Chapter 4 – Ecosystems, Their Properties, Goods, and Services

Coordinating Lead Authors
Andreas Fischlin (Switzerland), Guy F. Midgley (South Africa)

Lead Authors
Andrei Velichko (Russia), Pauline Dube (Botswana), Rik Leemans (Netherlands), Mark Rounsevell (Belgium), Jeff Price (USA), Juan Tarazona (Peru), Brij Gopal (India), Carol Turley (U.K.)

Contributing Authors
Julius Atlhopheng (Botswana), Martin Beniston (Switzerland), William J. Bond (South Africa), Keith Brander (Denmark), Terry V. Callaghan (U.K.), Jacqueline de Chazal (Belgium), Oagile Dikinya (Australia), Dimitrios Gyalistras (Switzerland), Lesley Hughes (Australia), Christian Körner (Switzerland), Wolfgang Lucht (Germany), Nick Lunn (Canada), Ron Neilson (USA), Martin Pecheux (France), Wilfried Thuiller (South Africa), Rachel Warren (U.K.)

Review Editors
Sandra Diaz (Argentina), Wolfgang Cramer (Germany)

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For reviewers: Please note that we have used 2007 as the year to earmark any reference which is currently in the status in print or similar, regardless whether it is likely to be printed next month or not. All those references will be removed from the final document, should they, contrary to current expectation, not be published in time. Please do not comment on that year.
Executive Summary

• Principle findings of the TAR remain relevant, but with more evidence and greater confidence from a broader ambit (high confidence; 4.4.1-4.4.11) that identifies substantial species and ecosystem vulnerabilities especially to temperature changes in excess of 2 degrees above pre-industrial levels (high confidence; 4.4.11). New concerns have emerged, including possible impacts of wildfire in terrestrial ecosystems (high confidence; 4.4.1), and ocean acidification in the marine realm (high confidence; 4.4.9).

• Climate change impacts may occur earlier than projected by TAR, such as negative impacts on global terrestrial carbon balance after an initial net gain until ~2030, induced mainly by impacts on forest systems turning the biosphere into a net carbon source (medium confidence; 4.4.1, 4.4.11) and there is growing evidence for a high vulnerability of a larger fraction of species (globally ~25% by 2100, for some biota as low as 1% or as high as 43%) becoming committed to extinction than previously assessed (medium confidence; 4.4.11).

• In addition to widespread species geographic shifts (high confidence; 4.4.11) major (high confidence) and rapid (low confidence) shifts in vegetation structure (medium confidence; 4.4.1, 4.4.10, 4.4.11) are possible in most regions (not only in the boreal zone as previously stressed) some driven by wildfire, and other ancillary stresses (medium confidence; 4.4.1) which may lead to mainly detrimental changes in ecosystem services (medium confidence; 4.4.1, 4.4.3, 4.4.4, 4.4.5, 4.4.6, 4.4.10, 4.4.11).

• Responses of endemic species geographic range size are overwhelmingly negative (high confidence; 4.4.11) with resulting impacts on biodiversity and biodiversity hotspots (medium confidence; 4.4.1, 4.4.10, 4.4.11) and strongly sensitive systems such as coral reefs (high confidence; Box 4.5; 4.4.9) arctic systems (high confidence; 4.4.6), mountains (high confidence; 4.4.7) Mediterranean-climate systems (medium confidence; 4.4.4) and savannas (medium confidence; 4.4.3).

• In many cases, endemic species persistence requires migration rates that exceed their natural adaptive capacity (for tree species high confidence (4.4.5), other species medium confidence, 4.4.11). These effects, combined with landscape fragmentation through land use change, limit natural adaptation especially for plant species, and increase their risk of extinction during this century (medium confidence; 4.4.1-4.4.6, 4.4.11). Marine ecosystems and species appear more able to shift range rapidly than many terrestrial species (medium confidence; 4.4.9).

• Changes in disturbance and other ancillary stresses such as wildfire and invasive species are likely to exacerbate climate change impacts, sometimes synergistically (medium confidence; 4.4.1).

• Conservation entities are generally poorly prepared (with a few exceptions of untested theoretical developments) for climate change impacts, especially if based solely on static protected areas systems (high confidence; 4.4.11, 4.6.1).

• The cascade of uncertainties from climate change projections (especially in rainfall patterns) through still developing theory and knowledge of species to ecosystem responses (high confidence; 4.4.1, 4.4.11, 4.8) remain significant barriers to developing coherent and detailed regional planning responses, thus responses remain limited to generalized no-regrets strategies (high confidence; 4.6), and impacts on sustainable development and livelihoods are difficult to estimate (high confidence; 4.7).
4.1 Introduction

The critical role of ecosystems in supporting human well-being is now beyond doubt (Millennium Ecosystem Assessment, 2005b), and the importance of ecosystem persistence under anthropogenic climate change is recognized in Article 2 of the UNFCCC. Humans have historically benefited from well known ecosystem-derived goods, but ecosystems also provide a range of equally important services that underpin human well-being in less well recognized ways through regulation of global carbon, nitrogen, water, and other biogeochemical cycles and global, regional and local environmental conditions.

4.1.1 Biomes, communities, population systems, and ecophysiology

For the purpose of this assessment an ecosystem is defined as a functional entity consisting of interacting groups of organisms and their environment. Ecosystems cover a hierarchy of spatial scales and can comprise the entire globe (e.g. global biogeochemistry), biomes at the continental scale, or small, well-circumscribed systems such as a single vernal pool. Thus, this chapter considers the response to climate change of functional and structural properties of ecosystems, their goods and services and their underlying biodiversity for biomes and communities, populations, the underlying ecophysiological processes and any emergent biogeochemical impacts.

The focus is on natural ecosystems (Table 4.1) from the global to local scales as grouped by commonly accepted functional classifications and based on projections of future impacts for the scenarios from WG I (physics) and chapter 2 (scenarios). Certain ecosystem goods and services are treated in detail in other sectoral chapters: chapters 3 (water), 5 (food, fibre, fisheries), 6 (coasts), and 8 (health). Key findings from this chapter are further developed in the synthesis chapters 17 to 20. Region-specific aspects of ecosystems are discussed in chapters 9 to 16.

Projecting the impacts of climate change on ecosystems is complicated by an uneven understanding of time and spatial scales of response because these scales are often linked. Processes at broad spatial scales are characterised by long-term temporal responses (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). Conversely, responses of species and populations sometimes of putative keystone or indicator species, show intermediate temporal responses (months to centuries) that underpin changes in biodiversity. Finally, fast ecophysiological responses (seconds, hours, days, months) of microorganisms, plants, and animals operate at microscopic scales from a leaf or organ to the cellular level, and are assessed here if they scale up to have a significant impact at broader spatial scales, or where the mechanistic understanding assists in assessing key thresholds in higher level responses.

Biomes are assemblages of organisms with a characteristic structure that is repeated at broad spatial scales in similar climatic zones throughout the world. They are the result of long-term, self-organizing processes such as the formation of soils, development of disturbance regimes, and a balanced nutrient capital. Thus climate change is expected to trigger responses that may last up to millennia before biomes reach a state of dynamic equilibrium with respect to their biophysical and chemical environment.

The distribution of biomes has traditionally been explained only in terms of climate control (Schimper, 1903), but several recent findings agree that disturbance regimes such as fire or insects, may influence vegetation structure significantly. Biomes are differentially sensitive to climatic change (e.g. Kirschbaum and Fischlin, 1996; Gitay et al., 2001; Sala et al., 2000), with temperature-limited biomes prone to impacts of warming, and water-limited biomes prone to increasing levels of drought. Some biomes such as fire dependent biomes, may be in a meta-stable state that can fairly rapidly switch
under climate and other environmental changes (Bond et al., 2005; Scheffer et al., 2001).

**Table 4.1:** Major ecosystems addressed in this report, together with their global areal extent, NPP and total carbon stores (above plus below ground, compiled from 1 – Sabine et al., 2004; 2 – Bonan, 2002; 3 – Field et al., 1998; 4 – Hassan et al., 2005 Table C2; 5 – Mitra et al., 2005 Table 2, Figure 4; 6 – Foley et al., 2005), and brief listing of key goods and services.

<table>
<thead>
<tr>
<th>Biome / Ecosystem</th>
<th>Area (Mkm²)</th>
<th>% transformed</th>
<th>NPP (Pg C yr⁻¹)</th>
<th>Plant Carbon (Pg C)</th>
<th>Soil Carbon (Pg C)</th>
<th>Ecosystem services</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Deserts</strong></td>
<td>17.7</td>
<td>3.5</td>
<td>3.2 (1.5)</td>
<td>1.10</td>
<td>1.208</td>
<td>Pasture, food, medicinals, eco-tourism, biodiversity of droughts- and heat-adapted life forms, extraction of fossil fuels and water, military and recreational activities</td>
</tr>
<tr>
<td><strong>Grasslands and savannas</strong></td>
<td>27.6</td>
<td>28.6</td>
<td>1.13 (0.7)</td>
<td>0.79</td>
<td>1.345</td>
<td>Livestock production, wood production for fuel and construction, plant, mammal and bird diversity, carbon sequestration, soil protection.</td>
</tr>
<tr>
<td><strong>Tropical grassland/savanna</strong></td>
<td>27.6</td>
<td>28.6</td>
<td>1.13 (0.7)</td>
<td>0.79</td>
<td>1.345</td>
<td>Nature-based tourism, mammal and bird diversity, Livestock production, wood production for fuel and construction, carbon sequestration, soil protection.</td>
</tr>
<tr>
<td><strong>Temperate grassland</strong></td>
<td>18</td>
<td>5.1</td>
<td>1.6</td>
<td>1.172</td>
<td></td>
<td>Carbon sequestration, soil protection.</td>
</tr>
<tr>
<td><strong>Mediterranean ecosytems</strong></td>
<td>2.8</td>
<td>1.3</td>
<td>1.17</td>
<td>1.124</td>
<td></td>
<td>Biodiversity, wildflower and other extractive products, water yield, soil protection.</td>
</tr>
<tr>
<td><strong>Forests and woodlands</strong></td>
<td>36.9</td>
<td>10.1</td>
<td>1.340 (0.3)</td>
<td>1.692</td>
<td></td>
<td>Timber and non-wood products, ecotourism, soil conservation, carbon sequestration, recreation, research opportunities, spiritual/cultural values</td>
</tr>
<tr>
<td><strong>Tropical forests</strong></td>
<td>17.5</td>
<td>34.0</td>
<td>1.201 (0.18)</td>
<td>1.340</td>
<td>1.692</td>
<td>Biodiversity, ecotourism, carbon sequestration and regional climate amelioration</td>
</tr>
<tr>
<td><strong>Temperate forests</strong></td>
<td>10.4</td>
<td>67.4</td>
<td>1.139 (0.3)</td>
<td>1.262</td>
<td></td>
<td>Timber and non-wood products, carbon sequestration</td>
</tr>
<tr>
<td><strong>Boreal forests</strong></td>
<td>13.7</td>
<td>25.4</td>
<td>1.57</td>
<td>1.150</td>
<td></td>
<td>Carbon sequestration, regional climate amelioration</td>
</tr>
<tr>
<td><strong>Cultivated lands</strong></td>
<td>35.3</td>
<td>47.0</td>
<td>38.0</td>
<td>-</td>
<td>-</td>
<td>Carbon sequestration and climate regulation, biodiversity of low-temperature adapted species, fuel, food and fibre, cultural and spiritual values</td>
</tr>
<tr>
<td><strong>Tundra, other palearctic biomes</strong></td>
<td>5.6</td>
<td>0.3</td>
<td>0.5 (0.8)</td>
<td>1.144</td>
<td></td>
<td>Carbon sequestration and climate regulation, biodiversity of low-temperature adapted species, fuel, food and fibre, cultural and spiritual values</td>
</tr>
<tr>
<td><strong>Freshwater lakes, rivers, wetlands</strong></td>
<td>10.3</td>
<td>11.0</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>Freshwater supply, aquatic biodiversity, nutrient transport, pollution amelioration</td>
</tr>
<tr>
<td><strong>Inland wetlands, peatlands</strong></td>
<td>3.8-4.0</td>
<td>3.8</td>
<td>3.8-4.0</td>
<td>225-462</td>
<td></td>
<td>Carbon sequestration, biodiversity, fuel</td>
</tr>
<tr>
<td><strong>Terrestrial ecosystems</strong></td>
<td>148.2</td>
<td>40.0</td>
<td>53.7 (56.4)</td>
<td>658</td>
<td>2322-2559</td>
<td>Food, carbon sequestration, biodiversity, regional climate control, waste dilution, recreation, spiritual and cultural value</td>
</tr>
<tr>
<td><strong>Marine ecosystems</strong></td>
<td>349.3</td>
<td>48.5</td>
<td>48.5</td>
<td>-</td>
<td>650</td>
<td>Marine ecosystems</td>
</tr>
<tr>
<td><strong>Total</strong></td>
<td>497.5</td>
<td>-</td>
<td>102.2 (104.9)</td>
<td>650</td>
<td></td>
<td><strong>Total</strong></td>
</tr>
</tbody>
</table>

**Communities** are found at a level below biomes and are characterized by a specific species composition and thus a specific diversity.

**Population systems** are usually characterized by a particular species and may cut across communities and even biomes, e.g. many migratory birds inhabit forests or grasslands – depending on the season – and visit wetlands on their migratory routes.

**Ecophysiological responses** operate continuously, at a microscopic scale, and usually represent the initial response of individual organisms to environmental variability. Ecophysiological mechanisms underpin individual tolerance to environmental stress, and comprise a broad range of responses. 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 leaf stomata, 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; Gedney et al.,
2006; Matthews, 2006).

4.1.2 Ecosystem goods and services

Ecosystems provide many goods and services (Fig. 4.1) that 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 and pollination, are well known, but others,
such as carbon sequestration (e.g. Foley and Ramankutty, 2004), are less obvious and often hardly
recognized, yet they are vital for human welfare. Some ecosystems services are not of direct use to
humans yet are a necessary prerequisite for all goods and services of interest to humans. These are
termed supporting services, and comprise the processes resulting in soil formation, local climate
modifications, nutrient cycling, all of which are needed for primary and secondary ecosystem
production (Fig. 4.1). Critically moreover, ecosystems maintain biodiversity, an increasingly valuable
resource that underpins many of the goods and services humans enjoy from ecosystems.

This chapter discusses many supporting, provisioning, and regulating services that are obtained from
ecosystems: (i) primary production (as the basis for fuel wood, fibre, and fodder production) and (ii)
primary or secondary production as the basis for other products (such as nuts, spices, aromatic plants,
medicinal and cosmetic products, or games cf. Gitay et al., 2001); (iii) services of paramount
importance for human existence like a) climate and water regulation; b) water and air purification, c)
carbon sequestration; d) disease, pest, and pathogen regulation.
4.1.3 Key issues

Ecosystems are expected to tolerate some level of future climate change and in some form or another will continue to persist (e.g. Kirschbaum and Fischlin, 1996; Gitay et al., 2001). A key uncertainty, however, is whether the resilience (see glossary and e.g. Folke et al., 2004) that ecosystems have developed during past environmental change (e.g. Harrison and Prentice, 2003) will be sufficient to tolerate rapid climatic changes in the future (e.g. Jump and Penuelas, 2005). Ecosystems are, moreover, increasingly subject to human induced pressures, such as extractive use of goods, and increasing fragmentation and degradation of natural habitats (e.g. Bush et al., 2004). Climate change will exacerbate these human-induced pressures and this will tend to reduce taxonomic diversity. Furthermore, it is thought that passing critical thresholds may trigger non-linear responses that could lead to novel critical states that are poorly understood. Projected future climate change and other human-induced pressures are unprecedented compared with the past few millennia (WG I, chapter 2, and e.g. Petit et al., 1999).

The timing of such effects is also important. Many ecosystems may take several centuries or even millennia (where soil formation is involved) to reach a quasi-equilibrium with a 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 to anticipate critical effects and the possible risk of coincident occurrence of effects that would otherwise be tolerable, i.e. if they happened at different times.

A further key issue is whether such transformations include species extinctions, since these represent irreversible change. This is crucial since many studies postulate a link between diversity and ecosystem functioning and changes in ecosystem services. Three effects seem possible (Millennium Ecosystem Assessment, 2005b p43-46): First, the loss of ecologically equivalent or redundant species decreases resilience. Secondly, loss of key predators or mutualists such as pollinators, or other keystone species alters ecosystem functioning. Thirdly, the loss of species that are ecological complements reduces overall performance in terms of provided goods and services.

4.1.4 Conclusions from the IPCC Third Assessment Report (TAR)

The TAR chapter on ecosystems (Gitay et al., 2001) compiled a substantial set of observations on changed distribution ranges, which concur with theoretically expected responses and considered the role of other pressures apart from climate change. It presented losses of biomass over significant areas due to fragmentation, non-sustainable land use and desertification, and threats to biodiversity with 25% of mammals and 12% of birds being at significant risk of global extinction due to anthropogenic climate change. In spite of previously held views that aquatic ecosystems are buffered against warming, it was found that in many northern hemisphere lakes and rivers ice cover broke up earlier and froze later. The TAR also questioned the previously expected productivity increases due to CO₂ fertilization effects, and demonstrated that tundra and some peat lands have turned from a net carbon sink to a source of up to 0.7 Pg C yr⁻¹. This chapter is based on new findings since the TAR.

4.2 Current sensitivities

4.2.1 Insights from past climates

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 CO₂ increase and a global warming of about 6°C 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 ongoing climatic change (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). Furthermore, modern biota have evolved under relatively cool low CO2 conditions for most of the Pleistocene, which 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 that approximate those currently present arising after the Cretaceous age (Beerling and Woodward, 2001). Dropping atmospheric CO2 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 cool glacial and warm 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 Climatic variability and extremes

Earlier IPCC reports described several ecosystems to be resilient to warming up to 1° C (e.g. Kirschbaum and Fischlin, 1996), but recent studies have resulted in a new, more differentiated view of the sensitivity of ecosystems (e.g. Walther et al., 2002) that includes understanding of the role of climatic variability and extremes. The links between climate variability and natural ecosystems has improved following progress in the understanding of the behaviour 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 determine indirectly the response of vegetation, notably through the shifts in major controls (temperature, precipitation, snow cover). For example, the European Alps experience changes in regional climates that can be attributed in part to the shifting behaviour of the NAO (Hurrell and van Loon, 1997; Serreze et al., 1997; Wanner et al., 1997; Beniston and Jungo, 2002) such as the lack of snow in the late 1980s and early 1990s (Beniston, 2003). Reversals of precipitation regimes in the Pacific region and beyond during ENSO events can disrupt vegetation through drought, heat stress, spread of parasites and disease, and enhanced occurrence of fire (e.g., Diaz and Markgraf, 1992). 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).

Many significant impacts of climatic change may emerge through shifts in the intensity and the frequency of extreme weather events. Extreme events can cause mass mortality of individuals and contribute significantly to determining which species occur in ecosystems (Parmesan et al., 2000). Drought in particular plays an important role in forest dynamics, and has been responsible for pulses of tree mortality in the Argentinian Andes (Villalba and Veblen, 1997) and in the eastern Mediterranean (Körner et al., 2005b). In the Canadian Rockies, times of extreme cold have been identified as a cause of tree death, with a sustained period of cold summers in 1696–1701 apparently
being responsible for extensive tree mortality (Luckman, 1994). Conversely, in 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, 2004; Schär et al., 2004 see also Box 4.1) 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 the species. Some plants may in the year following a major desiccation or heat-stress event enhance their biological productivity in order to ensure reproduction and perpetuation in the face of adversity. 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). The December 1999 “storm-of-the-century” that affected western and central Europe destroyed trees at a rate of up to 10 times that of the annual felling rate in parts of France and Switzerland (Anonymous, 2001). 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).

### Box 4.1: Ecological Impacts by the European heat wave 2003

Anomalous hot and dry conditions affected Europe between June and mid-August of 2003 (Schär et al., 2004; Luterbacher et al., 2004). The risk of summers as warm as 2003 may increase by two orders of magnitude in the next 40 years (Stott et al., 2004) and similar heat wave conditions are likely to increase in frequency (Meehl and Tebaldi, 2004) becoming the norm by 2080 in an SRES A2 world (Schär et al., 2004; Beniston, 2004). Future projections for southern Europe suggest significant species richness losses even under mean climate change conditions (Thuiller et al., 2005), let alone for climate extremes. The major impact of the 2003 heat wave on vegetation and ecosystems appears to have been through drought stress (Gobron et al., 2005), causing reductions of up to 30% in GPP, resulting in a net carbon source of 0.5 Pg C yr\(^{-1}\) (Ciais et al., 2005). Some vegetation types may have recovered by 2004 from the effects of drought (Gobron et al., 2005), which conforms with the findings of modelling studies (Fischlin et al., 2007), but enhanced crown damage of dominant forest trees in 2004 suggests complex delayed impacts (Fischer, 2005). This suggests a certain resilience in some ecosystems, but only if such events remain rare (Gobron et al., 2005).

In addition, a record-breaking incidence of spatially extensive wildfires was observed in some countries (Barbosa et al., 2003). Although the number of fires was not exceptional in Spain, Italy, Greece, and France, in Portugal in particular forest fires were much more extensive than have been recorded before, due to particular conditions of drought, high temperature and reduced humidity. These burned over more than 450,000 ha, twice the previous extreme (1998), and four times the 1980-2004 average (Trigo et al., 2005). By the third week of August, fires had burned in excess of 5.6% of the total forest surface of the country, with an economical impact exceeding 1 billion Euro (De Bono et al., 2004). Furthermore, by promoting the regrowth of highly flammable, shrubby vegetation that is more frequently burned than other land-cover types (Nunes et al., 2005), and due to the tendency of burned woodlands to reburn at shorter intervals (Vazquez and Moreno, 2001; Salvador et al., 2005), this may have produced a long-term impact. In addition, the 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 feed back 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).
4.2.3 Disturbance and other drivers

Ecosystems are sensitive not only to the direct effects of climate, but also to multiple interacting pressures such as land use change, fire, pollution and invasive species (global change drivers), which may also be climate dependent (Reid et al., 2005; Sala et al., 2000). In the recent past many anthropogenic pressures exerted on ecosystems have intensified (Gitay et al., 2001). This includes unsustainable resource utilization, pollution (e.g. Lelieveld et al., 2002), alterations to nutrient cycles (e.g. Vitousek et al., 1997b), land-use change (Lepers et al., 2005), and redistribution of biota through greater human mobility (e.g. Mack et al., 2000). The explicit inclusion of non-climate drivers in analyses of climate change could lead to unexpected outcomes (Hansen et al., 2001; Duraipappah et al., 2005). Consequently, many impact studies of climate change may be conservative estimates (e.g. studies that assume species migration across fragmented landscapes Pearson and Dawson, 2005).

Land use change is recognized as a primary global change driver (Hansen et al., 2001; Guo, 2000; Lambin et al., 2003; Korner, 2003; Lepers et al., 2005; Jenkins, 2003; Carpenter et al., 2005; Sala et al., 2000). This may be replacement of one land cover type by another, e.g. forest to cultivated land, as well as subtle changes of management practices within a given cover, e.g. intensification of agricultural land, both of which have transformed ~40% of the terrestrial surface (reviewed by Foley et al., 2005). Several studies suggest that land use change has been and will continue to be a more important driver for ecosystems than climate change (Slaymaker, 2001; Sala et al., 2000; Jenkins, 2003; Carpenter et al., 2005; Heywood and Watson, 1995; Gaston et al., 2003). However, the interactions between climate, land use and biodiversity are problematic for any consideration of these factors in isolation (Hansen et al., 2001; Lambin et al., 2003; Carpenter et al., 2005).

Fire influences community structure by favouring species that tolerate fire or even enhance fire spread, resulting in evolved flammability in species and communities (Bond and Keeley, 2005), and vegetation types that are far from maximum biomass predicted by regional climate (Bond et al., 2005). Thus geographic shifts in key species or fire weather may cause fundamental community shifts (Brooks et al., 2004). Fire-prone vegetation types cover a total of 40% of the world’s land surface (Chapin et al., 2002), and are common in tropical and subtropical regions often in the southern Hemisphere (Bond et al., 2005), with the exception of the boreal forest (Kasischke et al., 1995; Peng and Apps, 1999). Changes in fire regime driven at least partly by 20th century climate change (Gillett et al., 2004), in conjunction with people (Wotton et al., 2003), appear to be changing vegetation structure and composition, such as shifts from Picea to Pinus-dominated communities and 75-95% reductions in tree densities (Lavoie and Sirois, 1998). Drought facilitated the spread of human-caused fire in tropical regions during the 97/98 El Nino (Randerson et al., 2005), affecting atmospheric trace gas concentrations (CO, CH4, H2) at hemispheric scales (Kasischke et al., 2005; Langenfelds et al., 2002; Novelli et al., 2003). Tropical forest fires are becoming more common (Cochrane, 2003). In the Amazon, anthropogenic fires have strong negative effects on vegetation (Cochrane and Laurance, 2002; Haugaasen et al., 2003), and may even alter regional rainfall patterns (Andreae et al., 2004). Significant progress on globally applicable models of fire represented in DGVMs has been made since TAR (Thonicke et al., 2001; Fosberg et al., 1999) and suggests increases in fire frequencies (e.g. Stocks et al., 1998) with associated impacts (4.4.1, 4.4.5). The implications of the importance of fire globally are manifold (Bond et al., 2005). Firstly, fire suppression strategies are often limited in effect (Schoennagel et al., 2004; Keeley, 2002; Van Wilgen et al., 2004). Therefore management needs to incorporate adaptation as well as protection options, as enhancement of vegetation flammability by more prevalent fire weather and resulting big wildfires threatens human settlements, infrastructure, and livelihoods (e.g. Allen Consulting Group, 2005). Secondly, in some ecosystems, including islands, introduced fires have transformed forests into more flammable shrublands and grasslands (Ogden et al., 1998). Thirdly, the drivers of flammability, such as ecosystem productivity, fuel accumulation, and environmental fire risk conditions, are all influenced by climate change (Williams et al., 2001).
Invasive alien species (IAS) are a major threat to biodiversity (Sala et al., 2000; Vitousek et al., 1997a). Causes of biological invasions are multiple and complex (Dukes and Mooney, 1999), yet some simplifying approaches have been developed (Crawley, 1989; Chytry et al., 2005; Deutschewitz et al., 2003; Facon et al., 2006). Complex interactions may have unexpected outcomes; for example, after invasive rabbits were eradicated from the Kerguelen Islands, recent drying slowed native vegetation recovery, allowing exotic vegetation to become dominant (Chapuis et al., 2004). Change in biotic and/or abiotic disturbance regime is a primary driver of IAS (Le Maitre et al., 2004), and communities become generally more susceptible to invasion following extreme events (Smith and Knapp, 1999). Human-disturbed habitats in Britain have been shown to be rich in alien species (Crawley et al., 1996). IAS can also introduce disturbance regimes by enhancing vegetation flammability (Brooks et al., 2004). Overall, ongoing shifts in human mediated disturbances, insect pests, IAS and fire regimes can be seen to interact to alter regional vegetation structure, diversity and function (e.g. Timoney, 2003).

4.3 Assumptions about future trends

Climate models remain the tool of choice for constructing future climate scenarios (WG I chapter 8) used to force ecosystem or species-based impacts models. Simple global or regional temperature and precipitation scenarios are usually sufficient to design experimental manipulations to test ecosystem and species impacts and sensitivities empirically, though these more recently include also manipulations of rainfall variability.

Box 4.2: Generating bioclimatic scenarios to quantify ecosystem impacts

To assess future impacts on ecosystems we need scenarios based on projections of future climate. Recent advances have enabled to generate such scenarios not only for the entire globe but also for particular regions of interest. For an IPCC SRES A2 scenario (Nakicenovic et al., 2000) the EU-PRUDENCE study (Christensen et al., 2002) projected 4°C warming for much of Europe by 2071-2100. This is expected to be accompanied by more short-duration but intense convective rainfall events (e.g. 30% frequency increase for a 2°C warming Frei et al., 1998), plus a general northward migration of climatic zones, leading in some southern Mediterranean areas to a warming that exceeds 8-10°C with concomitant droughts. A similar tendency of simultaneous increases in the frequencies of both drought and intense precipitation events has also been reported for other mid-latitudes, e.g. for North America (e.g., Houghton et al., 2001; Trenberth, 1999). Such changes can have significant repercussions for hydrology and ecosystems, with extreme temperatures likely to exert stronger controls on evaporation or desiccation, heat and water stress on plants than mean temperatures (Fuhrer et al., 2007). In complex terrain such as the Alps, scenario generation faces particular challenges, since...
ecosystem processes depend on steep environmental gradients and operate on fine spatial scales (Gyalistras et al., 1994; Fischlin and Gyalistras, 1997), e.g. beta diversity tends to be greater than in flat areas (cf. Körner, 2000). Systematic climate model errors may exceed the climate change signal (Gyalistras et al., 1994; Beniston, 2003; Antic et al., 2006). Nevertheless, nested, high-resolution RCM simulations (5-km and 1-km scales) simulate in a physically consistent manner processes such as surface runoff, infiltration, and evaporation (e.g., Arnell, 1999; Bergstrom et al., 2001), extreme precipitation events (Frei et al., 1998), and damaging wind storms (Goyette et al., 2003). Preliminary results from simulations of extreme winter storms (e.g. 1999 Lothar storm Wernli et al., 2002; Goyette et al., 2003) indicate a future increased frequency of strong winds originating in the Atlantic at the expense of Föhntype storms related to southerly flow across the Alps. Sensitivity of forests to wind-stress increases markedly if return time of wind-storms decreases (Thurig et al., 2005a; Fuhrer et al., 2007) thereby hindering recovery of forest ecosystems in particular if also subject to pollution from nitrogen deposition and acidification (Braun et al., 2003).

Climate change is one of several interacting forces of global change, but these are often not well integrated in impact studies. For example, despite the recognized importance of land use change, it is rarely included in current global climate models (Hansen et al., 2001; Zebisch et al., 2004; Holman et al., 2005b; Levy et al., 2004; Pielke Sr, 2005; Feddema et al., 2005). This is likely due to an insufficient understanding of the underlying causes, trends and outcomes 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 a better understanding of underlying processes and development of useful model projections (Lambin et al., 2003). Several studies incorporating land use change have been undertaken at local and regional scales (section 4.4.10). Explicit inclusion of land use change in climate change analyses may reveal unexpected outcomes, in particular for biodiversity (Hansen et al., 2001). Consequently, many impact studies of climate change that ignore land-use and other global change trends may represent conservative estimates of projected ecosystem responses.

4.4 Key future impacts and vulnerabilities

To project impacts on ecosystems to projected climatic changes, many assumptions can and are being made. They give rise to basically three approaches: (i) correlative, (ii) mechanistic, and (iii) analog 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.

4.4.1 Biogeochemical cycles and biotic feedback

The cycling of chemical elements and compounds underpins the function of the biosphere and links ecosystems and climate by regulating chemical concentrations in soil, biota, atmosphere and ocean. Substantial progress has been made since TAR in understanding the interactive responses of ecosystems and the climate system, as determined by plant physiological responses, interactions with soil, and their scaled-up effects on regional and global biogeochemical cycles (Gedney et al., 2006;
Buchmann, 2002). The most advanced tools to achieve this scaling-up to global scale are Dynamic Global Vegetation Models (DGVMs), that simulate time-dependent changes in vegetation distribution and properties, and allow mapping of changes in ecosystem function and services (Metzger et al., 2006; Schroter et al., 2005). Reliability of results has improved in relation to previous generations of models but several aspects remain incompletely tested. Yet, validation is ongoing (e.g. Woodward and Lomas, 2004b; Prentice et al., 2006), and testing at hierarchical levels from leaf to biome and over relevant time scales have shown encouraging matches with observations (Lucht et al., 2002; Bachelet et al., 2003; Harrison and Prentice, 2003; Kohler et al., 2005; Gerten et al., 2004; Joos and Prentice, 2004; Peylin et al., 2005).

Key vulnerabilities as from TAR: Based on methods that predate and developed into DGVMs (equilibrium biogeography models or global biogeochemical models Neilson et al., 1998), the world’s terrestrial ecosystems will likely continue to sequester carbon for a number of decades and possibly throughout the 21st century, with an initially ‘greening’ world due to longer growing seasons, more precipitation, and CO2 fertilization benefits. Substantial structural changes in biomes could occur towards 2100 with ecosystem shifts toward higher latitudes and altitudes as CO2-fertilization benefits saturate and increasing temperature effects on respiration and transpiration could reverse initial carbon sequestration gains resulting in net global ecosystem carbon losses relative to today (e.g. Cramer et al., 2001). Dieback of much of the Amazon rainforest, due to desiccation, was an identified major vulnerability, but with a high degree of uncertainty. TAR concluded the net global terrestrial carbon exchange would be between -6.7 PgC yr⁻¹ (uptake) and +0.4 PgC yr⁻¹ (loss), and that anthropogenic CO2 emissions would remain the dominant determinant of atmospheric CO2 concentration during the 21st century. Key ecosystem forecasting needs identified in TAR were for spatially and temporally dynamic models containing all processes that produce inertia and lags in ecosystem responses, allowing reproduction of non-linear behaviour if conditions reach critical levels that feedback to and interact with climate evolution.

Impacts: Scaled-up effects of direct atmospheric CO2 enrichment on plant and ecosystem biomass accumulation (CO2 fertilization) are largely responsible for the enhanced NPP in current global models (Leemans et al., 2002). However, based on improved experiments, magnitudes of this effect are being revised downwards. Three constraints increasingly acknowledged are element stoichiometry (nutrients), forest tree dynamics, and secondary effects of CO2 on water relations and biodiversity. Trends in empirical data suggest caution when estimating future carbon sequestration potentials of the biosphere as a contribution to mitigating climate change, in particular as these benefits may be smaller than counteracting impacts of land use change. Persistent grassland responses to elevated CO2, which range from 0 to 40 % biomass gain per season, mainly reflect CO2-induced water savings induced by scaled-up impacts of reduced stomatal conductance (Morgan et al., 2004; Gerten et al., 2005), and thus rely on current moisture regimes and lack atmospheric feedback. The only replicated test of multiple CO2 x climate/environment interactions (water, temperature, nutrient supply) yielded no overall CO2 biomass signal (Shaw et al., 2002) highlighting significant influence of co-limiting environmental variables. Current best estimates for stimulation of agricultural production have also been reduced to 7-12 % for CO2 doubling of pre-industrial CO2 concentrations (Kimball et al., 2003; Kimball et al., 2002). Similar trends are emerging for forests, although the interpretation is complicated by time lags in biomass response to the artefactual step-change when initiating CO2 treatments, requiring long observation periods before a new steady state is reached. All three tall forest test systems, loblolly pine plantation (Oren et al., 2001; Schafer et al., 2003), sweet gum plantation (Norby et al., 2002; Norby and Luo, 2004), and mixed deciduous forest (Körner et al., 2005a) exhibit significant initial biomass stimulation, rapidly diminishing with time except for one of the four pairs of test plots (treatment vs. control) in the joint Duke pine experiments (Schafer et al., 2003). Scrub oak in Florida shows similar diminishing responses as treatment proceeds (Hungate et al., 2006), even though this is a post-fire
regenerating system. All other test systems with trees are vigorously expanding and too young to permit analogies for steady state forest biomass responses. However, it has been suggested that greatest CO₂ fertilization impacts may be seen in such systems (Bond and Midgley, 2000; Bond et al., 2003), especially where nutrients and water are less limiting and trees require carbon reserves to re-establish in fire-prone grasslands (see section 4.2.2). However, such gains in early life stage growth rate may not translate into greater landscape carbon sequestration which is largely determined by recalcitrant carbon pools in soils and long-lived biomass. For tropical forests, the planet’s single largest biomass carbon reservoir, post-industrial atmospheric CO₂ enrichment seems to have enhanced growth dynamics (Phillips et al., 2002; Wright et al., 2004). A more dynamic forest is likely to store less rather than more carbon in future (Korner, 2004), especially given the exceptional CO₂ responsiveness of tropical lianas that may increase tree mortalities and population turnover.

Best estimates of CO₂-induced water savings in forests due to reduced stomatal aperture range between 5 and 15 % (Wullschleger and Norby, 2001; Cech et al., 2003) for humid soils and good weather conditions, diminishing with drying soils. Desert shrub systems profit from CO₂ enrichment only during exceptional wet periods and not in dry periods (Nowak et al., 2004), contrasting with earlier expectations. Evapotranspiration rate data for temperate zone ecosystems under future CO₂ scenarios suggests that these may be reduced by less than 10%, across all weather conditions. Water savings through elevated CO₂ have very little effect on trees during drought, because benefits disappear after a few days when the nutrient cycle in the top soil becomes interrupted, and when the initial water savings are used up (Leuzinger et al., 2005). Repeated drought associated with high temperatures as experienced by Europe in 2003 may reduce landscape-wide carbon stocks (Ciais et al., 2005). Experimental data for intact ecosystems suggest far smaller hydrological effects than those derived from gas exchange theory (Gedney et al., 2006). These effects should become further diminished as atmospheric feedback resulting from reduced evaporation comes into play.

Soil nitrogen availability may be key to predict future carbon sequestration by terrestrial ecosystems (Reich et al., 2006), especially in the light of global N-deposition trends (Matson et al., 2002). Loblolly pine forest (Lichter et al., 2005) and grassland experiments (Van Kessel et al., 2000) have significantly reduced hopes for CO₂-fertilization driven carbon accumulation in soils, probably because carbon sequestration to humus is more nutrient demanding (in particular of nitrogen), than is e.g. wood formation (Hungate et al., 2006). Carbon accretion in soil would therefore exert negative feedback on plant growth by immobilizing soil nutrients and as more multi-year study results become available, acclimation effects reducing the CO₂-fertilization indicate a faster diminishing of the biospheric sink than previously assumed (Fig. 4.2, Reich et al., 2006) and used in model projections (Fig. 4.2, e.g. Scholze et al., 2007). Build-up of seasonally transitory soil C-pools such as in fine roots has been found to increase net primary production, but the generality of such enhanced C-fluxes and what fraction might add to soil carbon stocks remain unresolved (Norby et al., 2004). Soil warming has raised fears of carbon release, but whether this alone will diminish soil humus pools is an open question. Because soil respiration is largely driven by net primary production, there is no obvious reason for a decoupling of respiration from carbon fixation in hot climates (e.g. Eliasson et al., 2005), though in regions with thawing permafrost and lengthening of an otherwise cool season (reduced periods of anaerobic soil conditions) a decay of previously accumulated soil carbon stocks could release large amounts of carbon to the atmosphere. Accordingly, recent observations show widespread carbon losses from soils (Bellamy et al., 2005; Schulze and Freibauer, 2005), though by contrast, soil respiration rate under in situ boreal forest trials increased due to warming but rapidly slowed once labile soil-C had equilibrated under warmer conditions (Eliasson et al., 2005).

Changing fire regimes (see 4.2.2) are of significant concern for terrestrial carbon balance. Simulation of fire has improved substantially (see 4.4.2), and some models simulate individual fire event impacts on ecosystems (Lenihan et al., 1998; Thonicke et al., 2001). Yet, individual fire-generated age-classes are not explicitly simulated (Smith et al., 2001; Moorcroft et al., 2001) thus projecting CO₂ effects...
inconsistently with implied age-class structure. Small-scale fluxes from heterogeneous vegetation can be summed for coupling to an atmospheric model (Woodward and Lomas, 2001), for future development. DGVMs also assume rapid migration of species that comprise the generic “plant functional types” that are modelled to shift in geographic distribution (e.g. Leemans and Eickhout, 2004), thus modelled carbon sequestration gains from northward migration of the boreal forest (Neilson et al., 2005) are likely to be overoptimistic (see 4.4.5).

Ecosystem changes associated with land use and land cover change (see 4.2.2) are complex, involving a number of feedbacks (Lepers et al., 2005; Reid et al., 2005). For example, conversion of natural vegetation to agricultural land drives climate change by altering regional albedo and latent heat flux, causing additional summer warming in key regions in boreal and Amazon region, and winter cooling in the Asian boreal zone (Feddema et al., 2005), by releasing CO2 via losses of biomass and soil carbon (Levy et al., 2004; Gitz and Ciais, 2003; Canadell et al., 2004), and through a ‘land use amplifier effect’ (Gitz and Ciais, 2003). 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 (Jones and Donnelly, 2004; Lal, 2004; Lal, 2003; de Koning et al., 2005; Wang et al., 2004a; King et al., 2004a), but are limited by decades-long time-lags to replace old-growth forests.

Ecosystems are likely to respond to increasing external forcing in a nonlinear manner. Most initial ecosystem responses appear to dampen change (Aber et al., 2001), but amplify it if thresholds in magnitude or rate of change are crossed. Transitions between states may be triggered or the ecosystem may even collapse (Rietkerk et al., 2004; Scheffer et al., 2001; Schroder et al., 2005).

The sequestration and cycling of carbon in terrestrial ecosystems is a key concern, given the above drivers and concerns about threshold impacts. Recent work with DGVM approaches has begun to elucidate the likelihood of occurrence of important thresholds, and positive feedback to the atmosphere through carbon release from ecosystems. Global estimates (IS92a, HACM2-SUL Cramer et al., 2001) suggest a reduced global sink relative to that expected under CO2 fertilization alone, both in 2000 (0.6± 3.0 PgCy±1) and 2100 (0.3±6.6 PgCy±1) as a result of climate change impacts on Net Ecosphere Productivity (NEP) of tropical and southern hemisphere ecosystems; the rate of increase of NEP slows around 2030 as CO2 fertilization itself saturates, and in four of six models show further, climate-induced NEP declines, due to increased heterotrophic respiration and declining tropical NPP post-2050. These trends could develop for a century beyond 2100, even with stabilized atmospheric CO2 concentration and instantaneously stabilized climate. 20th century climate, nitrogen deposition and CO2 change alone may have increased carbon sequestration by up to 2500 gCm-2 in tropical and boreal forest regions, but the terrestrial biosphere could become a net CO2 source after CO2 fertilization saturates post-2020 (HadCM3 with additional CO2 and warming feedback from global vegetation response, Woodward and Lomas, 2004b). More recent modelling based on projected deforestation and climate change (IS92a, CGCM1, CSIRO, ECHAM, HadCM3) in the tropics alone suggest an additional release of 101 to 367 Pg C, adding between 29 and 129 ppm to global atmospheric CO2 by 2100, with much of this, on balance, sourced from deforestation rather than climate change (Cramer et al., 2004). Climate scenario uncertainty provides a substantial variance in global terrestrial C balance by 2100, even under a single CO2 emission scenario (IS92a, reaching 703 ppm atmospheric CO2 concentration by 2100, excluding vegetation feedback), with five GCMs driving DGVM estimates of global terrestrial C-sequestration of between -106 to +201 Pg C (Schaphoff et al., 2006), though in four out of five, the sink had begun decreasing from a peak well before 2060. A risk assessment for terrestrial biomes and biogeochemical cycling shows that a terrestrial carbon source is predicted in almost half of 52 GCM x emissions scenario combinations (Scholze et al., 2007), and that wildfire frequency increases dramatically even for 2100 warming of <2°C.

Here we show model results for the most recent version of the DGVM LPJ (Schaphoff et al., 2006) of...
shifts in biome structure and the terrestrial carbon sink under more recent IPCC emission scenarios SRES A2 and B1 (Nakicenovic et al., 2000). This supports projections of diminishing terrestrial C-sequestration as soon as 2030 (Fig. 4.2), far earlier than suggested in TAR (WG I, Figure 3.10), and substantial shifts in biome structure (Fig. 4.4).

**Fig. 4.2:** Net carbon exchange of all terrestrial ecosystems as simulated by the dynamic global vegetation model (DGVM) LPJ (Gerten et al., 2004; Sitch et al., 2005 - negative values mean a carbon sink, positive values carbon losses to the atmosphere). Past century data are based on observations and climate model data were normalised to be in accord with these observations for the period 1961-1990 data (CRU-PIK). Transient future projections are for the IPCC SRES A2 and B1 emission scenarios (Nakicenovic et al., 2000) forcing the climate models HadCM3 and ECHAM5, respectively, (cf. Schaphoff et al., 2006). In contrast to previous global projections (TAR WG I, Figure 3.10) the world’s ecosystems tend to become a carbon source earlier (here sink peak ~2030) and more consistently, corroborating other projections of increased forcing from biogenic terrestrial sources (e.g. Cox et al., 2000; White et al., 2000a; Cox et al., 2004; Schaphoff et al., 2006; Scholze et al., 2007 - see Fig. 4.4 for maps on underlying ecosystem changes and Table 4.3). Note, these projections assume an effective CO2 fertilization (cf. 4.4.1).

Impact modeling (Fig. 4.2) and modeling that dynamically links the physical climate system and vegetation, using Ocean-Atmosphere-General Circulation Models (OAGCMs, e.g. Cox et al., 2000) projects a terrestrial C source that will exacerbate both climate and further vegetation change to some degree (e.g. Sarmiento, 2000; Dufresne et al., 2002; Canadell et al., 2004). Impacts include the collapse of the Amazon forest (e.g. White et al., 2000a; Cox et al., 2004), and an overall C source from the tropics that exceeds the boreal C sink (Berthelot et al., 2002), leading to an 11% (Cox et al., 2000) to 40% (Dufresne et al., 2002) higher atmospheric CO2 concentration by 2100. Carbon and water cycling, at least, are also affected by shifting biogeographic zones (Gerten et al., 2005) that will be lagged by migration constraints that are not yet incorporated in DGVM approaches (see also 4.4.5 and 4.4.6), leading to a potential overestimation of vegetation C-sequestration potential, especially for regions such as the boreal, that are projected to benefit from in-migration of woody vegetation, while biomes that experience dieback will do so with shorter time lags (Neilson et al., 2005).

Interactions between ocean and atmosphere and land and oceans may also be critical for future
evolution of climate. For example, dimethyl sulphide (DMS) is a significant source of cloud 
condensation nuclei and changes in air-sea fluxes caused by global warming at 2 × CO₂ can result in 
regional radiative impact, from −15% to 30% and impact regional climate (Bopp et al., 2004; Bopp et 
al., 2003). DMS produced by coccolithophores may be influenced by their sensitivity to high seawater 
CO₂ (Riebesell et al., 2000). As the largest producer of calcite on the planet (Holligan et al., 1993) 
reduced calcification may also influence the global carbon cycle (Raven et al., 2005) and the albedo 
effect of the Earth (Tyrrell et al., 1999). N₂O from marine origin contributes ~33% of total input to the 
atmosphere (Prather and Enhalt, 2001). Changes to the concentration and distribution of oxygen in the 
oceans, either through increased stratification of the surface waters (Sarmiento et al., 1998) or through 
a decrease in the strength of thermohaline circulation (Houghton et al., 2001), will impact the ocean 
nitrogen cycles, especially the processes of nitrification and denitrification which promote N₂O 
production. The extent to which the unanticipated and recently discovered methane release from plant 
foliage (Keppler et al., 2006), can be scaled from individual leaf to biome level, is an ongoing debate.

4.4.2 Deserts

Characteristics: The largest terrestrial biome, comprising extra-polar regions with mean annual 
precipitation <250mm and unfavourable precipitation to potential evaporation ratio (Table 4.1 
Nicholson, 2002; Warner, 2004; Reid et al., 2005). Deserts are sparsely populated with rural human 
population densities of between 1 and 3 km⁻², and among the lowest GDP of all ecosystems (Reid et 
al., 2005).

Goods and services: Food, including rangeland grazing and wild foods; Ecotourism; Cultural and 
sense of place; Genetic resources, especially of arid-adapted species (Hassan et al., 2005). Biodiversity 
(especially winter-rainfall deserts e.g. Myers et al., 2000). Air quality, atmosphere composition and 
climate regulation (Hassan et al., 2005); Wind-blown dust and desert albedo influences regional 
rainfall and biogeochemistry of remote terrestrial and marine ecosystems (Lioubimtseva and Adams, 

Key Vulnerabilities as from TAR: Deserts were grouped with semi-arid and dry sub-humid rangelands 
by TAR. Noted vulnerabilities (TAR p.239) included possible region-specific increases in productivity 
due to increases in rainfall and rising atmospheric CO₂, potentially offset by management and rising 
temperatures; some regions of desert expansion; extensive animal production threatened by rangeland 
over-utilization; secondary salinization in dry lands partly offset by elevated CO₂ and and associated 
increased biomass and soil organic matter in deserts (Poorter and Perez-Soba, 2001; Lioubimtseva and 
Adams, 2004); overall lack of infrastructure and investment in resource management.

Impacts: Despite substantial disagreement among GCM projections, future rainfall change in deserts 
seems likely to be season-specific, and inter-annual variation will increase. Mid-continent deserts 
could experience more severe, persistent droughts (Lioubimtseva and Adams, 2004; Schwinning and 
Sala, 2004). Combined with elevated CO₂, these changes are likely to favour desert shrub species and 
this may further increase evapo-transpiration (Bassirirad et al., 1997; Smith et al., 2000). Deserts may 
experience more episodic climate events in future (Smith et al., 2000). Deserts of the southwest USA 
may shrink by up to 60% (Bachelet et al., 2001; Hardy, 2003; Duraiappah et al., 2005). Desert 
biodiversity will be vulnerable to climate change (Reid et al., 2005): In the Chihuahuan desert habitats 
of about half the species could be eliminated (Lenihan et al., 2003), while in the Succulent Karoo, 
doubled CO₂ climate scenarios lead to a median 19.2% endemic species extinction for a range of 
assumptions (Malcolm et al., 2006). Daytime in situ warming experiments suggest high vulnerability 
of endemic succulent growth forms of the Succulent Karoo to high-end warming scenarios for 2100 
(mean 5.5°C above ambient), inducing appreciable mortality within only a few months (Musil et al., 
2005). Indeed, Namib desert tree aloes (Aloe dichotoma) show latitudinal trends in mortality, highest
at the equatorward edges of their range, that suggest ongoing directional range contraction in response
to increasingly unfavourable water balance (Foden et al., 2007).

Desert species that depend on rainfall events to initiate breeding will be severely affected as will be the case for migratory birds although others suggest that resident bird communities may be more vulnerable (Dukes and Mooney, 1999; Myers et al., 2000; Hardy, 2003). In South Africa, a wide-ranging bird species of the Nama-Karoo desert species, the Mountain Wheatear, was projected to lose 51% of its bioclimatic range by 2050 (HadCM3 A2 Simmons et al., 2004). In contrast, desert reptile species could be favoured by warming as will be the case for amphibians (depending on rainfall scenarios Currie, 2001).

Extremely wet periods will result in high vulnerability to invasive alien species and subsequent fire outbreaks and this combined with land use disturbances will increase vulnerability to desertification (Dube and Pickup, 2001; Holmgren and Scheffer, 2001; Geist and Lambin, 2004; Lioubimtseva and Adams, 2004; Dukes and Mooney, 1999; Brooks et al., 2004). Vulnerability to desertification will be enhanced due to existence of shallow soils with higher soluble salts and the slow recolonization of soil surfaces by different algae components (Evans and Belnap, 1999; Johansen, 2001; Billings et al., 2003; Duraipappah et al., 2005). Very low biomass (below 14% threshold limit) will make Kalahari desert dune system susceptible to aeolian erosion (Thomas and Leason, 2005), and with regional warming of between 2.5 to 3.5°C most dune fields could be reactivated by 2100 (Thomas and Leason, 2005). Increased dust flux may increase aridity and suppress rainfall outside deserts, with opposite effects under wetting scenarios (Bachelet et al., 2001; Hardy, 2003; Prospero and Lamb, 2003; Lioubimtseva and Adams, 2004), leading to indirect effects on the vulnerability of remote regions to climate change.

Key vulnerabilities: Winter-rainfall desert vegetation communities are vulnerable to drier and warmer conditions (Malcolm et al., 2006; Simmons et al., 2004). Up to 50% extinctions could occur with habitat elimination in the Chihuahuan desert (Lenihan et al., 2003), while doubled CO2 climate scenarios lead to a median 19.2% species extinction in South Africa’s succulent Karoo Biome (Malcolm et al., 2006). Using climate models and assuming a climate warming of 1.5 to 2°C about one third of the Sahel aridified by ~2050 t and vegetation zones generally shifted Equatorward (Box 4.3), though alternative climate scenarios showed less pronounced changes (van den Born et al., 2004). Changing rainfall amounts and variability might favour the greater success of shrub species, and together with rising CO2 may increase cover (Bachelet et al., 2001; Hardy, 2003; Duraipappah et al., 2005).

Box 4.3: Drought and Sahel ecosystems

The Sahel, a region bounded by the Great Sahara desert in the north and the Sudanian savanna in the south, has experienced large climate shifts in the past 6,000 years: during the mid-Holocene, wet conditions supporting mesic vegetation communities and abundant wildlife became markedly drier. A second change followed prolonged drought after 1969, and is used here, with caution as an analogue for potential future impacts of climate change (Foley et al., 2003; ECF, 2004 WGI chapter 11). The major limitation in this approach is the lack of ecological changes dating back to before the drought began (Taylor et al., 2002; Hein and Ridder, 2007). Under this recently established climate regime, the Sahel supports lower biomass than other arid lands with similar annual rainfall, e.g. the Kalahari Desert (Nicholson, 2002). Available water for plants is limited by soils that are prone to crusting and the highly unpredictable rains from seasonal monsoonal system (coefficient of variation > 20-30%, Hulme, 2001). Rains fall as high intensity convective events during the warm summer and, result in high potential evapo-transpiration (Turner, 1999; Hiernaux and Turner, 2002; Nicholson, 2002). Very
low levels of nitrogen and phosphorus limit vegetative growth during wet years. Except for the transition zones in the south, the Sahel vegetation lacks the succulent, perennial grass and shrub forms typical of similar environments (Nicholson, 2000; Hiernaux and Turner, 2002). Instead, herbaceous species dominate as an annual grass layer, with few scattered trees and shrubs, due to the severity and long duration of the dry season (Lejeune et al., 2004). Adaptive traits of annual plants to drought include short growing season, a large investment seed production – most of which are transient- and high tillering ability. Variation in yield and species composition is related to rainfall, effects of grazing history on soil properties and topography (Turner, 1999; Hiernaux and Turner, 2002). There are substantial variations in rain-use efficiency (RUE) between years for a given site. For E.g. C3 Leguminous and other dicotyledonous species with lower RUE increase in wet years contributing to reduction of the RUE of the plant community over these periods (Hein and Ridder, 2007). Others suggest that the relatively constant RUE found in remote sensing studies (1982 -1990s) over different years may indicate a process of human induced degradation of the plant cover (Hein and Ridder, 2007). Atmospheric dust loads increase during drought years (30–50% of the total atmospheric dust loading, Nicholson, 2000).

Woody plant species in the Sahel have a diverse phenology but have shown drought induced mass mortality, and regeneration capacity in wet periods (Gonzalez, 2001; Hiernaux and Turner, 2002). Woody plants are increasingly being subjected to greater pressure from an increase in the proportion of small ruminants replacing tradition cattle as an adaptation to severe droughts in addition to production of charcoal and periodic wildfires (Woomer et al., 2004). Recovery of land productivity from drought has been found for the Sahara/Sahel boundary tracking inter-annual fluctuations of rainfall. Infiltration and aquifer recharge occurred annually following the severe droughts (about 2 to 20% of rainfall) and this is believed to contribute to recovery of vegetation (Nicholson, 2000). However, extreme drought contributed to the decline of many Palearctic migratory birds that wintered in the savanna and steppe zones of the Sahel (Gitay et al., 2001). A decrease in Sahel biodiversity and in some cases reactivation of vegetated dunes has also been noted (ECF, 2004).

**Adaptation costs and opportunities:** Taking advantage of for instance, wildlife species that will be favoured under climate change such as reptiles and amphibian richness, which can be accomplished through promoting eco-tourism (Currie, 2001; Reid et al., 2005). Other eco-tourism related opportunities could include landform changes such as remobilised dunes (Thomas et al., 2005). But the cost of these activities could be high due to lack of water, unbearable temperatures and increased problems of dust. While in wetter periods and also increase in temperature in cold deserts could provide opportunities for crop production.

There are no clear insights on adaptation option for migratory birds that cross deserts. For human needs autonomous adaptation could be cost effective (Allen Consulting Group, 2005). Because of unsuitable conditions for agriculture, deserts are current less transformed by human activity than other biomes. As a result a cost effective adaptation strategy might be to maintain deserts as there are i.e. with limited intervention (Balmford et al., 2002). Where soil degradation has occurred, restoration efforts might provide carbon sequestration potential in soil and above ground biomass, but might be costly (Duraiappah et al., 2005).

**Implications for policy and sustainable development:** If desert climates become more variable there will be greater need to incorporate climate change projections in designing future management policies. The effect of ecosystem degradation (desertification) on remote biomes i.e. through dust loads should also be considered in development plans. Sustainable utilization of deserts will require land use systems that minimize the destruction of soil to maintain water and nutrients cycles for plant growth (Duraiappah et al., 2005). Prevention of land degradation might be cheaper than soil restoration.
Programs that focus on adjusting timing and intensity of grazing and fires where appropriate will minimise disturbances. Community participation in decision making and management along with public policy will be critical for adapting for E.g. adjusting stocking rates given the social value of animals in most desert societies, but also in addressing issues of diversification i.e. from livestock systems to eco-tourism (Duraiappah et al., 2005).

**Impacts summary as a function of ΔT:**

- Median 19.2% endemic species extinction under a range of biological response scenarios and doubled CO₂ climate scenarios from GISS, GFDL-R30, OSU, UKMO, MPI-T106, HadCM2GHG, HadGCM2-SUL (Malcolm et al., 2006).
- Karoo Desert bird species, Mountain Wheatear, projected range loss of 51% by 2050 (HadCM3 A2 Simmons et al., 2004).
- Thirty percent of Sahel aridified, vegetation zones shifted Equatorward; warming of 1.5 to 2°C (van den Born et al., 2004).
- Kalahari desert dune system remobilized with regional warming of between 2.5 to 3.5°C (Thomas and Leason, 2005).

### 4.4.3 Grasslands and savanna

**Characteristics:** These include tropical C₄ grasslands and savannas (C₄ grass-dominated with 10-50% tree cover, ~28 million km²) and temperate C₄ and/or C₃-grass and herb-dominated grasslands (15 million km²) (Bonan, 2002). These systems are disturbance-controlled, by fire (Bond et al., 2005) and/or grazing (Fuhlendorf et al., 2001; Scholes and Archer, 1997). These disturbance regimes are often managed, though fire regimes depend on seasonality of ignition events and rainfall (Brown et al., 2005b).

**Goods and services:** (Hassan et al., 2005) Supporting services – soil preservation, soil nutrient cycling and conservation, and primary production; Regulating services – water and climate regulation; Provisioning services – food (rangeland grazing and wild foods, fuel, fibre and construction material, biochemicals, freshwater; Cultural services – recreation and ecotourism, extensive animal diversity, especially in Africa, that in turn support nature-based tourism, wildlife ranching, and hunting.

**Key vulnerabilities as from TAR:** Reviewed as rangelands in TAR, and included deserts. Major threats include fragmentation of land cover, over-grazing and changing fire regimes; extensive ranching threatened by climate change, though greater threat from land-use change; impacts of rising CO₂ on ecosystem vegetation structure and fuel availability, with possible increases in carbon sequestration dependent on management regimes. Soil stability and desertification, especially on arid margins. Lack of infrastructure and investment in resource management increases vulnerability.

**Impacts:** Climate and atmospheric change will affect grassland and savanna ecosystem function and species composition through the relative physiological and growth responses of their component species and growth forms to rainfall change, CO₂ fertilization, warming, and emergent responses of herbivory and fire regime. Few experimental approaches have assessed ecosystem responses to multifactorial treatments such as these (Norby and Luo, 2004). Experiments on warming, rainfall change or atmospheric CO₂ level are virtually absent in savannas; many field-based ecosystem studies are confined to grasslands (Rustad et al., 2001).

Temperate and alpine grassland responses to future climate change may be strongly controlled by hydrological cycle and water balance change (Novick et al., 2004; Zha et al., 2005). A Canadian grassland fixed roughly 5x as much carbon in a year with 30% higher rainfall, while a 15% rainfall reduction led to a net carbon loss (Flanagan et al., 2002). A European grassland showed only minor...
Increasing dominance of C4 grasses at the expense of C3 forms such as herbs, shrubs and trees may occur in response to warming (Epstein et al., 2002), relevant for C4 invasions accelerated by human disturbance and fire (Sage and Kubien, 2003). Some African savanna tree phenologies appear sensitive to seasonal air temperatures (Chidumayo, 2001). Less than 4°C warming show woody vegetation expanding with positive carbon sequestration in North America, but waxing with greater warming, partly due to fire, with up to 50% expansion of savannas at the expense of forests (Bachelet et al., 2001). Warming of a mixed C3/C4 New Zealand grassland to simulate extreme heat wave events increased C4 species success within a single growing season, but reduced productivity by over 60% where C4 plants were absent (White et al., 2000b). Drying, increased wind speed, and warming of 2.5°C to 3.5°C by 2100 reduce vegetation cover and remobilize Kalahari dune systems in southern Africa as early as 2040 (Thomas et al., 2005).

Transient system responses to the IS92a scenario reduced mid-latitude carbon sinks (partly in savanna systems) that offset increasing carbon sinks at high latitudes, leading to a net global carbon source from vegetation (White et al., 2000a). This effect was muted and delayed by more than a century if atmospheric [CO2] stabilized at 550 ppm. Suggestions that rising atmospheric CO2 would differentially favour C3 types in mixed C3/C4 grasslands have not been conclusively supported by experiments (Wand et al., 1999). In short-grass prairie, field CO2 fumigation and 2.6°C warming increased production by 26-47%, regardless of photosynthetic type (Morgan et al., 2001a). In C4 tropical grassland, decades-long CO2 enrichment also revealed no increase in C3 success (Stock et al., 2005). Both field studies showed increased soil water content at elevated CO2 (see also Ferretti et al., 2001). Elevated CO2 stimulation of productivity in chalk grasslands can be attributed to effects on stomatal conductance and short term increased soil water (Niklaus and Koerner, 2004). Increasing temperature and rainfall changes may override the potential benefits of rising CO2 for C3 relative to C4 grasses (Winslow et al., 2003). In Namibian savannas, rising CO2 may favour shrubs and herbs at the expense of C4 grasses (Thuiller et al., 2007a). Current and future atmospheric CO2 levels may reduce the historic vulnerability of Sahelian systems to drought, by facilitating increased primary productivity and water balance through CO2 fertilization (Wang and Eltahir, 2002). Regional climate modeling indicates that the biological impacts of CO2 fertilization on grasslands may scale up to affect regional

### Responses to 3°C Rise in Temperature

A Mongolian steppe grassland switched from carbon sink to source in response to water stress (Li et al., 2005). Net ecosystem carbon exchange in a North Oklahoma grassland was dependent on intra and inter-seasonal rainfall change (Suyker and Verma, 2001; Suyker et al., 2003). Increased rainfall variability was more significant than rainfall amount for tall-grass prairie productivity (Fay et al., 2000; Fay et al., 2002), with 50% increased dry spell duration causing 10% reduction in NPP (Fay et al., 2003) and 13% reduction in soil respiration (Harper et al., 2005). Ecosystem models of C3/C4 mixed grasslands show positive NPP relationships with moderate rainfall variability (with unchanged MAP), but greater variability reduces both NPP and ecosystem stability (Mitchell and Csillag, 2001); empirical results for C4 grasslands show a similar monotonic relationship between NPP and rainfall variability (Nippert et al., 2006). Varying both winter and summer precipitation had immediate and carry-over effects on short grass prairie respiration rate (Chimner and Welker, 2005). Both plant cover and productivity reductions were simulated along an aridity gradient in southern African savanna in response to the drying trend of ~8 mm yr\(^{-1}\) since 1970 (MAR ranged from 299 to 918 mm yr\(^{-1}\) in 2000 Woodward and Lomas, 2004a). Changing savanna vegetation cover may also feed back to affect regional rainfall patterns. Modeled removal of savanna systems from global vegetation cover has larger effects on global precipitation than for any other biome (Snyder et al., 2004). 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). At the continental scale, modeled conversion of tropical forest to savanna reduced rainfall in tropical regions, but increased rainfall in central southern Africa (Semazzi and Song, 2001).

### Long-Term Responses to Climate Warming

Increasing CO2 stimulation of productivity in chalk grasslands can be attributed to effects on stomatal conductance and short term increased soil water (Niklaus and Koerner, 2004). Increasing temperature and rainfall changes may override the potential benefits of rising CO2 for C3 relative to C4 grasses (Winslow et al., 2003). In Namibian savannas, rising CO2 may favour shrubs and herbs at the expense of C4 grasses (Thuiller et al., 2007a). Current and future atmospheric CO2 levels may reduce the historic vulnerability of Sahelian systems to drought, by facilitating increased primary productivity and water balance through CO2 fertilization (Wang and Eltahir, 2002). Regional climate modeling indicates that the biological impacts of CO2 fertilization on grasslands may scale up to affect regional...
climate (Eastman et al., 2001).

Temperate grassland carbon sequestration likely increased strongly in response to increases of CO₂ from glacial to interglacial levels (Gill et al., 2002), but increases of 54% in net fixation expected CO₂ doubles (Hsieh et al., 2005). Both NPP and carbon stocks are modeled to increase in response to both elevated CO₂ and warming, except with positive precipitation changes at cool sites (Riedo et al., 2000). Elevated CO₂ impacts on temperate grassland carbon sequestration are also dependent on management (Harmens et al., 2004; Jones and Donnelly, 2004), and nutrient amendment, complicated by being species- but not functional-type specific (Niklaus et al., 2001; Hanley et al., 2004). Tropical savanna NPP is second only to that of tropical forests, yet they maintain only a quarter of the above-ground biomass of tropical forests because of fires, and are therefore a potential carbon sink contingent on 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., 2004b). Fire exclusion has the potential to transform savannas to forests (observed by e.g. Bowman et al., 2001), and could result in an increase of closed forest cover from current 27% to 56% of the vegetated surface of the world (Bond et al., 2005).

Elevated CO₂ impacts on temperate grassland carbon sequestration are also dependent on management (Harmens et al., 2004; Jones and Donnelly, 2004), and nutrient amendment, complicated by being species- but not functional-type specific (Niklaus et al., 2001; Hanley et al., 2004). Tropical savanna NPP is second only to that of tropical forests, yet they maintain only a quarter of the above-ground biomass of tropical forests because of fires, and are therefore a potential carbon sink contingent on 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., 2004b). Fire exclusion has the potential to transform savannas to forests (observed by e.g. Bowman et al., 2001), and could result in an increase of closed forest cover from current 27% to 56% of the vegetated surface of the world (Bond et al., 2005).

Tree-grass balance in savannas may be shifting towards trees with continuously rising atmospheric CO₂ concentrations, and diminishing ability of grasses to suppress ever faster growing tree saplings in grass fires (Bond and Midgley, 2000; Bond et al., 2003). However, there are no field tests of this hypothesis, and other factors such as long-term grazing pressure may explain observed shrub encroachment (Van Auken, 2000). Encroachment of woody plants in grasslands and savannas may increase carbon sequestration, not only by standing crop, but also by increased carbon sequestration in soils dried by greater transpiration (Huxman et al., 2005) especially in drier sites (e.g. McCarron et al., 2003); wetter sites lose enough soil carbon to offset greater standing crop (Jackson et al., 2002). Woodland expansion also reduces soil carbon turnover rate by almost 40% through soil cooling, leading to potential reduction in soil efflux of 19x10⁶ Mg C yr⁻¹ in the US Great Plains with canopy closure (Smith and Johnson, 2004).

Field experiments reveal impacts on soil, and nutrient controls of vegetation response. Long term CO₂ fumigation of southern African C₄ grassland revealed muted impacts on nitrogen cycling and soil C sequestration (Stock et al., 2005), in contrast to greater C sequestration in short term studies of grassland ecosystems (e.g. Williams et al., 2004a). Elevated CO₂ impacts on litter decomposition seem species-specific and relatively minor (Ross et al., 2002) but may cascade through the soil food web (Hungate et al., 2000). Acclimatization of soil respiration to moderate warming suggests a low feedback potential of ecosystem respiration to atmospheric CO₂ composition (Luo et al., 2001; Edwards et al., 2004). However, complex interactions between plants and fungal symbionts showed potential impacts on soil structure that may predispose them to accelerated erosion (Rillig et al., 2002). Warming of a tallgrass prairie showed increased plant growth, funding enhanced soil fungal success (Zhang et al., 2005).

Climate change impacts studies for savanna and grassland fauna are few. Substantial community compositional shifts of African mammal faunas have been projected for 2050 and 2080, including substantial increases in critically endangered and extinct species to between 10 and 40% of mammal species (Thuiller et al., 2006). For an African arid savanna raptor, general reductions in population success have been simulated under climate change scenarios (Wichmann et al., 2003). A 4% to 98% reduction in species ranges for ~80% of 179 animal species have been projected using IS92a-driven regionally interpolated climate scenarios, including many savanna birds and mammals (Erasmus et al., 2002). Observed population declines in three African savanna ungulates suggest that summer rainfall reductions could result in their local extirpation if regional warming continues (Ogutu and Owen-Smith, 2003).
Adaptation costs and opportunities: Similar to other chapters

Key vulnerabilities:
- Shifts in vegetation structure driven by water, CO₂ and temperature changes, with potential non-linear impacts with increasing climate change, and associated impacts on habitat-dependent fauna.
- Accelerating fire frequency and fire size due to more common warmer, drier fire danger weather, and greater fine herbaceous fuel accumulation with increasing rainfall and CO₂ fertilization.
- Projected shifts in species geographic ranges threatened by landscape fragmentation through land use change, especially of large mammals with large range requirements, threats to fauna limited in migration potential by restrictive protected area networks.

Impacts summary as a function of ∆T:
- North America: < 4°C, woody vegetation expands, positive carbon sequestration, >4°C up to 50% expansion of savannas at the expense of forests (Bachelet et al., 2001).
- Africa: 2.5°C to 3.5°C by 2100, reduction in vegetation cover and remobilization of Kalahari dune systems in southern Africa as early as 2040 (Thomas et al., 2005).
- Africa: HadCM3 A2, 2050 and 2080, Mammal community compositional shifts, increases in critically endangered and extinct species to between 10 and 40% of mammal species (Thuiller et al., 2006).
- Africa: IS92a Gga (HADCM2-SUL) 4% to 98% reduction in species ranges for ~80% of 179 animal species (Erasmus et al., 2002).

4.4.4 Mediterranean ecosystems

Characteristics: Mid-latitudes on all continents, in coastal regions (Mediterranean basin, Baja Mexico/California, south-western South Africa, south-western and southern Australia, central Chile), covering ~ 3.4x10^6 km². Climatically distinct with generally wet winters and dry summers, often on nutrient-poor soils (Cowling et al., 2005) and with high levels of biodiversity (Cowling et al., 1996). Generally shrub-dominated, but woodlands, forests and even grasslands occur in limited regions, and in the Mediterranean basin and increasingly elsewhere, heavily utilized landscapes are dominated by grasses, herbs and annual plant species where human impacts have favored disturbance-tolerant floras (Lavorel, 1999). Generally fire-prone, due to dry summers (Montenegro et al., 2004).

Goods and services: Valuable for high biodiversity overall (Myers et al., 2000), but also for grazing in the Mediterranean basin and Chile, products of extractive harvesting (e.g. wildflowers in South Africa and Australia, now also commercially produced for international trade), water yield for human consumption and agriculture in South Africa, and ground cover services on otherwise generally unproductive nutrient-poor soils.

Key vulnerabilities as from TAR: Mediterranean systems were not explicitly reviewed, but threats from desertification were projected due to expansion of adjacent semi-arid and arid systems under relatively minor warming and drying scenarios.

Impacts: These systems may be among the most impacted by global change (Sala et al., 2000). Diverse Californian vegetation types may show substantial cover change for temperature increases greater than about 2°C, including desert and grassland expansion at the expense of shrublands, and mixed deciduous forest expansion at the expense of evergreen conifer forest (Hayhoe et al., 2004). The Cape Fynbos Biome could lose 65% area under warming of 1.8°C, with ultimate species extinction of 23%
resulting in the long term (Thomas et al., 2004b; Rutherford et al., 2000). For Europe, only minor
biome-level shifts are projected for Mediterranean vegetation types for GMT increase of 3.2°C (Parry,
2000), contrasting with between 60 and 80% of current species projected not to persist in the southern
European Mediterranean region (GMT increase of 1.8°C, Bakkenes et al., 2002). Inclusion of CO2
fertilization effects in biome-level modelling may partly explain this contrast. Land abandonment
trends facilitate ongoing forest recovery (Mouillot et al., 2003) in the Mediterranean Basin,
complicating projections. In southwest Australia, high rates of vegetation migration rates may be
induced under double CO2 scenarios (Malcolm et al., 2002), and habitats of all frogs and many
mammal species would contract significantly with only a 0.5°C warming (Group, 2005).

Climate change impacts will interact with fire regime shifts. Greater fire frequencies are noted in
Mediterranean basin regions (Pausas and Abdel Malak, 2004) with exceptions (Mouillot et al., 2003).
Double CO2 scenarios increase fire escapes by 40-50% in California (Fried et al., 2004), and double
fire risk in western Fynbos (Midgley et al., 2005), favoring sprouting plants over seed regenerators in
Fynbos (Bond and Midgley, 2003), fire-tolerant shrub dominance in the Mediterranean Basin
(Mouillot et al., 2002), and vegetation structural change in California (needle-leaved to broad-leaved
trees, trees to grasses) reducing productivity and carbon sequestration (Lenihan et al., 2003).

Significant warming trends have been noted in the Cape Floristic Kingdom (Warburton et al., 2005)
and the Mediterranean Basin (Rebetez, 2004), extending growing season by 16 days (Penuelas and
Filella, 2001). Experimental warming shifted species relative dominance in Mediterranean Basin
communities (Wessel et al., 2004).

Projected rainfall changes are spatially complex (e.g. Sanchez et al., 2004; Sumner et al., 2003;
Vicente-Serrano et al., 2004). Rainfall frequency reductions projected for some Mediterranean regions
(e.g. Cheddadi et al., 2001) will exacerbate drought conditions. Soil water content controls ecosystem
water and CO2 flux in the Mediterranean Basin system below key water content thresholds (Rambal et
al., 2003), and could reduce ecosystem carbon and water flux (Reichstein et al., 2002). The 2003
European drought had major physiological impacts on Mediterranean vegetation and ecosystems, but
most showed a full recovery by 2004 (Gobron et al., 2005). A mediterranean dryland ecosystem
showed net losses of carbon in dry years (Hastings et al., 2005).

With increasing precipitation, forest expansion and greater carbon storage has been projected in
California (Bachelet et al., 2001). In the Mediterranean basin, vegetation structural change will depend
on rainfall change, 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).

Many species show marginal benefits from rising atmospheric CO2 (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 CO2 (De Angelis et al.,
2000).

Established Pinus halepensis (Borghetti et al., 1998) show high drought resistance, but Ponderosa pine
forests had reduced productivity and water flux during a 1997 heatwave, which did not recover for the
rest of the season, indicating 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 drought tolerance under field conditions. Experimental drying
differentially reduced 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 Penuelas, 2005), suggesting
complex changes in species relative success under drying scenarios. Species-specific tolerance to
drought in woody plants may be explained by xylem hydraulic characteristics (Martinez-Vilalta et al.,
Drought may also act indirectly on plants by reducing the availability of soil phosphorus (Sardans and Penuelas, 2004). Bioclimatic niche-based modeling projects reduced species persistence, species richness, and range reductions for the majority of species modeled in the Cape Floristic Region (Midgley et al., 2002; Midgley et al., 2003). Ranges of trees and shrubs may shift unpredictably, and fragment, under IS92a scenarios (Shafer et al., 2001). In southern Europe, species composition change may be high under a range of scenarios (Thuiller et al., 2005). Range size reductions increase species extinction risks, with up to 30-40% facing increased extinction probabilities beyond 2050 (Thomas et al., 2004a). Species of lowland plains may be at higher risk than montane species both in California (Peterson, 2003) and Cape Floristic Region (Midgley et al., 2003), although in the Mediterranean Basin, montane species show high risk (Thuiller et al., 2005).

Adaptation costs and opportunities: Adaptation strategies include optimising species persistence through identifying key geographic regions for higher levels of species protection (Hannah et al., 2005), and to facilitate species migration in protected areas (Williams et al., 2005). Costs have not been estimated.

Key vulnerabilities: Warming and drying trends threaten substantial species range reductions and imply a need for high rates of migration to allow natural adaptation, unlikely to be realized. Land-use, habitat fragmentation and intense human pressures will further limit natural adaptation responses, and fire regime shifts may threaten specific species and plant functional types. Vegetation structural change driven by dominant, common or invasive species may also threaten rare species. Overall a loss of biodiversity and carbon sequestration services may be realized over much of these regions.

Impacts summary as a function of $\Delta T$: Diverse vegetation types of California are projected to show substantial shifts in cover for temperature increase in excess of about 2°C, including desert and grassland expansion at the expense of shrublands, and mixed deciduous forest expansion at the expense of evergreen conifer forest (Hayhoe et al., 2004). Cape Fynbos Biome reduced in area by 65% (climate scenario HADCM2 n IS92a, for 2050). In southwest Australia, high rates of vegetation migration rates may be induced under double CO$_2$ scenarios (Malcolm et al., 2002), and the habitats for all frog and many mammal species would be significantly reduced with only a 0.5°C warming (Group, 2005). Double CO$_2$ scenarios increase fire escapes by 40-50% in California (Fried et al., 2004), and double fire risk in western Fynbos (Midgley et al., 2005).

Implications for policy and sustainable development: [for DGVM section: Rising atmospheric CO$_2$ may counteract some effects of drying trends on plant primary productivity (Ainsworth and Long, 1997), and may already have counteracted negative impacts of climate trends during the 20$^{th}$ century.

4.4.5 Forests and woodlands

Characteristics: Ecosystems with a densely treed canopy, covering a total of 41.6 Mkm$^2$ (~30% of all land) with 42% in the tropics, 25% in the temperate, and 33% in the boreal zone (Table 4.1, e.g. Sabine et al., 2004). Forests require benign environmental conditions (summer means >12°C, >300 mm total annual rainfall) and are among the most productive terrestrial ecosystems (Table 4.1). This makes them attractive both for agricultural uses (underlying the currently high deforestation and degradation rates in tropical and subtropical regions Hassan et al., 2005; leading to about one quarter of anthropogenic CO$_2$ emissions e.g. Houghton, 2003a), and also for climate change mitigation purposes (WG III Chapter 11, Watson et al., 2000). Nevertheless, forests, in particular the northern circumpolar boreal forests (70.2 Pg (~30%) Smith et al., 2004), still store in plant biomass and soils...
the largest portion of biospheric carbon stocks, recently estimated as high as 1’640 PgC (Sabine et al., 2004), corresponding to ~220% of atmospheric carbon.

Goods and services: In addition to well known commercial goods (Chapter 5 Shvidenko et al., 2005 Section 21.5 p.600-607) forests provide numerous goods for subsistent livelihoods, especially for the rural poor (Gitay et al., 2001 p. 239; Shvidenko et al., 2005). Many key supporting services essential for human existence are less recognised, such as habitat provision for an increasing fraction of biodiversity (in particular where subject to land-use pressures Hassan et al., 2005; Duraiappah et al., 2005). Carbon sequestration dominates regulating services, due to high forest productivity, but numerous other regulating services are as essential for human well-being: climate regulation, soil and water protection or purification (>75% of globally usable freshwater supplies come from forested catchments Shvidenko et al., 2005), and recreational, cultural, and spiritual services (Reid et al., 2005; Millennium Ecosystem Assessment, 2005c).

Key vulnerabilities as from TAR: Forests, in particular in the boreal region, were considered to have a high potential vulnerability to climate change, which would only become apparent in the long-term unless disturbance regimes (drought, insects, fire), partly due to climate change, were to cross critical thresholds (Gitay et al., 2001).

Impacts: Projections for some forests currently limited by their minimum climatic requirements indicate gains from climate change (references at end of sentence; Fig. 4.4, vegetation changes (VC) 1 to 3), but many may be impacted detrimentally (Fig. 4.4 VC 6), notably for stronger warming (Kirschbaum and Fischlin, 1996; Matarira and Mwamuka, 1996; Bugmann, 1997; Fischlin and Gyalistras, 1997; Solomon and Leemans, 1997; Foley et al., 1998; Ravindranath and Sukumar, 1998; Lischke et al., 1998; Villers-Ruiz and Trejo-Vazquez, 1998; Dixon et al., 1999; Price et al., 1999; Siddiqui et al., 1999; Still et al., 1999; White et al., 1999; Daly et al., 2000; Flannigan et al., 2000; Ni et al., 2000; White et al., 2000b; Bachelet et al., 2001; Bergengren et al., 2001; Ostendorf et al., 2001; Smith and Lazo, 2001; Xu and Yan, 2001; Arnell et al., 2002; Enquist, 2002; Iverson and Prasad, 2002; Bachelet et al., 2003; Lauenroth et al., 2004; Levy et al., 2004; Izaurralde et al., 2005; Schaphoff et al., 2006; Scholze et al., 2007 compare VC 6 Fig. 4.4a vs. b). Average productivity gains result from CO2 fertilization (although this effect has shown to be much weaker than previously assumed, see 4.4.1), warming in cold climates given concomitant precipitation increases do compensate for possibly increasing water vapour pressure deficits, and precipitation increases under water limited conditions. There is growing evidence that nutrients (e.g. enrichment as resulting from N deposition), species composition, dynamic age structure effects (albeit often poorly mimicked by several models from above cited studies, in particular at the global scale), pollution, and biotic interactions such as soil organisms, may moderate the direct CO2 or climate change effects affecting in particular also the net ecosystem productivity (e.g. Karnosky et al., 2003; King et al., 2004b; Heath et al., 2005; Körner et al., 2005a 4.4.1). Impacts of climate change on forests, in particular detrimental ones, however, depend not only on climate means, but also on the seasonal and diurnal rainfall and temperature patterns 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). Furthermore, precipitation is distributed worldwide in a highly uneven manner (WG I Chapters 9, 10) and thus conditions forests may experience in future climate change are associated with major uncertainties. While the upper bound for precipitation is of minor concern, since extreme persistent flooding alone results in tree mortality, the upper bound for temperature is more critical, since it is linked to precipitation in a complex, non-linear manner. In some regions such as Central and Northern Europe in addition to warming more winter precipitation was observed in the past (e.g. Gyalistras, 2003; Xoplaki et al., 2003) and is projected for the future (WG I, chapter 9, 10). However, since summer temperatures are projected to increase and precipitation to decline, there is potential for drought conditions during the critical growing phase, which is true for many locations on all continents (e.g. Cox et al., 2004; Schaphoff et al., 2006; Scholze et al., 2007 Fig. 4.4 VC 6). Since all these responses
potentially influence net forest carbon exchange (NEP), substantive biotic feedbacks may result, either through carbon releases or influences on regional climate contributing to further major uncertainties (e.g. Betts et al., 2000; Peng and Apps, 2000; Grace and Rayment, 2000; Gitay et al., 2001; Semazzi and Song, 2001; Berggren et al., 2001; Rasmussen et al., 2002; Leemans et al., 2002; Wicks and Curran, 2003; Korner, 2003b; Cox et al., 2004; Korner, 2004; Canadell et al., 2004; Gruber et al., 2004; Heath et al., 2005; Körner et al., 2005a; Leuzinger et al., 2005; Körner, 2007).

Effects of drought on forests, including resulting biotic feedbacks, remain incompletely understood (e.g. Reichstein et al., 2002; Betts et al., 2004). While increased mortality is a likely consequence of severe drought, impacts can be offset by fertile soils (Hanso and Weltzin, 2000). Moreover, the European heat wave 2003 (Box 4.1) enhanced growth at high elevation sites (Jolly et al., 2005) and more benign conditions in 2004 allowed forests to partly recover (Gobron et al., 2005; somewhat contested by Lloret et al., 2004), whereas contemporaneous drought was found to deplete resilience of e.g. Mediterranean forests (Lloret et al., 2004 4.4.4).

Drought conditions interact with disturbances such as fire (Flannigan et al., 2000) or insects (Hanso and Weltzin, 2000; Logan et al., 2003 Box 4.1). For instance, in boreal forest, tree defoliating insects affect an area about 50 times as large as that subject to fire (Logan et al., 2003) and can periodically cause economically as well as ecologically substantive damages (e.g. Gitay et al., 2001 Box 5-10). Insect pests are at least partly responsible for the decline and ultimate extirpation of stands at the southern margins of the range of their hosts, subject to warmer and drier conditions, where outbreaks occur at a higher frequency (Volney and Fleming, 2000). If climate warms and this ecotone becomes subject to increased drought, the effect is expected to become stronger (Gan, 2004; Logan et al., 2003).

At the northern ecotone, frosts and the cold temperature in general appear to limit insect outbreaks (Virtanen et al., 1996; Volney and Fleming, 2000), implying that outbreaks currently constrained from northern ranges will become more frequent in a warming world (Carroll et al., 2004). In general it is difficult to project overall effects of climate change on pest insects, since these are affected by temperature, humidity, and biotic interactions in a species-specific manner, which may perturb poorly understood ecological interactions such as natural biological control, tipping the balance on population control positively or negatively (Malmstrom and Raffa, 2000; Bale et al., 2002; Morecroft et al., 2002), whereas for diseases some findings suggest an increase in infected areas or increased abundances (e.g. Boland et al., 2004; Bergot et al., 2004).

Climate change is known to alter some disturbance regimes directly, e.g. increasing wildfire frequencies and event extents (e.g. Stocks et al., 1998; Podur et al., 2002; Gillett et al., 2004; Brown et al., 2004), but climate change induced stress on trees may also indirectly exacerbate disturbances (Dale et al., 2000). Relationships between storms and climate change are still poorly understood (WG I, Box 4.2, ice storms Irland, 2000), however windthrow impacts can be substantive (e.g. Peterson, 2000; Thurig et al., 2005b; Fuhrer et al., 2007), and for southern Finland some findings suggest increased susceptibility to windthrow by climate change (Peltola et al., 1999). Considerable progress has been made in understanding fire regimes and related processes (Kasischke and Stocks, 2000; Skinner et al., 2002; Stocks et al., 2002; Hicke et al., 2003; Podur et al., 2003; Wotton et al., 2003; Gillett et al., 2004) enabling improved projections of future fire regimes (Li et al., 2000; Flannigan et al., 2000; de Groot et al., 2003; Fried et al., 2004; Brown et al., 2004). Some authors (Flannigan et al., 2000) argue that altered fire regimes will impact boreal forests more than climate change itself. Climate changes including in El Niño events may also alter fire regimes in fire-prone regions such as Australia (Hughes, 2003; Williams et al., 2004b; Allen Consulting Group, 2005) the Mediterranean region (e.g. Mouillot et al., 2002 see 4.4.4) or as far North as Alaska (Hess et al., 2001). Finally, fire may severely impact the tundra (Rupp et al., 2000; Camill and Clark, 2000), possibly releasing large quantities of CO2 from carbon previously immobilized in frozen soils (Harden et al., 2000) which have accumulated to one of the largest carbon stocks in the biosphere since the last ice age (Smith et al., 2004).
Rapid encroachment of boreal tree species into present tundra could compensate for such losses. Vegetation models project that forest might replace between 11 and 50% of tundra over the period in which atmospheric CO₂ doubles (Harding et al., 2002; Kaplan et al., 2003; White et al., 2000b; Callaghan et al., 2005 Fig. 4.4 VC 1). However, estimates for migration rates of tree species from paleoecological records (as low as 10-25 m/yr for Norway spruce Udra, 1988; on average 200-300 m/yr, up as high as 500-1,000 m/yr for pioneers, or exceptionally 2,000 m/yr, Davis, 1981; Velichko et al., 1995; Ritchie and MacDonald, 1986; Kirschbaum and Fischlin, 1996; Symon et al., 2005 p. 570) and modelling studies reconstructing past (e.g. Solomon and Kirilenko, 1997; Prentice and Jolly, 2000; Lischke et al., 2002) or projecting future (Iversen et al., 2004; Neilson et al., 2005) dispersal, all indicate that more realistic migration rates will result in lagged northward shifts of taiga (150-250 yr Chapin and Starfield, 1997; Camill and Clark, 2000; Skre et al., 2002). Lower rates are probably realistic, since higher estimates typically ignore delaying factors such as herbivory or soil formation processes (Vlassova, 2002; Crawford et al., 2003; Nordengren et al., 2003; Cairns and Moen, 2004; Juday, 2005 see 4.4.6) and recent genetic analysis (McLachlan et al., 2005) indicates that commonly inferred estimates from pollen have overestimated dispersal rates, explaining observed pollen records by multifton recolonization from low-density refugees (Pearson, 2006). Slow encroachment of taiga into tundra is confirmed by satellite data showing no expansion of boreal forest stands in response to recent warming (Masek, 2001). Simultaneously, the rate of boreal forests loss at the southern boundary due to drought and fire (e.g. Kurz and Apps, 1999) indicates an incipient lag, with projections of major carbon losses, in particular from peatlands, with the potential to accelerate global climate change (e.g. Woodwell et al., 1998; Camill and Clark, 1998; Camill and Clark, 2000; Grant et al., 2003 4.4.6).

As reported in TAR (Gitay et al., 2001) the net global loss in forest cover appears to have slowed further (Stokstad, 2001; FAO, 2001), but in some tropical and subtropical regions, notably Southeast Asia and similarly the Amazon (e.g. Nepstad et al., 1999), deforestation rates are still high (0.01-2.01%/yr Lepers et al., 2005; Alcamo et al., 2007), while in some northern regions like Siberia, degradation rates are increasing largely due to non-sustainable logging (Lepers et al., 2005). Though uncertainties for rate estimates are considerable (e.g. FAO, 2001; Houghton, 2003b; Lepers et al., 2005), current trends in pressures (Nelson, 2005) will clearly lead to continued deforestation and degradation in critical areas (historically accumulated loss of 182–199 PgC DeFries et al., 1999; Canadell et al., 2004; Shvidenko et al., 2005; expected releases in 21st century 40 – 100 PgC Gruber et al., 2004) with concomitant implications for biodiversity (Duraiappah et al., 2005) and other supporting services (Hassan et al., 2005). In most industrialized countries forest areas are expected to increase (e.g. European forests by 2080 up to 6% for SRES B2, Karjalainen et al., 2002; Sitch et al., 2005 partly due to intensified, agricultural management and climate change.

Although land use changes may dominate impacts, especially in biodiversity hotspots, climate change generally exacerbates biodiversity risks (montane cloud forests Foster, 2001; Amazonia Miles, 2002; Hawai Benning et al., 2002; Costa Rica Enquist, 2002; Australia Williams et al., 2003) including tropical montane cloud forests where extinction of amphibian species has been attributed as very likely caused by recent climate change (Pounds et al., 2006). In a few exceptions climate change may increase diversity locally or regionally (Kienast et al., 1998) but in most cases extinction risks are increased and conservation management practices need to adapt to prevent this (e.g. Villers-Ruiz and Trejo-Vazquez, 1998; Bazzaz, 1998; Thompson et al., 1998; Malhi and Phillips, 2004).

Adaptation costs and opportunities: Few studies exist (e.g. Winnett, 1998) and quantitative estimates, i.e. model projections, diverge and prevent conclusive findings (Winnett, 1998).

Key vulnerabilities: Most scenarios based on an A2 world show significant forest dieback towards the end of this century and onwards in tropical, boreal, and mountain areas with a concomitant loss in diversity as well as key services such as local climate regulation and C sequestration. These forests
also harbor a major fraction of Earth’s biodiversity, in particular tropical forest diversity hotspots (e.g. northeastern Amazonia Miles, 2002). Mountain forests are increasingly encroached upon from adjacent lowlands, while simultaneously losing high altitude habitats due to warming.

**Impacts summary as a function of \(\Delta T\):**

- > 1°C substantive fraction of stands with severe climate change impact (Lexer *et al.*, 2002).
- > 1°C high sensitivity of humid tropical forests of north Queensland, Australia (Hilbert *et al.*, 2001).

### 4.4.6 Tundra and arctic ecosystems

**Characteristics:** “Tundra” is used here in its widest sense to denote vegetation and ecosystems north of the treeline of closed boreal forest, (tundra’s southern transition zone) which occurs at about 51°N in eastern Canada to about 84°N in Greenland (Callaghan *et al.*, 2005). This includes any ecosystems poleward from tundra, notably the sea ice biome in both hemispheres (e.g. Arrigo and Thomas, 2004 see 4.4.9), and Southern Ocean sub-antarctic islands, which enables an otherwise desert-like ice environment with the needed primary production to support life-forms up to top predators such as polar bears (Box 4.4).

**Goods and services:** Arctic ecosystem services include climate regulation, biodiversity maintenance, fuel, food and fibre production, and cultural enjoyment (Chapin *et al.*, 2005a p. 721-728). Climate regulation is likely to be dominated by positive feedbacks between climate and albedo changes through diminishing snow covers and expanding forests or woodlands (Chapin *et al.*, 2005b) plus changes in the greenhouse gas balance, notably of methane (Christensen *et al.*, 2004). The arctic harbors unique species (e.g. Box 4.4) contributing to global biodiversity (Usher *et al.*, 2005; Chapin *et al.*, 2005a), and are important as a renewable resource for local livelihoods. Local mixed economies of cash and subsistence depend strongly on household production, which involves the harvest of local resources, food preparation, storage, distribution, and consumption, all forming part of an intergenerational transmission of a unique body of knowledge (Hassol, 2004a).

**Key vulnerabilities as from TAR:** Arctic and sub-arctic ecosystems (ombrotrophic bog communities) above permafrost were considered likely to be most vulnerable to climatic changes, since impacts may turn arctic regions from a net sink to a net carbon source (Gitay *et al.*, 2001). Adverse impacts were projected for subsistence species like marine birds, seals, polar bears, tundra birds and tundra ungulates (Gitay *et al.*, 2001).

**Impacts:** Global warming is projected to be most pronounced at high latitudes (WG I, Chapters 10,11 - Phoenix and Lee, 2004). Ongoing rapid climatic changes will introduce poleward forcing of climatically near-equilibrium tundra at unprecedented rates (Velichko, 2002), causing largely lagged and critical responses in its slow growing plant communities (Camill and Clark, 2000; Velichko *et al.*, 2004; Chapin *et al.*, 2000; Callaghan *et al.*, 2004a; Callaghan *et al.*, 2004c): via (a) novel thermal regimes will tend to drive edificators (biome-specific environment-forming plants important for autonomous adaptation including soil formation) outside their current local zonal distribution, putting them at a competitive disadvantage relative to species immigrating from South; (b) rates of change together with flat tundra topography require large migration rates which exceed the migrational capacity of the edificators (Callaghan *et al.*, 2005). Taiga encroachment from South into tundra, which could compensate for these detrimental effects, is very likely to lag these changes (see 4.4.5 – e.g. Callaghan *et al.*, 2004b).

The thermally-stable oceanic climate of the sub-Antarctic Marion Island (47°S, 38°E) appears to be changing, with a rise in annual mean surface air temperature of 1.2°C between 1969 and 1999. Annual
precipitation decreased more or less simultaneously, and the 1990s was the driest in the island’s five decades with records (Smith, 2002). These changes may be linked to a shift in phase of the semiannual oscillation in the Southern Hemisphere after about 1980 (Rouault et al., 2005). Climatic change will directly affect the indigenous biota of sub-Antarctic islands, particularly through interactions with introduced biota (Smith, 2002; Barnes et al., 2006). Experimental droughting of the keystone cushion plant species *Azorella selago* on Marion Island revealed measurable negative impacts after only several months (Le Roux et al., 2005).

Projections of vegetation changes in the northern Arctic suggest for ~2080 that 17.6% (14-23%) of the current polar desert (1960 estimates) will be replaced by tundra vegetation (Callaghan et al., 2005). The BIOME4 model (Kaplan et al., 2003) projects for the Canadian Arctic a wide zone of shrub tundra displacing dwarf shrub tundra. Models operating at the biome level ignore key individualistic responses of tundra species, including differential migration rates (Chapin and Shaver, 1985). Thus in reality, communities without present analog are likely to form. Experimental manipulations of air temperature at 11 locations across the tundra also show that tundra plant communities change substantially by shifts in species dominance, canopy height, and diversity (Walker et al., 2006) and that cryptogams are particularly vulnerable (Cornelissen et al., 2001; van Wijk et al., 2004). A warming of 1 to 3°C caused a short-term diversity decrease, but generalisations are unwarranted because of insufficiently long experimentation (van Wijk et al., 2004; Graglia et al., 2001; Dormann and Woodin, 2002; Walker et al., 2006).

Moisture supply influences substantially the state of permafrost – one of the most important components of the tundra landscape (Anisimov and Nelson, 1996; Anisimov et al., 2002b; Anisimov et al., 2002a). A combined increase in temperature, active layer depth, and moisture content in the upper ground layer will not only promote paludification (Crawford et al., 2003) and thermokarst processes as well the dryness in the elevated elements of topography, but also influence ecosystem structures and functions (Velichko et al., 2005). Increasing instability of the active layer brings about more extensive mixing and lateral shifting of the mineral matrix in the soil, causing stress for the plant root systems, slowing decomposition. Generally this will favour moisture-loving species (e.g. sedges), while the peatbog vegetation over permafrost could experience drier conditions (Camill, 2005).

Projections for animal populations are still scanty and expected effects are manifold and sometimes detrimental, sometimes beneficial for the impacted population: While summer food availability may increase for some vertebrates (Hinzman et al., 2005), icing rise events at critical winter times may reduce abundance (Aanes et al., 2002; Yoccoz and Ims, 1999). Tundra wetland habitat for migrant birds may dry progressively (Smith et al., 2005; Hinzman et al., 2005). Many species of Arctic breeding shorebirds and waterfowl are projected to undergo major population declines as their tundra habitat disappears (Table 4.2, Box 4.6). In contrast, northern range extension of more southern species is expected, e.g. moose and red fox (Callaghan et al., 2005). Some new colonisers might achieve the character of “invasive” species with major impacts, such as the North American Mink (Neuvonen, 2004), but also the presently restricted populations of southern “weeds” that are likely to spread in a warmer climate (Forbes, 1995) may eventually lead to substantive carbon emissions (Sturm et al., 2001). For arctic species such as the polar bear major risks of extinction are associated with the large decrease in the extent of the sea ice biome and the sea ice cover (Box 4.4).

**Box 4.4: Polar bears – a species in peril?**

Polar bears (*Ursus maritimus*) live throughout the ice-covered waters of the circumpolar Arctic and can wander thousands of kilometres an year. There are an estimated 20,000 to 25,000 polar bears worldwide (IUCN/SSC Polar Bear Specialist Group, in prep.), mostly inhabiting the annual sea ice
over the continental shelves and inter-island archipelagos. They are specialized predators of ice-breeding seals and are therefore dependent on sea ice for survival. Mother bears give birth to cubs and nurse them in dens in snow banks. They emerge in spring after having fasted for 5 to 7 months (Ramsay and Stirling, 1988) and are in great need of a nourishing meal, a ringed seal best hunted from the ice before it breaks up. Polar bears having 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 in the tundra biome 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 have been found to bioaccumulate through the food chain and reach high levels in polar bears (Norstrom et al., 1998). Recent studies suggest that these contaminants not only affect the bear’s 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).

The greatest challenges facing polar bears relate to 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 the Western Hudson Bay population has declined from 1200 bears in 1987 to fewer than 950 in 2004. 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 U.S. Fish and Wildlife Service is also considering a petition to list the polar bear as a threatened species based in part on future risks to the species population from climate change. The principal cause of this decline is climatic warming and its consequent negative affects on the sea ice habitat of polar bears. Reductions in sea ice will drastically shrink marine habitat for polar bears, ice-inhabiting seals and other animals (Hassol, 2004b). If sea ice declines according to some projections (cf. WG I Chapter 10, Figure 10.3.10) polar bears are pushed towards extinction. Similar consequences are facing other ice-dependent species, not only in the Arctic but also in the Antarctic (Chapter 1, Barbraud and Weimerskirch, 2001; Croxall et al., 2002).

Changes in albedo associated with loss of snow cover, and invasion of low, snow-covered tundra vegetation by darker evergreen coniferous trees is very likely to decrease regional albedo significantly and lead to a warming effect that is greater than the cooling projected from the increased carbon uptake by advancing forests and shrubs (Betts, 2000; Sturm et al., 2001). Process based models generally indicate a net carbon sequestration in the Arctic due to enhanced production exceeding decomposition increases (McGuire et al., 2000; Sitch et al., 2007). However, the potential for carbon sequestration varies from region to region (Callaghan et al., 2005) and model uncertainties are high (Sitch et al., 2007), since changes in hydrology, permafrost, insect pest outbreaks, fire and human impacts are poorly represented, yet can have large effects on the carbon balance. The projected changes may cause an increase in carbon uptake (Camill and Clark, 2000; McGuire et al., 2000; Zavarzin and Kudeyarov, 2006), but also to increases in areas of mires, peatlands and waterlogged soils leading to greater methane emissions (Christensen et al., 2004). Moreover, models of methane fluxes project dramatic increases in CH₄ emissions as soils warm (Sitch et al., 2007), offsetting the possible overall carbon sink due to the great greenhouse warming potential of CH₄.
Adaptation costs and opportunities: There are no cost estimates available and adaptation costs are expected to be high (Symon et al., 2005).

Key vulnerabilities: Despite an overall increased carbon uptake, changes in albedo and increased releases of methane will lead to positive climate radiative forcing through the Arctic region. Unique endemic biodiversity (e.g. polar bear, Box 4.4) as well as tundra dependent, non-native species like migratory species (e.g. waterfowl, Box 4.6, section 4.4.8) are put at major extinction risks with the concomitant threats to food security for indigenous peoples (e.g. Inuit).

Impacts summary as a function of \(\Delta T\):
- A MGT rise of \(\sim 1^\circ C\) is expected to reduce the tundra area in Siberia and North America.
- \(4^\circ C\) polar bears at high risk of extinction

Implications for policy and sustainable development: Changes in the availability of traditional resources as well as changes in their accessibility are likely to add to the stresses on the livelihoods of Arctic residents arising already from socio-economic changes in globalisation and life styles (Hassol, 2004b). Subsistence activities, particularly associated with gathering natural resources, are important for maintaining healthy social relationships and cultural identity (Chapin et al., 2005a).

4.4.7 Mountains

Characteristics: Mountain regions (~20% of all land) exhibit many climate types within short horizontal distances that correspond to widely-separated latitudinal belts. Consequently, although species richness decreases with elevation, mountain ecosystems feature high beta diversity and in many regions are of the species richest ecosystems (e.g. Moser et al., 2005; Spehn and Körner, 2005).

Goods and services: Mountain ecosystem services such as water purification and climate regulation extend beyond their geographical boundaries and affect all continental mainlands (e.g. Woodwell, 2004). Local key services allow inhabitability of mountain areas, e.g. through slope stabilisation or protection from natural disasters, provide many goods for subsistent livelihoods, and are home to many indigenous peoples. They serve also increasingly as refuges for many endemic species and are attractive to recreational activities and tourism due to their relative remoteness.

Key vulnerabilities as from TAR: Shortened snow melt season leading to rapid water release and low elevation floods may cause water shortage during growing season, but these impacts may be dominated by ecosystem degradation pressures such as land-use changes, overgrazing, trampling, pollution, vegetation destabilization, and soil losses, in particular in highly diverse regions such as Caucasus and Himalayas (Gitay et al., 2001). While adaptive capacities are generally considered limited, high vulnerability is attributed to many alpine biota with their high levels of endemism.

Impacts: 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), leading to a loss of the coolest climatic zones at the mountain peaks and upslope shifts of vegetation belts (Gitay et al., 2001; Penuelas and Boada, 2003). Because mountain tops are smaller than their bases, upward shifting vegetation belts are reduced to increasingly smaller areas, which may genetically deteriorate species and make them more vulnerable to other environmental pressures such as pollution (Peters and Darling, 1985; Bortenschlager, 1993). However, despite the relative short distances to be covered, for many endemic species upward migration is constrained by genetically fixed climatic tolerances, and in general a reshuffling of competition pressures due to particular species-individualistic responses through varying longevities and survival rates drawing from a high
degree of evolutionary specialisation to the harsh mountain climate (e.g. Gottfried et al., 1999; Theurillat et al., 1998; Theurillat and Guisan, 2001; Klanderud, 2005; Klanderud and Totland, 2005; Dullinger et al., 2005; Huelber et al., 2006), including effects from invading alien species (e.g. Dukes and Mooney, 1999; Mack et al., 2000). In contrast to the short distance upward shifts, mountain ridges also represent considerable obstacles to dispersal for many species as evidenced by many paleoecological and modelling studies (e.g. Foster, 2001; Lischke et al., 2002; Neilson et al., 2005; Pounds et al., 2006).

Treelines representing the interface between subalpine forests and low-growing alpine meadows are partly controlled by carbon balance (Körner, 1999; Monson et al., 2002), but according to recent findings surprisingly strongly by temperature, in particular soil temperature, which moderates the growth rate of all small mountain plants (worldwide climatic treelines are characterized by seasonal mean ground temperature of 6.7±1.6°C (95% confidence interval) Körner, 1998; Grace et al., 2002; Korner and Paulsen, 2004; Millar et al., 2004; Zha et al., 2005; Lara et al., 2005). In many mountains the treeline is located below its potential climatic limit because of grazing, in case of the European Alps century-old, or disturbances such as wind or fire. In other regions such as the Himalayas, deforestation of past decades has transformed much of the environment and fragmented ecosystems (Becker and Bugmann, 2001). Although temperature control may well be dominant at equilibrium with climate, ability of tree populations to follow temperature have been challenged by recent findings (Shiyatov, 2003; Wilmking et al., 2004; Dullinger et al., 2004).

Where warmer and drier conditions are projected, mountain vegetation is expected to suffer from increased evapo-transpiration (Ogaya et al., 2003; Jasper et al., 2004; Stampfli and Zeiter, 2004; Rebetez and Dobbertin, 2004; Zierl and Bugmann, 2005; Jolly et al., 2005; Pederson et al., 2006), which has been projected to lead to subalpine forest dieback for continental climata in the interior of mountain ranges (e.g. Fischlin and Gyalistras, 1997; Lischke et al., 1998; Lexer et al., 2000; Bugmann et al., 2005) or areas under the influence of a Mediterranean climate. Even in humid tropical regions, plants and animals have been shown to be sensitive to water stress on mountains such as Mt. Kinabalu in the Malaysian part of Borneo (Kitayama, 1996) or Costa Rica (Still et al., 1999).

The duration and depth of snow cover, often correlated with mean temperature and precipitation (Keller et al., 2005; Monson et al., 2006), is a key factor in many alpine ecosystems (Körner, 1999; Daimaru and Taoda, 2004). Missing snow cover exposes plants and animals to frost and influences water supply in spring (Keller et al., 2005). If animal movements are disrupted by changing snow patterns, as has been found in Colorado (Inouye et al., 2000), increased wildlife mortality may result through a mismatch between wildlife and environment. Warming caused by increased precipitation lead at high altitudes to higher snowfall, so that earlier arriving altitudinal migrants are confronted with delayed snowmelt (Inouye et al., 2000).

In the harsh mountain climate disturbances such as avalanches, rock fall, fire, wind, and herbivore damage interact and are strongly dependent from climate (e.g. Whitlock et al., 2003; Beniston and Stephenson, 2004; Carroll et al., 2004; Cairns and Moen, 2004; Hodar and Zamora, 2004; Schumacher et al., 2004; Schoennagel et al., 2004; Pierce et al., 2004; Kajimoto et al., 2004). These effects are often detrimental, prevent recruitment which is crucial during phases needing adaptation, and are exacerbated by human land-use and other anthropogenic pressures (e.g. Lawton et al., 2001; Dirnbock et al., 2003; Huber et al., 2005).

The increasing interest in analysing climatic change sensitive ecotones such as the treeline in mountains (e.g. Camarero et al., 2000; Walther et al., 2001; Diaz, 2003; Sanz-Elorza et al., 2003) has shown that several mountain-restricted species will exhibit population loss (e.g. Beever et al., 2003; Florenzano, 2004). Most vulnerable ecotone species are those that are genetically poorly adapted to rapid environmental change, reproduce slowly, disperse poorly, and which are isolated or are highly
specialized, because of their high sensitivity to seemingly minor stresses (McNeely, 1990). Recent findings for Europe, despite the spatially coarse analysis, indicate that mountain species are disproportionately sensitive to climate change (~60% species loss Thuiller et al., 2005). Moreover, if in addition to climate change also human pressures exert impacts on mountain biota, substantial biodiversity losses are to be expected (Pounds et al., 1999; Lawton et al., 2001; Pounds, 2001; Pauli et al., 2001; Pauli et al., 2003; Solorzano et al., 2003; Halloy and Mark, 2003; 30% loss of birds Peterson, 2003; Pounds and Puschendorf, 2004; Pounds et al., 2006).

Adaptation costs and opportunities: No estimates available.

Key vulnerabilities: Disproportional high risk of extinction for many endemic species in diverse mountain ecosystems such as the montane cloud forests or forests in other tropical regions on several continents (Williams et al., 2003; Pounds and Puschendorf, 2004; Andreone et al., 2005; Pounds et al., 2006) or wherever endemic species loose their mountain habitat that is shrinking due to warming (Pauli et al., 2003; Thuiller et al., 2005)

Impacts summary as a function of $\Delta T$:

- > 2°C in highly diverse Australian wet tropics cause complete loss of core environments with high extinction risks for endemic species (Williams et al., 2003)
- > 3°C profound changes in all biota expected (Theurillat and Guisan, 2001)

4.4.8 Wetlands, Freshwater lakes and rivers

Characteristics: Inland aquatic ecosystems vary greatly in hydrological characteristics and global distribution. Most of the mighty rivers drain tropical and subtropical lands, and several of them pass through highly arid regions. A majority of natural freshwater lakes are located in the higher latitudes, whereas most of the artificial lakes occur in mid- and lower latitudes. Numerous saline lakes occur at altitudes up to 5000 m a.s.l., especially in the Himalaya and Tibet. The majority of natural wetlands (peatlands) is in the boreal region but most managed wetlands (rice paddies) are in the tropics and subtropics. Global estimates of area under rivers, lakes and wetlands vary greatly depending upon definition (Finlayson et al., 2005). This chapter follows the TAR in considering ‘wetlands’ as distinct from rivers and lakes. Wetlands encompass a most heterogenous spectrum of habitats along the hydrological and nutrient gradients and all key processes, including goods and services provided, depend on the catchment level hydrology. Inland waters are also subject to many pressures due to their position downstream from many human activities.

Goods and services: Aquatic ecosystems provide a wide range of goods and services (Gitay et al., 2001; Finlayson et al., 2005), which include water supply and waste assimilation. Rivers transport water and nutrients from the land to the oceans and provide crucial buffering capacity during drought spells by being fed from mountain springs and glaciers (e.g. European summer 2003). The endorheic lakes serve as sediment and carbon sinks (Cohen, 2003) and hereby provide crucial repositories of information on past climate changes. Wetlands are often biodiversity ‘hotspots’ (Reid et al., 2005) and play a major role for climate change through both carbon sequestration and methane emissions (Finlayson et al., 2005).

Key vulnerabilities as from TAR: Gitay et al. (2001) described some inland aquatic ecosystems (arctic, subarctic ombrothropic bog communities on permafrost, depressional wetlands with small catchments, drained or otherwise converted peatlands) as most vulnerable to climate change, and indicated the limits to adaptations due to the dependence on water availability controlled by outside factors.

Impacts: The climate change impacts on inland aquatic ecosystems will range from the direct effects of
the rise in temperature and CO₂ concentration to indirect effects through alterations in the hydrology
resulting from the changes in the regional or global precipitation regimes and the melting of glaciers
and ice cover (e.g. Cubasch et al., 2001 chapter 3, WGI chapters 4, 10).

Studies since TAR have confirmed and strengthened the earlier conclusions that the rising temperature
will lower the water quality in lakes through a fall in hypolimnetic oxygen concentrations, release of P
from the sediments, increased thermal stability, altered mixing patterns (McKee et al., 2003; Verburg
et al., 2003; Winder and Schindler, 2004; Jankowski et al., 2006). It will affect the microorganisms
and benthic invertebrates (Kling et al., 2003). The distribution range of many species of fish (Lake et
al., 2000; Poff et al., 2002; Kling et al., 2003), invertebrates, invasive biota and waterfowl will shift
northwards (Moss et al., 2003; Zalakevicius and Svazas, 2005) while some will become extinct
(Jackson and Mandrak, 2002; Chu et al., 2005). Major changes will occur in the species composition,
seasonality and production of planktonic communities (e.g., increase in nuisance bluegreen algal
blooms) and their food web interactions (Gerten and Adrian, 2002; Kling et al., 2003; Winder and
Schindler, 2004). Studies along an altitudinal gradient in Sweden show that NPP can increase by an
order of magnitude by a 6°C air temperature increase (Karlsson et al., 2005). Tropical lakes may
however respond by a decrease in NPP and a decline in fish yields (e.g. due to last century warming
reductions in lake Tanganyika: 20% NPP, 30% fish yields O'Reilly et al., 2003). Higher CO₂ levels
will generally increase NPP, whereas in the tropics this may also stimulate methane fluxes, particularly
from paddy fields, thereby negating the positive effects (Ziska et al., 1998; Freeman et al., 2004).

Boreal peatlands will be affected most by warming and increased winter precipitation as the species
composition of both plant and animal communities will change significantly (Keller et al., 2004;
Weltzin et al., 2003; Weltzin et al., 2001; Weltzin et al., 2000; Berendse et al., 2001). Numerous
Arctic lakes will disappear with a 2-3°C temperature rise (Smith et al., 2005; Symon et al., 2005). The
seasonal migration patterns and routes of many wetland species will change considerably and their
survival will be reduced up to the risk of extinction (Reid et al., 2005; Finlayson et al., 2005; Inkley et
al., 2004; Zalakevicius and Svazas, 2005).

Small increases in the variability in precipitation regimes suffice to significantly impact wetland plants
and animals at different stages of their life cycle (Keddy, 2000). In monsoonal regions increased
variability risks to diminish wetland biodiversity and prolonged dry periods promote terrestrialization
of wetlands as witnessed in Keoladeo National Park (India Gopal and Chauhan, 2001; Chauhan and
Gopal, 2001). In dryland wetlands, changes in the precipitation regimes may cause the loss of
biodiversity (Bauder, 2005). An increase or decrease in freshwater flows will affect the coastal
wetlands (chapter 6) by altering salinity, sediment inputs and nutrients (Schallenberg et al., 2001;
Flöder and Burns, 2004).

Adaptation costs and opportunities: The climate change will increase significantly the costs of
treatment of water and possibly, the control of invasive species. It provides opportunities for
developing adaptive management strategies, technologies that ensure environmental flows for aquatic
ecosystem goods and services, and for improving various biodiversity related policies and conventions
(Boere and Taylor, 2004).

Key vulnerabilities: vulnerability varies between different geographical regions (Van Dam et al., 2002)
but both the water quality and ecosystem goods and services will generally deteriorate. Their close
relationship to hard and costly to control hydrological regimes and their often cross-national
dependence on catchments leave little scope for adaptation.

Impacts summary as a function of ΔT: Currently available data are mostly inadequate to specify
impacts as a function of temperature change. A 2-3°C temperature rise can increase the DOC release
by up to 700% (Frey and Smith, 2005)
Implications for policy and sustainable development: Anthropogenic impacts on aquatic ecosystems and their terrestrial catchments complicate the climate change impacts (Hartig et al., 2002; Kabat et al., 2004; Pyke and Marty, 2005). Increasing demand for water, coupled with decreased supply in some regions (de Wit and Stankiewicz, 2006) in the wake of uncertainties of precipitation will threaten the inland aquatic ecosystems directly. The future of the aquatic ecosystems will therefore depend upon the adoption of integrated water management strategies at the continental scale that adapt to changing availability of water resources.

4.4.9 Oceans, shallow seas and their ecosystems

Characteristics: Oceans cover over 71% of the earth’s surface area, they extend from polar to tropical regions and due to this significant coverage and depth (mean is 4000m) comprise about 98% of the Earth’s biosphere (14 billion km³). They are a massive reservoir of carbon and harbour a great biodiversity throughout their depths, both in the pelagic and benthic realms, and at their interface with land. For this reason they support a diverse range of ecosystems from highly productive (e.g. upwelling regions) to those with low productivity (e.g. oceanic gyres). Nearly half of global primary production is carried out by phytoplankton and is the basis of the marine food web (Field et al., 1998). Ocean primary productivity depends on sunlight and nutrients supplied from deep waters (Sarmiento et al., 2004a).

Goods and services: Marine ecosystems provide important and valuable goods and services (e.g. fisheries and aquaculture, provision of energy, recreation and tourism, CO₂ draw down and climate stabilization, decomposition of organic matter and regeneration of nutrients, coastal protection and they harbour an extensive biodiversity) many of which are critical to the functioning of the Earth system (Chapter 5, Costanza et al., 1997; Hassan et al., 2005 Table 18.2, p. 79; McCarthy et al., 2001 Sections 6.3.2, 6.3.4, 6.3.5, 6.4.5 and 6.4.6). Coastal zones, particularly low lying areas, and the highly valuable local and global socioeconomic services they provide (e.g. agricultural land, human settlements and associated infrastructure and industry, aquaculture and fisheries and freshwater supply) are particularly vulnerable to climate change (Hassan et al., 2005 Section 19.3.2, Table 19.2, p. 531, p. 676; McCarthy et al., 2001 Section 6.5, p. 362).

Key vulnerabilities as from TAR: Impacts of high atmospheric CO₂ on lower surface ocean pH and carbonate ions were overlooked in TAR. Coral reefs, cold water corals, and ecosystems (e.g. Southern Ocean) where aragonite will decline or become undersaturated and where calcareous organisms (e.g. pteropods) play an important role will become vulnerable this century (reviewed by Raven et al., 2005). Synergistic impacts of higher seawater temperatures and declining carbonate make these ecosystems even more vulnerable (e.g. Raven et al., 2005; Turley et al., 2006 Box 4.5). Marginal sea ice and surrounding ecosystems are vulnerable to warming, particularly in the Northern Hemisphere (Sarmiento et al., 2004b).

Box 4.5: Coral reefs – are coral reefs endangered by climate change?

Reefs are habitat for about a quarter of marine species and are the most diverse among marine ecosystems (Roberts et al., 2002; Buddemeier et al., 2004). They underpin local shore protection, fisheries, tourism (Chapter 6 Hoegh-Guldberg et al., 2000; Hoegh-Guldberg, 2004; Hoegh-Guldberg, 2005; Cesar et al., 2003; Willig et al., 2003), and though supplying only about 2-5% of the global fisheries harvest, comprise a critical subsistence protein source (Pauly et al., 2002).

Corals are affected by rising atmospheric CO₂ concentrations (WG I Box 7.3 Orr et al., 2005; Raven et al., 2005) and by warming of ocean surface waters (Box 6.1 Reynaud et al., 2003; McNeil et al., 2004;
McWilliams et al., 2005) leading to declining calcification and bleaching: Experiments at expected aragonite concentrations demonstrated a reduction in coral calcification (Langdon et al., 2003; Marubini et al., 2001; Hallock, 2005), coral skeleton weakening (Marubini et al., 2003) and strong temperature dependence (Reynaud et al., 2003). Oceanic pH projections go lower and at a greater rate than known from the past 20 million years (Turley et al., 2006; Raven et al., 2005; Caldeira and Wickett, 2003). Doubling CO$_2$ will reduce calcification in aragonitic corals by 20%-60% (Kleypas et al., 1999; Kleypas and Langdon, 2002; Reynaud et al., 2003; Raven et al., 2005). By 2070 many reefs could reach marginal aragonite saturation states (Feely et al., 2004; Orr et al., 2005), resulting in reduced coral cover and physically and biologically more erodible reef frameworks (Kleypas et al., 2001; Guinotte et al., 2003). Many studies incontrovertibly link coral bleaching to warmer SSTs (e.g. McWilliams et al., 2005) and mass bleaching and coral mortality often results beyond key temperature thresholds (see Box 6.1). Near-annual exceedance of current bleaching thresholds are projected for several sites globally by GCMs around 2040, and in some places even by 2020 (Hoegh-Guldberg, 1999; Sheppard, 2003). After bleaching, algae quickly colonise dead corals, possibly inhibiting later coral recruitment (e.g. McClanahan et al., 2001; Szmant, 2001; Gardner et al., 2003; Jompa and McCook, 2003). Modelling predicts a switch to algal dominance in 2030 to 2050 (Wooldridge et al., 2005; Langmead and Sheppard, 2007).

Adaptation potential (Hughes et al., 2003) by reef organisms requires further experimental and applied study (Coles and Brown, 2003; Hughes et al., 2003). Natural adaptive shifts to symbionts with +2°C resistance will delay demise of reef facies to roughly 2100 (Sheppard, 2003), rather than mid-century (Hoegh-Guldberg, 2005). Around 30% of warm water corals have disappeared in the past 20 years (Hoegh-Guldberg, 2005) due largely to increasing higher SST frequency (Hoegh-Guldberg, 1999). In some regions, such as the Caribbean, coral losses have been estimated at 80% (Gardner et al., 2003). Coral migration to higher latitudes with more optimal SST is unlikely, due both to latitudinally decreasing aragonite concentrations and projected atmospheric CO$_2$ increases (Kleypas et al., 2001; Guinotte et al., 2003; Orr et al., 2005; Raven et al., 2005). Elevated SST and decreasing aragonite have a complex synergy (McNeil et al., 2004; Kleypas et al., 2005; Reynaud et al., 2003) but could produce major coral reef changes (Hoegh-Guldberg, 2005; Guinotte et al., 2003). Corals could become rare on tropical and subtropical reefs by 2050 (at 2-3 x CO2 Raven et al., 2005; Hoegh-Guldberg, 2005; Kleypas and Langdon, 2002). Other climate change (sea-level rise, storm impact, aerosols) and

Fig. 4.3: Impacts of climate change and non-climate impacts on coral reefs (modified from Kleypas and Langdon, 2002). The box sizes are scaled relative to literature on climate impacts on reefs (Bleaching, Box 6.1; non-climate impacts Chapter 16; case study of the World Heritage site, the Great Barrier Reef Box 11.1).
non-climate factors (over-fishing, pollution, nutrient and sediment load, Boxes 6.2, 11.1, and chapter 16) add multiple impacts on coral reefs (Fig. 4.3), increasing vulnerability and reducing resilience to climate change (Kleypas and Langdon, 2002; Buddemeier et al., 2004; Hallock, 2005; Cole, 2003; Koop et al., 2001).

**Impacts:** Climate change can impact marine ecosystems through ocean warming (Wang et al., 2004b) by increasing thermal stratification and reducing upwelling (Cox et al., 2000; Sarmiento et al., 2004a), sea level rise (Houghton et al., 2001), increase in wave height and frequency (Monahan et al., 2000; Wang et al., 2004b), loss of sea ice (Sarmiento et al., 2004b) and decreases in the pH and carbonate ion concentration of the surface oceans (Caldeira and Wickett, 2003; Feely et al., 2004; Sabine et al., 2004; Raven et al., 2005).

Coupled physical/biogeochemical models predict a net ~5% decrease in global phytoplankton productivity (-20% in the tropics and +30% at high latitudes) and ~6% decrease in export production under 2xCO₂ due to increased oceanic thermal stratification and reduced upwelling of nutrients (Cox et al., 2000). Theoretically, nutrient speciation could also be influenced by the lower pH expected this century (Zeebe and Wolf-Gladrow, 2001; Raven et al., 2005). Decreases in both upwelling and formation of deep water and increased stratification of the upper ocean will reduce the input of essential nutrients into the sunlit regions of oceans and reduce productivity (Sarmiento et al., 2004a; Cox et al., 2000; Loukos et al., 2003; Lehodey et al., 2003).

Projections of ocean biological response to climate warming by 2050 show contraction of the highly productive marginal sea ice biome by 42% and 17% in Northern and Southern Hemispheres under IS92a (Sarmiento et al., 2004b). The sea-ice biome accounts for a large proportion of primary production in polar waters and supports a substantial food web. As timing of the spring phytoplankton bloom is linked to the sea ice edge, loss of sea ice (Walsh and Timlin, 2003) and large reductions of the total primary production in the Northern Hemisphere marginal sea ice biome in the Northern Hemisphere (Behrenfeld and Falkowski, 1997; Marra et al., 2003) would have strong effects, for example, on the productivity of the Bering Sea (Stabeno et al., 2001). Expansion of the low productivity permanently stratified subtropical gyre biome by 4.0% (Northern) and 9.4% (Southern Hemisphere) and the subpolar gyre biome by 16% (Northern) and 7% (Southern Hemisphere) and contraction of the seasonally stratified subtropical gyre by 11% in both hemispheres are also predicted by 2050 due to climate warming under IS92a (Sarmiento et al., 2004b). If the predictions of reduced productivity and reduction and expansion of key biomes are accurate then it seems likely that marine ecosystems will respond but the extent of this is difficult to predict based on current understandings (Sarmiento et al., 1998).

Major changes to planktonic and benthic community composition and productivity have been observed in North Sea since 1955 (Clark and Frid, 2001) and changes since the mid-1980s may have reduced the survival of young cod (Beaugrand et al., 2003). Elevated temperatures increased mortality of winter flounder eggs and larvae (Keller and Klein-Macphee, 2000) and a 2°C rise in SST would result in removal of Antarctic bivalves and limpets from the Southern Ocean (Peck et al., 2004). Tuna populations may spread towards presently temperate regions, based on predicted warming of surface water and increasing primary production at mid and high latitudes (Loukos et al., 2003). Warming El Niño events have had a positive impact on skipjack populations in the Pacific Ocean (Lehodey et al., 2003).

CO₂ emitted to the atmosphere by human activities has been absorbed by the oceans, making them more acidic. Surface ocean pH has already decreased by 0.1 unit (equivalent to a 30% increase in hydrogen ion concentration) and is predicted to decreases by a further 0.3-0.4 units by 2100 (Caldeira and Wickett, 2003). This may impact a wide range of organisms and ecosystems (e.g. Box 4.5, reviewed by Raven et al., 2005) including juvenile planktonic, as well as adult, forms of benthic calcifying organisms (e.g.
Echinoderms, gastropods and shellfish) and impact recruitment (reviewed by Turley et al., 2006). Aragonite undersaturation is predicted to occur in polar and subpolar surface waters and the Southern Ocean will be undersaturated by 2100 (Orr et al., 2005). Organisms using aragonite to make their shells (e.g. pteropods) will be at risk and this will threaten ecosystems such as the Southern Ocean in which they play a dominant role in the foodweb and carbon cycling (Orr et al., 2005). Cold-water coral ecosystems, have been found in almost all the world’s oceans and their aerial coverage could equal or exceed that of warm-water coral reefs (Freiwald et al., 2004; Guinotte et al., 2006). They harbour a distinct and rich ecosystem, provide habitats and nursery grounds for a variety of species, including commercial fish and numerous new species and previously thought to be extinct species (Raven et al., 2005). These geologically ancient, long-lived, slow growing and fragile reefs will suffer reduced calcification rates and as the aragonite saturation horizon moves towards the ocean surface large parts of the oceans will be uninhabitable to them by 2100 (Feely et al., 2004; Orr et al., 2005; Guinotte et al., 2006; Raven et al., 2005). Since cold-water corals do not have symbiotic algae but depend on extracting food particles sinking from surface waters or carried by ocean currents they are also vulnerable to changes to ocean currents, primary productivity and flux of food particles (Guinotte et al., 2006). Warm-water coral reefs are also sensitive to multiple impacts including increased sea surface temperature (SST) and decreasing carbonate ion concentrations. (Raven et al., 2005).

Adaptation costs and opportunities: Increasing global temperatures would shift species’ geographic ranges to higher latitudes or tidal heights due to changes in sea-level (time of exposure), and air and water temperature, but also cause localized extinctions at “hot spots” (Helmuth et al., 2002) but abilities of calcifying organisms to adapt to decreasing carbonate ions by changing their geographical distribution or physiology are of concern (Box 4.5). Future reduction of CO2 emissions is the only method of reducing impacts from ocean acidification (Raven et al., 2005). Protection and/or adaptation of climate change impacts on coastal zones, particularly low lying areas, and the highly valuable local and global socioeconomic services will be very costly economically, socially and culturally (McCarthy et al., 2001 6.5, p.362; Hassan et al., 2005 p. 677).

Key vulnerabilities: Known key vulnerable ecosystems are warm-water coral reefs (Box 4.5), cold-water corals, the Southern Ocean and marginal sea ice ecosystems.

Impacts summary as a function of $\Delta T$:

- 2 x CO2: net ~5% decrease in global phytoplankton productivity (-20% in the tropics and +30% at high latitudes Cox et al., 2000).
- 2 x CO2: net ~6% decrease in export production (Cox et al., 2000).
- 2050: contraction of the highly productive marginal sea ice biome by 42% and 17% in Northern and Southern Hemispheres (Sarmiento et al., 2004b).
- 2050: expansion of the low productivity permanently stratified subtropical gyre biome by 4.0% (Northern) and 9.4% (Southern Hemisphere) and the subpolar gyre biome by 16% (Northern) and 7% (Southern Hemisphere Sarmiento et al., 2004b).
- 2050: contraction of the seasonally stratified subtropical gyre biome by 11% in both hemispheres (Sarmiento et al., 2004b).
- 2°C (SST): loss of Antarctic bivalves and limpets from the Southern Ocean (Peck et al., 2004).
- ~1°C (SST): Major changes to planktonic and benthic community composition and productivity in North Sea (Clark and Frid, 2001; Beaugrand et al., 2003).
- ~1°C (SST): reduced survival of young cod in North Sea (Beaugrand et al., 2003).
- ~2°C (SST): increasing loss of warm-water coral reefs due to bleaching (Hoegh-Guldberg, 2005).
- ~2°C (SST): loss of warm-water corals globally – any adaptive symbionts unlikely to function (Sheppard, 2003).
- 2 x CO2: reduced calcification in warm water aragonitic corals by 20%-60% (Raven et al., 2005; Hoegh-Guldberg, 2005; Kleypas and Langdon, 2002).
- 2100: 70% reduction in cold-water aragonitic corals (Guinotte et al., 2006).
• 2100: Loss of aragonitic pteropods from Southern Ocean (Feely et al., 2004; Orr et al., 2005).

Impacts for policy and sustainable development: Coral reef impacts will be significant to sustainability of the Small Island Nations and to fisheries and tourism of nations adjacent to reef systems. Loss of sea ice and reduced productivity in surrounding waters will impact fisheries and biodiversity. Reduction or contraction of ocean biomes will result in redistribution of ocean productivity which will impact fisheries policies and sustainable development. Ocean uptake of anthropogenic CO₂ may concern conventions for the protection of the Earth’s seas (e.g. The London Convention, OSPAR). Marine ecosystems are very important in regulating the earth life support system and climate change impacts on them will be very costly economically, socially and culturally (McCarthy et al., 2001 6.5, p.362; Hassan et al., 2005 p. 677). The only way of regulating the impacts of ocean acidification is substantial reductions in CO₂ emissions (Raven et al., 2005).

4.4.10 Cross-biome impacts

This section highlights issues that cut across biomes, such as large-scale geographic shifts of vegetation (Fig. 4.4) or animal migration patterns (e.g. Box 4.4, Box 4.6), and changes in land use and aquatic systems.

Boreal forest and Arctic tundra ecosystems are projected to experience increased growth as a consequence of longer and warmer growing seasons (Lucht et al., 2002 see also Fig. 4.4). Woody boreal vegetation is expected to expand to higher latitudes and higher elevations at the expense of tundra (Grace et al., 2002; Kaplan et al., 2003; Gerber et al., 2004). At the southern ecotone to the grasslands of the continental interiors, increased drought stress is projected to cause a recession of boreal forest due to increased vulnerability to drought, insects and fires (Bachelet et al., 2001; Scholze et al., 2007), and a lower rate of sapling survival (Hogg and Schwarz, 1997). This forest die-back could be partially mitigated by concurrent CO₂-induced enhanced water use efficiency (Gerten et al., 2005), small regional increases in precipitation and an increased depth of permafrost thawing. It is uncertain whether peak summer heat stress on boreal tree species could potentially cause regional transitions to grassland where continental winter climate remains too cold for temperate forest species to succeed (Gerber et al., 2004; Scholze et al., 2007). Vegetation change in the lower to mid latitudes is uncertain because transitions between tropical desert and woody vegetation covers are particularly difficult to forecast. Climate models disagree in pattern and magnitude of projected changes in atmospheric circulation and climate variability, particularly for precipitation (e.g. with respect to the Indian and West African monsoons). For the Sahel and other semi-arid regions, increasing drought is predicted by some models (Held et al., 2005), while increased water use efficiency is projected to cause more greening, though potentially associated with more frequent fires, in others (Bachelet et al., 2003; Woodward and Lomas, 2004b; Schaphoff et al., 2006; Ni et al., 2006). Substantial drought stress in the Mediterranean region is reasonably certain (Hayhoe et al., 2004; Schroter et al., 2005). In savannahs, woody encroachment is projected to be a consequence of enhanced water use efficiency and increased precipitation in some regions (Bachelet et al., 2001; Schaphoff et al., 2006; Ni et al., 2006). A general increase of deciduous over evergreen vegetation is predicted at all latitudes, although the forests in both the eastern U.S. and eastern Asia appear to be sensitive to drought stress and decline under some scenarios (Bachelet et al., 2001; Scholze et al., 2007; Gerten et al., 2005). Non-woody C₃ vegetation is projected to increase over C₄ vegetation due to carbon fertilisation despite the competitive advantage of C₄ plants at higher temperatures (Collatz et al., 1998; Gerten et al., 2005). Tropical ecosystems are expected to change particularly in the Amazon, where a subset of GCMs shows strong to moderate reductions in precipitation with the consequence of transitions of evergreen tropical forest to raingreen forest or grasslands (Cox et al., 2004; Cramer et al., 2004; Woodward and Lomas, 2004b). However, representations of tropical succession remains underdeveloped in current models. The global land biosphere is projected to loose carbon beyond temperature increases of 3
degrees (Gerber et al., 2004), mainly from temperate and boreal soils, with vegetation carbon declining beyond temperature increases above 5 degrees (Gerber et al., 2004). Carbon sinks persist mainly in the Arctic and in savannah grasslands (Woodward and Lomas, 2004b; Schaphoff et al., 2006). However, there is large variability between the projections of different vegetation (Cramer et al., 2001) and climate (Schaphoff et al., 2006) models for a given emission scenario (Fig. 4.4).

![Map of projected changes in terrestrial ecosystems](image)

**Fig. 4.4:** Projected changes in terrestrial ecosystems as simulated by DGVM LPJ (Sitch et al., 2003; Gerten et al., 2004) for two IPCC SRES emission scenarios (Nakicenovic et al., 2000) forcing two climate models: a) HadCM3 A2, b) ECHAM5 B1 (Schaphoff et al., 2006 - see Fig. 4.2 for explanations and Table 4.3).

Most global scenario studies agree that land use change will be the dominant driver of terrestrial biodiversity loss, before changes in climate and other drivers such as nitrogen deposition (Jenkins, 2003; Carpenter et al., 2005; Sala et al., 2000; Scharlemann et al., 2004; Thomas et al., 2004a; UNEP, 2002) through 2050 in many ecosystems (see below). In freshwater systems (World Commission on Dams, 2000) and oceans (Pauly et al., 2002), extractive use has degraded biodiversity habitats, communities and foodwebs. The relative importance of these drivers varies across biomes (Sala et al., 2000). Climate change is likely to be more important than land use change where the magnitude of...
projected climate change is greatest and human intervention is limited such as in the tundra, boreal,  
cool conifer forests and savannah biomes (Duraiappah et al., 2005; Sala et al., 2000). In South African  
Fynbos, climate change is projected to threaten more endemic plant species richness than land use  
change by 2020 (Bomhard et al., 2005). All global studies project a significant reduction in native  
vegetation (principally forest) in non-industrialised countries and arid regions due to expansion of  
aricultural or urban land use driven principally by population growth, especially in Africa, South  
America and in South Asia where a reduction in native habitat will result in biodiversity loss  
(Carpenter et al., 2005). Globally, biodiversity (represented by species richness and relative  
abundance) may decrease by 12-16% by 2050 (Duraiappah et al., 2005) and are likely in virtually all  
biomes (Duraiappah et al., 2005 Re105; Sala et al., 2000), with tropical forest and woodland, savanna  
and warm mixed forest accounting for 80% of the species lost (~30,000 species) by 2050 (Duraiappah  
et al., 2005). Biomes that have been subjected to large change in the past such as Mediterranean  
forests and temperate grasslands will be least affected by land use change over the next 50 years.  
Northern latitude countries may become increasingly important for biodiversity and species  
conservation as the ranges of species distributions move poleward in response to climate change  
(Berry et al., 2006). Northern latitude countries are also sensitive to the effects of climate change on  
land use (especially agriculture) and these regions could support adaptation strategies to mitigate the  
negative effects of future climate and land use change. Biomes at the highest latitudes that have not  
already been converted to agriculture are likely to remain relatively unchanged in the future  
(Duraiappah et al., 2005).

In industrialized countries most development scenarios propose a decrease in agricultural land,  
particularly in northern temperate regions (Carpenter et al., 2005; Jenkins, 2003). Regional studies  
concur with this trend (Ewert et al., 2005; Rounsevell et al., 2005; Verburg et al., 2006). Much  
aricultural land occurs in areas of high biological value (and this trend is likely to continue in the  
future Scharlemann et al., 2004). Agricultural land abandonment may, therefore, create opportunities  
for biodiversity through habitat restoration (Kankaanpää and Carter, 2004; Rounsevell et al., 2006  
Rounsevell, 2006, Ro148), notwithstanding interactions with other drivers such as increases in fires  
due to extreme temperature events. Where the importation of agricultural commodities contributes to  
arland abandonment there may, however, be negative biodiversity impacts on ecosystems from the  
regions producing these imports (Cyranoski, 2005 De Oliveira, 2005, Ethanol; Commission of the  
European Communities, 2006). Thus, the distribution and flow of global food commodities is  
important to understanding cross-biome impacts. An alternative to agricultural land abandonment is  
etensification (e.g. through organic production) that may also have benefits for biodiversity by better  
accommodating native species (Rounsevell et al., 2005; Rounsevell et al., 2006). Regional scale  
analyses that integrate climate and land use change have been dominated by European studies  
(Dirnbock et al., 2003; Stefansenescu et al., 2004; Holman et al., 2005b; Holman et al., 2005a; Scheller  
and Mladenoff, 2005; Araújo et al., 2006; de Chazal et al., 2007; Giupponi et al., 2006; Harrison et al.,  
2006; Metzger et al., 2006; Schroter et al., 2005; Rounsevell et al., 2006). Studies on biodiversity in  
other world regions (e.g. North America and China) have tended to focus on the impacts of land use  
change in isolation (Lu et al., 2004; Santelmann et al., 2004; Conway and Lathrop, 2005). The effects  
of land use change on species at the regional scale may further fragment the overall trends arising from  
climate change (Harrison et al., 2006; Del Barrio et al., 2006; Rounsevell et al., 2006; Holman et al.,  
2005a). Habitat fragmentation is exacerbated when species have relatively low dispersal distances (Del  
Barrio et al., 2006) and land use change favours some species over others, depending on the relative  
‘hostility’ of the new environment (Giupponi et al., 2006). For example, agricultural abandonment in  
mountain regions will increase the distribution of forest species as forests encroach and decrease the  
distribution of open habitat (grassland) species (Giupponi et al., 2006).

Vagile animals such as polar bears (sea ice biome, tundra; Box 4.4) and in particular migratory birds  
tundra, wetlands, lakes, tropical forests etc., Box 4.6) are representatives of the need to analyze  
impacts both within and across biomes. Many species breed in one area then move to another to spend
the non-breeding season (Robinson et al., 2005). Indeed, many migratory species may be more vulnerable to climate change than resident species (Price and Root, 2005). As migratory species often move annually in response to seasonal climate changes, their behaviour, including migratory routes are highly sensitive to climate change. Indeed numerous studies have found that many of these species are arriving earlier (Chapter 1, e.g. Root et al., 2003). Changes in the timing of biological events are of particular concern because of a potential disconnect between migrants and their food resources if the phenology of each advance at different rates (Inouye et al., 2000; Root et al., 2003; Visser et al., 2004). The potential impact of climate change on migratory birds has been especially well studied (Box 4.6).

Box 4.6: Crossing biomes – impacts of climate change on migratory birds

Migratory species can be impacted by climate change in their breeding, wintering and/or critical stopover habitats. Models project changes in the future ranges of many species (Crick, 2004; Peterson et al., 2002; Price and Glick, 2002), some suggesting that the ranges of migrants may shift to a greater extent than non-migrants (Price and Root, 2001). In some cases this may lead to a lengthening and in others to a shortening of migration routes. Moreover, changes in wind patterns, in particular in relation to seasonal migration timing, could help or hinder migration (Butler et al., 1997). Other expected impacts include continuing changes in phenology, behavior, population sizes, and possibly genetics (reviewed in Crick, 2004; Robinson et al., 2005).

Many migratory species must cross barriers, e.g., the Sahara Desert, in moving between their wintering and breeding areas. Many species must stop in the Sahel to refuel en route from their breeding to their wintering areas. Degradation of vegetation quality in the Sahel (Box 4.3) could potentially lead to population declines in these species (Robinson et al., 2005).

More than 80% of the species living within the Arctic Circle winter farther south (Robinson et al., 2005). However, climate induced habitat change may be greatest in the Arctic (Symon et al., 2005; Zockler and Lysenko, 2000). For example, the Red Knot could potentially lose 16%-33% of its tundra breeding habitat by 2100 (HadCM2a1, UKMO). Additionally, at least some populations of this species could lose critical migratory stopover habitat (Delaware Bay, U.S.A.) to sea-level rise (Galbraith et al., 2002).

The breeding areas of many Arctic breeding waterfowl are projected to decline by 10-93% (Zockler and Lysenko, 2000). In North America’s Prairie Pothole region, models have projected an increase in drought with a 3°C regional temperature increase and varying changes in precipitation, leading to large losses of wetlands and to declines in the populations of waterfowl breeding there (Johnson et al., 2005). Many of these species also winter in coastal areas vulnerable to sea-level rise (Inkley et al., 2004).

Declines or range shifts in almost any species will have at least some impact on ecosystem services. For migrants, these impacts are spread over two or more areas. Some of these ecosystem services are intrinsic (pollination, insect control, seed dispersal) and some are extrinsic (existence value, recreation, subsistence, spiritual, religion). These services were discussed in the TAR and a more detailed discussion of services and recreational values can be found elsewhere (Price, 2003; Price and Glick, 2002).

Impacts of climate change will transcend across biomes. Higher CO2 concentrations lower the nutritional quality of the terrestrial litter (Lindroth et al., 2001; Tuchman et al., 2002; Tuchman et al., 2003a; Tuchman et al., 2003b) which in turn will affect the food web relationships of benthic communities in rivers. Greater amount of DOC released in the peatlands at higher CO2 levels is...
exported to the streams and finally reaches the coastal waters (Freeman et al., 2004).

Increasing global temperatures would shift the geographic ranges of marine biomes to higher latitudes or tidal heights due to changes in sea-level (time of exposure), and air and water temperature, but could also cause localised extinctions at “hot spots” (Helmuth et al., 2002). The ability, however, of calcifying organisms to adapt to decreasing carbonate ions by changing their geographical distribution or physiology is unknown.

4.4.11 Global synthesis including impacts on biodiversity

Since the TAR considerable progress has been made in key fields that allow projection of climate change impacts on species and ecosystems, and the synthesis of these combined results provides a picture, still imperfect, of projected impacts. However, two of these key fields, namely bioclimatic (also called niche-based) modelling and dynamic global vegetation modelling (DGVM) remain quite removed from each other in approach, assumptions, limitations of scale and application. Bioclimatic modelling has been reinvigorated by the availability of significant species distribution data, together with finer scale climate data that have allowed this correlative method to be widely applied.

Uncertainties are many, and center on the problem of simulating the fundamental vs. realized niche, that in turn undermine confidence in range computation and future projection, ignorance of inter-species interactions, inadequate treatment of dispersal and migration potential, converting range loss projection to risk of extinction, and limited use for rare range-restricted species. In modelling ecosystem function and plant functional type response, there has been a considerably greater leap in understanding and enhanced confidence in the ecophysiological processes affected by climate change and the mechanisms by which climate change may impact biomes, ecosystem components such as soils, fire behaviour and vegetation structure (ie biomass distribution and leaf area index), bolstered by important experimental results. Limitations of DGVM now relate to assumptions of free dispersal of species comprising the abstract plant functional types that make up ecosystem components.

Nonetheless, where ecosystem degradation (loss of functional types) is projected using this method, these carry credibility of incorporating lags in ecosystem components and more defensible assumptions than for bioclimatic modelling. Thus, it is reasonable to expect major changes in the DGVM vegetation structure to equate to change in the structure of communities and abundances of populations, leading to geographical shifts of PFT-dominance with major changes in ecosystem function. These in turn will translate into impacts on ecosystem goods and services.

Lower levels of atmospheric CO2 rise and climate change may be beneficial in some regions, depending on latitude, on the CO2 responsiveness of plant functional types, and on the natural adaptive capacity of indigenous biota, mainly through range shifts that are now being observed. But as levels of change increase greater impacts are projected while ecosystem and species response may be lagged, especially in forests and in soils. At such tipping points (Fig. 4.5b) crucial services like carbon sequestration may cease to function as they do now. While these points are difficult to identify without substantial uncertainties, they may lead to irreversible effects such as biodiversity loss or, at the very least, impacts that have a slow recovery time (e.g. soils, corals). Climate changes are further imposed on ecosystems already experiencing substantial pressures of largely detrimental effects. Roughly 60% of evaluated ecosystems are currently unsustainably utilised and show increasing signs of degradation (Reid et al., 2005; Hassan et al., 2005). This alone is likely to cause widespread biodiversity loss (Chapin et al., 2000; Jenkins, 2003; Reid et al., 2005), given that 12% to 52% of major species groups (15,589 species) are threatened with global extinction (Baillie et al., 2006). Threat of extinction varies among groups, with birds and amphibians and continental species, particularly at high latitudes and altitudes, generally at greatest risk (Baillie et al., 2006). Climate change trends will exacerbate these pressures substantially, based on improving knowledge about their impacts on biodiversity as function of changes relative to the current climate (Table 4.2, Fig. 4.5).
Table 4.2: Projected impacts of climate change on ecosystems and population systems as reported in the literature for different levels of global mean annual temperature rise, $\Delta T_{\text{glob}}$, relative to preindustrial climate (event numbers as used in Fig. 4.5). Projections from the literature were harmonized into a common framework by upscaling (where necessary) from local to global temperature rise using the HadCM3 GCM regional output, upon which many of the studies were based, and by using a common global mean temperature reference point for the year 1990 (after Warren, 2006). Whilst some of the literature relates impacts directly to global mean temperature rises (column $\Delta T_{\text{reg}}$ left blank), many studies give only local temperature rises, $\Delta T_{\text{reg}}$, and hence require upscaling (HadCM2/3 IS92a outputs from Hulme et al., 1999 used). Whenever possible the table reports the GCM used.

<table>
<thead>
<tr>
<th>No</th>
<th>$\Delta T_{\text{glob}}$ above pre-industrial</th>
<th>$\Delta T_{\text{reg}}$ above 1990</th>
<th>Impacts to unique or widespread ecosystems or population systems</th>
<th>Region affected</th>
<th>Assessment / GCM</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>$&lt;1$</td>
<td></td>
<td>Increased coral bleaching causes loss of reefs</td>
<td>Caribbean, Indian Ocean, Great Barrier Reef</td>
<td>M / 4 GCMs</td>
<td>2</td>
</tr>
<tr>
<td>2</td>
<td>$&lt;1$</td>
<td></td>
<td>Risk extinctions in Dryandra forest</td>
<td>Australia</td>
<td>E</td>
<td>3</td>
</tr>
<tr>
<td>3</td>
<td>$&lt;1$</td>
<td></td>
<td>Range loss begins for Golden Bowerbird</td>
<td>Australia</td>
<td>M</td>
<td>4</td>
</tr>
<tr>
<td>4</td>
<td>$&lt;1$</td>
<td></td>
<td>Antarctic marine ecosystems affected by continued reductions in krill reducing Adelie penguin populations</td>
<td>Antarctica</td>
<td>E</td>
<td>42</td>
</tr>
<tr>
<td>5</td>
<td>$&lt;1$</td>
<td></td>
<td>Range losses begin for animal species in S Africa</td>
<td>S Africa</td>
<td>M / H2, H3</td>
<td>5</td>
</tr>
<tr>
<td>6</td>
<td>1.5</td>
<td>1.4</td>
<td>Coral reefs regionally functionally extinct</td>
<td>Great Barrier Reef, SE Asia, Caribbean, Indian Ocean</td>
<td>M</td>
<td>2</td>
</tr>
<tr>
<td>7</td>
<td>1.5</td>
<td>1.4</td>
<td>Coral reefs regionally functionally extinct</td>
<td>Great Barrier Reef, SE Asia, Caribbean, Indian Ocean</td>
<td>M</td>
<td>9</td>
</tr>
<tr>
<td>8</td>
<td>1.6</td>
<td>1</td>
<td>9 – 31% (mean 18%) all species extinct</td>
<td>N America</td>
<td>M / GCMs range</td>
<td>13</td>
</tr>
<tr>
<td>9</td>
<td>$&lt;1.7$</td>
<td>1</td>
<td>8% loss freshwater fish habitat, 15% loss in Rocky Mountains, 9% loss of salmon</td>
<td>N America</td>
<td>M / GCMs range</td>
<td>13</td>
</tr>
<tr>
<td>10</td>
<td>1.7</td>
<td>1</td>
<td>10% Global Ecosystems transformed; loss 47% wooded tundras (taiga), 23% cool conifer forest, 21% scrubland, 15% grassland/steppe, 14% savannah, 13% tundra and 12% temperate deciduous forest. Ecosystems variously lose 2 – 47% extent.</td>
<td>Globe</td>
<td>M / 5 GCMS</td>
<td>6</td>
</tr>
<tr>
<td>11</td>
<td>1.7</td>
<td>1</td>
<td>Extinction reptiles (7 – 14%) frogs (8 – 18%) birds (7 – 10%), mammals (10 – 15%) in Queensland as 47% of habitat lost</td>
<td>Australia</td>
<td>M / H2</td>
<td>1, 7</td>
</tr>
<tr>
<td>12</td>
<td>1.7</td>
<td>1</td>
<td>Extinction plants in Cerrado (38 – 45%, no dispersal only)</td>
<td>Brazil</td>
<td>M / H2</td>
<td>1</td>
</tr>
<tr>
<td>13</td>
<td>1.7</td>
<td>1</td>
<td>Extinction mammals (2 – 18%), birds (2 – 8%) and butterflies (1 – 11%)</td>
<td>Mexico</td>
<td>M / H2</td>
<td>1</td>
</tr>
<tr>
<td>14</td>
<td>1.8</td>
<td>1</td>
<td>Risk functional extinction of Golden Bowerbird: habitat reduced by 50%</td>
<td>Australia</td>
<td>M</td>
<td>5</td>
</tr>
<tr>
<td>15</td>
<td>1.8</td>
<td>1</td>
<td>Extensive loss/conversion of habitat in Kakadu wetland due to sea level rise and saltwater intrusion</td>
<td>Australia</td>
<td>M / CS</td>
<td>10</td>
</tr>
<tr>
<td>16</td>
<td>2</td>
<td>1</td>
<td>Functional extinction of most coral reefs</td>
<td>Globe Arctic</td>
<td>M</td>
<td>2</td>
</tr>
<tr>
<td>17</td>
<td>2</td>
<td>1</td>
<td>Total loss Arctic summer ice, high risk of extinction of polar bears, walrus, seals, whole ecosystems stressed</td>
<td>Arctic</td>
<td>M</td>
<td>11</td>
</tr>
<tr>
<td>18</td>
<td>2</td>
<td>1</td>
<td>42% existing Arctic tundra remains stable</td>
<td>Arctic</td>
<td>M</td>
<td>14</td>
</tr>
<tr>
<td>No</td>
<td>$\Delta T_{\text{glob}}$ above pre-industrial</td>
<td>$\Delta T_{\text{reg}}$ above 1990</td>
<td>Impacts to unique or widespread ecosystems or population systems</td>
<td>Region affected</td>
<td>Assessment / GCM</td>
<td>Source</td>
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<tr>
<td>19</td>
<td>2</td>
<td></td>
<td>Millions of Arctic nesting shorebirds lose 10%-45% of breeding area; high Arctic species most at risk.</td>
<td>Arctic</td>
<td>M</td>
<td>14</td>
</tr>
<tr>
<td>20</td>
<td>2</td>
<td></td>
<td>Millions of Arctic nesting geese lose up to 50% breeding area.</td>
<td>Arctic</td>
<td>M</td>
<td>14</td>
</tr>
<tr>
<td>21</td>
<td>2</td>
<td></td>
<td>Extinction of plants (3-14%)</td>
<td>Europe</td>
<td>M / H2</td>
<td>1</td>
</tr>
<tr>
<td>22</td>
<td>2</td>
<td></td>
<td>Severe damage (59% loss) to boreal forest</td>
<td>China</td>
<td>M</td>
<td>15</td>
</tr>
<tr>
<td>23</td>
<td>2</td>
<td>2</td>
<td>16% freshwater fish habitat loss, 28% loss in Rocky Mts, 18% loss salmon</td>
<td>N America</td>
<td>M / GCMs range</td>
<td>13</td>
</tr>
<tr>
<td>24</td>
<td>1.8 – 2.4</td>
<td>1 – 2</td>
<td>Alpine systems in Alps can tolerate local temperature rise of 1-2C, but tolerance likely to be negated by land use change</td>
<td>Europe</td>
<td>E</td>
<td>8</td>
</tr>
<tr>
<td>25</td>
<td>2.1</td>
<td></td>
<td>Extinction of plants (3 – 16%)</td>
<td>Europe</td>
<td>M</td>
<td>1</td>
</tr>
<tr>
<td>26</td>
<td>2.1</td>
<td>1.4 – 2.6</td>
<td>Extinction butterflies (13 – 23%)</td>
<td>Australia</td>
<td>M / H2</td>
<td>1</td>
</tr>
<tr>
<td>27</td>
<td>2.2</td>
<td>2</td>
<td>60% N American wood warblers ranges contract, whilst only 8% expand, 4 to 13 (34%) reach “vulnerable” status</td>
<td>N America</td>
<td>M</td>
<td>43</td>
</tr>
<tr>
<td>28</td>
<td>2.2</td>
<td></td>
<td>15 – 37% (mean 24%) species extinct</td>
<td>Globe</td>
<td>M</td>
<td>1</td>
</tr>
<tr>
<td>29</td>
<td>2.2</td>
<td></td>
<td>8 – 12% of 277 medium/large mammals from 28 families in 141 African national parks critically endangered or extinct; further 22 – 25% endangered</td>
<td>Africa</td>
<td>M / H2</td>
<td>23</td>
</tr>
<tr>
<td>30</td>
<td>2 – 2.5</td>
<td></td>
<td>Fish populations decline, wetland ecosystems dry and disappear</td>
<td>Malawi, African Great Lakes</td>
<td>E</td>
<td>20</td>
</tr>
<tr>
<td>31</td>
<td>2 – 2.5</td>
<td></td>
<td>New England maples at risk impacting tourism</td>
<td>USA (fall colour)</td>
<td>E</td>
<td>20</td>
</tr>
<tr>
<td>32</td>
<td>2.3</td>
<td>1.8</td>
<td>Extinctions (100% potential range loss) 10% endemics in hotspot for plant biodiversity; 51 – 65% loss of Fynbos</td>
<td>S Africa</td>
<td>M / H2, CSM</td>
<td>24, 44</td>
</tr>
<tr>
<td>33</td>
<td>2.3</td>
<td></td>
<td>Extinction mammals (2 – 20%), birds (3 – 8%) and butterflies (3 – 15%)</td>
<td>Mexico</td>
<td>M / H2</td>
<td>1</td>
</tr>
<tr>
<td>34</td>
<td>2.3</td>
<td>1.8</td>
<td>Extinction 48 – 57% Cerrado plants</td>
<td>Brazil</td>
<td>M / H2</td>
<td>1</td>
</tr>
<tr>
<td>35</td>
<td>2.3</td>
<td></td>
<td>Extinction of 21 – 40% of Proteaceae (based on range loss and species area assumptions)</td>
<td>South Africa</td>
<td>M / H2, M / H2</td>
<td>1, 44</td>
</tr>
<tr>
<td>36</td>
<td>2.3</td>
<td>1.8</td>
<td>Succulent Karoo reduced to 20% of area, threatening 2800 plants with extinction; 5 S African parks lose &gt; 40% plant species</td>
<td>S Africa</td>
<td>M / H2</td>
<td>5, 25</td>
</tr>
<tr>
<td>37</td>
<td>2.4</td>
<td></td>
<td>Transformation of ecosystems e.g. 32% of plants move from 44% European area with potential extinction of endemics</td>
<td>N Europe</td>
<td>M</td>
<td>16</td>
</tr>
<tr>
<td>38</td>
<td>2.4</td>
<td></td>
<td>63 of 165 rivers studied lose &gt;10% of their fish species</td>
<td>World</td>
<td>M / H3</td>
<td>19</td>
</tr>
<tr>
<td>39</td>
<td>2.4</td>
<td></td>
<td>Large range loss animals &amp; risk extinctions of 11% species</td>
<td>Mexico</td>
<td>M / H2</td>
<td>26</td>
</tr>
<tr>
<td>40</td>
<td>2.5</td>
<td></td>
<td>Complete loss alpine zone</td>
<td>Australia</td>
<td>E</td>
<td>20</td>
</tr>
<tr>
<td>41</td>
<td>2 – 3</td>
<td></td>
<td>Amazon collapse – huge loss of biodiversity</td>
<td>S America, Globe</td>
<td>M</td>
<td>21</td>
</tr>
<tr>
<td>42</td>
<td>2.6</td>
<td></td>
<td>Extinction of plants (4 – 21%)</td>
<td>Europe</td>
<td>M / H2</td>
<td>1</td>
</tr>
<tr>
<td>43</td>
<td>2.6</td>
<td></td>
<td>20 – 70% loss (average 44%) bird habitat at 4</td>
<td>USA</td>
<td>M</td>
<td>29</td>
</tr>
<tr>
<td>No</td>
<td>$\Delta T_{\text{glob}}$ above pre-industrial</td>
<td>$\Delta T_{\text{reg}}$ above 1990</td>
<td>Impacts to unique or widespread ecosystems or population systems</td>
<td>Region affected</td>
<td>Assessment / GCM</td>
<td>Source</td>
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<tr>
<td>44</td>
<td>2.6</td>
<td></td>
<td>major coastal sites</td>
<td>Globe</td>
<td>M</td>
<td>1</td>
</tr>
<tr>
<td>45</td>
<td>2.7</td>
<td></td>
<td>21 – 52% (mean 35%) species extinct</td>
<td>Globe</td>
<td>M / 5</td>
<td>6</td>
</tr>
<tr>
<td>46</td>
<td>2.7</td>
<td></td>
<td>2 – 10% European plants critically endangered or extinct; mean species turnover of 48% (spatial range 17 – 75%); mean species loss of 27% (spatial range 1- 68%)</td>
<td>Europe</td>
<td>M / H3</td>
<td>22</td>
</tr>
<tr>
<td>47</td>
<td>2.9</td>
<td>2.1 – 3.9</td>
<td>Extinction butterflies (21 – 36%)</td>
<td>Australia</td>
<td>M / H2</td>
<td>1</td>
</tr>
<tr>
<td>48</td>
<td>2.9</td>
<td>2.1 – 3.9</td>
<td>Extinction of plants (4 – 100%) as Amazon dries</td>
<td>Amazon</td>
<td>M / H2</td>
<td>1</td>
</tr>
<tr>
<td>49</td>
<td>2.9</td>
<td>2.1 – 3.9</td>
<td>&gt;50% range loss for 83% of 24 latitudinally restricted endemic butterflies</td>
<td>Australia</td>
<td>M / 10</td>
<td>30</td>
</tr>
<tr>
<td>50</td>
<td>3</td>
<td>3</td>
<td>21 – 52% (mean 35%) species extinct</td>
<td>Globe</td>
<td>M</td>
<td>1</td>
</tr>
<tr>
<td>51</td>
<td>3</td>
<td>3</td>
<td>Extinctions of 200 - 300 species (32 – 63%) of alpine flora</td>
<td>New Zealand</td>
<td>M</td>
<td>33</td>
</tr>
<tr>
<td>52</td>
<td>3</td>
<td></td>
<td>66 of 165 rivers studied lose &gt;10% of their fish species</td>
<td>Globe</td>
<td>M / H3</td>
<td>19</td>
</tr>
<tr>
<td>53</td>
<td>3</td>
<td></td>
<td>Increased fire frequency converts forest &amp; macquis to scrub, increased pest outbreaks</td>
<td>Mediterranean</td>
<td>M / H3</td>
<td>34</td>
</tr>
<tr>
<td>54</td>
<td>3</td>
<td></td>
<td>Risk extinction of 90% Hawaiian honeycreeper birds</td>
<td>Hawaii</td>
<td>M</td>
<td>18</td>
</tr>
<tr>
<td>55</td>
<td>3</td>
<td></td>
<td>Complete loss boreal forest</td>
<td>China</td>
<td>M</td>
<td>15</td>
</tr>
<tr>
<td>56</td>
<td>3</td>
<td></td>
<td>Large loss migratory bird habitat</td>
<td>Baltic, USA, Mediterranean Africa, Indonesia</td>
<td>M / H3</td>
<td>35, 36</td>
</tr>
<tr>
<td>57</td>
<td>3.1</td>
<td>2.1 – 2.5</td>
<td>Cloud forest regions lose hundreds of metres of elevational extent, potential extinctions</td>
<td>C. America, Tropical Africa, Indonesia</td>
<td>M / GE</td>
<td>17</td>
</tr>
<tr>
<td>58</td>
<td>3.1</td>
<td>3</td>
<td>Loss of 9 – 62% mammal species from mountainous areas of Great Basin</td>
<td>USA</td>
<td>M</td>
<td>32</td>
</tr>
<tr>
<td>59</td>
<td>3.1</td>
<td>3</td>
<td>24% loss freshwater fish habitat, 40% loss in Rocky Mountains, 27% loss of salmon. 50% loss world’s most productive duck habitat in prairie pothole region</td>
<td>N America</td>
<td>M / GCMs</td>
<td>13</td>
</tr>
<tr>
<td>60</td>
<td>3.1</td>
<td>3</td>
<td>66% animals lost from Kruger; 29 endangered species lost &gt;50% range; 4 species locally extinct</td>
<td>S Africa</td>
<td>M / H2</td>
<td>27</td>
</tr>
<tr>
<td>61</td>
<td>2.85 – 3.6</td>
<td>2 – 3</td>
<td>High risk extinction of Golden Bowerbird: at 2C local temperature rise habitat reduced by 90% and at 3C by 96%</td>
<td>Australia</td>
<td>M</td>
<td>4</td>
</tr>
<tr>
<td>62</td>
<td>3.3</td>
<td>2.5 – 3.0</td>
<td>Extinction mammals (24 – 59%), birds (28 – 40%), butterflies (13 – 70%), other invertebrates (18 – 80%), reptiles (21 – 45%)</td>
<td>South Africa</td>
<td>M / H2</td>
<td>1</td>
</tr>
<tr>
<td>63</td>
<td>3.3</td>
<td>3.7</td>
<td>Extinction birds (4 – 38%)</td>
<td>Europe</td>
<td>M / H3</td>
<td>1</td>
</tr>
<tr>
<td>64</td>
<td>3.3</td>
<td></td>
<td>77% loss low tundra</td>
<td>Canada</td>
<td>M</td>
<td>39</td>
</tr>
<tr>
<td>65</td>
<td>3.4</td>
<td></td>
<td>22% loss coastal wetlands</td>
<td>Globe</td>
<td>M / H2, H3</td>
<td>35</td>
</tr>
<tr>
<td>66</td>
<td>3.4</td>
<td></td>
<td></td>
<td></td>
<td></td>
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</tr>
</tbody>
</table>
### Impacts to unique or widespread ecosystems or population systems

<table>
<thead>
<tr>
<th>No</th>
<th>$\Delta T_{glob}$ above pre-industrial</th>
<th>$\Delta T_{reg}$ above 1990</th>
<th>Region affected</th>
<th>Assessment / GCM</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>67</td>
<td>3.7</td>
<td></td>
<td>Globe</td>
<td>M</td>
<td>6</td>
</tr>
<tr>
<td>68</td>
<td>3.7</td>
<td></td>
<td>Globe</td>
<td>M / 5 GCMs</td>
<td>6</td>
</tr>
<tr>
<td>69</td>
<td>3.7</td>
<td></td>
<td>Globe</td>
<td>M / GCMs range</td>
<td>6</td>
</tr>
<tr>
<td>70</td>
<td>3.7</td>
<td>3</td>
<td>Australia</td>
<td>M</td>
<td>12</td>
</tr>
<tr>
<td>71</td>
<td>3.7</td>
<td>2</td>
<td>Hawaii</td>
<td>M</td>
<td>18</td>
</tr>
<tr>
<td>72</td>
<td>3.7</td>
<td></td>
<td>Namibia</td>
<td>M / H3 range</td>
<td>39</td>
</tr>
<tr>
<td>73</td>
<td>3.7</td>
<td></td>
<td>Africa</td>
<td>M / H3 range</td>
<td>23</td>
</tr>
<tr>
<td>74</td>
<td>3.8</td>
<td></td>
<td>Globe</td>
<td>M</td>
<td>40</td>
</tr>
<tr>
<td>75</td>
<td>3.8</td>
<td></td>
<td>Globe</td>
<td>M</td>
<td>40</td>
</tr>
<tr>
<td>76</td>
<td>4</td>
<td></td>
<td>Australia</td>
<td>M</td>
<td>12</td>
</tr>
<tr>
<td>77</td>
<td>4.1</td>
<td></td>
<td>USA</td>
<td>M</td>
<td>43</td>
</tr>
<tr>
<td>78</td>
<td>4.2</td>
<td>2.5 – 3.5</td>
<td>Mexico</td>
<td>M / CCC, GF</td>
<td>28</td>
</tr>
<tr>
<td>79</td>
<td>4.2</td>
<td></td>
<td>Australia</td>
<td>M</td>
<td>7</td>
</tr>
<tr>
<td>80</td>
<td>4.3</td>
<td></td>
<td>Europe</td>
<td>M / H3 range</td>
<td>22</td>
</tr>
<tr>
<td>81</td>
<td>3.7 – 4.7</td>
<td>3 – 4</td>
<td>Europe</td>
<td>M</td>
<td>41</td>
</tr>
<tr>
<td>82</td>
<td>5.5</td>
<td></td>
<td>USA</td>
<td>M</td>
<td>29</td>
</tr>
<tr>
<td>83</td>
<td>5.7</td>
<td></td>
<td>Australia</td>
<td>M</td>
<td>7</td>
</tr>
<tr>
<td>84</td>
<td>~5.7</td>
<td>5</td>
<td>Australia</td>
<td>M</td>
<td>12</td>
</tr>
<tr>
<td>85</td>
<td>7.7</td>
<td></td>
<td>Australia</td>
<td>M</td>
<td>7</td>
</tr>
</tbody>
</table>

1. *the study of Leemans and Eickhout 2004 used the following GCMs: HadCM2, GFDL, ECHAM4, CSIRO-MK2, CGCM1
2. **relative to 1961-1990 temperatures.
Fig. 4.5: Critical climate change impacts on ecosystems for different levels of global mean annual temperature rise, $\Delta T$, relative to preindustrial climate: a) where, b) when (approach and event numbers as used in Table 4.2). It is important to note that these impacts do not take account of ancillary stresses on species due to over-harvesting, habitat destruction, landscape fragmentation, alien species invasions, fire regime change, pollution (such as nitrogen deposition), or for plants the potential beneficial effects of rising atmospheric $\text{CO}_2$. 
Unless major changes in policy and way of life take place (Jenkins, 2003), climate change impacts and an intensified human dominance of ecosystems (Vitousek et al., 1997b and others in Science, 277 issue 5325) will diminish the areal extent (e.g. reduction by 5-66% alone due to 2°C warming, Table 4.2, Fig. 4.5) and disrupt nearly all ecosystems of the globe (16% affected alone due to 2°C warming, Table 4.2, Fig. 4.5). Those impacts will vary regionally and across biomes and will lead to a global biodiversity loss, as expressed through area reductions of wild habitats and declines in the abundance of wild species putting those species at risk of extinction (e.g. 3-16% of European plants alone due to 2.1°C or up to 100% of Amazon plants alone due to 2.9°C warming, Table 4.2, Fig. 4.5).

Globally, biodiversity (represented by species richness and relative abundance) may decrease by 12-16% (Sala, 2005) due to a combination of climate and land use changes with tropical forest and woodland, savanna and warm mixed forest accounting for 80% of the species lost (~30,000 species) by 2050. Independent of climate change earlier estimates have projected that about 50% of species being at risk of extinction if 90% of ecosystems would be degraded by land use change pressures or other non-sustainable exploitatations (Soulé and Sanjayan, 1998). Overall, through 2050, climate change is estimated to be about twice as strong a driver for biodiversity loss in the Arctic tundra as land use change, whilst in cool conifer forest and savannas it is considered to be 2 to 3 times more important (Carpenter et al., 2005). In other ecosystems land use change may be a stronger driver of biodiversity loss. However, an analysis of the SRES scenarios to 2100 (Strengers et al., 2004) predicts that deforestation ceases in all scenarios except A2, suggesting that beyond 2050 climate change will be the major driver for biodiversity loss rather than land use change. Malcolm et al. (2006) find that by 2100 between 1% and 43% of endemic species (average 11.6%) will be committed to extinction, whereas Thomas et al. (2004a) report already by 2050 a range from 15% to 37%. Malcolm et al. (2006) argue that climate-change induced extinction rates in hotspots for biodiversity in the tropics exceed the predicted extinctions from deforestation, suggesting that at least in some cases climate change is a more serious threat to biodiversity than land use change. In the Mediterranean-climate region of South Africa, climate change may have at least as significant an impact in increasing species extinction risk of endemic Proteas as land use change does by 2020 (Bomhard et al., 2005). In summary, it is likely that only temperate and warm mixed forest species might have a smaller impact from climate change than from land use change. However, if there is a 44% risk of a terrestrial carbon source with a 3°C temperature rise above 1990 (Scholze et al., 2007 see below), this would imply a worldwide decline of forests, at least and suggest here too, that climate change could become a stronger driver of biodiversity loss than land use change.

Loss of biodiversity will lead to decreases in the provision of ecosystem goods and services with trade-offs between ecosystem services likely to intensify (Fig. 1 National Research, 1999; Duraipappah et al., 2005; Carpenter et al., 2005). Gains in provisioning services (e.g., food supply, water use) are projected to occur in part at the expense of other regulating and supporting services including genetic resources, habitat provision, climate and runoff regulation. Projected changes may also increase the likelihood of ecological surprises that are detrimental for human well-being (Duraipappah et al., 2005; Burkett et al., 2005). Ecological surprises include rapid and abrupt changes in temperature and precipitation leading to an increase in extreme events such as floods, fires and landslides, increases in eutrophication, or rapid and sudden increases in disease (Carpenter et al., 2005). This entails also entire ecosystems suddenly shifting their behavior to a new mode (Scheffer et al., 2001; Folke et al., 2004), once gradual pressures such as warming reach a critical level and any services dependent on the previous mode may get lost. Reparation may then become extremely costly or due to hysteresis effects very time consuming, if not impossible.

DGVM modelling of two climate scenarios that represent intermediate low and intermediate high rates (ECHAM5 B2 and HadCM3 A2 respectively) which in ensemble runs generate warming of 2 °C and 3.8°C relative to 1980-1999 baseline by 2100 respectively, reveal a potentially complex sequence of
ecosystem changes as climate change may evolve (Fig. 4.4, Table 4.3). In the B1 scenario (Fig. 4.4b) undergo 55% of extant ecosystems substantive changes. This scenario reveals some positive impacts especially in Africa and the southern Hemisphere, as desert green-up is projected in southern Africa, the Sahel region, central Australia, and also the Arabian Peninsula and parts of central Asia. As ensemble climate models suggest moderate drying in these regions, therefore this response may be due to a positive impact of rising atmospheric CO2, previously noted by (Thuiller et al., 2007b) in eastern Namibia, and attributed by them to CO2 by sensitivity analysis. However, in mid to high latitudes on all continents, substantial shifts in forest structure toward more rain-green, summer-green or deciduous rather than evergreen forest, and forest and woodland decline underlie the overall drop in global terrestrial carbon sequestration potential that occurs post 2030, and approaches a net source by 2100.

In the A2 scenario (Fig. 4.4a) undergo 69% of extant ecosystems substantive changes. Desert green-up persists in the regions described above, but substantial decline of forest and woodland at northern, tropical and subtropical latitudes underly a far more negative carbon balance. The current global sink provided by all terrestrial ecosystems deteriorates perhaps already post 2020, and by 2100 and beyond (not shown) the terrestrial biosphere turns globally into an increasing substantial carbon source (Fig. 4 Scholze et al., 2007).

Table 4.3: Major biome changes projected by LPJ forced by ECHAM5 B1 scenario and HadCM3 A2 scenario (1 – Sabine et al., 2004) (2 – savannas Bonan, 2002; + forest Sabine et al., 2004 - see Fig. 4.2 for explanations and Fig. 4.4 for maps on underlying ecosystem changes)

<table>
<thead>
<tr>
<th>Vegetation Change (VC)</th>
<th>B1 biome area change (ΔT2100-preind. +2ºC)(Mkm²)</th>
<th>A2 biome area change (ΔT2100-preind. +3.8ºC)(Mkm²)</th>
</tr>
</thead>
<tbody>
<tr>
<td>4: Desert green-up</td>
<td>27.7 (+22%)</td>
<td>6.2 (+22%)</td>
</tr>
<tr>
<td>6: Forest/woodland decline</td>
<td>-4.1 (-12%)</td>
<td>-12.1 (-29%)</td>
</tr>
<tr>
<td>1+2+3: Forest/woodland expansion</td>
<td>12.7 (+31%)</td>
<td>16.6 (+40%)</td>
</tr>
<tr>
<td>1+2+3-6: Net forest/woodland change</td>
<td>8.6 (+21%)</td>
<td>4.5 (+11%)</td>
</tr>
</tbody>
</table>

4.5 Costs and valuation of ecosystem goods and services

While many efforts have been made to estimate the economic value of ecosystem goods and services (Costanza, 2000; Costanza, 2001; Costanza et al., 1997; Costanza et al., 2000; Daily et al., 2000; Giles, 2005; Reid et al., 2005), others argue that such efforts are not only largely futile and flawed (Pearce, 1998; Toman, 1998b; Bockstael et al., 2000; Pagiola et al., 2004), but may also provide society a disservice (Ludwig, 2000; Kremen, 2005). The estimates range from unknown (incomparibility cf. Chang, 1997) or invaluable, since infinite (Toman, 1998b) because of lack of a human substitute, to ~38x10¹² USD/a (updated to 2000 levels Costanza et al., 1997; Balmford et al., 2002; Hassan et al., 2005), which is larger but of similar magnitude than the global gross national product (GNP) of 31 x10¹² USD/a (2000 levels). Such monetary estimates are targeted at policy-makers to assist assessments of the economic benefits of our natural environment (Farber et al., 2006). Some argue (Balmford et al., 2002; Reid et al., 2005; Balmford et al., 2005) unless those values are recognized also in economic terms, ecosystems will continue their decline and the planet’s ecological health is at stake (Millennium Ecosystem Assessment, 2005c). Others argue that ecosystems provide goods and services which are invaluable and need to be conserved on other, fundamental principles, i.e. the precautionary principle (maximin strategy) for not jeopardizing the conditions for a decent, healthy, and secure human existence on this planet (e.g. Costanza et al., 2000; van den Bergh, 2004). What is sometimes lost in the arguments is that natural capital (including ecosystem goods and services) is part of society’s capital assets (Arrow et al., 2004). The question then may be considered as whether one should maximize present value or try to achieve a measure of sustainability. In either case, it is the change in quantities of the capital stock that must be considered (including ecosystem services). One
A approach in considering valuation of ecosystem services is to calculate how much of one type of capital asset would be needed to compensate for the loss of one unit of another type of capital asset (Arrow et al., 2004). What is not disputed is that factoring in the full value of ecosystem goods and services, whether in monetary or non-monetary terms, distorts measures of economic wealth such that a country may be judged to be growing in wealth according to conventional indicators, while it actually becomes poorer due to the loss of natural resources (Balmford et al., 2002; Millennium Ecosystem Assessment, 2005c; Mock, 2005 p.33ff.). Ignoring such aspects also means loosing out on opportunities to gain. For instance Balmford et al. (2002) estimated benefit:cost ratio of at least 100:1 for an effective global conservation program setting aside 15% of current Earth surface if all aspects conventionally ignored are factored in. Finally it is argued that since the poor most directly depend on services from ecosystems, the degradation of these systems will exacerbate poverty, hunger, and disease and obstruct sustainable development (e.g. Mooney et al., 2005; Millennium Ecosystem Assessment, 2005c; Mock, 2005).

4.6 Acclimation and adaptation: practices, options, and constraints

Although climate change is a global issue, local efforts can help greatly in maintaining and enhancing resilience and in limiting the longer-term damage from climate change (e.g. Hughes et al., 2003; Opdam and Wascher, 2004; Singh, 2003). The following section discusses possible adaptation options that could minimize the potential impacts of climate change. Identifying adaptation options is a rapidly developing field, so this list should not be considered exhaustive.

4.6.1 Adaptation options

Adaptation starts with recognition that climate change is occurring and will result in changes in ecosystems (Fig. 4.6). As these changes occur, natural resource management techniques will need to respond in order to increase the resilience of ecosystems or strengthen “ecosystems' ability to respond naturally to climate change” (c.f. Article 2 of the UNFCCC). There are many opportunities to achieve this (Tomkins and Adger, 2003; Cropp and Gabrica, 2002). For effective response, managers must first understand the nature of the climatic and ecological changes that are likely to occur in their region. Monitoring environmental change including climate, and how ecosystems respond is important so that adjustments in management strategies can be made as needed (e.g. Adger et al., 2003; Moldan et al., 2005). Conservation planners and ecosystem managers need, therefore, to be better informed about the potential consequences of climate change. Although, many adaptation options are available to wildlife managers, uncertainty about the magnitude and timing of climate change may discourage some from adopting new management practices. However, management based on ‘no regrets’ or the ‘precautionary principle’ would certainly be prudent. Actions to reduce the impact of other threats could also enhance resilience to climate change (e.g. Opdam and Wascher, 2004). Such pro-active approaches would encourage conservation planning that is both relevant today and in the future. Techniques that allow the management of conservation resources in response to climate variability may ultimately prove to be the most beneficial way of preparing for abrupt climate change by increasing ecosystem resilience (Bharwani et al., 2005).

There are many relatively straightforward options to adapt to the impact of climate change in intensively managed ecosystems (See CH5, agriculture). In less intensively managed ecosystems, decisions have to be made about whether to minimize compositional changes in order to continue the supply of a service or to allow a change in order to supply other ecosystem services. This can be achieved by different management plans, but all require information on the ongoing changes in climate, ecosystems and ecosystem services.
Natural ecosystems or nature reserves (i.e. managed to conserve aspects of biodiversity) are not adaptable. The development of reserve systems, however, can reduce their vulnerability to climate change (McNeely and Schutyser, 2003). Reserve systems should be designed in considering the long-term shifts in plant and animal distributions. Furthermore, reserves should be protected from major disturbances, such as unusual fires and droughts, which increase the likelihood of local extinctions and invasions. Ultimately, adaptation possibilities are determined by the conservation priorities of each reserve. Strategies to cope with climate change should be part of conservation management plans. This is unfortunately rarely the case (Chopra et al., 2005). Adaptation in ecosystems does not occur automatically and it requires adequate resources and financing. This has until now not been widely recognized.

A primary adaptation strategy to climate change is to reduce and manage the other stresses on species and ecosystems (Duraiappah et al., 2005). This may lead to an increase in the resilience of habitats and species to both climate change and climate variability. In addition to removing other stressors it is necessary to maintain healthy, connected and genetically diverse populations. Isolated, small populations are often more prone to local extirpations than larger, more widespread populations (e.g. Gitay et al., 2002; Lovejoy and Hannah, 2005). Connected populations also facilitate the movement of species between them. Although connectivity, genetic diversity and population size are goals managers already strive to accomplish, climate change increases the importance of doing so.

Reducing stress on ecosystems is difficult especially in densely populated regions in Africa, Asia and Europe. Recent studies in southern Africa have signalled the need for policy to focus on managing areas outside protected areas such as subsistence rangelands (Von Maltitz et al., 2007). This can, in part, be achieved through the devolution of resource ownership and management to communities, securing community tenure rights and incentives for resource utilization. This argument is based on the observation that greater species diversity occurs outside protected areas that are more extensive (Scholes et al., 2004). Many species will need 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 contrasts with communal or private land use systems where migration may be encouraged by strategic policies.

Managers can use prescribed fires and other techniques to reduce fuel load and the potential for catastrophic fires. It may also be possible to minimize the effect of severe weather events by for example, securing water rights to maintain water levels through a drought, or by having infrastructure capable of surviving floods. Maintaining viable and widely dispersed populations of individual species also minimizes the probability that localized catastrophic events will cause significant negative effects.
Dispersed populations might also be less vulnerable to wide-spread events such as hurricanes and typhoons.

Climate change is likely to increase opportunities for invasive species because of their adaptability to disturbance (Lake and Leishman, 2004). Captive breeding for reintroduction and translocation is likely to be less successful if climate change is more rapid. 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 large changes in the climate. 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.

A costly 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). Meshing existing species with new species in a given habitat would, however, be difficult and expensive. In many cases the knowledge of ecosystem interactions and species requirements may be lacking. Engineering 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, the use of pesticides may be necessary. 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.

### 4.6.2 Adaptation costs

There are no factual studies that have established the effectiveness and costs of adaptation options in ecosystems. Unfortunately, this makes currently a comprehensive assessment of the avoided damages (i.e. benefits) and costs impossible (see also 4.5). But the costs involved in monitoring, increasing the resilience of conservation networks and adaptive management are probably large. For example, the money spent annually on nature conservation in the Netherlands was recently estimated to be 1 billion Euros (Milieu en, 2005). Of this amount, 285 Million Euros were used to manage national parks and reserves and 280 million Euros were used for new reserve network areas and habitat improvement; the main objective being to reduce fragmentation between threatened populations and to respond to other threats. The reserve network planned for the Netherlands (to be established by 2020) will increase the resilience of species, populations and ecosystems to climate change, but at a high cost. Although not defined explicitly in this way, a significant proportion of these costs can be interpreted as climate adaptation costs.

### 4.6.3 Implications for biodiversity

Many studies and assessments stress the adverse impacts of climate change on biodiversity (e.g. Gitay et al., 2002; Hannah and Lovejoy, 2003; Lovejoy and Hannah, 2005; Thomas et al., 2004a; Thuiller et al., 2005; Schroter et al., 2005; van Vliet and Leemans, 2006), but comprehensive appraisals of adaptation options to deal with declining biodiversity are rare.

The UN Convention on Biological Diversity (CBD, http://www.biodiv.org) aims to conserve biodiversity, to sustainably use biodiversity and its components and to fairly and equitably share
benefits arising from the utilization of biodiversity. This goes much further than most national biodiversity policies. The CBD explicitly recognizes the use of biodiversity, ecosystems and their services and frames this as a developmental issue. As such, it extents beyond UNFCCC’s objective of ‘avoiding dangerous human interference with the climate system at levels where ecosystems cannot adapt naturally, food security is not threatened and economic development can proceed in a sustainable manner’. The main tool proposed by the CBD is the ecosystem approach (EA, Smith and Malthby, 2003) based on integrated response options that intentionally and actively address ecosystem services (including biodiversity) and human well-being simultaneously and involve all stakeholders at different institutional levels. The EA resembles sustainable forest management projects (FAO, 2001). In theory the EA helps the conservation and sustainable use of biodiversity, but applications of the approach have had limited success (Brown et al., 2005a). Integrated responses include, however, learning by doing; a proactive approach that should increase the resilience of ecosystems and biodiversity.

4.6.4 Interactions with other policies and policy implications

An alternative towards designing integrated policies to address climate change and terrestrial systems is to formulate policies that cut across the UN conventions, such as UNFCCC, CBD and Convention to Combat Desertification (CCD, Nnadozie, 1998). Mitigation strategies aimed at combating desertification, for example, would produce a win-win situation. Apart from stopping 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.

4.7 Implications for sustainable development

All people in the world depend on natural and managed ecosystems and the services they provide. Over the past 50 years, humans have converted and modified these ecosystems more rapidly and over larger areas than in any comparable period of human history. These changes have been driven by the rapidly growing demands for food, fresh water, timber, fibre, and fuel and have contributed to substantial net gains in human well-being and economic development (Reid et al., 2005). However, it has also resulted in a substantial and largely irreversible loss of biodiversity and degradation in ecosystems and their services. The important drivers of ecosystem change are unlikely to disappear with climate change and excessive nutrient loading probably becoming more severe. The continued degradation of ecosystems and biodiversity would jeopardise the livelihoods and well-being of millions of people.

Furthermore, the consequences of policies for the vulnerability of ecosystems to climate change at both the national and international level are not yet fully understood. There is growing evidence that significant impacts on the environment may result from perverse or unintended effects of policies from other sectors, which directly or indirectly have adverse consequences on ecosystems and other environmental processes (Chopra et al., 2005). Land re-distribution policies, for example, designed to increase food self-sufficiency contribute to reducing carbon sequestration, and loss of biodiversity through extensive clear-cutting increases vulnerability to land degradation.

Effective mechanisms to analyze cross-sectoral impacts and to feed new scientific knowledge into policy-making are necessary (Schneider, 2004). For instance, understanding climate variability in arid lands could set a precedent for policies that account for delays in the response of ecosystems to management interventions. Knowing that because of their episodic nature unmanaged arid lands do show immediate negative impacts, policies could be formulated to reduce the temptation of land
managers to overstock during "good" periods. The latter is usually very profitable in the short term, but risks degrading the ecosystem in the long term (Ash et al., 2002).

There is a substantial evidence to suggest that developing and implementing policies and strategies to reduce the vulnerability of ecosystems to climate change is closely linked to the availability of capacity to address current needs (e.g. Chanda, 2001). Thus, prospects for successful adaptation to climate change will remain limited as long as factors that contribute to chronic vulnerability to, for example, drought and floods (e.g. population growth, poverty and globalization) are not resolved (Kates, 2000; Reid et al., 2005).

4.7.1 Ecosystems services and sustainable development

There are no simple solutions to the problem of ecosystem degradation and loss of services because of the large differences in natural and socio-economic conditions between regions. Many interactions, lags and feedbacks, including those that operate across a range of spatial, temporal and organizational scales generate complex patterns, which are not fully understood. Past actions to slow or reverse the degradation of ecosystems have yielded significant results, but these improvements have generally not kept pace with growing pressures (Reid et al., 2005).

Sustainable development means meeting the needs of the present without compromising the ability of future generations to meet their needs. The Millennium Development Goals (MDGs), which include poverty reduction, improved health, education, gender equality, sanitation and environmental sustainability, are a major international policy target towards sustainable development. Any progress achieved in addressing the MDGs is unlikely to be sustained if ecosystem services continue to be degraded. Sound management of ecosystem services provides cost-effective opportunities for addressing multiple development goals in a synergistic manner (Reid et al., 2005).

The role of ecosystems in sustainable development and in achieving the MDGs involves an array of stakeholders, from government, the private sector, international agencies, the local community to individuals (Jain, 2003; chapter 4 Adeel et al., 2005). 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 an ecosystem, let alone the situation under future climates. Responding and adapting to the impacts of climate change on ecosystems calls for a clear and structured system of decision making at all levels (Kennett, 2002). Impacts of climate change on ecosystems also show strong interrelationships with ecosystem processes and human activities at various scales over time. Addressing these impacts requires a coordinated, integrated, cross-sectoral policy framework with a long-term focus, a strategy that so far has not been easy to implement (Brown, 2003).

4.7.2 Subsistence livelihoods and indigenous peoples

The impacts of climate change on ecosystems and their services will not be distributed equally around the world. Dryland and Mediterranean regions are likely to be more vulnerable than others (IPCC-TAR) and ecosystem degradation is largest in these regions (Hassan et al., 2005). Climate change is likely to cause additional inequities, as its impacts are unevenly distributed over space and time and disproportionately affect the poor (Tol, 2001). The term double exposure has been used for regions, sectors, ecosystems and social groups that are confronted both by the impacts of climate change and by the consequences of economic globalization (O'Brien and Leichenko, 2000). Thus special attention needs to be given to indigenous peoples with subsistence livelihoods and groups with limited information access and few means of adaptation.
For example, in 1996 the southern African countries developed 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, 1996). However, the implementation of this policy is far from being realized. There is evidence to demonstrate that poverty amongst other factors plays a significant role in degrading ecosystems in developing countries, which also contributes to the failure to meet sustainable development goals (Tole, 1998; Harou, 2002). Unsustainable development and continued degradation affect the poor the most because they depend directly on their immediate ecosystems and have limited resources to invest in conservation or to avoid degradation (Hardy, 2003; Reid et al., 2005). As a result climate change and sustainable development need to incorporate issues of equity (Kates, 2000; Jain, 2003; Richards, 2003).

4.8 Key uncertainties and research priorities

Major sources of uncertainties for this assessment are (i) that projections for precipitation carry a significantly higher uncertainty than temperature, which is also of great relevance for ecosystems in the southern hemisphere; (ii) that within global vegetation models upscaling of ecophysiological processes to the global scale and the role of disturbance regimes are poorly understood, which hampers the assessment of related biotic feedbacks; (iii) that crucial information is drawn from species realized niches only instead of fundamental niches, which is particularly relevant for biodiversity projections; (iv) that the role of invasive alien species in a changing climate for both biodiversity and ecosystem functioning is still poorly understood and may result in major surprises; (v) that abrupt events, such as the slowing of the thermohaline circulation may affect marine ecosystems in unknown ways; (vi) that the only recently considered effect of increasing surface ocean CO2 and declining pH on marine productivity, biodiversity, biogeochemistry and ecosystem functioning is poorly understood, yet future research may confirm the initial findings that marine ecosystems will be impacted in significant ways; (vii) that changes in human use and management of ecosystems will interact with climate change in ways that are currently difficult to predict.

To project the future properties and services of ecosystems following research needs arise:

- Identifying tipping points of **permafrost–soil–vegetation interactions** at strongest warming high latitudes, where terrestrial as well as marine (below 100 m) permafrost and gas hydrates store more than 10,000 PgC (CH4, CO2), which, if released, could result in abrupt and significant climate forcing (e.g. Symon et al., 2005 1015; iLeaps, 2005; Schellnhuber, 2002).
- More robust modeling of interactions between biota and their geophysical environment using several, independently developed **DGVMs**. To expand such research from the stagnation since TAR better funding, in particular of the costly but needed validation (Price et al., 2001) leading beyond model intercomparisons, is required. The goal is to better project the long-term evolution of the carbon sequestration potential of ecosystems including any lagged responses (e.g. Scheffer et al., 2001; iLeaps, 2005).
- More emphasis on precipitation projections (e.g. Handel and Risbey, 1992) and resulting **water regime** effects to be studied in particular in mature forests in the northern hemisphere, seasonal tropical forests, and arid or semi-arid grassland and savanna systems in the southern hemisphere (e.g. Jasienski et al., 1998; Karnosky, 2003).
- Improved understanding of **disturbances**, in particular episodic events such as drought, fire, insect calamities, diseases, floods, and wind-storms including invasions by alien species, as they interact with ecosystem responses to climate change (e.g. Osmond et al., 2004; Opdam and Wascher, 2004).
- Urgent need to expand current under-investment in **large spatial scale, long-term field studies** (May, 1999b; Krauchi et al., 2000; Morgan et al., 2001b; Osmond et al., 2004; Opdam and...
Wascher, 2004; Symon et al., 2005 1019) also to better address scale mismatches between climatology and ecology (Root and Schneider, 1995).

- Urgent promotion of studies on impacts of rising atmospheric CO₂, ocean acidification, and warming on coral reef and other marine systems (Coles and Brown, 2003; Anonymous, 2007)

- Advances in understanding the relationship between biodiversity and ecosystem services at a scale relevant to human well-being or with Sir Robert May's (1999a) words: "The relatively rudimentary state of ecological science prevents us from making reliable predictions about how much biological diversity we can lose before natural systems collapse and deprive us of services upon which we depend".

- Progress towards identification of ecosystem service providers, ecosystem structures that determine functionality, environmental key factors influencing provision of services, and quantitative information on economic impacts (Toman, 1998a; Winnett, 1998; Kremen, 2005; Symon et al., 2005 1018).

- Integrative vulnerability studies on adaptive management responses to preserve biodiversity (including conservation and reservation management), ecosystem function and services in relation to pressures from land-use change and climate change (Kappelle et al., 1999; Symon et al., 2005 1018; Stenseth and Hurrell, 2005; Lorenzoni et al., 2005).
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