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

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Executive Summary

During the course of this century the resilience of many ecosystems (their ability to adapt naturally) is likely to be exceeded by an unprecedented combination of change in climate and in other global change drivers (especially land use change and overexploitation), if greenhouse gas emissions and other changes continue at or above current rates (high confidence). By 2100 ecosystems will be exposed to atmospheric CO_2 levels substantially higher than in the past 650 000 years, and global temperatures at least among the highest as those experienced in the past 740 000 years (very high confidence) [WGI, 6, 4.2, 4.4.10, 4.4.11]. This will alter the structure, reduce biodiversity and perturb functioning of most ecosystems, and compromise the services they currently provide (high confidence) [4.2, 4.4.1, 4.4.2-4.4.9, 4.4.10, 4.4.11, Figure 4.4, Table 4.1]. Present and future land use change and associated landscape fragmentation are very likely to impede species' migration and thus impair natural adaptation via geographic range shifts (very high confidence) [4.1.2, 4.1.3, 4.2, 4.4.5, 4.4.10].

Several major carbon stocks in terrestrial ecosystems are vulnerable to current climate change and/or land-use impacts and at a high degree of risk from projected unmitigated climate and land-use changes (high confidence). Several terrestrial ecosystems individually sequester as much carbon as is currently in the atmosphere (very high confidence) [4.4.1, 4.4.6, 4.4.8, 4.4.10, 4.4.11]. The terrestrial biosphere is likely to become a net source of carbon during the course of this century (medium confidence), possibly earlier than projected by IPCC's Third Assessment Report (low confidence) [4.1, Figure 4.2]. Methane emissions from tundra frozen loess ("yedoma", comprising ~500 Pg C) and permafrost (comprising ~400 Pg C) have accelerated in the past two decades, and are likely to accelerate further (high confidence) [4.4.6]. At current anthropogenic emission rates the ongoing positive trends in the terrestrial carbon sink will peak before mid-century, then begin diminishing, even without accounting for tropical deforestation trends and biosphere feedback, tending strongly towards a net carbon source before 2100 (high confidence) [Figure 4.2, 4.4.1, 4.4.10, Figure 4.3, 4.4.11], while the buffering capacity of the oceans will begin to saturate [WGI, *e.g.* 7.3.5.4]. While some impacts may include primary productivity gains with low levels of climate change (<~2ºC mean global change above pre-industrial levels), synergistic interactions are likely to be detrimental, *e.g.* increased risk of irreversible extinctions (very high confidence) [4.4.1, Figure 4.2, 4.4.10, Figure 4.3, 4.4.11].

Approximately one fifth to one third of species assessed so far (in an unbiased sample) are likely to be at increased risk of extinction if global mean temperatures exceed a warming of 2 to 3°C above pre-industrial levels (medium confidence) [4.4.10, 4.4.11, Figure 4.4, Table 4.1]. Projected impacts on biodiversity are significant and of key relevance, since global losses in biodiversity are irreversible (very high confidence) [4.4.10, 4.4.11, Figure 4.4, Table 4.1]. Endemic species richness is highest where regional palaeoclimatic changes have been muted, providing circumstantial evidence of their vulnerability to projected climate change (medium confidence) [4.2.1]. With global average temperature changes of 2°C above pre-industrial levels many terrestrial, freshwater, and marine species (particularly endemics across the globe) are at a far greater risk of extinction than in the geological past (medium confidence) [4.4.5, 4.4.11, Figure 4.4, Table 4.1]. Globally ~20% to ~30% of species (global uncertainty range from 10% to 40%, but varying among regional biota from as low as 1% to as high as 80%) will be at increasingly high risk of extinction by 2100 [4.2, 4.4.10, 4.4.11, Figure 4.4, Table 4.1]. Current conservation practices are generally poorly prepared to adapt to this level of change, and effective adaptation responses are likely to be costly to implement (high confidence) [4.4.11, Table 4.1, 4.6.1].

Substantial changes in structure and functioning of terrestrial ecosystems are very likely to occur with a global warming of > 2 to 3°C above pre-industrial levels (high confidence). Between

~26% (WGI B1 scenario; 1.7ºC warming) and ~37% (WGI A2 scenario, 3.7ºC warming) of extant ecosystems will reveal appreciable changes by 2100, with some positive impacts especially in Africa and the southern Hemisphere arid regions, but extensive forest and woodland decline in mid to high latitudes and in the tropics, associated especially with changing disturbance regimes (especially through wildfire and insects) [4.4.2, 4.4.3, 4.4.5, 4.4.10, 4.4.11, Figure 4.3].

Substantial changes in structure and functioning of marine and other aquatic ecosystems are very likely to occur with a mean global warming of > 2 to 3°C above pre-industrial levels and the associated increased atmospheric CO_2 levels (high confidence). Climate change (very high confidence) and ocean acidification (medium confidence) will impair a wide range of planktonic and shallow benthic marine organisms that use aragonite to make their shells or skeletons, such as corals [Box 4.4] and marine snails (Pteropods), with significant impacts particularly in the Southern Ocean, where cold water corals are likely to show large reductions in geographic range this century [4.4.9, B4.4]. Substantial loss of sea ice will reduce habitat for dependant species (*e.g.* Polar bears) (very high confidence [4.4.9, Box 4.4, 6.1.1.1, 11.6, Box 4.3, 5.4, 6.1]. Terrestrial tropical and sub-tropical aquatic systems are at significant risk under at least WGI A2 scenarios; negative impacts across ~25% of Africa by 2100 (especially southern and western Africa) will cause a decline in both water quality and ecosystem goods and services (high confidence) [4.4.2, 4.4.7].

Ecosystems and species are very likely to show a wide range of vulnerabilities to climate change, depending on imminence of exposure to ecosystem-specific, critical thresholds (very high confidence). Most vulnerable ecosystems include coral reefs, the sea ice biome and other high latitude ecosystems (*e.g.* boreal forests), mountain ecosystems and mediterranean-climate ecosystems (high confidence) [Figure 4.4, Table 4.1, 4.4.9, Box 4.4, 4.4.5, 4.4.6, Box 4.3, 4.4.7, 4.4.4, 4.4.10, 4.4.11]. Least vulnerable ecosystems include savannas and species–poor deserts, but this assessment is especially subject to uncertainty relating to the CO_2 fertilization effect and disturbance regimes such as fire (low confidence) [Box 4.1, 4.4.1, 4.4.2, Box 4.2, 4.4.3, 4.4.10, 4.4.11].

4.1 Introduction

An ecosystem can be practically defined as a dynamic complex of plant, animal, and micro-organism communities and the nonliving environment, interacting as a functional unit (Millennium Ecosystem Assessment, Reid *et al.*, 2005). Ecosystems may be usefully identified through having strong interactions between components within their boundaries and weak interactions across boundaries (Reid *et al.*, 2005, part 2). Ecosystems are well recognised as critical in supporting human well-being (Reid *et al.*, 2005), and the importance of their preservation under anthropogenic climate change is explicitly highlighted in Article 2 (the objective) of the United Nations Framework Convention of Climate Change (UNFCCC).

In this chapter the focus is on the properties, goods and services of non-intensively managed and unmanaged ecosystems and their components (as grouped by widely accepted functional and structural classifications, Figure 4.1), and their potential vulnerability to climate change as based on scenarios mainly from IPCC (IPCC, 2007, Working Group I – WGI; this volume, 2). Certain ecosystem goods and services are treated in detail in other sectoral chapters (this volume): 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 (this volume). Region-specific aspects of ecosystems are discussed in chapters 9 to 16 (this volume). This chapter is based on work published since the Third Assessment Report of the IPCC (TAR Gitay *et al.*, 2001). We do not summarize TAR findings here, but refer back to relevant TAR results where appropriate in the chapter to indicate confirmation or revision of major findings.

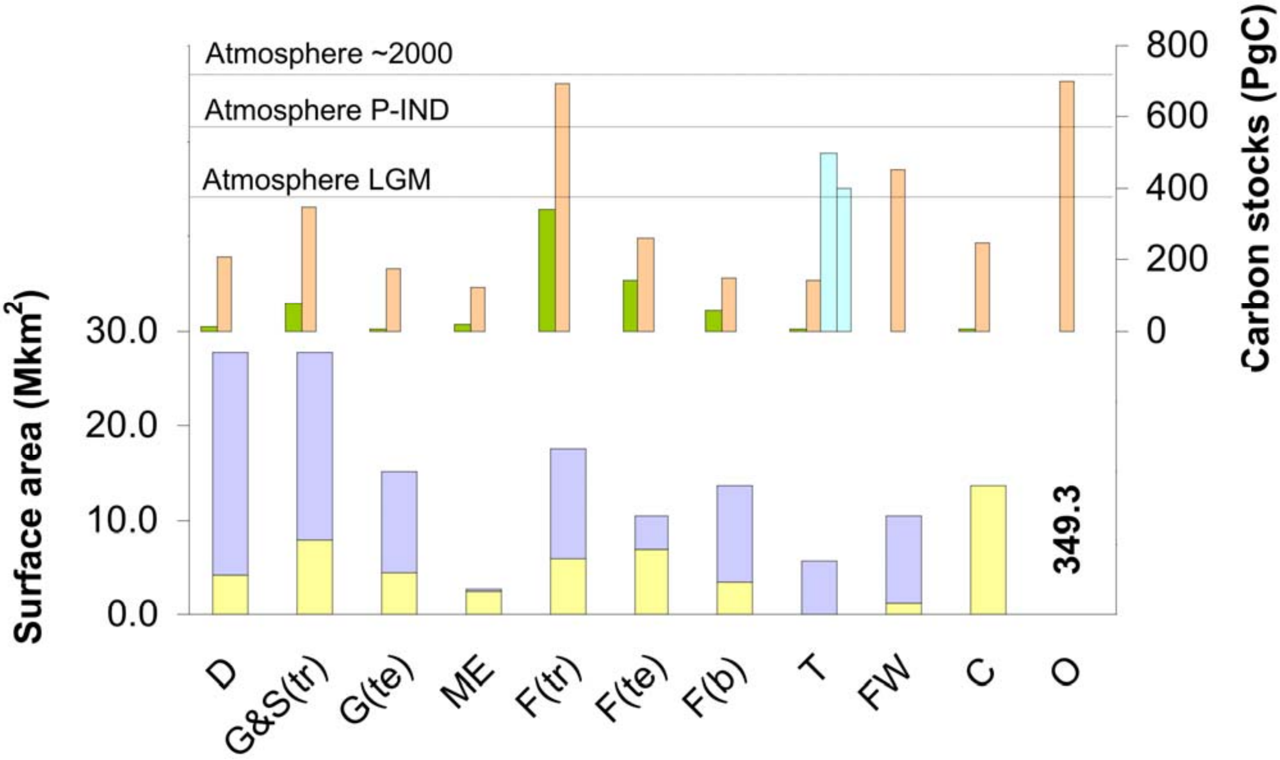

Figure 4.1: *Major ecosystems addressed in this report, with their global areal extent (lower panel, Mkm^2, transformed by land use in yellow, untransformed in blue, from Hassan et al. 2005, except for Mediterranean climate ecosystems, where transformation impact is from Myers et al (2000), and total carbon stores (upper panel, PgC) in plant biomass (green), soil (brown), yedoma/permafrost (light blue). D=Deserts, G&S(tr)=Tropical Grasslands and Savannas, G(te)=Temperate grasslands, ME=Mediterranean Ecosystems, F(tr)=Tropical Forests, F(te)=Temperate Forests, F(b)=Boreal forests, T=Tundra, Fw=Freshwater lakes and wetlands, C=Croplands, O=Oceans. Data are from Table 2.2, p23, Sabine (2004), except for carbon content of yedoma permafrost and permafrost (light blue columns, left and right, respectively, Zimov et al., 2006), ocean organic carbon content (dissolved plus particulate organic WGI, 7.3.4.1), and ocean surface area from Hassan et al. (2005, Summary, Table C2, p. 15, inserted as a number). Figures here update the TAR (WGI, 3), especially through considering soil C to 3m depth (Jobbagy and Jackson, 2000), as opposed to 1m. Approximate carbon content of the atmosphere (PgC) is indicated by the dotted lines for last glacial maximum (LGM), pre-industrial (P-IND) and current (~2000).*

Projecting the impacts of climate change on ecosystems is complicated by an uneven understanding of the interlinked temporal and spatial scales of ecosystem responses. Processes at large spatial scales, *i.e.* the biosphere at the global scale, are generally characterised by slow response times on the order of centuries, and even up to millennia (WGI, 6). However it is also important to note that some large scale responses in the palaeorecord (WGI, 6) and to current climate anomalies such as El Niño events may emerge at much shorter timescales (Holmgren *et al.*, 2001; Sarmiento and Gruber, 2002; Stenseth *et al.*, 2002; van der Werf *et al.*, 2004). At continental scales, biomes (see glossary) respond at decadal to millennial time scales (*e.g.* Davis, 1989; Prentice *et al.*, 1991; Lischke *et al.*, 2002; Neilson *et al.*, 2005), and groups of organisms forming ecological communities at the regional scale have shorter response times of years to centuries. Responses of populations (*i.e.* interbreeding individuals of the same species), occur at intermediate temporal scales of months to centuries, and underpin changes in biodiversity. These include changes at the genetic level that may be adaptive, as demonstrated for example for trees (Jump *et al.*, 2006) and corals (Coles and Brown, 2003). Fast physiological response times (*i.e.* seconds, hours, days, months) of micro-organisms, plants, and animals operate at small scales from a leaf or organ to the cellular level; They underlie organism responses to environmental conditions, 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.

The spatial distribution of ecosystems at biome scale has traditionally been explained only in terms of climate control (Schimper, 1903), but it is increasingly apparent that disturbance regimes such as fire or insects may strongly influence vegetation structure somewhat independently of climate (*e.g.* Andrew and Hughes, 2005; Bond *et al.*, 2005). Biomes are differentially sensitive to climatic change (*e.g.* Kirschbaum and Fischlin, 1996; Sala *et al.*, 2000; Gitay *et al.*, 2001), with temperature-limited biomes prone to impacts of warming, and water-limited biomes prone to increasing levels of drought. Some, such as fire dependent biomes, may be in a meta-stable state that can change rapidly under climate and other environmental changes (Scheffer *et al.*, 2001; Sankaran *et al.*, 2005). Marine biome responses, too, have been shown at decadal scales (Beaugrand *et al.*, 2002) with more rapid regime shifts within decades (Edwards *et al.*, 2002; Richardson and Schoeman, 2004; Edwards *et al.*, 2006). Biomes therefore provide a useful level of ecological organization at which to summarize climate change impacts, being of large enough extent to conduct a global synthesis, yet having a response time relevant to anthropogenic climate change.

4.1.1 Ecosystem goods and services

Ecosystems provide many goods and services that are of vital importance for the functioning of the biosphere, and provide the basis for the delivery of tangible benefits to human society. Hassan *et al.* (2005) define these to include supporting, provisioning, regulating, and cultural services. In this chapter we discuss mainly: (i) supporting services, such as primary and secondary production, and biodiversity, a resource that is increasingly recognised to sustain many of the goods and services humans enjoy from ecosystems. These provide a basis for three higher level services, namely (ii) provisioning services, such as products (cf. Gitay *et al.*, 2001) *i.e.* food (including game, roots, seeds, nuts and other fruit, spices; fodder), fibre (including wood, textile) and medicinal and cosmetic products (including aromatic plants, pigments); (ii) regulating services, which are of paramount importance for human society such as a) carbon sequestration; b) climate and water regulation; c) protection from natural hazards such as floods, avalanches, or rock fall; d) water and air purification, e) disease and pest regulation; (iii) cultural services, which satisfy human spiritual and aesthetic appreciation of ecosystems and their components.

4.1.2 Key issues

Based on new findings for ecosystems since the TAR, we highlight here five overarching key issues pertinent to assessing the vulnerability of ecosystems to anthropogenic climate change, and related adaptation responses.

Firstly, 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), as they have done repeatedly with palaeoclimatic changes (WGI, 6). A primary key issue, however, is whether ecosystem resilience (understood as the disturbance an ecosystem can tolerate before it shifts into a different state (see glossary) and *e.g.* Scheffer *et al.*, 2001; Cropp and Gabrica, 2002; Folke *et al.*, 2004) inferred from these responses (*e.g.* Harrison and Prentice, 2003) will be sufficient to tolerate future anthropogenic climate change (*e.g.* Chapin *et al.*, 2004; Jump and Peñuelas, 2005). The implications of possibly transient increases in productivity for resilience are also very relevant. These may occur in certain terrestrial ecosystems through likely atmospheric CO_2 fertilization effects and/or modest warming (*e.g.* Baker *et al.*, 2004; Lewis *et al.*, 2004b; Malhi and Phillips, 2004), and demonstrated consequences of increased radiation due to reduced tropical cloudiness (Nemani *et al.*, 2003). Ecosystem resilience thus seems usefully equivalent to the critical ecosystem property highlighted in Article 2 of the UNFCCC, *i.e.* an "ability to adapt naturally".

Secondly, ecosystems are increasingly subjected to other human induced pressures, such as extractive use of goods, and increasing fragmentation and degradation of natural habitats (*e.g.* Bush *et al.*, 2004). In the medium term (*i.e.* decades) especially, climate change will increasingly exacerbate these human-induced pressures, causing a progressive decline in biodiversity (Lovejoy and Hannah, 2005). However, this is likely to be a complex relationship that may also include some region-specific reductions in land-use pressures on ecosystems (*e.g.* Goklany, 2005; Rounsevell *et al.*, 2006).

A third key issue involves exceeding critical thresholds and triggering non-linear responses in the biosphere that could lead via positive feedback to novel states that are poorly understood. Projected future climate change and other human-induced pressures are virtually certain to be unprecedented compared with the past several hundred millennia (*e.g.* Petit *et al.*, 1999; Augustin *et al.*, 2004; Siegenthaler *et al.*, 2005; IPCC, 2007, WGI, 2).

Fourthly, the understanding of time lags in ecosystem responses is still developing, including, for example, of broad scale biospheric responses or shifting species geographic ranges. Many ecosystems may take several centuries (vegetation) or even possibly millennia (where soil formation is involved) before responses to a changed climate are played out (*e.g.* Lischke *et al.*, 2002). A better understanding of transient responses and the functioning of ecosystems under continuously changing conditions is needed to narrow uncertainties about critical effects and to develop effective adaptation responses at the time scale of interest to human society.

A fifth key issue relates to species extinctions, and especially global extinction as distinct from local extinctions, since the latter represents irreversible change. This is crucial especially because of a very likely link between biodiversity and ecosystem functioning in the maintenance of ecosystem services (Duraiappah *et al.*, 2005; Hooper *et al.*, 2005; Diaz *et al.*, 2006; Worm *et al.*, 2006), and thus extinctions are virtually certain to reduce societal options for adaptation responses.

4.2 Current sensitivities

4.2.1 Climatic variability and extremes

The biosphere has been exposed to large variability and extremes of CO_2 and climate throughout geological history (WGI, 6, Augustin *et al.*, 2004; Siegenthaler *et al.*, 2005) – this provides some insight into the current sensitivities of ecosystems even though it is not possible to match past climate analogues precisely with future warming, due to differences in forcing factors (Overpeck *et al.*, 2006), dominant ecosystems, and species (*e.g.* Velichko *et al.*, 2002). What can be learned is that, firstly, significant biological changes including species extinctions have accompanied large climate perturbations of the past (*e.g.* Overpeck *et al.*, 2005). Secondly, endemic biodiversity is concentrated in regions that have experienced lower variability during the Pleistocene (from ~2 Mya Jansson, 2003), during which glacial and interglacial conditions have alternated for roughly the past two million years. Thirdly, range shifts have been a major species response (Lovejoy and Hannah, 2005), although genetic and physiological responses (Davis and Shaw, 2001) have also occurred, which can be broadly defined as "natural adaptation" at species level, and by aggregation, at the ecosystem level.

While earlier IPCC reports described several ecosystems to be resilient to warming up to 1°C (*e.g.* Kirschbaum and Fischlin, 1996), recent studies provide a more differentiated view of ecosystem sensitivity (*e.g.* Walther *et al.*, 2002) that includes understanding of the role of climatic variability and extremes. Knowledge about climate variability and natural ecosystems has improved with better understanding of the behaviour of decadal-scale climatic oscillations and their impacts, including ENSO (El Niño/Southern Oscillation) and the NAO (North Atlantic Oscillation) (WGI, 3.6). These low-frequency phenomena indirectly determine vegetation responses, notably through shifts in major controls (temperature, precipitation, snow cover). For example, the European Alps show changes in regional climates that can partly be attributed to NAO variability (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 more frequent fire (*e.g.*, Diaz and Markgraf, 1992). Similar effects have been reported for NAO (Edwards and Richardson, 2004; Sims *et al.*, 2004; Balzter *et al.*, 2005). Sea surface 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 climate 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 plays an important role in forest dynamics, driving pulses of tree mortality in the Argentinean Andes (Villalba and Veblen, 1997), North American woodlands (Breshears and Allen, 2002; Breshears *et al.*, 2005), and in the eastern Mediterranean (Körner *et al.*, 2005b). In both the Canadian Rockies (Luckman, 1994) and European Alps (Bugmann and Pfister, 2000) extreme cold through a period of cold summers in 1696–1701 caused extensive tree mortality. Heat waves such as the recent 2003 event in Europe (Beniston, 2004; Schär *et al.*, 2004; Box 4.1) have both short-term and long-term implications for vegetation, particularly if accompanied by drought conditions.

Hurricanes can cause widespread mortality of wild organisms, 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 the background rate (Anonymous, 2001). Loss of habitat due to hurricanes can also lead to greater conflict with humans. For example, fruit bats (*Pteropus spp.*) declined recently on American Samoa due to a combination of direct mortality events and increased hunting pressure (Craig *et al.*, 1994). Greater storminess and higher return of extreme events will also alter disturbance regimes in coastal ecosystems leading to changes in diversity and hence ecosystem functioning. Saltmarshes, mangroves and coral reefs are likely to be particularly vulnerable (Bertness and Ewanchuk, 2002).

Assessment of the impacts of climate variability, their trends, and development of early warning systems has been strongly advanced since the TAR by satellite-based remote sensing efforts. Notable contributions have included insights into phenological shifts in response to warming (*e.g.* Badeck *et al.*, 2004) and other environmental trends (*e.g.* Nemani *et al.*, 2003), complex Sahelian vegetation changes (*e.g.* Prince *et al.*, 1998; Rasmussen *et al.*, 2001; Anyamba and Tucker, 2005; Hein and Ridder, 2006), wildfire impacts (*e.g.* Isaev *et al.*, 2002; Barbosa *et al.*, 2003; Hicke *et al.*, 2003; Kasischke *et al.*, 2003), coral bleaching events (*e.g.* Yamano and Tamura, 2004), cryosphere changes

(Walsh, 1995; Prentice *et al.*, 2007), ecotone (see glossary) responses to climate (*e.g.* Masek, 2001), deforestation (*e.g.* Asner *et al.*, 2005), and even feedback to regional climate (*e.g.* Durieux *et al.*, 2003), the impacts of extreme climate events (*e.g.* Gobron *et al.*, 2005; Lobo and Maisongrande, 2006), and even in monitoring soil water (Wagner *et al.*, 2003).

Box 4.1: Ecological impacts of the European heat wave 2003.

Anomalous hot and dry conditions affected Europe between June and mid-August, 2003 (Fink *et al.*, 2004; Luterbacher *et al.*, 2004; Schär *et al.*, 2004). Since similarly warm summers may occur at least every second year by 2080 in an SRES A2 world, for example (Schär *et al.*, 2004; Beniston, 2004), effects on ecosystems observed in 2003 provide a conservative analogue of future impacts. The major effects of the 2003 heat wave on vegetation and ecosystems appears to have been through heat and drought stress, and wildfires.

Drought stress impacts on vegetation (Gobron *et al.*, 2005; Lobo and Maisongrande, 2006) reduced gross primary production (GPP) in Europe by 30% overall, resulting in a net carbon source of 0.5 PgC yr^{-1} (Ciais *et al.*, 2005). However, vegetation responses to the heat varied on environmental gradients such as altitude (Jolly *et al.*, 2005). Some vegetation types as monitored by remote sensing were found to recover to a normal state by 2004 (*e.g.* Gobron *et al.*, 2005), but enhanced crown damage of dominant forest trees in 2004, for example, indicate complex delayed impacts (Fischer, 2005). Freshwater ecosystems experienced prolonged depletion of oxygen in deeper layers of lakes during the heat wave (Jankowski *et al.*, 2006), and there was a significant decline and subsequent poor recovery in species richness of molluscs in the River Saône (Mouthon and Daufresne, 2006). Taken together, this suggests quite variable resilience across ecosystems of different types, with very likely progressive impairment of ecosystem composition and function if such events increase in frequency (*e.g.* Lloret *et al.*, 2004; Rebetez and Dobbertin, 2004; Jolly *et al.*, 2005; Fuhrer *et al.*, 2006).

High temperatures and greater dry spell durations increase vegetation flammability (*e.g.* Burgan *et al.*, 1997), and during the 2003 heat wave a record-breaking incidence of spatially extensive wildfires was observed in European countries (Barbosa *et al.*, 2003), with roughly 650 000 ha of forest burned across the continent (De Bono *et al.*, 2004). Fire extent (area burned), though not fire incidence, was exceptional in Europe in 2003, as found for the extraordinary 2000 fire season in the USA (Brown and Hall, 2001), and noted as an increasing trend in the USA since the 1980s (Westerling *et al.*, 2006). In Portugal, area burned was more than twice the previous extreme (1998) and four times the 1980-2004 average (Trigo *et al.*, 2005; Trigo *et al.*, 2006). Over 5% of the total forest area of Portugal burned, with an economic impact exceeding 1 billion Euros (De Bono *et al.*, 2004).

Long term impacts of more frequent similar events are very likely to cause changes in biome type, particularly by promoting highly flammable, shrubby vegetation that burns more frequently than less flammable vegetation types such as forests (Nunes *et al.*, 2005), and as seen in the tendency of burned woodlands to reburn at shorter intervals (Vazquez and Moreno, 2001; Salvador *et al.*, 2005). The conversion of vegetation structure in this way on a large enough scale may even cause accelerated climate change through losses of carbon from biospheric stocks (Cox *et al.*, 2000). Future projections for Europe suggest significant reductions in species richness even under mean climate change conditions (Thuiller *et al.*, 2005b), but an increased frequency of such extremes (as indicated *e.g.* by Schär *et al.*, 2004) is likely to exacerbate overall biodiversity losses (Thuiller *et al.*, 2005b).

4.2.2 Other ecosystem change drivers

Ecosystems are sensitive not only to changes in climate and atmospheric trace gas concentrations but also to other anthropogenic changes such as land use, nitrogen deposition, pollution and invasive species (Vitousek *et al.*, 1997; Mack *et al.*, 2000; Sala *et al.*, 2000; Hansen *et al.*, 2001; Körner, 2003a; Lelieveld *et al.*, 2002; Lambin *et al.*, 2003; Reid *et al.*, 2005). In the recent past, these pressures have significantly increased due to human activity (Gitay *et al.*, 2001). Natural disturbance regimes (*e.g.* wildfire and insect outbreaks) are also important climate-sensitive drivers of ecosystem change. Projecting the impacts of the synergistic effects of these drivers presents a major challenge, due to the potential for non-linear, rapid, threshold-type responses in ecological systems (Burkett *et al.*, 2005).

Land use change represents the anthropogenic replacement of one land cover type by another, *e.g.* forest to cultivated land (or the reverse), 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). Land use change and related habitat loss and fragmentation have long been recognized as important drivers of past and present ecosystem change, particularly of biodiversity (Fahrig, 2003; Heywood and Watson, 1995).

Fire influences community structure by favouring species that tolerate fire or even enhance fire spread, resulting in a relationship between the relative flammability of a species and its relative abundance in a particular community (Bond and Keeley, 2005). As a result many vegetation types are far from the maximum biomass predicted by regional climate alone (Bond *et al.*, 2005). Geographic shifts in key species or fire may therefore cause fundamental community shifts (Brooks *et al.*, 2004; Schumacher and Bugmann, 2006). 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 (Bond *et al.*, 2005), and the boreal region (Harden *et al.*, 2000) in particular. Intensified wildfire regimes driven at least partly by 20th century climate change (Gillett *et al.*, 2004; Westerling *et al.*, 2006) appear to be changing vegetation structure and composition with shifts from *Picea* to *Pinus*-dominated communities and 75-95% reductions in tree densities observed in forest-tundra transition in Eastern Canada (Lavoie and Sirois, 1998). By contrast, in Quebec, fire frequency appears to have dropped during the 20th century (Bergeron *et al.*, 2001), a trend projected to continue (Bergeron *et al.*, 2004; 4.4.5). Across the entire North American boreal region, however, total burned area from fires increased by a factor of 2.5 between the 1960s and 1990s, while the area burned from human-ignited fires remained constant (Kasischke and Turetsky, 2006). In Southeast Asia, by contrast, human activities have significantly altered fire regimes in ways that may be detrimental to the affected ecosystems (Murdiyarso and Lebel, 2007).

Drought facilitated the spread of human-caused fire in tropical regions during the 97/98 El Niño (Randerson *et al.*, 2005), affecting atmospheric trace gas concentrations such as CO, CH_4, and H_2 (Kasischke *et al.*, 2005; Langenfelds *et al.*, 2002; Novelli *et al.*, 2003) and CO_2 emissions (van der Werf *et al.*, 2004) at hemispheric and global scales. Drought conditions increase Amazon forest flammability (Nepstad *et al.*, 2004). Tropical forest fires are becoming more common (Cochrane, 2003), and have strong negative effects on Amazonian vegetation (Cochrane and Laurance, 2002; Haugaasen *et al.*, 2003), possibly even intensifying rainfall events (Andreae *et al.*, 2004, but see 4.4.1 and 4.4.5 on forest productivity trends).

Significant progress on globally applicable models of fire has been made since the TAR (Thonicke *et al.*, 2001). Modelling suggests increases in wildfire impacts (4.4.1, 4.4.5) during the 21st century under a wide range of scenarios (*e.g.* Scholze *et al.*, 2006). The implications of the regional and global importance of fire are manifold (Bond *et al.*, 2005). Firstly, fire suppression strategies often have limited impact (Schoennagel *et al.*, 2004; Keeley, 2002; Van Wilgen *et al.*, 2004), and enhancement of vegetation flammability through more prevalent fire weather (Brown *et al.*, 2004) and resulting big wildfires threatens human settlements, infrastructure, and livelihoods (*e.g.* Allen Consulting Group, 2005). Secondly, in some ecosystems, including islands, human-caused 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; 4.4.3, 4.4.4, 4.4.5).

The spatial impact of insect damage is significant and exceeds that of fire in some ecosystems, but especially in boreal forests (Logan *et al.*, 2003). Spruce bud worm, for example, defoliated over 20 times the area burned in eastern Ontario from between 1941-1996 (Fleming *et al.*, 2002). Furthermore, fires tended to occur 3-9 years after a SBW outbreak (Fleming *et al.*, 2002), suggesting greater interaction between these disturbances with further warming. Disturbance by forest tent caterpillar has also increased in western Canada in the past 25 years (Timoney, 2003). In the Mediterranean region, defoliation of Scots Pine shows a significant association with previous warm winters, implying that future climatic warming may intensify insect damage (Hodar and Zamora, 2004; 4.4.5).

Invasive alien species (IAS, Chornesky and Randall, 2003) represent a major threat to endemic or native biodiversity in terrestrial and aquatic systems (Sala *et al.*, 2000; Occhipinti-Ambrogi and Savini, 2003; Scavia *et al.*, 2002). Causes of biological invasions are multiple and complex (Dukes and Mooney, 1999), yet some simple models have been developed (Crawley, 1989; Chytry *et al.*, 2005; Deutschewitz *et al.*, 2003; Facon *et al.*, 2006). Alien species invasions also interact with other drivers, sometimes resulting in some unexpected outcomes (Chapuis *et al.*, 2004). Change in biotic and/or abiotic disturbance regimes are recognised a primary drivers of IAS (Le Maitre *et al.*, 2004), with communities often becoming more susceptible to invasion following extreme events (Smith and Knapp, 1999), such as are projected under future climate change. IAS can also change disturbance regimes through increasing vegetation flammability (Brooks *et al.*, 2004). Overall, ongoing shifts in human mediated disturbances, insect pests, IAS and fire regimes are very likely to be important in altering regional ecosystem structure, diversity and function (*e.g.* Timoney, 2003).

4.3 Assumptions about future trends

The work reviewed in this chapter is dependent on assumptions of various types that are important in assessing the level of confidence that can be associated with its results (Moss and Schneider, 2000), but can be challenging to quantify and aggregate. Assumptions and uncertainties associated with climate scenarios (WGI, 8) are not considered here, other than to identify the greenhouse gas emission trends or socio-economic development pathways (*e.g.* SRES Nakicenovic *et al.*, 2000) assumed in the literature we review (see also Table 4.1, especially scaling methodology and associated uncertainties). Since the TAR many global or regional scenarios have become available to quantify future impacts (Christensen *et al.*, 2002; WGI, 10, 11), and confidence in future climate projections has increased recently (WGI, 8) and (Nakicenovic *et al.*, 2000). However, many assumptions must be made, due to imperfect knowledge, in order to project ecosystem responses to climate scenarios. We provide here a brief outline and guide to the literature of those that are most relevant.

To project impacts of climate change on ecosystems there are basically three approaches: (i) correlative, (ii) mechanistic, and (iii) analogue approaches. For the correlative (i) and mechanistic (ii) approach, studies and insights from the present give rise to the assumption that the same relationships will hold in the future. Three modelling approaches in particular have provided relevant results since the TAR: Firstly, correlative models use knowledge of the spatial distribution of species to derive functions (Guisan and Thuiller, 2005) or algorithms (Pearson *et al.*, 2004) that relate the probability of their occurrence to climatic and other factors (Guisan and Zimmermann, 2000). Criticised for assumptions of equilibrium between species and current climate, inability to account for species interactions, lack of physiological mechanism and inability to account for population processes and migration (see Pearson and Dawson, 2003; Pearson, 2006), these methods have nonetheless proved capable of simulating known species range shifts in the distant (Martinez-Meyer *et al.*, 2004) and recent past (Araújo *et al.*, 2005), and provide a pragmatic first-cut assessment of risk to species decline and extinction (Thomas *et al.*, 2004a). Secondly, mechanistic models include the modelling of terrestrial ecosystem structure and function. They are based on current understanding of energy, biomass, carbon, nutrient and water relations, and their interacting dynamics with and among species like primary producers. Such approaches generate projections of future vegetation structure, *e.g.* as the likely balance of plant functional types (PFTs) after permitting competitive interaction and accounting for wildfire (Woodward and Lomas, 2004b; Lucht *et al.*, 2006; Prentice *et al.*, 2007; but see Betts and Shugart, 2005; for more complete discussion). Extrapolated to global scale, these are termed Dynamic Global Vegetation Models (DGVMs, see glossary). An equivalent approach for oceans is lacking (but see Field *et al.*, 1998). Thirdly, Earth system models have begun to incorporate more realistic and dynamic vegetation components, which quantify positive and negative biotic feedbacks by coupling a dynamic biosphere to atmospheric circulations with a focus on the global carbon cycle (Friedlingstein *et al.*, 2003; Cox *et al.*, 2004; Cox *et al.*, 2006; Friedlingstein *et al.*, 2006).

Ecosystem and species-based models are typically applied at scales much finer than are resolved or reliably represented in global climate models. The requisite downscaling techniques of various types (dynamic, empirical) have matured and are increasingly used to provide the needed spatio-temporal detail (IPCC-TGCIA, 1999; Mearns *et al.*, 2003; Wilby *et al.*, 2004; WGI, 11). Physically consistent bioclimatic scenarios can now be derived for almost any region, including developing countries (*e.g.* Jones *et al.*, 2005) and complex, mountainous terrain (*e.g.* Gyalistras and Fischlin, 1999; Hayhoe *et al.*, 2004). However, major uncertainties relating to downscaling remain in the impact projections presented in this chapter, centring mainly on soil water balance and weather extremes which are key to many ecosystem impacts, yet suffer from low confidence in scenarios for precipitation and climate variability, despite recent improvements (WGI, 8).

Despite the recognized importance of multiple drivers of ecosystem change, they are rarely included in current global climate models (Hansen *et al.*, 2001; Levy *et al.*, 2004; Zebisch *et al.*, 2004; Feddema *et al.*, 2005; Holman *et al.*, 2005b; Pielke Sr, 2005). The explicit inclusion of non-climatic drivers and their associated interactions in analyses of future climate change impacts could lead to unexpected outcomes (Hansen *et al.*, 2001; Sala, 2005). Consequently, many impact studies of climate change that ignore land use and other global change trends may represent inadequate estimates of projected ecosystem responses.

4.4 Key future impacts and vulnerabilities

The scope of this section satisfies that required by IPCC plenary in relation to future impacts on properties, goods and services of major ecosystems and on biodiversity. However, to assess ecosystem goods and services more completely, issues relating to biogeochemical cycling and other supporting or regulating services are also deemed appropriate for consideration under this heading. Following reviews of impacts on individual ecosystems, impacts that cut across ecosystems (such as large scale vegetation shifts and migratory species) are elaborated. Finally the overall implications for biodiversity are highlighted in a global synthesis. Within relevant subsections, we describe briefly ecosystem properties, goods and services, we summarize key vulnerabilities as identified by the TAR, and then review what new information is available on impacts focusing on supporting and regulating services (for provisioning services see this volume, 3, 5, 6).

4.4.1 Biogeochemical cycles and biotic feedback

The cycling of chemical elements and compounds sustains 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 the TAR in understanding the interactive responses of terrestrial 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 (Buchmann, 2002; Cox *et al.*, 2006; Friedlingstein *et al.*, 2006; Gedney *et al.*, 2006). Interactions between ocean and atmosphere and land and oceans are also critical for the future evolution of climate (4.4.9, but mainly WGI, *e.g.* 7.3.5.4).

Among the most advanced tools to achieve scaling-up of terrestrial systems 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; Schröter *et al.*, 2005). Testing at hierarchical levels from leaf to biome and over relevant time scales have shown encouraging agreement with observations (Lucht *et al.*, 2002; Bachelet *et al.*, 2003; Harrison and Prentice, 2003; Gerten *et al.*, 2004; Joos and Prentice, 2004; Kohler *et al.*, 2005; Peylin *et al.*, 2005), and validation is ongoing (*e.g.* Woodward and Lomas, 2004b; Prentice *et al.*, 2007). Recently, full coupling between DGVMs and climate models has progressed from earlier work (*e.g.* Woodward and Lomas, 2001) to explore feedback effects between biosphere and atmospheric processes (Cox *et al.*, 2006; Friedlingstein *et al.*, 2006), that were initially reported as having significant implications for the carbon cycle (Cox *et al.*, 2000).

Key vulnerabilities: 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 surpassed. Transitions between states may be triggered, or the ecosystem may even “collapse” *i.e.* show a rapid transition to a much less productive and/or species-poor assemblage with lower biomass and other impairments such as degrading soils (*e.g.* Scheffer *et al.*, 2001; Rietkerk *et al.*, 2004; Schröder *et al.*, 2005). Changing fire regimes provide an important example (see 4.2.2 for a more complete treatment), as these are of significant concern for the terrestrial carbon balance (Schimel and Baker, 2002; van der Werf *et al.*, 2004; Westerling *et al.*, 2006), especially because they can be self-reinforcing (Bond and Keeley, 2005). However, even less extreme responses of ecosystems are likely to have important ramifications for the biosphere because of their spatial extent.

Based on early versions of DGVMs (equilibrium biogeography models or global biogeochemical models Neilson *et al.*, 1998), the world’s terrestrial ecosystems were projected to continue as a net carbon sink for a number of decades and possibly throughout the 21st century, with an initially ‘greening’ world due to longer growing seasons, more precipitation, and CO_2 fertilization benefits. Substantial structural changes in biomes were projected towards 2100 with ecosystem shifts toward higher latitudes and altitudes. A reversal of initial carbon sequestration gains were projected during the 21st century, as CO_2-fertilization benefits approach saturation and temperature effects on respiration and transpiration increase, potentially resulting in net global ecosystem carbon losses relative to today (*e.g.* Cramer *et al.*, 2001). With feedback from the global carbon cycle to the atmosphere accounted for, dieback of much of the Amazon rainforest due to desiccation was an identified major vulnerability, but with a high degree of uncertainty (Cox *et al.*, 2000). The TAR concluded the net global terrestrial carbon exchange would be between -6.7 PgC yr^{-1} (uptake 1 PgC) and +0.4 PgC yr^{-1}, and that anthropogenic CO_2 emissions would remain the dominant determinant of atmospheric CO_2 concentration during the 21st century. Key ecosystem forecasting needs identified in the TAR were for spatially and temporally dynamic models to simulate processes that produce inertia and lags in ecosystem responses. Progress on this issue has now allowed initial assessments of the potential for feedbacks from ecosystems to atmospheric composition and climate change.

Impacts: Observations for global NPP from 1982-1999 show an increase of 6%, concentrated in the tropics and due virtually certainly to greater solar radiation with reduced cloud cover (Nemani *et al.*, 2003), broadly concurring with the projection in the TAR of an increasing biospheric sink in the initial stages of climate change. Scaled-up effects of direct atmospheric CO_2 enrichment on plant and ecosystem biomass accumulation (CO_2 fertilization) are largely responsible for the projected continued enhancement of NPP in current global models (Leemans *et al.*, 2002). By contrast, impacts in oceans, especially through acidification, have been largely negative (see 4.4.9).

Despite improved experiments, the magnitude of the terrestrial CO_2 fertilization effect remains uncertain, though improved simulation of major vegetation types (particularly forests and savannas) at the last glacial maximum by incorporating CO_2 effects (Harrison and Prentice, 2003) are encouraging. The three main constraints that have been observed to limit the fertilization effect are element stoichiometry (nutrient balance), forest tree dynamics, and secondary effects of CO_2 on water relations and biodiversity. Trends in some 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 CO_2, which range from 0 to 40% biomass gain per season, mainly reflect CO_2-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 the realistic atmospheric feedback of the future negate this benefit. The only replicated test of multiple CO_2 x climate/environment interactions (water, temperature, nutrient supply) yielded no overall CO_2 biomass signal in a grassland system (Shaw *et al.*, 2002) highlighting significant influence of co-limiting environmental variables.

Similar trends are emerging for forests, although the interpretation is complicated by time lags in biomass response to the artifactual step-change when initiating CO_2 treatments, requiring longer observation periods before a new steady state (*e.g.* in terms of leaf area index, fine root dynamics and nutrient cycling) is reached. Three tall forest test systems, loblolly pine plantation (Oren *et al.*, 2001; Schäfer *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 that diminish with time except for one of the four pairs of test plots (treatment vs. control) in the joint Duke pine experiments (Schäfer *et al.*, 2003). A European boreal forest system also showed smaller CO_2 growth stimulation in mature trees under field conditions than expected from results for saplings (Rasmussen *et al.*, 2002). It is uncertain if test systems with regrowing trees (*e.g.* Karnosky, 2003) provide valid analogies for biomass responses in mature forests, which are at a steady state nutrient cycle. Nonetheless, a recent analysis (Norby *et al.*, 2005) suggests that the NPP response of forests to elevated CO_2 is relatively predictable across a broad range of sites, with a stimulation of 23 +/- 2% at a median CO_2 of double the pre-industrial level. The logarithmic biotic growth factor derived from this is 0.60 (β-factor, expressing the response as a function of the relative CO_2 increase).

It has been suggested that greatest CO_2 fertilization impacts may be seen in savanna systems post-fire (Bond and Midgley, 2000; Bond *et al.*, 2003), especially where nutrients are less limiting and in systems in which trees require carbon reserves to re-establish after fire (see section 4.2.2). Scrub oak in Florida shows diminishing CO_2 responses as treatment proceeds (Hungate *et al.*, 2006), even though this is a post-fire regenerating system. For tropical forests, the planet's single largest biomass carbon reservoir, post-industrial atmospheric CO_2 enrichment seems to have enhanced growth dynamics (Phillips *et al.*, 2002; Laurance *et al.*, 2004; Wright *et al.*, 2004). A more dynamic forest might ultimately store less rather than more carbon in future if long-term species compositional changes are realised (Laurance *et al.*, 2004; Malhi *et al.*, 2006), especially given the exceptional CO_2 responsiveness of tropical lianas that may increase tree mortalities and population turnover (Körner, 2004).

Based on experimental data, best estimates of instantaneous CO_2-induced water savings due to reduced stomatal aperture range from 5 to 15 % (Wullschleger and Norby, 2001; Cech *et al.*, 2003) for humid conditions, diminishing with drying soils. Desert shrub systems increase production in elevated CO_2 only during exceptional wet periods and not in dry periods (Nowak *et al.*, 2004), contrasting with earlier expectations (Morgan *et al.*, 2004). Evapotranspiration data for temperate zone ecosystems under future CO_2 scenarios suggests that this may be reduced by less than 10% across all weather conditions. Water savings through elevated CO_2 hold limited benefits for trees during drought, because nutrient availability in drying top soil becomes interrupted, and initial water savings are exhausted (Leuzinger *et al.*, 2005). Repeated drought with high temperatures (*e.g.* Europe in 2003, Box 4.1) may reduce landscape-wide carbon stocks (Ciais *et al.*, 2005). Data for intact ecosystems reveal a hydrological response up to the global scale of increasing run-off (*e.g.* Gedney *et al.*, 2006) that is consistent with stomatal responses to rising CO_2 (*e.g.* Hetherington and Woodward, 2003; Gedney *et al.*, 2006).

Soil nitrogen availability is key to predict future carbon sequestration by terrestrial ecosystems (Reich *et al.*, 2006), especially in light of global N-deposition trends (2-10 fold increase in some industrialized areas, Matson *et al.*, 2002). The future ability of ecosystems overall to sequester additional carbon is very likely to be constrained by levels of nitrogen availability and fixation, and other key nutrients such as phosphorous that may also become increasingly limiting (Hungate *et al.*, 2003). Carbon accumulation and sequestration in critical soil stocks (see Figure 4.1) has been found to be strongly nitrogen-constrained, both because levels well above typical atmospheric inputs are needed to stimulate soil C-sequestration, and because natural N_2-fixation appears to be particularly strongly limited by key nutrients (van Groenigen *et al.*, 2006).

Results from a loblolly pine forest (Lichter *et al.*, 2005) and grassland experiments (Van Kessel *et al.*, 2000) suggest a reduced likelihood for CO_2-fertilization driven carbon accumulation in soils, probably because carbon sequestration to humus is more nutrient demanding (not only nitrogen), than is wood formation, for example (Hungate *et al.*, 2006). Carbon accretion in soil is therefore itself likely to exert negative feedback on plant growth by immobilizing soil nutrients (in addition to cation depletion by acidic precipitation), contributing to a faster diminishing of the biospheric sink (Figure 4.2, Reich *et al.*, 2006) than implemented in model projections (Figure 4.2, *e.g.* Scholze *et al.*, 2006).

Accumulation of seasonally transitory soil C-pools such as in fine roots has been found at elevated ambient CO_2 concentrations, but the general validity of such enhanced C-fluxes and what fraction of these might be sequestered to recalcitrant (see glossary) soil carbon stocks remains unresolved (Norby *et al.*, 2004). Soil warming may enhance carbon emissions, especially by reducing labile soil organic carbon pools (Davidson and Janssens, 2006). This results in the commonly observed short term (<decadal) loss of carbon in warming experiments, followed by the re-establishment of a new equilibrium between inputs and losses of soil C (*e.g.* Eliasson *et al.*, 2005; Knorr *et al.*, 2005). Recent observations indeed show widespread carbon losses from soils (Bellamy *et al.*, 2005; Schulze and

Freibauer, 2005) that are consistent with this formulation. However, in regions with thawing permafrost, a decay of historically accumulated soil carbon stocks (yedoma, >10 000 years old, Figure 4.1) due to warming (Zimov *et al.*, 2006) and nutrient deposition (Blodau, 2002; Mack *et al.*, 2004) could release large amounts of carbon to the atmosphere (see also 4.4.6). Increased NPP (but see Angert *et al*., 2005) and vegetation change (see 4.4.5 and *e.g.* Sturm *et al.*, 2001) may partly counterbalance this carbon release (see 4.4.6, and Sitch *et al.*, 2007), thus complicating projections (Blodau, 2002; for a full discussion see 4.4.6).

Ecosystem changes associated with land use and land cover change (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 (Chapin *et al.*, 2005b; Feddema *et al.*, 2005), by releasing CO_2 via losses of biomass and soil carbon (Gitz and Ciais, 2003; Canadell *et al.*, 2004; Levy *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 (Lal, 2003; Jones and Donnelly, 2004; King *et al.*, 2004a; Lal, 2004; Wang *et al.*, 2004a; de Koning *et al.*, 2005; WGIII, 9), but are probably limited to reducing the ultimate atmospheric CO_2 increase by 2100 by between 40-70 ppm (House *et al.*, 2002), and by ~century-long time lags until mature forests are established (4.4.5, 4.4.6, 4.4.10), and offset by regional warming effects of lower albedo with poleward boreal forest expansion (*e.g.* Betts, 2000; for a full discussion see 4.4.6).

The sequestration and cycling of carbon in terrestrial ecosystems is a key vulnerability, given the above drivers, their generally global extent, their potential irreversibility and likely existence of threshold-type impacts. The extent to which the recently discovered methane release from plant foliage (Keppler *et al.*, 2006) can be scaled to biome level is under debate (Houweling *et al.*, 2006), and highlights the currently limited understanding of the methane cycle, and its exclusion from Earth system models (*e.g.* Betts and Shugart, 2005). Nonetheless, recent work especially 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 (*e.g.* Friedlingstein *et al.*, 2006; Lucht *et al.*, 2006; Scholze *et al.*, 2006).

Global estimates (IS92a, HACM2-SUL, Cramer *et al.*, 2001) suggest a reduced global sink relative to that expected under CO_2 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 Biome Productivity (NBP) of tropical and southern hemisphere ecosystems. According to these models, the rate of NBP increase slows by around 2030 as CO_2 fertilization itself saturates, and in four of six models shows further, climate-induced NBP declines, due to increased heterotrophic respiration and declining tropical NPP after 2050. These trends are projected to continue until mid-century, even with stabilized atmospheric CO_2 concentration and instantaneously stabilized climate beyond 2100 (Woodward and Lomas, 2004b; see also next paragraph, Figure 4.2). 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 PgC, adding between 29 and 129 ppm to global atmospheric CO_2 by 2100, mainly due to deforestation (Cramer *et al.*, 2004).

Climate scenario uncertainty provides a substantial variance in global terrestrial C balance by 2100, even under a single CO_2 emission scenario (IS92a, projected to reach 703 ppm atmospheric CO_2 concentration by 2100, excluding vegetation feedback). Using five GCMs to drive DGVMs, global terrestrial C-sequestration is estimated at between -106 to +201 PgC (Schaphoff *et al.*, 2006), though in four out of five, the sink service decreased 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, and that wildfire frequency increases dramatically even for a warming of <2ºC by 2100 (Scholze *et al.*, 2006). Here we show model results for the most recent version of the DGVM Lund-Potsdam-Jena-Model (LPJ) (Schaphoff *et al.*, 2006) highlighting changes in biome structure (relative cover of dominant growth forms) 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 early as 2030 (Figure 4.2), earlier than suggested in the TAR (WGI, Figure 3.10), and substantial shifts in biome structure (Figure 4.3), discussed more fully in 4.4.10 and 4.4.11.

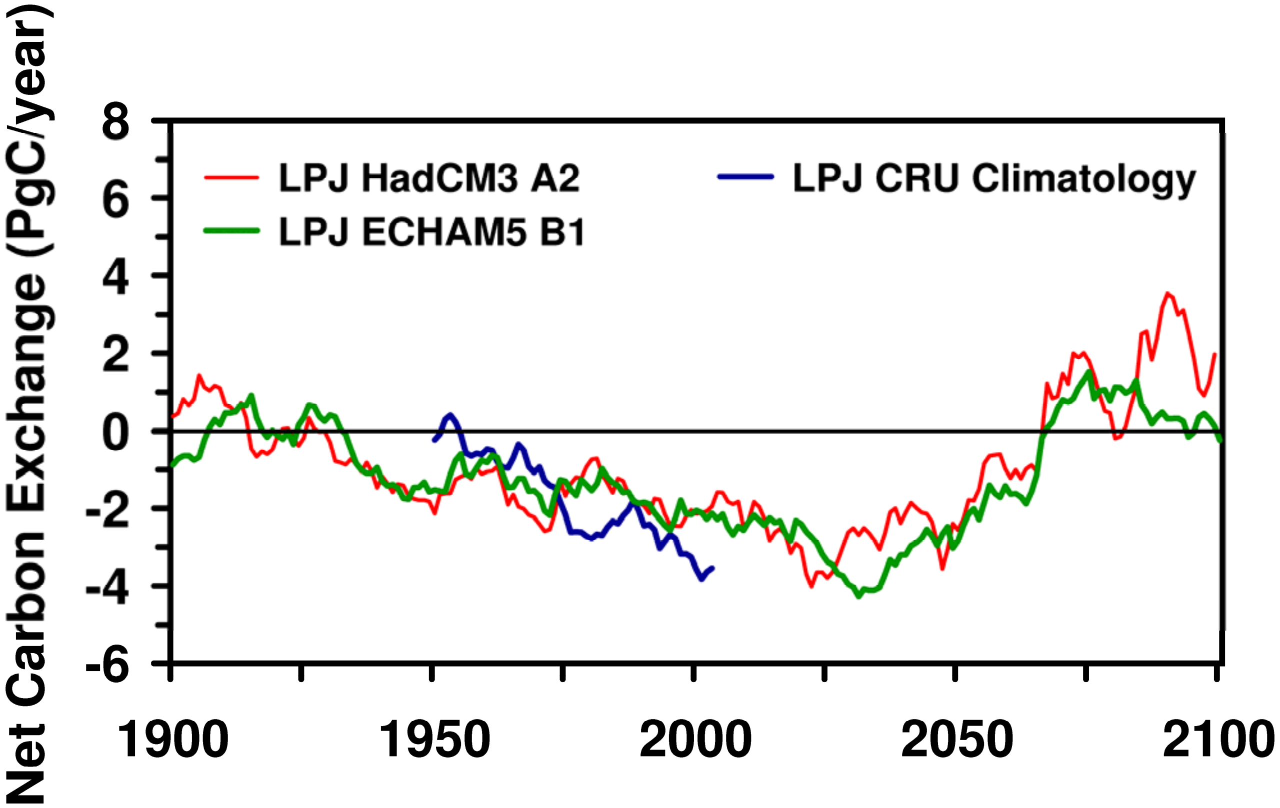

Figure 4.2: *Net carbon exchange of all terrestrial ecosystems as simulated by the dynamic global vegetation model (DGVM) LPJ (Gerten et al., 2004; Sitch et al., 2005 - negative values mean a carbon sink, positive values carbon losses to the atmosphere). Past century data are based on observations and 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. Lucht et al., 2006; Schaphoff et al., 2006). In contrast to previous global projections (TAR WGI, Figure 3.10) the world's ecosystems sink service saturates earlier (~2030) and the terrestrial biosphere tends to become a carbon source earlier (~2070) 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; Lucht et al., 2006; Schaphoff et al., 2006; Scholze et al., 2006, - see Figure 4.3 for maps on underlying ecosystem changes). Note these projections assume an effective* CO_2 *fertilization (4.4.1).*

Projections from modelling that dynamically links the physical climate system and vegetation, using Ocean-Atmosphere-General Circulation Models (OAGCMs, *e.g.* Cox *et al.*, 2000), suggest a terrestrial C source that will exacerbate both climate and further vegetation change to at least 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 CO_2 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. This is especially so for boreal regions, due to unrealistically high projections of in-migration rates of trees and shrubs (Neilson *et al.*, 2005).

Changes in air-sea fluxes of dimethyl sulphide (DMS, from –15% to 30%) caused by global warming of ~2°C is projected to have a regional radiative and related climatic impact (Bopp *et al.*, 2004; Bopp *et al.*, 2003), as DMS is a significant source of cloud condensation nuclei. DMS is produced by coccolithophores, which are sensitive to high seawater CO_2 (Riebesell *et al.*, 2000). As the largest producer of calcite on the planet (Holligan *et al.*, 1993) reduced calcification by these organisms may also influence the global carbon cycle (Raven *et al.*, 2005) and global albedo (Tyrrell *et al.*, 1999). N_2O of marine origin contributes ~33% of total input to the atmosphere (Enhalt and Prather, 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 (IPCC, 2001), will impact the ocean nitrogen cycles, especially the processes of nitrification and denitrification which promote N_2O production.

4.4.2 Deserts

Properties, goods and services: One of the largest terrestrial biomes, deserts cover 27.7 Mkm^2, comprising extra-polar regions with mean annual precipitation <250mm and an unfavourable precipitation to potential evaporation ratio (Nicholson, 2002; Warner, 2004; Reid *et al.*, 2005). Deserts support on the order of 10 people per km^2, in sparse populations with among the lowest GDP of all ecosystems (Reid *et al.*, 2005). Recent estimates suggest between 10-20% of deserts and drylands are degraded due to an imbalance between demand for and supply of ecosystem services (Adeel *et al.*, 2005). Critical provisioning goods and services include wild food sources, forage and rangeland grazing, fuel, building materials, and water for humans and livestock, for irrigation, and for sanitation, and genetic resources especially of arid-adapted species (Adeel *et al.*, 2005; Hassan *et al.*, 2005). Regulating services include air quality, atmosphere composition and climate regulation (Hassan *et al.*, 2005), especially through wind-blown dust and desert albedo influences on regional rainfall and biogeochemistry of remote terrestrial and marine ecosystems (Warner, 2004).

Key vulnerabilities: TAR noted several vulnerabilities in drylands (Gitay *et al.*, 2001, p.239) but chiefly that human over-use and land degradation, exacerbated by an overall lack of infrastructure and investment in resource management, would very likely overwhelm climate change impacts, with the exception of impacts of increased dry and wet extremes due to ENSO frequency increase, and negative impacts of projected warming and drying in high biodiversity regions. On the other hand, evidence for region-specific increases in productivity and even community compositional change due to rising atmospheric CO_2 was reported, with associated increased biomass and soil organic matter. Overall impacts of elevated CO_2 were reported as comparable, though usually opposite in sign, to climate change projections. Since TAR, further work shows that desert biodiversity is likely to be vulnerable to climate change (Reid *et al.*, 2005); with winter-rainfall desert vegetation, and plant and animal species especially vulnerable to drier and warmer conditions (Lenihan *et al.*, 2003; Malcolm *et al.*, 2006; Musil *et al.*, 2005; Simmons *et al.*, 2004), and continental deserts vulnerable to desiccation and even soil mobilisation, especially with human land-use pressures (Thomas and

Leason, 2005). However, the potentially positive impact of rising atmospheric CO_2 remains a significant uncertainty, especially because it is likely to increase plant productivity, particularly of C_3 plants (Thuiller *et al.*, 2006b), and together with rainfall change, could even induce wildfires (Bachelet *et al.*, 2001; Hardy, 2003; Duraiappah *et al.*, 2005). The uncertain impact of elevated CO_2 on vegetation productivity and biogeochemical cycling in deserts is an important source of contrasting projections of impacts and vulnerability for different desert regions and vegetation types. Climate change and direct human land-use pressure are likely to have synergistic impacts on desert ecosystems and species that may be offset at least partly by vegetation productivity and carbon sequestration gains due to rising atmospheric CO_2. The net effect of these trends is very likely to be region-specific.

Impacts: Deserts are likely to experience more episodic climate events and inter-annual variability may increase in future, though there is substantial disagreement among GCM projections and across different regions (Smith *et al.*, 2000; Duraiappah *et al.*, 2005). Continental deserts could experience more severe, persistent droughts (Lioubimtseva and Adams, 2004; Schwinning and Sala, 2004). Vulnerability to desertification will be enhanced due to the indicated increase in incidents of severe drought globally (Burke *et al.*, 2006). In the Americas, temperate deserts are projected to expand substantially under doubled CO_2 climate scenarios (Lauenroth *et al.*, 2004). However, dry spell duration and warming trends on vegetation productivity may at least partly be offset by rising atmospheric CO_2 effects on plants (Bachelet *et al.*, 2001; Thuiller *et al.*, 2006b), leading to sometimes contrasting projections for deserts that are based on different modelling techniques that either incorporate or ignore CO_2 fertilization effects.

Elevated CO_2 has been projected to have significant potential impacts on plant growth and productivity in drylands (Lioubimtseva and Adams, 2004). This projection has been confirmed for cool desert shrub species (Hamerlynck *et al.*, 2002), and both desert shrubs and invasive (but not indigenous) grasses in wet years only (Smith *et al.*, 2000). On the whole, evidence for CO_2 fertilization effects in deserts is conflicted, and species-specific (Lioubimtseva and Adams, 2004; Morgan *et al.*, 2004). In the southwest USA the total area covered by deserts may decline by up to 60% if CO_2 fertilization effects are realised (Bachelet *et al.*, 2001). There are limited direct impacts of atmospheric CO_2 on N-fixing in soil biological crusts (Billings *et al.*, 2003), but soil microbial activity beneath shrubs was increased, thus reducing plant-available nitrogen (Billings *et al.*, 2002).

Soil vulnerability to climate change is indicated by shallow desert substrates with high soluble salts and the slow recolonisation of disturbed soil surfaces by different algae components (Evans and Belnap, 1999; Johansen, 2001; Duraiappah *et al.*, 2005). Very low biomass (a drop below a 14% cover threshold) is very likely to make the Kalahari desert dune system in southern Africa 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. About one third of the Sahel was projected to aridify with warming of 1.5 to 2°C by ~2050, with and a general equatorward shift of vegetation zones (Box 4.2). Alternative climate scenarios show less pronounced changes (van den Born *et al.*, 2004).

Box 4.2: Vegetation response to rainfall variability in the Sahel.

The Sahel falls roughly between the 100–200 mm/year (northern boundary) and 400–600 mm/year rainfall isohyets (southern boundary), and supports dry savanna vegetation forming transition zones with the Sahara and humid tropical savanna (Nicholson, 2000; Hiernaux and Turner, 2002; Anyamba and Tucker, 2005). These transition zones have historically fluctuated in response to rainfall changes (Hiernaux and Turner, 2002), in the clearest example of multi-decadal variability measured during the past century (Hulme, 2001). Ecosystem responses to past rainfall variability in the Sahel are potentially useful as an analogue of future climate change impacts, in the light of projections that extreme drought-affected terrestrial areas will increase from 1% to ~30 % globally by the 2090s (Burke *et al.*, 2006).

During the mid-Holocene, conditions supporting mesic vegetation and abundant wildlife deteriorated rapidly (ECF, 2004; Foley *et al.*, 2003), highlighting Sahel sensitivity to forcing effects. The Sahel has shown the largest negative trends in annual rainfall observed globally in the past century, though these reversed somewhat after the late 1970s (WGI, 3). Since ~1900, multi-decadal scale rainfall variability persisted, with drying trends between ~1930–50 and 1960–1985 (Hulme, 2001; Nicholson, 2001). Conditions apparently improved between 1950 and 1960, with limited evidence suggesting increased human and livestock numbers (Reij *et al.*, 2005). Severe drought prevailed in the early 1980s (WGI, 3, Hulme, 2001), and ground-water levels declined, species-specific woody plant mortality increased (mainly of smaller plants), and even dominant perennial C_4 grasses with high water-use efficiency declined. Exposed soil caused increased atmospheric dust loads (Nicholson, 2000; Nicholson, 2001). These events stimulated the concept of desertification and subsequent debates on its causes (Herrmann and Hutchinson, 2005).

The persistence of drought during the latter 20th century prompted suggestions that land cover change had exerted a positive feedback to reinforce drought conditions, but the modelled vegetation change necessary to induce this effect does not reflect reality (Hulme, 2001). During relatively wet periods (Nicholson *et al.*, 2000; Anyamba and Tucker, 2005, WGI, 3) spatially variable regeneration in both the herbaceous and the woody layer have been observed (Gonzalez, 2001; Rasmussen *et al.*, 2001; Hiernaux and Turner, 2002). Remote sensing shows resilience of Sahelian vegetation to drought, with no directional change in either desert transition zone position or vegetation cover (Nicholson and Tucker, 1998 drought). Sahel green-up between the years 1982 and 1998 (Hickler *et al.*, 2005; Prince *et al.*, 1998) and 1994–2003 (Anyamba and Tucker, 2005) has been noted, but this interpretation has recently been challenged (Hein and Ridder, 2006).

Drivers of Sahel vegetation change remain uncertain (Hutchinson *et al.*, 2005) especially because the correlation between rainfall and NDVI appear weak, signalling that greening cannot be fully explained by increasing rainfall (Olsson *et al.*, 2005), and greening may not comprise a return to the initial species composition, cover and surface soil conditions (Warren, 2005). Inconclusive interpretations of vegetation dynamics in the Sahel may reflect complex combined effects of human land use and climate variability on arid environments (Rasmussen *et al.*, 2001). It is far from clear how the interactive effect of climate change, land use activities, and rising CO_2 will influence the Sahel in future. Green-up, or desert amelioration (Figure 4.3, vegetation class 4) due to rising CO_2 and enhanced water-use efficiency (as observed by Herrmann *et al.*, 2005) may accrue only in wet years (Morgan *et al.*, 2004).

Episodic wet periods may increase vulnerability to invasive alien species and subsequent fire outbreaks and this, combined with land over-use, will increase vulnerability to degradation and 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). Wet spells with elevated humidity and warmer temperatures will increase the prevalence of plant diseases (Harvell *et al.*, 2002).

Desert biodiversity is likely to be vulnerable to climate change (Reid *et al.*, 2005), especially in so-called "biodiversity hotspots (Myers *et al.*, 2000). In the Succulent Karoo of South Africa, 2800 plant species face potential extinction as bioclimatically suitable habitat is reduced by 80% with a global warming of 1.5-2.7°C above pre-industrial levels (see Table 4.1). Daytime *in situ* warming experiments suggest high vulnerability of endemic succulent (see glossary) growth forms of the Succulent Karoo to high-end warming scenarios for 2100 (mean 5.5ºC above current ambient temperatures), inducing appreciable mortality in some succulent species tested (but not all) within only a few months (Musil *et al.*, 2005). Desert species that depend on rainfall events to initiate breeding, such as resident birds, and migratory birds whose routes cross deserts, will be severely affected (Dukes and Mooney, 1999; Hardy, 2003; Box 4.5). The Mountain Wheatear in South Africa was projected to lose 51% of its bioclimatic range by 2050 under an SRES A2 scenario (Simmons *et al.*, 2004). In contrast, desert reptile species could be favoured by warming, but depending on rainfall scenarios (Currie, 2001).

4.4.3 Grasslands and savannas

Properties, goods and services: Dominated by a spatially and temporally variable mix of grass and tree growth forms (Sankaran *et al.*, 2005), these include tropical C_4 grasslands and savannas (C_4 grass-dominated with 10-50% tree cover, ~28 Mkm^2) and temperate C_4 and/or C_3-grass and herb-dominated grasslands (15 million km^2, Bonan, 2002). Generally rich in grazing, browsing and other fauna (especially but not only in Africa), these systems are strongly controlled by fire (Bond *et al.*, 2005) and/or grazing regimes (Fuhlendorf *et al.*, 2001; Scholes and Archer, 1997). Disturbance regimes are often managed (*e.g.* Sankaran, 2005), though fire regimes depend also on seasonality of ignition events and rainfall dependent accumulation of flammable material (Brown *et al.*, 2005b). Temperate and tropical systems provide somewhat distinct goods and services. Temperate grasslands contain a substantial soil carbon pool, are important for maintaining soil stability, and provide fodder for wild and domestic animals. Tropical savanna systems possess significant wild faunal diversity that supports nature-based tourism revenue (both extractive and non-extractive) and subsistence livelihoods (food, medicinals and construction material), in addition to cultural, regulating and supporting services.

Key vulnerabilities: The structure, productivity and carbon balance of these systems appear more sensitive than indicated in the TAR to variability of and changes in major climate change drivers. The direct CO_2 fertilization impact and warming effect of rising atmospheric CO_2 have contrasting effects on their dominant functional types (trees and C_3 grasses may benefit from rising CO_2 but not from warming; C_4 grasses may benefit from warming, but not from CO_2 fertilization), with uncertain, non-linear and rapid changes in ecosystem structure and carbon stocks likely. Carbon stocks are very likely to be strongly reduced under more frequent disturbance, especially by fire, and disturbance and drought impacts on cover may exert regional feedback effects. On balance, savannas and grasslands are likely to show reduced carbon sequestration due to enhanced soil respiratory losses through warming, fire regime changes and increased rainfall variability, but possible regional gains in woody cover through direct CO_2 fertilization, and increased plant carbon stocks, cannot be excluded. Scientific predictive skill is currently limited by very few field-based, multi-factorial experiments, especially in tropical systems. Projected range shifts of mammal species will be limited by fragmented habitats and human pressures, as suggested in the TAR, with declines in species richness likely, especially in protected areas. Because of the important control by disturbance, management options exist to develop adaptive strategies for carbon sequestration and species conservation goals.

Impacts: Ecosystem function and species composition of grasslands and savanna are likely to respond mainly to precipitation change and warming in temperate systems, but in tropical systems, CO_2 fertilization and emergent responses of herbivory and fire regime will also exert strong control. Very few experimental approaches have assessed ecosystem responses to multi-factorial treatments such as listed above (Norby and Luo, 2004), and experiments on warming, rainfall change or atmospheric CO_2 level are virtually absent in savannas, with many ecosystem studies confined mainly to temperate grasslands (Rustad *et al.*, 2001).

Rainfall change and variability is very likely to affect vegetation in tropical grassland and savanna systems, with, for example, a reduction in cover and productivity simulated along an aridity gradient in southern African savanna in response to the observed drying trend of ~8 mm yr^{-1} since 1970 (Woodward and Lomas, 2004a). Sahelian woody plants, for example, have shown drought induced mass mortality and subsequent regeneration during wetter periods (Hiernaux and Turner, 2002). Large-scale changes in savanna vegetation cover may also feed back to regional rainfall patterns. Modelled removal of savannas from global vegetation cover has larger effects on global precipitation than for any other biome (Snyder *et al.*, 2004), and in four out of five savannas studied globally, modelled savanna-grassland conversion resulted in 10% lower rainfall, suggesting positive feedback between human impacts and changing climate (Hoffmann and Jackson, 2000). At continental scale, modelled forest-savanna conversion reduced rainfall in tropical African regions, but increased it in central southern Africa (Semazzi and Song, 2001).

Changing amounts and variability of rainfall may also strongly control temperate grassland responses to future climate change (Novick *et al.*, 2004; Zha *et al.*, 2005). A Canadian grassland fixed roughly five times 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). Similarly, Mongolian steppe grassland switched from carbon sink to source in response to seasonal water stress, though carbon balance was neutral on an annual basis (Li *et al.*, 2005). Non-linear responses to increasing rainfall variability may be expected, as ecosystem models of mixed C_3/C_4 grasslands show initially positive NPP relationships with increasing rainfall variability, but greater variability ultimately reduces both NPP and ecosystem stability even if rainfall total is kept constant (Mitchell and Csillag, 2001). Empirical results for C_4 grasslands confirm a similar monotonic (hump-backed) relationship between NPP and rainfall variability (Nippert *et al.*, 2006). 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).

The CO_2 fertilization and warming effect of rising atmospheric CO_2 have generally opposite effects on savanna and grassland dominant functional types, with CO_2 fertilization favouring woody C_3 plants (Ainsworth and Long, 2005), and warming favouring C_4 herbaceous types (Epstein *et al.*, 2002). Simulated heat wave events increased C_4 dominance in a mixed C_3/C_4 New Zealand grassland within a single growing season, but reduced productivity by over 60% where C_4 plants were absent (White *et al.*, 2000b). Some African savanna trees are sensitive to seasonal high air temperature extremes (Chidumayo, 2001). North American forest vegetation types could spread with up to 4ºC warming; but with greater warming, forest cover could be reduced by savanna expansion of up to 50%, partly due to the impacts of fire (Bachelet *et al.*, 2001).

Elevated CO_2 has important effects on production and soil water balance in most grassland types, mediated strongly by reduced stomatal conductance and resulting increases in soil water (Leakey *et al.*, 2006) in many grassland types (Nelson *et al.*, 2004; Niklaus and Körner, 2004; Stock *et al.*, 2005). In short-grass prairie, elevated CO_2 and 2.6ºC warming increased production by 26-47% , regardless of grass photosynthetic type (Morgan *et al.*, 2001a). In C_4 tropical grassland, no relative increase in herbaceous C_3 success occurred in double-ambient CO_2 (Stock *et al.*, 2005). Regional climate modelling indicates that CO_2 fertilization effects on grasslands may scale up to affect regional climate (Eastman *et al.*, 2001).

Differential effects of rising atmospheric CO_2 on woody relative to herbaceous growth forms are very likely (Bond and Midgley, 2000). Trees and shrubs show higher CO_2 responsiveness than do herbaceous forms (Ainsworth and Long, 2005). Savannas may thus be shifting towards greater tree dominance as atmospheric CO_2 rises, with diminishing grass suppression of faster growing tree saplings (Bond *et al.*, 2003). Simulations suggest that rising CO_2 may favour C_3 forms at the expense of African C_4 grasses (Thuiller *et al.*, 2006b), even under projected warming. Continuing atmospheric CO_2 rise could increase resilience of Sahelian systems to drought (Wang and Eltahir, 2002). However, without definitive tests of the CO_2 fertilization effect on savanna trees, other factors can be invoked to explain widely observed woody plant encroachment in grassland systems (Van Auken, 2000).

Above-ground carbon stocks in savannas are strongly contingent on disturbance regimes. Australian savanna systems are currently a net carbon sink of 1– 3 t C ha^{-1} y^{-1}, depending on fire frequency and extent (Williams *et al.*, 2004b). Fire exclusion can transform savannas to forests (*e.g.* Bowman *et al.*, 2001), with an upper (albeit technically unfeasible) global estimate of potential doubling of closed forest cover (Bond *et al.*, 2005). Thus savanna structure and carbon stocks are very likely to be responsive to both individual and interactive effects of disturbance regime (Sankaran *et al.*, 2005; Bond *et al.*, 2003) and atmospheric CO_2 change (Bond and Midgley, 2000).

There are few factorial experiments on multiple changing factors, but they suggest interactions that are not predictable from single factor experiments – such as the dampening effect of elevated CO_2 on California C_3 grassland responses to increased rainfall, nitrate, and air temperature (Shaw *et al.*, 2002). Increasing temperature and rainfall changes are seen to override the potential benefits of rising CO_2 for C_3 relative to C_4 grasses (Winslow *et al.*, 2003), and European C_3 grassland showed minor responses to 3ºC rise in temperature possibly due to concomitant drying impacts (Gielen *et al.*, 2005). Elevated CO_2 impacts on grassland carbon sequestration also seem dependent on management practices (Harmens *et al.*, 2004; Jones and Donnelly, 2004), and are complicated by being species- but not functional-type specific (Niklaus *et al.*, 2001; Hanley *et al.*, 2004).

Soil-mediated responses are important in biogeochemical controls of vegetation response. Long term CO_2 enrichment of southern African C_4 grassland revealed limited 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). Likewise, elevated CO_2 impacts on litter decomposition and soil fauna seem species-specific and relatively minor (Ross *et al.*, 2002; Hungate *et al.*, 2000). Warming of a tallgrass prairie showed increased plant growth that supported enhanced soil fungal success (Zhang *et al.*, 2005). 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). Soil respiration shows a ~20% increase in response to ~2.4°C warming (Norby *et al.*, 2007), though acclimatization of soil respiration (Luo *et al.*, 2001) and root growth (Edwards *et al.*, 2004) to moderate warming has also been observed. Soil carbon loss from UK soils, many in grasslands, confirm carbon losses of ~2% per annum in carbon-rich soils, likely related to regional climate change (Bellamy *et al.*, 2005). In an African savanna system, rainfall after a dry spell generates substantial soil respiration activity and soil respiratory carbon losses (Veenendaal *et al.*, 2004), suggesting strong sensitivity to rainfall variability.

Climate change impacts studies for savanna and grassland fauna are few. The proportion of threatened mammal species may increase to between 10 and 40% between 2050 and 2080 (Thuiller *et al.*, 2006a). Changing migration routes especially threaten migratory African ungulates and their predators (Thirgood *et al.*, 2004). Observed population declines in three African savanna ungulates suggest that summer rainfall reductions could result in their local extirpation if regional climate change trends are sustained (Ogutu and Owen-Smith, 2003). For an African arid savanna raptor, population declines have been simulated for drier, more variable rainfall scenarios (Wichmann *et al.*, 2003). A 4% to 98% species range reduction for ~80% of mainly savanna and grassland animal species in South Africa is projected under an IS92a scenario (Erasmus *et al.*, 2002).

4.4.4 Mediterranean ecosystems

Properties, goods and services: Mediterranean-type ecosystems (MTEs) are located in mid-latitudes on all continents (covering ~ 3.4 Mkm^2), often on nutrient-poor soils and in coastal regions. These biodiverse systems (Cowling *et al.*, 1996) are climatically distinct with generally wet winters and dry summers (Cowling *et al.*, 2005), and thus fire-prone (Montenegro *et al.*, 2004). Vegetation structure is mainly shrub-dominated, but woodlands, forests and even grasslands occur in limited regions. Heavily utilized landscapes are dominated by grasses, herbs and annual plant species (Lavorel, 1999). MTEs are valuable for high biodiversity overall (Myers *et al.*, 2000) and thus nature-based tourism, but many extractive uses include wildflower harvesting in South Africa and Australia, medicinal herbs and spices, and grazing in the Mediterranean basin and Chile. Water yield for human consumption and agriculture are critical in South Africa, and these systems provide overall soil protection services on generally unproductive nutrient-poor soils.

Key vulnerabilities: Mediterranean-type ecosystems were not explicitly reviewed in the TAR, but threats from desertification were projected due to expansion of adjacent semi-arid and arid systems under relatively minor warming and drying scenarios. Warming and drying trends are likely to induce substantial species range shifts, and imply a need for migration rates that will exceed the capacity of many endemic species. 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: These systems may be among the most impacted by global change drivers (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 bioclimatic zone of the Cape Fynbos Biome could lose 65% of its area under warming of 1.8°C relative to 1961-1990 (2.3°C, pre-industrial), with ultimate species extinction of 23% resulting in the long term (Thomas *et al.*, 2004b). For Europe, only minor biome-level shifts are projected for Mediterranean vegetation types (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 hypothetical and uncertain CO_2 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, substantial vegetation shifts are projected under double CO_2 scenarios (Malcolm *et al.*, 2002b).

Climate change is likely to increase fire frequency and fire extent. Greater fire frequencies are noted in Mediterranean basin regions (Pausas and Abdel Malak, 2004) with some exceptions (Mouillot *et al.*, 2003). Double CO_2 climate scenarios increase wildfire events by 40-50% in California (Fried *et al.*, 2004), and double fire risk in Cape Fynbos (Midgley *et al.*, 2005), favouring re-sprouting plants 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) and reducing productivity and carbon sequestration (Lenihan *et al.*, 2003).

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, and has now been observed in the eastern Mediterranean (Körner *et al.*, 2005b). Soil water content controls ecosystem water and CO_2 flux in the Mediterranean Basin system (Rambal *et al.*, 2003), and reductions are very likely to 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 appeared to have recovered from drought by 2004 (Gobron *et al.*, 2005, Box 4.1).

Many MTE species show apparently limited benefits from rising atmospheric CO_2 (Dukes *et al.*, 2005), with constrained increases in above-ground productivity (*e.g.* Blaschke *et al.*, 2001; Maroco *et al.*, 2002). Yet modelling suggests that under all but extremely dry conditions, CO_2 increases over the past century have already increased NPP and LAI in the Mediterranean basin, despite warming and drying trends (Osborne *et al.*, 2000). Rising atmospheric CO_2 appears increasingly unlikely to have a major impact in MTEs over the next decades, especially because of consistent projections of reduced rainfall. Elevated CO_2 is projected to facilitate forest expansion and greater carbon storage in California if precipitation increases (Bachelet *et al.*, 2001). In the Mediterranean basin, CO_2 fertilization impacts such as increased forest success in the eastern Mediterranean and Turkey and increased shrub cover in northern Africa are simulated if rainfall does not decrease (Cheddadi *et al.*, 2001). There is currently insufficient evidence to project elevated CO_2–induced shifts in ecosystem carbon stocks in MTE, but nutrient-limited systems appear relatively unaffected (de Graaff *et al.*, 2006). Established *Pinus halepensis* (Borghetti *et al.*, 1998) show high drought resistance, but Ponderosa pine forests had reduced productivity and water flux during a 1997 heat wave, and 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 (Peñuelas *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 Peñuelas, 2004) and tree species (Ogaya and Peñuelas, 2003), but delayed flowering and reduced flower production of Mediterranean basin shrub species (Llorens and Peñuelas, 2005), suggesting complex changes in species relative success under drying scenarios. Drought may also act indirectly on plants by reducing the availability of soil phosphorus (Sardans and Peñuelas, 2004).

Bioclimatic niche-based modelling studies project reduced endemic species geographic ranges and species richness in the Cape Floristic Region (Midgley *et al.*, 2002; Midgley *et al.*, 2003; Midgley *et al.*, 2006). 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.*, 2005b). 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.*, 2005b).

4.4.5 Forests and woodlands

Properties, goods and services: Ecosystems with a dense tree 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 (Figure 4.1, *e.g.* Sabine *et al.*, 2004). Forests require relatively favourable environmental conditions and are among the most productive terrestrial ecosystems (Figure 4.1). This makes them attractive both for climate change mitigation (WGIII, 9; Watson *et al.*, 2000) and agricultural uses. The latter underlies 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). Nevertheless, forests sequester the largest fraction of terrestrial ecosystem carbon stocks, recently estimated at 1640 PgC (Sabine *et al.*, 2004, Figure 4.1), equivalent to ~220% of atmospheric carbon. In addition to commercial timber goods (this volume, 5, Shvidenko *et al.*, 2005 Section 21.5 p.600-607) forests provide numerous non-timber forest products, important for subsistence livelihoods (Gitay *et al.*, 2001 p. 239; Shvidenko *et al.*, 2005). Key ecosystem services include 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, 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 benefits (Reid *et al.*, 2005; Millennium Ecosystem Assessment, 2005).

Key vulnerabilities: Forests, especially in the boreal region, have been identified as having a high potential vulnerability to climate change in the long-term (Kirschbaum and Fischlin, 1996), but more immediately if disturbance regimes (drought, insects, fire), partly due to climate change, cross critical thresholds (Gitay *et al.*, 2001). Since the TAR, most DGVM models based on A2 scenarios show significant forest dieback towards the end of this century and beyond in tropical, boreal, and mountain areas, with a concomitant loss of key services (Figure 4.3.). Species-based approaches suggest losses of diversity in particular in tropical forest diversity hotspots (*e.g.* north-eastern Amazonia, Miles, 2002) and tropical Africa (McClean *et al.*, 2005), with medium confidence. Mountain forests are increasingly encroached upon from adjacent lowlands, while simultaneously losing high altitude habitats due to warming (see also 4.4.7).

Impacts: Projections for some forests currently limited by their minimum climatic requirements indicate gains from climate change (Figure 4.3, vegetation changes 1 and 2), but many may be impacted detrimentally (Figure 4.3, vegetation change 6), notably for strong warming and its concomitant effects on water availability (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; Matsui *et al.*, 2004; Izaurralde *et al.*, 2005; Fuhrer *et al.*, 2006; Lucht *et al.*, 2006; Schaphoff *et al.*, 2006; Scholze *et al.*, 2006; cf. Figure 4.3a vs. b, vegetation change 6). Productivity gains may result through three mechanisms: (i) CO_2 fertilization (although the magnitude of this effect remains uncertain in these long-lived systems, see 4.4.1); (ii) warming in cold climates, given concomitant precipitation increases to compensate for possibly increasing water vapour pressure deficits; (iii) precipitation increases under water limited conditions.

There is growing evidence (see also this volume, 5.4.1.1) that several factors may moderate direct CO_2 or climate change effects on net ecosystem productivity in particular, namely nutrient dynamics (*e.g.* either enrichment or leaching resulting from N deposition), species composition, dynamic age structure effects, pollution, and biotic interactions, particularly via soil organisms, (*e.g.* Karnosky *et al.*, 2003; King *et al.*, 2004b; Heath *et al.*, 2005; Körner *et al.*, 2005a; 4.4.1). Climate change impacts on forests will result not only through changes in mean climate, but also through changes in seasonal and diurnal rainfall and temperature patterns (as influenced by the hydrologically relevant surroundings of a forest stand, *e.g.* Zierl and Bugmann, 2005). Recently observed moderate climatic changes have induced forest productivity gains globally (reviewed in Boisvenue and Running, 2006) and possibly enhanced carbon sequestration, especially in tropical forests (Baker *et al.*, 2004; Lewis *et al.*, 2004a; Lewis *et al.*, 2004b; Malhi and Phillips, 2004; Phillips *et al.*, 2004), where these are not reduced by water limitations (*e.g.* Boisvenue and Running, 2006) or offset by deforestation or novel fire regimes (Nepstad *et al.*, 1999; Nepstad *et al.*, 2004; Alencar *et al.*, 2006) or by hotter and drier summers at mid and high latitudes (Angert *et al.*, 2005).

Potential increases in drought conditions have been quantitatively projected for several regions (*e.g.*, Amazon, Europe) during the critical growing phase, due to increasing summer temperatures and precipitation declines (*e.g.* Cox *et al.*, 2004; Schaphoff *et al.*, 2006; Scholze *et al.*, 2006; Figure 4.3, vegetation change 6). Since all these responses potentially influence net forest ecosystem productivity (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; Semazzi and Song, 2001; Bergengren *et al.*, 2001; Leemans *et al.*, 2002; Körner, 2003b; Cox *et al.*, 2004; Canadell *et al.*, 2004; Gruber *et al.*, 2004; Heath *et al.*, 2005; 4.4.1). Effects of drought on forests include mortality, a potential reduction in resilience (*e.g.* Lloret *et al.*, 2004; Hogg and Wein, 2005), and can cause major biotic feedbacks (*e.g.* Ciais *et al.*, 2005; Box 4.1). However, these effects remain incompletely understood and vary from site to site (*e.g.* Reichstein *et al.*, 2002; Betts *et al.*, 2004). For example, drought impacts can be offset by fertile soils (Hanson and Weltzin, 2000), or if due to a heat wave, drought may even be accompanied by enhanced tree growth at cooler high elevation sites due to a longer growing season and enhanced photosynthetic activity (Jolly *et al.*, 2005; Box 4.1).

Drought conditions further interact with disturbances such as insects (Hanson and Weltzin, 2000; Fleming *et al.*, 2002; Logan *et al.*, 2003; Schlyter *et al.*, 2006; Box 4.1) or fire (Flannigan *et al.*, 2000). Tree defoliating insects, especially in boreal forests, periodically cause substantial damage (*e.g.* Gitay *et al.*, 2001 Box 5-10; Logan *et al.*, 2003). Insect pests were found to be at least partly responsible for the decline and ultimate extirpation of stands at the southern margins of the range of their hosts, subjected to warmer and drier conditions (Volney and Fleming, 2000). At the poleward ecotone (see glossary), frosts and general low temperatures appear to limit insect outbreaks (Virtanen *et al.*, 1996; Volney and Fleming, 2000), thus outbreaks currently constrained from northern ranges could become more frequent in the future (Carroll *et al.*, 2004). If climate warms and this ecotone becomes exposed to more droughts, insect outbreaks will become a major factor (Logan *et al.*, 2003; Gan, 2004). With A2 and B2 SRES scenarios downscaled to regional level in northern Europe, projected climate extremes by 2070-2100 will increase the susceptibility of Norway spruce to secondary damage through pests and pathogens, matched by an accelerated life cycle of spruce bark beetle populations (Schlyter *et al.*, 2006).

Climate change is known to alter the likelihood of increased wildfire sizes and frequencies (*e.g.* Stocks *et al.*, 1998; Podur *et al.*, 2002; Brown *et al.*, 2004; Gillett *et al.*, 2004), while also inducing stress on trees that indirectly exacerbate disturbances (Dale *et al.*, 2000; Fleming *et al.*, 2002; Schlyter *et al.*, 2006). This suggests an increasing likelihood of more prevalent fire disturbances, as has recently been observed (van der Werf *et al.*, 2004; Westerling *et al.*, 2006; 4.2.2).

Considerable progress has been made since TAR 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; Gillett *et al.*, 2004) enabling improved projections of future fire regimes (Flannigan *et al.*, 2000; Li *et al.*, 2000; de Groot *et al.*, 2003; Fried *et al.*, 2004; Brown *et al.*, 2004). Some argue (*e.g.* Harden *et al.*, 2000) that the role of fire regimes in the boreal region have previously been underestimated. About 10% of the 2002/2003 global carbon emission anomaly can be ascribed to Siberian fires by inverse modelling (van der Werf *et al.*, 2004), as supported by remote sensing (Balzter *et al.*, 2005). Climate changes including El Niño events 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), Indonesia, and Alaska (Hess *et al.*, 2001), but also introduce fire into regions where it was previously absent (*e.g.* Schumacher *et al.*, 2006). Intensified fire regimes are likely to impact boreal forests at least as much as climate change itself (Flannigan *et al.*, 2000), and may accelerate transitions, *e.g.* between taiga and tundra, through facilitating invasion of pioneering trees and shrubs into tundra (Landhäusser and Wein, 1993; Johnstone and Chapin, 2006).

Will forest expansions be realised as suggested by DGVMs (Figure 4.3)? Vegetation models project that forest might eventually replace between 11 and 50% of tundra with a doubling of atmospheric CO_2 (White *et al.*, 2000b; Harding *et al.*, 2002; Kaplan *et al.*, 2003; Callaghan *et al.*, 2005; Figure 4.3, vegetation change 1) moderated by many processes (*e.g.* Gamache and Payette, 2005). Other studies using a wide range of GCMs and forcing scenarios indicate that forests face globally the risk of major change (non-forested to forested and vice-versa within at least 10% of non cultivated land area) with a probability of ~43% if warming remains below 2°C globally relative to pre-industrial and of 88% if warming exceeds 3°C globally over pre-industrial (Scholze *et al.*, 2006). Those risks have been estimated as especially high for the boreal zone (44%, 88% respectively) whereas they were estimated as smaller for tropical forests in Latin America (19%, 38% respectively; see also Figure 4.3). One key process controlling such shifts is migration (*e.g.* Higgins and Harte, 2006). Estimates for migration rates of tree species from palaeoecological records are on average 200-300 m yr^{-1}, which is significantly below that of anticipated future climate change (≥1 km yr-1, Gitay *et al.*, 2001 Box 5-2) exceeding considerably that of postglacial warming. However, considerable uncertainties remain: (i) Though not completely quantified, many species can achieve rapid large-scale migrations (Reid's paradox (see glossary), *e.g.* Clark, 1998), but estimates at the low extreme imply a considerable range of lagged responses (Clark *et al.*, 2001; *e.g.* lag 0-20 yr Tinner and Lotter, 2001; lag several millennia, Johnstone and Chapin, 2003); (ii) recent genetic analysis (<100 m yr-1, McLachlan *et al.*, 2005) indicates that commonly inferred estimates from pollen have overestimated dispersal rates, explaining observed pollen records by multifront recolonisation from low-density refugees (Pearson, 2006); (iii) future landscapes will differ substantially from past climate change situations and landscape fragmentation creates major obstacles to migration (*e.g.* Collingham and Huntley, 2000); (iv) processes moderating migration such as competition, herbivory, or soil formation (Juday, 2005; land use, Vlassova, 2002; paludification, Crawford *et al.*, 2003; herbivory, Cairns and Moen, 2004; pathogens, Moorcroft *et al.*, 2006; 4.4.6); (v) tree species do not only respond to a changing climate by migration, but also by local adaptation, including genetic adaptation (Davis and Shaw, 2001; Davis *et al.*, 2005).

Modelling studies reconstructing past (*e.g.* Lischke *et al.*, 2002) or projecting future (Malcolm *et al.*, 2002b; Iverson *et al.*, 2004; Neilson *et al.*, 2005) dispersal, all indicate that more realistic migration rates will result in lagged northward shifts of taiga (lag length 150-250 yr, Chapin and Starfield, 1997; Skre *et al.*, 2002). While shrub expansion and a movement of the tree line (see glossary) were found to have advanced polewards in response to recent warming (Sturm *et al.*, 2001; Lloyd, 2005; Tape *et al.*, 2006; this volume, chapter 1), the expected slow encroachment of taiga into tundra is confirmed by satellite data showing no expansion of boreal forest stands (Masek, 2001) indicating century long time lags for the forest limit (see glossary) to move northward (Lloyd, 2005). All these findings suggest considerable uncertainties in how fast forests will shift northward (*e.g.* Clark *et al.*, 2003; Higgins *et al.*, 2003; Chapin *et al.*, 2004; Jasinski and Payette, 2005; McGuire *et al.*, 2007) and in the resulting consequences for the climate system (discussed in 4.4.6). Lower rates for the majority of species are probably realistic, also because future conditions comprise both unprecedented climate characteristics, including rapid rates of change (4.2.1, 4.4.11), and a combination of impediments to local adaptation and migration (with the exception of some generalists).

Compared to the TAR (Gitay *et al.*, 2001) the net global loss due to land use change 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-1 Lepers *et al.*, 2005; Alcamo *et al.*, 2006), while in some northern regions like Siberia, degradation rates are increasing largely due to unsustainable logging (Lepers *et al.*, 2005). Though uncertainties in 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, 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 in some areas, climate change generally exacerbates biodiversity risks especially in biodiversity hotspots and particularly for the first half of the 21st century, (montane cloud forests, Foster, 2001; Hawai Benning *et al.*, 2002; Costa Rica Enquist, 2002; Amazonia Miles, 2002; Australia Williams *et al.*, 2003). In tropical montane cloud forests, extinctions of amphibian species has been attributed to recent climate change (Pounds *et al.*, 2006; 4.4.7; Table 4.1 #2). In a few exceptions climate change may increase diversity locally or regionally (Kienast *et al.*, 1998) but in most cases extinction risks are increased.

4.4.6 Tundra and Arctic/Antarctic ecosystems

Properties, goods and services: Tundra denotes vegetation and ecosystems north of the closed boreal forest treeline (~5.6 Mkm^2), but here we also include ecosystems at circumpolar latitudes, notably the sea ice biome in both hemispheres (*e.g.* Arrigo and Thomas, 2004; 4.4.9), and sub-Antarctic islands (but see also this volume, 15). Ecosystem services include carbon sequestration, climate regulation, biodiversity and cultural maintenance, fuel, and food and fibre production (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 cover and, eventually, expanding forests (Chapin *et al.*, 2005b) and net emissions of greenhouse gasses, notably methane. The arctic significantly contributes to global biodiversity (Usher *et al.*, 2005; Chapin *et al.*, 2005a). Local mixed economies of cash and subsistence depend strongly on the harvest of local resources, food preparation, storage, distribution, and consumption. This forms a unique body of cultural knowledge traditionally transmitted from generation to generation (Hassol, 2004a).

Key vulnerabilities: Arctic and sub-arctic ecosystems (particularly ombrotrophic bog communities, see glossary) above permafrost were considered likely to be most vulnerable to climatic changes, since impacts may turn arctic regions from a net carbon sink to a net source (Gitay *et al.*, 2001). Literature since the TAR suggests that changes in albedo and an increased release of methane from carbon stocks (*e.g.* Christensen *et al.*, 2004), whose magnitudes were previously substantially underestimated, will lead to positive radiative climate forcing through the Arctic region (Camill, 2005; Lelieveld, 2006; Walter *et al.*, 2006; Zimov *et al.*, 2006). Adverse impacts, including pollution (see also this volume, 15), were projected for species such as marine birds, seals, polar bears, tundra birds and tundra ungulates (Gitay *et al.*, 2001). Unique endemic biodiversity (*e.g.* polar bears, Box 4.3) as well as tundra dependent species like migratory birds (*e.g.* waterfowl, Box 4.5, 4.4.8, Table 4.1) have been confirmed to be facing increasing extinction risks with concomitant threats to the livelihoods and food security for indigenous peoples.

Impacts: Global warming is projected to be most pronounced at high latitudes (WGI, 10,11; Phoenix and Lee, 2004). Ongoing rapid climatic changes will force tundra polewards at unprecedented rates (Velichko, 2002), causing lagged responses in its slow growing plant communities (Camill and Clark, 2000; Chapin *et al.*, 2000; Callaghan *et al.*, 2004a; Callaghan *et al.*, 2004c; Velichko *et al.*, 2004). Movements of some species of habitat-creating plants (edificators) require large spread rates exceeding their migrational capacity (Callaghan *et al.*, 2005). Poleward taiga encroachment into tundra is also likely to lag these changes (4.4.5 and *e.g.* Callaghan *et al.*, 2004b). Projections of vegetation changes in the northern Arctic suggest that by ~2080, 17.6% (14-23%) replacement of the current polar desert by tundra vegetation will have begun (Callaghan *et al.*, 2005). An eventual replacement of dwarf shrub tundra by shrub tundra is projected for the Canadian Arctic by 2100 (Kaplan *et al.*, 2003). Experimental manipulations of air temperature at 11 locations across the tundra also show that tundra plant communities change substantially through shifts in species dominance, canopy height, and diversity (Walker *et al.*, 2006) with cryptogams being particularly vulnerable (Cornelissen *et al.*, 2001; van Wijk *et al.*, 2004). A warming of 1-3°C caused a short-term diversity decrease, but generalisations are unwarranted because of insufficiently long experimentation time (Graglia *et al.*, 2001; Dormann and Woodin, 2002; van Wijk *et al.*, 2004; Walker *et al.*, 2006).

The thermally stable oceanic climate of the sub-Antarctic Marion Island appears to be changing, with a rise in annual mean surface air temperature of 1.2ºC 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 semi-annual oscillation in the Southern Hemisphere after about 1980 (Rouault *et al.*, 2005). Climatic change will directly affect the indigenous biota of sub-Antarctic islands (Smith, 2002; Barnes *et al.*, 2006). Experimental drying of the keystone cushion plant species *Azorella selago* on Marion Island revealed measurable negative impacts after only a few months (Le Roux *et al.*, 2005).

While summer food availability may increase for some vertebrates (Hinzman *et al.*, 2005), formation of ice-crust at critical winter times may reduce abundance of food below snow (Yoccoz and Ims, 1999; Aanes *et al.*, 2002; Inkley *et al.*, 2004). 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 tundra habitat shrinks (Box 4.5, Table 4.1). In contrast, northern range expansions of more southern species are expected, *e.g.* moose and red fox (Callaghan *et al.*, 2005). Some colonisers might ultimately need to be considered "invasive" species (*e.g.*, North American Mink, Neuvonen, 2004), and presently restricted populations of southern shrub species that are likely to spread in a warmer climate (Forbes, 1995) and possibly increased carbon sequestration (Sturm *et al.*, 2001; Tape *et al.*, 2006; for a discussion of overall consequences for climate, see end of 4.4.6). For arctic species such as the polar bear, increasing risks of extinction are associated with the projected large decrease in the extent of the sea ice biome and the sea ice cover (Box 4.3).

Box 4.3: Polar bears – a species in peril?

There are an estimated 20,000 to 25,000 polar bears (*Ursus maritimus*) worldwide, mostly inhabiting the annual sea ice over the continental shelves and inter-island archipelagos of the circumpolar Arctic, where they may wander for thousands of kilometres per year. They are specialized predators on ice-breeding seals and are therefore dependent on sea ice for survival. Female bears require nourishment after emerging in spring from a 5 to 7 month fast in nursing dens (Ramsay and Stirling, 1988), and are thus very dependent on close proximity between land and sea ice before it breaks up. Continuous access to sea ice allows bears to hunt throughout the year, but in areas where the sea ice melts completely each summer, they are forced to spend several months in tundra fasting on stored fat reserves until freeze-up.

Polar bears face great challenges from the effects of climatic warming (Stirling and Derocher, 1993; Stirling *et al.*, 1999; Derocher *et al.*, 2004), as projected reductions in sea ice will drastically shrink marine habitat for polar bears, ice-inhabiting seals and other animals (Hassol, 2004b). Break up of the sea ice on western Hudson Bay, Canada already occurs ~3 weeks earlier than 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, similar impacts on other subpopulations of polar bears can be reasonably expected. 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 from climate change. If sea ice declines according to some projections (WGI, 10, Figure 10.3.10; Figure 4.4, Table 4.1) polar bears will face a high risk of extinction with warming of 2.8°C above pre-industrial (range 2.5-3.0°C, Table 4.1). Similar consequences are facing other ice-dependent species, not only in the Arctic but also in the Antarctic (this volume, 1; Barbraud and Weimerskirch, 2001; Croxall *et al.*, 2002).

Significant changes in tundra are of two main types (Velichko *et al.*, 2005), namely in vegetation structure (and related albedo), and in below-ground processes related to a combined increase in temperature, increase in depth of the active layer (see glossary), and moisture content. These will promote paludification (see glossary, Crawford *et al.*, 2003), thermokarst processes (see glossary), and increase the dryness of raised areas. Moisture supply substantially influences the state of permafrost, one of the most important components of the tundra landscape (Anisimov *et al.*, 2002a; Anisimov *et al.*, 2002b). Increasing active layer instability causes greater mixing and shifting of the soil's mineral matrix, damaging plant roots. Generally this will favour moisture-loving species (*e.g.* sedges), while the peat bog vegetation over permafrost could experience drier conditions (Camill, 2005).

Substantial recent upward revisions (Zimov *et al.*, 2006) of carbon stocks (Figure 4.1) in permafrost and yedoma (see glossary), and measurements of methane releases from North Siberian thaw lakes (Walter *et al.*, 2006), Scandinavian mires (Christensen *et al.*, 2004) and Canadian permafrost (Camill, 2005) now show tundra to be a significantly larger atmospheric methane source than previously recognized. Current estimates of northern wetland methane emissions increase by 10–63%

based on Northern Siberian estimates alone. This methane source comprises a positive feedback to climate change, as thaw lakes (Walter *et al.*, 2006) and mires (Christensen *et al.*, 2004) are expanding in response to warming. While thermokarst-derived emissions are currently modest relative to anthropogenic sources, a potential stock of ~500 Pg of labile carbon in yedoma permafrost (Figure 4.1) could greatly intensify the positive feedback to high-latitude warming trends that are currently projected (Sazonova *et al.*, 2004; Mack *et al.*, 2004; Lelieveld, 2006; Zimov *et al.*, 2006).

Changes in albedo associated with snow cover loss, and eventual invasion of tundra vegetation by evergreen coniferous trees is likely to decrease regional albedo significantly and lead to a warming effect greater than the cooling projected from the increased carbon uptake by advancing trees (4.4.5) and shrubs (Betts, 2000; Sturm *et al.*, 2001; Chapin *et al.*, 2005b; Sturm *et al.*, 2005; McGuire and Chapin, 2006; McGuire *et al.*, 2007). Remote sensing already shows that tundra has greened over the past 20 years (Sitch *et al.*, 2007). However, the potential for CO_2 sequestration varies from region to region (Callaghan *et al.*, 2005) and model uncertainties are high (Sitch *et al.*, 2007), since migration rates (4.4.5), changes in hydrology, fire, insect pest outbreaks and human impacts relevant to the carbon cycle are poorly represented (4.4.1, 4.4.5).

4.4.7 Mountains

Properties, goods and services: Mountain regions (ca. 20-24% of all land, scattered throughout the globe) exhibit many climate types corresponding to widely-separated latitudinal belts within short horizontal distances. Consequently, although species richness decreases with elevation, mountain regions support many different ecosystems and have among the highest species richness globally (*e.g.* Moser *et al.*, 2005; Spehn and Körner, 2005; Väre *et al.*, 2003). Mountain ecosystems have a significant role in biospheric carbon storage and carbon sequestration, particularly in semi-arid and arid areas (*e.g.*, western US, Tibetan plateau, Schimel *et al.*, 2002; Piao and Fang, 2006). 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 habitability of mountain areas, such as through slope stabilisation or protection from natural disasters such as avalanches and rockfall, provide many goods for subsistent livelihoods, and are home to many indigenous peoples. Mountains harbour a significant fraction of biospheric carbon (28% of forests are in mountains). Finally, they serve increasingly as refuges from direct human impacts for many endemic species and are attractive for recreational activities and tourism.

Key vulnerabilities: The TAR identified mountain regions as having experienced above-average warming in the 20th century, a trend likely to continue (Beniston *et al.*, 1997; Liu and Chen, 2000). Related impacts included a shortened snow melt period, with rapid water release and downstream floods, which in combination with reduced glacier extent, could cause water shortage during the growing season. The TAR suggested the these impacts may be exacerbated 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 were generally considered limited, high vulnerability was attributed to the many highly endemic alpine biota (Pauli *et al.*, 2003). Since the TAR, the literature has confirmed a disproportionately high risk of extinction for many endemic species in various mountain ecosystems, such as tropical 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), and globally where habitat loss due to warming threatens endemic species (Pauli *et al.*, 2003; Thuiller *et al.*, 2005b).

Impacts: Because temperature decreases with altitude by 5-10°C/km, a first-order approximation of species vulnerability to climate change is that upward migration is necessary for persistence (*e.g.*, MacArthur, 1972; Beniston, 2000; Theurillat and Guisan, 2001). However, this is not possible for the coolest climatic and vegetation zones near mountain peaks (Gitay *et al.*, 2001; Peñuelas and Boada, 2003). Mountain ridges, by contrast, represent considerable obstacles to dispersal for many species (*e.g.* Foster, 2001; Lischke *et al.*, 2002; Neilson *et al.*, 2005; Pounds *et al.*, 2006). Upward migration necessarily reduces a species' geographic range (mountain tops are smaller than their bases), and likely reduces genetic diversity within species and increases the risk of stochastic extinction due to ancillary stresses (Peters and Darling, 1985; Bortenschlager, 1993). A reshuffling of species on altitude gradients is to be expected as a consequence of individualistic species responses that are mediated by varying longevities and survival rates. These in turn are the result of a high degree of evolutionary specialisation to harsh mountain climates (*e.g.* Theurillat *et al.*, 1998; Gottfried *et al.*, 1999; Theurillat and Guisan, 2001; Klanderud, 2005; Klanderud and Totland, 2005; Dullinger *et al.*, 2005; Huelber *et al.*, 2006), and in some cases they include effects induced by invading alien species as well (*e.g.* Dukes and Mooney, 1999; Mack *et al.*, 2000). Some genetic evidence for *Fagus sylvatica* suggests populations may show some capacity for an *in situ* adaptive response to climate change (Jump *et al.*, 2006). However ongoing distributional changes (Peñuelas and Boada, 2003) show this response will not allow this species to persist throughout its range.

Upper tree lines, which represent the interface between sub-alpine forests and low-stature alpine meadows, have long been thought to be partly controlled by carbon balance (Körner, 1999; Monson *et al.*, 2002). However, recent findings show there is a strong relationship of global treeline position and soil temperature, which drives the growth rate of all small mountain plants. Worldwide, climatic tree lines appear to be characterized by seasonal mean air temperatures of ca. 6°C (Körner, 1998; Grace *et al.*, 2002; Körner and Paulsen, 2004; Millar *et al.*, 2004; Zha *et al.*, 2005; Lara *et al.*, 2005). In many mountains, upper treeline is located below its potential climatic position because of grazing, or disturbances such as wind or fire. In other regions such as the Himalaya, deforestation of past decades has transformed much of the environment and has led to fragmented ecosystems (Becker and Bugmann, 2001). Although temperature control may well be a dominant determinant of geographic range, tree species may well be unable to migrate and keep pace with changing temperature zones (Shiyatov, 2003; Dullinger *et al.*, 2004; Wilmking *et al.*, 2004).

Where warmer and drier conditions are projected, mountain vegetation is expected to be subject to increased evapotranspiration (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). This leads to increased drought, which has been projected to induce forest dieback in continental climates, particularly in the interior of mountain ranges (*e.g.* Fischlin and Gyalistras, 1997; Lischke *et al.*, 1998; Lexer *et al.*, 2000; Bugmann *et al.*, 2005) and Mediterranean areas. Even in humid tropical regions, plants and animals have been shown to be sensitive to water stress on mountains (*e.g.*, Borneo, Kitayama, 1996; Costa Rica, Still *et al.*, 1999). There is very high confidence that warming is a driver of amphibian mass extinctions at many highland localities, by creating increasingly favourable conditions for the pathogenic *Batrachochytrium* fungus (Pounds *et al.*, 2006).

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). A lack of 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. At higher altitudes, the increased winter precipitation likely to accompany warming leads to greater snowfall, so that earlier arriving altitudinal migrants are confronted with delayed snowmelt (Inouye *et al.*, 2000).

Disturbances such as avalanches, rock fall, fire, wind, and herbivore damage interact and are strongly dependent on climate (*e.g.* Peñuelas and Boada, 2003; Whitlock *et al.*, 2003; Beniston and Stephenson, 2004; Cairns and Moen, 2004; Carroll *et al.*, 2004; Hodar and Zamora, 2004; Kajimoto *et al.*, 2004; Pierce *et al.*, 2004; Schoennagel *et al.*, 2004; Schumacher *et al.*, 2004). These effects may prevent recruitment and thus limit adaptive migration responses of species, 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).

Ecotonal (see glossary) sensitivity to climate change, such as tree lines in mountains (*e.g.* Camarero *et al.*, 2000; Walther *et al.*, 2001; Diaz, 2003; Sanz-Elorza *et al.*, 2003), has shown that populations of several mountain-restricted species are likely to decline (*e.g.* Beever *et al.*, 2003; Florenzano, 2004). The most vulnerable ecotone species are those that are genetically poorly adapted to rapid environmental change, reproduce slowly, disperse poorly, and are isolated or are highly specialized, because of their high sensitivity to environmental stresses (McNeely, 1990). Recent findings for Europe, despite a spatially coarse analysis, indicate that mountain species are disproportionately sensitive to climate change (~60% species loss, Thuiller *et al.*, 2005b). Substantial biodiversity losses are likely if human pressures on mountain biota occur in addition to climate change impacts (Pounds *et al.*, 1999; Lawton *et al.*, 2001; Pounds, 2001; Halloy and Mark, 2003; Peterson, 2003; Solorzano *et al.*, 2003; Pounds and Puschendorf, 2004; Pounds *et al.*, 2006).

4.4.8 Freshwater wetlands, lakes and rivers

Properties, goods and services: Inland aquatic ecosystems (covering ~10.3 Mkm^2) vary greatly in characteristics and global distribution. The majority of natural freshwater lakes are located in the higher latitudes, most artificial lakes occur in mid- and lower latitudes, and many saline lakes occur at altitudes up to 5000 m, especially in the Himalaya and Tibet. The majority of natural wetlands (peatlands) are in the boreal region but most managed wetlands (rice paddies) are in the tropics and subtropics (where peatlands also occur). Global estimates of the 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 heterogeneous spectrum of habitats following hydrological and nutrient gradients and all key processes, including goods and services provided, depend on the catchment level hydrology. Inland waters are subject to many pressures from human activities. Aquatic ecosystems provide a wide range of goods and services (Gitay *et al.*, 2001; Finlayson *et al.*, 2005). Wetlands are often biodiversity ‘hotspots’ (Reid *et al.*, 2005), as well as functioning as filters for pollutants from both point and non-point sources, and being important for carbon sequestration and emissions (Finlayson *et al.*, 2005). Rivers transport water and nutrients from the land to the oceans and provide crucial buffering capacity during drought spells especially if fed by mountain springs and glaciers (*e.g.*, European summer 2003; Box 4.1; this volume, 12.6.1). Endorheic (see glossary) lakes serve as sediment and carbon sinks (Cohen, 2003), providing crucial repositories of information on past climate changes.

Key vulnerabilities: Gitay *et al.* (2001) have described some inland aquatic ecosystems (arctic, sub-arctic ombrotrophic 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. More recent results show vulnerability varying by geographic region (Van Dam *et al.*, 2002; Stern *et al.*, 2006). This includes significant negative impacts across 25% of Africa by 2100 (SRES B1, de Wit and Stankiewicz, 2006) with both water quality and ecosystem goods and services deteriorating. Since it is generally difficult and costly to control hydrological regimes, the interdependence between catchments across national borders leaves often little scope for adaptation.

Impacts: The climate change impacts on inland aquatic ecosystems will range from the direct effects of the rise in temperature and CO_2 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.* WGII,1,3; Cubasch *et al.*, 2001; this volume, 3; WGI, 4, 10).

Studies since TAR have confirmed and strengthened the earlier conclusions that rising temperature will lower water quality in lakes through a fall in hypolimnetic (see glossary) oxygen concentrations, release of phosphorus (P) from sediments, increased thermal stability, and altered mixing patterns (McKee *et al.*, 2003; Verburg *et al.*, 2003; Winder and Schindler, 2004; Jankowski *et al.*, 2006). In northern latitudes, ice cover on lakes and rivers will continue to break up earlier and the ice-free periods to increase (this volume, 1; Weyhenmeyer *et al.*, 2004; Duguay *et al.*, 2006). Higher temperatures will negatively affect micro-organisms and benthic invertebrates (Kling *et al.*, 2003) and the distribution of many species of fish (Lake *et al.*, 2000; Poff *et al.*, 2002; Kling *et al.*, 2003), invertebrates, waterfowl and tropical invasive biota are likely to shift polewards (Moss *et al.*, 2003; Zalakevicius and Svazas, 2005) with some potential extinctions (Jackson and Mandrak, 2002; Chu *et al.*, 2005). Major changes will likely occur in the species composition, seasonality and production of planktonic communities (*e.g.*, increase in toxic blue-green algal blooms) and their food web interactions (Gerten and Adrian, 2002; Kling *et al.*, 2003; Winder and Schindler, 2004) with consequent changes in water quality (Weyhenmeyer, 2004). Enhanced UV-B radiation and increased summer precipitation will significantly increase dissolved organic carbon (DOC, see glossary) concentrations, altering major biogeochemical cycles (Phoenix and Lee, 2004; Zepp *et al.*, 2003; Frey and Smith, 2005). Studies along an altitudinal gradient in Sweden show that NPP can increase by an order of magnitude for a 6°C air temperature increase (Karlsson *et al.*, 2005). However, tropical lakes may respond with a decrease in NPP and a decline in fish yields (*e.g.* 20% NPP and 30% fish yield reduction in Lake Tanganyika due to warming over the last century O'Reilly *et al.*, 2003). Higher CO_2 levels will generally increase NPP in many wetlands, although in bogs and paddy fields it may also stimulate methane flux, thereby negating positive effects (Ziska *et al.*, 1998; Schrope *et al.*, 1999; Freeman *et al.*, 2004; Megonigal *et al.*, 2005; Zheng *et al.*, 2006).

Boreal peatlands will be affected most by warming (see also 4.4.5, 4.4.6) and increased winter precipitation as the species composition of both plant and animal communities will change significantly (Weltzin *et al.*, 2000; Berendse *et al.*, 2001; Weltzin *et al.*, 2001; Weltzin *et al.*, 2003; Keller *et al.*, 2004; 4.4.5, 4.4.6). Numerous Arctic lakes will dry out 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 need to change and some may be threatened with extinction (Inkley *et al.*, 2004; Reid *et al.*, 2005; Finlayson *et al.*, 2005; Zalakevicius and Svazas, 2005; Box 4.5).

Small increases in the variability of precipitation regimes will 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 terrestrialisation 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 biodiversity loss (Bauder, 2005). Changes in climate and land use will place additional pressures on already stressed riparian ecosystems along many rivers in the world (Naiman *et al.*, 2005). An increase or decrease in freshwater flows will also affect coastal wetlands (this volume, 6) by altering salinity, sediment inputs and nutrient loadings (Schallenberg *et al.*, 2001; Flöder and Burns, 2004).

4.4.9 Oceans and shallow seas

Properties, goods and services: Oceans cover over 71% of the earth's surface area from polar to tropical regions to a mean depth of 4000 m, comprising about 14 billion km^3, yet containing only 698–708 PgC (organic), 13–23 PgC of which is in living and dead biomass (Figure 4.1; WGI, 7.3.4.1). Despite low biomass, phytoplankton carries out almost half of global primary production, and is the basis of the marine food web (Field *et al.*, 1998). Substantial biodiversity exists in both pelagic and benthic realms and along coastlines, in a diverse range of ecosystems from highly productive (*e.g.* upwelling regions) to those with low productivity (*e.g.* oceanic gyres). Ocean primary productivity depends on sunlight and nutrients supplied from deep waters (Sarmiento *et al.*, 2004a). Marine ecosystems provide goods and services such as fisheries, provision of energy, recreation and tourism, CO_2 sequestration and climate regulation, decomposition of organic matter and regeneration of nutrients and coastal protection - many of which are critical to the functioning of the Earth system (Chapter 5, Costanza *et al.*, 1997; McLean and Tsyban, 2001, Sections 6.3.2, 6.3.4, 6.3.5, 6.4.5 and 6.4.6; Hassan *et al.*, 2005, Table 18.2, p. 79). Marine biodiversity supports ecosystem function and the services it provides (Worm *et al.*, 2006) with over 1 billion people relying on fish as their main animal protein source, especially in developing nations (Pauly *et al.*, 2005). 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 (McLean and Tsyban, 2001, Section 6.5, p. 362; Hassan *et al.*, 2005, Section 19.3.2, Table 19.2, p. 531, p. 676).

Key vulnerabilities: Impacts of high atmospheric CO_2 on lower surface ocean pH and carbonate ions were overlooked in TAR. Coral reefs, cold water corals, and ecosystems (*e.g.* Southern Ocean and Artic Ocean) where aragonite (used by many organisms to make their shells or skeletons) will decline or become under-saturated and where calcareous organisms (*e.g.* pteropods) play an important role will become vulnerable this century (reviewed by Raven *et al.*, 2005; Haugan *et al.*, 2006). 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 4.4). Marginal sea ice and surrounding ecosystems are vulnerable to warming, particularly in the Northern Hemisphere (Sarmiento *et al.*, 2004b; WGI, 11). Since the TAR, literature has confirmed that salient vulnerable ecosystems are warm-water coral reefs (Box 4.4), cold-water corals, the Southern Ocean and marginal sea ice ecosystems. Impacts of high atmospheric CO_2 on lower surface ocean pH and carbonate ions were overlooked in TAR. This impact affects coral reefs, cold water corals, and ecosystems (*e.g.* Southern Ocean), where aragonite (used by many organisms to make their shells or skeletons) will decline or become undersaturated. These and other ecosystems where calcareous organisms (*e.g.* pteropods, see glossary) play an important role will become vulnerable this century (reviewed by Raven *et al.*, 2005; Table 4.1). 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 4.4). Marginal sea ice and surrounding ecosystems are vulnerable to warming, particularly in the Northern Hemisphere (Sarmiento *et al.*, 2004b).

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 (IPCC, 2001), increase in wave height and frequency (Monahan *et al.*, 2000; Wang *et al.*, 2004b), loss of sea ice (Sarmiento *et al.*, 2004b; WGI, 10, 11), increased risk of diseases in marine biota (Harvell *et al.*, 2002) and decrease 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 2 x CO_2 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 (Cox *et al.*, 2000; Loukos *et al.*, 2003; Lehodey *et al.*, 2003; Sarmiento *et al.*, 2004a). In coastal areas and margins increased thermal stratification may lead to oxygen deficiency, loss of habitats, biodiversity and distribution of species and impact whole ecosystems (Rabalais *et al.*, 2002). Changes to rainfall and nutrient flux from land may exacerbate these hypoxic events (Rabalais *et al.*, 2002).

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 (Sarmiento *et al.*, 2004b; see also WGI, 10, 11). 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 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). Reductions in winter sea-ice will affect the reproduction, growth and development of fish, krill, and their predators, including seals and seal-dependent polar bears (*e.g.* Barber and Iacozza, 2004; Box 4.3), leading to further reductions in abundance and distribution changes (this volume, 15.4.3). An expansion by 4.0% (Northern) and 9.4% (Southern Hemisphere), and the sub-polar gyre biome by 16% (Northern) and 7% (Southern Hemisphere), has been projected for the permanently stratified, subtropical gyre biome with its low productivity. This effect has now been observed in the North Pacific and Atlantic (McClain *et al.*, 2004; Sarmiento *et al.*, 2004b). A contraction by 11% of the seasonally stratified subtropical gyre is also projected in both hemispheres by 2050 due to climate warming. These changes are likely to have significant impacts on marine ecosystem productivity globally, with uncertainties in projections of NPP using six mainly IS92a-based scenarios narrowing to an increase of between 0.7%-8.1% by mid-century (ΔT_{global} ~1.5-3ºC).

Changes to planktonic and benthic community composition and productivity have been observed in the North Sea since 1955 (Clark and Frid, 2001) and since the mid-1980s may have reduced the survival of young cod (Beaugrand *et al.*, 2003). Loose shifts in pelagic biodiversity (Beaugrand *et al.*, 2002) and in fish community composition have been seen (Genner *et al.*, 2004; Perry *et al.*, 2005). Changes in seasonality or recurrence of hydrographic events or productive periods could be affected by trophic links to many marine population, including exploited or cultured populations (Stenseth *et al.*, 2002; Stenseth *et al.*, 2003; Platt *et al.*, 2003; Llope *et al.*, 2006). Elevated temperatures increased mortality of winter flounder eggs and larvae (Keller and Klein-Macphee, 2000) and lead to later spawning migrations (Sims *et al.*, 2004). A 2°C rise in sea surface temperature (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).

Marine mammals, birds, cetaceans and pinnipeds (seals, sea lions and walruses), which feed mainly on plankton, fish and squid, are vulnerable to climate change-driven changes in prey distribution, abundance and community composition in response to climatic factors (Learmonth *et al.*, 2006). Changing water temperature also has an effect on the reproduction of cetaceans and pinnipeds, indirectly through prey abundance, either through extending the time between individual breeding attempts, or by reducing breeding condition of the mother (Whitehead, 1997). Current extreme climatic events provide an indication of potential future effects. For example, the warm water phase of the ENSO is associated with large-scale changes in plankton abundance and associated impacts on food-webs (Hays *et al.*, 2005), and changes to behaviour (Lusseau *et al.*, 2004), sex ratio (Vergani *et al.*, 2004) and feeding and diet (Piatkowski *et al.*, 2002) of marine mammals.

Melting Arctic ice-sheets will reduce ocean salinities (IPCC, 2001), causing species-specific shifts in the distribution and biomass of major constituents of Arctic food webs, including poleward shifts in communities and the potential loss of some polar species (such as Narwhal *Monodon monoceros*). Migratory whales (*e.g.* Grey Whale *Eschrichtius robustus*), that spend summer in Arctic feeding grounds are likely to experience disruptions in their food sources (Learmonth *et al.*, 2006). Nesting biology of sea turtles is strongly affected by temperature, both in timing and in the determination of the sex-ratio of hatchlings (Hays *et al.*, 2003), but implications for population size are unknown. A predicted sea-level rise of 0.5 metres will eliminate up to 32% loss of sea turtle nesting beaches in the Caribbean (Fish *et al.*, 2005).

Box 4.4: 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; Cesar *et al.*, 2003; Willig *et al.*, 2003; Hoegh-Guldberg, 2004; Hoegh-Guldberg, 2005), and though supplying only about 2-5% of the global fisheries harvest, comprise a critical subsistence protein and income source in the developing world (Pauly *et al.*, 2005; Sadovy, 2005; Whittingham *et al.*, 2003).

Corals are affected by warming of surface waters (this volume, Box 6.1; Reynaud *et al.*, 2003; McNeil *et al.*, 2004; McWilliams *et al.*, 2005) leading to bleaching (loss of algal symbionts, this volume Box 6.1). Many studies incontrovertibly link coral bleaching to warmer sea surface temperature (*e.g.* McWilliams *et al.*, 2005) and mass bleaching and coral mortality often results beyond key temperature thresholds (this volume, Box 6.1). Annual or bi-annual exceedance of bleaching thresholds is projected at the majority of reefs worldwide by 2030-2050 (Hoegh-Guldberg, 1999; Sheppard, 2003; Donner *et al.*, 2005). 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 phase switch to algal dominance on the Great Barrier Reef and Caribbean reefs in 2030 to 2050 (Wooldridge *et al.*, 2005).

Coral reefs will also be affected by rising atmospheric CO_2 concentrations (WGI, Box 7.3; Orr *et al.*, 2005; Raven *et al.*, 2005) resulting in declining calcification. Experiments at expected aragonite concentrations demonstrated a reduction in coral calcification (Marubini *et al.*, 2001; Langdon *et al.*, 2003; Hallock, 2005), coral skeleton weakening (Marubini *et al.*, 2003) and strong temperature dependence (Reynaud *et al.*, 2003). Oceanic pH projections decrease at a greater rate and to a lower level than that experienced over the past 20 million years (Caldeira and Wickett, 2003; Raven *et al.*, 2005; Turley *et al.*, 2006). 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 critical aragonite saturation states (Feely *et al.*, 2004; Orr *et al.*, 2005), resulting in reduced coral cover and greater erosion of reef frameworks (Kleypas *et al.*, 2001; Guinotte *et al.*, 2003).

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 may delay demise of some reefs to roughly 2100 (Sheppard, 2003), rather than mid-century (Hoegh-Guldberg, 2005) although this may vary widely across the globe (Donner *et al.*, 2005). Estimates of warm water coral cover reduction in the last 20-25 years are 30% or higher (Hoegh-Guldberg, 2005; Wilkinson, 2004) 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). Coral migration is also limited by lack of available substrate (Section 6.4.1.5). Elevated SST and decreasing aragonite have a complex synergy (Reynaud *et al.*, 2003; McNeil *et al.*, 2004; Kleypas *et al.*, 2005; Harvell *et al.*, 2002) but could produce major coral reef changes (Guinotte *et al.*, 2003; Hoegh-Guldberg, 2005). Corals could become rare on tropical and sub tropical reefs by 2050 due to the combined effects of increasing pCO_2 and increasing frequency of bleaching events (at 2-3 x CO_2 Kleypas and Langdon, 2002; Raven *et al.*, 2005; Hoegh-Guldberg, 2005). Other climate change factors (such as sea-level rise, storm impact and aerosols) and non-climate factors (such as over-fishing, invasion of non-native species, pollution, nutrient and sediment load (although this could also be related to climate changes through changes to precipitation and river flow; this volume, Boxes 6.1, 11.1, and chapter 16)) add multiple impacts on coral reefs (this volume, Box 16.2), increasing their vulnerability and reducing resilience to climate change (Koop *et al.*, 2001; Kleypas and Langdon, 2002; Cole, 2003; Buddemeier *et al.*, 2004; Hallock, 2005).

Surface ocean pH has decreased by 0.1 unit due to absorption of anthropogenic CO_2 emissions (equivalent to a 30% increase in hydrogen ion concentration) and is predicted to decrease 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.* coral reefs, Box 4.4, reviewed by Raven *et al.*, 2005) including juvenile planktonic, as well as adult, forms of benthic calcifying organisms (*e.g.* echinoderms, gastropods and shellfish). This will affect recruitment (reviewed by Turley *et al.*, 2006). Polar and sub-polar surface waters and the Southern Ocean will be aragonite under-saturated by 2100 (Orr *et al.*, 2005) and Arctic waters similarly threatened (Haugan *et al.*, 2006). Organisms using aragonite to make their shells (*e.g.* pteropods) will be at risk and this will threaten ecosystems such as the Southern and Arctic Oceans in which they play a dominant role in the food web and carbon cycling (Orr *et al.*, 2005; Haugan *et al.*, 2006).

Cold-water coral ecosystems exist 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; Raven *et al.*, 2005; Orr *et al.*, 2005; Guinotte *et al.*, 2006). 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 SST and decreasing aragonite concentrations within this century (Box 4.4).

4.4.10 Cross-biome impacts

This section highlights issues that cut across biomes, such as large-scale geographic shifts of vegetation (Figure 4.3) or animal migration patterns (*e.g.* Box 4.3, Box 4.5), and changes in land use and aquatic systems.

Biome shifts: Boreal forest and Arctic tundra ecosystems are projected generally to show increased growth due to longer and warmer growing seasons (Lucht *et al.*, 2002; Figure 4.3). Woody boreal vegetation is expected to spread into tundra at higher latitudes and higher elevations (Grace *et al.*, 2002; Kaplan *et al.*, 2003; Gerber *et al.*, 2004). At the southern ecotone (see glossary) with continental grasslands, a contraction of boreal forest is projected due to increased impacts of drought, insects and fires (Bachelet *et al.*, 2001; Scholze *et al.*, 2006), and a lower rate of sapling survival (Hogg and Schwarz, 1997). Drought stress could partially be counteracted by concurrent CO_2-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 cause regional transitions to grassland where continental winter climate remains too cold for temperate forest species to succeed (Gerber *et al.*, 2004; Lucht *et al.*, 2006). In temperate forests, milder winters may reduce winter hardening in trees, increasing their vulnerability to frost (Hänninen *et al.*, 2001; Hänninen, 2006). Vegetation change in the lower to mid latitudes is uncertain because transitions between tropical desert and woody vegetation types are 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 (Figure 4.3), though potentially associated with more frequent fires, in others (Bachelet *et al.*, 2003; Woodward and Lomas, 2004b; Ni *et al.*, 2006; Schaphoff *et al.*, 2006). In savannas, woody encroachment is projected to be a consequence of enhanced water use efficiency and increased precipitation in some regions (Bachelet *et al.*, 2001; Ni *et al.*, 2006; Lucht *et al.*, 2006; Schaphoff *et al.*, 2006; 4.4.3; Figure 4.3). The moderate drying, including desert amelioration, as projected in southern Africa, the Sahel region, central Australia, the Arabian Peninsula and parts of central Asia (Figure 4.3) may be due to a positive impact of rising atmospheric CO_2, as noted in eastern Namibia through sensitivity analysis (Thuiller *et al.*, 2006b). A general increase of deciduous at the expense of 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; Gerten *et al.*, 2005; Lucht *et al.*, 2006; Scholze *et al.*, 2006). 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 by some models to lose carbon beyond temperature increases of 3°C (Gerber *et al.*, 2004), mainly from temperate and boreal soils, with vegetation carbon declining beyond temperature increases above 5°C (Gerber *et al.*, 2004). Carbon sinks persist mainly in the Arctic and in savanna 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.

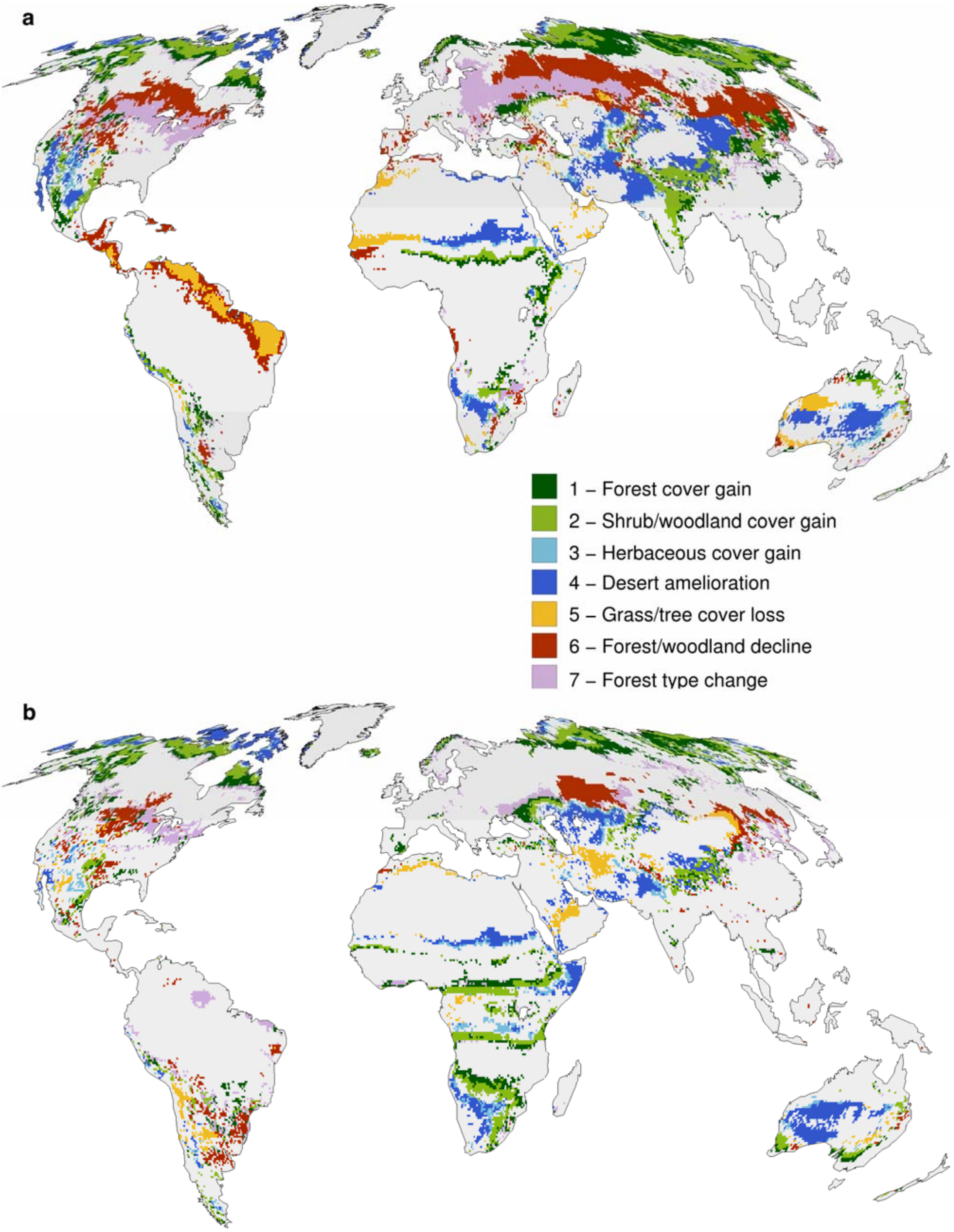

Figure 4.3: *Projected appreciable changes in terrestrial ecosystems by 2100 relative to 2000 as simulated by DGVM LPJ (Sitch et al., 2003; Gerten et al., 2004) for two IPCC SRES emission scenarios (Nakicenovic et al., 2000) forcing two climate models:* ***a****) HadCM3 A2,* ***b****) ECHAM5 B1 (Lucht et al., 2006; Schaphoff et al., 2006). Changes are considered appreciable and only shown if they exceed 20% of the area of a simulated grid cell (see Figure 4.2 for further explanations).*

Migration patterns: Vagile (see glossary) animals such as polar bears (sea ice biome, tundra; Box 4.3) and in particular migratory animals (tundra, wetlands, lakes, tropical forests, savannas, *etc.*, Box 4.5) respond to 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). 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 sensitive to climate. Numerous studies have found that many of these species are arriving earlier (this volume, 1 and *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 4.5).

Land use: The relative importance of key drivers on ecosystem change varies across regions and biomes (Sala *et al.*, 2000; Sala, 2005). Several global studies suggest that at least until 2050 land use change will be the dominant driver of terrestrial biodiversity loss in human dominated regions (Sala *et al.*, 2000; UNEP, 2002; Gaston *et al.*, 2003; Jenkins, 2003; Scharlemann *et al.*, 2004; Sala, 2005). Conversely, climate change is likely to dominate where human interventions are limited such as in the tundra, boreal, cool conifer forests, deserts and savanna biomes (Sala *et al.*, 2000; Duraiappah *et al.*, 2005). Assessment of impacts on biodiversity differ if other drivers than climate change are taken into account (Thomas *et al.*, 2004a; Sala, 2005; Malcolm *et al.*, 2006). Interactions among these drivers may mitigate or exacerbate the overall effects of climate change (Opdam and Wascher, 2004). The effects of land use change on species through landscape fragmentation at the regional scale may further exacerbate impacts from climate change (Holman *et al.*, 2005a; Del Barrio *et al.*, 2006; Harrison *et al.*, 2006; Rounsevell *et al.*, 2006).

Global land use change studies project a significant reduction in native vegetation (mostly forest) in non-industrialised countries and arid regions due to expansion of agricultural or urban land use driven principally by population growth, especially in Africa, South America and in South Asia (Hassan *et al.*, 2005). That reduction in native habitat will result in biodiversity loss (*e.g.* Duraiappah *et al.*, 2005; 4.4.11). Northern latitude countries and high altitude regions may become increasingly important for biodiversity and species conservation as the ranges of species distributions move poleward and upward in response to climate change (Berry *et al.*, 2006). Northern latitude countries and high altitude regions are also sensitive to the effects of climate change on land use, especially agriculture, which is of particular relevance if those regions are to 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).

Aquatic systems: Higher CO_2 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 CO_2 levels is exported to the streams and finally reaches the coastal waters (Freeman *et al.*, 2004).

Box 4.5: Crossing biomes – impacts of climate change on migratory birds

Migratory species can be affected by climate change in their breeding, wintering and/or critical stopover habitats. Models project changes in the future ranges of many species (Peterson *et al.*, 2002; Price and Glick, 2002; Crick, 2004), 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, especially in relation to seasonal migration timing, could help or hinder migration (Butler *et al.*, 1997). Other expected impacts include continuing changes in phenology, behaviour, population sizes, and possibly genetics (reviewed in Crick, 2004; Robinson *et al.*, 2005).

Many migratory species must cross geographic 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 4.2) 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 (Zöckler and Lysenko, 2000; Symon *et al.*, 2005). For example, the Red Knot could potentially lose 15%-37% of its tundra breeding habitat by 2100 (HadCM2a1, UKMO). Additionally, at least some populations of this species could also 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 shorebirds and waterfowl are projected to decline by up to 45% and 50% respectively (Folkestad, 2005) for global temperature increases of 2°C above pre-industrial. A temperature increase of 2.9°C above pre-industrial would cause larger declines of up to 76% for waterfowl and up to 56% for shorebirds. 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). One review of 300 migrant bird species found that 84% face some threat from climate change, almost half because of changes in water regime (lowered water tables and drought), and this was equal to the summed threats due to all other anthropogenic causes (Robinson *et al.*, 2005).

4.4.11 Global synthesis including impacts on biodiversity

Considerable progress has been made since the TAR in key fields that allow projection of future climate change impacts on species and ecosystems. Two of these key fields, namely climate envelope modelling (also called niche-based, or bioclimatic modelling) and dynamic global vegetation modelling (DGVMs) have provided numerous recent results. The synthesis of these results provides a picture of potential impacts and risks that is far from perfect, in some instances apparently contradictory, but overall highlights a wide array of key vulnerabilities (Figures 4.2, 4.4, 4.5, Table 4.1).

Climate envelope modelling has burgeoned recently due to increased availability of species distribution data, together with finer scale climate data and new statistical methods that have allowed this correlative method to be widely applied (*e.g.* Thuiller *et al.*, 2005b; Guisan and Thuiller, 2005; McClean *et al.*, 2005). Despite several limitations (4.3 and references cited therein) these models offer the advantage of assessing climate change impacts on biodiversity quantitatively (*e.g.* Thomas *et al.*, 2004a). Climate envelope models do not simulate dynamic population or migration processes, and results are typically constrained to the regional level, so that the implications for biodiversity at the global level are difficult to infer (Malcolm *et al.*, 2002a).

In modelling ecosystem function and plant functional type response, understanding has deepened since the TAR, though consequential uncertainties remain. 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 (*i.e.*, biomass distribution and leaf area index) are now explicitly modelled and have been bolstered by experimental results (*e.g.* Woodward and Lomas, 2004b). One emerging key message is that climate change impacts on the fundamental, regulating services may previously have been underestimated (4.4.1, 4.4.10, Figures 4.2, 4.4). Nevertheless, the globally applicable DGVMs are limited inasmuch as the few plant functional types used within the models aggregate numerous species into single entities (Sitch *et al.*, 2003). These are assumed to be entities with very broad environmental tolerances, which are immutable and immune to extinction. Therefore, underlying changes in species richness are not accounted for, and the simultaneous free dispersal of PFTs assumed (*e.g.* Neilson *et al.*, 2005; Midgley *et al.*, 2007). The strength of DGVMs is especially in their global application, realistic dynamics and simulation of ecosystem processes including essential elements of the global C-cycle (*e.g.* Malcolm *et al.*, 2002b). Thus, it is reasonable to equate changes in DGVM-simulated vegetation (*e.g.* Figure 4.3) to changes in community and population structures in the real world.

What overall picture emerges from the results reviewed here? It appears that moderate levels of atmospheric CO_2 rise and climate change relative to current conditions may be beneficial in some regions (Nemani *et al.*, 2003), depending on latitude, on the CO_2 responsiveness of plant functional types, and on the natural adaptive capacity of indigenous biota (mainly through range shifts that are now being widely observed, this volume, 1). But as change continues, greater impacts are projected, while ecosystem and species response may be lagged (4.4.5, 4.4.6). At key points in time (Figure 4.4) ecosystem services such as carbon sequestration may cease, and even reverse. While such “tipping points” (Kemp, 2005) are not possible 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 (*e.g.* soils, corals).

In the two simulations presented in Figure 4.2 (warming of 2.9ºC and 5.3ºC by 2100 over land relative to 1961-1990 baseline) the DGVM approach reveals salient changes in a key regulating service of the world’s ecosystems, carbon sequestration. Changes in the spatial distributions of ecosystems are given in Figure 4.3. (where it must be stressed that the figure highlights only key vulnerabilities through depicting appreciable vegetation type changes, *i.e.* PFT change over >20% of the area of any single pixel modelled). In the B1 scenario (Figure 4.3b) ~26% of extant ecosystems reveal appreciable changes by 2100, with some positive impacts especially in Africa and the southern Hemisphere. However, these positive changes are likely due to the assumed CO_2 fertilization effect (4.4.10, Figure 4.3). By contrast, 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 ~2070 (Figure 4.2, Figure 4.3). In the A2 scenario roughly 37% of extant ecosystems reveal appreciable changes by 2100. Desert amelioration persists in the regions described above, but substantial decline of forest and woodland is seen at northern, tropical and subtropical latitudes. In both scenarios the current global sink deteriorates after 2030, and by 2070 (ΔT ~2.5°C over pre-industrial) the terrestrial biosphere becomes an increasing carbon source (Figure 4.2; see also Scholze *et al.*, 2006) with the concomitant risk of positive feedback. Similar results were obtained by using a wide range of climate models which indicate that the biosphere becomes consistently within this century a net CO_2 source with a global warming of >3°C relative to pre-industrial (Scholze *et al.*, 2006). On the other hand it must be noted that by ~2100 the modelled biosphere has stored an additional 205–228 PgC (A2 and B1 respectively) relative to the year 2000 (Lucht *et al.*, 2006).

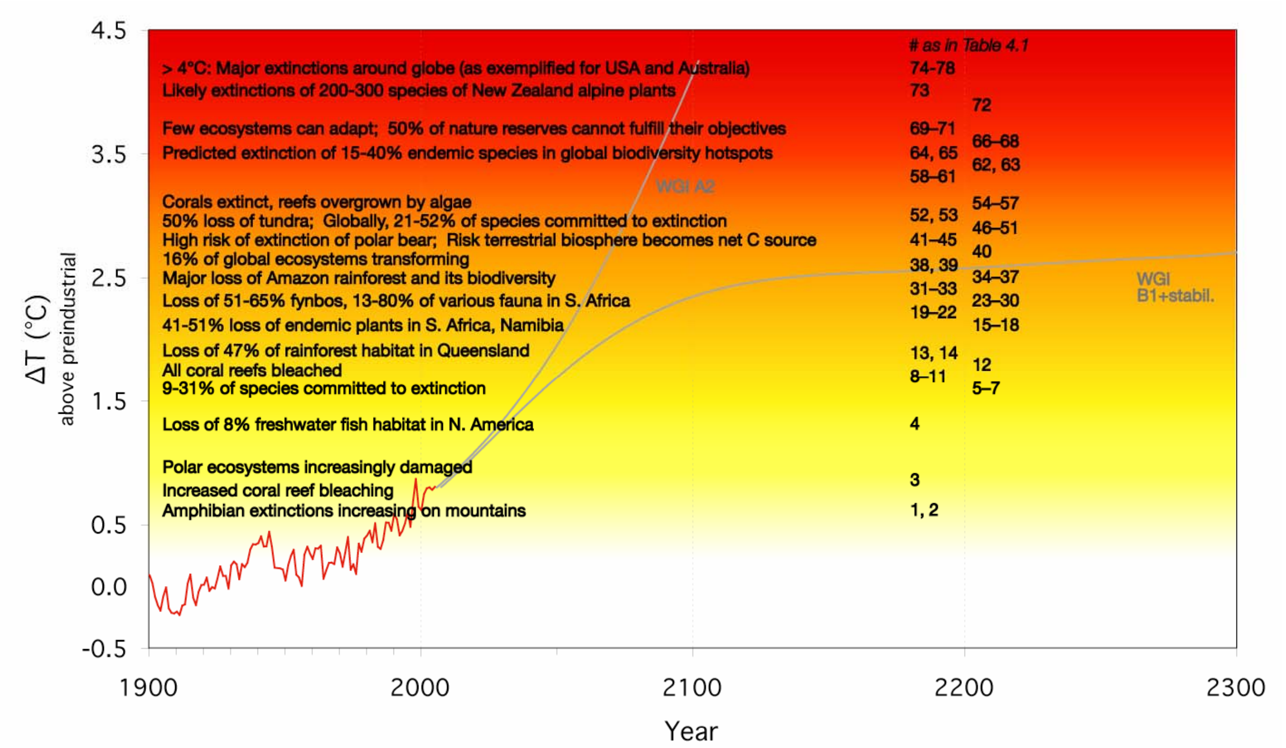

Figure 4.4: *Compendium of projected risks due to critical climate change impacts on ecosystems for different levels of global mean annual temperature rise, ΔT, relative to pre-industrial climate (approach and event numbers as used in Table 4.1). It is important to note that these impacts do not take 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* CO_2*. The red curve shows observed temperature anomalies for the period 1900-2005 (Brohan et al., 2006, see also WGI, Figure 3.6). The two grey curves provide examples of the possible future evolution of temperature against time (see WGI, Figure 10.4), providing examples of higher and lower trajectories for the future evolution of the expected value of ΔT. Shown are the simulated, multi-model mean responses to (i) the A2 radiative forcing scenario and (ii) an extended B1 scenario, where radiative forcing beyond the year 2100 was kept constant to the 2100 value (all data from WGI, Figure 10.4, see also WGI, Section 10.7).*

Climate envelope modelling suggests that climate change impacts will diminish the areal extent of some ecosystems (*e.g.* reduction by 2-47% alone due to 1.6°C warming above pre-industrial, Table 4.1 #6) and impact many ecosystems properties and services globally. Climate impacts alone will vary regionally and across biomes and will lead to increasing levels of 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 with 2.2°C (Table 4.1 #20) or major losses of Amazon rainforest with 2.5°C warming above pre-industrial, Figure 4.4, Table 4.1 #36). Globally, biodiversity (represented by species richness and relative abundance) may decrease by 13-19% due to a combination of land use change, climate change and nitrogen deposition under four scenarios by 2050 relative to species present in 1970 (Duraiappah *et al.*, 2005). Looking at projected losses due to land use change alone (native habitat loss), habitat reduction in tropical forests and woodland, savanna and warm mixed forest account for 80% of the species projected to be lost (~30,000 species Sala, 2005). The apparent contrast between high impacts shown by projections for species (climate envelope models) relative to PFTs (DGVMs) are likely to be due to a number of reasons – most importantly, real species virtually certainly have narrower climate tolerances than PFTs, a fact more realistically represented by the climate envelope models. DGVM projections reveal some increasing success of broad-range, generalist plant species, while climate envelope models results focus on endemics. Endemics, with their smaller ranges, have been shown to have a greater vulnerability to climate change (Thuiller *et al.*, 2005a), and may furthermore be dependent on keystone species in relationships that are ignored in DGVMs. Therefore, for assessing extinction risks, climate envelope modelling appears currently to offer more realistic results.

As indicated in the TAR, climate changes are being imposed on ecosystems experiencing other substantial and largely detrimental, pressures. Roughly 60% of evaluated ecosystems are currently utilized unsustainably and show increasing signs of degradation (Reid *et al.*, 2005; Hassan *et al.*, 2005; Worm *et al.*, 2006). This alone will likely cause widespread biodiversity loss (Chapin *et al.*, 2000; Jenkins, 2003; Reid *et al.*, 2005), given that 15,589 species, from every major taxonomic group, are already threatened with global extinction (Baillie *et al.*, 2006). The likely synergistic impacts of climate change and land-use change on endemic species has been widely confirmed (Warren, 2006; Thomas *et al.*, 2004a; Malcolm *et al.*, 2006; Lovejoy and Hannah, 2005; Leemans and Eickhout, 2004; Hughes, 2003; Hare, 2006; Hannah *et al.*, 2002a), as has overexploitation of marine systems (Worm *et al.*, 2006; this volume, 5, 6).

Overall, climate change has been estimated to be a major driver of biodiversity loss in cool conifer forests, savannas, mediterranean-climate systems, tropical forests, in the Arctic tundra, and in coral reefs (Carpenter *et al.*, 2005; Malcolm *et al.*, 2006; Thomas *et al.*, 2004a). In other ecosystems land use change may be a stronger driver of biodiversity loss at least in the near term. In an analysis of the SRES scenarios to 2100 (Strengers *et al.*, 2004) deforestation is reported to cease in all scenarios except A2, suggesting that beyond 2050 climate change is very likely to be the major driver for biodiversity loss globally. Due to climate change alone it has been estimated that by 2100 between 1% and 43% of endemic species (average 11.6%) will be committed to extinction (DGVM based study, Malcolm *et al.*, 2006), whereas following another approach (climate envelope modelling based study, Thomas *et al.*, 2004a) it has been estimated that on average 15% to 37% of species (combination of most optimistic assumptions 9%, most pessimistic 52%) will be committed to extinction by 2050 (*i.e.* that their range sizes will have begun shrinking and fragmenting in a way that guarantees their accelerated extinction). Climate-change induced extinction rates in tropical biodiversity hotspots are likely to exceed the predicted extinctions from deforestation during this century (Malcolm *et al.*, 2006). In the Mediterranean-climate region of South Africa, climate change may have at least as significant an impact on endemic Protea species extinction risk as land use change does by 2020 (Bomhard *et al.*, 2005). Based on all above findings and our compilation (Figure 4.4, Table 4.1) we estimate that on average 20% to 30% of species assessed may be at risk of extinction from climate change impacts within this century if global mean temperatures exceed 2° to 3°C relative to pre-industrial levels (this chapter). The uncertainties remain large, however, since for ~2°C the percentage may be as low as 10% or for ~3°C as high as 40%, and depending on biota the range is between 1% and 80% (Table 4.1 and Thomas *et al.*, 2004a; Malcolm *et al.*, 2006).

Losses of biodiversity will likely lead to decreases in the provision of ecosystem goods and services with trade-offs between ecosystem services likely to intensify (Figure 1; National Research Council, 1999; Duraiappah *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 (Duraiappah *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 could also entail sudden shifts of ecosystems to less desired states (Scheffer *et al.*, 2001; Folke *et al.*, 2004; *e.g.* Chapin *et al.*, 2004), through for example, the exeedance of critical temperature thresholds, possibly resulting in the irreversible loss of ecosystem services, which were dependent on the previous state (Reid *et al.*, 2005).

Table 4.1: *Projected impacts of climate change on ecosystems and population systems as reported in the literature for different levels of global mean annual temperature rise,* ΔT_g*, relative to pre-industrial climate – mean and range (event numbers as used in Figure 4.4). The table temperature is used as an indicator of the other associated climate changes that match particular amounts of* ΔTg*, e.g., precipitation change, and where considered, change in the concentration of greenhouse gases in the atmosphere. Projections from the literature were harmonized into a common framework by down/upscaling (where necessary) from local to global temperature rise using multiple GCMs, 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 or particular GCM scenarios, many studies give only local temperature rises,* ΔT_{reg}*, and hence require upscaling. The 13 GCM output datasets used are taken from the IPCC DDC at http://ipcc-ddc.cru.uea.ac.uk.*

No.	ΔT_g above pre-ind[i]	ΔT_g above pre-ind[ii] (range)	ΔT_{reg} above 1990 (range)	Impacts to unique or widespread ecosystems or population systems	Region	Ref. #
1	0.6			Increased coral bleaching	Caribbean, Indian Ocean, Great Barrier Reef	2
2	0.6			Amphibian extinctions/extinction risks on mountains due to climate-change induced disease outbreaks	Costa Rica, Spain, Australia	52, 54
3	<1			Marine ecosystems affected by continued reductions in krill possibly impacting Adelie penguin populations; Arctic ecosystems increasingly damaged	Antarctica, Arctic	42, 11, 14
4	1.3	1.1-1.6	1	8% loss freshwater fish habitat, 15% loss in Rocky Mountains, 9% loss of salmon	N America	13
5	1.6	1.2-2.0	0.7-15	9-31% (mean 18%) of species committed to extinction	Globe	1
6	1.6			Bioclimatic envelopes eventually exceeded leading to 10% transformation of global ecosystems; loss of 47% wooded tundra, 23% cool conifer forest, 21% scrubland, 15% grassland/steppe, 14% savanna, 13% tundra and 12% temperate deciduous forest. Ecosystems variously lose 2-47% areal extent.	Globe	6
7	1.6	1.1-2.1	1	Suitable climates for 25% of eucalypts exceeded	Australia	12
8	1.7	1-2.3	1°C SST	All coral reefs bleached	Great Barrier Reef, SE Asia, Caribbean	2

[i] The mean temperature change is taken directly from the literature, or is the central estimate of a range given in the literature, or is the mean of upscaling calculations. For further details see caption.

[ii] The range of temperature change represents the uncertainty arising from the use of different GCM models to calculate global temperature change.

9	1.7	1.2-2.6		38-45% of the plants in the Cerrado committed to extinction	Brazil	1, 44
10	1.7	1.3-3		2-18% of the mammals, 2-8% of the birds and 1-11% of the butterflies committed to extinction	Mexico	1, 26
11	1.7	1.3-2.4	2	16% freshwater fish habitat loss, 28% loss in Rocky Mountains, 18% loss of salmon	N America	13
12	<1.9	<1.6-2.4	<1	Range loss begins for Golden Bowerbird	Australia	4
13	1.9	1.6-2.4	1	7-14% of reptiles, 8-18% of frogs, 7-10% of birds, and 10-15% of mammals committed to extinction as 47% of appropriate habitat in Queensland lost	Australia	1, 7
14	1.9	1.6-2.4	1	Range loss of 40-60% for Golden Bowerbird	Australia	4
15	2.1			41-51% loss in plant endemic species richness	S Africa, Namibia	39
16	2.1	1.0-3.2	1-2	Alpine systems in Alps can tolerate local temperature rise of 1-2°C, tolerance likely negated by land use change	Europe	8
17	2.1		1.4-2.6	13-23% of butterflies committed to extinction	Australia	1, 30
18	2.1	1.4-2.6		Bioclimatic envelopes of 2-10% plants exceeded leading to endangerment or extinction; mean species turnover of 48% (spatial range 17-75%); mean species loss of 27% (spatial range 1- 68%)	Europe	22
19	2.2	1.5-2.8		Most areas experience 8-20% increase in number ≥7day periods with Forest Fire Weather Index > 45: increased fire frequency converts forest & Macquis to scrub, leads to more pest outbreaks	Mediterranean	34
20	2.2			3-16% of plants committed to extinction	Europe	1, 16
21	2.2	2.1-2.3	1.6-1.8	15-37% (mean 24%) of species committed to extinction	Globe	1
22	2.2	1.7-3.2		8-12% of 277 medium/large mammals in 141 national parks critically endangered or extinct; 22-25% endangered	Africa	23
23	2.3	1.5-2.7	2°C SST	Loss of Antarctic bivalves and limpets	Southern Ocean	4
24	2.3	2.0-2.5		Fish populations decline, wetland ecosystems dry and disappear	Malawi, Afr., Great Lakes	20
25	2.3	1.5-2.7	2.5-3.0	Extinctions (100% potential range loss) of 10% endemics; 51-65% loss of Fynbos; including 21-40% of *Proteaceae* committed to extinction; Succulent Karoo area reduced by 80%, threatening 2800 plant species with extinction; 5 parks lose > 40% of plant species	S Africa	1,5,24, 25
26	2.3	2.3-4.0	2.5-3.0	24-59% of mammals, 28-40% of birds, 13-70% of butterflies, 18-80% of other invertebrates, 21-45% of reptiles committed to extinction; 66% of animal species potentially lost from Kruger National Park	S Africa	1, 27
27	2.3	2.2-4.0		2-20% of mammals, 3-8% of birds and 3-15% of butterflies committed to extinction	Mexico	1, 26

28	2.3	1.6-3.2		48-57% of Cerrado plants committed to extinction	Brazil	1
29	2.3			Changes in ecosystem composition, 32% of plants move from 44% of area with potential extinction of endemics	N Europe	16
30	2.3	1.6-3.2	3	24% loss freshwater fish habitat, 40% loss in Rocky Mountains, 27% loss of salmon.	N America	13
31	2.4			63 of 165 rivers studied lose >10% of their fish species	Globe	19
32	2.4			Bioclimatic range of 25-57% (full dispersal) or 34-76% (no dispersal) of 5,197 plant species exceeded	Subsaharan Africa	60
33	>2.5			Sink service of terrestrial biosphere saturates and begins turning into a net carbon source	Globe	58,59
34	2.5		2°C SST	Extinction of coral reef ecosystems (overgrown by algae)	Indian Ocean	9
35	2.5	1.9-4.3		42% of UK land area with bioclimate unlike any currently found there; in Hampshire, declines in curlew and hawfinch and gain in yellow-necked mouse numbers; loss of montane habitat in Scotland; potential bracken invasion of Snowdonia montane areas		57
36	2.5	2.0-3.0		Major loss of Amazon rainforest with large losses of biodiversity	S America, Globe	21, 46
37	2.5			20-70% loss (mean 44%) of coastal bird habitat at 4 sites	USA	29
38	2.6	1.6-3.5		Most areas experience 20-34% increase in number ≥7day periods with Forest Fire Weather Index > 45: increased fire frequency converts forest & Macquis to scrub, causes more pest outbreaks	Mediterranean	34
39	2.6			4-21% of plants committed to extinction	Europe	1
40	2.7			Bioclimatic envelopes exceeded leading to eventual transformation of 16% of global ecosystems: loss of 58% wooded tundra, 31% cool conifer forest, 25% scrubland, 20% grassland/steppe, 21% tundra, 21% temperate deciduous forest, 19% savanna. Ecosystems variously lose 5-66% of their areal extent	Globe	6
41	2.8	1.2-4.5	1-3	Extensive loss/conversion of habitat in Kakadu wetland due to sea level rise and saltwater intrusion	Australia	10
42	2.8	2.5-3.0		Multimodel mean 62% (range 40-100%) loss Arctic summer ice extent, high risk of extinction of polar bears, walrus, seals; Arctic ecosystem stressed	Arctic	11,54
43	2.8	2.3-4.6	2.1-2.5	Cloud forest regions lose hundreds of metres of elevational extent, potential extinctions ΔT_{reg}2.1°C for C America and ΔT_{reg} 2.5°C for Africa	C. America, Tropical Africa, Indonesia	17
44	2.8	2.1-3.1	3	Eventual loss of 9-62% of the mammal species from Great Basin montane areas	USA	32
45	2.8	1.9-3.8	3	38-54% loss of waterfowl habitat in prairie pothole region	USA	37,38

46	2.9		3.2-6.6	50% loss existing tundra offset by only 5% eventual gain; millions of Arctic nesting shorebirds species variously lose up to 5-57% of breeding area; high Arctic species most at risk; geese species variously lose 5-56% of breeding area	Arctic	14
47	2.9			Lat. of N forest limits shifts N by 0.5° latitude in W Europe, 1.5° in Alaska, 2.5° in Chukotka and 4° in Greenland	Arctic	40
48	2.9	1.6-4.1		Threat of marine ecosystem disruption through loss of aragonitic pteropods	S Ocean	49
49	2.9	1.6-4.1		70% reduction in deep-sea cold-water aragonitic corals	Ocean Basins	48
50	2.9		2.1-3.9	21-36% of butterflies committed to extinction; >50% range loss for 83% of 24 lat. restricted species	Australia	1,30
51	2.9	2.6-3.3	2.1-2.8	21-52% (mean 35%) of species committed to extinction	Globe	1
52	3			66 of 165 rivers studied lose >10% of their fish species	Globe	19
53	3.0	1.9-3.5		20% loss of coastal migratory bird habitat	Delaware, USA	35, 36
54	3.1	2.3-3.7	2°C SST	Extinction of remaining coral reef ecosystems (overgrown by algae)	Globe	2
55	3.1	1.9-4.1	3-4	Alpine systems in Alps degraded	Europe	8
56	3.1	2.5-4.0	2	High risk of extinction of Golden Bowerbird as habitat reduced by 90%	Australia	4
57	3.1	1.8-4.2	3-4	Risk of extinction of Alpine species	Europe	41
58	3.3	2.0-4.5		Reduced growth in warm water aragonitic corals by 20%-60%; 5% decrease in global phytoplankton productivity	Globe	2,47,48
59	3.3	2.3-3.9	2.6-2.9	Substantial loss of alpine zone, and its assoc. flora and fauna (e.g., alpine sky lily, and mountain pygmy possum)	Australia	45
60	3.3	2.8-3.8	2	Risk of extinction of Hawaiian honeycreepers as suitable habitat reduced by 62-89%	Hawaii	18
61	3.3		3.7	4-38% of birds committed to extinction	Europe	1
62	3.4			Substantial loss of boreal forest	China	15
63	3.4			6-22% loss of coastal wetlands; large loss migratory bird habitat particularly in USA, Baltic and Mediterranean	Globe	35, 36
64	3.5	2.0-5.5		Predicted extinction of 15-40% endemic species in global biodiversity hotspots (narrow specificity)	Globe	50
65	3.5	2.3-4.1	2.5 – 3.5	Loss of temperate forest wintering habitat of Monarch butterfly	Mexico	28
66	3.6	2.6-4.3	3	Bioclimatic limits of 50% of eucalypts exceeded	Australia	12
67	3.6	2.6-3.7		30-40% of 277 mammals in 141 parks critically endangered/extinct; 15-20% endangered	Africa	23

68	3.6	3.0-3.9		Parts of the USA lose 30-57% neotropical migratory bird species richness	USA	43
69	3.7			Few ecosystems can adapt	Globe	6
70	3.7			50% all nature reserves cannot fulfil conservation objectives	Globe	6
71	3.7			Bioclimatic envelopes exceeded leading to eventual transformation of 22% of global ecosystems; loss of 68% wooded tundra, 44% cool conifer forest, 34% scrubland, 28% grassland/steppe, 27% savanna, 38% tundra and 26% temperate deciduous forest. Ecosystems variously lose 7-74% areal extent.	Globe	6
72	3.9			4-24% plants critically endangered/extinct; mean species turnover of 63% (spatial range 22-90%); mean species loss of 42% (spatial range 2.5-86%)	Europe	22
73	4.0	3.0-5.1	3	Likely extinctions of 200-300 species (32-63%) of alpine flora	New Zealand	33
74	>4.0		3.5	38-67% of frogs, 48-80% of mammals, 43-64% of reptiles and 49-72% of birds committed to extinction in Queensland as 85-90% of suitable habitat lost	Australia	1, 7
75	>>4.0		5	Bioclimatic limits of 73% of eucalypts exceeded	Australia	12
76	>>4.0		5	57 endemic frogs/mammals eventually extinct, 8 endangered	Australia	7
77	>>4.0		7	Eventual total extinction of all endemic species of Queensland rainforest	Australia	7
78	5.2			62-100% loss of bird habitat at 4 major coastal sites	USA	29

Sources: 1-Thomas *et al.*, 2004a; 2-Hoegh-Guldberg, 1999; 4-Hilbert *et al.*, 2004; 5-Rutherford *et al.*, 2000; 6-Leemans and Eickhout, 2004; 7-Williams *et al.*, 2003; 8-Theurillat and Guisan, 2001; 9-Sheppard, 2003; 10-Eliot *et al.*, 1999; 11-Symon *et al.*, 2005; 12-Hughes *et al.*, 1996; 13-Preston, 2006; 14-Zöckler and Lysenko, 2000; 15-Ni, 2001; 16-Bakkenes *et al.*, 2002; 17-Still *et al.*, 1999; 18-Benning *et al.*, 2002; 19-Xenopoulos *et al.*, 2005; 20-ECF, 2004; 21-Cox *et al.*, 2004; 22-Thuiller *et al.*, 2005b; 23-Thuiller *et al.*, 2006b; 24-Midgley *et al.*, 2002; 25-Hannah *et al.*, 2002a; 26-Peterson *et al.*, 2002; 27-Erasmus *et al.*, 2002; 28-Villers-Ruiz and Trejo-Vazquez, 1998; 29-Galbraith *et al.*, 2002; 30-Beaumont and Hughes, 2002; 31-Kerr and Packer, 1998; 32-McDonald and Brown, 1992; 33-Halloy and Mark, 2003; 34-Moriondo *et al.*, 2006; 35-Nicholls *et al.*, 1999; 36-Najjar, 2000; 37-Sorenson *et al.*, 1998; 38-Johnson *et al.*, 2005; 39-Broennimann *et al.*, 2006; 40-Kaplan *et al.*, 2003; 41-Theurillat *et al.*, 1998; 42-Forcada *et al.*, 2006; 43-Price and Root, 2005; 44-Siqueira and Peterson, 2003; 45-Pickering *et al.*, 2004; 46-Scholze *et al.*, 2006; 47-Raven *et al.*, 2005; 48-Cox *et al.*, 2000; 49-Orr *et al.*, 2005; 50-Malcolm *et al.*, 2006; 51-Peck *et al.*, 2004; 52-Pounds *et al.*, 2006; 53-Arzel *et al.*, 2006; 54-Bosch *et al.*, 2006; 57-Berry *et al.*, 2005; 58-Lucht *et al.*, 2006; 59-Schaphoff *et al.*, 2006; 60-McClean *et al.*, 2005.

There is detailed information on the derivation for each entry in Table 4.1 in Appendix 4.1

4.5 Costs and valuation of ecosystem goods and services

There is growing interest in developing techniques for environmental accounting. To that end, definitions of ecosystem goods and services are currently fluid. For example, ecosystem services accrue to society in return for investing in or conserving natural capital (Heal, 2006), or ecosystem services are ultimately the end products of nature, the aspects of nature that people make choices about (Boyd, 2006). Definitions aside, all humans clearly rely on ecosystem services (Reid *et al.*, 2005). While many efforts have been made to use standard economic techniques to estimate the economic value of ecosystem goods and services (Costanza *et al.*, 1997; Costanza, 2000; Costanza *et al.*, 2000; Daily *et al.*, 2000; Costanza, 2001; 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 actually provide society a disservice (Ludwig, 2000; Kremen, 2005). The estimates from these techniques range from unknown (incomparability cf. Chang, 1997) or invaluable, or infinite (Toman, 1998b) because of lack of human substitutes, to ~38x10^{12} 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 x10^{12} USD/a (2000 levels). These monetary estimates are usually targeted at policy-makers to assist assessments of the economic benefits of the natural environment (Farber *et al.*, 2006) in response to cost-benefit paradigms. Some argue (Balmford *et al.*, 2002; Reid *et al.*, 2005; Balmford *et al.*, 2005) that unless ecosystem values are recognized in economic terms, ecosystems will continue their decline, placing the planet's ecological health at stake (Millennium Ecosystem Assessment, 2005). Others argue that ecosystems provide goods and services which are invaluable and need to be conserved on more fundamental principles, *i.e.* the precautionary principle 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) or a moral and ethical responsibility to natural systems not to destroy them.

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 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, 2005; Mock, 2005 p.33ff.). Ignoring such aspects almost guarantees opportunity costs. 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. Additionally, many sectors and industries depend directly or indirectly on ecosystems and their services. The impacts of climate change could hold enormous costs for forests and coastal marine systems, as well as managed agricultural systems (Epstein and Mills, 2005; Stern *et al.*, 2006). Multiple industries, such as timber, fisheries, travel, tourism and agriculture, are threatened by disturbances caused by climate change. Impacts on these sectors will influence financial markets, insurance companies and large multinational investors (Mills, 2005).

The United Nations have recognized the need to develop integrated environmental and economic accounting. However, many difficulties remain, especially as ecosystems may be the most difficult of all environmental assets to quantify (Boyd, 2006). There is a growing recognition that national accounting procedures need to be modified to include values for ecosystem goods and services (Heal, 2006). Outside of the techniques mentioned above (often using contingent valuation) others have argued for developing a Green GDP to describe the state of nature and its worth, or an Ecosystem Services Index to account for all of nature's contributions to the welfare of human society (Banzhaf and Boyd, 2005; Boyd, 2006). Ultimately, it may be developing economies that are the most sensitive to the direct impacts of climate change because they are more dependent on ecosystems and agriculture (Stern *et al.*, 2006). As such, it is the poor that depend most directly on ecosystem services. Thus the degradation of these systems and their services will ultimately exacerbate poverty, hunger, and disease and obstruct sustainable development (*e.g.* Mooney *et al.*, 2005; Millennium Ecosystem Assessment, 2005; Mock, 2005; Stern *et al.*, 2006).

4.6 Acclimation and adaptation: practices, options, and constraints

Although climate change is a global issue, local efforts can help maintain and enhance resilience and limit some of the longer-term damages from climate change (*e.g.* Hughes *et al.*, 2003; Opdam and Wascher, 2004; Singh, 2003). This section discusses adaptation options with respect to natural ecosystems. Adaptation of these ecosystems involves only reactive, autonomous responses to ongoing climate change, including changes in weather variability and extremes. However, ecosystem managers can proactively alter the context in which ecosystems develop. In this way they can improve the resilience, *i.e.* the coping capacity of ecosystems (see glossary). Such ecosystem management involves anticipatory adaptation options. Identifying adaptation responses and adaptation options is a rapidly developing field, so the discussion below is not exhaustive. However, one should realize that beyond certain levels of climate change (Hansen *et al.*, 2003; Table 4.1, Figure 4.4) impacts on ecosystems are severe and largely irreversible.

4.6.1 Adaptation options

As climatic changes occur, natural resource management techniques can be applied to increase the resilience of ecosystems. Increasing resilience is consistent also with the "ecosystem approach" developed by the Convention on Biological Diversity (CBD) which is a "strategy for management of land, water and living resources that promotes conservation and sustainable use in an equitable way" (Smith and Malthby, 2003). There are many opportunities to increase resilience (Tompkins and Adger, 2003; Cropp and Gabrica, 2002), however they may only be effective for lower levels of climate change (<~2-3°C, Executive Summary, Figure 4.4, Table 4.1).

Effective responses depend on an understanding of likely regional climatic and ecological changes. Monitoring environmental change, including climate, and associated ecosystem responses is vital to allow for adjustments in management strategies (*e.g.* Adger *et al.*, 2003; Moldan *et al.*, 2005). Although many adaptation options are available to wildlife managers, uncertainty about the magnitude and timing of climate change and delayed ecosystem responses (*e.g.* 4.4.5) may discourage their application. Nevertheless, 'no regrets' decisions based on the 'precautionary principle' appear preferable. Actions to reduce the impact of other threats, such as habitat fragmentation or destruction, pollution and introduction of alien species, are very likely to enhance resilience to climate change (*e.g.* Opdam and Wascher, 2004; Goklany, 1998; Inkley *et al.*, 2004). Such proactive 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 possible abrupt climate change by increasing ecosystem resilience (Bharwani *et al.*, 2005).

A few key options to adapt at least to lower levels of climate change in intensively managed ecosystems (this volume, 5) have been suggested (*e.g.* Hannah *et al.*, 2002b; Hannah *et al.*, 2002a; Hannah and Lovejoy, 2003; Hansen *et al.*, 2003). Expansion of reserve systems can potentially reduce the vulnerability of ecosystems to climate change (McNeely and Schutyser, 2003). Reserve systems may be designed with some consideration of long-term shifts in plant and animal distributions, natural disturbance regimes, and the overall integrity of the protected species and ecosystems (*e.g.* Williams *et al.*, 2005). Ultimately, adaptation possibilities are determined by the conservation priorities of each reserve and by the magnitude and nature of the change in climate. Strategies to cope with climate change are beginning to be considered in conservation (Cowling *et al.*, 1999; Chopra *et al.*, 2005; Scott and Lemieux, 2005), and highlight the importance of planning guided by future climate scenarios.

A primary adaptation strategy to climate change and even current climate variability is to reduce and manage the other stresses on species and ecosystems, such as habitat fragmentation and destruction, overexploitation, eutrophication, desertification and acidification (Duraiappah *et al.*, 2005; Inkley *et al.*, 2004; Robinson *et al.*, 2005; Worm *et al.*, 2006). Robinson *et al.* (2005) suggest that this may be the only practical large-scale adaptation policy available for marine ecosystems. In addition to removing other stressors it is necessary to maintain viable, connected, and genetically diverse populations (Inkley *et al.*, 2004; Robinson *et al.*, 2005). Isolated, small populations are often more prone to local extirpations than larger, more widespread populations (*e.g.* Gitay *et al.*, 2002; Davis *et al.*, 2005; Lovejoy and Hannah, 2005). Although connectivity, genetic diversity and population size are important current conservation goals, climate change increases their importance. The reduction and fragmentation of habitats may also be facilitated through increases in agricultural productivity (*e.g.* Goklany and Trewavas, 2003) reducing pressures on natural ecosystems. However, increasing demand for some types of biofuels may negate this potential benefit (*e.g.* Busch, 2006).

Reducing stress on ecosystems is difficult especially in densely populated regions. Recent studies in southern Africa have signalled the need for policy to focus on managing areas outside protected areas (*e.g.*, subsistence rangelands; Von Maltitz *et al.*, 2006). 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). Species migration between protected areas in response to shifting climatic conditions is likely to be impeded, unless assisted by often costly interventions geared towards landscapes with greater ecological connectivity. Strategic national policies could co-ordinate with communal or private land-use systems, especially when many small reserves are involved and would be particularly cost-effective if they address climate change proactively. Finally, migration strategies are very likely to become substantially more effective when they are implemented over larger regions and across national borders (*e.g.* Hansen *et al.*, 2003).

Controlled burning and other techniques may be useful to reduce fuel load and the potential for catastrophic wildfires. 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 (*e.g.*, hurricane, typhoon, flood).

Climate change is likely to increase opportunities for invasive alien species because of their adaptability to disturbance (Lake and Leishman, 2004; Stachowicz *et al.*, 2002). Captive breeding for reintroduction and translocation or the use of provenance trials in forestry are expensive and 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 creation of new habitats in areas where natural colonization is unlikely to occur (Anonymous, 2000). In many cases the knowledge of ecosystem interactions and species requirements may be lacking. Engineering habitats to facilitate species movements may call for an entirely new field of study. Engineering interactions to defend coastlines, for example, that change the connectivity of coastal ecosystems facilitate the spread of non-native species (Bulleri, 2005) as well as warm temperate species advancing polewards (Mieszkowska *et al.*, 2006; Helmuth *et al.*, 2006).

Ultimately, managers may need to enhance or replace diminished or lost ecosystem services. This could mean manual seed dispersal or reintroducing pollinators. 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 is likely to be a major source of "surprises" from climate change.

4.6.2 Assessing the effectiveness and costs of adaptation options

There are few factual studies that have established the effectiveness and costs of adaptation options in ecosystems. Unfortunately, this makes 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 certainly large. For example, the money spent annually on nature conservation in the Netherlands was recently estimated to be 1 billion Euros (Milieu en Natuurplanbureau, 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; Thomas *et al.*, 2004a; Lovejoy and Hannah, 2005; Thuiller *et al.*, 2005b; Schröter *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 extends beyond UNFCCC's objective of 'avoiding dangerous human interference with the climate system at levels where ecosystems cannot adapt naturally'. The main tool proposed by the CBD is the ecosystem approach (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 Ecosystem Approach resembles sustainable forest management projects (FAO, 2001). In theory, the Ecosystem Approach 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

Formulating integrated policies that cut across multiple UN conventions, such as UNFCCC, CBD and Convention to Combat Desertification (CCD) could produce win-win situations in addressing climate change, increasing resilience and dealing with other policy issues (Nnadozie, 1998). Strategies aimed at combating desertification, for example, contribute towards increased soil carbon and moisture levels. Mitigation strategies focussed at afforestation, including such projects under the Clean Development Mechanism (CDM, see glossary), could help ecosystem adaptation through improved ecological connectivity. The Ecosystem Approach can fulfil objectives specified by different conventions (Reid *et al.*, 2005), and in assessing adaptation strategies, such synergies could be identified and promoted.

4.7 Implications for sustainable development

Over the past 50 years, humans have converted and modified natural ecosystems more rapidly and over larger areas than in any comparable period of human history (*e.g.* Steffen *et al.*, 2004). These changes have been driven by the rapidly growing demands for food, fish, fresh water, timber, fibre, and fuel (*e.g.* Vitousek *et al.*, 1997) and have contributed to substantial net gains in human well-being and economic development, while resulting in a substantial and largely irreversible loss of biodiversity and degradation in ecosystems and their services (Reid *et al.*, 2005).

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, while designed to increase food self-sufficiency also contribute to reducing carbon sequestration and loss of biodiversity through extensive clear-cutting.

Effective mechanisms to analyze cross-sectoral impacts and to feed new scientific knowledge into policy-making are necessary (Schneider, 2004). 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 (*e.g.* population growth, poverty and globalization) that contribute to chronic vulnerability to, for example, drought and floods are not resolved (Kates, 2000; Reid *et al.*, 2005).

4.7.1 Ecosystems services and sustainable development

Large differences in natural and socio-economic conditions among regions mitigate against simple solutions to the problem of ecosystem degradation and loss of services. 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). However, sound management of ecosystem services provides several cost-effective opportunities for addressing multiple development goals in a synergistic manner (Reid *et al.*, 2005).

Progress achieved in addressing the Millennium Development Goals (MDGs) is unlikely to be sustained if ecosystem services continue to be degraded (Goklany, 2005). The role of ecosystems in sustainable development and in achieving the MDGs involves an array of stakeholders (Jain, 2003; 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 of 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, mountain and Mediterranean regions are likely to be more vulnerable than others (Gitay *et al.*, 2001) 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 (Stern *et al.*, 2006; 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. 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

Key uncertainties listed here are those that limit our ability to project climate change impacts on ecosystems, but only if they have implications at sub-continental and higher spatial scale, are relevant for many species, populations, and communities, or significantly weaken a modelling result. In terms of climate uncertainty, it is important to highlight that projections for precipitation carry a significantly higher uncertainty than temperature, yet play a major role for many projections obtained from modelling approaches. In relation to projecting climate change impacts on ecosystems, we find key sources of uncertainty to include: (i) inadequate representation of the interactive coupling between ecosystems and the climate system, and furthermore, of the multiple interacting drivers of global change. This prevents a fully integrated assessment of climate change impacts on ecosystem services; (ii) major biotic feedbacks to the climate system, especially through trace gases from soils in all ecosystems, and methane from labile carbon stocks such as wetlands, peatlands, permafrost and yedoma (see glossary); (iii) how aggregation within current DGVMs (see glossary) with respect to the functional role of individual species and the assumption of their instantaneous migration biases impact estimates; (iv) the net result of changing disturbance regimes (especially through fire, insects and land-use change) on biotic feedbacks to the atmosphere, ecosystem structure, function, biodiversity and ecosystem services; (v) the magnitude of the CO_2 fertilization effect in the terrestrial biosphere and its components over time; (v) the limitations of climate envelope models used to project responses of individual species to climate changes, and for deriving estimations of species extinction risks; (vi) the synergistic role of invasive alien species in both biodiversity and ecosystem functioning; (vii) the effect of increasing surface ocean CO_2 and declining pH on marine productivity, biodiversity, biogeochemistry and ecosystem functioning; (viii) the impacts of interactions between climate change and changes in human use and management of ecosystems as well as other drivers of global environmental change.

Guided by the above, the following research needs can be identified as priorities for reducing uncertainties:

- Identify key vulnerabilities in **permafrost–soil–vegetation interactions** at high latitudes, and their potential feedback to the biosphere trace gas composition. Recent estimates suggest that terrestrial permafrost contains more than 1,000 PgC, which is increasingly emitting CO_2 and more importantly, methane (*e.g.* Walter *et al.*, 2006; Zimov *et al.*, 2006). The implications of this for abrupt and significant climate forcing are significant (*e.g.* Schellnhuber, 2002; iLEAPS, 2005; Symon *et al.*, 2005, p. 1015; Lelieveld, 2006; Zimov *et al.*, 2006).
- More robust modelling of interactions between biota and their geophysical environment using several, independently developed **DGVM**s and earth system models. Validation (Price *et al.*, 2001) beyond model intercomparisons is required, especially also with respect to the methane cycle. The goal should be to narrow uncertainties relating to the vulnerability of the carbon sequestration potential of ecosystems including more realistic estimates of lagged and threshold 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. These should emphasize interactions between vegetation and atmosphere, including CO_2 fertilization effects, in mature forests in the Northern Hemisphere, seasonal tropical forests, and arid or semi-arid grassland and savannas (*e.g.* Jasienski *et al.*, 1998; Karnosky, 2003).
- Improved understanding of the role of **disturbance regimes**, *i.e.* frequency and intensity of episodic events (drought, fire, insect outbreaks, diseases, floods, and wind-storms) and that of alien species invasions, as they interact with ecosystem responses to climate change itself and pollution (*e.g.* Osmond *et al.*, 2004; Opdam and Wascher, 2004).
- Development of integrated **large spatial scale remote sensing with long-term field studies** (May, 1999b; Kräuchi *et al.*, 2000; Morgan *et al.*, 2001b; Osmond *et al.*, 2004; Opdam and Wascher, 2004; Symon *et al.*, 2005, p. 1019) to better address scale mismatches between the climate system and ecosystems (Root and Schneider, 1995).
- Studies on impacts of rising atmospheric CO_2 on **ocean acidification**, and warming on coral reefs and other marine systems (Coles and Brown, 2003; Anonymous, 2004), and widening the range of terrestrial ecosystems for which CO_2 fertilization responses have been quantified (*e.g.* Bond *et al.*, 2003).
- Validating species-specific **climate envelope models** by testing model projections against the plethora of range shifts observed in nature (*e.g.* Walther *et al.*, 2001; this volume, 1).
- Advances in understanding the relationship between **biodiversity** and the **resilience** (see glossary) of ecosystem services at a scale relevant to human well-being, to quote Sir Robert

May (1999a): "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".

- Improve identification of environmental key factors influencing ecosystem structures that determine functionality and provisioning services of ecosystems together with quantitative information on **economic impacts** (including implications for adaptation costs; Toman, 1998a; Winnett, 1998; Kremen, 2005; Symon *et al.*, 2005, *e.g.* p. 1019).
- **Integrative vulnerability** studies on adaptive management responses to preserve biodiversity (including conservation and reservation management) and ecosystem services in relation to pressures from land use change and climate change (Kappelle *et al.*, 1999; Lorenzoni *et al.*, 2005; Stenseth and Hurrell, 2005; Symon *et al.*, 2005).

Appendix 4.1

The table below contains detailed information on models and how the upscaling and downscaling were performed for each entry in Table 4.1 of Fischlin and Midgley (2007, in the following referred to as "this chapter", table below uses same numbering scheme as Table 4.1).

In each case **E** indicates an empirical derivation, **M** indicates a modeling study, a **number** refers to how many GCMs (see glossary) were used in the original literature (for GCM abbreviations used here see below) other codes indicate if model projections included respectively, precipitation (**P**), ocean acidification (**pH**), sea ice (**SI**), sea level rise (**SLR**), sea surface temperature (**SST**) or anthropogenic water use (**W**); dispersal assumptions from the literature (**D** - estimate assumes dispersal; **ND** - estimate assumes no dispersal; **NR** - not relevant since species/ecosystem has nowhere to disperse to in order to escape warming (e.g. habitat is at top of isolated mountain or at southern extremity of austral landmass)).

IMAGE, **BIOME4**, **LPJ**, **MAPSS** refer to specific models as used in the study, e.g. LPJ denotes the Lund-Potsdam-Jena dynamic global vegetation model (LPJ-DGVM, Sitch *et al.*, 2003, see also glossary).

Lower case a-h refers to how the literature was addressed in terms of up/downscaling (**a** - clearly defined global impact for a specific ΔT against a specific baseline, upscaling not necessary; **b** - clearly defined regional impact at a specific regional ΔT where no GCM used; **c** - clearly defined regional impact as a result of specific GCM scenarios but study only used the regional ΔT; **d** - as c but impacts also the result of regional precipitation changes; **e** - as b but impacts also the result of regional precipitation change; **f** - regional temperature change is off-scale for upscaling with available GCM patterns to 2100, in which case upscaling is, where possible, approximated by using Figure 10.3.2 and 10.3.5 from WGI, chapter 10; **g** - studies which estimate the range of possible outcomes in a given location or region considering a multi-model ensemble linked to a global temperature change. In this case upscaling is not carried out since the GCM uncertainty has already been taken into account in the original literature; **h** - cases where sea surface temperature is the important variable, hence upscaling has been carried out using the maps from WGI, chapter 10, using figures 10.3.2 and 10.3.5, taking the increases in local annual mean (or where appropriate seasonal, from Figure 10.3.6) surface air temperature over the sea as equal to the local increases in annual mean or seasonal sea surface temperature. GCM abbreviations used here: **H2** – HadCM2, **H3** – HadCM3, **GF** - GFDL, **EC** - ECHAM4, **CS** - CSIRO, **CG** - CG, **PCM** - NCAR PCM.

The GCM outputs used in this calculation are those used in the Third Assessment Report (IPCC, 2001) and are at 5 degree resolution: HadCM3 A1FI, A2, B1, B2 where A2 is an ensemble of 3 runs and B2 is an ensemble of 2 runs; ECHAM4 A2 and B2 (not ensemble runs); CSIRO mark 2 A2, B1, B2; NCAR PCM A2 B2; CGCM2 A2 B2 (each an ensemble of 2 runs). Where GCM scenario names only were provided further details were taken from: HadCM2/3 (Mitchell *et al.*, 1995), http://ipcc-ddc.cru.uea.ac.uk (see also Gyalistras *et al.*, 1994; Gyalistras and Fischlin, 1999; IPCC-TGCIA, 1999; Jones *et al.*, 2005). All used GCMs/AOGCMs have been reviewed here: (IPCC, 1990; IPCC, 1996; Neilson and Drapek, 1998; IPCC, 2001).

No. (#)[1]	Details on type of study, models, model results, and methods used to derive the sensitivities as tabulated in Table 4.1 for each table entry
1	M, 4, SST
2	E
3	E, SI
4,11,30	M, 5, ND, c; ref. quotes 13.8% loss in Rocky Mountains for each 1°C rise in JJA temperature, upscaled with CS, PCM, CG
5	M, D&ND, a; 18% matches minimum expected climate change scenarios which Table 3 of ref. (supplementary material) lists as ΔT of 0.9°-1.7°C (mean 1.3°C) above 1961-1990 mean; 8 of 9 sub-studies used H2
6	M, 5, IMAGE, a; authors confirmed temperature baseline is year 2000 which is 0.1°C warmer than 1990
7	M, D, b; no GCM used in ref.; upscaled with H3, EC, CS, PCM, CG
8	M, SST, h
9	M, H2, P, ND, d; table 3 of ref. 1 gives global ΔT of 1.35°C above 1961-1990; HHGSDX of H3; downscaled with H3 then upscaled with H3, EC, CS, PCM, CG
10	M, H2, P, D&ND, d; table 3 of ref. 1 gives global ΔT of 1.35°C above 1961-1990; HHGGAX of H3; downscaled with H3 then upscaled with H3, EC, CS, PCM, CG
11	As for #4
12,14	M, P, NR, e; upscaled using H3, EC, CS, PCM, CG
13	M, D, b; upscaled using H3, EC, CS, PCM, CG
14	As for #12
15	M, H3, P, D, d; H3 2050 SRES mean
16	E, P, D, b; upscaled using H3, EC, CS, PCM, CG
17	M, H2, P, D, d, g; table 3 of ref. 1 gives global ΔT of 1.35°C above 1961-1990; upscaled with H3, EC, CS, PCM, CG; Uses a local ΔT range across Australia
18	M, H3, P, D&ND, d; ref. uses B1 of H3 in 2050 with a ΔT of 1.8°C above the 1961-1990 baseline; downscaled with H3 and then upscaled with H3, EC, CG
19	M, P, NR, d; HadRM3PA2 in 2050, figure 13 in ref. shows ΔT matching B2 of H3 of 1.6°C above 1961-1990 mean; downscaled with H3 and upscaled with H3, EC, CS, PCM, CG
20	M, H2, P, D&ND, d; studies used global annual mean ΔT of 1.7-2.0°C above 1961-1990 mean
21	M, P, D&ND, a; table 3 of ref. mid-range climate scenarios have a mean ΔT of 1.9°C above 1961-1990
22	M, H2, P, D&ND, d; ref. refers to A2 of H3 in 2050 that has a ΔT of gives as 1.9°C above 1961-1990 (Arnell *et al.*, 2004); downscaled with H3 then upscaled with H3, EC, CS, PCM, CG

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[1] Same numbers as used in the chapter text, first column in Table 4.1

23	H; upscaled using maps from WGI, chapter 10
24	E, P, NR, a
25	M, 2, P, NR, d; scenarios on CRU website used with ΔT of 2.0°C above 1961-1990, agrees with Table 3 of ref. 1 which gives ΔT of 2.0°C above 1961-1990 mean; downscaled with H3 then upscaled with H3, EC, CS, PCM, CG
26	M, H2, P, D, d; the 66% is from a suite of 179 representative species, table 3 of ref. 1 lists global ΔT of 3°C above 1961-1990 mean, upscaled with H3, EC, CS, CG
27	M, H2, P, D&ND, d; table 3 of ref. 1 which gives ΔT of 2.0°C above 1961-1990 mean using HHGGAX; downscaled with H3 then upscaled with H3, EC, CS, PCM, CG
28	M, H2, P, ND, d; as for #27
29	M, IMAGE, P, D&ND; ref. gives the global temperature change relative to 1990
30	As for #4
31	M, H3, W, a; ref. uses B2 of H3 in 2070 that has a ΔT rise of 2.1°C with respect to the 1961-1990 mean
32	M, P, D&ND; ref. uses B1 in H3 in 2080s from (Arnell *et al.*, 2004)
33	M, 2, P, LPJ; upscaled with H3, EC5 (see also this chapter, figures 4.2 and 4.3)
34	M, SST, h
35	M, P, D, d; UKCIP02 high emission scenario used as central value; upscaled for Hampshire from UKCIP02 regional maps using H3, EC, CS
36	M, a
37	M, SLR, a; analysis based on transient 50% probability of sea level rise using the US EPA scenarios for ΔT of 2°C above 1990 baseline
38	M, P, NR, d; see #24; HadRM3PA2 in 2050, taken from Figure 13 in ref.
39	M, H2, D&ND, d; ref. uses global ΔT of 2.3°C above 1961-1990 mean; downscaled with H3 and upscaled with H3, EC, CG
40	M, 5, IMAGE, a; authors confirmed temperature baseline is year 2000 which is 0.1°C warmer than 1990
41	M, CS, b; upscaled with H3, EC, CS, PCM, CG
42	M, 15, SI, a; Arzel (Arzel *et al.*, 2006) uses 15 GCMs with A1B for 2080s, ΔT A1B 2080s multi-model from WGI, chapter 10, figure 10.3.2 is 2.5°C above 1990; ACIA uses 4 GCMs with B2, multi-model ΔT is 2.2°C over 1961-1990 or 2.0°C above 1990
43	M, GE, P, NR, d; GENESIS GCM with 2.5°C rise for CO_2 doubling from 345 to 690ppm, 345 ppm corresponds quite closely to the 1961-1990 mean; upscaling then gives the range all locations used variously used H3, EC, CS, CG
44	M, NR, b; upscaled with H3, EC, CS, and CG
45	M, 2, P, d, g; range is due to importance of Δ P, GFDL CO_2 doubling is from 300 ppm which is close to 1900 climate sensitivity in SAR is 3.7; UKMO in 2050 is 1.6°C above 1961-1990 mean, 1.9°C above preindustrial
46,47	M, H2, BIOME4, P, NR, c; A1 scenario of H2GS has ΔT of 2.6°C relative to 1961-1990 mean
48,49	pH, g; IS92a in 2100 has 788 ppm CO_2 and ΔT of 1.1-3.6°C above 1990
50	M, 10, P, D, d, g; 2.6°C above 1961-1990 mean.upscaled with H3, EC, CS, CG at lower end, upper end out of range
51	M, P, D, ND, a; Table 3 of ref. maximum climate scenarios have a mean ΔT of 2.6°C above 1961-1990 or 2.3°C above 1990

52	M, H3, W, a; ref. uses A2 of H3 in 2070 that has a ΔT of 2.7°C with respect to the 1961-1990 mean and hence 2.5°C with respect to 1990
53	M, H3, SLR, a; IS92a median ΔT 2.0°C above 1990 (IPCC, 1996, Figure 6.20) and range 1.4-3.0°C
54	M, SST, h
55	E, P, D, e; upscaled with H3, EC, CS
56	M, P, NR, e; upscaled for several sites taken from maps in ref., using H3, EC, CS, CG
57	M, NR
58	pH, a; impact is at CO_2 doubling, T range given by WGI for equilibrium climate sensitivity
59	M, CS, P, d; upscaled with H3, EC, CS, CG
60	M, NR, b; % derived from Table 1 in ref. for all forest areas combined on the 3 islands studied; upscaling considers changes averaged over 3 islands and uses H3, EC, CS, CG
61	M, H3, P, D&ND, d, f; table 3 of ref. lists global ΔT of 3°C above 1961-1990 mean
62	M, BIOME3, P, d, f; H2 2080s has global ΔT of 3.4°C above preindustrial (Hulme *et al.*, 1999)
63	M, H2, SLR, NR, a; see #62
64	M, 7, BIOME3, MAPSS, P, D&ND, a; uses CO_2 doubling scenarios from Neilson & Drapek (Neilson and Drapek, 1998) table 2; control concentrations were obtained directly from modellers; thus deduced mean global mean ΔT for this study
65	M, 2, P, D, d; study used CO_2 doubling scenarios - CCC ΔT at doubling is 3.5°C relative to 1900 whilst GFDL R30 is 3.3°C relative to 1900; upscaling gives range H3, EC, CG
66	M, D, b; no GCM used, upscaled with H3, EC, CS
67	M, H3, P, D&ND, d; ref. uses A2 in H3 in 2080 that has a ΔT of 3.3°C above 1961-1990 (Arnell *et al.*, 2004)
68	M, CCC, P, D, d; CO_2 equilibrium doubling scenario has ΔT of 3.5°C relative to 1900; downscaled with CGCM and upscaled with H3, EC, CS, CG
69,70	M, 5, IMAGE, a; authors confirmed temperature baseline is year 2000 which is 0.1°C warmer than 1990
71	M, 5, IMAGE, P, a; see #70
72	M, H3, P, D, d, f; ref. lists ΔT of 3.6°C for A1 in H3 in 2080 relative to 1961-1990, downscaled with H3 and upscaled with H3, EC, CG
73	M, NR, b; upscaled with H3, EC, CG
74	M, NR, b, f; WGI, chapter 10, figures 10.3.5 and 10.3.2 suggest global ΔT of 3.5°C relative to 1990
75	M, D, f; WGI, chapter 10, figure 10.3.5 shows this occurs for $\Delta T \geq$3.5°C above 1990
76	M, NR, b, f; see #75
77	M, D, b, f
78	M, SLR, a; US EPA scenario of 4.7°C above 1990.

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