result of increased evapo-transpiration. This is most likely to occur in mountain climates under the influence of continental and Mediterranean regimes. Even in tropical regions, however, there are indications that plants are already sensitive to water stress on mountains such as Mt. Kinabalu in the Malaysian part of Borneo (Kitayama, 1996).
The length and depth of snow cover, often correlated with mean temperature and precipitation, is one of the key climatic factors in alpine ecosystems (Körner, 1999). Snow cover provides frost protection for plants in winter, and water supply in spring. Alpine plant communities are characterized by a very short growing season (i.e., the snow-free period) and require water to begin their growth cycle (Keller *et al.*, 2005).

4.4.9 Oceans, shallow seas and their ecosystems

With the exception of some restricted oceanic regions in some models, all regions of the globe are projected to show warming under the enhanced greenhouse gases (GHG) conditions that are anticipated to prevail in the late decades of the twenty-first century (Wang *et al*., 2004b). The corresponding rise in sea level is projected to be 0.09 to 0.88 m (Houghton *et al*., 2001a). Apart from first order changes in temperature and sea level rise, decreases in both upwelling and formation of deep water, stratification of the upper ocenan and reduce the input of essential nutrients have been demonstrated (Lehodey *et al*., 2003; Loukos *et al*., 2003). Increased stratification suppresses upwelling of nutrients into the upper, lighted region of the ocean, which leads to decreased production of phytoplankton. At high latitudes, this decrease could be offset somewhat by a lengthened growing season because the mixed layer shallows earlier and deepens later in the year (Bopp *et al*., 2001). Where phytoplankton production is limited by the length of the growing season, an increase in this length could increase overall system productivity if suitable nutrients are available. Coupled physical/biogeochemical models predict a net decrease (~5 percent) in global phytoplankton productivity under doubled CO2 conditions that increase oceanic thermal stratification and reduce upwelling of nutrients (Cox *et al*., 2000). Open ocean productivity is also affected by natural interannual climate variability, and climate-driven changes in the frequency, magnitude, or timing of any of these phenomena may lead to marked changes in water column mixing and stratification, and ultimately a reorganization of the ecosystem (Fasham *et al*., 2001). Under doubled atmospheric CO2 concentration, recent studies suggest a change of some 20% in oceanic primary production at both regional and global scales. The trend is negative at low latitudes, where lower surface nutrients decrease productivity, and positive at high latitudes where better light conditions extend the duration of the growing season.

Temperature extremes (both high and low) can be lethal to organisms. At sub-lethal levels, temperature influences growth and metabolism, governs animal behavior and distribution patterns of organisms, and acts in concert with other environmental variables such as dissolved oxygen. However, prediction of how increased temperature might affect webs or systems as a whole remains problematic, Scheffer (2001). Nevertheless, we can expect that some ecosystems may shift abruptly and with little warning to an alternative stable state (Kennedy *et al*., 2002).

Temperature influences the timing of reproduction and controls rates of egg and larval development (Kennedy *et al*., 2002). For example, recent declines in the abundance of winter flounder in New England might be due to the negative effects of warmer winters on egg survival and larval hatching and development (Keller and Klein-MacPhee, 2000). Evidence from the North Sea also suggests that fish recruitment is primarily influenced by environmental factors Clark (2001). Thus, fluctuations in fish abundance may be due to changes in distribution influenced by changes in water temperatures. For example, the abundance of some Lusitanian species, especially pogge (Agonus cataphractus), butterfish (Pholis gunellus), and eelpout (Zoarces viviparus), correlate with surface sea temperature and salinity, and warmer summers are also associated with an increased recruitment of taxa situated at the edge of their normal geographic distribution in the North Sea, such as bass (Dicentrarchus labrax) and red mullet (Mullus surmuletus). Climate change may also alter species interactions by changing the timing of physiological events (Penuelas and Filella, 2001b).

Ocean biological response to climate warming between the beginning of the industrial revolution and 2050 has been projected fro a range of six coupled climate model simulations (CSIRO, GFDL, HADLEY, IPSL, MPI, NCAR) Sarmiento *et al*. (2004). Based on this, climate warming leads to a contraction of the highly productive marginal sea ice biome by 42% in the Northern Hemisphere and 17% in the Southern Hemisphere, and leads to an expansion of the low productivity permanently stratified subtropical gyre biome by 4.0% in the Northern Hemisphere and 9.4% in the Southern Hemisphere (Table 4-2). In between these, the subpolar gyre biome expands by 16% in the Northern Hemisphere and 7% in the Southern Hemisphere, and the seasonally stratified subtropical gyre contracts by 11% in both hemispheres.

Table 4.2: *Percent change of the average response of biogeographical province areas to global warming averaged over the period 2040 to 2060 (Sarmiento et al., 2004).*

	Indian Ocean	Pacific Ocean	Atlantic Ocean	Global
	%Δ	%Δ	%Δ	%Δ
		Northern Hemisphere		
Marginal sea ice		-45.3	-37.5	-41.7
Subsolar		13.9	19.7	16.2
Subtropical seasonal		-13.4	-9.4	-10.9
Subtropical permanent	2.3	2.9	7.4	4.0
Low-latitude upwelling	-6.8	1.6	0.8	0.4
		5ºS to 5ºN		
Upwelling	10.1	1.3	2.4	3.2
Downwelling	-19.5	-4.3	-10.9	-9.7
		Southern Hemisphere		
Low-latitude upwelling	-0.7	-7.3	4.8	-2.5
Subtropical Permanent	7.3	9.7	10.8	9.4
Subtropical seasonal	-3.3	-14.7	-18.4	-10.6
Subpolar	18.7	3.3	3.4	7.9
Marginal sea ice	-23.7	-16.9	-10.3	-17.2

Variation in the areal coverage of sea ice and timing of the spring retreat has strong effects on the productivity of the Bering Sea marine ecosystem (Kennedy *et al*., 2002). For example, the timing of the spring phytoplankton bloom is directly tied to the location of the sea ice edge over the southeast Bering Sea shelf (Stabeno *et al.*, 2001).

The integrated primary production results summarized in Table 4-3 represent the combined effect of the primary production changes obtained by Sarmiento *et al.* (2004) with the area changes of Table 4.2. The large reduction in area of the Northern Hemisphere marginal sea ice biome overwhelms the increase in average primary production in this region, resulting in a net reduction of 33% and 39% in the total primary production calculated with two phytplankton primary productivity algorithms (Behrenfeld and Falkowski, 1997 primary production models; Marra *et al.*, 2003). By contrast, a large increase occurs in the Northern Hemisphere subpolar gyre biome, where the changes in area and average primary production have the same sign. Total primary production in the Southern Hemisphere subpolar biome also increases by a large amount for the same reason, but the marginal sea ice biome in this hemisphere changes only modestly due to a cancellation between the large increase in primary production and smaller area decrease in this hemisphere than in the Northern Hemisphere.

Table 4-3 *Percent changes of the predicted response of integrated primary production (Pg carbon yr*$^{-1}$*) to Global Warming for the Period 2040 to 2060 (Sarmiento et al., 2004).*

	Indian Ocean	Pacific Ocean	Atlantic Ocean	Global
	%Δ	%Δ	%Δ	%Δ
		B&F Model		
Northern Hemisphere				
Marginal sea ice		-36.2	-29.6	-33.2
Subpolar		37.4	22.7	29.4
Subtropical seasonal		-13.1	-4.4	-7.4
Subtropical permanent	-5.1	4.1	0.6	2.2
Low-latitude upwelling	-11.1	-2.3	-7.9	-5.8
5ºS to 5ºN				
Upwelling	2.6	-11.9	-7.8	-8.9
Downwelling	-22.3	-11.6	-9.3	-14.0
Southern Hemisphere				
Low-latitude upwelling	-7.4	-10.2	-0.9	-5.4
Subtropical permanent	3.4	5.4	14.7	7.4
Subtropical seasonal	0.1	-11.0	-12.0	-5.8
Subpolar	26.0	8.4	19.5	16.4
Marginal sea ice	-8.1	-6.3	6.9	-2.7
GRLobal	0.8	-1.0	2.1	0.7
		MARRA Model		
Northern Hemisphere				
Marginal sea ice		-41.0	-35.6	-38.9
Subpolar		30.5	22.2	26.1
Subtropical seasonal		-8.5	-0.6	-3.5
Subtropical permanent	12.5	16.3	15.0	15.3
Low-latitude upwelling	5.4	15.7	11.1	12.3
5ºS to 5ºN				
Upwelling	18.3	8.7	12.7	11.2
Downwelling	-11.3	4.5	1.1	-0.3
Southern Hemisphere				
Low-latitude upwelling	8.2	1.9	13.2	8.1
Subtropical permanent	17.6	15.7	18.9	16.8
Subtropical seasonal	-0.5	-11.3	-11.0	-6.0
Subpolar	21.3	6.5	16.2	13.6
Marginal sea ice	-16.4	-4.8	6.8	-4.1
GRLobal	8.2	7.1	8.3	8.1

In tropical marine ecosystems, coral reefs (Box 4-4) are threatened directly at local and regional scales by human activities, including over-harvesting, deforestation, modification and engineering practices, and general decline of coastal environments due to increasing population pressures. These threats are in addition to natural stresses that have always existed on coral reefs such as storms, freshwater inundation and seismic and volcanic events (Burke *et al.*, 2002; Wilkinson, 2002; Burke and Maidens, 2004; Wilkinson, 2004). Sea level change has the potential for affecting reef distribution globally, because many corals survive only in shallow water. Fossil corals found above or below present reefs demonstrate variations in past sea level.

However, most of the ~0.5 m of sea-level rise expected by 2100 due to heating of the oceans is not enough to "drown" coral reefs, but the potential degradation of water quality and increased erosion of shorelines resulting from flooded coastlines would mean a source of stress (Kleypas *et al*., 2001; Buddemeier *et al*., 2004; Henderson, 2005). On the other hand more rapid and frequent sea-level changes than explained by orbital forcing has registered as breaks in coral growth Thompson (2005).

In the case of organisms inhabiting the rocky intertidal zone, they have emerged in recent years as potential harbingers of the effects of climate change on species distribution patterns in nature, and would be the most affected due to changes in sea-level (time of exposure), and air and water temperature. Thus, climate change may not lead to a poleward shift in the distribution of intertidal organisms, as has been proposed, but instead will likely cause localised extinctions at a series of "hot spots" (Helmuth *et al*., 2002).

It is predicted that increasing global temperatures will shift species' geographic ranges to higher latitudes or altitudes (Helmuth *et al*., 2002). However, temperature changes may have more immediate effects on local populations by altering the interaction between a species and its competitors, mutualists, predators, prey, or pathogens. If these interactions are sensitive to temperature, small climatic changes could generate system-wide ecological changes. Wilkinson (2002; 2004) reported that approximately 16% of the world's reefs were seriously damaged in El Niño 1998, linked with droughts and higher than average sea surface temperatures, and that 40% of these reefs are either recovering well or have recovered. However, the recovery is not uniform and many reefs virtually destroyed in 1998 show minimal signs of recovery. This recovery could be reversed if the predicted increases in ocean temperatures occur as a result of increasing global climate change. Yet reefs are in serious decline; an estimated 30% are already severely damaged, and close to 60% may be lost by 2030 (Wilkinson, 2002).

Increasing sea surface temperatures and CO2 concentrations provide clear evidence of global climate change in the tropics (Fig. 4-4), and current predictions are that the extreme events of 1998 will become more common in the next 50 years, i.e. massive global bleaching mortality will be a regular event (Abram *et al*., 2003; Wilkinson, 2004). Hughes (2003) shows different models of future scenarios for coral reefs and climate change (Box 4-4). The first one shows a popular model with an invariant bleaching "threshold" at ~1°C above mean summer maximum temperatures (Fig. 4-5a). This threshold will be chronically exceeded as temperatures rise over the next 50 years, leading to predictions of massive losses of all corals. However, it must be consider that bleaching is conspicuously patchy and can be attributed to extrinsic environmental patchiness (e.g. temperature, light, turbulence), as well as intrinsic differences (phenotypic and genetic) among corals and their microalgal symbionts (Brown *et al*., 2000; Dunne and Brown, 2001). Therefore, bleaching thresholds are more realistically visualized as a broad spectrum of responses (Fig. 4-5b). Furthermore, bleaching susceptibilities may also change over time as a result of phenotypic and genetic responses (Fig. 4-5c). In particular, substantial geographic variation in bleaching thresholds within coral species provides circumstantial evidence for ongoing evolution of temperature tolerance (Hughes *et al*., 2003).

Box 4-4: Coral reefs and mass bleaching– are coral reefs endangered by climate change?

Since the Third Assessment Report a clearer picture has emerged that Climate Change is the main threat to Coral Reefs, and they have become one of the clearest indicators of its ecological impacts. Coral reef mass bleaching has now been observed worldwide since the early 1980's (see reviews of Williams and Bunkley-Williams, 1990; Glynn, 1996; Brown, 1997; Pêcheux, 1998; Hoegh-Guldberg, 1999; Hughes *et al*., 2003; Coles and Brown, 2003; Hoegh-Guldberg and Hoegh-Guldberg, 2004), with only the relatively cool Gulf of Aquaba (Red Sea) being exceptions.

Trends and vulnerabilities: Mass bleaching was first noted in 1979 in Bonaire (Williams & Bunkley-Williams, 1990). There was widespread bleaching in the Great Barrier Reef in early 1982. The late 1993 El Niño-Southern Oscillation (ENSO) was associated with 95% mortality in the East Pacific, and many other regions worldwide, notably the Caribbean. The most severe mass bleaching events on record were observed in 1987-1988, 1994-1995 and 1998, and with more than 95% mortality specifically in the Indian Ocean. The 2002 Great Barrier Reef bleaching event was also severe. Mass bleaching appears to be increasing in frequency and magnitude (Hoegh-Guldberg, 1999).

Differentiation of bleaching with depth is often observed, as for the 1998 Indian Ocean event where shallow corals were the most affected. But in other events, even reversed depth trends down to at least 100 m depth occurred a few kilometres apart (Williams and Bunkley-Williams, 1990), possibly due to local water circulation patterns. Mass bleaching occurs clearly at peak of summer warming with above normal temperatures, and is often associated with anticyclonic "doldrum" conditions, i.e. windless conditions, and a clear sky and high levels of illumination. In relation to a 1980 baseline, reefs are now roughly 20% destroyed, 24% critically damaged, and 26% threatened, with an overall 48% loss of coral cover (Wilkinson, 2004). Reefs are indeed affected by many local anthropogenic stresses (influx of nutrients, herbicides/pesticides, sewage and other pollutants, siltation, overfishing and destructive fishing practices, and the impacts of tourism and the aquarium trades) and global pressures (global warming, ocean acidification, weather changes**)**. The latter pressures are overwhelmingly responsible for so-called "mass bleaching" (Wilkinson, 2004), responsible of two third to nine tenth of coral loss (high confidence).

Corals bleach when temperature is anomalously high with "hot spots", 0.5-1°C for one month (or dose equivalents) above the mean of the hottest month (Goreau and Hayes, 1994, Strong *et al.*, 1997, Berkelmans, 2002, Lui and Strong, 2003, Hoegh-Guldberg and Hoegh-Guldberg, 2004). Bleaching affects all reef photosynthetic symbioses, the basis of this ecosystem. This includes not only hard corals but all other cnidarians associated with dinoflagelates, molluscs (such as the giant clam *Tridacna*) with dinoflagelates, sponges harbouring either dinoflagelates or cyanobacteria, unicellular large foraminifers bearing diatoms or chlorophytes, and probably ascidians (prochordate) in symbiosis with the "pro-eukaryote" *Prochloron* sp. (synthesis in Pêcheux, 1998; Cockey *et al.*, 1997). Calcification ceases in bleached corals, and sometimes even in non-bleached corals (Mendes and Woodley, 2002). Mortality in 700 year-old colonies (cf. in Hoegh-Guldberg, 1999), and an absence of species composition shift for the past 3000 years in a Belize lagoon, is indications that mass bleaching is a new phenomenon (Aronson *et al.*, 2002).

Adaptive responses: Corals often recover in the months following bleaching. Mortality occurs in roughly half of the events (low confidence). Recovering corals have depleted lipid reserves, and impaired reproduction (see Hoegh-Guldberg, 1999) as do symbiotic foraminifers (Talge *et al.*, 1997). About 40% of the reefs affected in 1998 are recovering with new recruitments (Wilkinson, 2004).

Corals are long lived and therefore have limited potential for rapid evolution (but see Hughes *et al*., 2003). However, it is possible that corals may bleach to shift to temperature-resistant symbionts (the Adaptative Bleaching Hypothesis, (Buddemeier and Fautin, 1993)). Much recent research suggests an as yet undefined capacity in corals and zooxanthellae to adapt to conditions that have induced coral bleaching (reviewed by Coles and Brown (2003)), but there is no conclusive evidence for symbiont shift as opposed to selection by differential mortality. Adaptation by a poleward migration is possible, as *Acropora* along Florida (Precht and Aronson, 2004) but is limited by the steep winter 18°C isotherm gradient, and the rate of climate change compared to glacial/interglacial shifts (cf. Hughes *et al*., 2003).

Interacting stresses: Rising atmospheric CO_2 reduces seawater pH (-0.103 pH unit, +26.7% H^+ concentration) and aragonite and calcite $CaCO_3$ saturation (-15.5%, %), prompting experiments to manipulate calcium availability (Gattuso *et al*. 1998). A consistent 20%-60% reduction of calcification in corals with a lowering $CaCO_3$ saturation from a doubling CO_2 has now been projected (refs in Reynaud *et al*., 2003), and symbiotic photosynthesis is also negatively affected (ter Kuile *et al*., 1989, Allemand *et al*., 1996, Reynaud *et al*., 2003), and perhaps by bleaching impacts (Pêcheux, 2002).

Reefs are becoming overgrown by algae world-wide (Szmant, 2001), in particular in Caribbean (Gardner *et al*., 2003), suggesting a disruption of the coral/algal competitive balance. Attribution to either bottom up (eutrophication) or top down mechanisms (drop in herbivory due to overfishing) is controversial (reviewed by McCook *et al*., 2001). After bleaching, dead corals are quickly colonised by turf algae, possibly inhibiting later corals recruitment (e.g. McClanahan *et al*., 2001). Foraging pressure of the sea urchin *Eucidaris* increased after bleaching (Robinson, 1985). Colonisation of partially bleached corals by algal mats allowed grazing by damselfishes inflicting additional coral mortality, and providing further substrates for algal mats (Glynn, 1990). In Panama 1983, the coral-eating sea star *Acanthaster* crossed the ring of pocilloporids, left unprotected by small crustaceans eliminated by bleaching, and devasted the lagoon (Glynn, 1985).

Projections of impacts: Model application of current bleaching thresholds using three ECHAM and one CSIRO Global Circulation Models for seven world reef locations predict near annual bleaching events by 2040, and even 2020 in particular places (Hoegh-Guldberg, 1999). Similar dates for high frequency bleaching events have been projected for the Indian Ocean using HadCM3 simulations (Sheppard, 2003). More complex modelling efforts for the Great Barrier Reef and Caribbean (Wooldridge *et al*. 2005; Langmead and Sheppard, subm.), show that the switch to algal dominance will occur in 2030 to 2050, with the additional influence of fishing management. If corals shift to, or are selected with symbionts with a +2°C resistance, demise of reef facies can be delayed to roughly 2100.

Ecosystem services: Reefs are essential for local shore protection (see Chapter 6), fisheries and tourism industries (see, for the Great Barrier Reef, full analysis of Hoegh-Guldberg *et al*., 2000, Hoegh-Guldberg and Hoegh-Guldberg, 2004). Reef fisheries supply only about 2-5% of the global harvest, but it is critical source of protein for local people (Pauly *et al*., 2002). The assessment of reef biodiversity is yet to be completed, but they may be more diverse than tropical forests.

Knowledge gaps: Monitoring has made good progress though still more is required. In relation to forest ecologists, the number of reef scientific researchers is lower by ~2 orders of magnitude. Particularly needed are climatologists to develop scenarios of change, intensified collaborations with biochemists in order to quantify the mechanisms that promote thermal tolerance (Coles and Brown, 2003), and the development of credible mechanistic models of coral reef response.

Although it has been sustained that bleaching is “adaptive” (Baker, 2001; Baker, 2003; Baker *et al.*, 2004; Rowan, 2004) offering a high-risk ecological opportunity for reef corals to rid themselves rapidly of suboptimal algae and to acquire new partners, bleaching is still followed by high mortality, reduced growth rates, and lower fecundity (Fitt *et al.*, 2001; Baird and Marshall, 2002; Douglas, 2003; Wilkinson, 2004). Baghooli & Hidaka (2003) suggest that even though zooxanthellae from different host have different stress susceptibilities to temperature, the host itself plays a significant role in determining bleaching susceptibility of corals. On the other hand, accelerating rate of environmental change could exceed the evolutionary capacity of coral and zooxanthellae species to adapt. In 1998, lethal SSTs varied by 5°C (from <29°C to >34°C) depending on location. According to Sheppard (2003) a modest acclimation or adaptation by corals would greatly prolong time, that is by raising the SST presumed to be lethal at a site by 2°C, corals would prolong nearly a century (Fig. 4-6) before their ‘extinction date’.

[Above text needs to be coordinated with Box 4-4 "Coral reefs"]

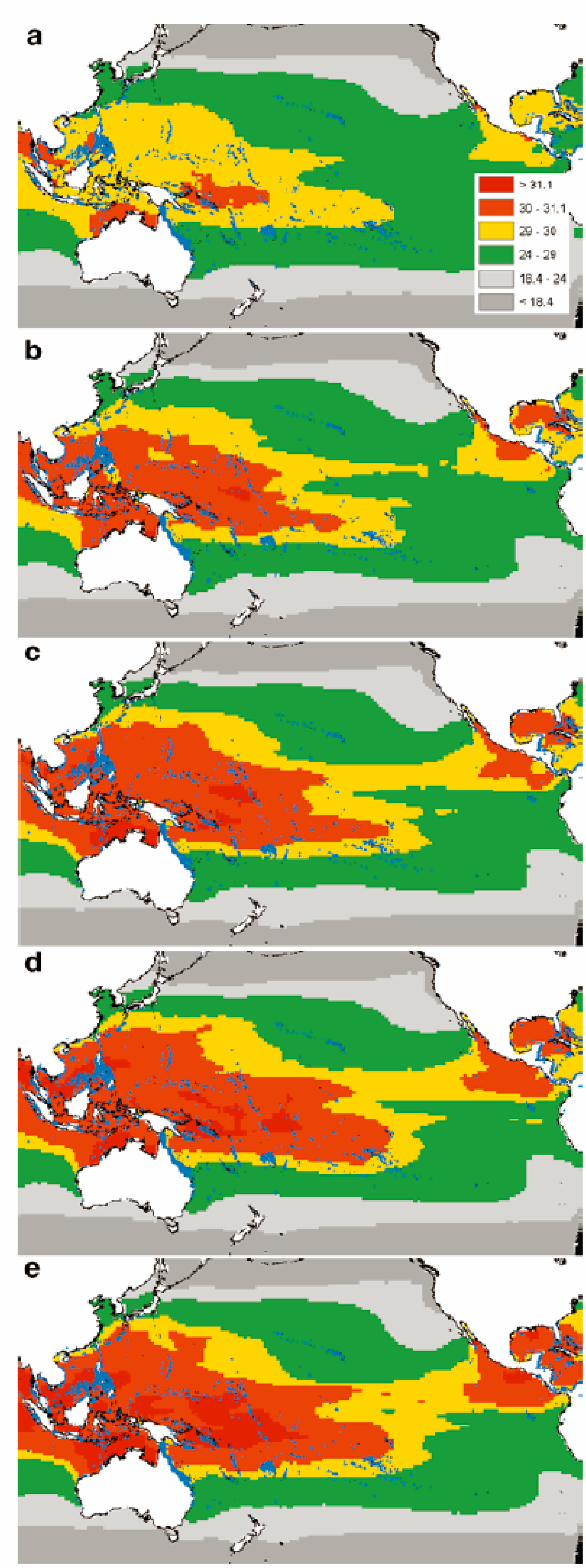

Fig. 4.4: *Maximum monthly sea surface temperature values.* ***a:*** *Observed 1982-1991 temperatures (Reynolds et al., 2002).* ***b:*** *Projected values, 2000-2009; pCO2=375 ppmv.* ***c:*** *Projected values, 2020-2029; pCO2=415 ppmv.* ***d:*** *Projected values, 2040-2049; pCO2=465 ppmv.* ***e:*** *Projected values, 2060-2069; pCO2=517 ppmv (Guinotte, 2003).*

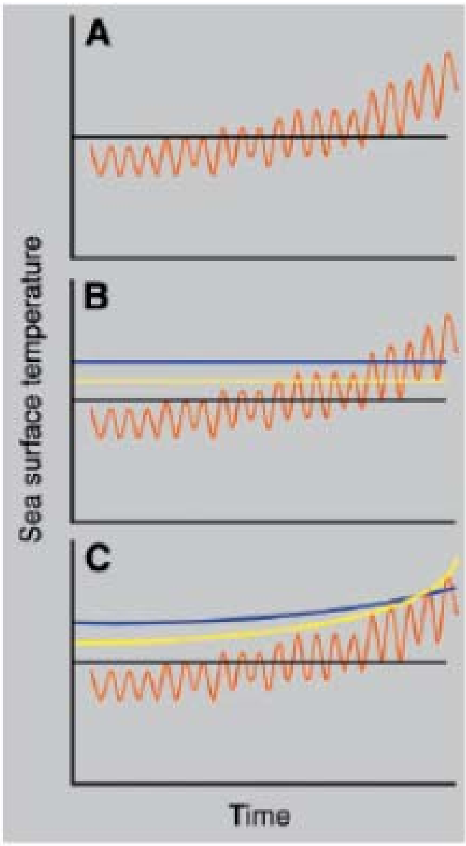

Fig. 4.5: *A) A model showing a constant coral bleaching threshold, which is likely to be chronically exceeded in the future as oceanic temperatures increase (Hoegh-Guldberg, 1999b). B) An alternative model that incorporates differences in bleaching thresholds (e.g., among species, depth, and locations}, indicated by parallel lines. C) A more realistic scenario where changes in thresholds also occur over time, attributable to acclimation and evolution (Hughes et al., 2003).*

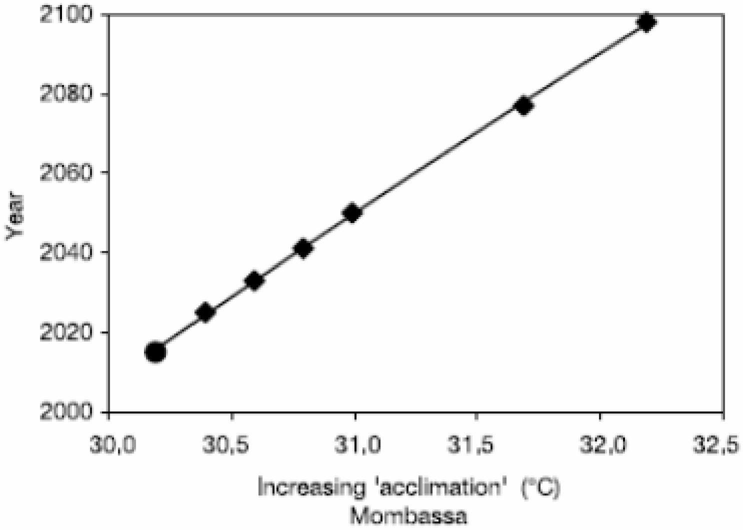

Fig. 4.6: *Recession of time to extinction date with imagined acclimation of corals by up to 2°C with Kenya as the example. Dot is the existing situation, diamonds are the extinction date given coral 'acclimation' (Kleypas et al., 2001).*

Most of the coupled global climate models simulations driven by increased greenhouse gases show similarities between the global warming conditions and those of an El Niño event, that is an extension of warm waters towards the central and eastern equatorial Pacific associated with a decrease in equatorial divergence and primary production. Tuna population would spread towards presently temperate regions, based on predicted warming of surface water and increasing primary production at mid and high latitudes. This pattern is already observed during El Niño events in the catch of local fishermen (Loukos et al., 2003). El Niño events have a positive impact on skipjack recruitment in the Pacific Ocean (Lehodey et al., 2003) and the skipjack population of recent years is at an all time high in relation to the last 30 yr (Hampton, 2002; Maunder, 2002). The explanation would be the increased frequency of El Niño events during the last three decades, in correspondence with the regime shift of the Pacific Decadal Oscillation (PDO) in 1976 (Loukos et al., 2003).

Recent coral bleaching events point convincingly to global warming as an immediate and acute threat to coral reefs (Spalding and Jarvis, 2002; Wilkinson, 2002; Wilkinson, 2004). However, evidence suggests that future increases in atmospheric CO2 will affect calcification in several major groups of marine organisms Kleypas (2002) (Fig. 4-7). This is a more direct, effect of fossil fuel emissions and is based on two relationships: 1) an increase in total CO2 in seawater results in a reduction in carbonate ion concentration; and 2) many independent experiments show a strong positive relationship between carbonate ion concentration and calcification rate in numerous species of calcifying algae, scleractinian corals, and coccolithophores.

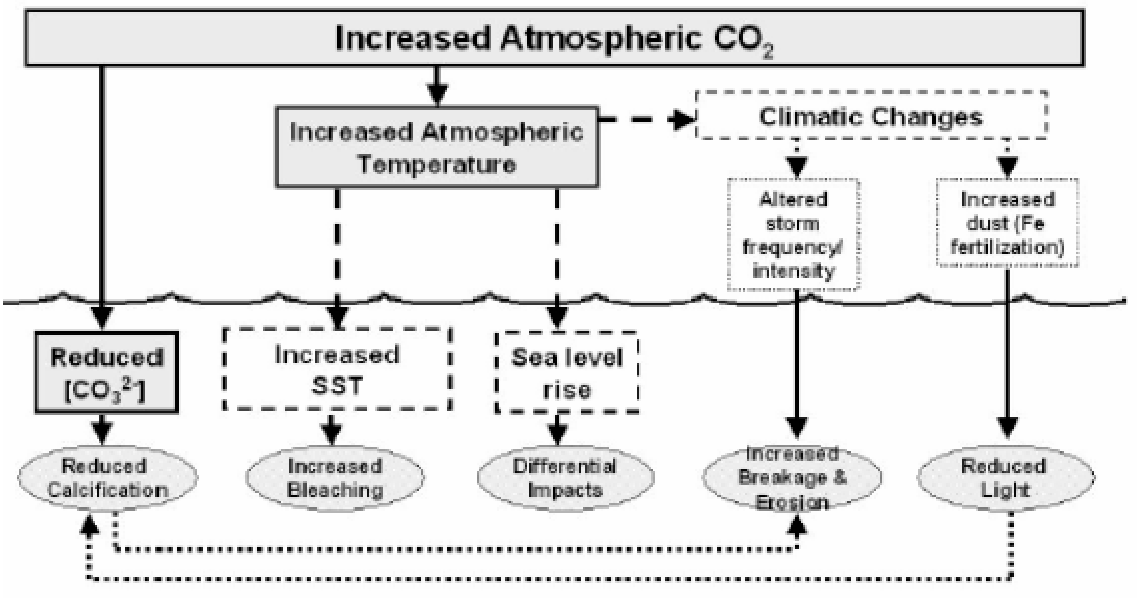

Fig. 4.7: Direct versus indirect impacts of increased atmospheric CO2 on coral reefs. Most direct effects are shown as solid lines; least direct as dotted. Direct impacts are not necessarily worse than indirect impacts, but are more predictable (Kleypas and Langdon, 2002).

Seawater chemistry of the surface ocean (ocean mixed layer) will change in direct thermodynamic response to increased atmospheric pCO2, and the magnitude of this response can be predicted with a high degree of confidence (Guinotte, 2003). It is known that the carbonate ion concentration is equal to the difference between total alkalinity and total CO2 (Kleypas et al., 2001). Therefore, as alkalinity decreases (such as through removal of Ca2+ during calcification) or as total CO_2 increases (such as through increased CO_2 from the atmosphere, and/or respiration), CO_3^{2-} decreases. Reef-builders require both Ca^{2+} and CO_3^{2-} ions for calcification. $[Ca^{2+}]$ in seawater is much higher than $[CO_3^{2-}]$ and is not considered limiting to calcification except under unusual circumstances, but [CO32-] is considered limiting to calcification. Even if fossil fuel emissions were stabilized today, atmospheric CO2 would still reach double pre-industrial levels by the end of this century, and best estimates indicate that it will double by the year 2065, affecting the aragonite saturation state used by the corals and other calcifying organisms (Figs. 4-5, Kleypas et al., 2001; Kleypas and Langdon, 2002; Guinotte, 2003). This doubling of CO2 will invoke an approximately 30% decrease in [CO32-]

Table 4.4*: Comparison of seawater carbonate equilibrium concentrations under the pre-industrial atmospheric CO2 level (1 × CO2) and its doubling (2 × CO2). Values were determined assuming a temperature = 25°C, salinity = 35 PSU, and alkalinity=2300 µequiv kg-1 (Kleypas and Langdon, 2002).*

Parameter	**Concentration at 1 x CO_2**	**Concentration at 2 x CO_2**
pCO_2 (µatm)	280	560
$H_2CO_3 + CO_2$ (µmol kg^{-1})	8	16
HCO_3^- (µmol kg^{-1})	1635	1867
CO_3^{2-} (µmol kg^{-1})	272	177
Total CO_2 (µmol kg^{-1})	1915	2061
pH	8.17	7.93

Environmental records of past atmospheric CO2 levels and ocean pH imply that projections of future atmospheric CO2 levels are higher than any levels of the past 24 million years. Reef-building corals and coralline algae will therefore be exposed to a very different ocean chemistry than has existed for millennia, and strong reductions in calcification rates are likely (Kleypas et al., 2001; Kleypas and Langdon, 2002).

Dimethyl sulphide (DMS) is the most abundant volatile sulphur compound at the surface and has a strong marine phytplanktonic origin. Once in the atmosphere, DMS is a significant source of cloud condensation nuclei in the unpolluted marine atmosphere. The radiative impact resulting from changes in DMS air-sea fluxes caused by global warming Bopp (2003; 2004)is a small increase {3%) in the global DMS flux to the atmosphere at 2xCO2, but with large spatial heterogeneities (from -15% to 30%). The radiative perturbation resulting from the DMS-induced change in cloud albedo is estimated to be -0.05 W·m–2 (Watt per m2), which represents only a small negative climate feedback on global warming. However, there are large regional changes, such as a perturbation of up to -1.5 W·m-2 in summer between 40°S and 50°S, that can impact the regional climate. The radiative impact of changes in DMS emissions at 2 × CO2 reaches 1 W·m-2 in the western Equatorial Pacific Ocean and off the west coasts of Angola and Chile, whereas in the Southern Ocean between 40°S and 60°S and in the eastern Equatorial Pacific Ocean the radiative impact of changes in DMS emissions reaches –1 W·m-2. In the Southern Ocean, the radiative impact resulting from changes in the DMS cycle may partly alleviate the radiative forcing resulting from anthropogenic CO2.

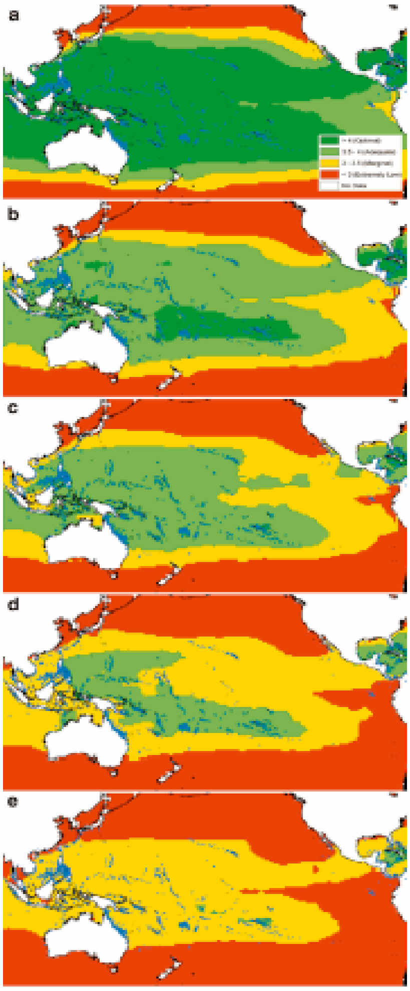

Fig. 4.8: *Aragonite saturation state.* ***a*** *Calculated preindustrial (1870) values; pCO2=280 ppmv.* ***b*** *Projected values, 2000-2009; pCO2=375 ppmv.* ***c*** *Projected values, 2020–2029; pCO2=415 ppmv.* ***d*** *Projected values, 2040–2049; pCO2= 465 ppmv.* ***e*** *Projected values, 2060–2069; pCO2=517 ppmv (Guinotte, 2003).*

The northeast Atlantic is projected to have increases in both winter and fall seasonal means and extremes of significant wave heights (SWH) in the twenty first century. These increases are generally accompanied by decreases in the mid-latitudes of North Atlantic and increases in the southwest North Atlantic. The nonstationary generalized extreme value analysis shows that global warming may result in changes in the spatial distribution of wave height extremes, eventually leading to changes in the size and frequency of extreme wave height events. For example, in the Norwegian Sea, an extreme wave height event that occurs on average once every 20-yr period in fall in the present-day (1990s) climate is expected to occur on average once every 4–12 yr in the climate projected for year 2080 under the A2 forcing scenario. Such significant changes will have an impact on the life span of marine and coastal infrastructure in the area. The possible changes in future wave extremes should be taken into account in the design, planning, and operation of coastal and offshore industries (Wang et al., 2004b).

It is possible that anthropogenic forcing may affect the ocean wave climate by changing the occupation statistics of atmospheric circulation regimes. Using the CGCM1 simulations of the

Canadian Centre for Climate Modelling and Analysis, Monahan (2000) concluded that under global warming, the episodic splitflow regime [which resembles the extreme negative phase of the North Atlantic Oscillation (NAO) in SLP] occurs less frequently while the standing oscillation regime (which resembles the Arctic Oscillation) occurs more frequently. In other words, global warming is associated with more frequent occurrence of the positive phase of NAO on the one hand, and with increases of wave height in the northeast Atlantic on the other hand. The implication here is that the projected wave height increases in the northeast Atlantic are associated with the anthropogenic changes that affect the NAO. Such a relationship between the NAO and wave height makes sense physically and is well supported by observational evidence. The significant increases in winter wave height observed in the northeast Atlantic in 1958–97 were found to be closely related to an ‘‘enhanced’’ positive phase of NAO (Wang and Swail, 2001; Wang and Swail, 2002).

For the northeast Atlantic, these scenarios are consistent with the double CO2 scenario of the STOWASUS-2100 (Kaas and STOWASUS Group, 2001), in which the mean SWH in this region was projected to have 5–35-cm increases in the climate of 2060–89 relative to that of 1970–99 in the cold seasons (September–February, Kaas and STOWASUS Group, 2001). Correspondingly, our scenarios projected 5–35-cm (5–20 cm) increases in the climate of 2070–99 relative to the 1961–90 climate for winter (fall) mean SWH (not shown).

Ecosystems are highly dynamic on a range of temporal scales from seconds to millennia. These dynamics are in response to changes in a multiplicity of forcing factors, which occur at a similar multitude of temporal scales. Clark (2001) have identified five major changes in North Sea zooplankton:

- a long-term decline in North Sea zooplankton from 1955 followed by a marked recovery after 1980 {from CPR survey data)
- change in the taxonomic composition of the zooplankton community between 1979 and 1980 observed in the Dove zooplankton time series, and the CPR time series for the central-east North Sea region
- change in the seasonal pattern and peak productivity of phytoplankton after 1979 in the central west North Sea region
- change from a phytoplankton community dominated by diatoms to one dominated by flagellates in the German Bight area
- long-term decline in Calanus finmarchicus and a long-term increase in C. helgolandicus over the 1962 to 1992 period.

The long-term dynamics of the North Sea plankton community in the majority of the northern and central areas of the North Sea appear to be controlled by climatic and meteorological factors. These act primarily through the influence of temperature and wind on thermocline formation and, therefore, influence the timing of the spring bloom (Clark and Frid, 2001).

In the case of the North Sea benthos, Clark (2001) has identified three major changes in the:

- biomass and abundance was higher during the 1980s compared with the 1970s in both littoral (Balgzand) and sublittoral (Northumberland, Skaggerak) stations.
- changes in abundance off Northumberland (M1 and P) occurred between 1980 and 1981, coinciding with a noticeable shift in community structure at one of the stations. At the 100-m-deep Skaggerak station, these changes were observed to occur a year earlier, between 1979 and 1980, while at Balgzand, the change in abundance and biomass that also occurred between 1979 and 1980 was accompanied by a shift from larger- to smaller-sized individuals.

- between the 1920s and the 1980s, three out of five communities in the central and southern North Sea showed a definite change, whilst between the 1950s and 1980s, the Dogger Bank benthos showed a decline in long-lived taxa, although total biomass had increased, mainly because of an increase in opportunistic species.

Although climate change is by definition a global issue, local conservation efforts can greatly help in maintaining and enhancing resilience and in limiting the longer-term damage from bleaching and related human impacts (Hughes et al., 2003).

The coral reef death of the Mentawai Islands (Sumatra, Indonesia) during the 1997 has been linked not only to increase of temperature, but also to the 1997 Indonesian wildfires. Widespread tropical wildfire is a recent phenomenon, the magnitude and frequency of which are increasing as population rises and terrestrial biomass continues to be disrupted. Where background nutrient supplies in reef waters are elevated or human activities have reduced upper trophic levels, reefs are likely to become increasingly susceptible to large algal blooms triggered by episodic nutrient enrichment from wildfires. Therefore, in addition to their impact on forest ecology and human health, tropical wildfires may pose a new threat to coastal marine ecosystems that could escalate into the 21st century (Abram et al., 2003).

Discharge of bilge and ballast water from ships releases a toxic mix of oil, nutrients, exotic marine species, and other pollutants (Burke and Maidens, 2004).

Competition between corals and algae is an important process on coral reefs, especially during reef degradation, when abundant corals are often overgrown by benthic macroalgae. Jompa (2003a; 2003b) suggest that the filamentous red algae, Anotrichium tenue and Corallophila huysmansii, are direct cause of coral tissue death rather than simply a symptom or consequence of previous tissue damage or coral stress.

4.4.10 Population systems

Population systems allow to link physical, and chemical aspects from the perspective of a single species. In particular vagile animal species may hereby connect several different habitats as e.g. migratory birds connecting the tundra, wetlands in the temperate zone, mediterranean ecosystems, savannas, subtropical and finally tropical ecosystems. An integrated assessment of the impacts of climate change requires therefore to use a view orthogonal to that of the usual biomes.

An illustrative example showing how various regions (ice, sea, land are linked to the life of a particular species) are connected via a species is that of the polar bear (Box 4-5).

Box 4.5: Polar bears – a species in peril?

Polar bears (*Ursus maritimus* PHIPPS) live throughout the ice-covered waters of the circumpolar Arctic. Although some occur in the permanent multi-year pack ice of the central Arctic basin, they are more often in the annual ice over the continental shelf and inter-island archipelagos that surround the polar basin. There are an estimated 20,000 to 25,000 polar bears worldwide (IUCN/SSC Polar Bear Specialist Group, in prep.). They are specialized predators of ice-breeding seals and are therefore dependent on sea ice for survival. Polar bears that have continuous access to sea ice are able to hunt throughout the year. However, in areas where the sea ice melts completely each summer, polar bears are forced to spend several months on land fasting on stored fat reserves until freeze-up.

Both anthropogenic and natural changes in the Arctic marine ecosystem threaten the welfare of polar bears. Many persistent organic pollutants are lipophilic and may bioaccumulate through the food chain to reach high levels in polar bears (Norstrom *et al.*, 1998). Recent studies of polar bears suggest that these contaminants not only affect the endocrine and immune systems (Bernhoft *et al.*, 2000; Skaare *et al.*, 2001; Lie *et al.*, 2004; Lie *et al.*, 2005) but may also affect subsequent reproductive success (Derocher *et al.*, 2003).

Perhaps the greatest challenge facing polar bears are the effects of climatic warming (Stirling and Derocher, 1993; Stirling *et al.*, 1999; Derocher *et al.*, 2004), particularly when the projections of various climate and ice models that suggest greatly diminished sea ice cover and extended open-water seasons are considered. Breakup of the sea ice on western Hudson Bay, Canada is already about 3 weeks earlier than it was in the early 1970s, resulting in polar bears in this area coming ashore earlier with reduced fat reserves (a 15% decline in body condition), fasting for longer periods of time, and having reduced productivity (Stirling *et al.*, 1999).

Preliminary estimates suggest that from the mid 1990s through the mid 2000s the Western Hudson Bay population has declined from 1200 to 1000 individuals. Although these changes are specific to one subpopulation, we might reasonably expect similar impacts on other subpopulations of polar bears. In 2005, the IUCN Polar Bear Specialist Group concluded that the IUCN Red List classification of the polar bear should be upgraded from Least Concern to Vulnerable based on the likelihood of an overall decline in the size of the total population of more than 30% within the next 35 to 50 years. The principal cause of this decline is climatic warming and its consequent negative affects on the sea ice habitat of polar bears. Similar consequences may face other ice-dependent species, not only in the Arctic but also in the Antarctic (Barbraud and Weimerskirch, 2001; Croxall *et al.*, 2002).

The results from further studies assessing climate change impacts on single species, albeit not necessarily studying entire population systems, are summarised in the subsequent section.

4.4.11 Global synthesis including impacts on biodiversity

[Summarises key results from the above sections]
Current trends will continue, with or without climate change. With respect to implications for biodiversity this is expected to lead to an overall loss in biodiversity (Millennium Ecosystem Assessment, 2005b Volume 1, p29-34).

The ATEAM project represents an attempt to assess climate change impacts in an integrated effort. It focused on a number of goods and services from its study region (Europe) as a whole (e.g. soil organic carbon, carbon sequestration, wood production, carbon storage, water resources, biodiversity changes) while the ACCELERATES project focused on biodiversity changes in agricultural areas only. Three studies examined projected changes at local and/or regional scales (Holman *et al.*, 2004b; de Chazal *et al.*, 2007; Rounsevell *et al.*, 2007a).

The ATEAM project developed a set of European land use scenarios for the seven coupled GCM and SRES scenario combinations, based on an interpretation of the SRES storylines (Ewert *et al.*, 2005; Rounsevell *et al.*, 2005). Broad projected land use changes has already been described above for Europe, largely representing the output of this work. The general trends was reductions in agricultural areas for food production, compensated somewhat by increases in areas for bio-energy production and forests, and small increases in urban and nature conservation areas. Decline is agricultural land in the A1F1 and A2 scenarios was particularly pronounced (Rounsevell *et al.*, 2005). VISTA downscaled these ATEAM scenarios for application to their study regions using qualitative interpretations and statistical methods (de Chazal *et al.*, 2007).

The ACCELERATES replicated its European scale focus in six case study regions: Belgium, Denmark, the Belluno Valley, Italy, Almeria, Spain, Lesvos, Greece, and East Anglia, UK. VISTA also took an agro-pastoral focus, however examined a range of ecosystem services (e.g. sheep grazing, hunting, agro-pastoral scenic value, sports and leisure opportunities, observing wildflowers, hiking) in five marginal agricultural regions: France, Germany, Portugal, Sweden and Greece. REGIS examined projected changes across four sectors (coasts, agriculture, water resources, biodiversity) in two areas in England: East Anglia and the North-West.

ACCELERATES developed their own agricultural land use change scenarios, based on the SRES storylines (Abildtrup *et al.*, 2007; Audsley *et al.*, 2007), and used them in combination with the ATEAM work for projected changes in other land uses. Loss of agricultural land was particularly pronounced in the agriculturally marginal areas (Audsley *et al.*, 2007). Overall, results were generally consistent with the ATEAM work (Rounsevell *et al.*, 2007a). REGIS derived a set of parameters based on their qualitative storylines for input into the range of sectoral models developed (coastal and river flooding, agriculture and hydrological), including changes in land use (Holman *et al.*, 2004a).

In attempts to quantify risks of biodiversity loss, SDMs were used to make assessments for many systems, including rare or endangered species.

For instance the ATEAM project used the BIOMOD niche-based modeling technique (Thuiller, 2003) to project the distributions of more than 2000 species across Europe (1350 plants, 157 mammals, 108 reptiles, 383 breeding birds) for the selected scenarios and climate models. Projections were derived under two extreme cases of dispersal, representing no migration versus universal migration. Land use change was not included explicitly as a variable, however (Thuiller *et al.*, 2004b) reports that the relatively coarse resolution of the data together with the magnitude of projected climate change would likely make the effect of land use of secondary importance for the analyses performed. ACCELERATES used the SPECIES model, an artificial neural network technique, (Pearson *et al.*, 2002) to project the distributions of 47 species associated with 10 habitats, representing a range of dominant through to rare or threatened species. Land use change was also not explicitly included as a variable in these European wide analyses. For the regional studies that included land use change, the ATEAM land use change scenarios (Ewert *et al.*, 2005; Rounsevell *et al.*, 2005) were used in conjunction with the SPECIES model, a dispersal model (Pearson and Dawson, 2005) and the ALCOR connectivity model (del Barrio, 1998).

At a European scale, all scenarios result in considerable change. ATEAM results suggest, that in terms of potential loss in species range, under the no migration hypothesis, more than 50% of the species have a projected range loss of >30% (equivalent to the ‘vulnerable’ category under the IUCN threat category), or a projected range loss of 100% in next 50 to 80 years (IUCN ‘extinct’ category) (Thuiller *et al.*, 2005). The A1F1-HadCM3 showed the greatest changes, with 22% of the species having a projected range loss of >80% (critically endangered IUCN category) and 2% extinct by 2080. Under the universal migration assumption, range changes are, not surprisingly, less severe. For all scenarios, 67% of species showing ?< 30% range loss ('low risk' IUCN category), and under the B1 scenario, 76% of the species would be at low risk (Thuiller *et al.*, 2005). For estimates of species loss and turnover, there is great variation across scenarios. The A1-HadCM3 scenario represents the greatest losses, leads to a mean species loss of 42%, and a turnover of 63%. This scenario represented the widest range of variability across Europe for both species loss (2.5–86%) and turnover (22–90%). The percentage of species loss exceeded 80% in some regions, such as north central Spain, and the Cevennes and Massif Central in France. B1-HadCM3 projected the lowest mean percentage of species loss (27%). Other scenarios show intermediate mean rates of species loss (~30%) and turnover (~50%) (Thuiller *et al.*, 2005).

This trend of greater loss of species in the Southern Europe is more obvious when the ATEAM results are grouped by biogeographical regions. Major patterns of dispersal were found to be similar over all scenarios, so trends are reported here based on the A1F1-HadCM3 scenario, representing the maximum change. Using the classification of (Metzger *et al.*, 2004), the northern Mediterranean (52%), Lusitanian (60%) and Mediterranean mountain (62%) regions were the regions showing the greatest species loss; with the Boreal (29%), northern Alpine (25%), and Atlantic (31%) regions showing the least losses. When species turnover was examined, a different pattern emerges, the greatest species turnover occurring at the transition between the Mediterranean and continental regions, with overall losses of Euro-Siberian species and increases in Mediterranean or Atlantic species. The Lusitanian region and the Mediterranean Mountains show the highest species turnover of 71%, with the Pannonian and Mediterranean South region showing a turnover of ~66%. Thus, these regions may lose a substantial part of their plant species diversity, and experience a major change in floristic composition. Southern Fennoscandia also represents an area of high potential species turnover with possible losses of boreal species and gain of Euro-Siberian species.

The REGIS project, projected distributions of 28 species within four habitats for the two regions, also using the SPECIES model (Pearson *et al.*, 2002). Outputs of the various sector models (coastal and river floods, agriculture and hydrological) were then used to examine changes in these projections in terms of changes in coastal habitats, changes in agricultural-land use and management, and changes in water quality and quantity.

Extrapolating the global population trends using the Living Planet Index for the years 1970-2000 we get estimates for the following drops of diversity per decade of approx. 12.3% globally, 0.5% in forests and woodlands, 11.6% in marine ecosystems, and 18.3% in freshwater habitats. This type of index suggests that tropical and southern temperate regions experience biodiversity losses at the fastest rate. Northern temperate regions are more stable, or are in a slower decline.

Similar efforts were made at the global scale. Here we make an attempt to summarize all those findings and to combine them with many other approaches. All together allow to assess the biodiversity implications in function of a quantitatively specified climate change relative to preindustrial climate (Table 4-5, Fig 4-9).

Table 4-5*: Climate change impacts on ecosystems in function of global mean temperature increases relative to preindustrial climate (compare Table 4-5, derived from Warren, 2005)*

Change of 0.7° - 1.0° C (i.e., 0.1° – 0.4° C additional warming)	
Global	Increased ecosystem disturbance by pests and disease, especially in Boreal forests, Australia and California [84] (Gitay *et al.*, 2001; Hare, 2003)
Europe	Increased overwinter survival in resident and wintering birds and northward expansion in ranges of butterflies [81,82] (Millennium Ecosystem Assessment, 2005b).
Africa	Increase in drought in Sahel could lead to loss of local flora and fauna [83] {ECF, 2004, unknown}. Reduction in extent of Karoo, the richest floral area in the world and declines in range sizes for some animal species in South Africa [87] (Rutherford *et al.*, 1999).
Australia	Coral reefs at high risk [85] (Hoegh-Guldberg, 1999a). Reduction in extent of Queensland's World Heritage Rainforest with loss of habitat and range declines in the Golden Bowerbird [86,89] (Hilbert *et al.*, 2001; Hilbert *et al.*, 2003). Risk of extinction of vulnerable species in SW Australia Dryandra forest [88] (Pouliquen-Young and Newman, 1999).
Small Island States	Coral reefs at high risk in Caribbean and Indian Ocean [85] (Hoegh-Guldberg, 1999a).
Change of 1.0° – 1.5° C	
Global	82% of coral reefs bleached [90] (Hoegh-Guldberg, 1999a). 10% of global ecosystems are transformed losing between 2 and 47% of their extents; only 53% of the wooded tundra remains stable [91] (Leemans and Eickhout, 2003).
Australia	50% loss of highland rainforest with range losses of endemic species and some risk of extinction of Golden Bowerbird [92] (Hilbert *et al.*, 2001; Hilbert *et al.*, 2003; Williams *et al.*, 2003). Greater than 50% loss of Kakadu wetlands [94] (Hare, 2005).
Small Island States	Potential extinction of coral reefs in the Indian Ocean [93] (Sheppard, 2003).
Change of 1.0° – 2.0° C	
Global	Risks for many ecosystems [95] (Leemans and Eickhout, 2003).
Australia	Many eucalypts at risk from range shifts [96] (Hughes *et al.*, 1996). Significant loss of alpine zone [98] (Busby, 1988). Extinction risks for frogs and mammals in Queensland rainforest [100] (Williams *et al.*, 2003).
North America	Large impacts to salmonid fishes [97] {, impacts on ecosystems} (Keleher and Rahel, 1996).
Africa	Severe loss of extent of Karoo [99] (Rutherford *et al.*, 1999).
Antarctic	Threats to key mollusk species [101] (Peck *et al.*, 2004).
Arctic	Severe damage to Arctic ecosystems [103] (Hassol, 2004). 60% loss of lemmings with concomitant ecosystem impacts on predators [104] (Kerr and Packer, 1998).
Europe	Moderate stress to alpine zone [102] (Fischlin and Gyalistras, 1997).
Change of 2.0° – 2.5° C	
Global	Coral reefs 97% bleached [105] (Hoegh-Guldberg, 1999a). 16% of global ecosystems transformed losing between 5% and 66% of extent [107] (Leemans and Eickhout, 2003). Further ecosystem disturbance by fire and pests [108] (Gitay *et al.*, 2002).
Arctic	Total loss of Arctic summer loss leading to ecosystem stress and a high risk of

	extinction to polar bears, walrus and some seals [106] (Hassol, 2004). Up to a 50% loss of breeding area for tundra breeding shorebirds and geese [111,112] {Folkestad, 2005, unknown}.
Asia	50% loss of the Sundarban wetlands in Bangladesh [109] (Qureshi and Hobbie, 1994; Smith *et al.*, 1998; Hare, 2005). Up to a 50% loss of boreal forest in China [114] (Ni, 2001; Hare, 2003). Cloud forests lose hundreds of meters of elevational extent in Indonesia region [118] (Still *et al.*, 1999).
North America	Greater than 50% loss of Salmonid fish habitat [115] (Hare, 2005). Local extinctions of mammals from mountains in the Great Basin of the United States [120] {Hannah, 2002, unknown}. Maple fall colors lost in the United States [133] {ECF, 2004, unknown}.
Central America	Cloud forests lose hundreds of meters of elevational extent [118] (Still *et al.*, 1999). Loss of Monarch butterfly wintering habitat in Mexico [121] (Villers-Ruiz and Trejo-Vazquez, 1998). Large range losses and risk of extinctions in animals in Mexico [123] (Peterson *et al.*, 2002a).
Small Island States	Extinction of Hawaiian Island endemic bird species [119] (Benning *et al.*, 2002).
Europe	Transformation of ecosystems with 32% of the plants moved from 44% of Northern Europe with potential extinctions [116] {ECF, 2004, unknown} (Bakkenes *et al.*, 2002).
Africa	Cloud forests lose hundreds of meters of elevational extent [118] (Still *et al.*, 1999). Karoo region reduced to 20% of original area placing 2800 species of plants at risk of extinction [124] {Rutherford, 1999, South African Country; Hanah, 2002, unknown}. Extensive losses to range sizes in protected areas of South Africa [125] (Erasmus *et al.*, 2002) {, impacts on ecosystems}. Wetland ecosystems disappear and large declines in fish populations in Malawi and the African Great Lakes [126] {ECF, 2004, unknown}. Extinctions of 10% of the endemics in Fynbos with a loss of 51%-65% of extent of coverage [131] (Midgley *et al.*, 2002).
Australia	Inflection point at which extinction risk increases for forest mammals in Queensland rainforest [117] (Williams *et al.*, 2003). High risk of extinction of Golden Bowerbird [122] {Hilbert, 2004, unknown}. Complete loss of alpine zone [132] (Pouliquen-Young and Newman, 1999; Hare, 2005).
Change of 2.0° – 3.0° C	
Australia	Total loss of Kakadu wetlands [128] (Hare, 2005). Extinctions of alpine flora in New Zealand [129] (Halloy and Mark, 2003).
South America	Collapse of the Amazon ecosystem [127] (Cox *et al.*, 2004).
Asia	Permafrost shifts north by 1° – 2° of latitude with melting of glaciers and desertification on the Tibetan Plateau [130] (Ni, 2000).
North America	Average loss of 44% of migratory shorebird habitat at 4 major sites in the United States [134] (Galbraith *et al.*, 2002; Hare, 2005).
Change of 3.0° – 4.0° C – [Note: These impacts are not on the map]	
Global	Few ecosystems can adapt to temperature increases above 3° C and 50% of all nature reserves can no longer fulfill their objectives; 22% of global ecosystems transformed with losses in extent of 7%-74% (Leemans and Eickhout, 2003). 22% loss of coastal wetlands (Nicholls *et al.*, 1999). 60% loss of tundra and 44% loss of taiga ecosystems {Neilson, 1997, unknown}.
Europe	Severe fire and pest problems with habitat conversion in the Mediterranean (Mouillot *et al.*, 2002). Risk of loss of up to 60% of the species, especially in

	southern Europe {ECF, 2004, unknown}. Large losses of migratory bird habitat in the Baltic region and the Mediterranean (Nicholls *et al.*, 1999; Najjar *et al.*, 2000). Alpine species near extinction (Bugmann, 1997; Fischlin and Gyalistras, 1997). 38% of European alpine species lose 90% of their ranges {, impacts on ecosystems}.
Small Island States	Risk of extinction of 90% of the Hawaiian Honeycreepers (Benning *et al.*, 2002).
Asia	Complete loss of boreal forest ecosystem in China (Ni, 2001).
North America	50% loss of duck nesting habitat in the prairie pothole region of U.S. and Canada (Sorenson *et al.*, 1998). 77% loss of low tundra in Canada {Neilson, 1997, unknown}.
Australia	50% loss of eucalypt species (Hughes *et al.*, 1996). 50% range loss of endemic butterflies {Beumont and Hughes, 2002, unknown}.

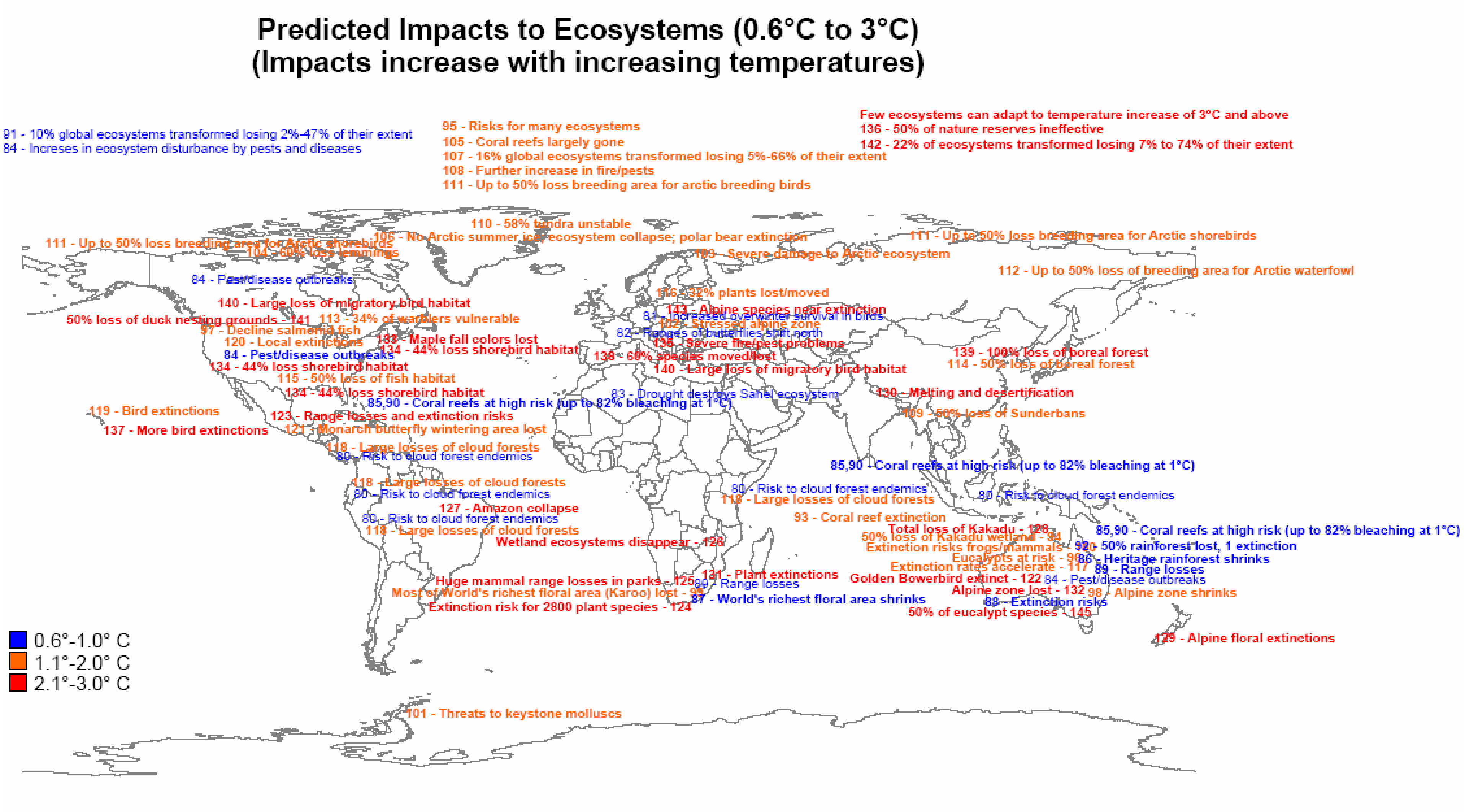

***Fig 4.9:** Climate change impacts on ecosystems in function of global mean temperature increases relative to preindustrial climate (compare Table 4-5).*

Global maps on hotspots can be used to point at areas where ‘reason of concern’ for impacts of climate on ecosystems are particularly large (Fig. 4.10)

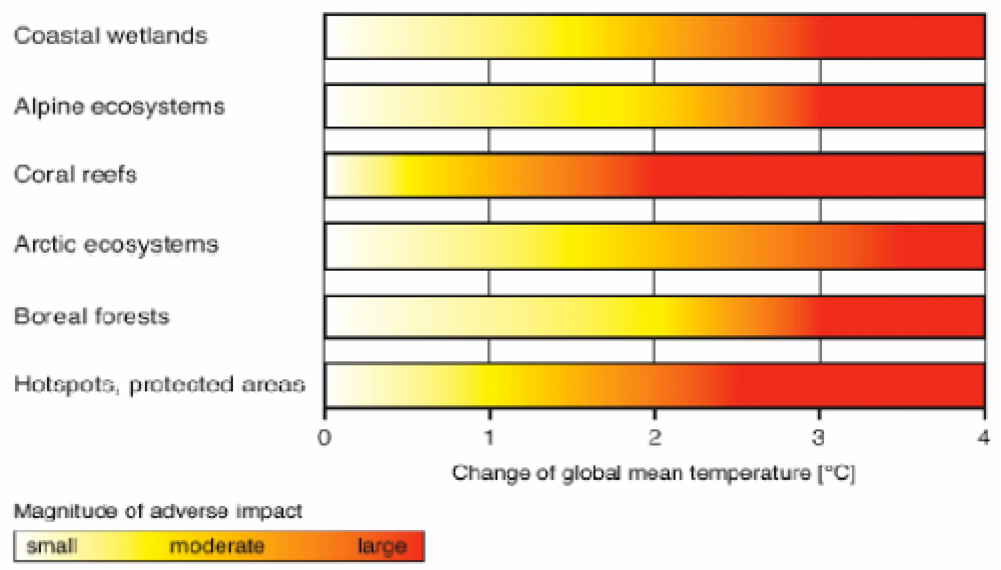

Fig 4.10: *Magnitude of adverse impacts on ecosystems in function of global mean temperature increases relative to preindustrial climate.*

4.5 Costs and other socio-economic aspects

There is a great deal of controversy as to whether ecosystems should be valued. Regardless of the controversy, there are now results available from attempts to value ecosystems, the basis for valuing them as affected by climate change.

4.5.1 Ecosystems services

For the first time at the global scale Costanza *et al*. (1997) estimated the global mean economic value of ecosystems (7 services,16 biomes) to amount to 33 T US$/a, while the current global GDP totals to 18 T US$/a *[needs checking whether this is still the most recent figure, will also need to be elaborated more by adding more recent estimates]*.

Such valuations are questioned by these arguments: (i) ecosystem services are only to a small fraction on the market, thus no reliable market prices are available; (ii) questioning values depends more on the knowledge of the interviewed than the true value; (iii) where no human substitute is available, which is the case for several key services (ecosystems form basis of all life) no valuation is possible, since service is basically invaluable; (iv) ecosystems may respond non-linearly, which is not properly reflected by a linearly measuring monetary estimate; *[Needs to be completed]*

Such valuations are justified by these arguments: whether a given service meets a specific goal or objective can be expressed (i) monetarily (Costanza, 2001 values, valuation) or (ii) as a risk function (insurance approach); (iii) human choices affecting ecosystems can be valued by interviewing involved people; (iv) where no human made substitute is available, partial substitutions can be used to asses fractions and then extrapolate; *[Needs to be completed]*

[Reminder: There are many ecosystem services and goods which have to be considered here, e.g.:
Supporting: Cost estimates of changes in supporting services due to climate change
Provisioning: Cost estimates of changes in provisioning services due to climate change
Regulating: Cost estimates of changes in regulating services due to climate change
Cultural services (e.g. use of ecosystems (landscapes, parks, reserves etc.)): Cost estimates of changes in recreational uses (aestehetic, recreation, education, spiritual) due to climate change]

Ecosystems services range from provisioning services such as in food and water, regulating for instance climate regulation, cultural as in cultural heritage and supporting services such as in nutrient cycling and primary production (chapter 2Millennium Ecosystem Assessment, 2005a). Literature shows that ecosystems services are generally poorly understood relative to other forms of capital, and often their role is widely appreciated only upon their loss (Daily *et al.*, 2000). Because ecosystems have no standard market value the practice for cost-benefit analysis is to assume that "nature" is either constant or irrelevant such that ecological services such as pest control or waste recycling are ignored (Schneider, 2003). Another key problem is the relative weight put on current versus future costs and benefits of ecosystems. Partly due to this, collateral benefits from biosphere GHG management (carbon credits as an economic instrument) that are public goods such as biodiversity preservation and flow regulation in watersheds are not fully considered in market-based models (Kennett, 2002).

Alternative approaches to this problem include those based on avoidance of costs but these have been noted to provide only partial, lower bound indications of value, especially for services without adequate substitute such as global climate regulation (Daily *et al.*, 2000). One way could be to assess the costs of climate change on ecosystems by evaluating the historic losses from extreme climatic events, such as floods, droughts, and hurricanes (Schneider, 2003). Others have suggested that the costs of climate change should be assessed in terms of the "Five Numeraires": Monetary loss, loss of life, quality of life (including coercion to migrate, loss of cultural heritage sites, etc.), species or biodiversity loss, and distribution/equity (change in income distributions) (Schneider, 2003). Policies need to be formulated to promote activities that further non-monetary values of ecosystems and establish mechanisms to ensure that non-market values and public goods receive appropriate attention in the valuation of ecosystems (chapter 2Millennium Ecosystem Assessment, 2005a).

However, some changes have been observed for example, an Australian US$25-million firm, Earth Sanctuaries, Ltd., was listed on the Australian Stock Exchange, making it the world's first conservation company to go public (Daily *et al.*, 2000). This trend calls for a need to begin, cataloging of the sources and consumers of ecosystem services, for example establishing for a given location a documentation of service flows occurring locally, across regions and globally. Nevertheless, there is need for caution for instance, ecosystems typically respond nonlinearly to perturbation, what is true in one region may not apply well elsewhere and land use adaptation to changes in ecosystems add another complexity (Daily *et al.*, 2000). For example, changes in vegetation due to desertification may not give rise to change in livestock production due to various human interventions for instance, a switch from cattle ranching to game farming or a change in the product, for example form meat production to wool for sheep production (Ash *et al.*, 2002). While for highly managed systems decoupling of climate and primary production from secondary production could be through supplementary feeding.

However, to attain sustainable development interventions are required. This is because maintaining a secondary production that is decoupled from carrying capacity minimizes feedback to managers on the role of their actions on primary production. This minimizes the incentive to reduce negative impacts on ecosystems although, on the long-term, primary production is certainly essential.

Coral reefs are used as an example to illustrate valuation approaches. Coral reefs represent a very high value for humankind, supporting millions of people whose lives depend on these natural resources for a source of food and income. Over geological time, future changes in atmospheric CO_2 can affect coral reefs at global scale in a number of ways (Fig. 4-7): by altering seawater chemistry, by greenhouse warming (temperature and sea-level fluctuations), and by other climatic changes (Kleypas *et al.*, 2001; Kleypas and Langdon, 2002; Wilkinson, 2004; Henderson, 2005; Thompson and Goldstein, 2005). Of these three, global changes in both seawater chemistry and temperature will probably have the most profound impacts on marine ecosystems within the next century (Clark and Frid, 2001; Kleypas et al., 2001; Helmuth et al., 2002; Kleypas and Langdon, 2002; Lehodey et al., 2003; Loukos et al., 2003).

Cesar (2003) estimates that coral reefs provide each year nearly US$ 30 billion in net benefits in goods and services to world economies, including, tourism, fisheries and coastal protection (Table 4-6); and the Hawaiian reefs produced an estimated total annual economic benefit of US$363 million (Cesar et al., 2002).

Table 4.6: *Potential net benefit streams per year and net present value (NPV) of the world's coral reefs (in billion US$) (Cesar et al., 2003).*

Good/service	**Amount**
Fisheries	5.7
Coastal protection	9.0
Tourism/recreation	9.6
Biodiversity value	5.5
Total	29.8
NPV (50 year; 3%)	797.4

Reefs provide a variety of goods and services, which create economic benefits to society. These economic benefits are often taken for granted, yet if these goods and services were taken away or destroyed, we would be forced to provide other methods to supply these benefits at significant costs. Table 4.7 illustrates the potential net benefit streams for the world in the order of US$ 30 billion per year if coral reefs were well managed and intact, based on new calculations estimated to be valued at US$ 9.0 billion per year (Cesar et al., 2003).

Table 4.7 *Potential net benefit streams per year and net present value (NPV) of coral reefs per region (in US$ million) (Cesar et al., 2003).*

	Southeast Asia	Caribbean (ex. USA)	Indian Ocean	Pacific (ex. USA)	Japan	USA	Australia	World
Reef area (km^2)	89,000	19,000	54,000	67,000	3,000	3,000	49,000	284,00
Fisheries	2,281	391	969	1,060	89	70	858	5,71
Coastal protection	5,047	720	1,595	579	268	172	629	9,00
Tourism/recreation	4,872	663	1,408	269	779	483	1,147	9,62
Biodiversity value	458	79	199	172	529	401	3,645	5,48
Total	12,658	1,853	4,171	2,079	1,665	1,126	6,278	29,83
NPV (at 3%)	338,348	49,527	111,484	55,584	44,500	30,097	167,819	797,35

4.5.2 Biodiversity and risk of species extinction

Species becoming rare or risking even extinction due to climate change are difficult to value, yet are of great concern to many. Given the ecosystem persists, internal changes may imply dramatic changes for many species. Some species may give way to others, which are able to provide albeit altered, yet still comparable services or services of similar value for human use. In some cases threatened species have key functions and the feedback to the ecosystem may be dramatic, e.g., general ecosystem productivity, health, and resilience, pollination, seed dispersal, natural pest control, and decomposition. Overall effect can be changes in the structure and function of the affected ecosystems (positive or negative due to reduced services) or even diminishing in functionality by increasing losses in local biodiversity (National Research Council, 1999). Extinction is well known not to be a linear process and depends also on thresholds. Finally, populations feedback onto the "encompassing" ecosystem: Once a biodiversity has fallen below a certain threshold, the entire ecosystem may suddenly and surprisingly collapse.

[Reminder: There are many other services and goods which have to be considered in addition to those already listed here:
Supporting (Pollinators, seed dispersal, decomposition, soil maintenance/formation): Cost estimates of changes in pollination, seed dispersal, decomposition, soil maintenance/formation due to climate change
Provisioning, Regulating (Control of Pest Species and Disease Vectors): Cost estimates of changes in pest regimes due to climate change
Cultural (Recreational use of wildlife, game, and plants): Cost estimates of changes in recreational uses of populations impacted by climate change]

4.5.3 Ecosystem goods

[Reminder:
Supporting (Productivity, Nutrient cycling,Soil formation): Cost estimates of changes in productivity, nutrient cycling, soil degradation/destabilisation etc. due to climate change
Provisioning, Regulating: Cost estimates of changes in provisioning (food, fresh water, wood and fiber, fuel) (reference to other chapters) regulating (climate, flood, disease, water/air purification)]

We use here examples from ocean ecosystems to illustrate this aspect.

The oceans are an important component of the climate system and ocean borne commerce is sensitive to the state of the ocean surface. Among other surface characteristics, ocean wave height could be affected by anthropogenic forcing of the climate system. Since the design of offshore oil platforms and other marine and coastal infrastructure is constrained by the largest wave height event anticipated during a fixed design period, increases in the extremes of wave height could have an impact on the life span of these installations that will be in excess of impacts anticipated from the rising sea level (Wang et al., 2004b).

Projections of ocean wave height are therefore useful for the design and operation of coastal and offshore industries. However, previous studies in this field are limited. The STOWASUS-2100 (Regional Storm, Wave, and Surge Scenarios for the 2100 Century) Group (Kaas and STOWASUS Group, 2001) carried out two 30-yr time-slice experiments for the northeast Atlantic: a control run for the period of 1970–99, and a double CO2 run for the period of 2060–89. The WASA (Waves and Storms in the North Atlantic) Group (1998) carried out a similar pair of 5-yr time-slice experiments for the North Atlantic as well as producing a statistical projection of future anomalies of intramonthly quantiles of wave height at Brent and near Ekofisk (in the northern and central North Sea, respectively). Given the intensity of industrial activity in parts of the North Atlantic, and the intensity of ship traffic throughout the basin, there is a need for additional projections of the future wave height climate that span the entire basin. The purpose of this study is to construct climate change scenarios of wave height in the North Atlantic, and to carry out a detailed assessment of changes in the projected wave heights. Sustained higher temperatures also may have positive effects Kennedy et al. (2002). Some commercially valuable estuarine-dependent species in the lower latitudes have higher growth rates and larger annual harvests when temperatures are higher. For example, shrimp harvest is generally highest in tropical climates and declines northward, so an increase in temperatures could increase the annual yield of shrimp in temperate waters like the Gulf of Mexico, as long as temperatures did not exceed lethal levels. In the mid-Atlantic region, where severe winters are thought to result in low blue crab catches, less severe winters may enhance harvests if other factors do not intervene.

Similarly, the commercially important ocean quahog clam lives near its southern limit in the mid-Atlantic region. An increase of 2°C would extend the period of thermal stress for ocean quahog larvae from the present 167 days to 231 days (May 14-December 31), lowering survival of the larvae and depressing their recruitment to the resident population, which could eventually die out over a period of decades Weinberg (2002).

Future global sea-level rise induced by climate change could inundate over 20,000 km2 (8,000 mi2) only in the United States, and this could cost cumulative impacts to coastal property of US$ 20 billion (assuming an economically efficient adaptation) to roughly US$ 150 billion (if vulnerable areas are inundated) Neumann (2000). The large difference between estimates suggests that there should be major efforts to encourage understanding strategies to more efficiently respond to sea-level rise. Major coastal cities such as New Orleans, Miami, New York, and Washington, DC, will have to upgrade flood defenses and drainage systems of risk adverse consequences.

If inundation outpaces accretion, marsh or mangrove forest will be submerged, and succumb. The continued loss of wetlands in Louisiana indicates that, under the present rate of sea-level rise of 2.3 mm per year and land subsidence of 4 mm per year, total marsh area will be critically reduced as humans alter water and sediment distributions (Kennedy et al., 2002).

4.6 Acclimation and adaptation: practices, options, and constraints

[Jeff wanted to develop text on adaptation, as he felt that the ZOD section on this was well reviewed]
The term adaptation has multiple meanings: Among biologists it refers to how a species evolves including how a species might evolve to climate changes. Other biologists have used the term adaptation to refer to ways in which a species might respond to a changing climate (e.g., by moving poleward). In this section we will use the term adaptation to discuss ways that human's can adapt management practices to help ecosystems respond to climate change.
Links are made here to, and impacts on ecosystems, ecosystem services and biodiversity

4.6.1 Background

The UN Convention on Biological Diversity (CBD) calls for the Ecosystem Approach as a robust way to mainstream biodiversity into other policy areas. The ecosystem approach resembles integrated forestry management and coastal management. The ecosystem approach provides a framework for finding a balance among different needs, for example, through integrated natural resource management systems and through various policy, legal, institutional and economic measures. Case studies evaluating implementation of the ecosystem approach are limited. The few existing case studies (CBD, 2003) have suggested a need to "dispel the myth that 'win-win' situations between development and conservation objectives were widely achievable, and concentrate instead on understanding how trade-offs and equitable compromises could be attained". Although the ecosystems approach does not explicitly aim to reduce climate-change related vulnerabilities but the adopted adaptive management strategies could provide early warnings and therefore more timely and adequate responses.

4.6.2 Natural adaptation

4.6.2.1 In place adaptations:
Ecophysiological, population, midterm and longterm structural responses,
Genetic adaptation (micro-, and macroevolution)

4.6.2.2 Spatial adaptation
Shifting biome borders and compositional changes
Migration

4.6.3 Adaptation options to climate change

4.6.3.1 Adaptation options to reduce the negative impacts of climate change on biodiversity
[Following text needs to be revised]

"Adaptation refers to the actions that people take in response to, or in anticipation of projected or actual changes in climate, to reduce adverse impacts or take advantage of the opportunities posed by climate change" (Tompkins and Adger, 2003). As the climate changes, it will become increasingly important to adapt natural resource management techniques in order to help ecosystems build resilience to better allow them to respond naturally.

To plan and respond effectively, managers must first understand the nature of the climatic and ecological changes that are likely to occur in their region. Monitoring climate change and how ecosystems respond is important so that adjustments in management strategies can be made as needed. Numerous adaptations and combinations of approaches will emerge as experienced wildlife managers gain an understanding of the changes that are likely to occur in habitats and species in a given biome. Unfortunately, uncertainty as to the magnitude and timing of climate change may discourage some from adapting management practices. However, moving towards what various authors have termed 'no regrets' management or 'precautionary principle' management would certainly be prudent. This approach would encourage conservation planning that is both relevant today and which models, or first principles, suggests will still be relevant in the future. The following section discusses some of the potential adaptations that might be considered in order to minimize the potential impacts of climate change. This is a rapidly developing field, especially in the area of preserve design, so this list should not be considered exhaustive.

Adaptation starts with recognition that climate change is occurring. Conservation planners and managers need to become better informed about the potential consequences of climate change and variability on the resources they work with. Developing techniques that allow conservation resources to be adequately managed in the face of climate variability may ultimately prove the most beneficial way to be prepared for abrupt climate change. For example, the ability to properly manage natural resources during El Nino events potentially acting as a surrogate for the potential changes that could occur with an abrupt climate change.

Managers also need to be better prepared for climate change surprises. These surprises could come about in terms of climate or in terms of wildlife or habitat responses. For example, one type of surprise may be repeated "100-year" floods because of changes in precipitation cycles. Another example might a change in climate leading to an insect pest switching from one generation per year to two generations per year – with increased habitat damage as a result. To a certain extent, use of existing adaptive management techniques can deal with some of these issues.

One of the primary adaptations to climate change is to reduce as many of the other stresses on the resource as possible. This, in turn, may lead to an increase in the resiliency of habitats and species to the effects of climate change and variability. Accompanying the removal of other stressors is the need to maintain healthy, connected and genetically diverse populations. Isolated, small populations are often more prone to local extirpations than larger, more widespread populations. Connected populations also provide an opportunity for members of a population in one portion of a species range to move to another portion. Certain mitigation activities can assist in this area (see below). Finally, the more genetically diverse a population is to begin with the greater the possibility that a species might be able to naturally adapt to climate change. Thus, healthier or more robust species and habitats should be better able to adapt to climate change as an additional stress. Although these are goals managers already strive to accomplish, climate change increases their importance.

Fire is a natural part of many ecosystems. Climate change, however, could lead to more frequent fires and/or a greater probability of catastrophic fires. For instance, in areas that experience lower precipitation from climate change, reductions in soil moisture can increase drought stress on plants, making them more vulnerable to disease and pest outbreaks, thereby increasing mortality. This factor, in turn, could lead to more frequent fires or a greater probability of catastrophic fires. Managers can use prescribed fires and other techniques to reduce fuel load and the potential for catastrophic fires. An increased in the intensity of severe weather also places wildlife at risk. Although it is not possible to avoid the disturbance itself, it may be possible to minimize the effect of the event. For example, securing water rights to maintain water levels through a drought, or having an infrastructure capable of surviving floods might minimize some impacts. Maintaining widely dispersed and viable populations of individual species also minimizes the probability that localized catastrophic events will cause significant negative effects. Having multiple, widely spaced populations might also offset some of the population losses attributable to wide-spread events such as hurricanes and typhoons.

Rapidly changing climates and habitats will likely increase opportunities for invasive species to spread because of their adaptability to disturbance. Captive breeding for reintroduction and translocation is likely to be less successful if climate change is more dramatic. Such change could result in large-scale modifications of environmental conditions, including the loss or significant alteration of existing habitat over some or all of a species' range. Captive breeding and translocation should therefore not be perceived as panaceas for the loss of biological diversity that might accompany dramatic climate change, especially given the current state of the environment. Populations of many species are already perilously small and further loss of habitat and stress associated with severe climate change may push many taxa to extinction.

One extreme adaptation option would be the restoration of habitats currently under serious threat or the creation of new habitats in areas where natural colonization is unlikely to occur (Anonymous, 2000). Much as a translocated animal may act as an endangered species, the prospects of trying to mesh existing species with new species in a given habitat is quite daunting, and likely quite expensive. In many cases the knowledge of ecosystem interactions and species requirements may not even be fully known. The use of engineered habitats to facilitate species movements may require the development of an entirely new field of study.

Ultimately, managers may need to enhance or replace diminished or lost ecosystem services. This could mean manual seed dispersal or reintroducing pollinators for some plant species. In the case of pest outbreaks, it may mean the use of pesticides. Enhancing or replacing other services, such as contributions to nutrient cycling, ecosystem stability, and ecosystem biodiversity may be much more difficult. The loss or reduced capacity of ecosystem services may be one of the major sources of surprise from climate change and variability.

4.6.3.2 Consequences of adaptation activities on ecosystems and biodiversity

4.6.3.3 The contribution of biodiversity to adaptation options

4.6.3.4 Adaptation options in various ecosystems

4.6.4 Adaptation costs

4.6.5 Overall implications for biodiversity

4.6.5.1 Projected changes in biodiversity

4.6.5.2 Project changes in conservation areas

4.6.6 Interactions (with other sectors)

4.6.6.1 Changes in the ecosystem services

Although climate change is by definition a global issue, local conservation efforts can greatly help in maintaining and enhancing resilience and in limiting the longer-term damage from bleaching and related human impacts (Hughes et al., 2003).

The coral reef death of the Mentawai Islands (Sumatra, Indonesia) during the 1997 has been linked not only to increase of temperature, but also to the 1997 Indonesian wildfires. Widespread tropical wildfire is a recent phenomenon, the magnitude and frequency of which are increasing as population rises and terrestrial biomass continues to be disrupted. Where background nutrient supplies in reef waters are elevated or human activities have reduced upper trophic levels, reefs are likely to become increasingly susceptible to large algal blooms triggered by episodic nutrient enrichment from wildfires. Therefore, in addition to their impact on forest ecology and human health, tropical wildfires may pose a new threat to coastal marine ecosystems that could escalate into the 21st century (Abram et al., 2003).

4.6.6.2 Regional and global biotic feedbacks

Nutrient enrichment (either particulate or dissolved) is considered one of the main causes for environmental decline in oligotrophic waters harbouring coral reefs (Koop *et al.*, 2001a; Koop *et al.*, 2001b; Roberts *et al.*, 2002; Bongiorni *et al.*, 2003). High levels of nutrients stimulate macroalgal growth enabling the macroalgae to rapidly cover living coral tissues (Jompa and McCook, 2003a). High nutrient loads also lead to an increase in phytoplankton biomass, resulting in turbidity and reduced submarine light levels which affect zooxanthellae photosynthesis (Koop et al., 2001a; Koop et al., 2001b). However, experiments in fish-farms have shown that nutrients released from intensive mariculture may not necessarily lead to the demise of coral reefs, as is commonly presumed; on the contrary, corals can grow faster (Bongiorni et al., 2003).

4.7 Conclusions: implications for sustainable development

Ecosystems are the ultimate foundation of the global economy, as a result it is pertinent that social and economic development goals are integrated with information on functioning of these systems and their degree of vulnerability and sensitivity to multiple stresses including those related to human activities and climate change (World Resources, 2003 Voice and Power). For this to be achieved there has to be structured environmental governance in place to facilitate a forum where ecosystem science and monitoring can be infused into the decision making process at various levels to accomplish sustainable development.

The role of ecosystems in sustainable development involves an array of stakeholders, from government, the private sector, international agencies, the local community to individuals within a household (Jain, 2003; chapter 4, Millennium Ecosystem Assessment, 2005a). Evidence from different parts of the world shows that in most cases it is far from clear who is "in charge" of the long term sustainability of the ecosystem, let alone of the situation under future climates. Documentation of ongoing and future impacts of climate change on ecosystems calls for a clear and structured system of decision making at all levels, that is established as a key part of any environmental governance (Kennett, 2002). Environmental Governance is essential to establish a balance between human requirements satisfied through ecosystems while making sure that critical ecosystem thresholds under different climatic conditions are not exceeded (World Resources, 2003 Voice and Power).

Simple decisions on when and where to set a veld fire have been found to give rise to a wide range of consequences on the ecosystem and on other stakeholders whose interests may be jeopardized by for instance a wrong timing of the burning. Equally important are decisions on post fire management plans. This can help to avoid e.g. a switch towards the spread of weeds in the ecosystem, which may further increase the risk of future unplanned fires.

While it has been shown that the processes that trigger desertification sometimes lie far from the desertified area, for instance demand for certain products driven by international markets have triggered a chain of reactions ultimately leading to land degradation (Tole, 1998; Harou, 2002; chapter 4, Millennium Ecosystem Assessment, 2005a). To tackle these problems others argue that what is required is environmental governance that aims to connect local people with the ecosystem on which they directly depend. This will be achieved through decentralized management of natural resources where local people take primary role in governing the ecosystem within their locality (World Resources, 2003 Voice and Power).

Recent climate change adaptation assessments on biodiversity and impacts of climate change in southern Africa signal the need for policy to focus on managing areas outside protected areas for instance, subsistence rangelands (Von Maltitz et al., 2007). It is proposed that among others this can be achieved through devolution of resources ownership and management to communities, securing community tenure rights and incentives for utilization of resources. This argument is based on observations that greater diversity of species occurs outside protected areas which are more extensive (Scholes et al., 2004). Many species will need to have the ability to track suitable habitats in response to climate change. Species migration will be difficult to achieve in protected areas without costly interventions such as the establishment of corridors. This is in contrast to open communal and some private land use systems if strategic policies are put in place. Such policies could be supported by for example the approach hat has been adopted under the UNEP Division of Early Warning Africa Environmental Outlook Report 2 where environment is considered an economic opportunity. This approach provides a framework to consider climate and human factors that might individually or in combination reduce the ability to take advantage of environmental economic opportunities that support sustainable development for instance in Africa {http://www.unep.org/dewa/africa /publications/aeo-2/aeo-2report.asp}.

Impacts of climate change on ecosystems and resulting feed back show strong interrelationships involving climate, terrestrial ecosystem processes and human activities at various scales over time. Addressing these impacts require coordinated integrated cross-sectoral policy framework with long-term focus, a strategy that so far has not been easy to implement (Kennett, 2002). Further, the impacts of a wide range of policies on increasing vulnerability of ecosystems to climate change at both the national and international level are yet to be fully understood. There is growing evidence that significant impacts on the environment result mostly from perverse or unintended effects of policies from other sectors, which directly or indirectly have adverse consequences on ecosystems and other environment processes (chapter 4, Millennium Ecosystem Assessment, 2005a). Land re-distribution policies designed to increase food self-sufficiency ultimately contribute in reducing carbon sequestration, and loss of biodiversity through extensive clear-cutting increases vulnerability to land degradation.

At the national level assigning institutional mandates for multisectoral programs is a problem because government departments and agencies are organised along classical sectoral lines without effective mechanisms for handling complex issues that overlap agency jurisdictions (Nnadozie, 1998). An ecosystem-based management approach supported by institutional infrastructure with integrated legal and policy frameworks, operating at a variety of scales, to ensure that cumulative impacts of land-use and resource management result in resilient ecosystems will be the basis for sustainable development (Kennett, 2002).

One alternative towards designing integrated policies to address climate change and terrestrial systems is to formulate policies that cut across the three UN conventions; biodiversity, desertification and climate change conventions (Nnadozie, 1998). Mitigation strategies aimed at combating desertification would produce a win-win situation. For instance, apart from halting or reversing desertification, revegetation or re- or afforestation programs contribute towards increased carbon sequestration, increased soil moisture level and ultimately at a regional scale this may positively influence rainfall.

Effective mechanisms to analyse cross-sectoral impacts and to feed evolving scientific knowledge at the appropriate scale into policy-making are necessary (Schneider, 2003). For instance, understanding climate variability in arid lands could set a basis for policies that make it possible to put management strategies in place that take into account the fact that in arid lands responses of ecosystems to management interventions may be delayed until conducive climatic conditions return. Equipped with knowledge that because of their episodic nature arid, unmanaged lands take longer for negative impacts to become evident, policies could be formulated that will reduce the temptation by land managers to overstock during "good" periods. The latter is usually very profitable in the short term, but risks to undermine a balanced use of the ecosystem (Ash et al., 2002).

Despite the above arguments there is a substantial literature showing that developing and implementing policies and strategies for reducing vulnerability of ecosystems to climate change is closely linked to availability of capacity to address current needs (Chanda, 2001).

Sustainable development means meeting the needs of the present without compromising the ability of future generations to meet their needs. Prospects for successful adaptation and mitigation to climate change will remain limited as long as factors that contribute to persistent vulnerability to, for instance, drought and floods such as population growth, poverty and global economies are not resolved (Kates, 2000). For instance the southern Africa countries developed the in 1996, a policy and strategy for environment and sustainable development. Its goal was “to accelerate economic growth with greater equity and self-reliance; to improve health, income and living conditions of the poor majority: and to ensure equitable and sustainable use of the environment and natural resources for the benefit of present and future generations” (SADC ELMS, 1996).

However, the implementation of this policy is far from being realized. There is adequate evidence to demonstrate that poverty among other factors plays a significant role in the currently experienced degradation of ecosystems in developing countries, a fact which contributes to the failure to meet sustainable development goals (Tole, 1998; Harou, 2002). Unsustainable development and continued degradation affect those poor the most, who depend directly on their immediate ecosystems and have limited resources to invest in conservation or to seek alternatives to escape degradation (Hardy, 2003).

As a result climate change and sustainable development need to incorporate issues of equity (Kates, 2000; Jain, 2003; Richards, 2003). This is more the case given increasing evidence suggesting that climate change invokes additional inequities, as its impacts are unevenly distributed over space and time and disproportionally affect the poor (Tol, 2001). There are indications that there is an ongoing element of double exposure in terms of regions, sectors, ecosystems and social groups being confronted both by the impacts of climate change and by the consequences of economic globalization (O'Brien and Leichenko, 2000). However, there is limited work available, which studies the effect of the on-going processes of economic globalization onto ecosystems in an integrated manner. Yet it may well be that this has the effect of modifying and or exacerbating existing vulnerabilities to climate change (Richards, 2003). For instance what are the links between adaptations to the new global economy and adaptation strategies to address long-term climate change (O'Brien and Leichenko, 2000)? Again we are using coral reefs to illustrate with a specific case those concepts. Coral reefs, the most diverse of all marine ecosystems, are increasingly threatened by human activities, climate change and disease. The vulnerability of reefs to climate change became fully evident in 1997-1998 when elevated sea surface temperatures (SSTs) linked to global warming and a strong El Niño caused widespread coral bleaching and mortality throughout the tropical oceans (Abram et al., 2003).

Wilkinson (2002) reports there are two sets of counteracting human activities are affecting the destiny of the world’s coral reefs:

- reefs are continuing to decline in many areas around the world due to steadily increasing threats from direct human pressures and indirect pressures of Global Climate Change; but
- there are many conservation and management projects being initiated at international, regional, national and local levels to arrest the declines in coral reef health in specific areas, and some of these initiatives are showing considerable success.

Reefs are of critical importance to human survival (especially in developing countries) because they provide subsistence food for a substantial portion of the population, serve as the principle coastal protection structures for most tropical islands, and contribute major income and foreign exchange earnings from tourism (Burke *et al.*, 2002; Burke and Maidens, 2004).

The major global initiative for coral reef conservation during the last years was passed by the Parliament of Australia in early 2004, with the declaration of 33% of the whole province of the Great Barrier Reef (the GBR World Heritage Area) as highly protected status (or no-take zones), following a careful analysis using the best available science and extensive consultation with major stakeholders. This is an increase from approximately 5% that was the case when the GBR was first zoned for protection in 1981 (Wilkinson, 2004).
Coral reefs are highly productive hotspots of biodiversity that support social and economic development. Their protection, therefore, is a socioeconomic imperative, as well as an environmental one. Global warming, coupled with preexisting human impacts, is a grave threat

that has already caused substantial damage. However, the available evidence indicates that, at a global scale, reefs will undergo major changes in response to climate change rather than disappear entirely (Hughes et al., 2003). The capacity of coral reef ecosystems to continue to generate the valuable goods and services (on which human welfare depends) has to be better understood and more actively managed. Therefore "hot spots" of corals (centres of higher biodiversity) must be kept in order to maintain biodiversity. Marine protected areas (MPAs) are currently the best management tool for conserving coral reefs and many other marine systems. Although MPAs can help preserve fish stocks because they change human behaviour, but they do not prevent or hold back warm water, or stop bleaching (Roberts et al., 2002).

Research in support of reef management urgently needs to increase the scale of experiments, sampling, and modeling to match the scale of impacts and key biological processes (e.g., dispersal, bleaching, and overfishing) and go beyond the current emphasis on routine monitoring and mapping. Emerging research on marine reserves and how they work to protect harvested stocks and spread risk also needs to be expanded and applied specifically to the tropics. These approaches must be integrated with socioeconomic aspects of coral reef resilience, incorporating adaptive management systems that operate locally, regionally, and globally. International integration and scaling-up of reef management is an urgent priority. Ecological modeling studies indicate that, depending on the level of exploitation outside no-take areas (NTAs), at least 30% of the world's coral reefs should be NTAs to ensure long-term protection and maximum sustainable yield of exploited stocks (Roberts et al., 2002; Hughes et al., 2003). Yet, even in affluent countries, such as the United States and Australia, less than 5% of reefs today are NTAs. Wealthy countries have an obligation to take the lead in increasing the proportion of reefs that are NTAs, while simultaneously controlling greenhouse-gas emissions.

4.7.1 Relationships to cross-cutting themes

Relatively few studies have attempted to couple land use and climate change scenarios in the analysis of impacts on ecosystem processes, goods and services. Global scale studies include the (Millennium Ecosystem Assessment, 2005b), the PAGE studies (Sala *et al.*, 2000b; World Resources Institute, 2000). Global scale scenarios are not appropriate, however, for the analysis of ecosystem impacts at local to regional scales. Moreover, there has been increasing interest in regional scale studies to promote development of similarly scaled policies, and to enable stakeholders to engage with climate related issues at a scale where they interact in terms of daily activities and management (Holman *et al.*, 2004b). Only a few such studies have been undertaken at the regional or local scale (Dirnbock *et al.*, 2003; Holman *et al.*, 2004a; Holman *et al.*, 2004b; Stefanescu *et al.*, 2004; Scheller and Mladenoff, 2005; Araújo *et al.*, 2007; de Chazal *et al.*, 2007; Giupponi *et al.*, 2007; Harrison *et al.*, 2007; Metzger *et al.*, 2007; Rounsevell *et al.*, 2007a; Schröter *et al.*, 2007). *[need to update with MEA sub-global studies].*

A number of these studies have been carried out in Europe, usually under the auspices of European Union funded projects, such as ATEAM, Advanced Terrestrial Ecosystem Analysis and Modelling (Smith *et al.*, 2005; Schröter *et al.*, 2007), ACCELERATES, Assessing Climate Change Effects on Land use and Ecosystems: from Regional Analysis to the European Scale (Rounsevell *et al.*, 2007a) and VISTA, Vulnerability of ecosystem services to land use change in traditional agricultural landscapes (de Chazal *et al.*, 2007).

4.7.2 Policy implications

An overview of recent rapid global land-cover change for the period 1981-2000, including changes in deforestation and other forest cover changes, dry land degradation (desertification), changes in cropland and urban extent has been compiled from a wide range of global, regional and sub-regional remote sensing and other data sources (Lepers *et al.*, 2005). Data sets are not well distributed over the globe, with a more data available in some parts of the world than others *[details to come after consultation with LUCC, details not mentioned in papers]*.

This implies that areas of rapid land use change may be occurring in locations not currently identified. A review of recent land use and land changes, with an emphasis on tropical regions, has also recently been completed (Lambin *et al.*, 2003). The following summary is drawn from these two reviews.

Deforestation has received much research attention in comparison to the other land use changes, with cropland change receiving the least. Areas of most rapid deforestation have occurred in the tropics, particularly in the Amazon basin and South-east Asia, with a growing trend of forest degradation in Eurasia. The net global decrease in forest cover was 9.4 million ha per year for the period 1990-2000, with tropical forests losing 15.2 million ha per year although overall, the net change was positive for temperate regions and negative in tropical regions. Boreal forests are also undergoing change, particularly with respect to altered fire frequencies. For example, more than 7.5 million ha of forest was burnt in a 6 year period in the late 1990s (ibid.).

For dry land degradation, the Asian continent has experienced the greatest change. However, given major data gaps in particularly sensitive arid zones such as the Mediterranean basin, eastern Africa, parts of south America, and in the U.S, it is possible that similar or at least some yet undetected changes may have also occurred in these regions.

For cropland changes, major areas of cropland increase are spread across a number of continents. Principal locations include south-east Asia, Bangladesh, along the Indus Valley, parts of the middle east and central Asia, Great Lakes region of eastern Africa, the southern border of Amazon basin, and in the Great Plains region of the U.S. In terms of cropland decrease, the lowlands of southeastern U.S. have experienced the greatest decrease, followed by Asia (eastern China), South America (parts of Brazil and Argentina) and Europe. Africa was the only continent where no decrease in cropland was identified. For changes in urban extent, the greatest change is located in non-industrialised countries throughout the tropical belt. It is estimated that 1 to 2 million ha of cropland are being taken out of production every year in these regions to meet the land demand for housing, industry, and associated infrastructure. Urban populations are growing faster than rural populations, while the number of mega-cities (more than 10 million inhabitants) has increased from 1 in 1950 to 17 in 2000, the majority being in developing countries (ibid.).

Analysis of the “root causes” of land use change suggest that, for tropical deforestation, more important triggers than local pressures of poverty are created by changing international economic opportunities (particularly in developing countries), resulting in decisions by governments to undertake new settlement schemes, other development projects, or logging and mining projects (Lambin *et al.*, 2001). The argument is made that the motivation for these decisions is the desire to secure territory, attract international capital, facilitate market opportunities, or promote interests of specific interest groups by exploiting state owned resources. For rangeland degradation the usual attribution to over-grazing, misrepresents a general trend in agricultural intensification, resulting in the conversion and fragmentation of formerly more contiguous grassland areas are greater drivers of change. For agricultural intensification, three main factors are implicated: first, land scarcity in developing countries, particularly those experiencing rapid population growth, secondly, commodification of selected crops, following commercial intensification, and thirdly, intervention, by state, donor or NGO sponsored projects to promote development in a region or economic sector as an attempt to increase income for individuals and/or the state. Finally, for urbanization, is recognised as substantially increasing the overall ecological footprint, both regionally and globally.

Moreover, urban expansion, particularly in developing countries tends to expand into prime cropland and grassland areas.

4.7.2.1 Resource management policies

4.7.2.2 Subsistence livelihoods and indigenous peoples

4.7.2.3 Other policy implications

Because mountain ecosystems are often referred to as “hotspots of biodiversity” (Price *et al.*, 2000), they warrant protection in order to maintain ecosystem integrity and adaptability. Furthermore, montane vegetation is important in terms of its protective role against slope erosion and as a component of mountain hydrology and water quality. Whatever the ecosystem response to multiple environmental stress factors, adaptation of natural ecosystems to climatic change in many regions cannot be achieved without some kind of human intervention, in the form of ecosystem management. Reforestation would in some cases be a viable adaptation option, and so would afforestation of abandoned agricultural land. Freshwater biological systems can be assisted in a number of ways which could help mitigate the impacts of climate change, particularly through the increase and protection of riparian vegetation, and restoring river and stream channels to their natural morphologies.
Becker (2001) have outlined suggested strategies for research under the auspices of the International Geosphere-Biosphere Program (IGBP), the International Human Dimensions Program (IHDP) and the Global Terrestrial Observing System (GTOS), in a coordinated scheme known as the Mountain Research Initiative MRI. In this context, four main research activities and tasks have been put forward, namely:

- long-term monitoring and analysis of indicators of environmental change in mountain regions, with a particular focus on cryospheric indicators, watershed hydrology, and terrestrial and acquatic ecosystems;
- integrated model-based studies of environmental change in different mountain regions, in particular coupled ecological, hydrological and land-use models, models allowing the study of feedbacks between land surfaces and the atmosphere, and integrated (physical, biological and economic) analyses of environmental change for policy purposes;
- process studies along altitudinal gradients and in associated headwater basins; such studies allow an assessment of environmental change at ecological and climatological boundaries, and thus the sensitivity of vegetation, snow, ice, and water resources to a range of forcings;
- sustainable land-use and natural resource management, with priorities for changes in forest resources, shifts in mountain agriculture and food security, and modifications to water resources driven by economic and demographic factors.

These diverse research and education initiatives, beyond their intrinsic value, also have the objective of feeding into the policy sphere. According to the outcome of ongoing and future research, policy response should lead to coordinated action by UN, governmental and non-governmental organizations in favor of environmental protection in mountains and uplands, and to help local populations adapt to changing ecological, economic, and health-related impacts. Policy should also aim to convince key global actors such as the World Trade Organization (WTO) to take mountain issues into consideration in the planning of future trade accords and commercial practices. Furthermore, a strengthening of ties between the "mountain chapter" of Agenda 21 and the UN Conventions on Climate Change, Biodiversity, and Desertification may lead to a more efficient, holistic approach to the problems currently facing many mountain regions.

4.7.3 Development implications

4.8 Key uncertainties and research priorities

[These need to be developed, and will include:
Identification of "tipping points" in the global carbon cycle and the interaction between drivers of sink activity (primarily CO2 fertilization, forest latitudinal shifts and reforestation/revegetation of abandoned land) and source activity (fire, tundra warming, deforestation, soil exposure and erosion)
The maximum carbon sequestration potential of natural ecosystems, and their likely time evolution.

The understanding of fire responses to climate and vegetation drivers
Projections of biodiversity responses, including migration (or spread), extinction and other population level processes, and the assumptions and uncertainties that underpin these, and the associated threats to global conservation efforts.

The relationship between biodiversity and ecosystem services at a scale relevant to human well-being.

The impacts of rising atmospheric CO2 and temperatures on coral reef and other marine systems, in an evolutionary context.

Adaptive management responses to preserve biodiversity, ecosystem function and ecosystem services.

Identification of biological and biospheric indicators to provide warning of the exceedance of key thresholds, and their links to policy decisions.

An urgent need for a research agenda examining how and where the synergy between invasive alien species and climate change threaten ecosystem services, integrity and sustainability and develop a strategy for anticipating and dealing with these global threats, especially in developing countries.]

References

Note, not all references are included in the following list, since some authors provided their input too late for merging.

Abildtrup, J., P. Rosato, M. Gylling, M. Fekete-Farkas, E. Audlsey, C. Giupponi, and M.D.A. Rounsevell, 2007: Socio-economic scenarios for agricultural land use studies. *Environmental Science and Policy*, pp. (submitted).

Abram, N.J., M.K. Gagan, M.T. McCulloch, J. Chappell, and W.S. Hantoro, 2003: Coral reef death during the 1997 Indian Ocean dipole linked to Indonesian wildfires. *Science*, 301, pp. 952-955.

Ad Hoc Technical Expert Group on Forest Biological Diversity, 2002: *Review of the status and trends of, and major threats to, the forest biological diversity.* CBD Technical Series, Secretariat of the Convention on Biological Diversity, Montreal, 164pp.

Ainsworth, E.A., and S.P. Long, 2005a: What have we learned from 15 years of free-air CO2 enrichment (FACE)? A meta-analytic review of the responses of photosynthesis, canopy. *New Phytologist*, 165(2), pp. 351-371.

Ainsworth, E.A., and S.P. Long, 2005b: What have we learned from 15 years of free-air CO2 enrichment (FACE)? A meta-analytic review of the responses of photosynthesis, canopy properties and plant production to rising CO2. *New Phytologist*, 165(2), pp. 351-372.

Alcamo, J., K. Kok, G. Busch, J. Priess, B. Eickhout, M. Rounsevell, D. Rothman, and H. M., 2005: Searching for the future of land: scenarios from the local to global scale. In: *LUCC Synthesis book* [Geist, H., and E.F. Lambin (eds.)]. pp. Chapter 8, in press.

Alford, R.A., and S.J. Richards, 1999: Global amphibian declines: A problem in applied ecology. *Annual Review of Ecology and Systematics*, 30, pp. 133-165.

Alverson, K., R. Bradley, K. Briffa, J. Cole, M. Hughes, I. Larocque, T. Pedersen, L. Thompson, and S. Tudhope, 2001: A global paleoclimate observing system. *Science*, 293 (5527), pp. 47-48.

Anderson, L., Maherali H, H. Johnson, H. Polley, and R. Jackson, 2001: Gas exchange and photosynthetic acclimation over subambient to elevated CO2 in a C3-C4 grassland. *Global Change Biol*, *7*, pp. *693-707*.

Anonymous, 2000: *Potential U.K. adaptation strategies for climate change.* DETR Summary Report, Wetherby, West Yorkshire, 19pp.

Araújo, M.B., D. Nogues-Bravo, I. Reginster, M.D.A. Rounsevell, and R.J. Whittaker, 2007: Human-biodiversity conflicts in a changing Europe. pp. (submitted).

Araújo, M.B., and R.G. Pearson, 2005: Equilibrium of species' distributions with climate. *Ecography*, In press

Araújo, M.B., W. Thuiller, and R.G. Pearson, 2006: Climate warming and the decline of amphibians and reptiles in Europe. *Journal of Biogeography*, In press

Araújo, M.B., R.J. Whittaker, R. Ladle, and M. Erhard, 2005: Reducing uncertainty in projections of extinction risk from climate change. *Global Ecology and Biogeography*, In press

Arnell, N., 1999: The effect of climate change on hydrological regimes in Europe. *Global Environmental Change*, 9, pp. 5-23.

Ash, A.J., D.M.S. Smith, and N. Abel, 2002: Land degradation and secondary production in semi-arid and grazing systems: What is the evidence? In: *Global desertification. Do humans cause deserts?* [Reynolds, J.F., and D.M. Stafford-Smith (eds.)]. Dahlem University Press, Berlin, Germany, pp. 111-134.

Audsley, E., K.R. Pearn, C. Simota, G. Cojocaru, E. Koutsidou, M.D.A. Rounsevell, M. Trnka, and V. Alexandrov, 2007: What can scenario modelling tell us about future European scale land use, and what not? *submitted*

Augustin, L., C. Barbante, P.R.F. Barnes, J.M. Barnola, M. Bigler, E. Castellano, O. Cattani, J.

Chappellaz, D. Dahljensen, B. Delmonte, G. Dreyfus, G. Durand, S. Falourd, H. Fischer, J. Fluckiger, M.E. Hansson, P. Huybrechts, R. Jugie, S.J. Johnsen, J. Jouzel, P. Kaufmann, J. Kipfstuhl, F. Lambert, V.Y. Lipenkov, G.V.C. Littot, A. Longinelli, R. Lorrain, V. Maggi, V. Masson-Delmotte, H. Miller, R. Mulvaney, J. Oerlemans, H. Oerter, G. Orombelli, F. Parrenin, D.A. Peel, J.R. Petit, D. Raynaud, C. Ritz, U. Ruth, J. Schwander, U. Siegenthaler, R. Souchez, B. Stauffer, J.P. Steffensen, B. Stenni, T.F. Stocker, I.E. Tabacco, R. Udisti, R.S.W. van de Wal, M. van den Broeke, J. Weiss, F. Wilhelms, J.G. Winther, E.W. Wolff, and M. Zucchelli, 2004: Eight glacial cycles from an Antarctic ice core. *Nature (London),* 429(6992), pp. 623-628.

Bachelet, D., R.P. Neilson, T. Hickler, R.J. Drapek, J.M. Lenihan, M.T. Sykes, B. Smith, S. Sitch, and K. Thonicke, 2003: Simulating past and future dynamics of natural ecosystems in the United States. *Global Biogeochemical Cycles,* 17(2), pp. 1045.

Bachelet, D., R.P. Neilson, J.M. Lenihan, and R.J. Drapek, 2001: Climate change effects on vegetation distribution and carbon budget in the United States. *Ecosystems*, 4(3), pp. 164-185.

Baird, A.H., and P.A. Marshall, 2002: Mortality, growth and reproduction in scleractinian corals following bleaching on the Great Barrier Reef. *Mar. Ecol. Prog. Ser.*, 237, pp. 133-141.

Baker, A., 2003: Flexibility and specificity in coral-algal symbiosis: Diversity, ecology, and biogeography of Symbiodinium. *Annu. Rev. Ecol. Evol. Syst.*, 34, pp. 661–689.

Baker, A.C., 2001: Reef corals bleach to survive change. *Nature*, 411, pp. 765-766.

Baker, A.C., C.J. Starger, T.R. McClanahan, and P.W. Glynn, 2004: Corals adaptive response to climate change: Shifting to new algal symbionts may safeguard devastated reefs from extinction. *Nature*, 430, pp. 741.

Bakkenes, M., J.R.M. Alkemade, F. Ihle, R. Leemans, and J.B. Latour, 2002: Assessing effects of forecasted climate change on the diversity and distribution of European higher plants for 2050. *Global Change Biology*, 8(4), pp. 390-407.

Barbraud, C., and H. Weimerskirch, 2001: Emperor penguins and climate change. *Nature (London),* 411(6834), pp. 183-186.

Bassiri, R., H.J.F. Reynolds, R. A.Virginia, and M.H. Brunelle, 1998: Growth and root $NO3^-$ and $PO4^{3-}$ uptake capacity of three desert species in response to atmospheric CO2 enrichment. *Australian Journal of Plant Physiology*(24), pp. 353-358.

Batterbury, S., and A. Warren, 2001: The African Sahel 25 years after the great drought: assessing progress and moving towards new agendas and approaches. *Global Environmental Change,* 11(1), pp. 1-8.

Beare, D., F. Burns, E. Jones, K. Peach, E. Portilla, T. Greig, E. McKenzie, and D. Reid, 2004: An increase in the abundance of anchovies and sardines in the north-western North Sea since 1995. *Global Change Biology*, 10(7), pp. 1209-1213.

Beare, D.J., S. Batten, M. Edwards, and D.G. Reid, 2002: Prevalence of boreal Atlantic, temperate Atlantic and neritic zooplankton in the North Sea between 1958 and 1998 in relation to temperature, salinity, stratification intensity and Atlantic inflow. *Journal of Sea Research*, 48(1), pp. 29-49.

Beaugrand, G., K.M. Brander, J.A. Lindley, S. Souissi, and P.C. Reid, 2003: Plankton effect on cod recruitment in the North Sea. *Nature (London)*, 426(6967), pp. 661-664.

Beaugrand, G., P.C. Reid, F. Ibanez, J.A. Lindley, and M. Edwards, 2002: Reorganization of North Atlantic marine copepod biodiversity and climate. *Science*, 296(5573), pp. 1692-1694.

Becker, A., and H. Bugmann, 2001: Global change and mountain regions. *IGBP Report 49*

Beerling, D.J., and F.I. Woodward, 2001: *Vegetation and the Terrestrial Carbon Cycle. Modelling the first 400 million years.* Cambridge University Press, Cambridge, 405pp.

Behrenfeld, M.J., and P.G. Falkowski, 1997: A consumers guide to phytoplankton primary production models. *Limnol. Oceanogr.*, 42(7), pp. 1479-1491.

Beniston, M., 2000: *Environmental Change in Mountains and Uplands.* Arnold, London, 172pp.

Beniston, M., 2003: Climatic change in mountain regions: a review of possible impacts. *Climatic Change*, 59, pp. 5-31.

Beniston, M., 2004a: The 2003 heat wave in Europe: A shape of things to come? An analysis based on Swiss climatological data and model simulations. *Geophysical Research Letters*, 31(2), pp. art. no.-L02202.

Beniston, M., 2004b: *Climatic change and its impacts: An overview focusing on Switzerland.* Kluwer Academic Publishers, Dordrecht and Boston, 296 pp.pp.

Beniston, M., and P. Jungo, 2002: Shifts in the distributions of pressure, temperature and moisture in the alpine region in response to the behavior of the North Atlantic Oscillation. *Theor. and Appl. Clim.*, 71, pp. 29-42.

Benning, T.L., D. LaPointe, C.T. Atkinson, and P.M. Vitousek, 2002: Interactions of climate change with biological invasions and land use in the Hawaiian Islands: Modeling the fate of endemic birds using a geographic information system. *Proceedings of the National Academy of Sciences*, 99, pp. 14246-14249.

Bergström, S., B. Carlsson, M. Gardelin, G. Lindström, A. Pettersson, and M. Rummukainen, 2001: Climate change impacts on runoff in Sweden - assessments by global climate models, dynamical downscaling and hydrological modelling. *Climate Research*, 16(101-112)

Bernhoft, A., J.U. Skaare, O. Wiig, A.E. Derocher, and H.J.S. Larsen, 2000: Possible immunotoxic effects of organochlorines in polar bears (Ursus maritimus) at Svalbard. *Journal of Toxicology and Environmental Health - Part A*, 59(7), pp. 561-574.

Berkelmans R, 2002, Time-integrated thermal bleaching thresholds of reefs and their variation on the Great Barrier Reef. Marine Ecology Progress Series 229:73-82.

Bhagooli, R., and M. Hidaka, 2003: Comparison of stress susceptibility of in hospite and isolated zooxanthellae among five coral species. *J. Exp. Mar. Biol. Ecol.*, 291, pp. 181–197.

Blaschke, L., M. Schulte, A. Raschi, N. Slee, H. Rennenberg, and A. Polle, 2001: Photosynthesis, Soluble and Structural Carbon Compounds in Two Mediterranean Oak Species (Quercus pubescens and Q. ilex) after Lifetime Growth at Naturally Elevated CO2 Concentrations. *Plant Biology*(3), pp. 288-298.

Bond, W.J., and S. Archibald, 2003: Confronting complexity: Fire policy choices in south African savanna parks. pp. 381-389.

Bond, W.J., and G.F. Midgley, 2000: A proposed CO2-controlled mechanism of woody plant invasion in grasslands and savannas. *Global Change Biology*, 6(8), pp. 865-869.

Bond, W.J., G.F. Midgley, and F.I. Woodward, 2003: The importance of low atmospheric CO2 and fire in promoting the spread of grasslands and savannas. *Global Change Biology*, 9(7), pp. 973-982.

Bond, W.J., and J.J. Midgley, 2003: The evolutionary ecology of sprouting in woody plants. *International Journal of Plant Sciences*, 164(3), pp. S103.

Bond, W.J., and B.W. Van Wilgen, 1996: *Fire and plants.* Chapman and Hall, London.

Bond, W.J., F.I. Woodward, and G.F. Midgley, 2005: The global distribution of ecosystems in a world without fire. *New Phytologist*, 165(2), pp. 525-537.

Bongiorni, L., S. Shafir, D. Angel, and B. Rinkevich, 2003: Survival, growth and gonad development of two hermatypic corals subjected to in situ fish-farm nutrient enrichment. *Mar. Ecol. Prog. Ser.*, 253, pp. 137-144.

Bopp, L., O. Aumont, S. Belviso, and P. Monfray, 2003: Potential impact of climate change on marine DMS (dimethylsulfide) emissions. *Tellus*, 55B, pp. 11-22.

Bopp, L., O. Boucher, O. Aumont, S. Belviso, J.-L. Dufresne, M. Pham, and P. Monfray, 2004: Will marine dimethylsulfide emissions amplify or alleviate global warming? A model study. *Can. J. Fish. Aquat. Sci.*, 61, pp. 826-835.

Bopp, L., P. Monfray, O. Aumont, J.-L. Dufresne, H. Le Treut, G. Madec, L. Terray, and J. Orr, 2001: Potential impact of climate change on marine export production. *Global Biogeochemical*

Cycles, 15, pp. 81-99.

Borghetti, M., S. Cinnirella, F. Magnani, and A. Saracino, 1998: Impact of long-term drought on xylem embolism and growth in Pinus halepensis Mill. *Trees - Structure and Function*, 12(4), pp. 187-195.

Bortenschlager, S., 1993: Das höchst gelegene Moor der Ostalpen "Moor am Rofenberg" 2760 m. *Festschrift Zoller, Diss. Bot.*, 196, pp. 329-334.

Brander, K.M., 2005: Cod recruitment is strongly affected by climate when stock biomass is low. *ICES Journal of Marine Science*, 62(3), pp. 339-343.

Brander, K.M., G. Blom, M.F. Borges, K. Erzini, G. Henderson, B.R. MacKenzie, H. Mendes, J. Ribeiro, A.M.P. Santos, and R. Toresen, 2003: Changes in fish distribution in the eastern North Atlantic: Are we seeing a coherent response to changing temperature? *ICES Marine Science Symposia*, 219, pp. 261-270.

Breshears, D.D., and F.J. Barnes, 1999: Interrelationships between plant functional types and soil moisture heterogeneity for semiarid landacspes within the grassland/forest continuum: a unified conceptual model. *Landscape Ecology*, 14, pp. 465-478.

Brooks, M.L., C. D'Antonio, D.M. Richardson, J.B. Grace, J.E. Keeley, J.M. Di Tomaso, R.J. Hobbs, M. Pellant, and D. Pyke, 2004a: Effects of invasive alien plants on fire regimes. *Bioscience*, 54(7), pp. 677-688.

Brooks, M.L., C.M. D'Antonio, D.M. Richardson, J.B. Grace, J.E. Keeley, J.M. DiTomaso, R.J. Hobbs, M. Pellant, and D. Pyke, 2004b: Effects of invasive alien plants on fire regimes. *Bioscience*, 54(7), pp. 677-688.

Brown, B.E., R.P. Dunne, M.S. Goodson, and A.E. Douglas, 2000: Bleaching patterns in coral reefs. *Nature*, 404, pp. 142-143.

Brown BE, 1997, Coral bleaching: causes and consequences. Coral Reefs 16:129-138

Buchmann, N., 2002: Plant ecophysiology and forest response to global change. *Tree Physiology*, 22(15-16), pp. 1177-1184.

Buddemeier, R.W., J.A. Kleypas, and R.B. Aronson, 2004: *Coral reefs & Global Climate Change: Potential Contributions of Climate Change to Stresses on Coral Reefs Ecosystems.* Pew Center on Global Climatic Change, Arlington, USA, Arlington, 44 pp.pp.

Buddemeier RW, Fautin DG, 1993, Coral bleaching as an adaptative mechanism – a testable hypothesis. Bioscience, 43, 320-326.

Bugmann, H., 1997: Sensitivity of forests in the European Alps to future climatic change. *Climate Research*, 8(1), pp. 35-44.

Bugmann, H., and C. Pfister, 2000: Impacts of interannual climate variability on past and future forest composition. *Regional Environmental Change*, 1, pp. 112-125.

Burke, L., and J. Maidens, 2004: *Reefs at risk in the Caribbean.* Words Resources Institute, USA, 80 pp.pp.

Burke, L., E. Selig, and M. Spalding, 2002: *Reefs at risk in Southeast Asia.* Words Resources Institute, USA, 72 pp.pp.

Burns, C.E., K.M. Johnson, and O.J. Schmitz, 2003: Global climate change and mammalian species diversity in U.S. national parks. *Proceedings of the National Academy of Science, USA*, 100(20), pp. 11474-11477.

Burrowes, P.A., R.L. Joglar, and D.E. Green, 2004: Potential causes for amphibian declines in Puerto Rico. *Herpetologica*, 60(2), pp. 141-154.

Busby, J.R., 1988: *Potential implications of climate change on Australia's flora and fauna.* CSIRO Division of Atmospheric Research, Melbourne, 397-388pp.

Bush, M.B., M.R. Silman, and D.H. Urrego, 2004: 48,000 years of climate and forest change in a biodiversity hot spot. *Science,* 303(5659), pp. 827-829.

Canadell, J.G., P. Ciais, P. Cox, and M. Heimann, 2004: Quantifying, understanding and managing the carbon cycle in the next decades. *Climatic Change*, 67(2-3), pp. 147-160.

CBD, 2003: *Expert Meeting On The Ecosystem Approach.* Expert meeting Report, Convention on Biological Diversity, Montreal, 31pp.

Cesar, H., L. Burke, and L. Pet-Soede, 2003: *The economics of worldwide coral reefs degradation.* Cesar Environmental Economics Consulting (CEEC), 6828GH Arnhem, The Netherlands, 23 pp.pp.

Cesar, H., P. van Beukering, S. Pintz, and J. Dierking, 2002: *Economic valuation of the coral reefs of Hawaii. Final report to National Oceanic and Atmospheric Administration Coastal Ocean Program.* Cesar Environmental Economics Consulting, Arnhem, The Netherlands.

Chanda, R., 2001: Safe-guarding the globe or basic well-being? Global environmental change issues and Southern Africa. In: *Globalisation, Democracy and Development in Africa.* [T., A., R. S.M., and A. A.G.M. (eds.)]. Organisation of Social Science Reserach in Eastern and Southern Africa (OSSREA), Addis Ababa, pp. 246-262.

Chapin, F.S., P.A. Matson, and H.A. Mooney, 2002: *Principles of terrestrial ecosystem ecology.* Springer., New York.

Chapuis, J.L., Y. Frenot, and M. Lebouvier, 2004: Recovery of native plant communities after eradication of rabbits from the subantarctic Kerguelen Islands, and influence of climate change. *Biological Conservation*, 117(2), pp. 167-179.

Chauhan, M., and B. Gopal, 2001: Biodiversity and management of Keoladeo National Park (India) - a wetland of international importance. In: *Biodiversity in Weylands: Assessment, Function and Conservation*. Backhuys Publishers, Leiden, The Netherlands, pp. 217-256.

Cheddadi, R., J. Guiot, and D. Jolly, 2001: The Mediterranean vegetation: what if the atmospheric CO2 increased? *Landscape Ecology*, 16(7), pp. 667-675.

Chidumayo, E.N., 2001: Climate and phenology of savanna vegetation in southern Africa. *Journal Of Vegetation Science*, 12(3), pp. 347-354.

Christensen, J., T. Carter, and F. Giorgi, 2002: PRUDENCE employs new methods to assess European climate change. *EOS*, 83, pp. 13.

Christensen, J., and O. Christensen, 2003: Severe summer-time flooding in Europe. *Nature*, 421, pp. 805-806.

Clark, R.A., and C.L.J. Frid, 2001: Long-term changes in the North Sea ecosystem. *Environ. Rev.*, 9, pp. 131-187.

Claussen, M.C., V. Kubatzki, A. Brovkin, P. Ganopolski, H. Hoelzmann, and J. Pachur, 1999: Simulation of an abrupt change in Saharan vegetation in the mid-Holocene. *Geophysical Research Letters*, 26, pp. 2037-2040.

Cokey E, Hallock P, Lidz BH, 1996, Decadal-scale changes in benthic foraminiferal assemblages off Key Largo, Florida. Coral Reefs, 15, 237-248.

Coles, S.L., Brown BE, 2003, Coral Bleaching – Capacity for Acclimatization and Adaptation. Adv Mar Biol 46:183-223

Costanza, R., 2001: Visions, values, valuation, and the need for an ecological economics. *Bioscience*, 51(6), pp. 459-468.

Costanza, R., R. D’arge, R. de Groot, S. Farber, M. Grasso, B. Hannon, K. Limburg, S. Naeem, R.V. O’neill, J. Paruelo, R.G. Raskin, P. Sutton, and M. van den Belt, 1997: The value of the world’s ecosystem services and natural capital. *Nature (London)*, 387(15 May), pp. 253-260.

Cowling, R.M., P.W. Rundel, B.B. Lamont, M.K. Arroyo, and M. Arianoutsou, 1996: Plant diversity in mediterranean-climate regions. *Trends in Ecology and Evolution*, pp. 362-366.

Cox, P.M., R.A. Betts, M. Collins, P.P. Harris, C. Huntingford, and C.D. Jones, 2004: Amazonian forest dieback under climate-carbon cycle projections for the 21st century. *Theoretical and Applied Climatology*, 78, pp. 137-156.

Cox, P.M., R.A. Betts, C.D. Jones, S.A. Spall, and I.J. Totterdell, 2000: Acceleration of global warming due to carbon-cycle feedbacks in a coupled climate model. *Nature (London),* 408(9.Nov.), pp. 184-187.

Craig, P., T. Pepper, and T.E. Morrell, 1994: The decline of fruit bats in America Samoa due to hurricanes and overhunting. *Biological Conservation*, 69, pp. 261-266.

Cramer, W., A. Bondeau, F.I. Woodward, I.C. Prentice, R.A. Betts, V. Brovkin, P.M. Cox, V. Fisher, J.A. Foley, A.D. Friend, C. Kucharik, M.R. Lomas, N. Ramankutty, S. Sitch, B. Smith, A. White, and C. Young-Molling, 2001: Global response of terrestrial ecosystem structure and function to CO2 and climate change: results from six dynamic global vegetation models. *Global Change Biology*, 7(4), pp. 357-373.

Crawley, M.J., 1989: Chance and timing in biological invasions. In: *Biological Invasions. A Global Perspective*. John Wiley and Sons, Chichester, pp. 407-424.

Crawley, M.J., P.H. Harvey, and A. Purvis, 1996: Comparative ecology of the native and alien floras of the British Isles. *Philosophical Transactions of the Royal Society of London Series B-Biological Sciences*, 351(1345), pp. 1251-1259.

Croxall, J.P., P.N. Trathan, and E.J. Murphy, 2002: Environmental change and Antarctic seabird populations. *Science*, 297(5586), pp. 1510-1514.

Cubasch, U., G.A. Meehl, H.J. Boer, R.J. Stouffer, M. Dix, A. Noda, C.A. Senior, S. Raper, and K.S. Yap, 2001: Projections of future climate change. Canbridge University Press, pp. 525-582.

Daily, G.C., T. Söderqvist, S. Aniyar, K. Arrow, P. Dasgupta, P.R. Ehrlich, C. Folke, A. Jansson, B. Jansson, N. Kautsky, S. Levin, J. Lubchenco, K. Mäler, D. Simpson, D. Starrett, D. Tilman, and B. Walker, 2000: The value of nature and the nature of value. *Science*, 289, pp. 395-396.

Davis, A.J., L.S. Jenkinson, J.H. Lawton, B. Shorrocks, and S. Wood, 1998: Making mistakes when predicting shifts in species range in response to global warming. *Nature (London)*, 39(19 Febr.), pp. 783-786.

Davis, M., 1989: Lags in vegetation response to greenhouse warming. *Climatic Change*, 15, pp. 75-82.

Davis, M.B., and R.G. Shaw, 2001: Range shifts and adaptive responses to Quaternary climate change. *Science*, 292(5517), pp. 673-679.

De Angelis, P., K.S. Chigwerewe, and G.E. Scarascia Mugnozza, 2000: Litter quality and decomposition in a CO2-enriched Mediterranean forest ecosystem. *Plant and Soil*, 224(1), pp. 31.

De Bono, A., P. Peduzzi, G. Giuliani, and S. Kluser, 2004: *Impacts of summer 2003 heat wave in Europe*. Nairobi, Kenya, 4pp.

de Chazal, J., F. Quetier, S. Lavorel, and A. Van Doorn, 2007: Vulnerability assessment in extensive agro-pastoral landscapes of Europe. pp. (submitted).

de Koning, F., R. Olschewski, E. Veldkamp, P. Benitez, M. Lopez-Ulloa, T. Schlichter, and M. de Urquiza, 2005: The ecological and economic potential of carbon sequestration in forests: Examples from South America. *Ambio*, 34(3), pp. 224-229.

de Menocal, P.B., 2004: African climate change and faunal evolution during the Pliocene-Pleistocene. *Earth and Planetary Science Letters*, 220, pp. 3-24.

del Barrio, G., 1998: to come, not listed in bibilography of source article. *to come*

Demenocal, P., J. Ortiz, T. Guilderson, J. Adkins, M. Sarnthein, L. Baker, and M. Yarusinsky, 2000: Abrupt onset and termination of the African Humid Period: rapid climate responses to gradual insolation forcing. *Quaternary Science Reviews*, 19(1-5), pp. 347-361.

Derocher, A.E., N.J. Lunn, and I. Stirling, 2004: Polar bears in a warming climate. *Integrative and Comparative Biology*, 44(2), pp. 163-176.

Derocher, A.E., H. Wolkers, T. Colborn, M. Schlabach, T.S. Larsen, and O. Wiig, 2003: Contaminants in Svalbard polar bear samples archived since 1967 and possible population level effects. *Science of the Total Environment*, 301(1-3), pp. 163-174.

Diaz, H., and N. Graham, 1997: Monitoring climatic variability within ecoregions: a strategy for climate change detection. *Wengen-97 Workshop on Past, Present and Future Climate*

Variability and Extremes: The Impacts of Forests. Wengen, Switzerland, pp. 30.

Diaz, H., and V. Markgraf, 1993: *El Niño: historical and paleoclimatic aspects of the Southern Oscillation.* Cambridge University Press, Cambridge, 476pp.

Dirnbock, T., S. Dullinger, and G. Grabherr, 2003: A regional impact assessment of climate and land-use change on alpine vegetation. *Journal Of Biogeography*, 30(3), pp. 401-417.

Douglas, A.E., 2003: Coral bleaching–how and why? *Mar. Pollut. Bull.*, 46, pp. 385-392.

Drake, B.G., M.A. Gonzàlez-Meler, and S.P. Long, 1997: More efficient plants: a consequence of rising atmospheric CO2? *Annual Review of Plant Physiology and Plant Molecular Biology*, 48, pp. 609-639.

Dube, O.P., and R.M. Kwerepe, 2000: Human induced change in the Kgalagadi sands: Beyond the year 2000. In: *Towards sustainable management in the Kalahari region - Some essential background and critical issues. Proceedings of the Botswana Global Change Committee-START Kalahari Transect Meeting held in Gaborone, 1998.* Directorate of Research and Development, University of Botswana, Gaborone, pp. 244-258.

Dube, O.P., and G. Pickup, 2001: Effects of rainfall variability and commnal and semi-commercial grazing on land cover in southern Africa rangelands. *Climate Research*, pp. 195-208.

Dukes, J.S., N.R. Chiariello, E.E. Cleland, L.A. Moore, M.R. Shaw, S. Thayer, T. Tobeck, H.A. Mooney, and C.B. Field, 2005: Responses of Grassland Production to Single and Multiple Global Environmental Changes. *PLoS Biology*, 3(10)

Dukes, J.S., and H. Mooney, 1999a: Does global change increase the success of biological invaders? *Trends in Ecology and Evolution*, 14, pp. 135-139.

Dukes, J.S., and H.A. Mooney, 1999b: Does global change increase the success of biological invaders? *Trends in Ecology and Evolution*, 14(4), pp. 135-139.

Dunne, R.P., and B.E. Brown, 2001: The influence of solar radiation on bleaching of shallow water reef corals in the Andaman Sea, 1993-1998. *Coral reefs*, 20, pp. 201-210.

Eastman, J.L., M.B. Coughenour, and R.A. Pielke, 2001: The regional effects of CO2 and landscape change using a coupled plant and meteorological model. *Global Change Biology*, 7(7), pp. 797.

Edwards, M., and A.J. Richardson, 2004: Impact of climate change on marine pelagic phenology and trophic mismatch. *Nature (London)*, 430(7002), pp. 881-884.

Eidg. Forschungsanstalt WSL, B.f.U., Wald und Landschaft BUWAL, 2001: *Lothar: Der Orkan 1999.* WSL, BUWAL, Birmensdorf, Bern.

Ellis, W.N., J.H. Donner, and J.H. Kuchlein, 1997: Recent shifts in phenology of Microlepidoptera, related to climatic change (Lepidoptera). *Entomologische Berichten, Amsterdam*, 57(4), pp. 66-72.

Enquist, B.J., E.P. Economo, T.E. Huxman, A.P. Allen, D.D. Ignace, and J.F. Gillooly, 2003: Scaling metabolism from organisms to ecosystems. *Nature (London)*, 423(6940), pp. 639-642.

Erasmus, B.F.N., A.S. Van Jaarsveld, S.L. Chown, M. Kshatriya, and K.J. Wessels, 2002: Vulnerability of South African animal taxa to climate change. *Global Change Biology*, 8(7), pp. 679.

Erwin, T.L., 1988: The tropical forest canopy. The heart of biotic diversity. In: *Biodiversity*. Nationak Academy Press, Washington, D.C., pp. 123-129.

Everett, J.T., A. Krovnin, D. Lluch-Belda, E. Okemwa, H.A. Regier, and J.-P. Troadec, 1996: Fisheries. In: *Climate change 1995 - Impacts, adaptations and mitigation of climate change: scientific-technical analysis. Contribution of Working Group II to the Second Assessment Report of the Intergovernmental Panel of Climate Change.* [Watson, R., M.C. Zinyowera, and R.H. Moss (eds.)]. Cambridge University Press, Cambridge a.o., pp. 511-537.

Ewert, F., M.D.A. Rounsevell, I. Reginster, M.J. Metzger, and R. Leemans, 2005: Future scenarios of European agricultural land use: Estimating changes in crop productivity. *Agriculture Ecosystems & Environment*, 107, pp. 101-116.

Fasham, M.J.R., B.M. Baliño, and M.C. Biwkes (eds.), 2001: *A new vision of ocean biogeochemistry after a decade of the Joint Global Ocean Flux Study (JGOFS).* AMBIO, 4-31pp.

Fay, P.A., J.D. Carlisle, A.K. Knapp, J.M. Blair, and S.L. Collins, 2003: Productivity responses to altered rainfall patterns in a C4-dominated grassland. *Oecologia*, 137, pp. 245 -251.

Ferretti, D., E. Pendall, P.P. Tans, A.R. Mosier, J. Morgan, J.W. White, and J.A. Nelson, 2001: Evaporative Water Loss from Soil Reduced by Elevated Atmospheric CO2. In: *American Geophysical Union, Fall Meeting 2001*, pp. abstract #B12A-0107.

Fischlin, A., and D. Gyalistras, 1997: Assessing impacts of climatic change on forests in the Alps. *Global Ecology and Biogeography Letters*, 6, pp. 19-37.

Fischlin, A., D. Gyalistras, and C. Theato, 2007: Impacts of the summer 2003 heat wave on forests as assessed by simulation models. *Climate Dynamics*, pp. in prep.

Fitt, W.K., B.E. Brown, M.E. Warner, and R.P. Dunne, 2001: Coral bleaching: interpretation of thermal tolerance limits and thermal thresholds in tropical corals. *Coral Reefs*, 20, pp. 51-65.

Flannigan, M.D., B.J. Stocks, and B.M. Wotton, 2000: Climate change and forest fires. *Science of the Total Environment*, 262(3), pp. 221-229.

Flöder, S., and C.W. Burns, 2004: Phytoplankton diversity of shallow tidal lakes: influence of periodic salinity changes on diversity and species number of a natural assemblage. *Journal of Phycology*, 40(1), pp. 54-61.

Foley, J.A., M.H. Costa, C. Delire, N. Ramankutty, and P. Snyder, 2003: Green surprise? How terrestrial ecosystems could affect earth's climate. *Front Ecol Environ*, 1, pp. 38-44.

Foley, J.A., S. Levis, I.C. Prentice, D. Pollard, and S.L. Thompson, 1998: Coupling dynamic models of climate and vegetation. *Global Change Biology*, 4(5), pp. 561-579.

Fosberg, M.A., W. Cramer, V. Brovkin, R. Fleming, R. Gardner, A.M. Gill, J.G. Goldammer, R. Keane, P. Koehler, J. Lenihan, R. Neilson, S. Sitch, K. Thornicke, S. Venevski, M.G. Weber, and U. Wittenberg, 1999: Strategy for a fire module in dynamic global vegetation models. *International Journal of Wildland Fire*, 9(1), pp. 79-84.

Fowbert, J.A., and R.I.L. Smith, 1994: Rapid population increases in native vascular plants in the Argentine Islands, Antarctic Peninsula. *Arctic and Alpine Research*, 26(3), pp. 290-296.

Frei, C., C. Schär, D. Lüthi, and H. Davies, 1998: Heavy precipitation processes in a warmer climate. *Geophys. Res. Lett.*, 25, pp. 1431-1434.

Fried, J.S., M.S. Torn, and E. Mills, 2004: The Impact of Climate Change on Wildfire Severity: A Regional Forecast for Northern California. *Climatic Change*, 64(1 - 2), pp. 169-191.

Frolking, S., C. Li, R. Braswell, and J. Fuglestvedt, 2004: Short- and long-term greenhouse gas and radiative forcing impacts of changing water management in Asian rice paddies. *Global Change Biology*, 10, pp. 1180-1196.

Fuhlendorf, S.D., D.D. Briske, and F.E. Smeins, 2001: Herbaceous vegetation change in variable rangeland environments: The relative contribution of grazing and climatic variability. *Applied Vegetation Science*, 4(2), pp. 177-188.

Galbraith, H., R. Jones, R. Park, J. Clough, S. Herrod-Julius, B. Harrington, and G. Page, 2002: Global climate change and sea level rise: potential losses of intertidal habitats for shorebirds. *Waterbirds*, 25, pp. 173-183.

Gardner, T.A., Côté IM, Gill GA, Grant A, Watkinson AR, 2003, Long-term region wide declines in Caribbean coral reefs. Science, 301, 958-960.

Gattuso, J.P., Frankignoulle M, Bourge I, 1998, Effect of calcium carbonate saturation of seawater on coral calcification. Global Planet Change, 18, 37–46.

Gerten, D., and R. Adrian, 2002: Species-specific changes in the phenology and peak abundance of freshwater copepods in response to warm summers. *Freshwater Biology,* 47, pp. 2163-2173.

Gitay, H., S. Brown, W. Easterling, and B. Jallow, 2001: Ecosystems and their goods and services. In: *Climate Change 2001 - Impacts, Adaptation, and Vulnerability. Contribution of Working*

Group II to the Third Assessment Report of the Intergovernmental Panel of Climate Change (IPCC) [McCarthy, J.J., O.F. Canziani, N.A. Leary, D.J. Dokken, and K.S. White (eds.)]. Cambridge University Press, Cambridge a.o., pp. 237-342.

Gitay, H., A. Suárez, R.T. Watson, and D.J. Dokken (eds.), 2002: *Climate change and biodiversity*. Intergovernmental Panel on Climate Change, 77pp.

Gitz, V., 2003: Amplifying effects of land use change on future atmospheric C02 levels. *Global biogeochemical cycles*, 17(1), pp. 1-15.

Giupponi, C., M. Ramanzin, E. Strararo, and S. Fuser, 2007: Land use change, biodiversity and agricultural policy in the Belluno province, Italy. *Environment Science and Policy*, pp. (submitted).

Glynn, P.W., 1985, El Niño -associated disturbance to coral reefs and post disturbance mortality by *Acanthaster plancii*. Mar. Ecol. Prog. Ser., 26, 295-300.

Glynn, P.W., 1990, Feeding ecology of selected coral-reef macroconsumers: patterns and effects on coral community structure. In : Dubinsky, Z., Coral Reefs. Ecosystems of the world, 25. Elsevier, Amsterdam, 365-400.

Glynn, P.W., 1996, Coral reef bleaching: facts, hypothesis and implications. Global Change Biol 2:495-509

Gobron, N., B. Pinty, F. Melin, M. Taberner, M.M. Verstraete, A. Belward, T. Lavergne, and J.L. Widlowski, 2005: The state of vegetation in Europe following the 2003 drought. *International Journal of remote sensing*, 26(9), pp. 2013-2020.

Goldammer, J.G., 1999: Forests on fire. *Science*, 284(5421), pp. 1782-1783.

Goldstein, A.H., N.E. Hultman, J.M. Fracheboud, M.R. Bauer, J.A. Panek, M. Xu, Y. Qi, A.B. Guenther, and W. Baugh, 2000: Effects of climate variability on the carbon dioxide, water, and sensible heat fluxes above a ponderosa pine plantation in the Sierra Nevada (CA). *Agricultural and Forest Meteorology*, 101(2-3), pp. 113-129.

Gopal, B., and M. Chauhan, 2001: South Asian wetlands and their biodiversity: The role of monsoons. In: *Biodiversity in Weylands: Assessment, Function and Conservation*. Backhuys Publishers, Leiden, The Netherlands, pp. 257-275.

Gopal, B., and W.J. Junk, 2000: Biodiversity in wetlands: An introduction. In: *Biodiversity in Wetlands: Assessment, Function and Conservation.* Backhuys Publishers, Leiden, The Netherlands, pp. 1-10.

Goreau, T.J., Hayes RL, 1994, Coral bleaching and “ocean hot spots”. Ambio 23:176-180

Goyette, S., O. Brasseur, and M. Beniston, 2003: Application of a new wind gust parameterization: Multiscale case studies performed with the Canadian regional climate model. *Journal of Geophysical Research (Atmosphere)*, 108(D13), pp. art. no.-4374.

Graham, C.H., S. Ferrier, F. Huettmann, C. Moritz, and A.T. Peterson, 2004: New developments in museum-based informatics and applications in biodiversity analysis. *Trends in Ecology and Evolution*, 19(9), pp. 497-503.

Graham, R.W., and E.C. Grimm, 1990: Effects of global climate change on the patterns of terrestrial biological communities. *Trends in Ecology and Evolution*, 5(9), pp. 289-292.

Graham, T.B., 2003: *Climate change and ephemeral pool ecosystems: potholes and vernal pools as potential indicator systems*.

Guinotte, J.M., R. W. Buddemeier and J. A. Kleypas, 2003: *Future coral reef habitat marginality: temporal and spatial effects of climate change in the Pacific basin*. *Coral Reefs*. 22, pp. 551-558.

Guisan, A., and J.-P. Theurillat, 2000: Equilibrium modelling of alpine plant distribution: how far can we go? *Phytocoenologia*, 30(3-4), pp. 353-384.

Guisan, A., and W. Thuiller, 2005: Predicting species distribution: offering more than simple habitat models. *Ecology Letters*, in press

Guo, Q.F., 2000: Climate change and biodiversity conservation in Great Plains agroecosystems.

Global Environmental Change-Human And Policy Dimensions, 10(4), pp. 289-298.

Gyalistras, D., 2003: Development and validation of a high-resolution monthly gridded temperature and precipitation data set for switzerland (1951-2000). *Climate Research*, 25(1), pp. 55-83.

Gyalistras, D., and A. Fischlin, 1999: Towards a general method to construct regional climatic scenarios for model-based impacts assessments. *Petermanns geographische Mitteilungen*, 143(4), pp. 251-264.

Gyalistras, D., H. von Storch, A. Fischlin, and M. Beniston, 1994: Linking GCM-simulated climatic changes to ecosystem models: case studies of statistical downscaling in the Alps. *Climate Research*, 4(3), pp. 167-189.

Halloy, S.R.P., and A.F. Mark, 2003: Climate change effects on alpine plant biodiversity: A New Zealand perspective on quantifying the threat. *Arctic antarctic and Alpine Research*, 35, pp. 248-254.

Hamerlynck, E., T. Huxman, T. Charlet, and S. Smith, 2002: Effects of elevated CO2 (FACE) on the functional ecology of the drought-deciduous Mojave Desert shrub, Lycium andersonii. *Environ Exper Bot*, 48, pp. 93-106.

Hamerlynck, E.P., T.E. Huxman, R.S. Nowak, S. Redar, M.E. Loik, D.N. Jordan, S.F. Zitzer, J.S. Coleman, J.R. Seeman, and S.D. Smith, 2000: Photosynthetic responses of Larrea tridentata to a step-increase in atmospheric CO2 at the Nevada Desert FACE Facility. *Journal of Arid Environments*, 44, pp. 425-436.

Hampton, J., 2002: *Stock assessment of skipjack tuna in the Western and Central Pacific Ocean. 15th SCTB, Hawaii.* Oceanic Fisheries Programme, Secretariat of the Pacific Community, Noumea, New Caledonia, 36 pppp.

Hansen, A.J., R.P. Neilson, V.H. Dale, C.H. Flather, L.R. Iverson, D.J. Currie, S. Shafer, R. Cook, and P.J. Bartlein, 2001: Global change in forests: Responses of species, communities, and biomes. *Bioscience*, 51(9), pp. 765-779.

Harden, J.W., S.E. Trumbore, B.J. Stocks, A. Hirsch, S.T. Gower, K.P. O'Neill, and E.S. Kasischke, 2000: The role of fire in the boreal carbon budget. *Global Change Biology*, 6, pp. 174-184.

Hardy, J.T., 2003: *Climate change. causes, effects and solutions.* John Wiley and Sons Ltd., San Francisco., 247pp.

Hare, W., 2003: *Assessment of Knowledge on Impacts of Climate Change - Contribution to the Specification of Articale 2 on the UNFCCC.* WGBU, Berlin.

Hare, W., 2005: Relationship between increases in global mean temperature and impacts on ecosystems, food production, water and socio-economic systems. In: *Avoiding Dangerous Climate Change*, Exeter, U.K., pp. in press.

Harou, P.A., 2002: What is the role of markets in altering the sensitivity of arid land systems to perturbation. In: *Global desertification. Do humans cause deserts?* [Reynolds, J.F., and D.M. Stafford-Smith (eds.)]. Dahlem University Press, Berlin, Germany, pp. 253-274.

Harrison, P.A., P.M. Berry, N. Butt, and M. New, 2007: Modeling climate change impacts on species' distributions at the European scale: implications for conservation policy. pp. (submitted).

Harrison, S.P., and A.I. Prentice, 2003: Climate and CO2 controls on global vegetation distribution at the last glacial maximum: analysis based on palaeovegetation data, biome modelling and palaeoclimate simulations. *Global Change Biology*, 9(7), pp. 983-1004.

Hartig, E.K., A.S. Kolker, and V. Gornit, 2000: *Climate change impacts on saltmarsh morphology in Jamaica Bay, New York City. International Peat Congress, session 124. Peatland Geochemistry.* Quebec, Canada.

Hassol, S.J. (ed.), 2004: *Impacts of a warming Arctic: Arctic climate impact assessment*. Cambridge Univ. Press, Cambridge, UK, 146pp.

Hastings, S.J., W.C. Oechel, and A. Muhlia-Melo, 2005: Diurnal, seasonal and annual variation in the net ecosystem CO2 exchange of a desert shrub community (Sarcocaulescent) in Baja

California, Mexico. *Global Change Biology*, 11(6), pp. 927-939.

Hattenschwiler, S., and C. Korner, 2003: Does elevated CO2 facilitate naturalization of the non-indigenous Prunus laurocerasus in Swiss temperate forests? *Functional Ecology*, 17(6), pp. 778-785.

Helmuth, B., C.D.G. Harley, P.M. Halpin, M. O'Donnell, G.E. Hofmann, and C.A. Blanchette, 2002: Climate change and latitudinal patterns of intertidal thermal stress. *Science*, 298, pp. 1015-1017.

Henderson, G.M., 2005: Coral clues to rapid sea-level change. *Science*, 308, pp. 361-362.

Hetherington, A.M., and F.I. Woodward, 2003: The role of stomata in sensing and driving environmental change. *Nature*, 424(6951), pp. 901.

Hiernaux, P., and M.D. Turner, 2002: The influence of farmer and Pastoral Management Practices on Desertification processes in the Sahel. In: *Global desertification. Do humans cause deserts?* [Reynolds, J.F., and D.M. Stafford-Smith (eds.)]. Dahlem University Press, Berlin, Germany.

Hilbert, D.W., M. Bradford, T. Parker, and D.A. Westcott, 2003: Golden Bowerbird (Prionodura newtonia) habitat in past, present, and future climates: predicted extinction of a vertebrate in tropical highlands due to global warming. *Biological Conservation*, in press

Hilbert, D.W., B. Ostendorf, and M.S. Hopkins, 2001: Sensitivity of tropical forests to climate change in the humid tropics of north Queensland. *Austral Ecology*, 26(6), pp. 590-603.

Hoegh-Guldberg, O., 1999a: Climate change, coral bleaching and the future of the world's coral reefs. *Marine and Freshwater Research*, 50(839-866)

Hoegh-Guldberg, O., 1999b: Climate change, coral bleaching and the future of the world's coral reefs. 50, 839–66. *Mar. Freshwater Res.*, 50, pp. 839-866.

Hoegh-Guldberg, O. 1999, Climate change, coral bleaching and the future of the world's coral reefs. Marine and Freshwater Ecology 50: 839-866

Hoegh-Guldberg, H., Hoegh-Guldberg O, 2004, The Implication of Climate Change for Australia Great Barrier Reef. WWF Australia, Ultimo NSW, Australia, 345 p., www.reefbase.org

Hoegh-Guldberg, O., Hoegh-Guldberg H, Stout D.K., and Cesar H. Timmerman A., 2000, Pacific in peril. Biological, economic and social impacts of climate change on Pacific coral reefs. Greenpeace ISBN 1876221100, www.reefbase.org

Hoelzmann, P., D. Jolly, S.P. Harrison, F. Laarif, R. Bonnefille, and H.J. Pachur, 1998: Mid-Holocene land-surface conditions in northern Africa and the Arabian Peninsula: A data set for the analysis of biogeophysical feedbacks in the climate system. *Global Biogeochemical Cycles*, 12(1), pp. 35-51.

Hoffmann, W.A., and R.B. Jackson, 2000: Vegetation-Climate Feedbacks in the Conversion of Tropical Savanna to Grassland. *Journal of Climate*, 13(9), pp. 1593-1602.

Hogg, I.D., and D.D. Williams, 1996: Response of stream invertebrates to a global warming thermal regime. *Ecology*, 77, pp. 395-407.

Holman, I.P., M.D.A. Rounsevell, S. Shackley, P.A. Harrison, R.J. Nicholas, P.M. Berry, and E. Audsley, 2004a: A regional, multi-sectoral and integrated assessment of the impacts of climate and socio-economic change in the UK Part I: methodology. *Climatic Change*?), pp. 1-3.

Holman, I.P., M.D.A. Rounsevell, S. Shackley, P.A. Harrison, R.J. Nicholas, P.M. Berry, and E. Audsley, 2004b: A regional, multi-sectoral and integrated assessment of the impacts of climate and socio-economic change in the UK Part II: results. *Climatic Change*, pp. 1-31.

Holmgren, M., and M. Scheffer, 2001: El Nino as a window of opportunity for the restoration of degraded arid ecosystems. *Ecosystems*, 4, pp. 151-159.

Holmgren, M., M. Scheffer, E. Ezcurra, J.R. Gutierrez, and G.M.J. Mohren, 2001: Elnino effects on the dynamics of terresrial ecosystems. *Trends in Ecology and evolution*, 16(2), pp. 89-94.

Houghton, J.T., Y. Ding, D.J. Griggs, M. Noguer, P.J. van der Linden, X. Dai, K. Maskell, and C.A. Johnson (eds.), 2001a: *Climate Change 2001: The Scientific Basis*. Cambridge University Press, 881pp.

Houghton, J.T., Y. Ding, D.J. Griggs, M. Noguer, P.J. van der Linden, X. Dai, K. Maskell, and C.A. Johnson (eds.), 2001b: *Climate change 2001: The scientific basis. Contribution of Working Group I to the Third Assessment Report of the Intergovernmental Panel of Climate Change (IPCC)*. Cambridge University Press, Cambridge a.o., 881pp.

Hughes, T.P. and 16 other authors, 2003, Climate Change, Human Impacts, and the Resilience of coral Reefs. Science, 301, 929-933.

Hughes, L., E.M. Cawsey, and M. Westoby, 1996: Climatic range sizes of Eucalyptus species in relation to future climate change. *Global Ecology and Biogeography Letters*, 5, pp. 23-29.

Hughes, T.P., A.H. Baird, D.R. Bellwood, M. Card, S.R. Connolly, C. Folke, R. Grosberg, O. Hoegh-Guldberg, J.B.C. Jackson, J. Kleypas, J.M. Lough, P. Marshall, M. Nystro, S.R. Palumbi, J.M. Pandolfi, B. Rosen, and J. Roughgarden, 2003: Climate change, human impacts, and the resilience of coral reefs. *Science*, 301, pp. 929-933.

Hulme, M., 2001: Climatic perspectives on Sahelian desiccation: 1973-1998. *Global Environmental Change*, 11, pp. 19-29.

Hurrell, J., and H. van Loon, 1997: Decadal variations in climate associated with the North Atlantic Oscillation. *Climatic Change*, 36, pp. 301-326.

Iverson, L.R., M.W. Schwartz, and A. Prasad, 2004: How fast and far might tree species migrate in the eastern United States due to climate change? *Global Ecology & Biogeography*, 13, pp. 209-219.

Jackson, D.A., and N.E. Mandrak, 2002: Changing fish biodiversity: predicting the loss of cyprinid biodiversity due to global climate change. In: *Fisheries in a Changing Climate, American Fisheries Society Symposium 32*. American Fisheries Society, Bethesda MD., pp. 89-98.

Jacobs, A.F.G., B.G. Heusinkveld, S.M. Berkowicz, and, 2000: Dew measurements along a longitudinal sand dune transect,Negev Desert, Israel. *Int. J Biometeorol*, 43, pp. 184-190.

Jain, R., 2003: Sustainable development: differing views and policy alternatives. *Clean Technologies and Environmental Policy*, 4(4), pp. 197-198.

Jansson, R., 2003: Global patterns in endemism explained by past climatic change. *Proceedings of the Royal Society London B*, 270, pp. 583-590.

Jenkins, M., 2003: Prospects for biodiversity. *Science*, 302(5648), pp. 1175-1177.

Jensen, M.N., 2003: Climate change - Consensus on ecological impacts remains elusive. *Science,* 299(5603), pp. 38-38.

Jompa, J., and L.J. McCook, 2003a: Contrasting effects of turf algae on corals: massive Porites spp. are unaffected by mixed-species turfs, but killed by the red alga Anotrichium tenue. *Mar. Ecol. Prog. Ser.*, 258, pp. 79-86.

Jompa, J., and L.J. McCook, 2003b: Coral-algal competition: macroalgae with different properties have different effects on corals. *Mar. Ecol. Prog. Ser.*, 258, pp. 87-95.

Jones, M.B., and A. Donnelly, 2004: Carbon sequestration in temperate grassland ecosystems and the influence of management, climat and evevated CO2. *Ney Phytologist*, 164, pp. 423-439.

Kaas, E., and STOWASUS Group, 2001: *Regional storm, wave and surge scenarios for the 2100 century (STOWASUS-2100). Final Report.*

Kankaanpää, S., and T. Carter, 2004: *Construction of European forest land use scenarios for the 21st century.* The Finnish Environment, Helsinki.

Karlsson, J., A. Jonsson, and M. Jansson, 2005: Productivity of high latitude lakes: climate effect inferred from altitude gradient. *Global Change Biology*, 11, pp. 710-715.

Kasischke, E.S., N.L. Christensen Jr., and B.J. Stocks, 1995: Fire, global warming, and the carbon balance of boreal forests. *Ecological Applications*, 5, pp. 437-451.

Kasischke, E.S., and J.E. Penner, 2004: Improving global estimates of atmospheric emissions from biomass burning. *Journal of Geophysical Research (Atmosphere),* 109(D14)

Kasischke, E.S., and B.J. Stocks (eds.), 2000: *Fire, climate change and carbon cycling in the boreal forest*. Springer-Verlag, New York, 461pp.

Kates, R.W., 2000: Cautionary tales: Adaptation and the global poor. *Climate Change,* 45(1), pp. 5-17.

Keeley, J.E., 2002: Fire management of California shrubland landscapes. *Environmental Management*, 29(3), pp. 395-408.

Keleher, C.J., and F.J. Rahel, 1996: Thermal limits to salmonid distributions in the Rocky Mountain region and potential habitat loss due to global warming: A geographic information system (GIS) approach. *Transactions of the American Fisheries Society*, 125, pp. 1-13.

Keller, A.A., and G. Klein-MacPhee, 2000: Impact of elevated temperature on the growth, survival, and trophic dynamics of winter flounder larvae: a mesocosm study. *Canadian Journal of Fisheries and Aquatic Science,* 57, pp. 2382-2392.

Keller, F., S. Goyette, and M. Beniston, 2005: Sensitivity analysis of snow cover to climate change scenarios and their impact on plant habitats in alpine terrain. *Climatic Change,* in press

Keller, J.K., J.R. White, S.D. Bridgham, and J. Pastor, 2004: Climate change effects on carbon and nitrogen mineralization in peatlands through changes in soil quality. *Global Change Biology,* 10(7), pp. 1053-1064.

Kennedy, V.S., R.R. Twilley, J.A. Kleypas, C. J.H., and S.R. Hare, 2002: *Coastal and Marine Ecosystems & Global Climate Change: Potential Effects on U.S. Resources.* Pew Center on Global Climate Change, Arlington, USA, 52pp.

Kennett, S.A., 2002: FORUM National Policies for Biosphere Greenhouse Gas Management: Issues and Opportunities. *Environmental Management,* 30(5), pp. 595-608.

Kerr, J., and L. Packer, 1998: The impact of climate change on mammal diversity in Canada.

King, J.A., R.I. Bradley, R. Harrison, and A.D. Carter, 2004a: Carbon sequestration and saving potential associated with changes to the management of agricultural soils in England. *Soil Use And Management*, 20(4), pp. 394-402.

King, J.Y., A.R. Mosier, J.A. Morgan, D.R. Lecain, D.G. Milchunas, and W.J. Parton, 2004b: Plant nitrogen dynamics in shortgrass steppe under elevated atmospheric carbon dioxide. *Ecosystems*, 7(2), pp. 147-160.

Kirschbaum, M., and A. Fischlin, 1996: Climate change impacts on forests. In: *Climate change 1995 - Impacts, adaptations and mitigation of climate change: scientific-technical analysis. Contribution of Working Group II to the Second Assessment Report of the Intergovernmental Panel of Climate Change.* [Watson, R., M.C. Zinyowera, and R.H. Moss (eds.)]. Cambridge University Press, Cambridge a.o., pp. 95-129.

Kitayama, K., 1996: Climate of the summit region of Mount Kinabalu (Borneo) in 1992, an El Niño year. *Mountain Research and Development,* 16, pp. 65-75.

Kleypas, J.A., R.W. Buddemeier, and J.P. Gattuso, 2001: The future of coral reefs in an age of global change. *Int. J. Earth Sciences*, 90, pp. 426-437.

Kleypas, J.A., and C. Langdon, 2002: Overview of CO2-induced changes in seawater chemistry. In: *Proc. 9th Int. Coral Reef Sym.,* 23-27 Oct. 2000, Bali, Indonesia, pp. 1085-1089.

Kling, J., K. Hayhoe, L.B. Johnsoin, J.J. Magnuson, S. Polasky, S.K. Robinson, B.J. Shuter, M.M. Wander, D.J. Wuebbles, and D.R. Zak, 2003: *Confronting climate change in the Great Lakes region: Impacts on our communities and ecosystems.* Union of Concerned Scientists and the Ecological Society of America, Cambridge, MA, USA and Washington, D.C., 92pp.

Koop, K., D. Booth, A. Broadbent, J. Brodie, D. Bucher, D. Capone, J. Coll, W. Dennison, M. Erdmann, P. Harrison, O. Hoegh-Guldberg, P. Hutchings, G.B. Jones, A.W. Larkum, J. O'Neil, A. Steven, E. Tentori, S. Ward, J. Williamson, and D. Yellowlees, 2001a: ENCORE: the effect of nutrient enrichment on coral reefs. Synthesis of results and conclusions. *Mar. Pollut. Bull.,* 42(2), pp. 91-120.

Koop, K., A. Steven, R. McGill, E. Drews, and B. MacDonalds, 2001b: Use of a telemetered dispensing system for controlling nutrient additions to experimental patch reefs in the ENCORE study at one tree island, Great Barrier Reef, Australia. *Mar. Pollut. Bull.*, 42, pp.

121–126.

Koren, I., Y.J. Kaufman, L.A. Remer, and J.V. Martins, 2004: Measurement of the Effect of Amazon Smoke on Inhibition of Cloud Formation. *Science*, 303(5662), pp. 1342-1345.

Korner, C., 2003: Ecological impacts of atmospheric CO2 enrichment on terrestrial ecosystems. *Philosophical Transactions of the Royal Society of London - Series A*, 361(1810), pp. 2023-2041.

Korner, C., 2004: Through enhanced tree dynamics carbon dioxide enrichment may cause tropical forests to lose carbon. *Philosophical Transactions of the Royal Society of London - Series A*, 359(1443), pp. 493-498.

Kurc, S.A., and E.E. Small, 2004: Dynamics of evapotranspiration in semiarid grassland and shrubland ecosystems during the summer monsoon season, central New Mexico. *Water Resources Research*, 40(9)

Kurz, W.A., and M.J. Apps, 1999: A 70-year retrospective analysis of carbon fluxes in the Canadian Forest Sector. *Ecological Applications*, 9(2), pp. 526-547.

Körner, C., 1995: Towards a better experimental basis for upscaling plant responses to elevated CO2 and climate warming. *Plant, Cell and Environment*, 18, pp. 1101-1110.

Körner, C., 1998: Worldwide positions of alpine treelines and their causes. *Lecture Notes in Earth Sciences*, 74, pp. 221-229.

Körner, C., 1999: *Alpine plant life - functional plant ecology of high mountain ecosystems.* Springer, Berlin, 343pp.

Körner, C., D. Sarrus, and D. Christodoulakis, 2005: Long-term increase in dryness in the East-Mediterranean evidenced for the island of Samos. *Regional Environmental Change*, in press

Lake, P.S., P. M.A., P. Biro, J. Cole, A.P. Covich, C. Dahm, J. Gibert, W. Goedkoop, K. Martens, and J. Verhoeven, 2000: Global change and the biodiversity of freshwater ecosystems; Impacts on linkages between above-sediment and sediment biota. *Bioscience*, 50, pp. 1099-1107.

Lal, R., 2003: Offsetting global CO2 emissions by restoration of degraded soils and intensification of world agriculture and forestry. *Land Degradation & Development*, 14(3), pp. 309-322.

Lal, R., 2004: Soil carbon sequestration to mitigate climate change. *Geotherma*, 123, pp. 1-22.

Lambin, E.F., H.J. Geist, and E. Lepers, 2003: Dynamics of land-use and land-cover change in tropical regions. *Annual review of environmental resources*(28), pp. 205-41.

Lambin, E.F., B.L. Turner, H.J. Geist, S.B. Agbola, A. Angelsen, J.W. Bruce, O.T. Coomes, R. Dirzo, G. Fischer, C. Folke, P.S. George, K. Homewood, J. Imbernon, R. Leemans, X.B. Li, E.F. Moran, M. Mortimore, P.S. Ramakrishnan, J.F. Richards, H. Skanes, W. Steffen, G.D. Stone, U. Svedin, T.A. Veldkamp, C. Vogel, and J.C. Xu, 2001: The causes of land-use and land-cover change: moving beyond the myths. *Global Environmental Change-Human And Policy Dimensions*, 11(4), pp. 261-269.

Langmead, O. and Sheppard C, submitted, Effect of repeated climatic impacts on a coral community: a simulation study. Global Change Biology.

Lavoie, C., and S. Payette, 1994: Recent fluctuations of the lichen-spruce forest limit in subarctic Quebec. *Journal of Ecology*, 82(4), pp. 725-734.

Lavorel, S., 1999: Ecological diversity and resilience of Mediterranean vegetation to disturbance. *Diversity & Distributions*, 5(1-2), pp. 3-13.

Le Maitre, D.C., D.M. Richardson, and R.A. Chapman, 2004: Alien plant invasions in South Africa: driving forces and the human dimension. *South African Journal of Science*, 100(1), pp. 103-112.

Leathwick, J.R., and M.P. Austin, 2001: Competitive interactions between tree species in New Zealand's old-growth indigenous forests. *Ecology*, 82(9), pp. 2560-2573.

LeCain, D., J. Morgan, A. Mosier, J. Nelson, and, 2003: Soil and plant water relations, not photosynthetic pathway, primarily influence photosynthetic responses in a semi-arid ecosystem under elevated CO2. *Annals of Botany*, 92, pp. 41-52.

Leemans, R., and B. Eichhout, 2004: Another reason for concern: regional and global impacts on ecosystems for different levels of climate change. *Global Environmental Change*, 14, pp. 219-228.

Leemans, R., and B. Eickhout, 2003: *Analysing ecosystems at different levels of climate change.* OECD Working Party on Global and Structural Policies.

Leemans, R., and A. Kleidon, 2002: Regional and global assessment of the dimensions of desertification. In: *Global desertification. Do humans cause deserts?* [Reynolds, J.F., and D.M. Stafford-Smith (eds.)]. Dahlem University Press, Berlin, Germany, pp. 215-232.

Lehmkuhl, D.M., 1974: Thermal regime alterations and vital environmental physiological signals i n aquatic systems. In: *thermal Ecology* [Gibbons, J.W., and R.R. Sharitz (eds.)]. pp. 216-222.

Lehodey, P., F. Chai, and J. Hampton, 2003: Modelling climate-related variability of tuna populations from a coupled ocean-biogeochemical-populations dynamics model. *Fish. Oceanogr.*, 12, pp. 483-494.

Leibold, M.A., 1995: The niche concept revisited: mechanistic models and community context. *Ecology*, 76(5), pp. 1371-1382.

Lelieveld, J., H. Berresheim, S. Borrmann, P.J. Crutzen, F.J. Dentener, H. Fischer, J. Feichter, P.J. Flatau, J. Heland, R. Holzinger, R. Korrmann, M.G. Lawrence, Z. Levin, K.M. Markowicz, N. Mihalopoulos, A. Minikin, V. Ramanathan, M. de Reus, G.J. Roelofs, H.A. Scheeren, J. Sciare, H. Schlager, M. Schultz, P. Siegmund, B. Steil, E.G. Stephanou, P. Stier, M. Traub, C. Warneke, J. Williams, and H. Ziereis, 2002: Global Air Pollution Crossroads over the Mediterranean. *Science*, 298(5594), pp. 794-799.

Lenihan, J.M., R. Drapek, D. Bachelet, and R.P. Neilson, 2003: Climate change effects on vegetation distribution, carbon, and fire in California. *Ecological Applications*, 13(6), pp. 1667-1681.

Lepers, E., E.F. Lambin, A.C. Janetos, R. DeFries, F. Achard, N. Ramankutty, and R.J. Scholes, 2005: A synthesis of information on rapid land-cover change for the period 1981-2000. *Bioscience*, 55(2), pp. 115-124.

Leuzinger, S., G. Zotz, R. Asshoff, and C. Korner, 2005: Responses of deciduous forest trees to severe drought in Central Europe. *Tree Physiology*, 25(6), pp. 641-650.

Levy, P.E., M.G.R. Cannell, and A.D. Friend, 2004: Modelling the impacts of future changes in climate, CO2 concentration and land use on natural ecosystems and the terrestrial carbon sink. *Global Environmental Change*, 14, pp. 21-30.

Lie, E., H.J.S. Larsen, S. Larsen, G.M. Johansen, A.E. Derocher, N.J. Lunn, R.J. Norstrom, O. Wiig, and J.U. Skaare, 2005: Does high organochlorine (OC) exposure impair the resistance to infection in polar bears (Ursus maritimus)? Part II: Possible effect of OCs on mitogen- and antigen-induced lymphocyte proliferation. *Journal of Toxicology and Environmental Health - Part A*, 68(6), pp. 457-484.

Lie, E., H.J.S. Larsen, S. Larsen, G.M. Johnsen, A.E. Derocher, N.J. Lunn, R.J. Norstrom, O. Wiig, and J.U. Skaare, 2004: Does high organochlorine (OC) exposure impair the resistance to infection in polar bears (Ursus maritimus)? part 1: effect of OCs on the humoral immunity. *Journal of Toxicology and Environmental Health - Part A*, 67(7), pp. 555-582.

Lindroth, R.L., 2001: Consequences of elevated carbon dioxide and ozone for foliar chemical composition and dynamics in trembling aspen (*Populus tremuloides*) and paper birch (*Betula papyrifera*). *Environmental Pollution*, 115, pp. 395-304.

Lioubimtseva, E., and J.M. Adams, 2004: Possible implications of increased carbon dioxide levels and climate change for desert ecosystems. *Environmental Management*, 33(Supplement 1), pp. S388-S404.

Lischke, H., A.F. Lotter, and A. Fischlin, 2002: Untangling a Holocene pollen record with forest model simulations and independent climate data. *Ecological Modelling*, 150(1), pp. 1-21.

Llorens, L., J. Penuelas, and M. Estiarte, 2003: Ecophysiological responses of two Mediterranean shrubs, Erica multiflora and Globularia alypum, to experimentally drier and warmer conditions. *Physiologia Plantarum*, 119(2), pp. 231-243.

Llorens, L., J. Penuelas, M. Estiarte, and P. Bruna, 2004: Contrasting Growth Changes in Two Dominant Species of a Mediterranean Shrubland Submitted to Experimental Drought and Warming. *Ann Bot*, 94(6), pp. 843-853.

Llorens, L., and J. Peñuelas, 2005: Experimental evidence of future drier and warmer conditions affecting flowering of two co-occurring mediterranean shrubs. *International Journal of Plant Sciences*, 166(2), pp. 235-245.

Lodge, D.M., 1993: Biological invasions: Lessons for ecology. *Trends in Ecology and Evolution*, 8(4), pp. 133-137.

Loeb, V., V. Siegel, O. Holmhansen, R. Hewitt, W. Fraser, W. Trivelpiece, and S. Trivelpiece, 1997: Effects of sea-ice extent and krill or salp dominance on the antarctic food web. *Nature*, 387(6636), pp. 897-900.

Logan, J.A., J. Regniere, and J.A. Powell, 2003: Assessing the impacts of global warming on forest pest dynamics. *Frontiers in Ecology and the Environment*, 1(3), pp. 130-137.

Los, S.O., 1998: *Linkages between global vegetation and climate. An analysis based on NOAA Advanced Very High Resolution Radiometer Data*. PhD-thesis, Vrije Universiteit, Amsterdam, 179pp.

Loukos, H., P. Monfray, L. Bopp, and P. Lehodey, 2003: Potential changes in skipjack tuna (*Katsuwonus pelamis*) habitat form a global warming scenario: modelling approach and preliminary results. *Fish. Oceanogr.*, 12(4/5), pp. 474-482.

Lucht, W., I.C. Prentice, R.B. Myneni, S. Sitch, P. Friedlingstein, W. Cramer, P. Bousquet, W. Buermann, and B. Smith, 2002: Climatic control of the high-latitude vegetation greening trend and Pinatubo effect. *Science*, 296(5573), pp. 1687-1689.

Luckman, B., Mountain Environments in Changing Climates, M. Beniston, (ed.), R, London and New York., 1994: Using multiple high-resolution proxy climate records to reconstruct natural climate variability: an example from the Canadian Rockies. [Beniston, M. (ed.)]. Routledge Publishing Company, pp. 42-59.

Lui, G, Strong AE, 2003, Remote sensing of sea surface temperature during 2002 Barrier Reef coral bleaching. EOS 84:137-144.

Luo, Y., S. Wan, D. Hui, and L.L. Wallace, 2001: Acclimatization of soil respiration to warming in a tall grass prairie. 413(6856), pp. 622.

Luterbacher, J., D. Dietrich, E. Xoplaki, M. Grosjean, and H. Wanner, 2004: European seasonal and annual temperature variability, trends, and extremes since 1500. *Science*, 303(5663), pp. 1499-1503.

MacArthur, R., 1972: *Geographical Ecology*. Harper and Row, New York.

Macdonald, I.A.W., 2004: Recent research on alien plant invasions and their management in South Africa: A review of the inaugural research symposium of the Working for Water programme. *South African Journal of Science*, 100(1), pp. 21-26.

Mack, R.N., D. Simberloff, W.M. Lonsdale, H. Evans, M. Clout, and F. Bazzaz, 2000: Biotic invasions: Causes, epidemiology, global consequences and control. *Issues in Ecology*, 5, pp. 2-22.

Marra, J., C. Ho, and C.C. Trees, 2003: *An algorithm for the calculation of primary productivity from remote sensing data*. Lamont-Doherty Earth Obs., Palisades, N.Y., 27pp.

Martinez-Meyer, E., A.T. Peterson, and W.W. Hargrove, 2004: Ecological niches as stable distributional constraints on mammal species, with implications for Pleistocene extinctions and climate change projections for biodiversity. *Global Ecology and Biogeography*, 13(4), pp. 305-314.

Martinez-Vilalta, J., and J. Pinol, 2002: Drought-induced mortality and hydraulic architecture in pine populations of the NE Iberian Peninsula. *Forest Ecology and Management*, 161(1-3), pp.

247-256.

Martinez-Vilalta, J., J. Pinol, and K. Beven, 2002: A hydraulic model to predict drought-induced mortality in woody plants: an application to climate change in the Mediterranean. *Ecological Modelling*, 155(2-3), pp. 127-147.

Maunder, M.N., 2002: *Status of skipjack tuna in the Eastern Pacific Ocean in 2001 and outlook for 2002.15th SCTB, Hawaii.* Oceanic Fisheries Programme, Secretariat of the Pacific Community, Noumea, New Caledonia, 52 pppp.

McCoock JL, Jompa J, Diaz-Pullido G, 2001, Competition between corals and algae on coral reefs: a review of evidence and mechanisms. Coral Reefs, 19, 400-425.

McClanahan TR, Muthiga NA, Mangi S, 2001, Coral and algaal changes after the 1998 coral bleaching: interaction with reef management and herbivores on Kenyan reefs. Coral Reefs, 19, 380-391.

McNeely, J., 1990: Climate change and biological diversity: policy implications. In: *Landscape-Ecological Impact of Climatic Change* [Boer, M., and R. de Groot (eds.)]. IOS Press, Amsterdam.

Meehl, G.A., and C. Tebaldi, 2004: More intense, more frequent, and longer lasting heat waves in the 21st century. *Science*, 305(5686), pp. 994-997.

Mendes JM, Woodley JD, Effect of the 1995-1996 bleaching event on polyp tissue depth, growth, reprouction and skeletal band formation in *Montastrea annularis*. Mar Ecol Prog Ser, 235, 93-102.

Metzger, M.J., R.G.H. Bunce, R.H.G. Jongman, C.A. Mücher, and J.W. Watkins, 2004: A statistical stratification of the environment of Europe. *Global Ecology and Biogeography (submitted)*

Metzger, M.J., M.D.A. Rounsevell, L. Acosta-Michlik, R. Leemans, and D. Schröter, 2007: The vulnerability of ecosystem services to land use change. pp. (submitted).

Midgley, G.F., L. Hannah, D. Millar, M.C. Rutherford, and L.W. Powrie, 2002: Assessing the vulnerability of species richness to anthropogenic climate change in a biodiversity hotspot. *Global Ecology and Biogeography*, 11(6), pp. 445-451.

Midgley, G.F., L. Hannah, D. Millar, W. Thuiller, and A. Booth, 2003: Developing regional and species-level assessments of climate change impacts on biodiversity in the Cape Floristic Region. *Biological Conservation*, 112(1-2), pp. 87-97.

Milchunas, D.G., J.A. A.R. Mosier, D.R.L. Morgan, J.Y.K. J.A., and J.A. Nelson, 2004: Root Production And Tissue Quality In A Shortgrass Steppe Exposed To Elevated CO2: Using A New Ingrowth Method. *Plant and Soil*, 0, pp. 1-12.

Millennium Ecosystem Assessment (ed.), 2005a: *Ecosystems and human well-being - Desertification synthesis*. Island Press, Washington, DC, 36pp.

Millennium Ecosystem Assessment (ed.), 2005b: *Ecosystems and human well-being - synthesis*. Island Press, Washington, DC, 155pp.

Mitsch, W.J., and J.G. Gosselink, 2000: *Wetlands*. John Wiley, New York, Third ed., xiii+919pp.

Moberg, A., x. xyz, x. xyz, and x. xyz, 2005: in press in Nature. *Nature*, pp. in press.

Molau, U., 1997: Responses to natural climatic variation and experimental warming in two tundra plant species with contrasting life forms: Cassiope tetragona and Ranunculus nivalis. *Global Change Biology*, 3 Suppl. 1, pp. 97-107.

Monahan, A.H., J.C. Fyfe, and G.M. Flato, 2000: A regime view of Northern Hemisphere atmospheric variability and change under global warming. *Geophys. Res. Lett.*, 27, pp. 1139-1142.

Montenegro, G., R. Ginochio, A. Segura, J.E. Keeley, and M. Gomez, 2004: Fire regimes and vegetation responses in two Mediterranean-climate regions. *Revista chilena de historia natural*, 77(3), pp. 455-464.

Morgan, J.A., D.R. Lecain, A.R. Mosier, and D.G. Milchunas, 2001a: Elevated CO2 enhances water relations and productivity and affects gas exchange in C3 and C4 grasses of the Colorado shortgrass steppe. *Global Change Biology*, 7(4), pp. 451.

Morgan, J.A., D.R. LeCain, A.R. Mosier, D.G. Milchunas, and., 2001b: Elevated CO2 enhances water relations and productivity and affects gas exchange in C3 and C4 grasses of the Colorado shortgrass steppe. *Global Change Biology*, 7, pp. 451-466.

Morgan, J.A., A.R. Mosier, D.G. Milchunas, D.R. LeCain, J.A. Nelson, and W.J. Parton, 2004a: CO2 enhances productivity but alters species composition and reduces forage quality in the Colorado shortgrass steppe. *Ecol. Applic.*, 14, pp. 208-219.

Morgan, J.A., D.E. Pataki, C. Körner, H. Clark, S.J. Del Grosso, J.M. Grünzweig, A.K. Knapp, A.R. Mosier, P.C.D. Newton, P.A. Niklaus, J.B. Nippert, R.S. Nowak, W.J. Parton, H.W. Polley, and M.R. Shaw, 2004b: Water relations in grassland and desert ecosystem responses exposed to elevated atmospheric CO2. *Oecologia*, 140, pp. 11-25.

Mosier, A.R., J.A. Morgan, J.Y. King, D. LeCain, and D.G. Milchunas, 2002: Soil-atmosphere exchange of CH₄, CO₂, NO_x, and N₂O in the Colorado shortgrass steppe under elevated CO₂. *Plant and Soil*, 240(2), pp. 201.

Mouillot, F., S. Rambal, and R. Joffre, 2002: Simulating climate change impacts on fire frequency and vegetation dynamics in a Mediterranean-type ecosystem. *Global Change Biology*, 8(5), pp. 423-437.

Mouillot, F., J.-P. Ratte, R. Joffre, J.M. Moreno, and S. Rambal, 2003: Some determinants of the spatio-temporal fire cycle in a mediterranean landscape (Corsica, France). *Landscape Ecology*, 18(7), pp. 665-674.

Myers, N., R.A. Mittermeier, C.G. Mittermeier, G.A.B. da Fonseca, and J. Kent, 2000: Biodiversity hotspots for conservation priorities. *Nature*, 403(6772), pp. 853-858.

Myneni, R.B., C.D. Keeling, C.J. Tucker, G. Asrar, and R.R. Nemani, 1997: Increased plant growth in the northern high latitudes from 1981 to1991. *Nature (London)*, 386(17.Apr.), pp. 698-702.

Nagel, J.M., T.E. Huxman, K.L. Griffin, and S.D. Smith, 2004: Co-2 enrichment reduces the energetic cost of biomass construction in an invasive desert grass. *Ecology*, 85(1), pp. 100-106.

Najjar, R.G., H.A. Walker, P.J. Anderson, E.J. Barron, R.J. Bord, J.R. Gibson, V.S. Kennedy, K. C.G., J.P. Megonigal, R.E. O'Connor, C.D. Polsky, N.P. Psuty, B.A. Richards, L.G. Sorenson, E.M. Steele, and R.S. Swanson, 2000: The potential impacts of climate chanege on the mid-Atlantic coastal region. *Climate Research*, 14, pp. 219-233.

Nakicenovic, N., J. Alcamo, G. Davis, B. de Vries, J. Fenhann, S. Gaffin, K. Gregory, A. Grübler, T.Y. Jung, T. Kram, E. Lebre la Rovere, L. Michaelis, S. Mori, T. Morita, W. Pepper, H. Pitcher, L. Price, K. Riahi, A. Roehrl, H.H. Rogner, A. Sankovski, M. Schlesinger, P. Shukla, S. Smith, R. Swart, S. van Rooijen, N. Victor, and Z. Dadi (eds.), 2000: *Emissions Scenarios - A special report of the Intergovernmental Panel on Climate Change (IPCC).* Cambridge University Press, Cambridge, 509pp.

National Research Council, 1999: *Perspectives on Biodiversity: Valuing its Role in an Everchanging World.* National Academy Press, Washington, D.C.

Naumburg, E., D.C. Housman, T.E. Huxman, T.N. Charlet, M.E. Loik, and S.D. Smith, 2003: Photosynthetic responses of Mojave Desert shrubs to free air CO2 enrichment are greatest during wet years. *Global Change Biol.*, 9, pp. 276-285.

Nelson, J.A., J.A. Morgan, D.R. LeCain, A.R. Mosier, D.G. Milchunas, P.B. A., and, 2003: Elevated CO2 increases soil moisture and enhances plant water relations in a long-term field study in semi-arid shortgrass steppe of Colorado. *Plant and Soil.*, 0, pp. 1-11.

Neumann, J.E., G. Nicholls, and M. Manion, 2000: *Sea-level rise & Global climate change: A Review of Impacts to U.S. Coasts.* Pew Center on Global Climate Change, Arlington, USA, 38 pppp.

Ni, J., 2000: A simulation of biomes on the Tibetan Plateau and their responses to global climate change. *Mountain Research and Development*, 20, pp. 80-89.

Ni, J., 2001: Carbon storage in terrestrial ecosystems of China: Estimates at different spatial scales.

Nicholls, R.J., F.M.J. Hoozemans, and M. Marchand, 1999: Increasing flood risk and wetland losses due to global sea-level rise: regional and global analysis. *Global Environmental Change - Human and Policy DImensions*, 9, pp. S69-S87.

Nicholson, S., 2000: Land surface processes and Sahel climate. *Reviews of Geophysics*, 38(1), pp. 117-139.

Nicholson, S.E., 2002: What are the key components of climate as a driver of desertification? In: *Global desertification. Do humans cause deserts?* [Reynolds, J.F., and D.M. Stafford-Smith (eds.)]. Dahlem University Press, Berlin, Germany, pp. 41-57.

Nnadozie, K.C., 1998: *Legal and policy approaches at linking the desertification and climate change agenda in Nigeria*, pp. 15.

Norby, R.J., and R.B. Jackson, 2000: Root dynamics and global change: seeking an ecosystem perspective. *New Phytol.*, 147, pp. 3-12.

Norby, R.J., and Y. Luo, 2004: Evaluating ecosystem responses to rising atmospheric CO2 and global warming in a multi-factor world. *New Phytologist*, 162(2), pp. 281.

Norstrom, R.J., S.E. Belikov, E.W. Born, G.W. Garner, B. Malone, S. Olpinski, M.A. Ramsay, S. Schliebe, I. Stirling, M.S. Stishov, M.K. Taylor, and O. Wiig, 1998: Chlorinated hydrocarbon contaminants in polar bears from eastern Russia, North America, Greenland, and Svalbard: Biomonitoring of Arctic pollution. *Archives of Environmental Contamination and Toxicology*, 35(2), pp. 354-367.

Notholt, J., Z. Kuang, C.P. Rinsland, G.C. Toon, M. Rex, N. Jones, T. Albrecht, H. Deckelmann, J. Krieg, C. Weinzierl, H. Bingemer, R. Weller, and O. Schrems, 2003: Enhanced Upper Tropical Tropospheric COS: Impact on the Stratospheric Aerosol Layer. *Science*, 300(5617), pp. 307-310.

O'Brien, K.L., and R.M. Leichenko, 2000: Double exposure: assessing the impacts of climate change within the context of economic globalization. *Global Environmental Change*, 10, pp. 221-232.

O'Reilly, C.M., S.R. Alin, P.-D. Plisnier, A.S. Cohen, and B.A. Mckee, 2003: Climate change decreases aquatic ecosystem productivity of Lake tanganyika, Africa. *Nature*, 424, pp. 766-768.

Ogaya, R., and J. Penuelas, 2003: Comparative seasonal gas exchange and chlorophyll fluorescence of two dominant woody species in a holm oak forest. *Flora*, 198(2), pp. 132.

Ogaya, R., and J. Penuelas, 2004: Phenological patterns of Quercus ilex, Phillyrea latifolia, and Arbutus unedo growing under a field experimental drought. *Ecoscience*, 11(3), pp. 263-270.

Ogden, J., L. Basher, and M. McGlone, 1998: Fire, forest regeneration and links with early human habitation: Evidence from New Zealand. *Annals of Botany*, 81(6), pp. 687-696.

Ogle, K., and J.F. Reynolds, 2004: Plant responses to precipitation in desert ecosystems: integrating functional types, pulses, thresholds, and delays. *Oecologia*, 141, pp. 282-294.

Ogutu, J.O., and N. Owen-Smith, 2003: ENSO, rainfall and temperature influences on extreme population declines among African savanna ungulates. *Ecology Letters*, 6(5), pp. 412-419.

Osborne, C.P., P.L. Mitchell, J.E. Sheehy, and F.I. Woodward, 2000: Modelling the recent historical impacts of atmospheric CO2 and climate change on Mediterranean vegetation. *Global Change Biology*, 6(4), pp. 445-458.

Osborne, T., K. Briffa, S. Tett, P. Jones, and R. Trigo, 1999: Evaluation of the North Atlantic Oscillation as simulated by a coupled climate model. *Climate Dynamics*, 15, pp. 685-702.

Ostrofsky, M.L., 1997: Relationship between chemical characteristics of autumn-shed leaves and aquatic processing rates. *Journal of the North American Benthological Society*, 16, pp. 750-759.

Parmesan, C., and H. Galbraith, 2004: *Observed impacts of global climate change in the U.S.* PEW Report, Pew Center on Global Climate Change, Washington DC., 56pp.

Parmesan, C., T.L. Root, and M.R. Willig, 2000: Impacts of extreme weather and climate on terrestrial biota. *Bulletin of the American Meteorological Society*, 81(3), pp. 443-450.

Parmesan, C., N. Ryrholm, C. Stefanescu, J.K. Hill, C.D. Thomas, H. Descimon, B. Huntley, L. Kaila, J. Kullberg, T. Tammaru, W.J. Tennent, J.A. Thomas, and M. Warren, 1999: Poleward shifts in geographical ranges of butterfly species associated with regional warming. *Nature*, 399(6736), pp. 579-583.

Parmesan, C., and G. Yohe, 2003: A globally coherent fingerprint of climate change impacts across natural systems. *Nature (London)*, 421(6918), pp. 37-42.

Pauly D, Christensen V, Guenette S, Pitcher TJ, Rashid Sumaila U, Walters CJ, 2002, Towards sustainability in world fisheries. Nature, 418, 689-695

Pausas, J.G., and D. Abdel Malak, 2004: Spatial and temporal patterns of fire and climate change in the eastern Iberian Peninsula (Mediterranean Basin). In: *Ecology, Conservation and Management of Mediterranean Climate Ecosystems of the World.10th International Conference on Mediterranean Climate Ecosystems* [Arianoutsou, M., and V.P. Papanastasis (eds.)]. Millpress, Rotterdam, Rhodes, Greece.

Pearson, R.G., and T.P. Dawson, 2003: Predicting the impacts of climate change on the distribution of species: Are bioclimate envelope models useful? *Global Ecology and Biogeography*, 12(5), pp. 361-371.

Pearson, R.G., and T.P. Dawson, 2005: Long-distance plant dispersal and habitat fragmentation: identifying conservation targets for landscape planning under climate change. *Biological Conservation*, 123, pp. 389-401.

Pearson, R.G., T.P. Dawson, P.M. Berry, and P.A. Harrison, 2002: SPECIES: A spatial valuation of climate impact on the envelope of species. *Ecological Modelling*, 154(3), pp. 289-6.

Pearson, R.G., T.P. Dawson, and C. Liu, 2004: Modelling species distributions in Britain: a hierarchical integration of climate and land-cover data. *Ecography*, 27, pp. 285-298.

Pêcheux M, 1998 in press, Review on coral reef bleaching. Accepted by Atoll Res. Bull., 214p., www.reefbase.org.

Pêcheux M, 2002, CO_2 increase, a direct cause of coral reef mass bleaching? Mar. Life, 12, 63-68.

Peck, L.S., K.E. Webb, and D.M. Bailey, 2004: Extreme sensitivity of biological function to temperature in Antarctic marine species. *Functional Ecology*, 15(5), pp. 625-630.

Peng, C., and M.J. Apps, 1999: Modelling the response of net primary productivity (NPP) of boreal forest ecosystems to changes in climate and fire disturbance regimes. *Ecological Modelling*, 122(3), pp. 175-193.

Peñuelas, J., and I. Filella, 2001a: Responses to a warming world. *Science*, 294(5543), pp. 743-795.

Peñuelas, J., and I. Filella, 2001b: Responses to a warming world. *Science*, 294, pp. 793-795.

Peñuelas, J., F. Lloret, and R. Montoya, 2001: Severe drought effects on Mediterranean woody flora in Spain. *Forest Science*, 47(2), pp. 214.

Peters, R.L., and J.D.S. Darling, 1985: The greenhouse effect and nature reserves - global warming would diminish biological diversity by causing extinctions among reserve species. *Bioscience*, 35(11), pp. 707-717.

Peterson, A.T., 2003: Projected climate change effects on Rocky Mountain and Great Plains birds: generalities of biodiversity consequences. *Global Change Biology*, 9(5), pp. 647-655.

Peterson, A.T., M.A. Ortega-Huerta, J. Bartley, V. Sanchez-Cordero, J. Soberon, R.H. Buddemeier, and D.R.B. Stockwell, 2002a: Future projections for Mexican faunas under global climate change scenarios. *Nature (London)*, 416(6881), pp. 626-629.

Peterson, A.T., M.A. Ortega-Huerta, J. Bartley, V. Sánchez-Cordero, J. Soberón, R.H. Buddemeier, and D.R.B. Stockwell, 2002b: Biodiversity consequences of climate change in Mexico.

Peterson, D., R. Rochefort, R. Little, and A. Woodward, 1994: Changes in sub-alpine tree distribution in western North America: a review of climatic and other causal factors. *The Holocene*, 4, pp. 89-100.

Petit, J.R., J. Jouzel, D. Raynaud, N.I. Barkov, J.-M. Barnola, I. Basile, M. Benders, J. Chappellaz, M. Davis, G. Delaygue, M. Delmotte, V.M. Kotlyakov, M. Legrand, V.Y. Lipenkov, C. Lorius, L. Pépin, C. Ritz, E. Saltzman, and M. Stievenard, 1999: Climate and atmospheric history of the past 420,000 years from the Vostok ice core, Antarctica. *Nature (London)*, 399(3.Jun.), pp. 429-436.

Poff, N.L., M.M. Brinson, and J.W.J. Day, 2002: *Aquatic ecosystems and global climate change: Potential impacts on inland freshwater and coastal wetland ecosystems in the United States.* Pew Center on Global Climate Change, Arlington, VA, USA, 44pp.

Polley, H.W., C.R. Tischler, H.B. Johnson, and J.D. Derner, 2002: Growth rate and survivorship of drought: CO2 effects on the presumed tradeoff in seedlings of five woody legumes. *Tree Physiology*, 22(6), pp. 383-391.

Poorter, H., 1993: Interspecific variation in the growth response of plants to an elevated CO2 concentration. *Vegetatio*, 104/105, pp. 77-97.

Poorter, H., and M. Perez-Soba, 2001: Growth response of plants to elevated CO2 under sub-optimal environmental conditions. *Oecologia*, 129, pp. 1-20.

Pouliquen-Young, O., and P. Newman, 1999: *The implications of climate change for land-based nature conservation strategies.* Australian Greenhouse Office, Environment Australia, Canberra Institute for Sustainability and Technology Policy, Murdoch University, Perth.

Pounds, J.A., 2001: Climate and amphibian declines. *Nature*, 410(6829), pp. 639-640.

Prentice, I.C., and D. Jolly, 2000: Mid-Holocene and glacial-maximum vegetation geography of the northern continents and Africa. *Journal of Biogeography*, 27(3), pp. 507-519.

Price, M., T. Kohler, and T. Wachs (eds.), 2000: *Mountains of the world: Mountain Forests and Sustainable Development*. Paul Haupt Publishers, Bern.

Prospero, J.M., and P.J. Lamb, 2003: African droughts and dust transport to the Caribbean: Climate change implications. *Science*, 302(5647), pp. 1024-1027.

Pulliam, H.R., 2000: On the relationship between niche and distribution. *Ecology Letters*, 3, pp. 349-361.

Pyne, S.J., 2001: The Fires This Time, and Next. *Science*, 294(5544), pp. 1005-1006.

Qureshi, A., and D. Hobbie, 1994: *Climate change in Asia.* Asian Development Bank, Manila.

Rahel, F.J., C.J. Keleher, and J.L. Anderson, 1996: Habitat loss and population fragmentation for coldwater fishes in the Rocky Mountain region in response to climate warming. *Limnology and Oceanography*, 41, pp. 1116-1123.

Rambal, S., J.-M. Ourcival, R. Joffre, F. Mouillot, Y. Nouvellon, M. Reichstein, and A. Rocheteau, 2003: Drought controls over conductance and assimilation of a Mediterranean evergreen ecosystem: scaling from leaf to canopy. *Global Change Biology*, 9(12), pp. 1813-1824.

Rappè, G., 2003: Mariene kustorganismen als bio-indicatoren van klimaatsveranderingen in de zuidelijke Noordzee. *De Levende Natuur*, 104(2), pp. 94-98.

Rebetez, M., 2004: Summer 2003 maximum and minimum daily temperatures over a 3300 m altitudinal range in the Alps. *Climate Research*, 27(1), pp. 45-50.

Reichstein, M., J.D. Tenhunen, O. Roupsard, J.M. Ourcival, S. Rambal, F. Miglietta, A. Peressotti, M. Pecchiari, G. Tirone, and R. Valentini, 2002: Severe drought effects on ecosystem CO2 and H2O fluxes at three Mediterranean evergreen sites: revision of current hypotheses? *Global Change Biology*, 8(10), pp. 999-1017.

Reynaud S, Leclercq N, Romaine-Lioud S, Ferrier-Pagès C, Jaubert J, Gattuso JP, 2003, Interacting effects of CO2 partial pressure and temperature on photosynthesis and calcification in a scleratinian coral. Global Change Biol 9:1-9

Reynolds, J.F., P.R. Kemp, K. Ogle, and R.J. Fernández, 2004: Modifying the pulse-reserve paradigm for deserts of North America: precipitation pulses, soil water, and plant responses. *Oecologia*, 141, pp. 194-210.

Reynolds, J.F., and D.M. Stafford-Smith, 2002: Do human cause deserts? In: *Global desertification.*

Do humans cause deserts? [Reynolds, J.F., and D.M. Stafford-Smith (eds.)]. Dahlem University Press, Berlin, Germany, pp. 1-21.

Reynolds, R.W., N.A. Rayner, T.M. Smith, D.C. Stokes, and W. Wang, 2002: An improved in situ and satellite SST analysis for climate. *J. Climate*, 15, pp. 1609-1625.

Richards, M., 2003: *Poverty Reduction, Equity and Climate Change: Global Governance Synergies or Contradictions? Globalisation and Poverty Programme* Overseas Development Institute.

Richardson, A.J., and D.S. Schoeman, 2004: Climate impact on plankton ecosystems in the Northeast Atlantic. *Science*, 305(5690), pp. 1609-1612.

Richardson, D.M., and B.W. van Wilgen, 2004: Invasive alien plants in South Africa: how well do we understand the ecological impacts? *South African Journal of Science*, 100(1), pp. 45-52.

RIVM, CBS, and S. DLO, 2003: *Natuurcompendium 2003: Natuur in cijfers.* KNNV Uitgeverij, Utrecht, 494pp.

Roberts, C.M., C.J. McClean, J.E.N. Veron, J.P. Hawkins, G.R. Allen, D.E. McAllister, C.G. Mittermeier, F.W. Schueler, M. Spalding, F. Wells, C. Vynne, and T.B. Werner, 2002: Marine biodiversity hotspots and conservation priorities for tropical reefs. *Science*, 295, pp. 1280-1284.

Robinson, A.D., 2004: Growth and abundance of desert annual in an arid woodland in Oman. *Plant Ecology*, 174, pp. 137-145.

Robinson G, 1985, Influence of the 1982-83 El Niño on Galapagos marine life. In: Robinson, G., Del Piño, E. M., El Niño en las Islas Galàpagos : el evento de 1982-1983. Fundaciòn Charles Darwin para las Islas Galàpagos, Quito, Ecuador, 153-190.

Roerink, G.J., M. Menenti, and W.S. Soepboer, Z., 2003: Assessment of climate impact on vegetation dynamics by using remote sensing. *Physics and Chemistry of The Earth*, 28(1-3), pp. 103-109.

Root, T.L., J.T. Price, K.R. Hall, S.H. Schneider, C. Rosenzweig, and J.A. Pounds, 2003: Fingerprints of global warming on wild animals and plants. *Nature (London)*, 421(6918), pp. 57-60.

Rounsevell, M., F. Ewert, I. Reginster, R. Leemans, and T. Carter, 2005: Future scenarios of European agricultural land use II projecting changes in cropland and grassland. *Agriculture Ecosystems & Environment*, 107, pp. 117-135.

Rounsevell, M.D.A., P.M. Berry, and P.A. Harrison, 2007a: Future environmental change impacts on rural land use and biological resources: a synthesis of the ACCELERATES project. pp. (submitted).

Rounsevell, M.D.A., I. Reginster, M.B. Araújo, T.R. Carter, N. Dendoncker, F. Ewert, J.I. House, S. Kankaanpää, R. Leemans, M.J. Metzger, C. Schmit, P. Smith, and G. Tuck, 2007b: A coherent set of future land use change scenarios for Europe. *Agriculture Ecosystems & Environment*, pp. (under review).

Rowan, R., 2004: Thermal adaptation in reef coral symbionts. *Nature*, 430, pp. 742.

Rustad, L.E., J.L. Campbell, G.M. Marion, R.J. Norby, M.J. Mitchell, A.E. Hartley, J.H.C. Cornelissen, and J. Gurevitch, 2001: A meta-analysis of the response of soil respiration, net nitrogen mineralization, and aboveground plant growth to experimental ecosystem warming. *Oecologia*, 126, pp. 543-562.

Rutherford, M.C., M.F. Midgeley, W.J. Bond, L.W. Powrie, C.F. Musil, R. Roberts, and J. Allsopp, 1999: *South African Country Study on Climate Change.* Terrestrial Plant Diversity Section, Vulnerability and Adaptation, Department of Environmental Affairs and Tourism, Pretoria.

Sabine, C.L., M. Heimann, P. Artaxo, D.C.E. Bakker, C.-T.A. Chen, C.B. Field, N. Gruber, C. Le Quere, R.G. Prinn, J.E. Richey, P.R. Lankao, J.A. Sathaye, and R. Valentini, 2004: Current status and past trends of the global carbon cycle. In: *The global carbon cycle: Integrating humans, climate and the natural world.* Island Press, Washington, pp. 17-44.

SADC ELMS, 1996: *Policy and strategy for environment an sustainable development. Towards equity-led growth and sustainable development in Southern Africa.*, Southern Africa

Development Community, 55pp.

Sala, O.E., F.S. Chapin, III, J.J. Armesto, E. Berlow, J. Bloomfield, R. Dirzo, E. Huber-Sannwald, L.F. Huenneke, R.B. Jackson, A. Kinzig, R. Leemans, D. Lodge, H.A. Mooney, M. Oesterheld, L.N. Poff, M.T. Sykes, B.H. Walker, M. Walker, and D. Wall, 2000a: Global biodiversity scenarios for the year 2100. *Science*, 287, pp. 1770-1774.

Sala, O.E., I.F.S. Chapin, J.J. Armesto, E. Berlow, J. Bloomfield, R. Dirzo, E. Huber Sanwald, L.F. Huenneke, R.B. Jackson, A. Kinzig, R. Leemans, D.H. Lodge, H.A. Mooney, M. Oesterheld, N. Leroy Poff, M.T. Sykes, B.H. Walker, M. Walker, and D.H. Wall, 2000b: Global biodiversity scenarios for the year 2100. *Science*, 287(5459), pp. 1770-1774.

Sardans, J., and J. Penuelas, 2004: Increasing drought decreases phosphorus availability in an evergreen Mediterranean forest. *Plant and Soil*, 267(1 - 2), pp. 367-377.

Sarmiento, J.L., R. Slater, R. Barber, L. Bopp, S.C. Doney, A.C. Hirst, J. Kleypas, R. Matear, U. Mikolajewicz, P. Monfray, V. Soldatov, S.A. Spall, and R. Stouffer, 2004: Response of ocean ecosystems to climate warming. *Global Biogeochemical Cycles*, 18(3), pp. GB3003 1-23.

Schallenberg, M., C.J. Hall, and C.W. Burns, 2001: Climate change alters zooplankton community structure and biodiversity in coastal wetlands.

Schar, C., and G. Jendritzky, 2004: Hot news from summer 2003. *Nature (London)*, 432(7017), pp. 559-560.

Scheffer, M., S. Carpenter, J.A. Foley, C. C. Folke, and B. Walker, 2001: Catastrophic shifts in ecosystems. *Nature*, 413, pp. 145-154.

Scheller, R.M., and D.J. Mladenoff, 2005: A spatially interactive simulation of climate change, harvesting, wind, and tree species migration and projected changes to forest composition and biomass in northern Wisconsin, USA. *Global Change Biology*, 11(2), pp. 307-321.

Schiermeier, Q., 2005: That sinking feeling. *Nature (London)*, 435(9 June), pp. 732-733.

Schimper, A.F.W., 1903: *Plant geography on a physiological basis.* Clarendon Press, Oxford.

Schlesinger, W.H., 1997: *Biogeochemistry: An analysis of global change.* Academic press, San Diego, Second ed.

Schneider, S.H., 2003: *Abrupt non-linear climate change, irreversibility and surprise. working party on global and structural policies oecd:oecd workshop on the benefits of climate policy: improving information for policy makers.* Organisation for Economic Co-operation and Development (OECD).

Schoennagel, T., T.T. Veblen, and W.H. Romme, 2004: The interaction of fire, fuels, and climate across rocky mountain forests. *Bioscience*, 54(7), pp. 661-676.

Scholes, R.J., and S.R. Archer, 1997: Tree-grass interactions in savannas. *Annual Review of Ecology and Systematics*, 28, pp. 517-544.

Scholes, R.J., and R. Biggs (eds.), 2004: *Ecosystems services in Southern Africa: A regional assessment*. CSIRO, Pretoria.

Scholes, R.J., G. von Maltitz, M. de Wit, G.O. Hughes, G. Midgley, and B. Erasmus, 2004: Helping biodiversity adapt to climate change. *Science Deve. Network*

Schröter, D., W. Cramer, R. Leemans, C. Prentice, M.B. Araújo, N.W. Arnell, A. Bondeau, H. Bugmann, T. Carter, C.A. Garcia, A.C. de la Vega-Leinert, M. Erhard, F. Ewert, M. Glendining, J.L. House, S. Kankaanpää, R.J.T. Klein, S. Lavorel, M. Lindner, M.J. Metzger, J. Meyer, T. Mitchell, I. Reginster, M. Rounsevell, S. Sabaté, S. Sitch, B. Smith, J. Smith, P. Smith, M.T. Sykes, K. Thonicke, W. Thuiller, G. Tuck, S. Zaehle, and B. Zierl, 2007: Ecosystem service supply and human vulnerability to global change in Europe. *Science*, pp. (submitted).

Schwinning, S., and O.E. Sala, 2004: Hierarchy of responses to resource pulses in arid and semi-arid ecosystems. *Oecologia*, 141, pp. 211-220.

Schär, C., P.L. Vidale, D. Lüthi, C. Frei, C. Häberli, M.A. Liniger, and C. Appenzeller, 2004: The role of increasing temperature variability in European summer heatwaves. *Nature (London)*, 427(6972 (22.Jan.)), pp. 332-336.

Semazzi, F.H.M., and Y. Song, 2001: A GCM study of climate change induced by deforestation in Africa. *Climate Research*, 17(2), pp. 169-182.

Serreze, M., F. Carse, R. Barry, and J. Rogers, 1997: Icelandic low cyclone activity: Climatological features, linkages with the North Atlantic Oscillation, and relationships with recent changes in the Northern Hemisphere circulation. *J. Climate*, 10, pp. 453-464.

Shafer, S.L., P.J. Bartlein, and R.S. Thompson, 2001: Potential Changes in the Distributions of Western North America Tree and Shrub Taxa under Future Climate Scenarios. *Ecosystems*, 4(3), pp. 200.

Sheppard, C.R.C., 2003: Predicted recurrences of mass coral mortality in the Indian Ocean. *Nature*, 425, pp. 294-297.

Sheppard CRC, 2003, Predicted recurrences of coral mass mortality in the Indian Ocean. Nature 424, pp.294-297

Silvertown, J., 2004: Plant coexistence and the niche. *Trends in Ecology and Evolution*, 19(11), pp. 605-611.

Skaare, J.U., A. Bernhoft, O. Wiig, K.R. Norum, E. Haug, D.M. Eide, and A.E. Derocher, 2001: Relationships between plasma levels of organochlorines, retinol and thyroid hormones from polar bears (Ursus maritimus) at Svalbard. *Journal of Toxicology and Environmental Health - Part A*, 62(4), pp. 227-241.

Slaymaker, O., 2001: Why so much concern about climate change and so little attention to land use change? *Canadian Geographer*, 45(1), pp. 71-78.

Smant AM (ed), 2001, Coral Reef algal community Dynamics. Coral Reefs, 19, 1-425.

Smith, J.B., A. Rahman, S. Haq, and M.Q. Mirza, 1998: *Considering adaptation to climate change in the sustainable development of Bangladesh.* World Bank, Washington, D.C.

Smith, L.C., G.M. MacDonald, A.A. Velichko, D.W. Beilman, O.K. Borisova, K.E. Frey, K.V. Kremenetski, and Y. Sheng, 2004: Siberian peatlands a net carbon sink and global methane source since the early Holocene. *Science*, 303(5656), pp. 353-356.

Smith, M.D., and A.K. Knapp, 1999: Exotic plant species in a C-4-dominated grassland: invasibility, disturbance, and community structure. *Oecologia*, 120(4), pp. 605-612.

Smith, P., J. Smith, M. Wattenbach, J. Meyer, M. Lindner, S. Zaehle, R. Hiederer, R. Jones, L. Montanarella, M.D.A. Rounsevell, and I. Reginster, 2005: Projected changes in mineral soil carbon of European forests, 1990-2100. *Canadian Journal of Soil Science*

Smith, S.D., T.E. Huxman, S.F. Zitzer, T.N. Charlet, D.C. Housman, J.S. Coleman, L.K. Fenstermaker, J.R. Seemann, and R.S. Nowak, 2000a: Elevated CO2 increases productivity and invasive species success in an arid ecosystem. *Nature*, 408, pp. 79-82.

Smith, S.D., T.E. Huxman, S.F. Zitzer, T.N. Charlet, D.C. Housman, J.S. Coleman, L.K. Fenstermaker, J.R. Seemann, and R.S. Nowak, 2000b: Elevated CO2 increases productivity and invasive species success in an arid ecosystem. *Nature*, 408(6808), pp. 79-82.

Smith, W., and A. Knapp, 1990: Ecophysiology of high mountain forests. In: *Plant biology of the Basin and Range* [Osmond, C., L. Pitelka, and G. Hidy (eds.)]. Springer-Verlag, pp. 87-142.

Snyder, P.K., C. Delire, and J.A. Foley, 2004: Evaluating the influence of different vegetation biomes on the global climate. *Climate Dynamics*, 23(3 - 4), pp. 279.

Snyman, H.A., 2005: Rangeland degradation in a semi-arid South Africa-I: influence on seasonal root distribution, root/shoot ratios and water-use efficiency. *Journal of Arid Environments*, 60 (3), pp. 457-481.

Sorenson, L., R. Goldberg, T.L. Root, and M.G. Anderson, 1998: Potential effects of global warming on waterfowl populations breeding in the northern Great Plains. *Climatic Change*, 40, pp. 343-369.

Spalding, M.D., and G.E. Jarvis, 2002: The impact of the 1998 coral mortality on reef fish communities in the Seychelles. *Mar. Pollut. Bull.*, 44, pp. 309–321.

Stabeno, P.J., N.A. Bond, N.B. Kachel, S.A. Salo, and J.D. Schumacher, 2001: On the temporal variability of the physical environment over the south-eastern Bering Sea. *Fisheries Oceanography*, 10, pp. 81-98.

Stapp, P., G.A. Polis, and F. Sanchez Pinero, 1999: Stable isotopes reveal strong marine and El Nino effects on island food webs. *Nature*, 401, pp. 467-69.

Stefanescu, C., S. Herrando, and F. Páramo, 2004: Butterfly species richness in the north-west Mediterranean Basin: the role of natural and human-induced factors. *Journal of Biogeography*, 31(6), pp. 905-915.

Still, C.J., P.N. Foster, and S.H. Schneider, 1999: Simulating the effects of climate change on tropical montane cloud forests. *Nature*, 398, pp. 608-610.

Stirling, I., and A.E. Derocher, 1993: Possible impacts of climatic warming on polar bears. *Arctic*, 46(3), pp. 240-245.

Stirling, I., N.J. Lunn, and J. Iacozza, 1999: Long-term trends in the population ecology of polar bears in western Hudson Bay in relation to climatic change. *Arctic*, 52(3), pp. 294-306.

Stirling, I., and T.G. Smith, 2004: Implications of warm temperatures, and an unusual rain event for the survival of ringed seals on the coast of southeastern Baffin Island. *Arctic*, 57(1), pp. 59-67.

Stock, W.D., F. Ludwig, C. Morrow, G.F. Midgley, S.J.E. Wand, N. Allsopp, and T. Bell, 2005: Long-term effects of elevated atmospheric CO2 on species composition and productivity of a southern African C4 dominated grassland in the vicinity of a CO2 exhalation. *Plant Ecology*, 178, pp. 211-224.

Stocks, B.J., M.A. Fosberg, T.J. Lynham, L. Mearns, B.M. Wotton, Q. Yang, J.Z. Jin, K. Lawrence, G.R. Hartley, J.A. Mason, and D.W. McKenney, 1998: Climate change and forest fire potential in Russian and Canadian boreal forests. *Climatic Change*, 38(1), pp. 1-13.

Stott, P.A., D.A. Stone, and M.R. Allen, 2004: Human contribution to the European heatwave of 2003. *Nature (London)*, 432(7017), pp. 610-614.

Strong AE, Barrientos CB, Duda C, Sapper J, 1997, Improved satellite techniques for monitoring coral reef bleaching. Proc. 8th Int. Coral Reef Symp., Panama, June 1996, 1:1495-1498. See coralreefwatch.noaa.gov/satellite

Sturm, M., C. Racine, K. Tape, and P. Albersen, 2001: Increasing shrub abundance in the Arctic. *Nature*, 411, pp. 546.

Sutherst, R.W., H.F. Maywald, and B.L. Russel, 2000: Estimating vulnerability under global change: modular modelling of pests. *Agriculture, Ecosystems and Environment*, 82, pp. 303-319.

Talge HK, Williams DE, Hallock P, Harney J, 1997, Symbiont loss in reef Foraminifera: consequence for affected populations. Proc. 8th Int. Coral Reef Symp., Panama, 1, 589-594

Taylor, C.M., E.F. Lambin, N. Stephenne, R.J. Harding, and R.L.H. Essery, 2002: The influence of land use change on climate in the Sahel. *Journal of Climate*, 15(24), pp. 3615-3629.

Theurillat, J.P., and A. Guisan, 2001: Potential impact of climate change on vegetation in the european Alps: a review. *Climatic Change*, 50, pp. 77-109.

Thomas, C.D., A. Cameron, R.E. Green, M. Bakkenes, L.J. Beaumont, Y.C. Collingham, B.F.N. Erasmus, M.F. de Siqueira, A. Grainger, L. Hannah, L. Hughes, B. Huntley, A.S. van Jaarsveld, G.F. Midgley, L. Miles, M.A. Ortega Huerta, A.T. Peterson, O.L. Phillips, and S.E. Williams, 2004a: Extinction risk from climate change. *Nature*, 427(6970), pp. 145-148.

Thomas, C.D., A. Cameron, R.E. Green, M. Bakkenes, L.J. Beaumont, Y.C. Collingham, B.F.N. Erasmus, M.F. de Siqueira, A. Grainger, L. Hannah, L. Hughes, B. Huntley, A.S. van Jaarsveld, G.F. Midgley, L. Miles, M.A. Ortega-Huerta, A.T. Peterson, O.L. Phillips, and S.E. Williams, 2004b: Extinction risk from climate change. *Nature (London)*, 427(6970), pp. 145-148.

Thomas, C.D., and J.J. Lennon, 1999: Birds extend their ranges northwards. *Nature*, 399(6733), pp. 213.

Thomas, D.S.G., M. Knight, and G.F.S. Wiggs, 2005: Remobilization of southern African desert dune systems by twenty-first century global warming. *Nature (London)*, 435(7046), pp. 1218-1221.

Thompson, W.G., and S. Goldstein, 2005: Open-system coral ages reveal persistent suborbital sea-level cycles. *Science*, 308, pp. 401-404.

Thuiller, W., 2003: BIOMOD: Optimising predictions of species distributions and projecting potential future shifts under global change. *Global Change Biology*, 9, pp. 1353-1362.

Thuiller, W., 2004: Patterns and uncertainties of species' range shifts under climate change. *Global Change Biology*, 10, pp. 2020-2027.

Thuiller, W., M.B. Araujo, R.G. Pearson, R.J. Whittaker, L. Brotons, and S. Lavorel, 2004a: Uncertainty in predictions of extinction risk. *Nature (London)*, 430(6995)

Thuiller, W., M.B. Araújo, and S. Lavorel, 2004b: Do we need land-cover data to model species distributions in Europe? *Journal of Biogeography*, 31, pp. 353-361.

Thuiller, W., S. Lavorel, M.B. Araujo, M.T. Sykes, and I.C. Prentice, 2005: Climate change threats to plant diversity in Europe.
Proceedings of the National Academy of sciences of the United States of America, 102(23), pp. 8245-8250.

Tiedemann, A.R., and J.O. Klemmedson, 2004: Responses of desert grassland vegetation to mesquite removal and regrowth. *Journal of Range Management*, 57(5), pp. 455-465.

Tol, R.S.J., 2001: Equitable cost-benefit analysis of climate change policies. *Ecological Economics*, 36, pp. 71-85.

Tole, L., 1998: Source of defeorestation in tropical developing countries.
Environmental Management, 22, pp. 19-33.

Tompkins, E.L., and W.N. Adger, 2003: *Building resilience to climate change through adaptive management of natural resources.* 19pp.

Trenberth, K., 1999: Conceptual framework for changes of extremes of the hydrological cycle with climate change. *Climatic Change*, 42, pp. 327-339.

Tuchman, N.C., 2002: Elevated atmospheric CO_2 lowers leaf litter nutritional quality for stream ecosystem food webs. *Global Change Biology*, 8, pp. 163-170.

Tuchman, N.C., 2003a: Elevated atmospgeric CO2 alters leaf litter nutritional quality for stream ecosystems. An in situ leaf decomposition study. *Hydrobiologia*

Tuchman, N.C., 2003b: Nutritional quality of leaf detritus altered by elevated atmospheric CO2: Effects on development of mosquito larvae. *Freshwagter Biology*

Tucker, C.J., and S.E. Nicholson, 1999: Variations in the size of the Sahara Desert from 1980 to 1997. *Ambio*, 28(7), pp. 587-591.

van der Werf, G.R., J.T. Randerson, G.J. Collatz, L. Giglio, P.S. Kasibhatla, A.F. Arellano, Jr., S.C. Olsen, and E.S. Kasischke, 2004: Continental-Scale Partitioning of Fire Emissions During the 1997 to 2001 El Nino/La Nina Period. *Science*, 303(5654), pp. 73-76.

van Herk, C.M., A. Aptroot, and H.F.S.O. van Dobben, 2002: Long-tern monitoring in the Netherlands suggests that lichens respond to global warming. *Lichenologist*, 34(2), pp. 141-154.

van Meijl, H., T. van Rheenen, A.B. Tabeau, and B. Eickhout, 2005: The impact of different policy environments on land use in Europe. *Agriculture, Ecosystems and Environment.*

van Oene, H., W.N. Ellis, M.M.P.D. Heijmans, D. Mauquoy, W.L.M. Tamis, A.J.H. van Vliet, F. Berendse, B. van Geel, R. van der Meijden, and S.A. Ulenberg, 2001: *Long-term effects of climate change on biodiversity and ecosystem processes.* NRP Report, Dutch National Research Programme on Global Air Pollution and Climate Change, Bilthoven, 306pp.

Van Wilgen, B.W., N. Govender, H.C. Biggs, D. Ntsala, and X.N. Funda, 2004: Response of Savanna fire regimes to changing fire-management policies in a large African National Park. *Conservation Biology*, 18(6), pp. 1533-1540.

Velichko, A.A., N. Catto, A.N. Drenova, V.A. Klimanov, K.V. Kremenetski, and V.P. Nechaev, 2002: Climate changes in East Europe and Siberia at the late glacial- holocene transition. *Quaternary International*, 91(1), pp. 75-99.

Verburg, P., R.E. Hecky, and H. Kling, 2003: Ecological consequences of a century of warming in Lake Tanganyika. *Science*, 301, pp. 505-507.

Verburg, P.H., J.E. Schulp, N. Witte, and A. Veldkamp, 2007: Downscaling of land use changes to assess the dynamics of European landscapes. pp. (submitted).

Villalba, R., A. Lara, J.A. Boninsegna, M. Masiokas, S. Delgado, J.C. Aravena, F.A. Roig, A. Schmelter, A. Wolodarsky, and A. Ripalta, 2003: Large-scale temperature changes across the southern Andes: 20th-century variations in the context of the past 400 years. *Climatic Change*, 59(1-2), pp. 177-232.

Villalba, R., and T. Veblen, 1997: Regional patterns of tree population age structure in northern Patagonia: climatic and disturbance influences. *J Ecol*, 85, pp. 113-124.

Villalba, R., T.T. Veblen, and J. Ogden, 1994: Climatic influences on the growth of subalpine trees in the colorado front range. *Ecology*, 75(5), pp. 1450-1462.

Villers-Ruiz, L., and I. Trejo-Vazquez, 1998: Impact of climatic change in forests and natural protected areas of Mexico. *Interciencia*, 23(1), pp. 10-+.

Vitousek, P.M., C.M. Dantonio, L.L. Loope, M. Rejmanek, and R. Westbrooks, 1997a: Introduced species: A significant component of human-caused global change. *New Zealand Journal of Ecology*, 21(1), pp. 1-16.

Vitousek, P.M., H.A. Mooney, J. Lubchenco, and J.M. Melillo, 1997b: Human domination of Earth's ecosystems. *Science*, 277(25. Jul.), pp. 494-499.

Von Maltitz, G.P., R.J. Scholes, B. Erasmus, and A. Letoalo, 2007: Adapting conservation strategies to accommodate impacts of climate change. In: *AIACC Adaptation Synthesis*. pp. (In press).

Walsh, J.E., 1995: Long-term observations for monitoring of the cryosphere. *Climatic Change*, 31(2-4), pp. 369-394.

Walther, G.R., E. Post, P. Convey, A. Menzel, C. Parmesan, T.J.C. Beebee, J.M. Fromentin, O. Hoegh-Guldberg, and F. Bairlein, 2002: Ecological responses to recent climate change. *Nature (London)*, 416(6879), pp. 389-395.

Wang, G.L., and E.A.B. Eltahir, 2000: Role of vegetation dynamics in enhancing the low-frequency variability of the Sahel rainfall. *Water Resources Research*, 36(4), pp. 1013-1021.

Wang, G.L., and E.A.B. Eltahir, 2002: Impact of CO2 concentration changes on the biosphere-atmosphere system of West Africa. *Global Change Biology*, 8(12), pp. 1169-1182.

Wang, S.Q., J.Y. Liu, G.R. Yu, Y.Y. Pan, Q.M. Chen, K.R. Li, and J.Y. Li, 2004a: Effects of land use change on the storage of soil organic carbon: A case study of the Qianyanzhou Forest Experimental Station in China. *Climatic Change*, 67(2-3), pp. 247-255.

Wang, X.L., and V.L. Swail, 2002: Trends of Atlantic wave extremes as simulated in a 40-yr wave hindcast using kinematically reanalyzed wind fields. *J. Climate*, 15, pp. 1020-1035.

Wang, X.L., and V.R. Swail, 2001: Changes of extreme wave heights in Northern Hemisphere oceans and related atmospheric circulation regimes., 14, 2204–2221. *J. Climate*, 14, pp. 2204-2221.

Wang, X.L., F.W. Zwiers, and V.L. Swail, 2004b: North Atlantic Ocean Wave Climate Change Scenarios for the Twenty-First Century. J. Climate. *J. Climate*, 17, pp. 2368-2383.

Wanner, H., R. Rickli, E. Salvisberg, C. Schmutz, and M. Schuepp, 1997: Global climate change and variability and its influence on Alpine climate - Concepts and observations. *Theoretical and Applied Climatology*, 58, pp. 221-243.

Warburton, M., R.E. Schulze, and M. Maharaj, 2005: *Is South Africa's temperature changing? An analysis of trends from daily records, 1950-2000.*

Wardle, D.A., G. Hornberg, O. Zackrisson, M. Kalela-Brundin, and D.A. Coomes, 2003: Long-

Term Effects of Wildfire on Ecosystem Properties Across an Island Area Gradient. *Science*, 300(5621), pp. 972-975.

Warren, R., 2005: Impacts of global temperature change at different annual mean global temperature increases. In: *Avoiding Dangerous Climate Change*, Exeter, UK.

Weinberg, J.R., T.G. Dahlgren, and K.M. Halanych, 2002: Influence of rising sea temperatures on commercial bivalve species of the U.S. Atlantic coast. In: *Fisheries in a Changing Climate* [McGinn, N.A. (ed.)]. American Fisheries Society, Bethesda, Maryland, pp. 131–140.

Weltzin, J.F., S.D. Bridgham, J. Pastor, and etal, 2003: Potential effects of warming and drying on peatland plant community composition. *Global Change Biology*, 9, pp. 141-151.

Weltzin, J.F., C. Harth, S.D. Bridgham, and etal, 2001: Production and microtopography of bog bryophytes: response to warming and water table manipulations. *Oecologia*, 128, pp. 557-565.

Weltzin, J.F., J. Pastor, C. Harth, and etal, 2000: Response of bog and fen communities to warming and water level manipulations. *Ecology*, 81, pp. 3464-3478.

Wessel, W.W., A. Tietema, C. Beier, B.A. Emmett, J. Peñuelas, and T. Riis-Nielsen, 2004: A Qualitative Ecosystem Assessment for Different Shrublands in Western Europe under Impact of Climate Change. *Ecosystems*, 7(6), pp. 662-671.

Wetzel, R.G., 2001: *Limnology: lake and river ecosystems.* Academic Press, San Diego, Third ed., 1006pp.

White, A., M.G.R. Cannell, and A.D. Friend, 2000a: CO2 stabilization, climate change and the terrestrial carbon sink. *Global Change Biology*, 6(7), pp. 817-833.

White, T.A., B.D. Campbell, P.D. Kemp, and C.L. Hunt, 2000b: Sensitivity of three grassland communities to simulated extreme temperature and rainfall events. *Global Change Biology*, 6(6), pp. 671.

Whittaker, R.H., 1975: *Communities and ecosystems.* Macmillan, New York, 2 ed., 385pp.

Wichmann, M.C., F. Jeltsch, W.R.J. Dean, K.A. Moloney, and C. Wissel, 2003: Implication of climate change for the persistence of raptors in arid savanna. *Oikos*, 102(1), pp. 186-202.

Wickensa, G.E., 1998: Ecophysiology of economic plants in arid and semi-arid lands. Adapation of desert organisms. In: *Adapation of desert organisms* [Cloudsley-Thompson, J.L. (ed.)]. Springer, Berlin, pp. 343.

Wikelski, M., and C. Thom, 2000: Marine iguanas shrink to survive El Nino. *Nature*, 403, pp. 37.

Wiley, J.W., and J.M. Wunderle, 1994: The effects of hurricanes on birds, with special reference to the Caribbean islands. *Bird COnservation International*, 3, pp. 319-349.

Wilkinson, C., 2002: Status of coral reefs of the world: 2002. In: *Global Coral Reef Monitoring Network and Australian Institute of Marine Science*, Townsville, Australia, pp. 378 pp.

Wilkinson C (ed) and 239 other authors, 2004, Status of Coral Reefs of the World: 2004. Aust Inst Mar Sci, 557p., www.aims.gov.au.

Wilkinson, C., 2004: Status of coral reefs of the world: 2004. In: *Global Coral Reef Monitoring Network and Australian Institute of Marine Science*, Townsville, Australia, pp. 557 pp.

Williams, A.A.J., D.J. Karoly, and N. Tapper, 2001: The sensitivity of Australian fire danger to climate change. *Climatic Change*, 49(1-2), pp. 171-191.

Williams, P.H., L. Hannah, S. Andelman, G.F. Midgley, M.B. Araújo, G.O. Hughes, L. Manne, E. Martinez-Meyer, and R.G. Pearson, 2005: Planning for climate change: identifying minimum-dispersal corridors for the Cape Proteaceae. *Conservation Biology*, in press

Williams, R.J., L.B. Hutley, G.D. Cook, J. Russell-Smith, A. Edwards, and X.Y. Chen, 2004: Assessing the carbon sequestration potential of mesic savannas in the Northern Territory, Australia: approaches, uncertainties and potential impacts of fire. *Functional Plant Biology*, 31(5), pp. 415-422.

Williams, S.E., E.E. Bolitho, and S. Fox, 2003: Climate change in Australian tropical rainforests: an impending environmental catastrophe. *Proceedings of the Royal Society of London B*, 270(1527), pp. 1887-1892.

Williams, EH, Bunkley-Williams L, 1990, The world-wide coral reef bleaching cycle and related sources of coral mortality. Atoll Res. Bull., 335, 1-71

Winder, M.A., and D.E. Schindler, 2004: Climatic effects on the phenology of lake processes. *Global Change Biology*, 10, pp. 1844-1856.

Wingfield, J.C., G. Ramos-Fernandez, A. Nunez-de la Mora, and H. Drummond, 1999: The effects of an "El Nino" southern oscillation event on reproduction in male and female blue-fotted boobies, Sula nebouxii. *General and Comparative Endocrinology*, 114, pp. 163-172.

Woodward, F.I., and M.R. Lomas, 2004: Simulating vegetation processes along the Kalahari transect. *Global Change Biology*, 10(3), pp. 383.

Woodwell, G.M., F.T. Mackenzie, R.A. Houghton, M. Apps, E. Gorham, and E. Davidson, 1998: Biotic feedbacks in the warming of the earth. *Climatic Change*, 40(3-4), pp. 495-518.

Wooldridge, S, Done T, Berkelmans R, Jones R, Marshall P, 2005, Precursors for resilience in coral communities in a warming climate: a belief network approach. Mar Ecol Prog Ser, 295, 157-169.

World Resources, 2003: *World resources 2002-2004. Decisions for the earth. Balance, voice and power.* World Resource Institute, Washington DC.

World Resources Institute, 2000: *World Resources 2000-20001 People and Ecosystems: The Fraying Web of Life.* World Resources Institute, Washington, D.C.

Xoplaki, E., J.F. Gonzalez-Rouco, D. Gyalistras, J. Luterbacher, R. Rickli, and H. Wanner, 2003: Interannual summer air temperature variability over Greece and its connection to the large-scale atmospheric circulation and Mediterranean SSTs 1950-1999. *Climate Dynamics*, 20(5), pp. 537-554.

Xue, Y., and M.J. Fennessy, 2002: Under what conditions does land-cover change impact regional climate. In: *Global desertification. Do humans cause deserts?* [Reynolds, J.F., and D.M. Stafford-Smith (eds.)]. Dahlem University Press, Berlin, Germany.

Zebisch, M., F. Wechsung, and H. Kenneweg, 2004: Landscape response functions for biodiversity - assessing the impact of land-use changes at the county level. *Landscape And Urban Planning*, 67(1-4), pp. 157-172.

Zeng, N., J.D. Neelin, K.M. Lau, and C.J. Tucker, 1999: Enhancement of interdecadal climate variability in the Sahel by vegetation interaction. *Science*, 286(5444), pp. 1537-1540.

Zierl, B., and H. Bugmann, 2005: Global change impacts on hydrological processes in Alpine catchments. *Water Resources Research*, 41(2)

Ziska, L.H., T.B. Moya, R. Wassmann, O.S. Namuco, R.S. Lantin, J.B. Aduna, E.J. Abao, K.P. Bronson, H.U. Neue, and D. Olszyk, 1998: Long-term growth at elevated carbon dioxide stimulates methane emission in tropical paddy rice. *Global Change Biology*, 4, pp. 657-665.