3 Future Environmental Impacts and Vulnerabilities

Coordinating lead author: Andreas Fischlin

Lead authors: Matthew Ayres, David Karnosky, Seppo Kellomäki, Bastiaan Louman, Chin Ong, Gian-Kasper Plattner, Heru Santoso and Ian Thompson

Contributing Authors: Trevor H. Booth, Nico Marcar, Bob Scholes, Chris Swanston and Dmitry Zamolodchikov

Abstract: The focus of this chapter is on climate-change impacts on the environment, the structure and functioning of forests, on their biodiversity, and on the services and goods provided by forests in order to identify key vulnerabilities. Based on the findings of the IPCC Fourth Assessment Report (IPCC 2007d), we first introduce four clusters (*unavoidable*, *stable*, *growth*, and *fast growth*) of climate change scenarios commonly used to quantitatively assess climate change impacts (sub-chapter 3.2). At the global scale (sub-chapter 3.3) as well as in the four domains (boreal – sub-chapter 3.4; temperate – 3.5; subtropical – 3.6; tropical – 3.7), our CCIAV-assessment (see glossary) for forests shows that many forests can adapt to a moderate climate change if water is sufficiently available, notably in currently temperature limited areas (*unavoidable*, lower end *stable*). In some temperate or boreal regions, certain forests can even increase their primary productivity in a moderate climate change. However, some of these benefits are easily offset as climate warms and the adaptive capacity of currently water limited, fire or insect prone forests is frequently exceeded already by a limited climate change (*unavoidable*, *stable*). Many other forests become also vulnerable to an unmitigated climate change (*growth*, *fast growth*) as their adaptive capacity is exceeded. Forests currently sequester significant amounts of carbon; a key vulnerability consists in the loss of this service, and forests may even turn into a net source. Among land ecosystems, forests currently house the largest fraction of biodiversity; unmitigated climate change threatens to put significant parts of it at risk. The boreal domain, being especially sensitive, serves as a model case and is treated in particular depth. Finally, conclusions are drawn to summarize all findings on the global as well as regional scales (sub-chapter 3.8).

Keywords: Climate change scenarios, climate change impacts, forest properties, forest functioning, forest services, climate triggered disturbances, autonomous adaptation, climate change opportunities, adaptive capacity, forest resilience, key vulnerabilities

■

3.1 Introduction

Forests provide many ecosystem services that are key to human well-being (cf. Chapter 1). This chapter focuses on impacts of climate change on these services, and elucidates how different scenarios of climate change can and will affect forests and their services, mostly only indirectly through a multitude of interdependent processes in a complex manner.

Many forest services have not yet been recognized as having value by markets (for a recent review on these issues cf. Fischlin et al. 2007, sub-chapter 4.5). Yet other approaches such as the recurrent themes (cf. Chapter 1) allow for the roles of forests to be described within a more market-oriented context. For a Climate Change Impacts, Adaptation, and Vulnerability (CCIAV) Assessment we need to know how forests will be exposed to climate change, how sensitive they are to that exposure, and in how far they have the capacity to adapt. Future exposure and sensitivity (cf. Chapter 2) determine future impacts and are typically given in a climate change scenario as simulated by a climate model forced by an emission scenario attempting to capture future human behaviour (cf. sub-chapter 3.2). Any response by a forest ecosystem – either at the scale of leaves, branches, trees, stands or up to the scale of entire biomes – can be modelled by an impact model and is understood as autonomous adaptation (cf. Chapter 2), since that response by the forest is not directed at avoiding or minimizing adverse impacts (cf. Chapter 4). This is in contrast to human adaptation, typically attempting at avoiding adverse impacts or exploiting beneficial opportunities, e.g. through silviculture (cf. Chapter 5) or through policy measures (cf. Chapter 7). When the adaptive capacity is sufficient to counteract the impacts from climate change, the forest ecosystem may continue to behave in a mode similar to the past. Otherwise, when the forest system's resilience breaks down and causes the ecosystem to switch to an entirely new mode of behaviour, for instance when a forest becomes grassland, such a forest is considered to be vulnerable to climate change.

Given that forests cover about a third of the Earth's land in many climates, store about half of all carbon (Fischlin et al. 2007), and very likely house the majority of biodiversity of land ecosystems, in accordance with the precautionary principle (cf. Chapter 1, 7), impacts of future climate change on forest properties, structures, goods, and services are of major interest to humankind (cf. Chapter 7). Moreover, since forests may not only be impacted by climate change, but play also a major role in the global carbon cycle, their fate is of decisive relevance also for the future fate of the climate system. Unfortunately, current approaches and models do not yet allow studying this interplay between forests and the rest of the climate system in a fully coupled manner. Nevertheless, impacts and possible feedbacks can be assessed systematically, enabling us to address the risks of climate change in an appropriate framework.

3.2 Climate-Change Scenarios

Any CCIAV assessment (see glossary) is based on particular, assumed environmental and socio-economic conditions and requires scenarios of climate change that are internally as consistent as possible and portray plausible representations of the future. Currently a large number of climate change scenarios are used for CCIAV assessment scenarios that are based on various assumptions about basically unknown future socio-economic conditions and their associated anthropogenic emissions that are used to create climate models. For the sake of simplicity, future climate change scenarios are grouped into four scenario clusters, thereby reducing the number of options for discussion of climate change impacts in the context of this report: *unavoidable*, *stable*, *growth* and *fast growth*. These categories relate mostly to current carbon dioxide (CO_2) emission paths and should be of particular relevance in the current climate change debate as it relates to impacts on forests. However, other clusters could have been chosen. This sub-chapter briefly introduces and describes some of the scenarios most often used in the context of CCIAV studies on forest ecosystems at the global as well as the regional scale, in particular as they pertain to the case studies discussed in this report.

Future climate change depends on many uncertain factors. There is still much debate not only about the causes of climate change and climate sensitivity (see glossary), but also the likely impact of future anthropogenic emissions of greenhouse gases, aerosols, the cycling of key elements like carbon and nitrogen, land-use change and various land-use or land management related effects. Despite these uncertainties, projections of future climate change are needed to address the potential human influence on climate and to decide on mitigation and adaptation measures.

Climate change projections are based on plausible, quantitatively specified assumptions about the possible evolution of demographic, socio-economic, technological and environmental factors. They all affect human emissions of greenhouse gases (GHGs) and aerosols and, thus, impact the Earth's radiation balance and ultimately climate. For example, future growth of human population, together with technological advances, will determine to some extent the usage of fossil fuels and associated greenhouse gas emissions. Some of these factors are also impacted by a changing climate, e.g. carbon sequestration by forests, or technology use dependent on infrastructures (Wilbanks et al. 2007). Thus, feedbacks emerge, which complicates the situation.

Moreover, the climate system's response to external forcings needs to be studied at the proper time scales. This is particularly important in the context of forests, since they respond more slowly than many other ecosystems and need to consider fast processes such as photosynthesis responding within seconds, as well as slow ones such as forest succession lasting centuries. Therefore, response times of forests are comparable to those of the climate as they respond to changes in radiative forcing (see glossary) resulting from changes in the chemical composition of the atmosphere. In the case of forest succession or soil formation response times are similar to the slowest components of the climate system such as the oceans, which operate at time scales of centuries to millennia. Century-long time scales contrast sharply with some human decision-making. This creates particular challenges for the consistency of scenarios, especially when projected far into the future. Consequently, socio-economically based emission scenarios cover typically only the 21st century. A few scenarios extend beyond 2100 to study the longer-term response of the climate system. Those scenarios are, of course, particularly welcome if we wish to study impacts of climate change on forest ecosystems.

3.2.1 Commonly Used Scenarios

The majority of emission scenarios and concentration pathways for all relevant GHGs and aerosols as used in current CCIAV assessments have been developed in the context of the Intergovernmental Panel on Climate Change (IPCC) reports in the course of the last two decades. Among those, the most commonly used are CO_2-only stabilization pathways (Wigley et al. 1996, Plattner et al. 2008) and multi-gas emission scenarios from the Special Report on Emission Scenarios (SRES) of IPCC (Nakicenovic et al. 2000). Some impact studies still use the older, simpler business-as-usual scenarios (IS90, IS92a, IPCC, 1990, 1992) as well as $2xCO_2$ scenarios, but the majority now uses the IPCC SRES scenarios. There are also new successor scenarios modifying or extending IPCC SRES scenarios such as constant radiative forcing after 2100, or zero emissions after 2000 (e.g. CMIP – Meehl et al. 2007) or 2100 (Plattner et al. 2008).

The CO_2-only stabilization profiles usually prescribe the pathway of the atmospheric CO_2 concentration following projections based on a particular emission scenario (Meehl et al. 2007, Plattner et al. 2008) up to a certain point in time and then allow the CO_2 concentration to stabilize at a given level. The SRES emission scenarios on the other hand are based on a set of storylines representing different demographic, social, economic, technological and environmental developments. The 40 IPCC SRES scenarios have been grouped into four scenario 'families' characterized by common narratives. Six scenarios are the most often used: A1B, A1FI, A1T, A2, B1 and B2.

The A1 family describes a future with a relatively low population growth but rapid economic growth and high energy and material demands moderated by rapid technological change. The A1 scenario family develops into three groups that describe alternative directions of technology change in the energy system. The A1FI scenario is representative of a fossil-intensive energy sector. Non-fossil energy sources are emphasized in the A1T scenario, whereas non-fossil energy sources and fossil sources are 'balanced' in the A1B scenario. The A2 family describes a heterogeneous world with economic development regionally oriented, slower economic growth and relatively high population growth. The B1 family describes a convergent world with low population growth as in A1 but with rapid changes in economic structure toward a service and information economy and the introduction of clean technologies. The B2 family describes a world in which the emphasis is on local solutions, with moderate population growth, intermediate levels of economic development, and less rapid technological change than in A1 or B1. Further details on the SRES scenario 'families' can be found in Nakicenovic et al. (2000) or in the IPCC Third (Houghton et al. 2001) and Fourth (IPCC 2007a) Assessment Reports.

IPCC insists that there is no basis to assign probabilities to any given scenario (Nakicenovic et al. 2000, cf. also the debates on these issues, e.g. Grübler and Nakicenovic 2001, Schneider 2001, Carter et al. 2007, Fisher et al. 2007). The emission scenarios must also not be interpreted as containing any policy recommendations. In addition, none of these scenarios include any future policies that explicitly address climate change. The more recent newer scenarios explicitly take climate mitigation actions in the scenario set-up into account (e.g. EMF 21, Weyant et al. 2006). However, these new mitigation scenarios have so far only been applied in a few climate/carbon cycle studies (see e.g. Van Vuuren et al. 2008) and are currently not yet much in use by CCIAV studies.

3.2.2 Climate Projections: Global Aspects

Climate projections based on the best currently available coupled climate models and using the previously introduced illustrative SRES emission scenarios have been presented in the recent IPCC AR4 (Meehl et al. 2007). The main findings from (Meehl et al. 2007) focus on two key climate parameters: global mean surface air temperature and global mean precipitation (Figures 3.1 and 3.2). Note that uncertainties associated with precipitation projections are still larger than those associated with temperature projections. Recent advances, however, now allow more robust precipitation projections for large parts of the globe (Figure 3.2; note ratio of coloured vs. white areas and fraction of stippled areas representing varying degrees of model agreement).

Projected global mean surface air temperatures do not differ substantially among scenarios until ~2030 but then start to diverge quickly (Figure 3.1). Global mean surface air temperatures still rise in all scenarios by the end of this century and reach a warming of 1.8 (B1), 2.8 (A1B), 3.4 (A2) and 4.0 (A1FI) °C by 2100 relative to present levels (IPCC 2007e, p. 70, Table TS.6), thereby covering an actual range 1.1 to 6.4°C warming by the end of this century

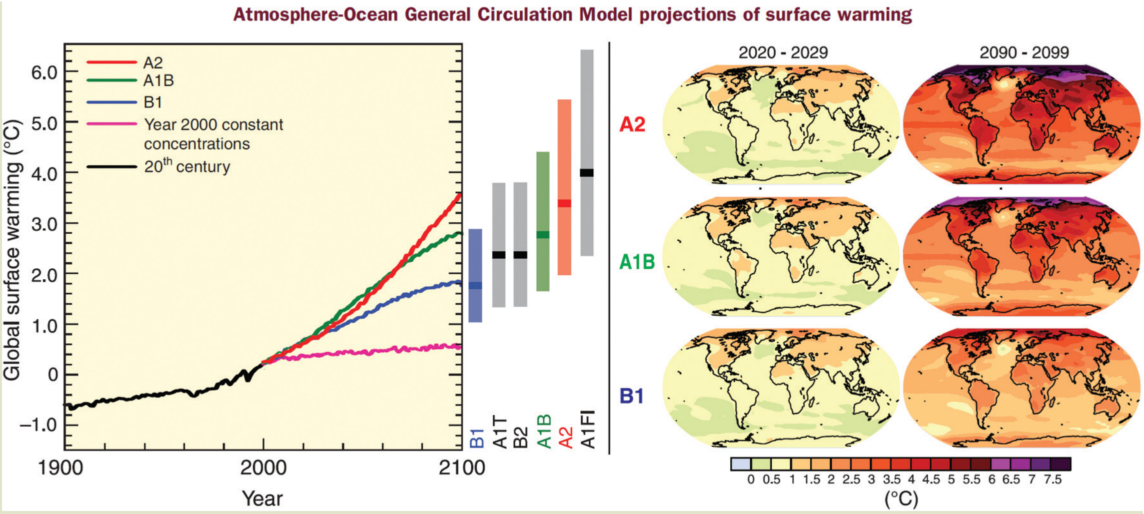

Figure 3.1 Left panel: Solid lines are multi-model global averages of surface warming (relative to 1980–1999) for the SRES scenarios A2, A1B and B1, shown as continuations of the 20th-century simulations. The pink line stands for the experiment where concentrations were held constant at year 2000 values. The bars in the middle of the figure indicate the best estimate (solid line within each bar) and the likely range assessed for the six SRES marker scenarios at 2090–2099 relative to 1980–1999. The assessment of the best estimate and likely ranges in the bars includes the Atmosphere-Ocean General Circulation Models (AOGCMs) in the left part of the figure, as well as results from a hierarchy of independent models and observational constraints. Right panels: Projected surface temperature changes for the early and late 21st century relative to the period 1980–1999. The panels show the multi-AOGCM average projections for the A2 (top), A1B (middle) and B1 (bottom) SRES scenarios averaged over decades 2020–2029 (left) and 2090–2099 (right) (IPCC 2007d, p. 46, Figure 3.2, reprinted with the permission of IPCC. See also IPCC 2007c, Meehl et al. 2007, section 10.4, 10.8, Figures 10.28, 10.29).

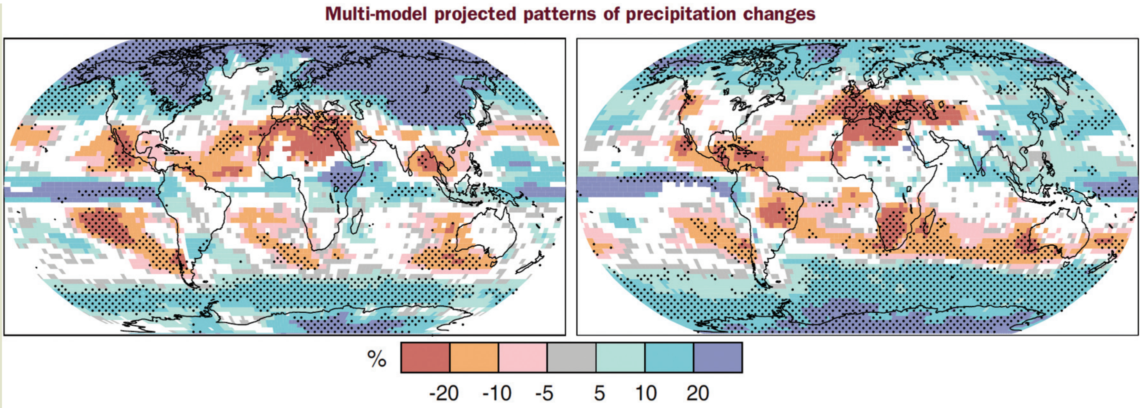

Figure 3.2 Relative changes in precipitation (in percent) for the period 2090–2099, relative to 1980–1999. Values are multi-model averages based on the SRES A1B scenario for December to February (left) and June to August (right). White areas are areas where less than 66% of the models agree in the sign of the change and stippled areas are those where more than 90% of the models agree in the sign of the change (IPCC 2007d, p. 47, Figure 3.3, reprinted with the permission of IPCC. See also IPCC 2007c, Meehl et al. 2007, Figure 10.9).

(33% confidence interval). The regional distribution is such that high latitudes, particularly in the Arctic, warm much faster than low latitudes, and land masses warm much faster than the oceans.

Precipitation is projected to wane further in regions that are already dry today (subtropics, e.g. Mediterranean basin), whereas regions that are relatively wet today tend to become even wetter (high latitudes, inner tropics, Figure 3.2).

While figures 3.1 and 3.2 indicate how climate might change on average and in the long term, no information about short-term variability, extreme events, in particular, is shown. However, short-term variability such as storms of any kind are as relevant as changes in the means, since they can cause serious damage, not least to forests. Current understanding

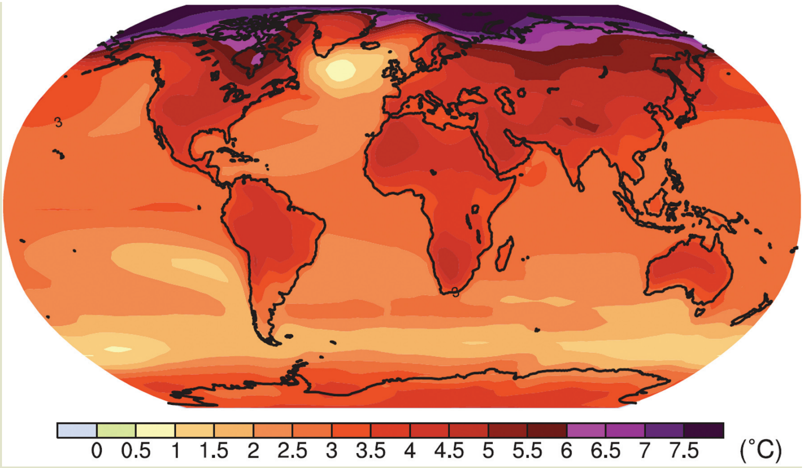

Figure 3.3 Multi-model mean of annual mean surface warming (surface air temperature change, °C) for the scenario A2 by time period 2080 to 2099. Anomalies are relative to the average of the period 1980 to 1999 (Meehl et al. 2007, p. 766, Figure 10.8, reprinted with the permission of IPCC).

and climate models indicate that, for example, future tropical cyclones are likely to be more intense due to the ongoing increases of tropical sea surface temperatures (IPCC 2007c). The projections indicate larger speeds of peak winds and more heavy precipitation events. There are indications that the frequency of tropical cyclones may decrease. However, confidence in those projections is much smaller. Since 1970 the proportion of very intense storms has been observed to increase in some regions, whereas current climate models simulate for that period a much smaller proportion. Similarly, changes in extra-tropical storms are projected; for example, there may be a northward shift of storm tracks.

3.2.3 Climate Projections: Regional Aspects

Although of great interest for studies of impacts and adaption, regional projections are associated with larger uncertainties than global projections. Nevertheless, recent advances in climate models now allow more reliable projections of regional climate change (e.g. Christensen et al. 2007, Figure 11.15 [p. 895], Figure 11.2 [p. 869], Figure 11.17 [p. 901], Figure 11.5 [p. 875]). Within this report, the focus is on four particular regions where several case studies investigate climate-change impacts on forests. Focus areas discussed include the Amazon, South Africa (Box 3.2), Southern Australia, and Northern Europe (Box 3.1). Figure 3.3 shows multi-model mean temperature projections at a scale suitable for these case studies featuring the IPCC SRES A2 scenario.

Special downscaling techniques would need to be applied in order to increase the reliability of regional projections based on global coupled climate models, in particular in the context of assessments of impacts on ecosystems (e.g. Gyalistras et al. 1994, Gyalistras and Fischlin 1999, Jones et al. 2005). Unfortunately, only a limited number of impact and adaptation studies use such techniques, which are of particular relevance in complex terrains where downscaling would actually be a necessity (e.g. Gyalistras et al. 1994, Fischlin and Gyalistras 1997, Gyalistras and Fischlin 1999).

3.2.4 Scenario Clusters

The four scenario clusters *fast growth*, *growth*, *stable*, and *unavoidable* stress commonalities among scenarios in the current trends of emissions and weigh possible later differences among pathways in the second half of this century much less.

Growth: With no major technological changes and without stringent climate policies, emissions are expected to continue growing and would still do so at the end of the century as captured in the IPCC SRES reference scenarios A1FI, A1B and A2. As a consequence, atmospheric CO_2 concentrations are expected to continue rising for quite some time after 2100 and the climate system will be out of equilibrium for centuries thereafter (e.g. Christensen et al. 2007, IPCC 2007d, IPCC 2007c, Meehl et al. 2007).

Stable: With major technological changes CO_2 emissions are expected to start declining during the course of this century as captured by the IPCC

SRES reference scenarios A1T, B2 and B1. As a consequence, atmospheric CO_2 concentrations are expected to approach a new equilibrium towards the end of this century (e.g. Christensen et al. 2007, IPCC 2007d, IPCC 2007c, Meehl et al. 2007). Such a stabilization of atmospheric CO_2 concentrations would be in accordance with the ultimate goal of the UNFCCC in its Article 2. However, whether the particular stabilization levels of the scenarios belonging to this cluster would avoid any dangerous interference with the climate system is and remains an unanswered question (Solomon et al. 2009). Moreover, judgements about dangerous interference cannot be properly addressed merely on scientific grounds (e.g. IPCC 1996).

The majority of CCIAV studies fall into the cluster *growth* while only a limited number look at scenarios belonging to the cluster *stable*. Studies assessing minimal adaptation or scenarios facing particularly rapid climate change are special cases of special interest in the context of this report. Consequently, two additional clusters have been introduced: *unavoidable* and *fast growth*.

Unavoidable: IPCC AR4 published for the first time multi-model simulations of climate system responses to an arbitrary freeze of atmospheric CO_2 concentrations at year 2000 levels (Figure 3.1, left panel, pink line) (IPCC 2007d, IPCC 2007c). This scenario is artificial and is very unlikely to be attainable in reality, since it implies as of 2000 negative and later zero emissions (unless atmospheric CO_2 would be sequestered in large amounts by forests and new technologies). However, the resulting climate scenarios allow the assessment of minimal impacts and minimum adaptation requirements.

Fast growth: Since about 2000, global emissions have been accelerating. CO_2 emissions rise currently by over 3% annually, whereas annual growth rates in the 1990s were on average only 1.1% (Raupach et al. 2007). These trends are not captured by the commonly used IPCC emission scenarios and even more importantly they are beyond the emission rates of the SRES reference scenario with the highest emissions for the present, i.e. the A1FI scenario. It is clear that this most recent trend in global emissions forms a particular challenge for humanity, including the forest sector (e.g. Schellnhuber et al. 2006, Ramanathan and Feng 2008), given the multi-millennial lifetime of the human CO_2 perturbation.

Whenever possible, this report refers back to these scenario clusters while discussing impacts, adaptation options, vulnerabilities and policy options.

3.3 Global Changes and Impacts

Given a climate-change scenario (cf. sub-chapter 3.2.4) and state-of-the-art Dynamic Global Vegetation Models (DGVMs, see Glossary), one can project future land vegetation under any climate-change scenario (e.g. Prentice et al. 2007). Unfortunately models of this type represent forests only at the biome level, i.e. they work with 'plant functional types' (PFT, see Glossary) instead of actual species. Other forest models are available, such as patch dynamics models that do operate at the species level and can also be applied in a CCIAV assessment context (e.g. Kirschbaum and Fischlin 1996, Box 1–4, p. 105). Patch models are most attractive for being able to mimic realistically the temporal characteristics of responses to a changing climate and their species specificity. However, they have the disadvantage of a limited geographical applicability and most of them are limited to the temperate and boreal domain (e.g. Solomon and Leemans 1990, p. 312), whereas DGVMs have the advantage of being applicable globally. Thus, current projections at the global scale are based on DGVM simulations such as those provided by IPCC (Figure 3.4, Table 3.1, Fischlin et al. 2007)

Table 3.1 Major biome changes projected by LPJ forced by a scenario from cluster *stable* (sub-chapter 3.2.4, ECHAM5 B1) and from cluster *growth* (sub-chapter 3.2.4, HadCM3 A2) (assumed forest/woodland area estimates for 2000: 41.6 Mkm^2 from Bonan 2002, Sabine et al. 2004, see Figure 3.4 for maps on underlying ecosystem changes and numbers used to denote types of vegetation changes)

Vegetation change	Scenario *stable* area change ($\Delta T_{2100\text{-preind.}}$ +2°) (Mkm^2)	Scenario *growth* area change ($\Delta T_{2100\text{-preind.}}$ +3.8°C) (Mkm^2)
6: Forest/woodland decline	–4.1 (–12%)	–12.1 (–29%)
1+2+3: Forest/woodland expansion	12.7 (+31%)	16.6 (+40%)
1+2+3–6: Net forest/woodland change	8.6 (+21%)	4.5 (+11%)

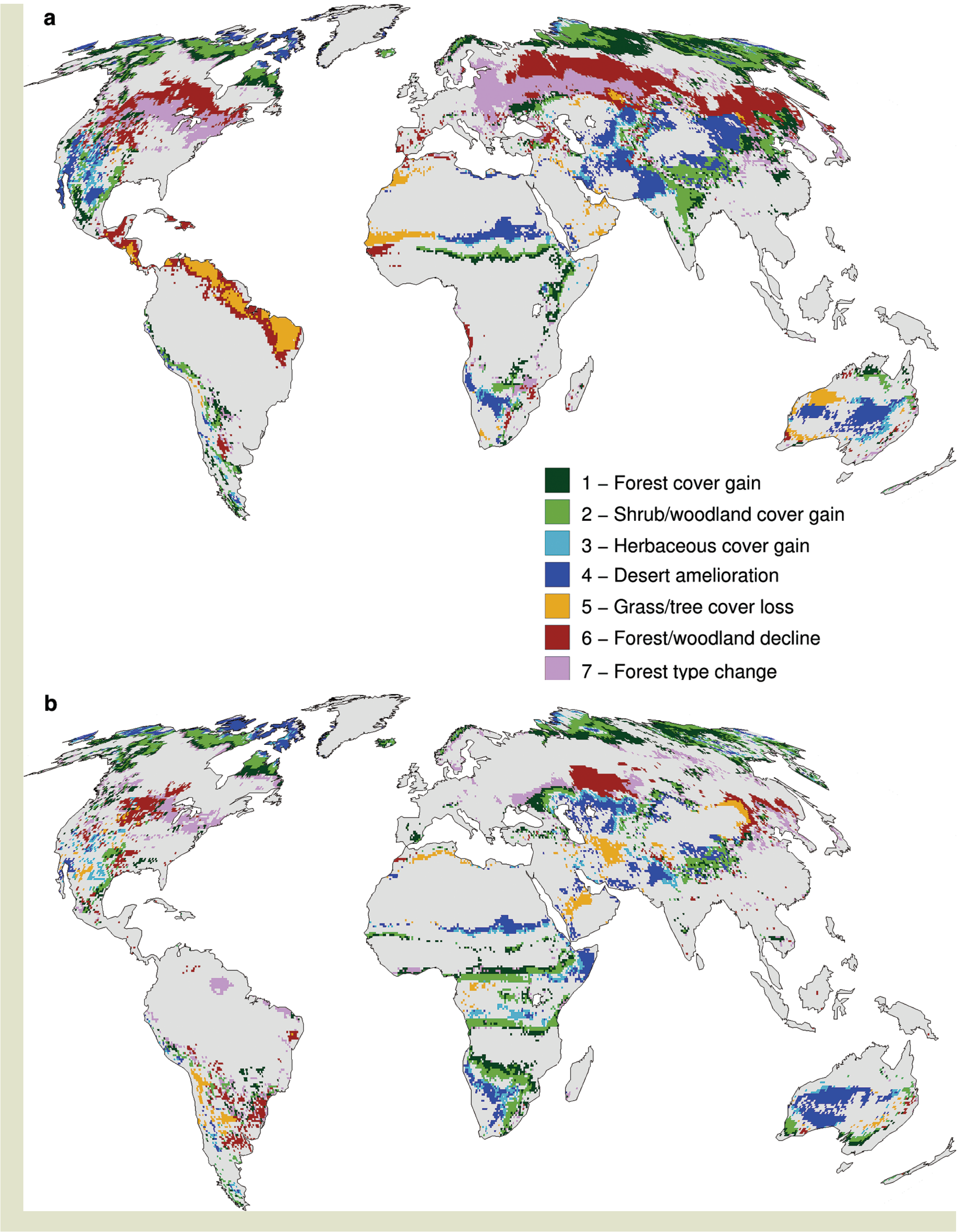

Figure 3.4 Projected appreciable changes in terrestrial ecosystems by 2100 relative to 2000 as simulated by DGVM LPJ (Sitch et al. 2003, Gerten et al. 2004) for two scenarios forcing two climate models: (a) scenario cluster *growth* (sub-chapter 3.2.4, HadCM3 A2), (b) scenario cluster *stable* (sub-chapter 3.2.4, ECHAM5 B1) (Lucht et al. 2006, Schaphoff et al. 2006). Changes are considered appreciable and are only shown if they exceed 20% of the area of a simulated grid cell (Fischlin et al. 2007, p. 238, Figure 4.3, reprinted with the permission of IPCC. See also Table 3.1).

The same climatic change impacts forests in a different manner, depending on the locally specific bioclimatic and edaphic conditions and the species composition. Furthermore, the management of forests and land use will modify the ecological responses of the ecosystems to climate change. This further emphasizes the need to analyze the impacts of climate change in a local context to gain a better

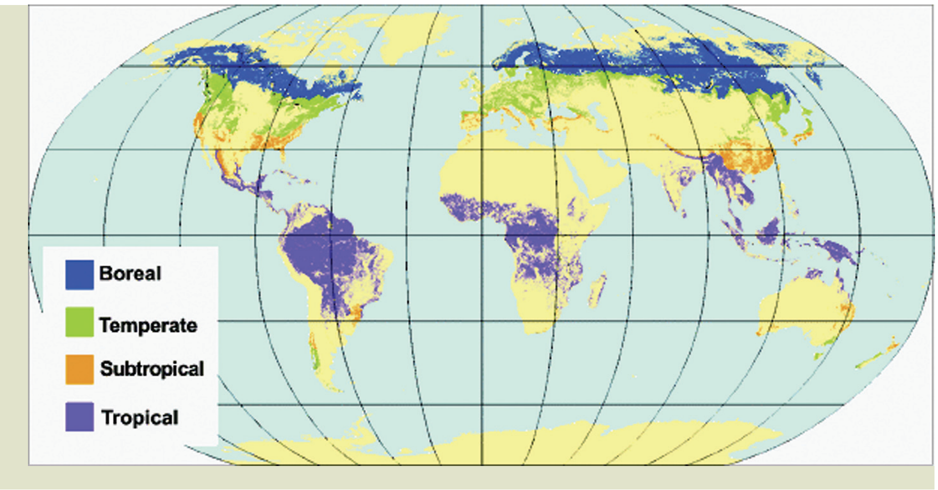

Figure 3.5 The four forest domains: boreal, temperate, subtropical and tropical as distinguished in this report (FAO 2001b, p. 5, Figure 1–4).

understanding of how climate change may affect provisioning and other services in the future, including the potential for forestry. This chapter will therefore make a separate CCIAV assessment for each of the four domains (see Chapter 1): boreal (sub-chapter 3.4), temperate (3.5), subtropical (3.6) and tropical (3.7) (Figure 3.5).

The following text is organized in such a way that both views, i.e. ecosystem services and the recurrent themes view, as alluded to above, are covered. The boreal domain serves as a model case and will be discussed in greater depth than the other domains. Topics covered for other domains treat complementary aspects, in particular those that call for special emphasis in the respective domain. Some of the regional biases in the following ought to be seen as exemplary and otherwise as being rather coincidental, since this chapter, given its scope, had to be written by a relatively small team of authors.

3.4 Boreal Domain

3.4.1 Types of Boreal Forests

The boreal forests (forests and other woodlands) cover 1270 million ha of land including boreal coniferous forests (730 million ha), boreal tundra (130 million ha) and boreal mountains (410 million ha), mainly in North America (Canada, Alaska), the Nordic countries (Finland, Sweden, Norway) and Russia (FAO 2001b). The boreal biome is the second largest terrestrial biome and has 33% of the Earth's forested area (FAO 2001b, Fischlin et al. 2007). These circumpolar forests (Figure 3.5) represent the environmental conditions characterized by the annual mean temperature of –5°C to +5°C, and the annual precipitation is 300–1500 mm. In these conditions, the potential evapotranspiration is about 400–450 mm but the actual evapotranspiration is substantially less (300–350 mm). The mean maximum temperature of the warmest summer month is more than 10°C, and the duration of summer is not longer than four months. The boreal zone is humid and typically characterized by coniferous tree species. Because of the cold winter and thin cover of snow, permafrost covers large areas in Alaska and the high-continental boreal zone in Canada and Siberia, where soil temperature regularly remains below 0°C even in summer.

The mean stem wood stocking in the boreal forest is about 120 m^3/ha, with a total mean stem wood growth of 1.6 m^3/ha/a (Table 3.2). In these forests, the most important coniferous species are pines (*Pinus*), spruces (*Picea*), firs (*Abies*), larches (*Larix*), junipers (*Juniperus*), thujas or cedars (*Thuja*) and hemlocks (*Tsuga*), while the most common deciduous species in these forests are poplars (*Populus*), birches (*Betula*), willows (*Salix*), and alders (*Alnus*). Most boreal tree genera occur throughout the zone representing transcontinental distributions across Eurasia or North America. The number of conifer species is greatest in North America, but also large in the southern part of the Far East. The number of tree species is particularly small in the north-western areas of Eurasia, where Scots pine (*Pinus sylvestris*) and Norway spruce (*Picea abies*) dominate the forested landscapes.

Table 3.2 Stocking and growth of forests in the major boreal forest regions (Kuusela 1990, FAO 2001b).

Region	Stocking, m^3/ha	Net annual growth, $m^3/ha/a$
Alaska	280	0.8
Canada	110	1.7
Nordic countries	90	3.3
Russia	130	1.4
Total mean	120	1.6

Table 3.3 Forest resources in the boreal regions (Kuusela 1990, FAO 2001b).

Region	Growing stock (10^9 m^3)	Net annual increment (10^6 m^3)	Annual removals (10^6 m^3)
Alaska	1.3	3.6	3.1
Canada	23.0	356.0	152.0
Nordic countries	4.4	158.0	102.0
Russia	67.0	750.0	357.0
Total	95.7	1300.0	642.0

3.4.2 Main Services Provided

Globally, timber production and carbon sequestration are the main forest goods and services, but the boreal forests are also important for conserving global biodiversity and supporting the production of many other goods and services. Their general temperature limitation results in particular characteristics, such as a higher production of humic substances which may lead to particular soil characteristics and the production of non-wood products such as berries and fungi.

3.4.3 Current Opportunities and Vulnerabilities

Timber: The total growing stock of trees in the boreal forests is 100 000 million m^3, of which 80 000 million m^3 represent coniferous tree species (Table 3.3). Boreal forests stock is about 45% of that of all forests and about 50% of that of the coniferous species. This stock increases by 1300 million m^3/a, which corresponds to about 30% of the global forest growth. The coniferous species growth is about 45% of that of the global coniferous net production. Annually about 600 million m^3 are harvested, which represents 20% of the global removal: 500 million m^3 are softwood corresponding to 45% of the global softwood harvest. 500 million m^3 are industrial wood corresponding to 37% of the global industrial wood harvest. The boreal forests in northern Europe or Fennoscandia (including Norway, Sweden, Finland and north-western Russia) provide about 40% of the timber used in Europe on 85% of the total forest area in Europe (956 million ha).

Carbon: The boreal region has been estimated to contain a total of 703 Pg of carbon and about 30% of all the carbon contained in the terrestrial biomes. (Symon et al. 2005, p. 550). Recent estimates by IPCC (Fischlin et al. 2007) that include soil carbon of forest soils to a depth of 3 m (Jobbagy and Jackson 2000) give a different picture: Boreal forests alone contain only 207 PgC, which corresponds to about 13% of all carbon contained in forests. This correction is also in line with other studies since the IPCC Third Assessment Report (Kauppi 2003). In Finland roughly three-quarters of this carbon is held in soils and can be as high as 88% (Kauppi et al. 1997), since cold temperatures slow decomposition resulting in an accumulation of soil carbon. The carbon budget of the boreal forests indicate a net sink between 0.5 and 2.5 MgC/ha/a (Shvidenko and Nilsson 2003). However, given the relatively small annual growth rates vis-à-vis the high rates of net felling, boreal forests are most sensitive to disturbances and any interannual variability in harvesting (Kurz and Apps 1999). Unfortunately, the global forest statistics exclude any changes in the frequency and severity of disturbances, which makes it difficult to assess the source/sink relationship and its changes over time.

Detailed analyses of forest inventory data, together with observed changes in disturbance over time, indicate that Canadian forest ecosystems changed from a modest sink (0.075 GtC/a) between 1920–1970 to a small net source of 0.050 GtC/a as of 1994 (Kurz and Apps 1999). In Russia between 1983 and 1992, managed forests from the European part were a sink of 0.051 GtC/a, while the less intensively managed Siberian forests were a net source of 0.081–0.123 GtC/a. (Shepashenko et al. 1998). These estimates are based on bottom-up methods that exclude factors such as CO_2-fertilization (e.g. Schimel et al. 2001), nitrogen deposition (e.g. Kauppi et al. 1992) and/ or climate change (e.g. Zhou et al. 2001, McMillan et al. 2008). This may have biased these estimates, a view which is also supported by remote sensing-based estimates (e.g. Myneni et al. 2001). Growth of tree species in the boreal conditions and elsewhere representing C_3-plants is sensitive to elevating CO_2 whenever the availability of nitrogen or other nutrients is not limiting (e.g. Jarvis and Aitken 1998).

Biodiversity: In general, the number of species per unit area is low at high latitudes. However, the total species richness in the boreal region is greater than in the poleward tundra, but less than in the temperate forests at mid-latitudes. Roughly, the species richness is correlated to the productivity of an ecosystem and, thus, increasing along the meridional temperature gradients across the boreal vegetation zone (Ympäristöministeriö 2007). The boreal forests frequently give way to mires and small lakes, leading to a mosaic structure of forest and wetland, which provides a huge variability in available habitats and, thus, increases the species and genetic richness at the landscape level. On the other hand, the boreal forests are characterized by large numbers of individuals of few tree species with a wide ecological amplitude, in contrast to tropical forests that sustain a small number of individuals of many species with a narrow ecological amplitude. Genetic diversity in any species is in part the result of the opportunity the species offers for gene recombinations. The genotypic variability represents adaptations to the specific conditions of local environments, suggesting a high degree of local adaptation within the boreal domain.

In the continental parts of the boreal forests, fire controls the natural dynamics of the forests and consequently influences biodiversity. Some species are adapted to using the resources provided by standing and lying burnt trees in different stages of decay. In particular, fire sustains a set of species in early post-fire communities that are distinct from later successional species. These include species from a range of groups, including birds, beetles, spiders and vascular and non-vascular plants (Esseen et al. 1993). If regular fires are absent, many species can build large populations only in situations with a reduced species richness. Such effects can be observed in many managed boreal forests in Nordic countries. Moreover, where effective fire-fighting has made fire events rare, species which depend on fire-modified habitats are now threatened. In Finland, 14 species, mostly beetles (*Coleoptera*) and bugs (*Hemiptera*), associated with burnt forest land are threatened (Ympäristöministeriö 2007).

Despite their relatively low number, the species in managed boreal forests represent an important part of the global biological diversity. This holds good for many countries in the boreal domain such as Finland (see also Box 3.1), where more than 90% of the forest area is managed for timber production. There the estimated total number of species is about 50 000, out of which about 43 000 are known. The reason for the relatively low total number of species is the short time that has elapsed since the last glaciations (10 000 years ago), with the consequence that immigration is still going on (e.g. Johnstone and Chapin 2003, Callaghan et al. 2004, Harris 2008). These species and the subsequent biodiversity involve a large contribution from natives of the eastern taiga (flying squirrel, Ural owl, Siberian jay, to name just a few). Most of these taiga species are connected with spruce forests. On the other hand, a high proportion of the forest species (20–25%) are dependent on dead wood (800 coleopterans, 1000 dipterans, 1000 fungi, 200 lichens, etc.). Many of these species are specialized in living on recently burnt tree material, while a high proportion live in peatland forests or on mires (Kellomäki et al. 2001, Ympäristöministeriö 2007).

3.4.4 Projected Future Impacts and Autonomous Adaptation

The boreal domain will experience more warming than equatorial zones (Anisimov et al. 2007, Christensen et al. 2007). Consequently, and because boreal forests are generally temperature limited, they are expected to be particularly impacted by future climate changes as stated by IPCC (Kirschbaum and Fischlin 1996, Anisimov et al. 2007, Fischlin et al. 2007).

Biome shifts: A key impact of climate change will be the effect on the living conditions of many species and their distribution will be altered. Although evidence from past climate changes shows that species respond individually, the boreal domain is nevertheless expected to shift polewards as an entire biome. In Canada, Price and Scott (2006) used the IBIS model to predict changes in the extent of the boreal and sub-boreal forests (Figure 3.6). Their work predicts a marked northward migration of the boreal forest and a considerable increase in parkland or savanna and grassland in previously boreal zones in central and southern Canada. Depending on the model scenario, carbon stocks increase or decrease in North America with climate change 'business as usual' scenarios by 2100 (Neilson et al. 1998, Price and Scott 2006). This difference is affected by different assumptions in the models and depends considerably on response to CO_2-fertilization and expected rates of fires. Thompson et al. (1998) projected fewer old-growth forests and more young forests across boreal landscapes under an increased fire regime.

Productivity: A main factor underlying the future impacts of climate change on the dynamics and vulnerability of boreal forests is how climate change affects the primary productivity of those forests. In the boreal domain, primary productivity is in general expected to increase through the following three main mechanisms: (i) CO_2-fertilization; (ii) temperature increases and lengthening of growing seasons; and (iii) precipitation increases under water-limited conditions that lead to a greater water availability. These effects tend to enhance regenerative, physiological and growth processes of trees. Based on the find-

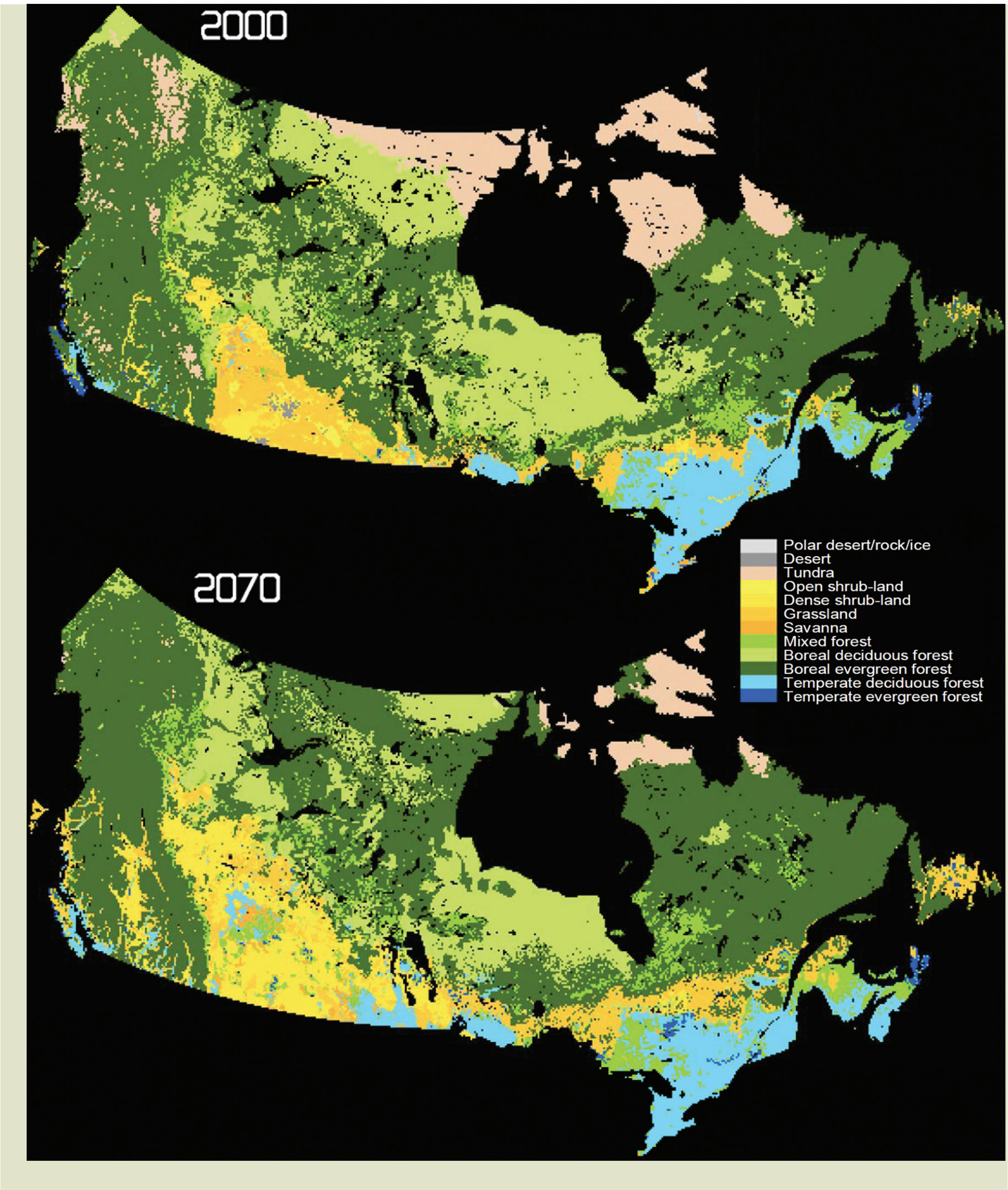

Figure 3.6 Distribution of major vegetation types as simulated by the vegetation model IBIS for 2000 and under a scenario from cluster *growth* (IPCC ISN92a) in 2070 in Canada. Note the band of grasslands extending across Ontario and south-western Quebec incorrectly simulated within a zone of otherwise continuous forest, suggesting the difficulty in accurately projecting future vegetation cover (Price and Scott 2006, reproduced with permission of the authors).

ings of satellite monitoring, IPCC reports a recent increase in global net primary production (NPP) by 12% in Eurasia and by 8% in North America from 1981 to 1999 (Fischlin et al. 2007, Rosenzweig et al. 2007). The underlying studies relate these changes to the elevation of the ambient atmospheric CO_2-concentration, lengthening of the growing season, nitrogen deposition, or changes in management. These estimates are well in line with the greening of the Northern Hemisphere as observed via remote sensing (e.g. Myneni et al. 2001), which is most probably due to the lengthening of the growing season at high latitudes due to the elevation of spring temperatures. The model-based analysis for Finland (Box 3.1) illustrates how climate change may affect forest growth in the boreal domain.

Box 3.1 Impacts of climate change on the growth of managed boreal forests in Finland (Kellomäki et al. 2008).

The simulations cover 26 million ha of forest land represented by the permanent sample plots of the Finnish National Forest Inventory located across the boreal forest zone at N 60°–70°. The current climate (1961–1990) used in the reference simulations represented the same spatial scale as the grid of the permanent sample plots of the National Forest Inventory. The climate-change scenarios were based on the IPCC SRES A2 emission scenario (cluster *growth*). By 2070–2099, the mean temperatures are projected to increase almost 4°C in the summer and more than 6°C in the winter. The annual precipitation is expected to increase by 10% in southern and up to 40% in northern Finland, mainly in winter. At the start of simulations in 1990, the atmospheric concentration of CO_2 was 350 ppm, compared with 840 ppm at the end of simulation in 2099. Current management practices were assumed in the simulation (Ruosteenoja et al. 2005).

Figure 3.7 shows that the growth integrated over the tree species varies currently from less than 1 m^3/ha/a in the north up to 6 m^3/ha/a in the south of Finland depending on the site fertility, tree species and age (or developmental phase) of tree populations. Climate change results in the largest change in growth in the northernmost part of the boreal region; i.e. any increase to a low growth rate may result in a large percentage change. Throughout northern Finland and Canada, the growth increase is several tens of percentages. In southern Finland, the increase is much less, ranging mainly from 10% to 20%, i.e. the integrated growth may increase up to 7 m^3/ha/a in the south. This implies that the growth at the rate of 3–4 m^3/ha/a currently prevailing in the central part of Finland may shift up to the Arctic circle (66°N). In southern Finland the growth may increase up to 12% in this century due to climate change. This is substantially less than in northern Finland, where the growth may be doubled compared to the growth under the current climate. Over the whole country, an increase of 44% was obtained, mostly effected by the large increase in the northern part of the country. However, the changes in the growth of Norway spruce are in many locations (mainly south from the latitude 62°N) small or even negative due largely to the more frequent drought periods occurring during the latter part of this century.

The increase in forest growth in the northern boreal region implies an increase in the potential timber harvest and carbon sequestration. The simulations showed that under southern boreal conditions the potential cutting drains may increase up to 50% by the end of this century. In the boreal forests of northern Finland, the increase is much larger (up to 170%), but there the absolute value (3 m^3/ha/a) is still less than two-thirds of that in the south (5 m^3/ha/a). At the same time, the duration and depth of soil frost will reduce substantially, which makes the winter-time timber harvest more difficult and reduces the overall profitability of timber harvest (Venäläinen et al. 2001).

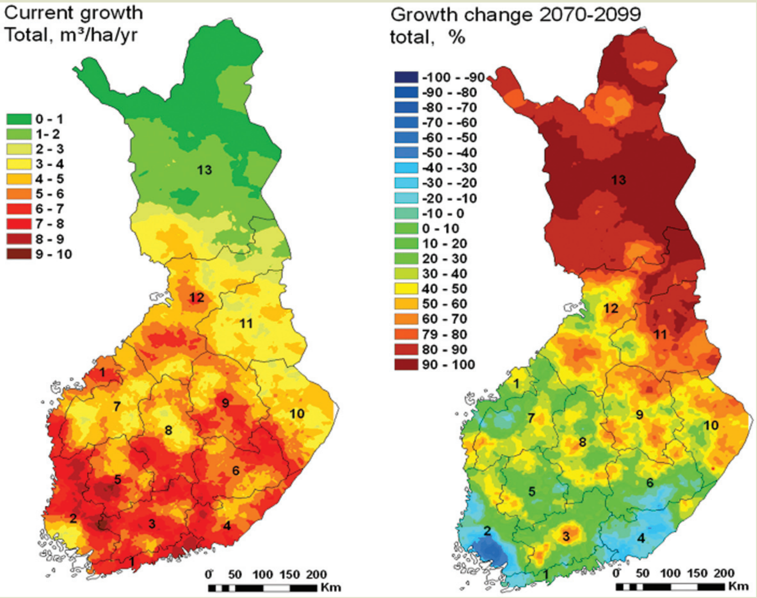

Figure 3.7 Current growth of stem wood (left) and the percentage change by the end of this century if the change in climate as described in the text (cluster *growth*) is assumed (numbers denote provinces within Finland)(Kellomäki et al. 2008).

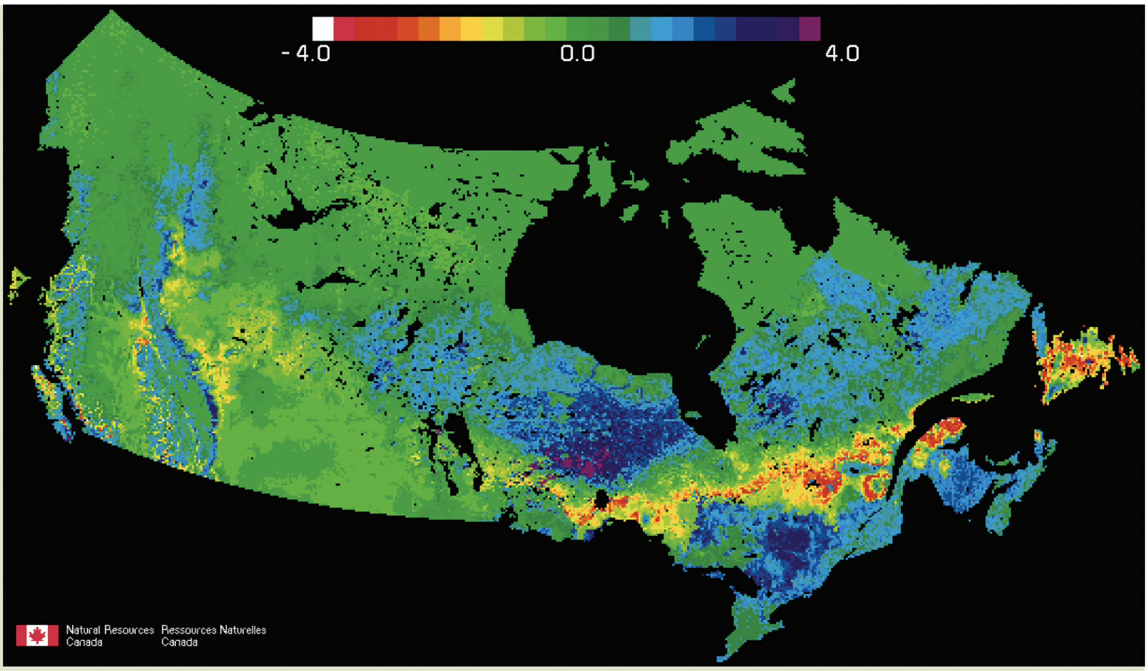

Figure 3.8 Changes in NPP (kgC/m²/a) as simulated by IBIS for the period 2070 relative to 2000 (Price and Scott 2006, reproduced with permission of the authors) for a scenario from cluster *growth* (IPCC ISN92a).

Similar results were found for Canada by Price and Scott (2006) with broad increases on average for the boreal region in excess of 0.2 kgC/m²/a. They suggested a wide range of variability, however, with some locations (central western and central eastern areas) showing only small increases in NPP, while other areas, such as those west of Hudson Bay, were found to have an increase of biomass by up of 0.5 kgC/m²/a (Figure 3.8). Nevertheless, they observed disparity among the results projected, depending on the emission scenarios and/or the various climate models used (*growth*: IPCC ISN92a, SRES A2; *stable*: IPCC SRES B2; GCMs: HADCM3, CGCM2, CSIRO Mk2). The MC1 model of Neilson (Lenihan et al. 1998, Daly et al. 2000, Bachelet et al. 2001) actually projects broad carbon losses for much of the same forests, which illustrates the complexity of these issues and raises questions and key uncertainties about the assumptions used in the models.

These model projections are illustrative examples that are based on complex assumptions. They encompass not only changes in climate such as increasing precipitation and warming temperatures, but also other effects such as CO_2 fertilization and species migrations. Some of these assumptions are associated with considerable uncertainties. The availability of sufficient nutrients or physiological acclimation to elevated CO_2 concentrations could significantly limit and reduce, respectively, the realized productivity gains from the CO_2 fertilization. An assumption of optimal dispersal of species results in projections of rapid shifts in geographical ranges.

Hungate (2003) argued that DGVM models make unrealistic assumptions about nitrogen availability. Since those nitrogen requirements as formulated in the models could not be met in reality, the model projections would be too optimistic, particularly in respect to sequestration services. A slackening of carbon sequestration would then result in an acceleration of climate change, which could lead eventually to environmental conditions where primary productivity would start to decrease even in the boreal domain. Fischlin (2007, section 4.4.1) discusses these issues in detail.

Current DGVMs also assume plant functional types that always have sufficient dispersal capabilities to track climate change optimally (e.g. Prentice et al. 2007). Real plant species, however, given the evidence from past climate changes (cf. Fischlin et al. 2007, section 4.4.5), are known to have limited dispersal capabilities. This is of particular relevance for tree species that are generally not expected to be able to track the rapid climate changes projected for this century (cf. Fischlin et al. 2007, section 4.4.5). This would lead to considerably lagged responses to climate change, perhaps century-long ones and, particularly where major soil formations are necessary, and the boreal timberline would advance polewards considerably slower than projected by current DGVMs (cf. Fischlin et al. 2007, section 4.4.5, 4.4.6).

In the boreal domain, climate change can also cause a decline in the primary productivity as has been documented for a substantial portion of forests in North America due to more frequent drought conditions. The concurrent increase in the productivity of the tundra, probably due to longer and warmer growing seasons, will in the long run cause northern boreal forests to invade the tundra, while boreal forests at the southern ecotone are likely to retreat due to increasing drought, insects and more prevalent fires (Denman et al. 2007, Fischlin et al. 2007, Figure 4.4–2). Since the rate of loss at the southern ecotone due to relatively fast processes such as fire is likely to be higher than the rate of gain at the northern ecotone due to the slow growth conditions, the overall effect of these two processes for the boreal forests is likely to be negative during the transient phase, i.e. until a new equilibrium between climate and vegetation is established. In this context it is also important to remember that climate-change scenarios from cluster *growth*, let alone *fast growth*, are generally not yet available beyond 2100, yet climate itself has not yet reached stabilization and, thus, the impacts assessed up to 2100 are not representative for the situation in the next century and beyond (compare also Box 3.1). However, in equilibrium a general increase in deciduous vegetation at the expense of evergreen vegetation is predicted at all latitudes, although the forests in both the eastern USA and eastern Asia appear to be sensitive to drought stress and already show declines under some scenarios in this century.

Box 3.1 illustrates these processes. While forest growth is projected to increase in general in Finland, the growth conditions in the southern boreal region are reduced because of the declining growth of Norway spruce due to the increasing frequency of drought periods. In the north, the primary productivity of the forest ecosystems may be increased substantially, but it will still be less than that currently present in the south. However, the special features of northern forests and terrestrial ecosystems may be diminished even above the current timberline. This development is probably quite inevitable, and little can be done in order to conserve the present character of the northern boreal forests. The northern forests may provide many opportunities for the forestry and timber industry, while the forest environment may turn suboptimal, e.g. for reindeer husbandry and recreation business, which are currently the main uses of the sub-artic and sub-alpine landscapes in the north.

Frost: In the boreal forests, the timing of bud-burst is related to spring temperatures, as found for birches and Scots pine (Myking and Heide 1995, Häkkinen et al. 1998). The bud-burst is preceded by low chilling temperatures during winter. Even under elevated temperatures, the chilling requirements of trees are likely to be fulfilled, and earlier bud-burst may be expected. On the other hand, there is no empirical evidence that earlier bud-burst under climatic warming would lead to catastrophic frost damage. On the contrary, in old provenance transfer experiments, where northern provenances of Norway spruce and Scots pine were grown in southern Finland, thus undergoing considerable ‘climatic change’ (increase of temperature sum by up to 600 degree days), bud-burst was hastened, but growth was also increased (Beuker 1994, Beuker et al. 1996). This is in line with the findings that in the phase of bud-burst the frost hardiness of Scots pine is still remarkable, i.e. it then still tolerates frost conditions below –20°C.

Storms: Strong winds blow down and break trees with large economic losses in timber production and productivity of forest ecosystems. The occurrence of wind damage is tightly linked with the occurrence of high wind speeds. The risk of wind damage is increasing with the maturing of trees, taller trees being at higher risk than shorter ones. On average, wind damage does not occur under boreal conditions up to a maximum mean regional wind speed of 15 m/s given that gusts also stay below 30 m/s (Peltola et al. 1999).

The overall risk of wind damage is greatest in stands adjacent to newly clear-felled areas and within newly thinned stands, especially if stands not previously thinned are suddenly thinned intensively. This is because wind is able to penetrate deeper into the canopy following thinning, with a subsequent increase in the wind load imposed on the trees, while dense stands dissipate incoming winds. The probability of damage decreases, however, with the time elapsed since thinning. However, changes in the occurrence of extreme wind speeds, along with the changing climate, are of the greatest importance. Except for the increasing risk of local wind extremes, it is still an open question whether climate change may induce changes in boreal wind patterns. However, the higher frequency of strong winds during periods of unfrozen soils in late autumn and early spring might be the most alarming scenario (Päätalo et al. 1999, Peltola et al. 1999, Venäläinen et al. 2001), although current climate-change scenarios do not allow boreal-domain projections for changes in storm patterns, i.e. intensities, frequencies and/or exact geographical or seasonal occurrence (cf. sub-chapter 3.2). The changing climate may decrease the duration of snow cover and frozen soil with a reduction in the overall anchorage and an obvious increase of wind-induced forest damage.

Snow: The severity of snow damage mainly depends on the amount of snowfall and the attachment of snow on crowns. Snow attachment is probable at temperatures around 0°C. In these conditions, snowfalls of 20–40 cm or more appear to represent a low to moderate risk, whereas snowfalls of about 60 cm or more increase damage risks to very high

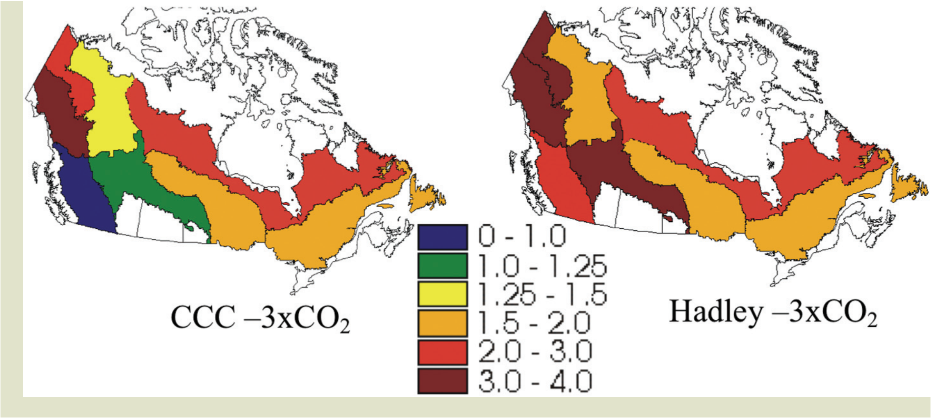

Figure 3.9 Projections of changes in area burnt based on weather/fire danger relationships shown as a ratio relative to a 1975–1990 baseline. These results suggest a 75–120% increase in area burnt (average ratios 1.75 and 2.2) by the end of this century according to scenarios from cluster *growth* (3xCO_2) as generated by two climate models (Canadian CCC and Hadley Centre HADCM3) (Flannigan et al. 2005 Figure 5, p. 11–12, copyright Springer. Reprinted with permission of Springer Science and Bussiness Media).

levels (Päätalo et al. 1999). Wind speeds less than 9 m/s appear to intensify the risk of snow damage otherwise induced by the accumulation of wet snow, whereas snow would more probably be dislodged from the tree crowns by wind speeds greater than 9 m/s (Peltola et al. 1999). In the Nordic countries, the mean return period of severe snow damages is 5–15 years.

A changing climate may affect the risk of snow damage in several ways. The share of snowfall from the total winter precipitation may be reduced with an obvious reduction of risk. On the other hand, increased winter precipitation may increase weather episodes with temperatures that enhance the attachment of snow and the concurrent accumulation of snow on tree crowns. Furthermore, increasing winter precipitation may lead to more intense snowfall and accumulation of snow. This would increase risks of snow damage if the temperature and wind conditions favour excessive snow accumulation, as seems to be the case for boreal conditions during the next 50 years (Venäläinen et al. 2001). These factors probably balance in such a way that at higher altitudes or most northern areas the risk of snow damage may increase, but at lower altitudes or areas outside the northern regions the risk may decrease. Later in this century, the risk of major snow damage will probably reduce due to a reduction of precipitation in the form of snow.

Fire: Temperature and precipitation are the main climatic factors affecting incidences of wildfires in forests. However, since precipitation is projected to increase with temperature in some cases (e.g. Bergeron 1991, Carcaillet et al. 2001) but to decrease in others (e.g. Hallett et al. 2003, Lynch and Hollis 2004), fire risk is expected both to increase and decrease with climate change (cf. Fischlin et al. 2007, section 4.4.5).

In boreal conditions, less rainfall during the growing season leads to more frequent fires during the same year (Flannigan and Wotton 2001). In Canada and Alaska, the temperature has been found to be the most important predictor of area burnt, with warmer temperatures associated with increased area burnt annually (Flannigan et al. 2001, Flannigan and Wotton 2001, Duffy et al. 2005, Flannigan et al. 2005). Boreal forest fire seasons have two peaks: In early spring large amounts of dry plant debris from the previous summer, without much green vegetation, increase the probability of fire ignitions (Zackrisson 1977). Furthermore, earlier melting of snow may dry out soils unless precipitation is simultaneously increased. The second peak is during the late summer when soils have dried out at a sufficiently deep level. Wotton and Flannigan (1993) estimated that the fire season length in Canada will increase by 22% or by 30 days, on average, in response to a climate scenario from cluster *stable* (2xCO_2). However, seasonal variability in precipitation may affect the development of the forest fire potential considerably. On the other hand, increased precipitation has also been found to mask the effects of warmer temperatures on forest fire (e.g. Bergeron and Archambault 1993). Flannigan et al. (2005) modelled two possible fire change scenarios for Canada under a scenario from the cluster *growth* (Figure 3.9). The models do not show strong concurrence except that the area burnt will increase by the largest amount in the extreme north-west area of the boreal region. More recent models for Alaska and northern and western Canada predicted even higher rates of increased fire, of up to 5.5 times the recent baseline, using scenarios from the clusters *growth* (IPCC SRES A2) and *stable* (IPCC SRES B2) (Balshi et al. 2008).

In more inhabited northern Europe, thanks to fire control, forest fires are rare, the percentage of forest land burnt annually being less than 0.05%. The mean size of fires is less than one hectare (Zackrisson 1977). The return period of fires is 50–100 years on average on dry upland sites, and much longer for moist upland sites. In boreal conditions, one to two weeks without rain is needed to significantly increase the fire risk even under current precipitation (250–700 mm/a). The projected more frequent drought spells, especially in southern Finland, indicate that the risk of wild fires may increase substantially. The main remaining uncertainties are related to the seasonal distribution of precipitation. Warmer temperatures in spring and early summer may lead to earlier melting of snow and drying of the soil in summer (Zackrisson 1977). A temperature increase alone (assuming no change in precipitation) of 3–5°C in summer (June–August) has been projected to increase the fire area in western Europe 15 to 50 times (Suffling 1992).

Insects: Pest insects are of considerable relevance in the boreal domain (e.g. Logan et al. 2003). Climate change affects insect outbreaks through several mechanisms, by altering (Evans et al. 2002): (i) survival and reproduction of the insects, (ii) natural enemies of the pests, (iii) nutrient content of the host trees, (iv) vigour and defence capabilities of the host trees, and (v) phenological synchrony of the pest and host trees (cf. Fischlin et al. 2007, section 4.4.5).

The northward expansion of several insect species and forest pests is likely to occur (Battisti 2004). However, the net impact of climate change on pests is complex to predict owing to interactions among plant defence mechanisms, food quality (Niemelä et al. 2001) including C:N ratio and effects of N-deposition on host plants, fire, altered ranges of forest species including enemies, feedbacks, weather and other factors (Williams et al. 2000). The complexity among these interactions results in a high degree of uncertainty with respect to future damage from outbreaks of endemic invasive insect pests (Fleming and Candau 1998, Fischlin et al. 2007). However, forest pests generally are likely to increase in frequency and intensity under climate change, particularly in the margins of the host tree species (Harrington et al. 2001, Fischlin et al. 2007, Ward and Masters 2007, see also Chapter 2).

The winter minimum temperature is the most important factor limiting pest distribution in the north. A warmer climate could provoke increases of outbreaks towards the north and accelerate the intensity and frequency of population peaks, although parasitoids and other natural enemies may cause higher mortality of larvae in the summer (Niemelä et al. 2001). The overall effect of this is not only poorly understood, but can hardly be generalized. However, a drier and warmer summer is favourable to the life strategy of many pests, and since pest insects multiply easily, they have in general a large potential for genetic adaptation to new environments, including their ability to defy control.

Although it is unlikely that insect pest species would have lower short-term success in a changing climate, Fleming and Candau (1998) suggested that certain feedbacks such as loss of host trees may actually reduce outbreaks, at least regionally. Nevertheless, it is known from many pests that they are very likely to have higher success: the European spruce bark beetle *Ips typographus* is one of them, the occurrence of which is typically related to the prevailing temperature conditions (Parry 2000). In spring the flight of *I. typhographus* occurs when daily mean temperatures exceed 18°C. The increasing spring and summer temperatures may increase the number of generations and the success of this species throughout Europe (e.g. Schlyter et al. 2006, Dobbertin et al. 2007). Similarly, the populations of *Neodiprion sertifer*, *Diprion pini* and *Panolis flammea* may grow due to temperature elevation by 2–3°C during the summer (cf. Virtanen 1996). Temperature elevation may also expand the occurrence of *Lymantria monacha* far up above the 60th latitude over Scandinavia. Currently, this insect damages Norway spruce mainly in central and southern Europe (Bejer 1988), wherever the mean temperature of July exceeds 16°C and the mean temperature of September exceeds 10.5°C (Parry 2000). In western Canada, recent climate change has been linked to the loss of millions of hectares of lodgepole pine (*Pinus contorta*) forest due to the mountain pine beetle (*Dendroctonus ponderosae*) (Logan et al. 2003, Carroll et al. 2004, Fischlin et al. 2007, Kurz et al. 2008).

Pathogens: High summer temperature combined with drought may damp down the epidemics of damaging fungi, but they may flourish in cool rainy summers. On the other hand, higher winter temperatures may enhance epidemics of damaging fungi like *Gremmenniella abietina* and *Lophodermella sulcigena*. The frequency of root rot induced by *Heterobasidion annosum* may also be larger, if the autumn, winter and spring temperatures are higher and the duration of frozen soil is shorter. In northern Europe, especially, climate change seems to enhance the occurrence of root rot with an increase in loss of timber and forest productivity (Parry 2000).

Alien invasive species: Alien invasive species are becoming a problem globally, and climate change will interact to increase the likelihood of their success (cf. Fischlin et al. 2007, section 4.4.11 [p. 218], Ward and Masters 2007). A good example of new organisms with large potential to damage trees is *Bursaphalencus xylophilus* nematode originating from North America. This pine nematode is quite easily transported in fresh timber, but its success is quite closely related to temperature. Until now, low summer temperatures and short growing seasons are effectively limiting the success of this species out-

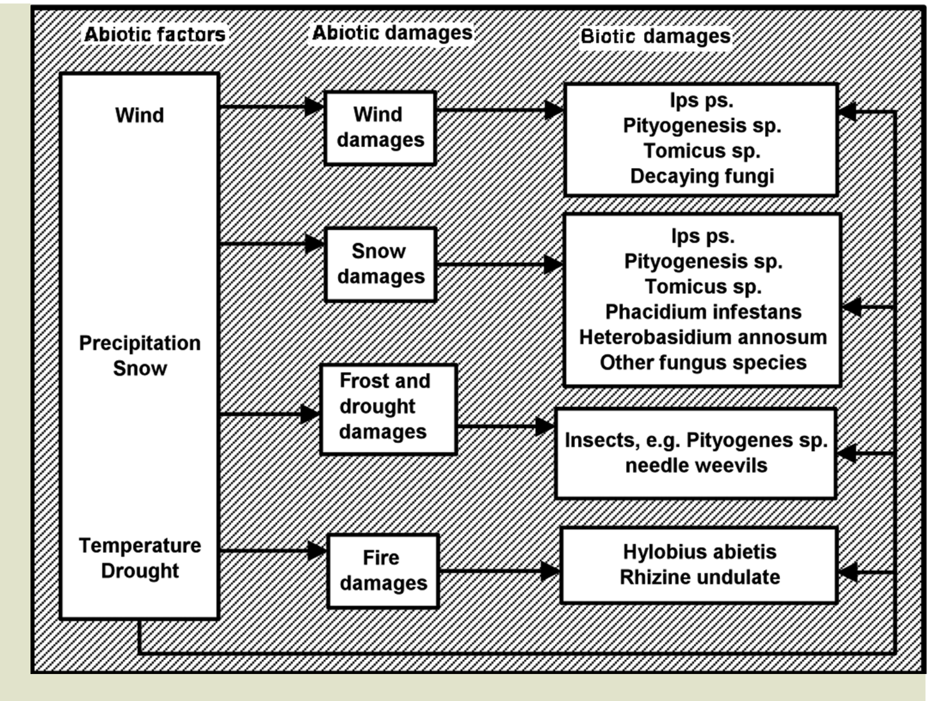

Figure 3.10 Weather and climate not only directly impact trees and trigger outbreaks of many populations of damaging insects and fungi but may also indirectly cause additional damage through weakening host trees and allowing infestations to develop further, thus exacerbating the damage (Parry 2000).

side northern Europe, even though it has frequently occurred in imported timber elsewhere within the boreal domain (see also Chapter 2).

Interaction between abiotic and biotic disturbances: Climate change may increase the mortality of trees and increase the risk of abiotic and biotic damage in boreal forests. Mortality of trees is endogenously related to the growth and life cycle of trees or exogenously related to the abiotic (frost, wind, snow, fire) and biotic (insect and fungal pests) factors.

Whenever climate change increases the growth of trees, endogenous mortality is expected to be larger due to a more rapid life cycle. An increased endogenous mortality may also increase the risk of exogenous mortality, since major outbreaks of many damaging insects and fungi are closely related to the presence of dead or dying trees or trees weakened by abiotic damage (Figure 3.10). Trees broken and uprooted by wind and snow provide more breeding material for bark beetles and increase the susceptibility of the entire stand to further attacks, which may even lead to an epidemic outbreak (Christiansen and Bakke 1988). Subsequent high summer temperatures and concomitant possible drought may weaken tree growth, while further increasing the growth of insect populations through enhanced physiological activity and more generations during the growing season (e.g. Wermelinger and Seifert 1999, Schlyter et al. 2006).

Herbivores: Currently, large herbivores like moose (*Alces alces*) and white-tailed deer (*Odocoileus virginianus*) are one of most important factors defining species composition of young forests in northern Europe and central-eastern North America. Current and future distribution of large herbivores is affected by high temperature in summer and the depth and seasonal distribution of snow. Thinner snow cover makes it easier for moose to move in winter. However, moose become thermally stressed by temperatures above –5°C in winter and above 14°C in summer, and deer are stressed below 5°C in winter (Schwartz and Renecker 2007, see also Chapter 2). Furthermore, composition and palatability of tree species also influence the success of mammalian herbivores. For instance, elevated levels of CO_2 can decrease the palatability of birch (*Betula pendula*) for hare (*Lepus timidus*) (Mattson et al. 2004).

3.4.5 Future Opportunities and Services at Risk

Timber: For North America, IPCC (Denman et al. 2007, Fischlin et al. 2007) reports a slow increase (1% per decade) in forest growth in the boreal regions, where the growth is limited by the low summer temperature and short growing season. The increase is most probably in the ecotone between the boreal and tundra vegetation, indicating the northward shift of boreal forests. On the other hand, the forest growth may be reduced locally at water-limited sites due to increasing drought episodes, with a consequent northward shift of boreal forests. The same trends are expected for the Siberian boreal forests. Based on the expected changes in the primary productivity, a slight increase in the potential timber harvest may be possible in the boreal forests of North America, Russia and the Nordic countries.

Carbon: Climate change can affect high-latitude carbon cycling through changes in the regeneration and growth of trees and changes in decomposition of organic matter in the forest floor and mineral soil. These changes are caused by changes in tree-species composition and enhanced decay due to elevated temperatures. Both processes are further controlled by fire- and wind-induced disturbances with impacts on the regeneration, growth and decay. The increase in March and April temperatures in high-latitude boreal forests results in earlier snow melt and lengthens the growing season, which, along with higher summer temperatures and higher atmospheric $CO_{2,}$ should enhance the total carbon uptake and, thus, an increase in summer carbon gain, balancing the increasing winter respiration in the ecosystem (decomposition, respiration of living organisms).

Whenever climate change increases tree growth, one may expect enhanced carbon sequestration in trees and soils. This applies especially to the northern and middle boreal forests, where the growth and litter yield will increase more rapidly than the decomposition of soil organic matter. However, the mean total amount of carbon will probably remain smaller in the north than in the south, even though the productivity of forests in the south is likely to decrease in many instances. Over Finland, the increase in the total amount of sequestered carbon in upland sites may be close to 30% higher than today (Kellomäki et al. 2008, Box 3.1).

Fire releases carbon to the atmosphere but it also converts a small fraction of decomposable plant material into stable charcoal. In old-growth forests, fire reverses forest succession and creates younger forests with higher growth rates, but it alters also the soil's thermal and moisture conditions, affecting decomposition and the availability of soil nutrients (Kasischke et al. 1995). The effects of climate change on long-term carbon sequestration depend greatly on the characteristics of the fire regimes. Fire has a highly variable direct and long-term effect on carbon losses, which depend on fire intensity and extent. Both depend on soil moisture and the quantity and quality of litter and other organic material on the soil surface. Given recent observations and model projections (cf. Fischlin et al. 2007, p. 228), an increase in fire episodes throughout the boreal forests is very likely and is expected to substantially increase carbon emissions from the boreal domain. Direct and indirect fire-generated carbon emissions from boreal forests may even exceed 20% of the global emissions from all biomass burning (Conard and Ivanova 1997).

Epidemic insect outbreaks can release significant amounts of carbon to the atmosphere. Pest insects are also of high relevance in the boreal domain and affect an area about 50 times larger than fire with a significantly larger economic impact (e.g. Logan et al. 2003), as the following example demonstrates. Warming at the end of the last century has allowed the build-up of significant outbreaks of the mountain pine beetle (*Dendroctonus ponderosae*), allowing the pest to invade new territories. Thanks to colder winter temperatures in the first half of the 20th century, this species used to cause no outbreaks in western Canada (Carroll et al. 2004). The recent, unprecedented outbreaks were estimated to continue to release up to 2020 large amounts of carbon, i.e. 270 MtC, turning those forests from a small sink into a large source (Kurz et al. 2008).

Biodiversity: The ecotone between boreal forest and tundra is a prominent feature of the northern boreal region, with a high value for biodiversity. In general, the higher productivity of boreal forests may increase species-richness and biodiversity in the long term, especially in the northern parts of the boreal zone. On the other hand, the change in tree-species composition alters substantially the properties of forest habitats. This may imply reduced success for true taiga species, which may be partly replaced by more southern species even in the central and northern parts of the boreal zone. However, assuming current management, the amount of decaying wood may increase in a changing climate. This is due to the higher primary productivity and faster maturation of trees, which result in a shorter life span of trees and increase the mortality of trees that are not removed by felling. More dead wood may increase the success of many rare and endangered species that fully depend on decaying dead wood (Kellomäki et al. 2001).

Climate change will eventually expand the treeline communities northwards. However, the geographic ranges of certain species such as white spruce (*Picea glauca*) are not expected to shift uniformly. In Alaska and north-western Canada, northward advances may be slow in the dry central parts of the northern boreal forest, whereas improved growth conditions are expected in moister habitats. On the other hand, the large-scale death of white spruce forests due to more prevalent attacks of spruce bark beetle (*Dendroctonus rufipennis*) will probably reduce the existing species richness temporarily, while giving space for more southern species to invade. At the southern tundra boundary in North America, spruce may be replaced by aspen (*Populus tremuloides*) (Hogg and Hurdle 1995). The poor success of Norway spruce

(*Picea abies*) in the southern ecotone between the boreal and temperate vegetation zones in northern Europe is identified in several model exercises (Kellomäki et al. 2008). All these processes are likely to reduce the role of taiga species in these locations, while it is highly uncertain precisely which species may replace them or at what time.

3.4.6 Key Vulnerabilities

Boreal forests from the Northern Hemisphere provide key provisioning services. Although primary productivity is still expected to increase, in general, for climate-change scenarios from the clusters *growth* and *stable* in the boreal domain and, in particular, in its northern forests, the same climate change is also likely to have negative impacts; these may particularly affect the currently more productive southern forests and all boreal forests through fire (e.g. Stocks et al. 2002, Fischlin et al. 2007) and insect incidences (e.g. Fischlin et al. 2007, Kurz et al. 2008). Both effects appear to have the potential for being significant for key services, but the overall balance can only be assessed if quantitative estimates become more reliable. This corroborates previous IPCC assessments pointing at the wide swings in provisioning services of boreal forests (Solomon 1996). On the other hand relatively well-understood temporal characteristics of these responses indicate high risks for overall negative effects to occur, more likely than not from now on and during a possibly century-long transient period.

3.5 Temperate Domain

3.5.1 Types of Temperate Forest

Temperate forests are found at mid-latitudes (~30° and <50° N and S, respectively) and cover an area of about 10.4 Mkm^2 (Fischlin et al. 2007). They can be grouped into warm deciduous or summergreen, and broad-leaved or conifer south-temperate forests (Olson et al. 1983) or, alternatively, into the ecological zones: oceanic, continental and mountain temperate forests (FAO 2001b, FAO 2006). Annual mean temperatures are below 17°C but above 6°C, annual precipitation is at least 500 mm and there is a markedly cool winter period (Walter 1979). Within the temperate zone one finds steep climatic gradients of precipitation and temperature, in particular with changes in altitude, and from oceanic to continental areas, all resulting in a considerable diversity among temperate forest types. The biome occurs primarily in the Northern Hemisphere, and in the south it is limited to areas of Chile, Argentina, New Zealand, South Africa and eastern Australia. Canada, the USA and Russia together hold 70% of temperate forests. China also has extensive areas of temperate forests, although most of these are second-growth and plantations. Several of the smaller areas maintain high levels of endemic biodiversity in part owing to their long-term isolation. They are important hotspots, including some in South Africa, New Zealand and Australia. Temperate forests are dominated by broad-leaf species with smaller amounts of evergreen broad-leaf and needle-leaf species (Melillo et al. 1993).

Tree species diversity is highest in the Asian temperate zone, where >900 woody species occur, nearly four times the North American species richness for this biome (Ohsawa 1995). Common species include the oaks (*Quercus*), eucalypts (*Eucalyptus*), acacias (*Acacia*), beeches (*Fagus* and *Nothofagus*), pines (*Pinus*) and birches (*Betula*). Temperate rainforests occur in several areas including western North America, Australia, New Zealand, Chile, South Africa and south-eastern Asia. The three main natural disturbances in temperate forests are wind, fire and herbivory (Frelich 2002). These vary in importance depending on rainfall and temperature (cool v. warm, and forest composition is mediated through the long-term interaction among these disturbance types (Kira 1991). Climate change is predicted to alter all these disturbances (Meehl et al. 2007), leading to uncertainty in future forest species composition. Changes as a result of climate-mediated herbivory are discussed in Chapter 2, but these are considerably altered by anthropogenic influences such as land-use change, introduction of invasive alien species, and predator control.

Annual net primary productivity of natural northern temperate forests is 900–1000 $\mathrm{g/m}^2$ while more southerly stands can produce up to 1400 $\mathrm{g/m}^2$ (Lieth and Whittaker 1975). Soil carbon ranges from 1 to >4 $\mathrm{kg/m}^2$, depending on forest type in North America (Finzi et al. 1998) but up to 7.7 $\mathrm{kg/m}^2$ in central Europe (Balesdent et al. 1993). The primary productivity of most temperate forest ecosystems may be limited by the availability of nitrogen, except where moisture may limit the system (Aber 1992, Rastetter et al. 2005). However, the addition of nitrogen increases productivity only to a certain extent, limited later by other minerals such as aluminium or as a result of elevated pH (e.g. Schulze 1989). Furthermore the C:N ratio is important for the rate at which carbon may be sequestered; hence there is debate over the functionality of nitrogen fertilization (e.g. Nadelhoffer et al. 1999, Martin et al. 2001). Recent evidence suggests, however, that in temperate forests, after the effects of disturbance have been accounted for, net carbon sequestration is mostly driven by nitrogen deposition, coming mostly from anthropogenic activities, and that this relationship is positive over a range of nitrogen deposition rates (Magnani et al. 2007). However, global warming seems to be reducing the total carbon uptake in temperate forests through losses from the soil in autumn, offsetting spring gains, which suggests that temperate forests may become in the future relatively poor carbon sinks if warming continues (Piao et al. 2008). Altered nitrogen and carbon levels will have effects on species composition of these forests (Parry 2000). Furthermore, while forest age is also a factor in carbon cycling, Luysseart et al. (2008) suggested that old-growth forests continue to sequester carbon at a high rate of 1.36 + 0.5 GtC/a. In North America, the recent invasion by native and exotic earthworms is altering forest function and the C:N ratios (e.g. Bohlen et al. 2004). This northward expansion will be enhanced by climate change and alter ecosystem properties, including the rate of carbon loss.

Owing to the large number of people living near temperate forests or in temperate forested lands, the entire range of goods and services from these forests is important. However, because of their mid-latitude position in a climate highly favourable to humans, temperate forests are, historically and pre-historically, the most extensively altered forests among all forest biomes. In Europe, temperate forests cover 160 million ha, which represents <50% of the original forest cover, and in both Europe and the USA, less than 1% of these deciduous forests are original primary forests (Reich and Frelich 2002). Furthermore, high levels of pollutants have entered many temperate forest areas since the beginning of the Industrial Revolution. Many of the major factors that influence these forests are due to human activities, including land-use and landscape fragmentation, pollution, soil nutrients and chemistry, fire suppression, alteration to herbivore populations, species loss, alien invasive species, and now climate change (Reich and Frelich 2002).

3.5.2 Main Services Provided

Of particular concern in the temperate domain is the loss of provisioning and cultural services, partly due to the high primary productivity of temperate forests at sites with high water availability and high nutrient levels, and to their proximity to densely populated areas in industrialized countries. In the latter areas, temperate forests increasingly provide many socio-economic and cultural services and often serve conservation goals directly or indirectly. More and more tourism, leisure and sports activities take place in those forests. The many species they harbour get under considerable anthropogenic pressure due to intensification of agricultural practices and urbanization. Only recently, in particular in western Europe, a reverse of some of these trends could be observed: Pressure on forests lessened due to intensification of agriculture (e.g. Rounsevell et al. 2006).

Timber: Northern Hemisphere temperate forests are important sources of round wood and pulpwood, with the types of products varying greatly depending on the region and forest types. Wood produced by Northern Hemisphere forests was largely responsible for the lumber supply of the rapid post-industrialization housing and urban growth of Western civilizations. Now, the region's timber production is increasingly being met from plantations (Easterling et al. 2007, Bosworth et al. 2008) and round-wood production for these forests is projected to increase some 25% over the next 20 years (Turner et al. 2006).

Carbon: Northern Hemisphere temperate forests are important sinks for atmospheric CO_2 (Goodale et al. 2002). However, estimates of the magnitude and distributions of this sink vary greatly and depend on temperature, nitrogen fertilization, fire, invasive species, age of the forest and levels of pollution. Temperate forest regions in the highly productive forests of western Europe (Liski et al. 2002), eastern USA (Birdsey et al. 2006) and east Asia (Saigusa et al. 2008) are known to be robust carbon sinks, although increased temperature may reduce this effect through loss of carbon from soils (Piao et al. 2008). Current carbon sink strength estimates for northern Caucasian forests, which were influenced by recent trends of forest exploitation, were found to be presently three times smaller than the figure estimated for 1970–1990 and five times smaller compared to the period 1950–1970 (Bakaeva and Zamolodchikov 2008). Less certain are the sink strengths of old-growth temperate forests (Pregitzer and Euskirchen 2004, Luyssaert et al. 2008), which, until recently, were thought not to be as strong sinks as younger, more rapidly growing forests. Weaker carbon sinks or even carbon losses are also seen for temperate forests in areas prone to periodic drought, such as the western USA, southern Europe, many parts of Australia and the southern Russian Far East (Moiseev and Alyabina 2007).

Non-timber products and uses: The world's temperate forests are responsible for a host of regionally dependent non-timber forest products and services. Firewood and indigenous people's speciality products such as botanical and medicinal products, mushrooms, fruits and nuts, and crafts materials are all supplied from these forests. Fuelwood is an important product from temperate forests, which provide roughly 10% of the global fuelwood harvest (FAOSTAT 2003). In addition, these forests protect water quality and quantity by harbouring water reservoirs for many of the world's major cities, provide wildlife habitat for game birds and animals, and are important refuges of biologically diverse fauna and flora. Increasingly, the world's temperate forests are utilized for recreational activities such as hiking, cross-country skiing and camping.

3.5.3 Current Opportunities and Vulnerabilities

Current climatic trends (e.g. Trenberth et al. 2007) indicate an increase in primary productivity in all humid regions, particularly the mesic regions and to a lesser degree the oceanic regions – due to slower warming rates. These forests are therefore expected to continue the strengthening of the carbon sequestration regulating services in the near future (at least roughly two decades, e.g. Fischlin et al. 2007, p. 222, Fig. 4.2) and do currently mitigate climate change. However, this is challenged by other authors as highly dependent on temperature, nitrogen and other mineral levels (e.g. Hungate et al. 2003, comprehensively reviewed by Fischlin et al. 2007 [section 4.4.1], Magnani et al. 2007, Piao et al. 2008). Moreover, this increased productivity for the near future is expected to sustain provisioning services, notably lumber production in the short- and mid-term (Easterling et al. 2007, IPCC 2007c).

However, primary productivity is expected to decrease as the drier regions of the temperate domain covering semi-arid to sub-humic climates in regions adjacent to the subtropical domain continue to experience more drought spells and, in general, a decrease in summer precipitation. Moreover, increased fire frequencies and areas involved and/or more intense fire events are expected as a result. Drought as well as fires will also lead to substantive carbon releases. For instance, in summer 2003 drought impacts on vegetation (Gobron et al. 2005, Lobo and Maisongrande 2006) reduced gross primary production in Europe by 30%. Respiration was also reduced, but to a lesser degree. The overall effect was a net carbon loss of 0.5 PgC/a (Ciais et al. 2005). Record-breaking incidences of wildfires in terms of spatial extent were observed throughout Europe in 2003 (Barbosa et al. 2003), with roughly 650 000 ha of forest burnt across the Continent (De Bono et al. 2004). Finally, as warming continues to accelerate according to the scenarios in the clusters *fast growth* or *growth* (IPCC 2007d [p. 45, section 3.2.1], IPCC 2007c [p. 12–13]), many forests in the temperate domain currently showing increasing productivity are likely to switch into a mode where production decreases as their climate moves toward sub-humic or even drier conditions, leading at some unknown point in the future to an overall productivity loss in the temperate domain (Lucht et al. 2006, Schaphoff et al. 2006, Scholze et al. 2006, Canadell et al. 2007, Fischlin et al. 2007, Raupach et al. 2007).

3.5.4 Projected Future Impacts and Autonomous Adaptation

Forest productivity has been increasing in two major temperate forest regions: eastern North America (Soule and Knapp 2006, Field et al. 2007b), and western Europe (Carrer and Urbinati 2006). This is thought to be from increasing CO_2 in the atmosphere (Field et al. 2007b), anthropogenic nitrogen deposition (Hyvönen et al. 2007, Magnani et al. 2007), warming temperatures (Marshall et al. 2008), and associated longer growing seasons (Chmielewski and Rötzer 2001, Parmesan 2006). Most models predict continuing trends of modestly increasing forest productivity in eastern North America and western Europe over this century (Alcamo et al. 2007, Field et al. 2007b, Alo and Wang 2008). Regional declines in forest productivity have also been seen in some areas of temperate forests due primarily to water scarcity as a result of recent droughts in Australia (Pitman et al. 2007), western North America (Breshears et al. 2005, Grant et al. 2006, Cook et al. 2007), and the European heat wave of 2003 (Schär et al. 2004, Ciais et al. 2005). There is a high likelihood of decreased summer precipitation and there is a high probability of an increased occurrence of heat waves over the next century (Alcamo et al. 2007, Field et al. 2007b) so that occurrences of drought will become more frequent, particularly at the southern end of the temperate forests from the Northern Hemisphere and in Australia. Thus, these events are likely to continue to have a negative impact on forest productivity in those areas.

Projections for the time near the end of the next century generally suggest decreasing growth and a reduction in primary productivity enhancement as temperatures warm, CO_2 saturation is reached for photosynthetic enhancement, and reduced summer precipitation all interact to decrease temperate zone primary productivity (for lodgepole pine Rehfeldt et al. 2001, Lucht et al. 2006, Scholze et al. 2006, Alo and Wang 2008). What is further contributing to decreased long-term primary productivity in some regions of temperate forests under climate change is the projected increased occurrence of forest pests, particularly in drought-stressed regions (Williams et al. 2000, Williams and Liebhold 2002), prolonging current trends of recent climate change-induced pest infestations (e.g. Logan and Powell 2001, Tran et al. 2007, Friedenberg et al. 2008).

Timber: Sustainable forest management is becoming more common in productive temperate forests, increasing the likelihood of sustainable management in the face of climate change. Temperate forest plantations are increasing and these are expected to provide an ever-increasing percentage of the roundwood products over the next century (Sedjo 1992, Birdsey et al. 2006). Timber volumes are likely to follow similar trends to those of primary productivity as discussed above.

Carbon: Climate change resulting from the enhanced greenhouse effect, together with the direct effects of increasing amounts of atmospheric CO_2 and increasing nitrogen deposition, are all expected to produce changes in the cycling of carbon in the temperate forest ecosystem (Morales et al. 2007). Increases in carbon sink strength are expected in some productive regions under intensive forest management such as central western Europe (Morales et al. 2007), while decreasing sink strength is projected for temperate forest areas facing increasing drought occurrence, such as southern western Europe (Morales et al. 2007), the southern part of the Russian Plain (Golubyatnikov and Denisenko 2001, Kolomyts 2006) and in ageing eastern North American forests (Birdsey et al. 2006).

Biodiversity: One of the most dramatic predictions of temperate forest model projections is the substantial range shifts which are expected to occur at the northern and southern borders of temperate forest (Iverson and Prasad 2001, Parmesan 2006, Fischlin et al. 2007, Gessler et al. 2007) and at higher levels on mountains (Breshears et al. 2008, Kelly and Goulden 2008). The ranges of northern temperate forests are predicted to extend into the boreal forest range in the north and upward on mountains (Iverson and Prasad 2001, Ohlemüller et al. 2006, Fischlin et al. 2007, Golubyatnikov and Denisenko 2007, Figure 4.3, p. 238). The distribution of temperate broad-leaved tree species is typically limited by low winter temperatures (Perry et al. 2008). Since the latter are projected to rise more rapidly than summer temperatures in Europe and North America (Christensen et al. 2007, sections 11.3, 11.5), temperate broad-leaved tree species may profit and invade currently boreal areas more rapidly than other temperate species. The area of temperate forests is projected to decrease at boundaries with the forest-steppe biome (Kolomyts 2006, Golubyatnikov and Denisenko 2007).

A major concern for biodiversity is that some species and certainly many populations within species may not be able to migrate quickly enough to find their suitable temperature niches due to the unprecedented rapidness of global warming (Fischlin et al. 2007). The few studies that have shown evidence of range shifts have reflected the limited capacity to disperse (Davis et al. 1986, Davis 1989). The main form of forest tree migration is via seed dispersal. However, only a few temperate-zone tree species, such as trembling aspen, have those very small seeds displayed in ultra-light pubescence so that they are readily dispersed by wind over long distances. However, global warming is expected to expatriate even trembling aspen from the north-eastern US temperate forests (Iverson and Prasad 2001). On the other hand, recent modelling studies incorporating population (O'Neill et al. 2008) and provenance variation (Reich and Oleksyn 2008) suggest that there is more plasticity than previously thought for response of temperate-forest trees to global warming in some regions.

Drought: Models suggest that the greatest climate-change threat to temperate forest ecosystems is reduced summer precipitation, leading to increased frequency and severity of drought (Christensen et al. 2007, Fischlin et al. 2007, IPCC 2007c, Meehl et al. 2007, Schneider et al. 2007, Chapter 3.2). This will probably be most prominent in temperate forest regions that have already been characterized as prone to drought stress, such as the western USA, northern China, southern Europe and the Mediterranean, and Australia (Photo 3.1). However, drought may also have widespread impacts on other northern temperate forests, particularly in limiting growth (Ciais et al. 2005, Leal et al. 2008) and triggering dieback and decline (Breshears et al. 2005) for species or populations within species near the southern borders of their range, such as paper birch in the Lake States of the USA (Jones et al. 1993), Austrian pine in the Alps (Leal et al. 2008), and European beech in southern Europe (Gessler et al. 2007).

Some effects of drought on primary productivity may be offset by near-term increases in water-use efficiency in a CO_2-enriched atmosphere (Aber et al. 2001) or by soil fertility (Hanson and Weltzin 2000). In the long term, however, productivity of temperate forests constrained by drought in the next century will be reduced, and declines and dieback episodes will occur more commonly under global warming (Breda et al. 2006). Drought during canopy development can have a long-lasting impact on carbon balance (Noormets et al. 2008). Drought-stricken forests are also more susceptible to opportunistic pests and fire (e.g. Hanson and Weltzin 2000). Together, these related effects can potentially change large areas of temperate forest ecosystems from carbon sinks to sources.

Fire: Fire is expected to be an ever increasing problem over the next century in the temperate forest as summer precipitation decreases, temperatures increase and drying conditions predominate, particularly in the Australian temperate forests (Pitman et al. 2007), as well as those in western North America (Breda et al. 2006, Cook et al. 2007), southern Europe (Ohlemüller et al. 2006), and northern Asia (Groisman et al. 2007).

Pests: Warming temperatures in temperate forests and increased occurrence of water stress are both likely to have important consequences for pest outbreaks. Warmer temperatures will mean more rapid growth of insects, shorter generation times for insects, and movement of temperature-sensitive insects into more poleward regions (Marshall et al. 2008).

John Innes

Photo 3.1 Dieback of *Eucalyptus gunnii* in the central Highlands of Tasmania, Australia. While the exact sequence of events leading to the tree mortality is uncertain, drought has been strongly implicated.

Unprecedented mountain pine beetle outbreaks have already been documented in northern British Columbia, Canada, related to global warming (Kurz et al. 2008), birch defoliations are extending farther north into Fennoscandia (Jepsen et al. 2008) than previously, and the highly damaging processionary moth is expanding northward and into the mountains from its traditional Mediterranean distribution (Battisti et al. 2005, Battisti et al. 2006).

Furthermore, warmer temperatures tend to remove bioclimatic barriers to the spread of alien pests, pathogens and plants. These alien invasives can quickly and permanently alter the composition of forests: for example, the emerald ash borer and hemlock woolly adelgid in eastern North America, sirex woodwasp in the southern hemisphere, pinewood nematode in Asia and western Europe, and sudden oak death in western North America and Europe (Dwinell 1997, Bergot et al. 2004, Butin et al. 2005, Hurley et al. 2007, Anulewicz et al. 2008). Some other insects and diseases are expected to increase in areas impacted by tropospheric O_3 (Chakraborty et al. 2008) or drought (Desprez-Loustau et al. 2006, Desprez-Loustau et al. 2007).

Under all scenarios of climate change (this report, sub-chapter 3.2.4), it seems likely that biological disturbance from established pests and pathogens will tend to increase on the warming poleward margins of temperate forests. Change will also occur along any other margins where water availability for trees is going either up or down. Furthermore, increasing commerce, in combination with modest climatic change, is very likely to produce additional biological invasions that will lead to further changes (generally increases) in forest disturbance. On the plus side, climate change will probably produce net decreases in pestilence in some regions (perhaps generally in subequatorial margins of temperate forests), and primary productivity will tend to go up overall, meaning that more losses to pests and pathogens can potentially be tolerated without losses in ecosystem services. However, even if the average level of pestilence remains the same (but probabilities change among regions), there would still be a tendency for transient reductions in the extent of mature forest because disturbance reduces a mature forest quickly while new mature forests can arise only slowly.

Other Disturbances: While fire, pest outbreaks and extreme weather events are well known to shape ecosystems (Field et al. 2007a) the contribution of land-use change is a very large driver of the temperate forest carbon budget, both in the Northern (Breshears and Allen 2002, Easterling and Apps 2005, Albani et al. 2006) and Southern (Wilson et al. 2005) Hemispheres. For example, in the northern temperate zone, increasing carbon stocks were seen in the USA during the past century as forests were regrown after extensive early logging, and as increas-

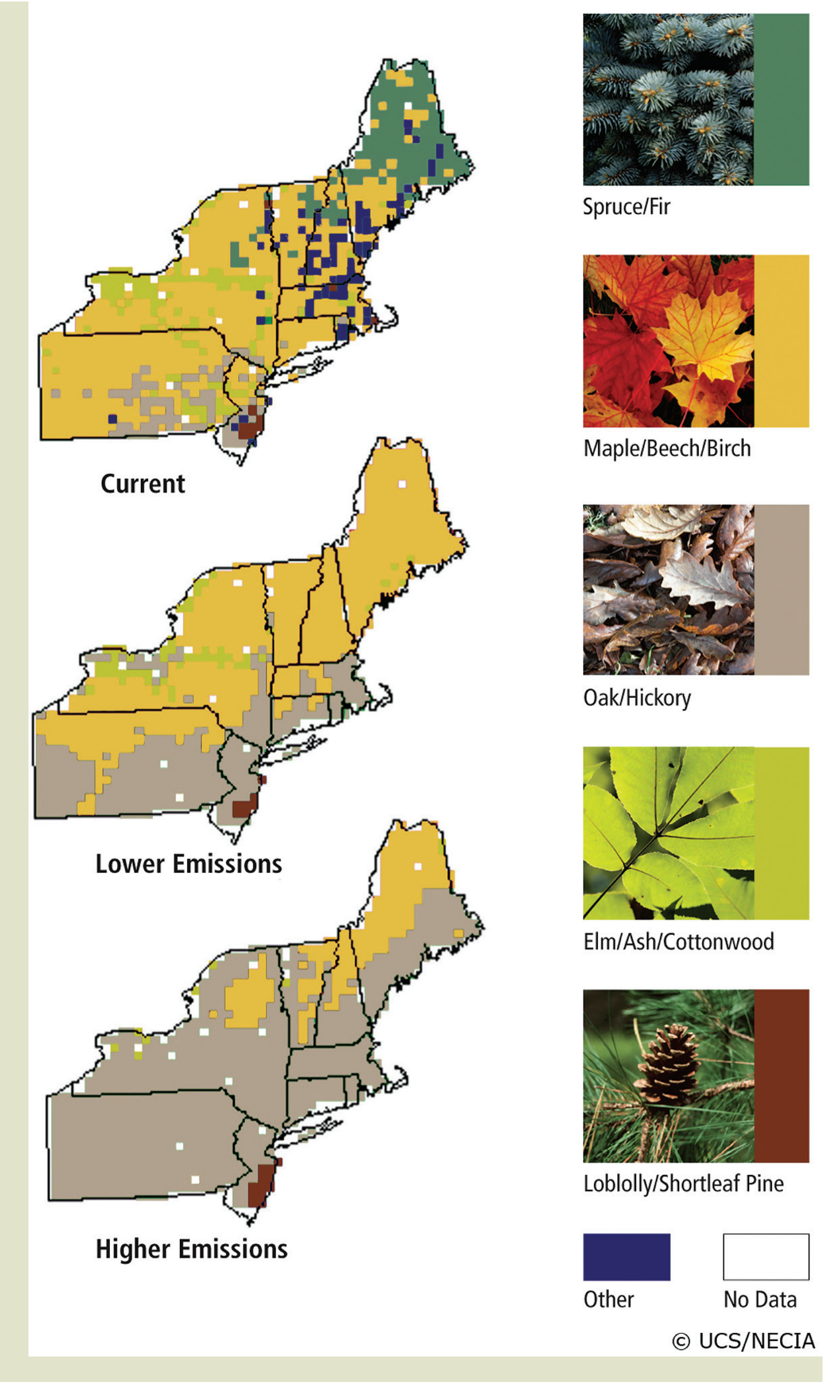

Figure 3.11 Projected distribution of forest types by major species groups of temperate forests in north-eastern USA from Frumhoff et al. (2007, Reprinted with permission of the Union of Concerned Scientists). Lower emissions correspond to a scenario from cluster *stable*, higher emissions to one from cluster *growth*.

ing amounts of marginal farm land were returned to forests (Caspersen et al. 2000, Birdsey et al. 2006). Without substantial carbon management practices such as the development of extensive energy plantations, the rate of carbon sequestration for these previously disturbed forests is expected to diminish over the next century as these forests mature (Albani et al. 2006, Birdsey et al. 2006).

Conversion of large areas of Southern Hemisphere temperate forests to exotic species plantations continues to be a concern for the next century (Wilson et al. 2005). Among the concerns regarding conversion in biologically and ecologically diverse areas is that habitat fragmentation will exacerbate climate-change effects related to species migrations with global warming (Honnay et al. 2002).

Air Pollution: Most industrialized and some of the largest urban areas are within the temperate domain. Consequently, temperate forests are particularly exposed to air pollution. Total deposition of nitrogen to temperate forests, in wet or dry, and oxidized or reduced form, is between 1 and 100 kg/ha/a (Hyvönen et al. 2007). The larger amounts are from industrialized regions such as the north-eastern USA and Central Europe. While this nitrogen has generally been thought to have a positive effect by stimulating primary productivity, as most forests occur on N-limited soils (Lebauer and Treseder 2008), there is a great deal of uncertainty as to whether detrimental effects of nitrogen saturation will eventually appear (Aber et al. 1998). A lively debate has started over the role of nitrogen deposition in the future carbon sequestration potential of temperate forests (Högberg 2007, Magnani et al. 2007, Sutton et al. 2008).

A second air pollutant projected to increase in the next century over large areas of temperate forests that are downwind of major metropolitan areas, is tropospheric ozone (Meehl et al. 2007, section 10.4.3). Tropospheric O_3 is a secondary pollutant that is generated from nitrogen oxides reacting with volatile organic compounds in the presence of sunlight. It is a highly reactive pollutant that can reduce the growth and carbon sequestration capacity of sensitive species (Karnosky et al. 2005, McLaughlin et al. 2007). Ozone is generally increased during extreme heat events such as the European heat wave of 2003 (Guerova and Jones 2007, Solberg et al. 2008), leading scientists to predict increasing ozone over large portions of the temperate forest in the next century as global warming continues (Fowler et al. 1999, Vautard et al. 2007, Andreani-Aksoyoglu et al. 2008).

As with all biomes, forest types and tree species are projected to change their distributions with climate warming. For example, in the north-eastern USA and south-eastern Canada, tree species are predicted to range northwards by up to 700 km, and certain forest types that occur unusually far south owing to montane conditions, are expected to disappear (Figure 3.11, Price and Scott 2006, Frumhoff et al. 2007). Similarly in Europe, reduced growth is already seen in southern-growing *Fagus sylvatica* (Jump et al. 2006). All authors agree that there is a high degree of uncertainty surrounding the relative species composition of future temperate forests, but that the area will remain well forested.

3.5.5 Future Opportunities and Services at Risk

Globally, temperate forests are among the world's most stable forests. As such, they are less likely to suffer severe consequences from climate change than some other forest types. However, there are rather dramatic regional risks that can have large impacts on temperate forests. Among these, the most widespread are related to projected decreases in summer rain as global warming continues. Droughts over expanded regions and with greater intensity and frequency are predicted for large areas of temperate forests over the next century. These droughts will probably lead to more frequent fires and will also predispose large areas of forests to opportunistic pests and pathogens such as bark beetles, Armillaria and wilt diseases. Regions of increasing precipitation may experience decreased risks from these pests and pathogens but increased risks from others.

In many temperate forests windthrow is the most important natural disturbance (e.g. Thürig et al. 2005). Since extra-tropical stormtracks are projected to move poleward (IPCC 2007d, p. 46) frequency and even intensity may increase and cause major forest disturbances. The fact that current climate models underestimate recent observations (IPCC 2007b, p. 10) is of particular relevance in a context of managing climate risks for temperate forests.

Finally, sensitivity to increasing air pollution loads, particularly nitrogen deposition and tropospheric O_3, will impact large areas of the northern temperate forest over the next century. Humans will also increasingly impact temperate forests as they cause land-use change and habitat fragmentation, which will interact with the above-mentioned risks to exacerbate biodiversity issues relating to species migration and wildlife habitat management.

3.5.6 Key Vulnerabilities

In some productive temperate regions, moderate climate change is expected to lead to improvements in timber production as well as regulating services, such as increases in carbon-sink strength, in particular under intensive forest management. However, towards the end of this century and beyond, in particular for scenarios from clusters *growth* and *fast growth*, reductions in primary productivity are projected due to above optimum temperatures, declining water availability during the growing season, and CO_2 saturation effects. Forests that are already prone to drought stress such as the western USA, northern China, southern Europe and the Mediterranean, and Australia, are projected to be affected not only by reduced summer precipitation but also by increased frequency and severity of drought and fires, with all the concomitant negative effects. In the temperate domain, air pollution is expected to interact with climate change; while the fertilization effects from nitrogen deposition are still highly uncertain, pollutants such as O_3 are known to diminish primary productivity, impacting provisioning as well as regulating services. Biological disturbances from established pests and pathogens will tend to increase on the warming poleward margins of temperate forests. Similar effects are expected along any other margins where water availability for trees is diminished. While intensification of agriculture may lead in some areas to a decrease of pressures on vegetation and wildlife in forested or woodland areas serving often as refuges, the current trends of habitat fragmentation and impoverishment of the landscape are expected to continue, including increasing opportunities for invasive alien species. This is likely to increase the many threats to biodiversity in the temperate domain, exacerbating the extinction risks climate change is causing for many species inhabiting the temperate domain.

3.6 Subtropical Domain

3.6.1 Types of Subtropical Forests

Subtropical regions are generally found in mid-latitudes between 25° and 40° in the Southern and Northern hemispheres. As described and mapped by the Food and Agriculture Organization of the United Nations (FAO 2001b, 2001a), the subtropical domain includes areas with at least eight months of over 10°C mean monthly temperatures. Subtropical forest areas include regions of humid forest, dry forest, steppe or savanna woodlands and subtropical mountain systems.

The humid subtropical forests are found in regions that receive >1000 mm of annual rainfall, with no distinct dry season, and where mean annual temperatures range from about 15–21°C. The four main regions of humid forest include south-eastern USA, south-eastern areas of South America (including parts of southern Brazil, Uruguay and Argentina), southern China and eastern coastal Australia. There are also some smaller regions, such as in the south-eastern coastal part of South Africa. Though much of these regions have been cleared for agriculture, they often support well-developed native and plantation forests, including important commercial species. For example, *Pinus elliottii* var. *elliottii* is native to the subtropical humid region of the USA and is an important timber species. It is also grown as a plantation species in some of the other major subtropical humid regions. Similarly, *Eucalyptus grandis* is a significant native forest species in humid subtropical Australia, but it has proved useful as a plantation species in parts of the other major humid regions. However, the use of eucalypts in humid subtropical regions of the USA and China is limited by occasional frost events associated with cold air movements from the north.

Major regions of subtropical dry forest include Mediterranean areas (including parts of Spain, Italy, Greece, Turkey and North Africa), southern Chile, parts of California, coastal parts of the Western Cape region of South Africa and the south west of Western Australia. These regions have hot dry summers and humid mild winters, with annual rainfall in the 400–900 mm range. FAO (2001a) described typical forest types including: Maquis dominated by *Quercus ilex* in the Mediterranean region; chaparral in California; Chilean Matorral; Fynbos in the Cape Region of South Africa; and Eucalyptus forest in south-west Australia.

Subtropical steppe or savanna areas are semi-arid with long hot summers and generally short mild winters. They have an annual rainfall ranging from 250 mm to about 1000 mm where they transition into subtropical humid forest. Total annual evaporation generally exceeds precipitation. Grasslands dominate in lower rainfall areas with shrubs and trees becoming more common as rainfall increases. These areas include woodlands satisfying the forest definition of UNFCCC (2001). The regions include inland areas in eastern and western Australia, parts of Argentina and parts of south-central USA. There is a belt of subtropical steppe in northern Africa between the subtropical dry region and the Sahara desert, but the FAO classification does not recognize subtropical steppe regions in southern Africa as true forests.

Subtropical mountain systems in the FAO (2001a) classification are generally found at elevations of approximately 800–1000 m. The main subtropical mountain systems are found in parts of the Andes, central Mexico, south-western USA, the mountains of the Middle East, western parts of the Himalayas and the high veldt region of South Africa.

The CABI (2005) database lists 508 tree species from the northern latitudinal range and 238 species from southern latitudes, but not all listed species are endemic to the subtropical domain.

3.6.2 Main Services Provided

Humid subtropical forests have been extensively converted to timber plantations, mainly with exotic species, so their primary functions are wood production and water catchment. Major regions of the subtropical dry forests, especially the Mediterranean areas, are important for agriculture, soil conservation and tourism. One of the fastest-growing economic sectors in southern Africa is wildlife-based tourism, almost completely focused on subtropical forests as defined here. At about 9% of the regional GDP, tourism is already as important as the forestry and agricultural sectors in many countries (Scholes and Biggs 2004).

As described in CABI (2005) subtropical forests provide provisioning services such as 84 types of wood products and 19 non-wood products such as resins, oils and food. Subtropical forests provide 11 other land/environment services including regulating, supporting and cultural ecosystem services such as revegetation, land reclamation, soil improvement, soil conservation, erosion control, and aesthetic value.

3.6.3 Current Opportunities and Vulnerabilities

Opportunities: Subtropical species are partly already well adapted to warm and dry climates. There are many examples of species growing in managed forestry trials under considerably warmer conditions than those they experience within their natural distributions, i.e. their realized niche (discussed in Kirschbaum and Fischlin 1996, Box 1.3), or even in unmanaged forests if their dispersal is assisted by humans (Booth et al. 1988, Booth 2007). However, many subtropical species now exist in highly fragmented environments as islands of natural forest amongst oceans of agricultural land. Species at a particular location may not have access to new sites where they would be better adapted to the new climatic conditions. Less tolerant species may then decrease in abundance and hereby create for other, more tolerant resident species opportunities to become more abundant because of reduced competition. If well irrigated subtropical plantations can be highly productive, offering opportunities to contribute towards future demands for wood and other forest products.

Key vulnerabilities: Many species are vulnerable, since they have limited distributions and hence narrow climatic ranges, poor dispersal mechanisms, and are growing in areas of low relief. For example, an analysis using existing climatic ranges of 819 Eucalyptus species in the unmanaged dry subtropical zone, showed a large number of potentially vulnerable species (Hughes et al. 1996). This region is also relatively flat and eucalypt species have very poor dispersal mechanisms. While these species may indeed be at increased risk of extinction, it would be wrong to imply that a species will necessarily become extinct if climatic conditions become entirely different from those it currently experiences. For instance, species in mountainous areas may be able to colonize higher, cooler locations comparatively easily even if they have poor dispersal abilities. The lapse rate is typically about 0.5–0.7°C cooler per 100 m increase in elevation, a temperature change that corresponds to a poleward dispersal of ~100–200 km of flat land.

3.6.4 Projected Future Impacts and Autonomous Adaptation

Biodiversity: Though the impacts of climatic and atmospheric change on commercial forests may be significant, their vulnerability may generally not be very great. Potential impact is a function of exposure and sensitivity, while vulnerability is related to potential impact and adaptive capacity (Allen Consulting Group 2005). Commercially important species tend to be planted over wide geographic areas. Responses to disease problems have been demonstrated in the past, such as the replacement of susceptible eucalypt genotypes with resistant genotypes when guava rust became a problem in Brazil (Glen et al. 2007). Clearly, such adaptive capacity is most easily implemented in short-rotation species, so the longer the rotation the greater should be the concern with monitoring species performance under currently extreme conditions.

Productivity: Increasing CO_2 can affect tree growth through increased photosynthetic rates and through improved water-use efficiency (Steffen and Canadell 2005). However, the magnitude and extent to which effects are sustained under different conditions in different tree species are not clear. Booth et al. (2008) have summarized some of the conflicting observations that have been reported for Australia. Forest growth rates may well be increased in some cases by rising levels of atmospheric CO_2, but rising temperatures, higher evaporation rates and lower rainfall may lower growth rates in other cases. It is certain that there will be complex interactions. For example, benefits of increased water-use efficiency may not be realized in some cases because of poor soil nutrition.

Many subtropical forests, especially where water is limited, regularly experience daytime temperatures above 35 or even 40°C, and will do so more frequently in future. Temperature responses and adaptive potential at these extremes is an under-researched area. A possible consequence of increasing temperatures above the physiological optimum (which tends to be lower in C_3 than C_4 species) is declining primary productivity and decreasing soil and biomass carbon stocks. High temperatures and longer drought

Box 3.2 Impacts of climate change on biodiversity in South Africa

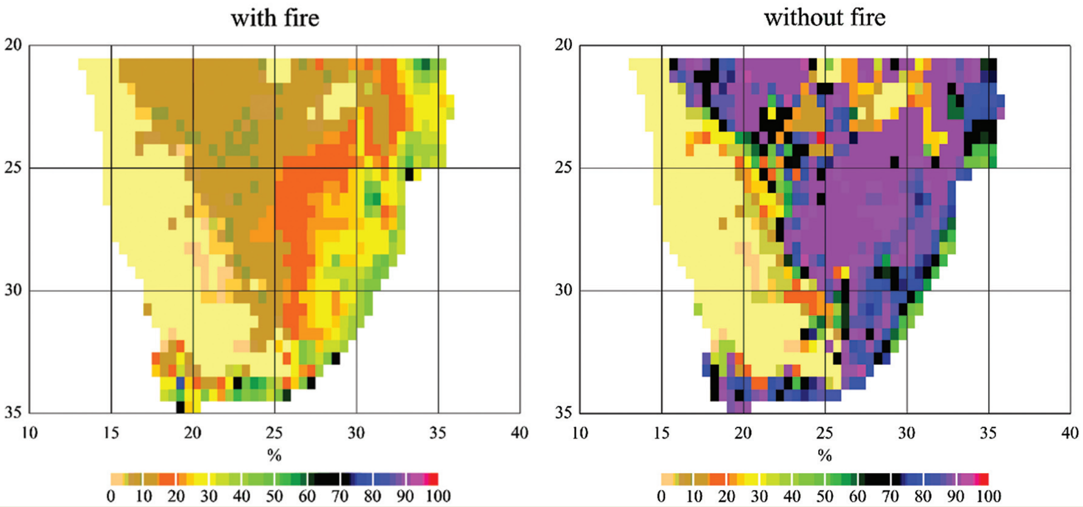

Figure 3.12 DGVM simulation of the current tree cover in southern Africa with (bottom left) and without (bottom right) the occurrence of fire (Bond et al. 2003b, copyright Elsevier. Reprinted with the permission of Elsevier. See also Bond et al. 2003a, Bond et al. 2005). The model indicates that the present preponderance of savanna woodlands and grasslands would probably be replaced by denser forests if fire frequency was much reduced or fire tolerance of trees and shrubs increases

Evidence of impacts of climate change on ecosystems is now emerging for southern Africa. For example, expansion of tree cover into the formerly open grasslands and savannas (bush encroachment) began around the 1960s, which may have been caused by the steadily rising global CO_2 concentration. In addition, the area considered climatically suitable for South Africa's seven existing terrestrial biomes could shrink by 35–55% by 2050 under scenarios from cluster *stable* (Midgley et al. 2001, p. 4). A disturbing prediction is the likelihood of the loss of the Succulent Karoo biome, home of the world's largest diversity of succulent flora and arguably the world's most botanically diverse arid region (Hannah et al. 2002). Countrywide, habitats are expected to shift along a west-to-east gradient of aridity, leading to an increased rate of extinction, as movement and available intact habitats are greatly restricted today. In southern Africa, fire-maintained systems have high species diversity compared to forest systems with fire suppression. The Cape Floral Kingdom (fynbos) is a biodiversity hotspot with over 7000 species, of which 68% occur nowhere else in the world (Gibbs 1987). The fynbos occurs in the winter rainfall regions and would be threatened by any change in rainfall that would alter the fire regime that is critical to the life cycle in the fynbos. With increasing atmospheric CO_2 levels, woody plants will reach fireproof levels more rapidly, as seen worldwide with tree density generally increasing in savanna woodlands (e.g. Bond and Midgley 2000). A great change in grassland biota will be expected if this level of bush encroachment into fire-dependent systems continues (e.g. Archer 1991). Simulations of current tree cover with and without fire occurrence show that savanna woodlands will be replaced by dense forests (Bond et al. 2003b, Bond et al. 2003a, Bond et al. 2005, Figure 3.12). Fire policy is contentious because human-induced fires are still frequent but not deliberately started by managers. The options are to manage for landscape heterogeneity, using patch mosaic burns, or use frequent intense burns over broader areas to maintain their current state.

increase vegetation flammability. According to the IPCC Fourth Assessment Report, drought stress has impacts on vegetation and has reduced gross primary production by as much as 30% in southern Europe, resulting in a net carbon source, particularly during the heat wave of 2003 (Fischlin et al. 2007, p. 217). In Portugal, the area burnt was almost twice that in the previous year and four times the 1980–2004 average.

Carbon: The structure, productivity and carbon balance of subtropical forests and savanna woodlands are sensitive to major climate-change drivers. CO_2 has contrasting direct effects on the dominant functional types – trees benefit from rising CO_2 but not from warming, while grasses benefit from warming but not from CO_2 increase – with uncertain, non-linear and rapid changes in ecosystem structures and carbon stock being likely (Fischlin et al. 2007). Carbon stocks are expected to be greatly reduced under more frequent disturbance, especially fire and droughts (e.g. Bond et al. 2005). In the savannas, reduced carbon sequestration is attributed to enhanced soil respiration through warming, fire regime changes and greater rainfall variability, but possible regional carbon gains through increased woody cover cannot be excluded.

Water: Climate change will bring drier, hotter and windier conditions to many regions, especially the areas with a mediterranean climate. These conditions will increase bushfire risks. Impacts of bushfires (as well as other factors such as drought) on species composition and re-growth, and consequent effects on catchment water yield, have been measured and modelled in Australia, but more for temperate than for subtropical forests. Factors such as bushfires affecting a site's capacity to store water in the canopy (via changes in canopy cover and leaf area index), litter layer (via changes in litter cover) and soil (via changes in soil water-holding capacity) will, in turn, affect water loss from interception, evaporation, transpiration and runoff.

Some rainfall runoff models using simple evaporation coefficients (e.g. Beare and Heaney 2002) have shown that climate change had the greatest effect in reducing annual stream flow in higher rainfall areas. To project reliably the influence on stream flow of various climate-change scenarios, modelling should account for the effect of changing climate on forest growth rates and how this will in turn affect evapotranspiration. For example, based on a projected 11% decrease in annual rainfall over 30 years, a change in water yield from a eucalypt-dominated catchment in south-western Australia of between 9 and 40%, depending on the changes in evaporation, resulted, but it was noted that confidence in the calculations of runoff could be significantly improved by using better estimates of the leaf-area index and potential evapotranspiration (Bari et al. 2005).

There is a statistically significant link between rainfall and stream flow and the El Niño-Southern Oscillation in eastern Australia (Australian Greenhouse Office 2003). In the drier regions of eastern Australian subtropical eucalypt-dominated woodland, the percentage change in runoff can be more than four times the percentage change in rainfall (Chiew and McMahon 2002). Most projections for decrease in runoff in the eastern Australian subtropical forest region are in the order of 7–35% by 2030 or 2050. In most parts of Australia, temperature increases alone have negligible impacts on runoff when compared with altered amounts of precipitation. However, most models used to project catchment water impacts of climate change, whether used in USA or Australia, do not take into account or adequately deal with vegetation impacts, such as the potential for reduced tree cover (Photo 3.2).

Dry land salinity is mainly a problem in the 400–800 mm rainfall zone, mostly in woodland-dominated 'crop-livestock' regions of Western Australia and the Murray-Darling Basin of eastern Australia (drier subtropics), and both land and stream are affected. A drier and hotter climate would result in reduced runoff and recharge to groundwater, with water tables being lower and hence salinity expression stabilized or reduced (Beare and Heaney 2002), a result already being experienced over the last decade in the northern wheat belt of Western Australia (George et al. 2008). This could be countered to some extent by an increased incidence of flooding. Native vegetation in these regions has been largely cleared for agriculture over the last century. The impact of planted forests (mainly eucalypts) on farm land to counter salinization would depend on growth and water-use efficiency as influenced by climate change.

Fire: Greater fire frequencies are already reported in the Mediterranean basin regions. Double CO_2 climate scenarios increase wildfire events by 40–50% in California and double fire risks in the Cape Fynbos, favouring re-sprouting plants, fire-tolerant shrub dominance in the Mediterranean basin, vegetation structural change in California and reducing net ecosystem productivity and, thus, carbon sequestration (Fischlin et al. 2007, p. 227).

Forests of subtropical areas are likely to be affected by changes in drought and fires. In both historical and future scenarios, fire is required for the co-existence of trees and grasses when deep soil water is available to trees. Simulations of tree/grass interactions under various climate-change scenarios indicate that more fires with higher temperatures resulted in decreased fuel moisture. Fire also increased in the deeply rooted grass scenarios because grass biomass, which serves as fine fuel source, was relatively high (Daly et al. 2000).

Pests and disease: For discussion of pests and disease in the subtropics and tropics, see this report, sub-chapter 3.7.4.

3.6.5 Future Opportunities and Services at Risk

Opportunities: Contrary to the pattern expected in boreal and temperate forests, both the frequency and intensity of fires in subtropical forests will eventually decrease after an initial phase of increase once rainfall has decreased so much that less grass fuel is available to support fires. Furthermore, the fraction of the landscape burnt tends to decrease with increasing human population density. A reduction in fire frequency and intensity, all else being equal, is expected to shift the tree/grass balance towards trees.

Ecological models do not suggest large near-term additional disturbances in native subtropical forests, and the largest impacts in the near future are likely to result from deforestation rather than from climate change (Gitay et al. 2001). However, many subtropical countries are increasing their share of the global timber market from plantations. Short-rotation exotic species, especially, are expected to be particularly suitable for adaptation during climate change, so that both tropical and subtropical countries could potentially benefit from climate change for increased timber production (Sohngen and Sedjo 2000). The effects in subtropical and tropical countries are directly linked to the size of higher primary productivity implied by climate change. It was projected that subtropical regions in Chile, Argentina, Brazil, South Africa, Australia and New Zealand could provide more than 30% of the market share in the middle of the century (Daigneault et al. 2008). If climate change drastically increases primary productivity in these plantations, large market impacts could result. Since most subtropical plantations focus on short-rotation species, of 10–20 years, timber managers can adjust and adapt rapidly if climate change has drastic effects. However, the sustainability of plantations is not beyond problems such as from pests or pathogens and on the long-run it may be preferable to manage plantations as part of an entire landscape within a framework of sustainable forest management (cf. e.g. Chapters 1, 6).

Vulnerabilities: Studies of the impacts of climate change on natural forest ecosystems projected contrasting scenarios, depending on precipitation patterns. In Mexico, simulations indicate that subtropical forests will increase in area because of projected increase in rainfall (Villers-Ruiz and Trejo–Vazquez 1998). In contrast, simulations for both Pakistan and Zimbabwe show a reduction in the area of natural forest ecosystems and an overall negative impact because of drier conditions (Matarira and Mwamuka 1996, Siddiqui et al. 1999). Drier conditions would also increase the risk of bushfires in many countries, especially in the Mediterranean basin.

The subtropical domain contains many key biodiversity hotspots in Latin America, southern Australia, the Fynbos or Succulent Karoo in South Africa, recognized as United Nations Educational, Scientific and Cultural Organization (UNESCO) World Heritage Sites. Many of these areas have been found in quantitative studies to be at particular risk from climate change (see also Box 3.2), since the majority of endemic species were projected to decline under a wide range of climate-change scenarios ranging from clusters *stable* to *growth* (e.g. Midgley et al. 2002, Thomas et al. 2004, Thuiller et al. 2006, Fischlin et al. 2007, Fitzpatrick et al. 2008).

Photo 3.2 Sub-tropical *Taxodium distichum* swamp in the Everglades, Florida, USA. Forests dependent on specific water levels will be particularly sensitive to climate change.

3.6.6 Key Vulnerabilities

Within the subtropical domain climate change is likely to increase fire frequency and fire extent in the near future and beyond, although fires will later diminish due to lack of fuel grass. According to the risk analysis by Scholze et al. (2006), using 16 climate models under various scenarios of climatic change, one of the main ecosystem services at risk is loss of water supply, because of more frequent drought and high fire risks in subtropical Africa, Central America, southern Europe and eastern USA. Substantially larger areas will be affected and/or much more negative impacts will result from global warming beyond 3°C (*growth*) compared to a warming of only 2°C (*stable*) relative to preindustrial levels. Climate change has been projected to pose a very severe threat to biodiversity, in particular, since the subtropical domain contains some of the most prominent biodiversity hotspots in Latin America, Australia and South Africa, leading to probable cascading consequences for ecosystem functioning and the production of goods and services.

3.7 Tropical Domain

3.7.1 Types of Tropical Forests

Tropical forests are found between 25°N and 25°S and cover an area of about 17.5 Mkm^2 (Fischlin et al. 2007). They can be minimally grouped into evergreen moist or rainforests, tropical seasonal or drought-deciduous forests (moist savannas) and tropical dry forests (dry savannas). Rainforests are characterized by warm temperatures (annual mean >24°C) and high (≥2.5m/a) and regular precipitation throughout the year. They are found along the equatorial zone between 5°N and 5°S, are evergreen or semi-evergreen and include various geographical landscapes: lowland, mountain and swamp. Precipitation is at least twice the potential evapotranspiration. Tropical seasonal forests are characterized by a ratio of precipitation to potential evapotranspiration between 2 and 1, whereas tropical dry forests are characterized by a ratio <1. Seasonal forests are found in tropical monsoon regions or other seasonal tropical wet-dry climate zones and are moist deciduous, i.e. the trees shed their leaves in the dry season. All tropical forests, as defined here, typically require monthly temperature means to remain above 15.5°C (Prentice et al. 1992). Subtropical forests, typically characterized by dry conditions as found north or near 25°N and south or near 25°S, respectively, are discussed together with tropical dry forests in a separate sub-chapter (this report, sub-chapter 3.6).

3.7.2 Main Services Provided

Productivity: Tropical forests provide a wide range of provisioning services that include not only production of the highly valued tropical timber for domestic and international markets, but also many non-wood products and goods for the local population, including the livelihood of many indigenous peoples (e.g. Gitay et al. 2001, Hassan et al. 2005, Reid et al. 2005).

Regulation: Tropical forests provide major regulating services. Regionally and locally, forests control air humidity, soil moisture and water evaporation, and therefore the local and regional climate, by regulating the hydrology through photosynthesis and the canopy cover. Tropical forests regulate not only the microclimate, as provided by forests in general, but also the continent-wide climate by sustaining higher precipitation levels compared to regions without a forest canopy (e.g. Laurance and Williamson 2001, Semazzi and Song 2001, Betts et al. 2004, Bruijnzeel 2004, Negri et al. 2004, Werth and Avissar 2004, Avissar and Werth 2005, Field et al. 2007a).

Long-term monitoring of plots in mature humid tropical forests concentrated in South America revealed that forests gain biomass by tree growth exceeding losses from tree death by 0.71+/– 0.34 tC/ha (Phillips et al. 1998). Several authors have reported such gains in primary productivity of many tropical forests (e.g. Phillips et al. 2002a, Baker et al. 2004, Laurance et al. 2004, Lewis et al. 2004a, Lewis et al. 2004b, Phillips et al. 2008, Lewis et al. 2009), whereas others have found a deceleration of growth (Feeley et al. 2007) possibly due to changes in the water regime (Malhi and Wright 2004, Boisvenue and Running 2006, Feeley et al. 2007), while others have pointed at the increasing dominance of the tree growth suppressing parasitic lianas (e.g. Phillips et al. 2002b, Wright et al. 2004) or other causes possibly slowing down growth (e.g. Malhi and Phillips 2004, Betts et al. 2008).These findings suggest that undisturbed tropical old-growth forests are currently a significant carbon sink. For the Amazon alone this sink was estimated to be 0.6 PgC/a (Phillips et al. 2008), for Africa 0.34 PgC/a and for all tropical forests 1.3 PgC/a (Lewis et al. 2009).

Biodiversity: Tropical forests, in particular rainforests, are estimated to harbour the highest, biodiversity of all land ecosystems (e.g. Gentry 1992, Leigh et al. 2004) amounting to more than half of terrestrial and about a quarter of global biodiversity (Myers et al. 2000), supporting a vast range of services to people (e.g. Fearnside 1999). 15 out of the worldwide 25 biodiversity hotspots are found in the global tropical domain (e.g. Myers et al. 2000, Webb et al. 2005). The diversity in these forests is not exactly known and can only be estimated approximately (May 1990, Gentry 1992). Nevertheless, many studies provide strong evidence that climate change may lead to major biodiversity losses on all continents (e.g. Bazzaz 1998, Ravindranath and Sukumar 1998, Miles et al. 2004, Possingham and Wilson 2005, Stokstad 2005, Malhi et al. 2008), with consequent effects on other goods and services from these forests (Photo 3.3).

3.7.3 Current Opportunities and Vulnerabilities

Opportunities: Under elevated atmospheric CO_2 concentrations many species show a physiological response, e.g. by changing their photosynthetic rate. In general, increased CO_2 concentration stimulates plant growth and is beneficial to forests and crops in the humid and sub-humid tropics, particularly if nutrient limitations are absent or marginal (Baker et al. 2004, Lewis et al. 2004a, Lewis et al. 2004b, Zhao et al. 2005).

Vulnerabilities: Tropical forests are sensitive to global climate change and may be so severely impacted in structure and function that their services are greatly threatened (e.g. Betts et al. 2008). Bazzaz (1998) argued that tropical forests are sensitive to climate change for the following reasons: Firstly, a small change in climate could affect phenological events (such as flowering and fruiting) – some highly tuned to current climatic conditions, which may escalate to major impacts on the forest's biodiversity. This may even lead to changes in the role of the entire ecosystem in the global carbon cycle. For example, fruit-dependent animals are vulnerable to the consequences of changes in plant phenology (Corlett and Lafrankie 1998). Secondly, co-evolution produced interactions among specific plant and animal species, such as pollination and seed dispersal, with a high degree of specialization and strongly interdependent (Bazzaz 1998). Thirdly, many species in tropical forests have narrow niches because the diversity per unit area is very high (e.g. Erwin 1988, Gentry 1992, Leigh et al. 2004, Wills et al. 2006). Since opportunities for upslope displacement of endemic species with low adaptive capacity are limited in the tropical domain (e.g. Australia – Williams et al. 2003, Africa – McClean et al. 2005, Latin America – Raxworthy et al. 2008), climate change is considered to pose considerable risks for tropical biodiversity (e.g. Miles et al. 2004, Fischlin et al. 2007, sections 4.4.5, 4.4.11). Fourthly, deforestation

Geoffrey Kay

Photo 3.3 Tropical forests, notably rainforests, harbour the majority of terrestrial biodiversity. Although research is less thorough than in other domains and quantitative estimates of diversity are difficult to obtain in the tropical domain, current knowledge robustly shows that tropical forests with their high endemism are of key relevance for the preservation of the Earth's biodiversity. Left: Primary rainforest stream scene within Gunung Mulu National Park, Sarawak, Borneo. Right – top: Flower of the worlds largest flowering plant from genus *Rafflesia*, at Poring Hot Springs, Borneo. Right – bottom: A Tomato Frog (*Dyscophus antongilii*) found at night in a tropical primary rainforest in the Makira Forest, Madagascar. Amphibian species such as the Golden Toad from Costa Rica's Monteverde cloud forests are among the first species possibly having gone extinct due to climate change (cf. review of extinction risks from climate change in Fischlin et al. 2007, section 4.4.11, p. 230, Figure 4.4, Table 4.1).

and other forms of anthropogenic disturbances may have significant ramifications, including impacts on tropical biodiversity (e.g. Pimm and Raven 2000, Pitman et al. 2002, Phillips et al. 2008), a situation in which climatic change is expected mainly to exacerbate the threats to biodiversity (e.g. Fischlin et al. 2007 [section 4.4.11], IPCC 2007b).

Non climatic drivers: Many humid and sub-humid tropical forests are degraded by human activities such as pasture and commercial agriculture expansion, high-intensity logging, including shifting cultivation, fire, mining, and generally an overexploitation of forest resources, e.g. unsustainable logging (Zhao et al. 2005).

In particular, the continued conversion of large areas of humid tropical forests to pasture or other agricultural land uses (Houghton 2007) is understood to be a major driver for ecosystem change and loss of biodiversity in Amazonia (Watson et al. 1997). Substantially large deforestation in Amazonia can reduce evapotranspiration that would lead to less rainfall during dry periods in large forest and rangeland areas, with mountain ecosystems and transitional zones between vegetation types. The superimposition of global warming-driven climate change could make these areas extremely vulnerable to change (Watson et al. 1997).

In tropical Asia climate change will also add to other pressures resulting from rapid urbanization, industrialization and economic development (Hassan et al. 2005). These trends have often led to unsustainable exploitation of natural resources, increased pollution, land degradation and numerous other environmental problems (Watson et al. 1997).

In Africa, tropical forests and rangelands are currently under threat from population pressures and land-use systems (e.g. Achard et al. 2002, Hassan et al. 2005). Apparent effects from these pressures include rapid deterioration in vegetation cover, biodiversity loss, and depletion of water availability through destruction of catchments and aquifers (Watson et al. 1997). Floristic biodiversity hotspots, such as the mountains of Cameroon and the Afromountain habitats that stretch from Ethiopia to the higher latitudes of Africa at altitudes above 2000 m, could be threatened by shifts in rainfall patterns.

Biodiversity on the mountains could be at risk from an increase in temperature, and because migration may be impeded (Zhao et al. 2005).

3.7.4 Projected Future Impacts and Autonomous Adaptation

Projected future responses of tropical forests to environmental change show significant variation, partly due to incomplete data from that region, to differences among the models of ecosystem function derived from the existing databases, and to differences in future climate scenarios generated by the GCMs (Aber et al. 2001, Zhao et al. 2005). Since particularly tropical forests are subject to many other human made pressures, notably land-use change, impacts of climate change need to be discussed together with those other changes. However, such research is challenging and is particularly lacking in the tropics, which impedes assessments of the impacts of climate change for this domain.

Ecosystem shifts: In the long term, significant shifts in the spatial distribution and extent of tropical forests are very likely, not least because of the interaction of climate-change impacts with the many non-climatic environmental changes taking place in the tropics (e.g. Huntingford et al. 2008, Nepstad et al. 2008).

In Thailand, for example, the area of tropical forest has been projected to increase from 45% to 80% of total forest cover, whereas in Sri Lanka, a significant increase in dry forest and a decrease in wet forest could occur due to climate change (Watson et al. 1997).

Major changes are also projected for the tropical rainforest of north Queensland in Australia (Hilbert et al. 2001), in part because of its constrained geography. An increase in global temperature by only 1ºC causes the area of lowland mesophyll vine forest environments to increase, and results in a loss of core environment for endemic vertebrate species in lowland and mid-altitude areas. Depending on the precipitation, the upland complex notophyll vine forest environments respond positively or negatively. Increased precipitation favours the rainforest types, whereas decreased rainfall increases the area suitable for forests dominated by sclerophyllous genera such as *Eucalyptus* and *Allocasuarina*. The habitats for many endemic vertebrates on the highlands are projected to decrease by 50% threatening many endemic species with eventual extinction. Many endemics are especially vulnerable because the capacity to latitudinal dispersal is relatively limited (Williams et al. 2003). A complete loss of the core environment would occur if the temperature increased by ≥5°C (see also Fischlin et al. 2007). Substantial elevation shifts of ecosystems in the mountain and upland areas of tropical Asia are projected for most climate-change scenarios (Watson et al. 1997). Some authors have reported significant effects of climate change on soil erosion from experiments in central Nepal, leading to deposits on agricultural lands, in irrigation canals and streams affecting crop production (Sivakumar et al. 2005).

In the tropics the ability of species to reach new climatically suitable areas will be further constrained by habitat loss and fragmentation and by their ability to migrate to and survive in appropriate surrogate eco-zones – autonomous adaptation processes – which could also be affected by alien invasive species (Thomas et al. 2004, Fischlin et al. 2007, Ward and Masters 2007). Also due to such mechanisms tropical forests in Central America and Amazonia, are at significant risk from climate change (Scholze et al. 2006). A number of climate models projected under scenarios from cluster *stable* (2xCO_2) suggest a reduction of low-level cloud formation in regions such as in Monteverde and elsewhere in Costa Rica. Changes in the dynamic equilibrium of the cloud forests trigger altitudinal shifts in species ranges, subsequent community reshuffling, biodiversity losses and possibly even forest dieback (Foster 2001).

Water: Possible consequences for water balance in combination with higher temperatures and changes in precipitation under *stable* to *growth* scenarios (global warming 2–3°C over pre-industrial levels) showed an increase in runoff in most parts of tropical Africa and north-west South America, and less runoff in west Africa and Central America (Scholze et al. 2006). High risks of reduced runoff resulted from simulations with an increase in global mean temperature >3°C (*growth* or *fast growth*) in Amazonia, Central America and western Africa. The large variations in rainfall, which cause either drought or flooding in South and Central America, are associated with the ENSO (see glossary) phenomenon (Sivakumar et al. 2005). The properties of a large proportion of tropical forests vary with the seasonal availability of soil water. Species lacking morphological or physiological adaptation, such as some evergreen species, may not survive under water-stress conditions, which ultimately could alter species composition (Zhao et al. 2005).

Losses of tropical forest cover due to climate change according to scenario *growth* (e.g. Huntingford et al. 2008) are expected to feedback on hydrology, both regionally and globally (cf. this report, sub-chapter 3.7.2, e.g. Webb et al. 2006, Bala et al. 2007, Cowling et al. 2008, Nepstad et al. 2008).

The impacts of climate variability and change in the arid and semi-arid tropics of Africa can be described as those related to projected temperature increases, probably leading to increased open water and soil/plant evaporation in combination with precipitation decreases. Desertification in Africa is an example of declining mean rainfall during the last half of the 20th century that has caused a 25–30 km south-west shift in Sahel, Sudan and Guinea vegetation zones at an average rate of 500–600 m/a (Zhao et al. 2005). Similarly, soil moisture is likely to decline in Asia, and therefore the least dryland type (dry sub-humid drylands) is expected to become semi-arid and semi-arid land is expected to become arid (Sivakumar et al. 2005).

Fire: Fire risks have generally increased because warmer temperature together with decreased land precipitation or prolonged drought is likely to accumulate fuels from dying vegetation (Nepstad et al. 2004). With a likely increase of droughts due to a prolonged dry season or the ENSO driven phenomenon of inter-annual variability, the incidence of forest fires is also expected to increase (Alencar et al. 2006). In Asia, climate change may influence fires, which in turn could significantly affect the structure, composition and age diversity of forests in that region. In particular *growth* and *fast growth* scenarios project significantly more frequent forest fires in the arid and semi-arid regions of Asia. More frequent wildfires are also likely in South America, including Amazonia, which are particularly pronounced under *growth* scenarios (>3°C, Scholze et al. 2006). Moreover, drier conditions can trigger insect damage or cause large-scale vegetation shifts (Shlisky et al. 2007).

Pests and disease: Primary forests, secondary forests, plantation forests and agroforestry systems of the subtropics and tropics all experience strong effects from plant pests and pathogens (e.g. Goyer 1991, Su-See 1999, Rice and Greenberg 2000, Mitchell 2002, Wingfield and Robison 2004, Bell et al. 2006, Heath et al. 2006, Ofori and Cobbinah 2007, Hall 2008b). Under all clusters of climate-change scenarios, consequential changes in the strength and form of forest pestilence are anticipated, with greater changes in pestilence accompanying greater changes in climate.

Compared to more poleward ecosystems, where temperature changes are expected to be the dominant driver, patterns of pestilence in the tropics and subtropics are likely to be more responsive to changes in moisture availability. Regions that become dryer are likely to experience increases in tree mortality from various insect herbivores and pathogens as the tree species that are presently there become physiologically mismatched with the changing climate (Van Bael et al. 2004, Desprez-Loustau et al. 2006). Regions that become wetter are likely to experience increases in tree mortality from hydrophyllic pathogens (Jönsson 2006).

On the other hand, some areas may become suitable for valuable tree species because of reduced climatic suitability for some pests and diseases. Increased progress in the development of models that predict pest and disease systems across a range of climates could allow the more detailed predictions that are needed for the adaptive responses of humans interacting with these forests (Wharton and Kriticos 2004, Battisti et al. 2006, Avelino et al. 2007, Nahrung et al. 2008). In intensively managed forests (plantations and agroforestry), the near-term future of forest pestilence will also be influenced by the human-aided movement of pests and pathogens (Roux et al. 2006, Andjic et al. 2007), the choice of tree genotypes for planting (Stone 2001, Ramirez et al. 2004, Dhakal et al. 2005), the extent and patterning of low-diversity stands (Folgarait et al. 1995, Schroth et al. 2000, Staver et al. 2001), and changes in the surrounding landscape that influence the natural enemies of plant pests and pathogens (Terborgh et al. 2001, Cunningham et al. 2005, Tylianakis et al. 2007).

3.7.5 Future Opportunities and Services at Risk

Opportunities: Climate change in general will result in adverse impacts on the natural resources including forests even though some areas would benefit, such as increased rainfall in the highlands of east Africa and equatorial central America that would make marginal lands more productive than they are now (Watson et al. 1997). Even if greenhouse gas emissions were brought to a halt (*unavoidable*), further warming would still occur (IPCC 2007c, Figure 3.1). Even with such a modest rate of warming (0.6°C/century) and assuming otherwise minimal anthropogenic disturbances, some ecosystems are still expected to be affected by changes in their species compositions. However, mitigation of climate change could avoid or minimize many further adverse impacts projected to occur later. For example, the loss of tropical forests and grasslands could be avoided, although in the long run forest may switch from a carbon sink to a net carbon source, perhaps only as late as 2170 (Arnell et al. 2002). The areas with high biodiversity extinction risk are also reduced considerably with climate-change mitigation (Thomas et al. 2004, Fischlin et al. 2007, IPCC 2007b, IPCC 2007d).

Services at risk: Natural disturbance regimes such as fire, insects and disease may potentially affect forests, including their goods and services. This is now more widely recognized in the forest management portfolios of the majority of countries having tropical forests within their territory. However, these are not enough to stop changes in the remaining tropical forests. What is called for are effective climate mitigation to protect existing tropical forests from the negative impacts of climate change and a different form of development in tropical countries from that of now industrialized nations (Gullison et al. 2007). Finally, carefully regulated markets appear to be required to halt or at least to slow down widespread impoverishment and/or losses in the remaining tropical forests (Lewis et al. 2004a, Hall 2008a).

The climate in tropical Asia is characterized by distinct seasonal patterns associated with the two monsoon seasons and the occurrences of tropical cyclones in three cyclogenesis core areas (Bay of Bengal, north Pacific and South China Sea). The climate records show that the ENSO phenomenon has been more frequent and stronger since the 1970s (e.g. Trenberth and Hoar 1996, Trenberth et al. 2007) and has escalated the risk of drought and fire, adding to other pressures such as rapid urbanization, industrialization, unsustainable exploitation of natural resources, increased pollution, land degradation, and other environmental problems (Watson et al. 1997).

Seasonal and inter-annual climate variability contributes particularly to the vulnerability of many regions, e.g. in South America and Australia. Through disturbances such as drought and fire the ENSO phenomenon adversely impacts socio-economies, if those depend heavily on the production of the region's natural ecosystems. If coastal, such regions may also be particularly at risk from other extreme events such as future tropical cyclones. With warming tropical sea-surface temperatures, hurricanes are likely to become more intense, to have stronger peak winds, and to bring heavier precipitation (IPCC 2007d). Although some climate models project globally decreasing frequencies for tropical cyclones, significant uncertainties remain (IPCC 2007d).

3.7.6 Key Vulnerabilities

Carbon storage: Effects from elevated atmospheric CO_2 concentrations on sequestration of carbon in tropical forests are still debated (e.g. Morgan et al. 2001) and the evidence is not unequivocal (Fischlin et al. 2007, section 4.4.1). Phillips et al. (1998) showed that neotropical forests have acted as carbon sinks for the last three decades. Under the simplest scenario of a steady rise in forest productivity over time, it is expected that relatively slow-growing, but otherwise little changing, tropical forests would still act as carbon sinks, perhaps for a century and beyond. The actual magnitude and spatial distribution of this C-sequestration service are influenced by changes in the vegetation structure through changing climate and water availability (Cramer et al. 2001). However, the contribution of tropical forests in slowing down climate change by sequestering carbon has also been projected to diminish in coupled vegetation-atmosphere models, which explicitly consider feedback mechanisms (e.g. Cox et al. 2000, Cox et al. 2004, Cox et al. 2006, Friedlingstein et al. 2006). Simulations using a scenario from cluster *growth* (IPCC IS92a) show a terrestrial carbon sink during the 1990s of 1.4–3.8 PgC/a, but ~2090 the sink is reduced to 0.3–6.6 PgC/a (Cramer et al. 2004). Another land carbon-sink simulation found for the late 20th and throughout the 21st century a persisting sink with a strength of ~1 PgC/a (<2°C, scenario cluster *growth*). A global warming of 2–3°C, however, showed an increasing sink only up to the middle of the century and thereafter it declined (Scholze et al. 2006). Such sink saturation effects were found to occur possibly as early as in the first half of this century (Fischlin et al. 2007, Figure 4.2, p. 222). A global warming of >3°C showed that the sink increases, but less strongly up to the middle of the century, then declined and turned in some cases into a net carbon source towards the end of this century (Scholze et al. 2006). Assuming continuation of current trends of emissions and land-use change, IPCC reported recently that it is very likely that land ecosystems turn into a net source before the end of this century, thus, significantly accelerating climate change (Fischlin et al. 2007, p. 213, IPCC 2007b, p. 11).

Biodiversity: Climate change could be the biggest cause of increased extinction rates in many regions, especially in the tropics (Thomas et al. 2004, Fischlin et al. 2007), and land-use change, such as deforestation, is also an important and synergistic driver (cf. Sala et al. 2000, for a recent, comprehensive review see Fischlin et al. 2007). Deforestation and degradation through infrastructure development, plus non-sustainable practices, result in fragmented forests and biomass losses at large spatial scales, which could be greater in CO_2-induced climate change (Zhao et al. 2005). The results are again impoverished forests with reduced productivity.

3.8 Conclusions

At a worldwide scale, global change pressures (climate change, land-use practices and changes in atmospheric chemistry) are increasingly affecting the supply of goods and services from forests (Easterling and Apps 2005). Moderate climate change alone (*unavoidable*, *stable*) would already put some sensitive ecosystems within the tropical domain at a considerable risk, especially those transitional between two different vegetation classes or eco-zones. Climate change threatens biodiversity, including some of the most valuable biodiversity hotspots of Earth, risking not only major changes in species compositions, but also highly significant and irreversible biodiversity losses that will result in the loss of ecosystem goods and services with severe consequences for forest communities. Considering also anthropogenic disturbances such as forest fragmentation and poor capacity for fire management, many forest species are expected to have difficulty in moving to climatically suitable areas to survive (i.e. to adapt to climate change). Under scenarios of *growth* ('business-as-usual') or *fast growth*, the resulting rapid global change will continue to impact forests, with important consequences for the ecosystem structure, its biodiversity and its many provisioning, regulating and socio-economic services; these include hydrological regulation, carbon sequestration, fires, pests, pathogens and forest health in general as well as ecotourism and the subsistence livelihoods of indigenous peoples.

Forests harbour a large fraction of the Earth biodiversity, perhaps as much as three quarters of the terrestrial biodiversity, with the tropical domain containing very likely already one quarter. Many studies show significant biodiversity losses at the ecosystem, species and genetic levels. Species extinction rates are driven by the magnitude or intensity of climate change, since they affect species distribution and composition (Thomas et al. 2004, Fischlin et al. 2007). One study estimated global extinction risks, ranging from average extinction rates of ~18% (*unavoidable*, lower end *stable*), over ~24% (*stable*), to ~35% (growth Thomas et al. 2004). The comprehensive meta-analysis by IPCC (Fischlin et al. 2007) estimated that, on average, roughly 20–30% of vascular plants and higher animals are at an increasing risk of extinction as temperatures increase by 2–3°C above pre-industrial levels (IPCC 2007b, p. 11). Although current knowledge does not permit predictions of precise tipping points, where some degree of biodiversity loss leads to substantial changes in structure and functioning of ecosystems, it is very likely that the projected losses in biodiversity are highly significant and will result in consequential changes in the ecosystem services currently provided.

Based on the presented CCIAV assessment at the global scale as well as for each of the four domains – boreal, temperate, subtropical and tropical – the following key vulnerabilities were identified:

- Globally, forest ecosystems are sufficiently resilient and can adapt to impacts of limited climate change according to scenarios from cluster *stable*, particularly in currently temperature limited or humid climates, by maintaining similar or increased levels of productivity. However, in drier medium wet, semi-arid to arid climates, forest productivity is projected to decline. Regardless of changes in productivity, species compositions are projected to be significantly altered, e.g. from boreal to mixed-deciduous, from boreal to grassland, from mixed-deciduous to deciduous, or deciduous to savanna.
- Globally, forest ecosystems have difficulty adapting to impacts from climate change according to scenarios from cluster *growth* or *fast growth*, in particular in submesic, semi-arid to arid climates, where productivity may decline to an extent that no longer supports forests or even trees. In such cases forest systems will become grasslands, savannas, or even deserts. In humid climates, forests are projected to continue to grow or expand. The overall balance is positive for scenarios at the lower bounds, but tends towards a negative balance for scenarios at the upper bounds of cluster *growth*. Several models project a significant risk (>40%) of losing entirely current carbon-regulating services, as land ecosystems turn globally into a net source of carbon beyond a global warming of 3°C or more relative to pre-industrial levels. Such effects are projected to be even more pronounced in the next century, as development pathways from the upper end of clusters *growth* and *fast growth* are still far from having reached a new climate equilibrium by ~2100.
- Boreal forests are projected to increase their productivity, in the northern taiga even under scenarios from cluster *growth*. However, at the same time those forests are projected to be impacted by an increased prevalence of fires and insect pests, and the overall balance of losses in the southern areas vs. the smaller gains in the northern parts is likely to be negative, particularly within this century. Moreover, the carbon emissions from thawing and burning peatlands in northern boreal taiga systems are projected to further enhance climate change.
- Temperate forests are projected to increase their productivity in northern poleward areas for climate-change scenarios from cluster *stable*, whereas the equatorial areas show declining productivity under the same scenarios. The overall balance is more likely than not positive. However, for scenarios from cluster *growth* and *fast growth*, the overall balance is highly uncertain with considerable simultaneous risks from drought, fire, pollution, habitat fragmentation and possibly more opportunities for invasive alien species arising towards the end of this century and beyond. They are projected to tip the balance further towards the dominance of negative effects.
- Productivity in most subtropical forests is projected to decrease under a wide range of climate-change scenarios. Fire frequencies are expected to increase, yet may reach saturation or may even diminish when conditions become so dry that decreased production leads to less fuel accumulation. In this domain, several biodiversity hotspots are threatened by a wide range of climate-change scenarios, and the well-being of many people depending on current productivity levels is increasingly at risk.

- Tropical forests are projected to increase their productivity wherever sufficient water is available. However, in seasonal dry or otherwise drier climates, tropical forests are projected to decline. Not only significant provisioning but also globally important regulating services are at risk. Climate feedbacks from local climate to the global carbon cycle may have major implications for the global climate and may contribute towards an acceleration of climate change. Moreover, the tropical domain harbours major amounts of the Earth's biodiversity, and substantial biodiversity losses are to be expected.
- High altitude systems that maintain forests are expected to lose biodiversity as the capacity for these species to move to suitable climate domains is extremely limited. According to current understanding, tropical mountain forest species systems are most at risk in this regard.

Since our analysis showed that many forests are highly vulnerable to unmitigated climate change (scenarios from cluster *growth*), merely strengthening adaptation will be insufficient to maintain, let alone enhance, the multitude of ecosystem services forests currently provide. Moreover, since forests may release large quantities of carbon if impacted by climate-change stressors or otherwise degraded, they may exacerbate climate change unless such feedbacks are slowed down. Thus, what is called for in addition to adaptation is climate mitigation and lessening non-climatic pressures, notably a large reduction in fossil-fuel emissions as well as stopping deforestation, that effectively curb climate change and enable forests to maintain their adaptive capacity.

References

Aber, J., McDowell, W., Nadelhoffer, K., Magill, A., Berntson, G., Kamakea, M., McNulty, S., Currie, W., Rustad, L. & Fernandez, I. 1998. Nitrogen saturation in temperate forest ecosystems – Hypotheses revisited. Bioscience 48(11): 921–934.

Aber, J., Neilson, R.P., McNulty, S., Lenihan, J.M., Bachelet, D. & Drapek, R.J. 2001. Forest processes and global environmental change: Predicting the effects of individual and multiple stressors. Bioscience 51(9): 735–751.

Aber, J.D. 1992. Nitrogen cycling and nitrogen saturation in temperate forest ecosystems. Trends Ecol. Evol. 7(7): 220–224.

Achard, F., Eva, H.D., Stibig, H.J., Mayaux, P., Gallego, J., Richards, T. & Malingreau, J.P. 2002. Determination of deforestation rates of the world's humid tropical forests. Science 297(9.Aug.): 999–1002.

Albani, M., Medvigy, D., Hurtt, G.C. & Moorcroft, P.R. 2006. The contributions of land-use change, CO2 fertilization, and climate variability to the Eastern US carbon sink. Glob. Chang. Biol. 12(12): 2370–2390.

Alcamo, J., Moreno, J.M., Nováky, B., Bindi, M., Corobov, R., Devoy, R.J.N., Giannakopoulos, C., Martin, E., Olesen, J.E. & Shvidenko, A. 2007. Europe. In: Parry, M.L., Canziani, O.F., Palutikof, J.P., van der Linden, P.J. & Hanson, C.E. (eds.). Climate change 2007: Impacts, adaptation and vulnerability. Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel of Climate Change (IPCC). Cambridge University Press, Cambridge, UK. p. 541–580.

Alencar, A., Nepstad, D. & Diaz, M.D.V. 2006. Forest understory fire in the Brazilian Amazon in ENSO and non-ENSO years: Area burned and committed carbon emissions. Earth Interactions 10(6): 1–17.

Allen Consulting Group 2005. Climate change risk and vulnerability: promoting an efficient adaptation response in Australia. Australian Greenhouse Office, in the Department of the Environment and Heritage, Canberra, Australia. 159 p.

Alo, C.A. & Wang, G.L. 2008. Potential future changes of the terrestrial ecosystem based on climate projections by eight general circulation models. J. Geophys. Res. G 113(G1).

Andjic, V., Barber, P.A., Carnegie, A.J., Pegg, G.S., Hardy, G., Wingfield, M.J. & Burgess, T.I. 2007. Kirramyces viscidus sp nov., a new eucalypt pathogen from tropical Australia closely related to the serious leaf pathogen, Kirramyces destructans. 36(5): 478–487.

Andreani-Aksoyoglu, S., Keller, J., Ordonez, C., Tinguely, M., Schultz, M. & Prevot, A.S.H. 2008. Influence of various emission scenarios on ozone in Europe. Ecol. Model. 217(3–4): 209–218.

Anisimov, O.A., Vaughan, D.G., Callaghan, T.V., Furgal, C., Marchant, H., Prowse, T.D., Vilhjálmsson, H. & Walsh, J.E. 2007. Polar regions (Arctic and Antarctic). In: Parry, M.L., Canziani, O.F., Palutikof, J.P., van der Linden, P.J. & Hanson, C.E. (eds.). Climate change 2007: Impacts, adaptation and vulnerability. Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel of Climate Change (IPCC). Cambridge University Press, Cambridge, UK. p. 653–685.

Anulewicz, A., McCullough, D., Cappaert, D. & Poland, T. 2008. Host range of the emerald ash borer (agrilus planipennis Fairmaire) (Coleoptera: Buprestidae) in North America: results of multiple–choice field experiments. Environ. Entomol. 31(1): 230–241.

Archer, S. 1991. Development and stability of grass/woody mosaics in a subtropical savanna parkland, Texas, USA. J. Biogeogr.(XI, 221): 453–462.

Arnell, N.W., Cannell, M.G.R., Hulme, M., Kovats, R.S., Mitchell, J.F.B., Nicholls, R.J., Parry, M.L., Livermore, M.T.J. & White, A. 2002. The consequences of CO_2 stabilisation for the impacts of climate change. Clim. Change 53(4): 413–446.

Avelino, J., Cabut, S., Barboza, B., Barquero, M., Alfaro, R., Esquivel, C., Durand, J. & Cilas, C. 2007. Topography and crop management are key factors for the development of American leaf spot epidemics on coffee in Costa Rica. Phytopathology 97(12): 1532–1542.

Avissar, R. & Werth, D. 2005. Global hydroclimatological teleconnections resulting from tropical deforestation. J. Hydrometeorol. 6(2): 134–145.

Bachelet, D., Lenihan, J.M., Daly, C., Neilson, R.P., Ojima, D.S. & Parton, W.J. 2001. MC1: a dynamic vegetation model for estimating the distribution of vegetation and associated carbon, nutrients, and water—technical documentation. Version 1.0. Gen. Tech. Rep. PNW-GTR-508, S. Department of Agriculture, Forest Service, Pacific Northwest Research Station, Portland, OR, USA. 95 p.

Bakaeva, Z.M. & Zamolodchikov, D.G. 2008. Assessment of carbon budget in forests according to actualized data of forest fund inventory (Republic of Ingushetia). Lesovedenie (For. Sci.) 3(Mai–Juni): 66–70 (in Russian, w. English abstract).

Baker, T.R., Phillips, O.L., Malhi, Y., Almeida, S., Arroyo, L., Di Fiore, A., Erwin, T., Higuchi, N., Killeen, T.J., Laurance, S.G., Laurance, W.F., Lewis, S.L., Monteagudo, A., Neill, D.A., Vargas, P.N., Pitman, N.C.A., Silva, J.N.M. & Martinez, R.V. 2004. Increasing biomass in Amazonian forest plots. Phil. Trans. R. Soc. Lond. B 359(1443): 353–365.

Bala, G., Caldeira, K., Wickett, M., Phillips, T.J., Lobell, D.B., Delire, C. & Mirin, A. 2007. Combined climate and carbon-cycle effects of large-scale deforestation. PNAS 104(16): 6550–6555.

Balesdent, J., Girardin, C. & Mariotti, A. 1993. Site-related delta-C-13 of tree leaves and soil organic-matter in a temperate forest. Ecology 74(6): 1713–1721.

Balshi, M., McGuire, A., Duffy, P., Flannigan, M., Walsh, J. & Melillo, J. 2008. Assessing the response of area burned to changing climate in western boreal North America using a Multivariate Adaptive Regression Splines (MARS) approach. Global Change Biol. 14: 1–23.

Barbosa, P., Libertà, G. & Schmuck, G. 2003. The European Forest Fires Information System (EFFIS) results on the 2003 fire season in Portugal by the 20th of August. Report 20030820, European Commission, Directorate General Joint Research Centre, Institute for Environment and Sustainability, Land Management Unit, Ispra (VA), Italy. 10 p.

Bari, M.A., Berti, M.L., Charles, S.P., Hauck, E.J. & Pearcey, M. 2005. Modelling of streamflow reduction due to climate change in Western Australia – A case study. In: Zerger, A. & Argent, R. (eds.), International Congress on Modelling and Simulation, MODSIM 2005, Modelling and Simulation Society of Australia and New Zealand, Canberra, Australia. December 2005. p. 482–488.

Battisti, A. 2004. Forests and climate change – lessons from insects. Forest@ 1(1): 17–24.

Battisti, A., Stastny, M., Buffo, E. & Larsson, S. 2006. A rapid altitudinal range expansion in the pine processionary moth produced by the 2003 climatic anomaly. Glob. Chang. Biol. 12(4): 662–671.

Battisti, A., Stastny, M., Netherer, S., Robinet, C., Schopf, A., Roques, A. & Larsson, S. 2005. Expansion of geographic range in the pine processionary moth caused by increased winter temperatures. Ecol. Appl. 15(6): 2084–2096.

Bazzaz, F.A. 1998. Tropical forests in a future climate: Changes in biological diversity and impact on the global carbon cycle. Clim. Change 39(2–3): 317–336.

Beare, S. & Heaney, A. 2002. Climate change and water resources in the Murray Darling Basin, Australia – Impacts and possible adaptation. Conference paper 02.11, ABARE, Canberra, A.C.T. 1–33 p.

Bejer, A. 1988. The nun moth in European spruce forests. In: Berryman, A. (ed.) Dynamics of forest insect populations: patterns, causes, implications. Plenum Publishing Corporation, New York, USA. p. 211–231.

Bell, T., Freckleton, R.P. & Lewis, O.T. 2006. Plant pathogens drive density-dependent seedling mortality in a tropical tree. Ecol. Lett. 9(5): 569–574.

Bergeron, Y. 1991. The Influence of Island and Mainland Lakeshore Landscapes on Boreal Forest-Fire Regimes. Ecology 72(6): 1980–1992.

Bergeron, Y. & Archambault, S. 1993. Decreasing frequency of forest fires in the southern boreal zone of Québec and its relation to global warming since the end of the 'Little Ice Age'. Holocene 3(3): 255–259.

Bergot, M., Cloppet, E., Pérarnaud, V., Déqué, M., Marçais, B. & Desprez-Loustau, M.L. 2004. Simulation of potential range expansion of oak disease caused by Phytophthora cinnamomi under climate change. Global Change Biol. 10(9): 1539–1552.

Betts, R.A., Cox, P.M., Collins, M., Harris, P.P., Huntingford, C. & Jones, C.D. 2004. The role of ecosystem–atmosphere interactions in simulated Amazonian precipitation decrease and forest dieback under global climate warming. Theor. Appl. Climatol. 78(1–3): 157–175.

Betts, R.A., Malhi, Y. & Roberts, J.T. 2008. The future of the Amazon: new perspectives from climate, ecosystem and social sciences. Phil. Trans. R. Soc. Lond. B 363(1498): 1729–1735.

Beuker, E. 1994. Long-term effects of temperature on the wood production of Pinus sylvestris L. and Picea abies (L.) Karst. in old provenance experiments. Scand. J. For. Res. 9: 34–45.

Beuker, E., Kolström, M. & Kellomäki, S. 1996. Changes in wood production of Picea abies and Pinus sylvestris under a warmer climate: comparison of field measurements and results of a mathematical model. Silv. Fen. 30(2–3): 239–246.

Birdsey, R., Pregitzer, K. & Lucier, A. 2006. Forest carbon management in the United States: 1600–2100. J. Environ. Qual. 35(4): 1461–1469.

Bohlen, P.J., Scheu, S., Hale, C.M., McLean, M.A., Migge, S., Groffman, P.M. & Parkinson, D. 2004. Non-native invasive earthworms as agents of change in northern temperate forests. Frontiers in Ecol. & Environ. 2(8): 427–435.

Boisvenue, C. & Running, S.W. 2006. Impacts of climate change on natural forest productivity – evidence since the middle of the 20th century. Global Change Biol. 12(5): 862–882.

Bonan, G.B. 2002. Ecological climatology: concepts and applications. Cambridge University Press, Cambridge, UK. 678 p.

Bond, W.J. & Midgley, G.F. 2000. A proposed CO_2-controlled mechanism of woody plant invasion in grasslands and savannas. Global Change Biol. 6(8): 865–869.

Bond, W.J., Midgley, G.F. & Woodward, F.I. 2003a. The importance of low atmospheric CO_2 and fire in promoting the spread of grasslands and savannas. Global Change Biol. 9(7): 973–982.

Bond, W.J., Midgley, G.F. & Woodward, F.I. 2003b. What controls South African vegetation – climate or fire? South Afr. J. Bot. 69(1): 79–91.

Bond, W.J., Woodward, F.I. & Midgley, G.F. 2005. The global distribution of ecosystems in a world without fire. New Phytol. 165(2): 525–537.

Booth, T.H. 2007. Climate change and Pinus radiata plantations in Australia. Aust. For. Grower: 44–48.

Booth, T.H., Kirschbaum, M.U. & Battaglia, M. 2008. Forestry. In: Csiro (ed.) An overview of climate change adaptation in the Australian agricultural sector – impacts, options and priorities. CSIRO, Canberra, Austrailia. p. 204–225.

Booth, T.H., Nix, H.A., Hutchinson, M.F. & Jovanovic, T. 1988. Niche analysis and tree species introduction. For. Ecol. Manage. 23: 47–59.

Bosworth, D., Birdsey, R., Joyce, L. & Millar, C. 2008. Climate change and the nation's forests: Challenges and opportunites. J. For. 106(4): 214–221.

Breda, N., Huc, R., Granier, A. & Dreyer, E. 2006. Temperate forest trees and stands under severe drought: a review of ecophysiological responses, adaptation processes and long-term consequences. Ann. For. Sci. 63(6): 625–644.

Breshears, D.D. & Allen, C.D. 2002. The importance of rapid, disturbance–induced losses in carbon management and sequestration. Global Ecol. Biogeogr. 11(1): 1–5.

Breshears, D.D., Cobb, N.S., Rich, P.M., Price, K.P., Allen, C.D., Balice, R.G., Romme, W.H., Kastens, J.H., Floyd, M.L., Belnap, J., Anderson, J.J., Myers, O.B. & Meyer, C.W. 2005. Regional vegetation die-off in response to global-change-type drought. PNAS 102(42): 15144–15148.

Breshears, D.D., Huxman, T.E., Adams, H.D., Zou, C.B. & Davison, J.E. 2008. Vegetation synchronously leans upslope as climate warms. PNAS 105(33): 11591–11592.

Bruijnzeel, L.A. 2004. Hydrological functions of tropical forests: not seeing the soil for the trees? Agric. Ecosystems & Environ. 104(1): 185–228.

Butin, E., Porter, A.H. & Elkinton, J. 2005. Adaptation during biological invasions and the case of Adelges tsugae. Evol. Ecol. Res. 7(6): 887–900.

CABI, C.f.A.B.I. 2005. Forestry compendium. CABI, Centre for Agricultural Bioscience International. Wallingford, UK.

Callaghan, T.V., Bjorn, L.O., Chernov, Y., Chapin, T., Christensen, T.R., Huntley, B., Ims, R.A., Johansson, M., Jolly, D., Jonasson, S., Matveyeva, N., Panikov, N., Oechel, W., Shaver, G., Elster, J., Henttonen, H., Laine, K., Taulavuori, K., Taulavuori, E. & Zockler, C. 2004. Biodiversity, distributions and adaptations of arctic species in the context of environmental change. Ambio 33(7): 404–417.

Canadell, J.G., Pataki, D.E., Gifford, R., Houghton, R.A., Luo, Y., Raupach, M.R., Smith, P. & Steffen, W. 2007. Saturation of the terrestrial carbon sink. In: Canadell, J.G., Pataki, D.E. & Pitelka, L.F. (eds.). Terrestrial ecosystems in a changing world. Springer-Verlag, Berlin. p. 59–78.

Carcaillet, C., Bergeron, Y., Richard, P.J.H., Frechette, B., Gauthier, S. & Prairie, Y.T. 2001. Change of fire frequency in the eastern Canadian boreal forests during the Holocene: does vegetation composition or climate trigger the fire regime? J. Ecol. 89(6): 930–946.

Carrer, M. & Urbinati, C. 2006. Long-term change in the sensitivity of tree-ring growth to climate forcing in Larix decidua. New Phytol. 170(4): 861–871.

Carroll, A.L., Taylor, S.W., Régnière, J. & Safranyik, L. 2004. Effects of climate change on range expansion by the mountain pine beetle in British Columbia. In: Shore, T.L., Brooks, J.E. & Stone, J.E. (eds.). Mountain Pine Beetle Symposium: Challenges and Solutions. Natural Resources Canada, Canadian Forest Service, Pacific Forestry Centre, Victoria, British Columbia, Information. p. 223–232.

Carter, T.R., Jones, R.N., Lu, X., Bhadwal, S., Conde, C., Mearns, L.O., O'Neill, B.C., Rounsevell, M.D.A. & Zurek, M.B. 2007. New assessment methods and the characterisation of future conditions. In: Parry, M.L., Canziani, O.F., Palutikof, J.P., van der Linden, P.J. & Hanson, C.E. (eds.). Climate change 2007: Impacts, adaptation and vulnerability. Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel of Climate Change (IPCC). Cambridge University Press, Cambridge, UK. p. 133–171.

Caspersen, J.P., Pacala, S.W., Jenkins, J.C., Hurtt, G.C., Moorcroft, P.R. & Birdsey, R.A. 2000. Contributions of land-use history to carbon accumulation in U.S. forests. Science 290(10.Nov.): 1148–1151.

Chakraborty, S., Luck, J., Hollaway, G., Freeman, A., Norton, R., Garrett, K.A., Percy, K., Hopkins, A., Davis, C. & Karnosky, D.F. 2008. Impacts of global change on diseases of agricultural crops and forest trees. 3(054): 1–15.

Chiew, F. & McMahon, T. 2002. Modelling the impacts of climate change on Australian streamflow. Hydrol. Processes 16(6): 1235–1245.

Chmielewski, F.M. & Rötzer, T. 2001. Response of tree phenology to climate change across Europe. Agric. For. Meteorol. 108(2): 101–112.

Christensen, J.H., Hewitson, B., Busuioc, A., Chen, A., Gao, X., Held, I., Jones, R., Kolli, R.K., Kwon, W.T., Laprise, R., Magaòa Rueda, V., Mearns, L., Menéndez, C.G., Räisänen, J., Rinke, A., Sarr, A. & Whetton, P. 2007. Regional climate projections. In: Solomon, S., Qin, D., Manning, M., Chen, Z., Marquis, M., Averyt, K.B., Tignor, M. & Miller, H.L. (eds.). Climate change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change (IPCC). Cambridge University Press, Cambridge, UK and New York, NY, USA. p. 847–940.

Christiansen, E. & Bakke, A. 1988. The spruce bark beetle of Eurasia. In: Berryman, A.A. (ed.) Dynamics of forest insect populations: patterns, causes, implications. Plenum Publishing Corporation, New York. p. 479–503.

Ciais, P., Reichstein, M., Viovy, N., Granier, A., Ogee, J., Allard, V., Aubinet, M., Buchmann, N., Bernhofer, C., Carrara, A., Chevallier, F., De Noblet, N., Friend, A.D., Friedlingstein, P., Grunwald, T., Heinesch, B., Keronen, P., Knohl, A., Krinner, G., Loustau, D., Manca, G., Matteucci, G., Miglietta, F., Ourcival, J.M., Papale, D., Pilegaard, K., Rambal, S., Seufert, G., Soussana, J.F., Sanz, M.J., Schulze, E.D., Vesala, T. & Valentini, R. 2005. Europe-wide reduction in primary productivity caused by the heat and drought in 2003. Nature 437(7058): 529–533.

Conard, S.G. & Ivanova, G.A. 1997. Wildfire in Russian boreal forests – Potential impacts of fire regime characteristics on emissions and global carbon balance estimates. Environ. Pollut. 98(3): 305–313.

Cook, E.R., Seager, R., Cane, M.A. & Stahle, D.W. 2007. North American drought: reconstructions, causes, and consequences. Earth Sci. Reviews 81(1–2): 93–134.

Corlett, R.T. & Lafrankie, J.V. 1998. Potential impacts of climate change on tropical Asian forests through an influence on phenology. Clim. Change 39(2–3): 439–453.

Cowling, S.A., Shin, Y., Pinto, E. & Jones, C.D. 2008. Water recycling by Amazonian vegetation: coupled versus uncoupled vegetation-climate interactions. Phil. Trans. R. Soc. Lond. B 363(1498): 1865–1871.

Cox, P.M., Betts, R.A., Collins, M., Harris, P.P., Huntingford, C. & Jones, C.D. 2004. Amazonian forest dieback under climate-carbon cycle projections for the 21st century. Theor. Appl. Climatol. 78(1–3): 137–156.

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

Cox, P.M., Huntingford, C. & Jones, C.D. 2006. Conditions for sink-to-source transitions and runaway feedbacks from the land carbon cycle. In: Schellnhuber, H.J.p.C.W., Nakicenovic, N., Wigley, T. & Yohe, G. (eds.). Avoiding dangerous climate change. Cambridge University Press, Cambridge, UK. p. 155–161.

Cramer, W., Bondeau, A., Schaphoff, S., Lucht, W., Smith, B. & Sitch, S. 2004. Tropical forests and the global carbon cycle: impacts of atmospheric carbon dioxide, climate change and rate of deforestation. Phil. Trans. R. Soc. Lond. B 359(1443): 331–343.

Cramer, W., Bondeau, A., Woodward, F.I., Prentice, I.C., Betts, R.A., Brovkin, V., Cox, P.M., Fisher, V., Foley, J.A., Friend, A.D., Kucharik, C., Lomas, M.R., Ramankutty, N., Sitch, S., Smith, B., White, A. & Young-Molling, C. 2001. Global response of terrestrial ecosystem structure and function to CO$_2$ and climate change: results from six dynamic global vegetation models. Global Change Biol. 7(4): 357–373.

Cunningham, S.A., Floyd, R.B. & Weir, T.A. 2005. Do Eucalyptus plantations host an insect community similar to remnant Eucalyptus forest? Austral Ecol. 30(1): 103–117.

Daigneault, A.J., Sohngen, B. & Sedjo, R. 2008. Exchange rates and the competitiveness of the United States timber sector in a global economy. For. Pol. & Econ. 10(3): 108–116.

Daly, C., Bachelet, D., Lenihan, J.M., Neilson, R.P., Parton, W. & Ojima, D. 2000. Dynamic simulation of tree-grass interactions for global change studies. Ecol. Appl. 10(2): 449–469.

Davis, M.B. 1989. Lags in vegetation response to greenhouse warming. Clim. Change 15: 75–82.

Davis, M.B., Woods, K.D., Webb, S.L. & Futyma, R.P. 1986. Dispersal versus climate – expansion of fagus and tsuga into the upper Great-Lakes region. Vegetatio 67(2): 93–103.

De Bono, A., Peduzzi, P., Giuliani, G. & Kluser, S. 2004. Impacts of summer 2003 heat wave in Europe. Early Warning on Emerging Environmental Threats 2, UNEP: United Nations Environment Programme, Nairobi, Kenya. 4 p.

Denman, K.L., Brasseur, G., Chidthaisong, A., Ciais, P., Cox, P.M., Dickinson, R.E., Hauglustaine, D., Heinze, C., Holland, E., Jacob, D., Lohmann, U., Ramachandran, S., da Silva Dias, P.L., Wofsy, S.C. & Zhang, X. 2007. Couplings between changes in the climate system and biogeochemistry. In: Solomon, S., Qin, D., Manning, M., Chen, Z., Marquis, M., Averyt, K.B., Tignor, M. & Miller, H.L. (eds.). Climate change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change (IPCC). Cambridge University Press, Cambridge, UK and New York, NY, USA. p. 499–587.

Desprez-Loustau, M.L., Marçais, B., Nageleisen, L.M., Piou, D. & Vannini, A. 2006. Interactive effects of drought and pathogens in forest trees. Ann. For. Sci. 63(2006): 597–612.

Desprez-Loustau, M.L., Robin, C., Reynaud, G., Deque, M., Badeau, V., Piou, D., Husson, C. & Marcais, B. 2007. Simulating the effects of a climate-change scenario on the geographical range and activity of forest-pathogenic fungi. Can. J. Plant Pathol. 29(2): 101–120.

Dhakal, L.P., Jha, P.K. & Kjaer, E.D. 2005. Mortality in Dalbergia sissoo (roxb.) following heavy infection by Aristobia horridula (hope) bettles – Will genetic variation in susceptibility play a role in combating declining health? For. Ecol. Manage. 218(1–3): 270–276.

Dobbertin, M., Wermelinger, B., Bigler, C., Burgi, M., Carron, M., Forster, B., Gimmi, U. & Rigling, A. 2007. Linking increasing drought stress to Scots pine mortality and bark beetle infestations. TheScientificWorldJournal 7(S1): 231–239.

Duffy, P.A., Walsh, J.E., Graham, J.M., Mann, D.H. & Rupp, T.S. 2005. Impacts of large-scale atmospheric-ocean variability on Alaskan fire season severity. Ecol. Appl. 15(4): 1317–1330.

Dwinell, L.D. 1997. The pinewood nematode: Regulation and mitigation. Annu. Rev. Phytopathol. 35: 153–166.

Easterling, W. & Apps, M. 2005. Assessing the consequences of climate change for food and forest resources: A view from the IPCC. Clim. Change 70(1–2): 165–189.

Easterling, W.E., Aggarwal, P.K., Batima, P., Brander, K.M., Erda, L., Howden, S.M., Kirilenko, A., Morton, J., Soussana, J.F., Schmidhuber, J. & Tubiello, F.N. 2007. Food, fibre and forest products. In: Parry, M.L., Canziani, O.F., Palutikof, J.P., van der Linden, P.J. & Hanson, C.E. (eds.). Climate change 2007: Impacts, adaptation and vulnerability. Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel of Climate Change (IPCC). Cambridge University Press, Cambridge, UK. p. 273–313.

Erwin, T.L. 1988. The tropical forest canopy – The heart of biotic diversity. In: Wilson, E.O. & Peter, F.M. (eds.). Biodiversity. National Academy Press, Washington, DC. p. 123–129.

Esseen, P., Ehnström, L., Eriksson, L. & Sjöberg, K. 1993. Boreal forest – the focal habitats of Fennoscandia. In: Hansson, L. (ed.) The Ecological principles of nature conservation : applications in temperate and borel environments. Elsevier Applied Science, London; New York. p. 252–325.

Evans, H., Straw, N. & Watt, A. 2002. Climate change implications for insect pests. In: Broadmeadow, M. (ed.) Climate Change and UK Forests. Forestry Commission, Edinburgh; UK. p. 99–118.

FAO 2001a. Global ecological zoning for the Global Forest Resources Assessment 2000. FAO Forestry Paper 56, Food and Agriculture Organization of the United Nations, Rome, Italy. 211 p.

FAO 2001b. Global forest resource assessment 2000 – main report. FAO Forestry Paper 140, Food and Agriculture Organization of the United Nations, Rome, Italy. 479 p.

FAO 2006. Global forest resource assessment 2005 – Progress towards sustainable forest management (main report). FAO Forestry Paper 147, Food and Agriculture Organization of the United Nations, Rome, Italy. 320 p.

FAOSTAT 2003. FAO statistics database on the world wide web. [Internet site]. Food and Agriculture Organization of the United Nations, Rome, Italy. Available at: http://faostat.fao.org/ [Cited 7 Jan 2009].

Fearnside, P.M. 1999. Biodiversity as an environmental service in Brazil's Amazonian forests: risks, value and conservation. Environ. Conserv. 26(4): 305–321.

Feeley, K.J., Wright, S.J., Supardi, M.N.N., Kassim, A.R. & Davies, S.J. 2007. Decelerating growth in tropical forest trees. Ecol. Lett. 10(6): 461–469.

Field, C.B., Lobell, D.B., Peters, H.A. & Chiariello, N.R. 2007a. Feedbacks of terrestrial ecosystems to climate change. Annu. Rev. of Environ. and Resour. 32: 1–29.

Field, C.B., Mortsch, L.D., Brklacich, M., Forbes, D.L., Kovacs, P., Patz, J.A., Running, S.W. & Scott, M.J. 2007b. North America. In: Parry, M.L., Canziani, O.F., Palutikof, J.P., van der Linden, P.J. & Hanson, C.E. (eds.). Climate change 2007: Impacts, adaptation and vulnerability. Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel of Climate Change (IPCC). Cambridge University Press, Cambridge, UK. p. 617–652.

Finzi, A.C., Canham, C.D. & Van Breeman, N. 1998. Canopy tree-soil interactions within temperate forests: Species effects on pH and cations (vol 8, p. 447, 1998). Ecol. Appl. 8(2): 447–454.

Fischlin, A. & Gyalistras, D. 1997. Assessing impacts of climatic change on forests in the Alps. Global Ecol. Biogeogr. Lett. 6(1): 19–37.

Fischlin, A., Midgley, G.F., Price, J.T., Leemans, R., Gopal, B., Turley, C., Rounsevell, M.D.A., Dube, O.P., Tarazona, J. & Velichko, A.A. 2007. Ecosystems, their properties, goods and services. In: Parry, M.L., Canziani, O.F., Palutikof, J.P., van der Linden, P.J. & Hanson, C.E. (eds.). Climate change 2007: Impacts, adaptation and vulnerability. Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel of Climate Change (IPCC). Cambridge University Press, Cambridge, UK. p. 211–272.

Fisher, B.S., Nakicenovic, N., Alfsen, K., Corfee Morlot, J., de la Chesnaye, F., Hourcade, J.C., Jiang, K., Kainuma, M., La Rovere, E., Matysek, A., Rana, A., Riahi, K., Richels, R., Rose, S., van Vuuren, D. & Warren, R. 2007. Issues related to mitigation in the long term context. In: Metz, B., Davidson, O.R., Bosch, P.R., Dave, R. & Meyer, L.A. (eds.). Climate change 2007: Mitigation. Contribution of Working Group III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change (IPCC). Cambridge University Press, Cambridge, UK. p. 95–116.

Fitzpatrick, M.C., Gove, A.D., Sanders, N.J. & Dunn, R.R. 2008. Climate change, plant migration, and range collapse in a global biodiversity hotspot: the Banksia (Proteaceae) of Western Australia. Glob. Chang. Biol. 14: 1337–1352.

Flannigan, M., Campbell, I., Wotton, M., Carcaillet, C., Richard, P. & Bergeron, Y. 2001. Future fire in Canada's boreal forest: paleoecology results and general circulation model – regional climate model simulations. Can. J. For. Res. 31(5): 854–864.

Flannigan, M. & Wotton, B. 2001. Climate, weather, and area burned. In: Johnson, E. & Miyanishi, K. (eds.). Forest Fires Behavior and Ecological Effects. Academic Press, San Diego. p. 351–373.

Flannigan, M.D., Logan, K.A., Amiro, B.D., Skinner, W.R. & Stocks, B.J. 2005. Future area burned in Canada. Clim. Chang. 72(1–2): 1–16.

Fleming, R.A. & Candau, J.N. 1998. Influences of climatic change on some ecological processes of an insect outbreak system in Canada's boreal forests and the implications for biodiversity. Environ. Monitor. Assess. 49(2–3): 235–249.

Folgarait, P.J., Marquis, R.J., Ingvarsson, P., Braker, H.E. & Arguedas, M. 1995. Patterns of attack by insect herbivores and a fungus on saplings in a tropical tree plantation. Environ. Entomol. 24(6): 1487–1494.

Foster, P. 2001. The potential negative impacts of global climate change on tropical montane cloud forests. Earth Sci. Rev. 55(1–2): 73–106.

Fowler, D., Cape, J.N., Coyle, M., Flechard, C., Kuylenstierna, J., Hicks, K., Derwent, D., Johnson, C. & Stevenson, D. 1999. The global exposure of forests to air pollutants. Wat. Air & Soil Poll. 116(1–2): 5–32.

Frelich, L.E. 2002. Forest dynamics and disturbance regimes. Studies from temperate evergreen-deciduous forests. Cambridge University Press, Cambridge, UK. 266 p.

Friedenberg, N.A., Sarkar, S., Kouchoukos, N., Billings, R.F. & Ayres, M.P. 2008. Temperature extremes, density dependence, and southern pine beetle (Coleoptera : Curculionidae) population dynamics in east Texas. Environ. Entomol. 37(3): 650–659.

Friedlingstein, P., Cox, P., Betts, R., Bopp, L., Von Bloh, W., Brovkin, V., Cadule, P., Doney, S., Eby, M., Fung, I., Bala, G., John, J., Jones, C., Joos, F., Kato, T., Kawamiya, M., Knorr, W., Lindsay, K., Matthews, H.D., Raddatz, T., Rayner, P., Reick, C., Roeckner, E., Schnitzler, K.G., Schnur, R., Strassmann, K., Weaver, A.J., Yoshikawa, C. & Zeng, N. 2006. Climate-carbon cycle feedback analysis: Results from the (CMIP)-M-4 model intercomparison. J. Clim. 19(14): 3337–3353.

Frumhoff, P.C., McCarthy, J.J., Melillo, J.M., Moser, S.C. & Wuebbles, D.J. 2007. Confronting climate change in the U.S. Northeast: science, impacts, and solutions. Synthesis report of the Northeast Climate Impacts Assessment (NECIA). Union of Concerned Scientists (UCS), Cambridge, MA, USA. 160 p.

Gentry, A.H. 1992. Tropical forest biodiversity – distributional patterns and their conservational significance. OIKOS 63(1): 19–28.

George, R.J., Speed, R.J., Simons, J.A., Smith, R.H., Ferdowsian, R., Raper, G.P. & Bennett, D.L. 2008. Long-term groundwater trends and their impact on the future extent of dryland salinity in Western Australia in a varibale climate. In, Proceedings 2nd International Salinity Forum, International Salinity Forum 2008, Adelaide, Australia. 29.3–2.4.2008.

Gerten, D., Schaphoff, S., Haberlandt, U., Lucht, W. & Sitch, S. 2004. Terrestrial vegetation and water balance – hydrological evaluation of a dynamic global vegetation model. J. Hydrol. 286(1–4): 249–270.

Gessler, A., Keitel, C., Kreuzwieser, J., Matyssek, R., Seiler, W. & Rennenberg, H. 2007. Potential risks for European beech (Fagus sylvatica L.) in a changing climate. Trees Struct. & Funct. 21(1): 1–11.

Gibbs, R.G.E. 1987. Preliminary floristic analysis of the major biomes in southern Africa. Bothalia 17(2): 213–227.

Gitay, H., Brown, S., Easterling, W. & Jallow, B. 2001. Ecosystems and their goods and services. In: McCarthy, J.J., Canziani, O.F., Leary, N.A., Dokken, D.J. & White, K.S. (eds.). Climate Change 2001 – Impacts, Adaptation, and Vulnerability. Contribution of Working Group II to the Third Assessment Report of the Intergovernmental Panel of Climate Change (IPCC). Cambridge University Press, Cambridge. p. 237–342.

Glen, M., Alfenas, A.C., Zauza, E.A.V., Wingfield, M.J. & Mohammed, C. 2007. Puccinia psidii: a threat to the Australian environment and economy a review. Australasian Plant Pathol. 36(1): 1–16.

Gobron, N., Pinty, B., Melin, F., Taberner, M., Verstraete, M.M., Belward, A., Lavergne, T. & Widlowski, J.L. 2005. The state of vegetation in Europe following the 2003 drought. Int. J. Remote Sensing 26(9): 2013–2020.

Golubyatnikov, L.L. & Denisenko, E.A. 2001. Response of the net primary production to climate change for the European Russia. Izvestia RAN Ser. Geogr. (Proc. Russian Acad. Sci. Geogr. Ser.)(6): 42–50 (in Russian, w. English abstract).

Golubyatnikov, L.L. & Denisenko, E.A. 2007. Model estimates of climate change impact on habitats of zonal vegetation for the plain territories of Russia. Biol. Bull. 34(2): 170–184.

Goodale, C.L., Apps, M.J., Birdsey, R.A., Field, C.B., Heath, L.S., Houghton, R.A., Jenkins, J.C., Kohlmaier, G.H., Kurz, W., Liu, S.R., Nabuurs, G.J., Nilsson, S. & Shvidenko, A.Z. 2002. Forest carbon sinks in the Northern Hemisphere. Ecol. Appl. 12(3): 891–899.

Goyer, R.A. 1991. Integrated pest-management of forest defoliators in the southeastern United States. For. Ecol. Manage. 39(1–4): 131–142.

Grant, R.F., Black, T.A., Gaumont-Guay, D., Kljun, N., Barrc, A.G., Morgenstern, K. & Nesic, Z. 2006. Net ecosystem productivity of boreal aspen forests under drought and climate change: mathematical modelling with Ecosys. Agric. For. Meteorol. 140(1–4): 152–170.

Groisman, P.Y., Sherstyukov, B.G., Razuvaev, V.N., Knight, R.W., Enloe, J.G., Stroumentova, N.S., Whitfield, P.H., Forland, E., Hannsen-Bauer, I., Tuomenvirta, H., Aleksandersson, H., Mescherskaya, A.V. & Karl, T.R. 2007. Potential forest fire danger over northern Eurasia: changes during the 20th century. Global Planet. Change 56(3–4): 371–386.

Grübler, A. & Nakicenovic, N. 2001. Identifying dangers in an uncertain climate. Nature 412(6842): 15.

Guerova, G. & Jones, N. 2007. A global model study of ozone enhancement during the August 2003 heat wave in Europe. Environ. chem. 4(5): 285–292.

Gullison, R.E., Frumhoff, P.C., Canadell, J.G., Field, C.B., Nepstad, D.C., Hayhoe, K., Avissar, R., Curran, L.M., Friedlingstein, P., Jones, C.D. & Nobre, C. 2007. Tropical forests and climate policy. Science 316(5827): 985–986.

Gyalistras, D. & Fischlin, A. 1999. Towards a general method to construct regional climatic scenarios for model-based impacts assessments. Petermanns geogr. Mitt. 143(4): 251–264.

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

Häkkinen, R., Linkosalo, T. & Hari, P. 1998. Effects of dormancy and environmental factors on timing of bud burst in Betula pendula. Tree Physiol. 18(10): 707–712.

Hall, A. 2008a. Better RED than dead: paying the people for environmental services in Amazonia. Phil. Trans. R. Soc. Lond. B 363(1498): 1925–1932.

Hall, J.S. 2008b. Seed and seedling survival of African mahogany (Entandrophragma spp.) in the Central African Republic: implications for forest management. For. Ecol. Manage. 255(2): 292–299.

Hallett, D.J., Mathewes, R.W. & Walker, R.C. 2003. A 1000-year record of forest fire, drought and lake-level change in southeastern British Columbia, Canada. Holocene 13(5): 751–761.

Hannah, L., Midgley, G.F., Lovejoy, T., Bond, W.J., Bush, M., Lovett, J.C., Scott, D. & Woodward, F.I. 2002. Conservation of biodiversity in a changing climate. Conserv. Biol. 16(1): 264–268.

Hanson, P.J. & Weltzin, J.F. 2000. Drought disturbance from climate change: response of United States forests. Sci. Total Environ. 262(3): 205–220.

Harrington, R., Fleming, R.A. & Woiwod, I.P. 2001. Climate change impacts on insect management and conservation in temperate regions: can they be predicted? Agric. For. Entomol. 3(4): 233–240.

Harris, S.A. 2008. Diversity of vascular plant species in the montane boreal forest of Western Canada in response to climatic changes during the last 25 ka, fire, and land use. Erdkunde 62(1): 59–73.

Hassan, R., Scholes, R. & Ash, N. (eds.). 2005. Ecosystems and human well-being – volume 1: current state & trends. Island Press, Washington, DC. 917 p.

Heath, R.N., Gryzenhout, M., Roux, J. & Wingfield, M.J. 2006. Discovery of the canker pathogen Chrysoporthe austroafricana on native Syzygium spp. in South Africa. Plant Dis. 90(4): 433–438.

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

Högberg, P. 2007. Environmental science – nitrogen impacts on forest carbon. Nature 447(7146): 781–782.

Hogg, E.H. & Hurdle, P.A. 1995. The aspen parkland in Western Canada: a dry-climate analog for the future boreal forest? Water Air & Soil Pollut. 82(1–2): 391–400.

Honnay, O., Verheyen, K., Butaye, J., Jacquemyn, H., Bossuyt, B. & Hermy, M. 2002. Possible effects of habitat fragmentation and climate change on the range of forest plant species. Ecol.

Lett. 5(4): 525–530.

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

Houghton, R.A. 2007. Balancing the global carbon budget. Annu. Rev. Earth Planetary Sci. 35: 313–347.

Hughes, L., Cawsey, E.M. & Westoby, M. 1996. Climatic range sizes of Eucalyptus species in relation to future climate change. Global Ecol. Biogeogr. Lett. 5(1): 23–29.

Hungate, B.A., Dukes, J.S., Shaw, M.R., Luo, Y.Q. & Field, C.B. 2003. Nitrogen and climate change. Science 302(5650): 1512–1513.

Huntingford, C., Fisher, R.A., Mercado, L., Booth, B.B.B., Sitch, S., Harris, P.P., Cox, P.M., Jones, C.D., Betts, R.A., Malhi, Y., Harris, G.R., Collins, M. & Moorcroft, P. 2008. Towards quantifying uncertainty in predictions of Amazon 'dieback'. Phil. Trans. R. Soc. Lond. B 363(1498): 1857–1864.

Hurley, B.P., Slippers, B. & Wingfield, M.J. 2007. A comparison of control results for the alien invasive woodwasp, Sirex noctilio, in the southern hemisphere. Agric. For. Entomol. 9(3): 159–171.

Hyvönen, R., Ågren, G.I., Linder, S., Persson, T., Cotrufo, M.F., Ekblad, A., Freeman, M., Grelle, A., Janssens, I.A., Jarvis, P.G., Kellomäki, S., Lindroth, A., Loustau, D., Lundmark, T., Norby, R.J., Oren, R., Pilegaard, K., Ryan, M.G., Sigurdsson, B.D., Strömgren, M., van Oijen, M. & Wallin, G. 2007. The likely impact of elevated [CO_2], nitrogen deposition, increased temperature and management on carbon sequestration in temperate and boreal forest ecosystems: a literature review. New Phytol. 173(3): 463–480.

IPCC 1996. IPCC Second Assessment Synthesis of Scientific-Technical Information relevant to interpreting Article 2 of the UN Framework Convention on Climate Change. Summary for Policymakers. Contribution of Working Group III to the Second Assessment Report of the Intergovernmental Panel of Climate Change (IPCC), IPCC Secretariat, WMO, Geneva, Switzerland. 27 p.

IPCC 2007a. Climate change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change (IPCC). In: Solomon, S., Qin, D., Manning, M., Chen, Z., Marquis, M., Averyt, K.B., Tignor, M. & Miller, H.L. (eds.). Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA. 996 p.

IPCC 2007b. Summary for policymakers. In: Parry, M.L., Canziani, O.F., Palutikof, J.P., van der Linden, P.J. & Hanson, C.E. (eds.). Climate change 2007: Impacts, adaptation and vulnerability. Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel of Climate Change (IPCC). Cambridge University Press, Cambridge, UK. p. 7–22.

IPCC 2007c. Summary for policymakers. In: Solomon, S., Qin, D., Manning, M., Chen, Z., Marquis, M., Averyt, K.B., Tignor, M. & Miller, H.L. (eds.). Climate change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change (IPCC). Cambridge University Press, Cambridge, UK and New York, NY, USA. p. 1–18.

IPCC 2007d. Climate Change 2007: Synthesis Report. Contribution of Working Groups I, II and III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change Core Writing Team, Pachauri, R.K and Reisinger, A. (eds.). IPCC, Geneva, Switzerland, 104 p

IPCC 2007e. Technical summary. In: Solomon, S., Qin, D., Manning, M., Chen, Z., Marquis, M., Averyt, K.B., Tignor, M. & Miller, H.L. (eds.). Climate change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change (IPCC). Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA. p. 19–91.

Iverson, L.R. & Prasad, A.M. 2001. Potential changes in tree species richness and forest community types following climate change. Ecosystems 4(3): 186–199.

Jarvis, P.G. & Aitken, A.M. 1998. European forests and global change. The likely impacts of rising CO_2 and temperature, Vol. XVIII. Cambridge University Press, Cambridge, UK. 379 p.

Jepsen, J.U., Hagen, S.B., Ims, R.A. & Yoccoz, N.G. 2008. Climate change and outbreaks of the geometrids Operophtera brumata and Epirrita autumnata in subarctic birch forest: evidence of a recent outbreak range expansion. J. Anim. Ecol. 77(2): 257–264.

Jobbagy, E.G. & Jackson, R.B. 2000. The vertical distribution of soil organic carbon and its relation to climate and vegetation. Ecol. Appl. 10(2): 423–436.

Johnstone, J.F. & Chapin, F.S. 2003. Non-equilibrium succession dynamics indicate continued northern migration of lodgepole pine. Global Change Biol. 9(10): 1401–1409.

Jones, E.A., Reed, D.D., Mroz, G.D., Liechty, H.O. & Cattelino, P.J. 1993. Climate stress as a precursor to forest decline: paper birch in northern Michigan 1985–1990. Can. J. Forest Res. 23(2): 229–233.

Jones, P., Amador, J., Campos, M., Hayhoe, K., Marín, M., Romero, J. & Fischlin, A. 2005. Generating climate change scenarios at high resolution for impact studies and adaptation: Focus on developing countries. In: Saloh, Y., Robledo, C. & Markku, K. (eds.). Tropical forests and adaptation to climate change: in search of synergies. Island Press, Washington, D.C., USA. p. 37–55.

Jönsson, U. 2006. A conceptual model for the development of Phytophthora disease in Quercus robur. New Phytol. 171(1): 55–68.

Jump, A.S., Hunt, J.M. & Peòuelas, J. 2006. Rapid climate change-related growth decline at the southern range edge of Fagus sylvatica. Glob. Chang. Biol. 12(11): 2163–2174.

Karnosky, D.F., Pregitzer, K.S., Zak, D.R., Kubiske, M.E., Hendrey, G.R., Weinstein, D., Nosal, M. & Percy, K.E. 2005. Scaling ozone responses of forest trees to the ecosystem level in a changing climate. Plant Cell & Environ. 28(8): 965–981.

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

Kauppi, P.E. 2003. New, low estimate for carbon stock in global forest vegetation based on inventory data. Silv. Fen. 37(4): 451–457.

Kauppi, P.E., Mielikäinen, K. & Kuusela, K. 1992. Biomass and carbon budget of European forests, 1971 to 1990. Science 256: 70–74.

Kauppi, P.E., Posch, M., Hänninnen, P., Henttonen, H.M., Ihalainen, A., Lappalainen, E., Starr, M. & Tamminen, P. 1997. Carbon reservoirs in peatlands and forests in the boreal regions of Finland. Silv. Fen. 31(1): 13–25.

Kellomäki, S., Kouki, J., Niemelä, P. & Peltola, H. 2001. Timber industry. In: Levin, S.A. (ed.) Encyclopedia of Biodiversity. Academic Press, San Diego, USA. p. 655–666.

Kellomäki, S., Peltola, H., Nuutinen, T., Korhonen, K.T. & Strandman, H. 2008. Sensitivity of managed boreal forests in Finland to climate change, with implications for adaptive management. Phil. Trans. R. Soc. Lond. B 363(1501): 2341–2351.

Kelly, A.E. & Goulden, M.L. 2008. Rapid shifts in plant distribution with recent climate change. Proc. Natl. Acad. Sci. USA 105(33): 11823–11826.

Kira, T. 1991. Forest ecosystems of east and southeast Asia in a global perspective. Ecol. Res. 6(2): 185–200.

Kirschbaum, M. & Fischlin, A. 1996. Climate change impacts on forests. In: Watson, R., Zinyowera, M.C. & Moss, R.H. (eds.).

Climate change 1995 – Impacts, adaptations and mitigation of climate change: scientific-technical analysis. Contribution of Working Group II to the Second Assessment Report of the Intergovernmental Panel of Climate Change. Cambridge University Press, Cambridge, UK. p. 95–129.

Kolomyts, E.G. 2006. Prognosis of the impact of global climate change on zonal ecosystems of the Volga river basin. Russ. J. Ecol. 37(6): 391–401.

Kurz, W.A. & Apps, M.J. 1999. A 70-year retrospective analysis of carbon fluxes in the Canadian Forest Sector. Ecol. Appl. 9(2): 526–547.

Kurz, W.A., Dymond, C.C., Stinson, G., Rampley, G.J., Neilson, E.T., Carroll, A.L., Ebata, T. & Safranyik, L. 2008. Mountain pine beetle and forest carbon feedback to climate change. Nature 452(7190): 987–990.

Kuusela, K. 1990. The dynamics of boreal coniferous forests, Vol. 112. The Finnish National Fund for Research and Development (SITRA), Helsinki, Suomi Finland. 172 p.

Laurance, W.F., Oliveira, A.A., Laurance, S.G., Condit, R., Nascimento, H.E.M., Sanchez-Thorin, A.C., Lovejoy, T.E., Andrade, A., D'Angelo, S., Ribeiro, J.E. & Dick, C.W. 2004. Pervasive alteration of tree communities in undisturbed Amazonian forests. Nature 428(6979): 171–175.

Laurance, W.F. & Williamson, G.B. 2001. Positive feedbacks among forest fragmentation, drought, and climate change in the Amazon. Conserv. Biol. 15(6): 1529–1535.

Leal, S., Eamus, D., Grabner, M., Wimmer, R. & Cherubini, P. 2008. Tree rings of Pinus nigra from the Vienna basin region (Austria) show evidence of change in climatic sensitivity in the late 20th century. Can. J. Forest Res. 38(4): 744–759.

Lebauer, D.S. & Treseder, K.K. 2008. Nitrogen limitation of net primary productivity in terrestrial ecosystems is globally distributed. Ecology 89(2): 371–379.

Leigh, E.G., Davidar, P., Dick, C.W., Puyravaud, J.P., Terborgh, J., Ter Steege, H. & Wright, S.J. 2004. Why do some tropical forests have so many species of trees? Biotropica 36(4): 447–473.

Lenihan, J.M., Daly, C., Bachelet, D. & Neilson, R.P. 1998. Simulating broad-scale fire severity in a Dynamic Global Vegetation Model. Northwest Sci. 72: 91–103.

Lewis, S.L., Lopez-Gonzalez, G., Sonké, B., Affum-Baffoe, K., Baker, T.R., Ojo, L.O., Phillips, O.L., Reitsma, J.M., White, L., Comiskey, J.A., Djuikouo K, M, Ewango, C.E.N., Feldpausch, T.R., Hamilton, A.C., Gloor, M., Hart, T., Hladik, A., Lloyd, J., Lovett, J.C., Makana, J., Malhi, Y., Mbago, F.M., Ndangalasi, H.J., Peacock, J., Peh, K.S.H., Sheil, D., Sunderland, T., Swaine, M.D., Taplin, J., Taylor, D., Thomas, S.C., Votere, R. & Wöll, H. 2009. Increasing carbon storage in intact African tropical forests. Nature 457(19.Feb.): 1003–1007.

Lewis, S.L., Malhi, Y. & Phillips, O.L. 2004a. Fingerprinting the impacts of global change on tropical forests. Phil. Trans. R. Soc. Lond. B 359(1443): 437–462.

Lewis, S.L., Phillips, O.L., Baker, T.R., Lloyd, J., Malhi, Y., Almeida, S., Higuchi, N., Laurance, W.F., Neill, D.A., Silva, J.N.M., Terborgh, J., Lezama, A.T., Martinez, R.V., Brown, S., Chave, J., Kuebler, C., Vargas, P.N. & Vinceti, B. 2004b. Concerted changes in tropical forest structure and dynamics: evidence from 50 South American long-term plots. Phil. Trans. R. Soc. Lond. B 359(1443): 421–436.

Lieth, H. & Whittaker, R.H. (eds.). 1975. Primary productivity of the biosphere. Springer, Berlin. 339 p.

Liski, J., Perruchoud, D. & Karjalainen, T. 2002. Increasing carbon stocks in the forest soils of western Europe. For. Ecol. Manage. 169(1): 159–175.

Lobo, A. & Maisongrande, P. 2006. Stratified analysis of satellite imagery of SW Europe during summer 2003: the differential response of vegetation classes to increased water deficit. Hydrol. Earth Syst. Sci. 10(2): 151–164.

Logan, J.A. & Powell, J.A. 2001. Ghost forests, global warming, and the mountain pine beetle (Coleoptera: Scolytidae). Am. Entomol.: 160–172.

Logan, J.A., Regniere, J. & Powell, J.A. 2003. Assessing the impacts of global warming on forest pest dynamics. Frontiers in Ecol. & Environ. 1(3): 130–137.

Lucht, W., Schaphoff, S., Erbrecht, T., Heyder, U. & Cramer, W. 2006. Terrestrial vegetation redistribution and carbon balance under climate change. Carbon Balance Manage. 1(6): 1–7.

Luyssaert, S., Schulze, E.D., Börner, A., Knohl, A., Hessenmöller, D., Law, B.E., Ciais, P. & Grace, J. 2008. Old-growth forests as global carbon sinks. Nature 455(7210): 213–215.

Lynch, J.A. & Hollis, J.L.H.F.S. 2004. Climatic and landscape controls of the boreal forest fire regime: Holocene records from Alaska. J. Ecol. 92(3): 477–489.

Magnani, F., Mencuccini, M., Borghetti, M., Berbigier, P., Berninger, F., Delzon, S., Grelle, A., Hari, P., Jarvis, P.G., Kolari, P., Kowalski, A.S., Lankreijer, H., Law, B.E., Lindroth, A., Loustau, D., Manca, G., Moncrieff, J.B., Rayment, M., Tedeschi, V., Valentini, R. & Grace, J. 2007. The human footprint in the carbon cycle of temperate and boreal forests. Nature 447(7146): 848–852.

Malhi, Y., Betts, R.A. & Roberts, T. (eds.) 2008. Climate change and the fate of the Amazon. Philosophical Transactions of the Royal Society of London B: Biological Sciences, 363(1498): 1727–1932.

Malhi, Y. & Phillips, O.L. 2004. Tropical forests and global atmospheric change: a synthesis. Phil. Trans. R. Soc. Lond. B 359(1443): 549–555.

Malhi, Y. & Wright, J. 2004. Spatial patterns and recent trends in the climate of tropical rainforest regions. Phil. Trans. R. Soc. Lond. B 359(1443): 311–329.

Marshall, J.D., Blair, J.M., Peters, D.P.C., Okin, G., Rango, A. & Williams, M. 2008. Predicting and understanding ecosystem responses to climate change at continental scales. Frontiers in Ecol. & Environ. 6(5): 273–280.

Martin, P.H., Nabuurs, G.J., Aubinet, M., Karjalainen, T., Vine, E.L., Kinsman, J. & Heath, L.S. 2001. Carbon sinks in temperate forests. Annu. Rev. Energ. Environ. 26: 435–465.

Matarira, C.H. & Mwamuka, F.C. 1996. Vulnerability of Zimbabwe forests to global climate change. Clim. Res. 6(2): 135–136.

Mattson, W.J., Kuokkanen, K., Niemelä, P., Julkunen-Tiitto, R., Kellomäki, S. & Tahvanainen, J. 2004. Elevated CO_2 alters birch resistance to Lagomorpha herbivores. Glob. Chang. Biol. 10(8): 1402–1413.

May, R.M. 1990. How many species? Phil. Trans. R. Soc. Lond. B 330(1257): 293–304.

McClean, C.J., Lovett, J.C., Kuper, W., Hannah, L., Sommer, J.H., Barthlott, W., Termansen, M., Smith, G.E., Tokamine, S. & Taplin, J.R.D. 2005. African plant diversity and climate change. Ann. of the Missouri Bot. Garden 92(2): 139–152.

McLaughlin, S.B., Nosal, M., Wullschleger, S.D. & Sun, G. 2007. Interactive effects of ozone and climate on tree growth and water use in a southern Appalachian forest in the USA. New Phytol. 174(1): 109–124.

McMillan, A.M.S., Winston, G.C. & Goulden, M.L. 2008. Age-dependent response of boreal forest to temperature and rainfall variability. Glob. Chang. Biol. 14(8): 1904–1916.

Meehl, G.A., Stocker, T.F., Collins, W.D., Friedlingstein, P., Gaye, A.T., Gregory, J.M., Kitoh, A., Knutti, R., Murphy, J.M., Noda, A., Raper, S.C.B., Watterson, I.G., Weaver, A.J. & Zhao, Z.C. 2007. Global climate projections. In: Solomon, S., Qin, D., Manning, M., Chen, Z., Marquis, M., Averyt, K.B., Tignor, M. & Miller, H.L. (eds.). Climate change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change (IPCC). Cambridge University Press, Cambridge, UK and New York, NY, USA. p. 747–845.

Melillo, J.M., McGuire, A.D., Kicklighter, D.W., Moore Iii, B., Vorosmarty, C.J. & Schloss, A.L. 1993. Global climate change and terrestrial net primary production. Nature 363(6426): 234–240.

Midgley, G.F., Hannah, L., Millar, D., Rutherford, M.C. & Powrie, L.W. 2002. Assessing the vulnerability of species richness to anthropogenic climate change in a biodiversity hotspot. Global Ecol. Biogeogr. 11(6): 445–451.

Midgley, G.F., Rutherford, M.C. & Bond, W.J. 2001. The heat is on – impacts of climate change on plant diversity in South Africa. WWF-sponsored booklet, NBI, Cape Town, South Africa. 10 p.

Miles, L., Grainger, A. & Phillips, O. 2004. The impact of global climate change on tropical forest biodiversity in Amazonia. Global Ecol. Biogeogr. 13(6): 553–565.

Mitchell, J.D. 2002. Termites as pests of crops, forestry, rangeland and structures in southern Africa and their control. Sociobiol. 40(1): 47–69.

Moiseev, B.N. & Alyabina, I.O. 2007. Assessment and mapping of components of carbon and nitrogen balances in main biomes of Russia. Izvestia RAN Ser. Geogr. (Proc. Russian Acad. Sci. Geogr. Ser.) September–Oktober(5): 116–127 (in Russian, w. English abstract).

Morales, P., Hickler, T., Rowell, D.P., Smith, B. & Sykes, M.T. 2007. Changes in European ecosystem productivity and carbon balance driven by regional climate model output. Glob. Chang. Biol. 13(1): 108–122.

Morgan, M.G., Pitelka, L.F. & Shevliakova, E. 2001. Elicitation of expert judgments of climate change impacts on forest ecosystems. Clim. Change 49(3): 279–307.

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

Myking, T. & Heide, O.M. 1995. Dormancy release and chilling requirement of buds of latitudinal ecotypes of Betula pendula and B. pubescens. Tree Physiol. 15(11): 697–704.

Myneni, R.B., Dong, J., Tucker, C.J., Kaufmann, R.K., Kauppi, P.E., Liski, J., Zhou, L., Alexeyev, V. & Hughes, M.K. 2001. A large carbon sink in the woody biomass of Northern forests. Proc. Natl. Acad. Sci. USA 98(26): 14784–14789.

Nadelhoffer, K.J., Emmett, B.A., Gundersen, P., Kjonaas, O.J., Koopmans, C.J., Schleppi, P., Tietema, A. & Wright, R.F. 1999. Nitrogen deposition makes a minor contribution to carbon sequestration in temperate forests. Nature 398(6723): 145–148.

Nahrung, H.F., Schutze, M.K., Clarke, A.R., Duffy, M.P., Dunlop, E.A. & Lawson, S.A. 2008. Thermal requirements, field mortality and population phenology modelling of Paropsis atomaria Olivier, an emergent pest in subtropical hardwood plantations. For. Ecol. Manage. 255(8–9): 3515–3523.

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

Negri, A.J., Adler, R.F., Xu, L.M. & Surratt, J. 2004. The impact of Amazonian deforestation on dry season rainfall. J. Clim. 17(6): 1306–1319.

Neilson, R.P., Prentice, I.C., Smith, B., Kittel, T. & Viner, D. 1998. Simulated changes in vegetation distribution under global warming. In: Watson, R.T., Zinyowera, M.C., Moss, R.H. & Dokken, D.J. (eds.). Regional impacts of climatic change: An assessment of vulnerability. A special report of the Intergovernmental Panel on Climate Change (IPCC) Working Group II. Cambridge University Press, Cambridge, UK. p. Annex C, 439–456.

Nepstad, D., Lefebvre, P., Da Silva, U.L., Tomasella, J., Schlesinger, P., Solorzano, L., Moutinho, P., Ray, D. & Benito, J.G. 2004. Amazon drought and its implications for forest flammability and tree growth: a basin-wide analysis. Global Change Biol. 10(5): 704–717.

Nepstad, D.C., Stickler, C.M., Filho, B.S. & Merry, F. 2008. Interactions among Amazon land use, forests and climate: prospects for a near-term forest tipping point. Phil. Trans. R. Soc. Lond. B 363(1498): 1737–1746.

Niemelä, P., Chapin Iii, F.S., Danell, K. & Bryant, J.P. 2001. Herbivory-mediated responses of selected boreal forests to climatic change. Clim. Change 48(2–3): 427–440.

Noormets, A., McNulty, S.G., Deforest, J.L., Sun, G., Li, Q. & Chen, J. 2008. Drought during canopy development has lasting effect on annual carbon balance in a deciduous temperate forest. New Phytol. 179(3): 818–828.

O'Neill, G.A., Hamann, A. & Wang, T.L. 2008. Accounting for population variation improves estimates of the impact of climate change on species' growth and distribution. J. Appl. Ecol. 45(4): 1040–1049.

Ofori, D.A. & Cobbinah, J.R. 2007. Integrated approach for conservation and management of genetic resources of Milicia species in West Africa. For. Ecol. Manage. 238(1–3): 1–6.

Ohlemüller, R., Gritti, E.S., Sykes, M.T. & Thomas, C.D. 2006. Quantifying components of risk for European woody species under climate change. Glob. Chang. Biol. 12(9): 1788–1799.

Ohsawa, M. 1995. Latitudinal comparison of altitudinal changes in forest structure, leaf-type, and species richness in humid monsoon Asia. Vegetatio 121(1–2): 3–10.

Olson, J.S., Watts, J.A. & Allison, L.J. 1983. Major world ecosystem complexes 1:30 000 000: ranked by carbon in live vegetation. Publ., Oak Ridge National Laboratory – Environmental Sciences Division, Oak Ridge, Tennessee, U.S.A.

Päätalo, M.L., Peltola, H. & Kellomäki, S. 1999. Modelling the risk of snow damage to forests under short-term snow loading. For. Ecol. Manage. 116(1–3): 51–70.

Parmesan, C. 2006. Ecological and evolutionary responses to recent climate change. Annu. Rev. Ecol. Evol. Syst. 37: 637–669.

Parry, M.L. (ed.) 2000. Assessment of potential effects and adaptations for climate change in Europe: The Europe ACACIA Project. Report of concerted action of the environment programme of the Research Directorate General of the Commission of the European Communities 1465-458X 1465-458X, Jackson Environment Institute, University of East Anglia, Norwich, UK. 320 p.

Peltola, H., Kellomäki, S. & Vaisanen, H. 1999. Model computations of the impact of climatic change on the windthrow risk of trees. Clim. Change 41(1): 17–36.

Perry, D.A., Oren, R. & Hart, S.C. 2008 (2nd ed.). Major forest types and their climatic controls. In: Forest ecosystems. Johns Hopkins University Press, Baltimore. p. 41–59.

Phillips, O.L., Lewis, S.L., Baker, T.R., Chao, K.J. & Higuchi, N. 2008. The changing Amazon forest. Phil. Trans. R. Soc. Lond. B 363(1498): 1819–1827.

Phillips, O.L., Malhi, Y., Higuchi, N., Laurance, W.F., Núòez, P.V., Vásquez, R.M., Laurance, S.G., Ferreira, L.V., Stern, M., Brown, S. & Grace, J. 1998. Changes in the carbon balance of tropical forests: evidence from long-term plots. Science 282(16.0ct.): 439–442.

Phillips, O.L., Malhi, Y., Vinceti, B., Baker, T., Lewis, S.L., Higuchi, N., Laurance, W.F., Vargas, P.N., Martinez, R.V., Laurance, S., Ferreira, L.V., Stern, M., Brown, S. & Grace, J. 2002a. Changes in growth of tropical forests: Evaluating potential biases. Ecol. Appl. 12(2): 576–587.

Phillips, O.L., Martinez, R.V., Arroyo, L., Baker, T.R., Killeen, T., Lewis, S.L., Malhi, Y., Mendoza, A.M., Neill, D., Vargas, P.N., Alexiades, M., Ceron, C., Di Fiore, A., Erwin, T., Jardim, A., Palacios, W., Saldias, M. & Vinceti, B. 2002b. Increasing dominance of large lianas in Amazonian forests. Nature 418(6899): 770–774.

Piao, S.L., Ciais, P., Friedlingstein, P., Peylin, P., Reichstein, M., Luyssaert, S., Margolis, H., Fang, J.Y., Barr, A., Chen, A.P., Grelle, A., Hollinger, D.Y., Laurila, T., Lindroth, A., Richardson, A.D. & Vesala, T. 2008. Net carbon dioxide losses of northern ecosystems in response to autumn warming. Nature 451(7174): 49–U3.

Pimm, S.L. & Raven, P. 2000. Extinction by numbers. Nature 403(24.Feb.): 843–845.

Pitman, A.J., Narisma, G.T. & McAneney, J. 2007. The impact of climate change on the risk of forest and grassland fires in Australia. Clim. Chang. 84(3–4): 383–401.

Pitman, N.C.A., Jorgensen, P.M., Williams, R.S.R., Leon-Yanez, S. & Valencia, R. 2002. Extinction-rate estimates for a modern neotropical flora. Conserv. Biol. 16(5): 1427–1431.

Plattner, G.K., Knutti, R., Joos, F., Stocker, T.F., von Bloh, W., Brovkin, V., Cameron, D., Driesschaert, E., Dutkiewicz, S., Eby, M., Edwards, N.R., Fichefet, T., Hargreaves, J.C., Jones, C.D., Loutre, M.F., Matthews, H.D., Mouchet, A., Muller, S.A., Nawrath, S., Price, A., Sokolov, A., Strassmann, K.M. & Weaver, A.J. 2008. Long-term climate commitments projected with climate-carbon cycle models. J. Clim. 21(12): 2721–2751.

Possingham, H.P. & Wilson, K.A. 2005. Turning up the heat on hotspots. Nature 436(7053): 919–920.

Pregitzer, K.S. & Euskirchen, E.S. 2004. Carbon cycling and storage in world forests: biome patterns related to forest age. Glob. Chang. Biol. 10(12): 2052–2077.

Prentice, I.C., Bondeau, A., Cramer, W., Harrison, S.P., Hickler, T., Lucht, W., Sitch, S., Smith, B. & Sykes, M.T. 2007. Dynamic global vegetation modelling: quantifying terrestrial ecosystem responses to large-scale environmental change. In: Canadell, J.G., Pataki, D.E. & Pitelka, L.F. (eds.). Terrestrial ecosystems in a changing world. Springer-Verlag, Berlin. p. 175–192.

Prentice, I.C., Cramer, W.P., Harrison, S.P., Leemans, R., Monserud, R.A. & Solomon, A.M. 1992. A global biome model based on plant physiology and dominance, soil properties and climate. J. Biogeogr. 19(2): 117–134.

Price, D.T. & Scott, D. 2006. Final Report to Government of Canada: Climate Change Impacts and Adaptation Program. Canadian Forest Service (CFS), Northern Forestry Centre, Edmonton, AB, Canada. 53 p.

Ramanathan, V. & Feng, Y. 2008. On avoiding dangerous anthropogenic interference with the climate system: Formidable challenges ahead. PNAS 105(38): 14245–14250.

Ramirez, C.C., Zamudio, F., Verdugo, J.V. & Nunez, M.E. 2004. Differential susceptibility of Poplar hybrids to the aphid Chaitophorus leticomelas (Homoptera : Aphididae). J. Econ. Entomol. 97(6): 1965–1971.

Rastetter, E.B., Perakis, S.S., Shaver, G.R. & Agren, G.I. 2005. Terrestrial C sequestration at elevated-CO_2 and temperature: The role of dissolved organic N loss. Ecol. Appl. 15(1): 71–86.

Raupach, M.R., Marland, G., Ciais, P., Le Quere, C., Canadell, J.G., Klepper, G. & Field, C.B. 2007. Global and regional drivers of accelerating CO_2 emissions. PNAS 104(24): 10288–10293.

Ravindranath, N.H. & Sukumar, R. 1998. Climate change and tropical forests in India. Clim. Change 39(2–3): 563–581.

Raxworthy, C.J., Pearson, R.G., Rabibisoa, N., Rakotondrazafy, A.M., Ramanamanjato, J.B., Raselimanana, A.P., Wu, S, Nussbaum, R.A. & Stone, D.A. 2008. Extinction vulnerability of tropical montane endemism from warming and upslope displacement: a preliminary appraisal for the highest massif in Madagascar. Glob. Chang. Biol. 14(8): 1703–1720.

Rehfeldt, G.E., Wykoff, W.R. & Ying, C.C. 2001. Physiologic plasticity, evolution, and impacts of a changing climate on Pinus contorta. Clim. Change 50(3): 355–376.

Reich, P.B. & Frelich, L. 2002. Temperate deciduous forests. In: Mooney, H.A. & Canadell, J. (eds.). The earth system: biological and ecological dimensions of global environmental change. Wiley, Chichester. p. 565–569.

Reich, P.B. & Oleksyn, J. 2008. Climate warming will reduce growth and survival of Scots pine except in the far north. Ecol. Lett. 11(6): 588–597.

Reid, W.V., Mooney, H.A., Cropper, A., Capistrano, D., Carpenter, S.R., Chopra, K., Dasgupta, P., Dietz, T., Duraiappah, A.K., Hassan, R., Kasperson, R., Leemans, R., May, R.M., McMichael, A.J., Pingali, P., Samper, C., Scholes, R., Watson, R.T., Zakri, A.H., Shidong, Z., Ash, N.J., Bennett, E., Kumar, P., Lee, M.J., Raudsepp-Hearne, C., Simons, H., Thonell, J. & Zurek, M.B. 2005. Ecosystems and human well-being: synthesis. Island Press, Washington, DC. 155 p.

Rice, R.A. & Greenberg, R. 2000. Cacao cultivation and the conservation of biological diversity. Ambio 29(3): 167–173.

Rosenzweig, C., Casassa, G., Karoly, D.J., Imeson, A., Liu, C., Menzel, A., Rawlins, S., Root, T.L., Seguin, B. & Tryjanowski, P. 2007. Assessment of observed changes and responses in natural and managed systems. In: Parry, M.L., Canziani, O.F., Palutikof, J.P., van der Linden, P.J. & Hanson, C.E. (eds.). Climate change 2007: Impacts, adaptation and vulnerability. Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel of Climate Change (IPCC). Cambridge University Press, Cambridge, UK. p. 79–131.

Rounsevell, M.D.A., Berry, P.M. & Harrison, P.A. 2006. Future environmental change impacts on rural land use and biodiversity: a synthesis of the ACCELERATES project. Environ. Sci. Pol. 9: 93–100.

Roux, J., Mthalane, Z.L., de Beer, Z.W., Eisenberg, B. & Wingfield, M.J. 2006. Quambalaria leaf and shoot blight on Eucalyptus nitens in South Africa. Aust. Plant Pathol. 35(4): 427–433.

Ruosteenoja, K., Jylhä, K. & Tuomenvirta, H. 2005. Climate scenarios for FINADAPT studies of climate change adaptation. FINDAPT Working Paper 15, Finnish Environment Institute Mimeographs 345, Helsinki, Finland. 32 p.

Sabine, C.L., Heimann, M., Artaxo, P., Bakker, D.C.E., Chen, C.T.A., Field, C.B., Gruber, N., Le Quéré, C., Prinn, R.G., Richey, J.E., Lankao, P.R., Sathaye, J.A. & Valentini, R. 2004. Current status and past trends of the global carbon cycle. In: Field, C.B. & Raupach, M.R. (eds.). Global Carbon Cycle: Integrating Humans, Climate, and the Natural World. Island Press, Washington, D.C. p. 17–44.

Saigusa, N., Yamamoto, S., Hirata, R., Ohtani, Y., Ide, R., Asanuma, J., Gamo, M., Hirano, T., Kondo, H., Kosugi, Y., Li, S.G., Nakai, Y., Takagi, K., Tani, M. & Wang, H.M. 2008. Temporal and spatial variations in the seasonal patterns of CO_2 flux in boreal, temperate, and tropical forests in East Asia. Agric. For. Meteorol. 148(5): 700–713.

Sala, O.E., Chapin, I.F.S., Armesto, J.J., Berlow, E., Bloomfield, J., Dirzo, R., Huber Sanwald, E., Huenneke, L.F., Jackson, R.B., Kinzig, A., Leemans, R., Lodge, D.H., Mooney, H.A., Oesterheld, M., Leroy Poff, N., Sykes, M.T., Walker, B.H., Walker, M. & Wall, D.H. 2000. Global biodiversity scenarios for the year 2100. Science 287(5459): 1770–1774.

Schaphoff, S., Lucht, W., Gerten, D., Sitch, S., Cramer, W. & Prentice, I.C. 2006. Terrestrial biosphere carbon storage under alternative climate projections. Clim. Change 74(1–3): 97–122.

Schär, C., Vidale, P.L., Lüthi, D., Frei, C., Häberli, C., Liniger, M.A. & Appenzeller, C. 2004. The role of increasing temperature variability in European summer heatwaves. Nature 427(6972): 332–336.

Schellnhuber, H.J.p.C.W., Nakicenovic, N., Wigley, T. & Yohe, G. (eds.). 2006. Avoiding dangerous climate change. Cambridge University Press, Cambridge, UK. 406 p.

Schimel, D.S., House, J.I., Hibbard, K.A., Bousquet, P., Peylin, P., Braswell, B.H., Apps, M.J., Baker, D., Bondeau, A., Canadell, J., Churkina, G., Cramer, W., Denning, A.S., Field, C.B., Friedlingstein, P., Goodale, C., Heimann, M., Houghton, R.A., Melillo, J.M., Moore Iii, B., Murdiyarso, D., Noble, I., Pacala, S.W., Prentice, I.C., Raupach, M.R., Rayner, P.J., Scholes, R.J., Steffen, W.L. & Wirth, C. 2001. Recent patterns and mechanisms of carbon exchange by terrestrial ecosystems. Nature 414(8 Nov): 169–172.

Schlyter, P., Stjernquist, I., Bärring, L., Jönsson, A.M. & Nilsson, C. 2006. Assessment of the impacts of climate change and weather extremes on boreal forests in northern Europe, focusing on Norway spruce. Clim. Res. 31(1): 75–84.

Schneider, S.H. 2001. What is 'dangerous' climate change? Nature 411(3.May): 17–19.

Schneider, S.H., Semenov, S., Patwardhan, A., Burton, I., Magadza, C.H.D., Oppenheimer, M., Pittock, A.B., Rahman, A., Smith, J.B., Suarez, A. & Yamin, F. 2007. Assessing key vulnerabilities and the risk from climate change. In: Parry, M.L., Canziani, O.F., Palutikof, J.P., van der Linden, P.J. & Hanson, C.E. (eds.). Climate change 2007: Impacts, adaptation and vulnerability. Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel of Climate Change (IPCC). Cambridge University Press, Cambridge, UK. p. 779–810.

Scholes, R.J. & Biggs, R. (eds.). 2004. Ecosystem services in southern African: A regional assessment – A contribution to the Millennium Ecosystem Assessment, prepared by the regional-scale team of the Southern African Millennium Ecosystem Assessment. Council for Scientific and Industrial Research, Pretoria, South Africa. 84 p.

Scholze, M., Knorr, W., Arnell, N.W. & Prentice, I.C. 2006. A climate change risk analysis for world ecosystems. PNAS 103(35): 13116–13120.

Schroth, G., Krauss, U., Gasparotto, L., Aguilar, J.A.D. & Vohland, K. 2000. Pests and diseases in agroforestry systems of the humid tropics. Agrofor. Syst. 50(3): 199–241.

Schulze, E.D. 1989. Air pollution and forest decline in a spruce (Picea abies) forest. Science 244(May, 19): 776–783.

Schwartz, C.C. & Renecker, L.A. 2007. Nutrition and energetics. In: Franzmann, A.W., Schwartz, C.C. & McCabe, R.E. (eds.). Ecology and management of the North American Moose. University Press of Colorado, Boulder, USA.

Sedjo, R.A. 1992. Temperate forest ecosystems in the global carbon cycle. Ambio 21(4): 274–277.

Semazzi, F.H.M. & Song, Y. 2001. A GCM study of climate change induced by deforestation in Africa. Clim. Res. 17(2): 169–182.

Shepashenko, D., Shvidenko, A. & Nilsson, S. 1998. Phytomass (live biomass) and carbon of Siberian forests. Biomass Bioenerg. 14(1): 21–31.

Shlisky, A., Waugh, J., Gonzalez, P., Gonzalez, M., Manta, M., Santoso, H., Alvarado, E., Nuruddin, A.A., Rodríguez-Trejo, D.A., Swaty, R., Schmidt, D., Kaufmann, M., Myers, R., Alencar, A., Kearns, F., Johnson, D., Smith, J. & Zollner, D. 2007. Fire, ecosystems and people: threats and strategies for global biodiversity conservation. GFI Technical Report 2007-2, The Nature Conservancy, Arlington, VA. 17 p.

Shvidenko, A. & Nilsson, S. 2003. A synthesis of the impact of Russian forests on the global carbon budget for 1961–1998. Tellus B 55(2): 391–415.

Siddiqui, K.M., Mohammad, I. & Ayaz, M. 1999. Forest ecosystem climate change impact assessment and adaptation strategies for Pakistan. Clim. Res. 12(2–3): 195–203.

Sitch, S., Smith, B., Prentice, I.C., Arneth, A., Bondeau, A., Cramer, W., Kaplan, J.O., Levis, S., Lucht, W., Sykes, M.T., Thonicke, K. & Venevsky, S. 2003. Evaluation of ecosystem dynamics, plant geography and terrestrial carbon cycling in the LPJ dynamic global vegetation model. Global Change Biol. 9(2): 161–185.

Sivakumar, M.V.K., Das, H.P. & Brunini, O. 2005. Impacts of present and future climate variability and change on agriculture and forestry in the arid and semi-arid tropics. Clim. Chang. 70(1–2): 31–72.

Sohngen, B. & Sedjo, R. 2000. Potential carbon flux from timber harvests and management in the context of a global timber market. Clim. Chang. 44(1–2): 151–172.

Solberg, S., Hov, O., Sovde, A., Isaksen, I.S.A., Coddeville, P., De Backer, H., Forster, C., Orsolini, Y. & Uhse, K. 2008. European surface ozone in the extreme summer 2003. J. Geophys. Res. D 113(D7): D07307.

Solomon, A. 1996. Wood production under changing climate and land use. In: Watson, R., Zinyowera, M.C. & Moss, R.H. (eds.). Climate change 1995 – Impacts, adaptations and mitigation of climate change: scientific-technical analysis. Contribution of Working Group II to the Second Assessment Report of the Intergovernmental Panel of Climate Change. Cambridge University Press, Cambridge, UK. p. 487–510.

Solomon, A.M. & Leemans, R. 1990. Climatic change and landscape-ecological response: issues and analysis. In: Boer, M.M. & de Groot, R.S. (eds.). Landscape-ecological impact of climatic change. IOS Press, Amsterdam. p. 293–316.

Solomon, S., Plattner, G.K., Knutti, R. & Friedlingstein, P. 2009. Irreversible climate change due to carbon dioxide emissions. PNAS 106: 6, in press.

Soule, P.T. & Knapp, P.A. 2006. Radial growth rate increases in naturally occurring ponderosa pine trees: a late-20th century CO_2 fertilization effect? New Phytol. 171(2): 379–390.

Staver, C., Guharay, F., Monterroso, D. & Muschler, R.G. 2001. Designing pest-suppressive multistrata perennial crop systems: shade-grown coffee in Central America. Agrofor. Syst. 53(2): 151–170.

Steffen, W. & Canadell, P. 2005. Carbon dioxide fertilisation and climate change policy. Australian Greenhouse Office, Department of the Environment and Heritage, Canberra, A.C.T. 1–36 p.

Stocks, B.J., Mason, J.A., Todd, J.B., Bosch, E.M., Wotton, B.M., Amiro, B.D., Flannigan, M.D., Hirsch, K.G., Logan, K.A., Martell, D.L. & Skinner, W.R. 2002. Large forest fires in Canada, 1959–1997. J. Geophys. Res. D 108(D1): 8149.

Stokstad, E. 2005. Experimental drought predicts grim future for rainforest. Science 308(5720): 346–347.

Stone, C. 2001. Reducing the impact of insect herbivory in eucalypt plantations through management of extrinsic influences on tree vigour. Austral Ecol. 26(5): 482–488.

Su-See, L. 1999. Forest health in plantation forests in South-East Asia. Aust. Plant Pathol. 28(4): 283–291.

Suffling, R. 1992. Climate change and boreal forest fires in Fennoscandia and central Canada. In: Boer, M. & Koster, E. (eds.). Greenhouse-Impact on Cold-Climate Ecosystems and Landscapes. Catena Verlag, Cremlingen-Destedt, Netherlands. p. 111–132.

Sutton, M.A., Simpson, D., Levy, P.E., Smith, R.I., Reis, S., van Oijen, M. & de Vries, W. 2008. Uncertainties in the relationship between atmospheric nitrogen deposition and forest carbon sequestration. Glob. Chang. Biol. 14(9): 2057–2063.

Symon, C., Arris, L. & Heal, B. (eds.). 2005. Arctic climate impact assessment (ACIA) – Scientific report. Cambridge University Press, Cambridge, UK. 1042 p.

Terborgh, J., Lopez, L., Nunez, P., Rao, M., Shahabuddin, G., Orihuela, G., Riveros, M., Ascanio, R., Adler, G.H., Lambert, T.D. & Balbas, L. 2001. Ecological meltdown in predator-free forest fragments. Science 294(5548): 1923–1926.

Thomas, C.D., Cameron, A., Green, R.E., Bakkenes, M., Beaumont, L.J., Collingham, Y.C., Erasmus, B.F.N., de Siqueira, M.F., Grainger, A., Hannah, L., Hughes, L., Huntley, B., van Jaarsveld, A.S., Midgley, G.F., Miles, L., Ortega-Huerta, M.A., Peterson, A.T., Phillips, O.L. & Williams, S.E. 2004. Extinction risk from climate change. Nature 427(6970): 145–148.

Thompson, I.D., Flannigan, M.D., Wotton, B.M. & Suffling, R. 1998. The effects of climate change on landscape diversity: An example in Ontario forests. Environ. Monitor. Assess. 49(2–3): 213–233.

Thuiller, W., Midgley, G.F., Rouget, M. & Cowling, R.M. 2006. Predicting patterns of plant species richness in megadiverse South Africa. Ecography 29(5): 733–744.

Thürig, E., Palosuo, T., Bucher, J. & Kaufmann, E. 2005. The impact of windthrow on carbon sequestration in Switzerland: a model-based assessment. For. Ecol. Manage. 210(1–3): 337–350.

Tran, J.K., Ylioja, T., Billings, R.F., Regniere, J. & Ayres, M.P. 2007. Impact of minimum winter temperatures on the population dynamics of Dendroctonus frontalis. Ecol. Appl. 17(3): 882–899.

Trenberth, K.E. & Hoar, T.J. 1996. The 1990–1995 El Nino Southern Oscillation event: Longest on record. Geophys. Res. Lett. 23(1): 57–60.

Trenberth, K.E., Jones, P.D., Ambenje, P., Bojariu, R., Easterling, D., Klein Tank, A., Parker, D., Rahimzadeh, F., Renwick, J.A., Rusticucci, M., Soden, B. & Zhai, P. 2007. Observations: surface and atmospheric climate change. In: Solomon, S., Qin, D., Manning, M., Chen, Z., Marquis, M., Averyt, K.B., Tignor, M. & Miller, H.L. (eds.). Climate change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change (IPCC). Cambridge University Press, Cambridge, UK and New York, NY, USA. p. 235–336.

Turner, J.A., Buongiorno, J. & Zhu, S.S. 2006. An economic model of international wood supply, forest stock and forest area change. Scand. J. For. Res. 21(1): 73–86.

Tylianakis, J.M., Tscharntke, T. & Lewis, O.T. 2007. Habitat modification alters the structure of tropical host-parasitoid food webs. Nature 445(7124): 202–205.

Van Bael, S.A., Aiello, A., Valderrama, A., Medianero, E., Samaniego, M. & Wright, S.J. 2004. General herbivore outbreak following an El Nino-related drought in a lowland Panamanian forest. J. Trop. Ecol. 20(6): 625–633.

Van Vuuren, D.P., Meinshausen, M., Plattner, G.K., Joos, F., Strassmann, K.M., Smith, S.J., Wigley, T.M.L., Raper, S.C.B., Riahi, K., Dela Chesnaye, F., den Elzen, M.G.J., Fujino, J., Jiang, K., Nakicenovic, N., Paltsev, S. & Reilly, J.M. 2008. Temperature increase of 21st century mitigation scenarios. PNAS 105(40): 15258–15262.

Vautard, R., Beekmann, M., Desplat, J., Hodzic, A. & Morel, S. 2007. Air quality in Europe during the summer of 2003 as a prototype of air quality in a warmer climate. Comptes Rendus Geosci. 339(11–12): 747–763.

Venäläinen, A., Tuomenvirta, H., Heikinheimo, M., Kellomäki, S., Peltola, H., Strandman, H. & Väisänen, H. 2001. Impact of climate change on soil frost under snow cover in a forested landscape. Clim. Res. 17(1): 63–72.

Villers-Ruiz, L. & Trejo-Vazquez, I. 1998. Impacto del cambio climático en los bosques y áreas naturales protegidas de México (Impact of climatic change in forests and natural protected areas of Mexico). Interciencia 23(1): 10–19.

Walter, H. 1979. Vegetation of the Earth and ecological systems of the geobiosphere. Springer-Verlag, New York, USA. 274 p.

Ward, N.L. & Masters, G.J. 2007. Linking climate change and species invasion: an illustration using insect herbivores. Glob. Chang. Biol. 13(8): 1605–1615.

Watson, R.T., Zinyowera, M.C., Moss, R.H. & Dokken, D.J. 1997. Regional impacts of climatic change: An assessment of vulnerability – Summary for Policymakers. A special report of the Intergovernmental Panel on Climate Change (IPCC) Working Group II. IPCC, Paris. 27 p.

Webb, T.J., Gaston, K.J., Hannah, L. & Woodward, F.I. 2006. Coincident scales of forest feedback on climate and conservation in a diversity hot spot. Proc. R. Soc. Lond. B 273(1587): 757–765.

Webb, T.J., Woodward, F.I., Hannah, L. & Gaston, K.J. 2005. Forest cover-rainfall relationships in a biodiversity hotspot: The Atlantic forest of Brazil. Ecol. Appl. 15(6): 1968–1983.

Wermelinger, B. & Seifert, M. 1999. Temperature-dependent reproduction of the spruce bark beetle *Ips typographus*, and analysis of the potential population growth. Ecol. Entomol. 24(1): 103–110.

Werth, D. & Avissar, R. 2004. The regional evapotranspiration of the Amazon. J. Hydrometeorol. 5(1): 100–109.

Weyant, J.P., de la Chesnaye, F.C. & Blanford, G.J. 2006. Overview of EMF-21: multigas mitigation and climate policy. The Energy J. 27(Special Issue #3): 1–32.

Wharton, T.N. & Kriticos, D.J. 2004. The fundamental and realized niche of the Monterey Pine aphid, Essigella californica (Essig) (Hemiptera : Aphididae): implications for managing softwood plantations in Australia. Diversity & Distribution 10(4): 253–262.

Wigley, T.M.L., Richels, R. & Edmonds, J.A. 1996. Economic and environmental choices in the stabilization of atmospheric CO_2 concentrations. Nature 379(6562): 240–243.

Wilbanks, T.J., Romero Lankao, P., Bao, M., Berkhout, F., Cairncross, S., Ceron, J.P., Kapshe, M., Muir-Wood, R. & Zapata-Marti, R. 2007. Industry, settlement and society. In: Parry, M.L., Canziani, O.F., Palutikof, J.P., van der Linden, P.J. & Hanson, C.E. (eds.). Climate change 2007: Impacts, adaptation and vulnerability. Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel of Climate Change (IPCC). Cambridge University Press, Cambridge, UK. p. 357–390.

Williams, D.W. & Liebhold, A.M. 2002. Climate change and the outbreak ranges of two North American bark beetles. Agric. For. Entomol. 4(2): 87–99.

Williams, D.W., Long, R.P., Wargo, P.M. & Liebhold, A.M. 2000. Effects of climate change on forest insect and disease outbreaks. In: Mickler, R.A., Birdsey, R.A. & Hom, J.L. (eds.). Responses of northern U.S. forests to environmental change. Springer-Verlag, Berlin, Germany. p. 455–494.

Williams, S.E., Bolitho, E.E. & Fox, S. 2003. Climate change in Australian tropical rainforests: an impending environmental catastrophe. Proc. R. Soc. Lond. Ser. B Biol. Sci. 270(1527): 1887–1892.

Wills, C., Harms, K.E., Condit, R., King, D., Thompson, J., He, F.L., Muller-Landau, H.C., Ashton, P., Losos, E., Comita, L., Hubbell, S., Lafrankie, J., Bunyavejchewin, S., Dattaraja, H.S., Davies, S., Esufali, S., Foster, R., Gunatilleke, N., Gunatilleke, S., Hall, P., Itoh, A., John, R., Kiratiprayoon, S., de Lao, S.L., Massa, M., Nath, C., Noor, M.N.S., Kassim, A.R., Sukumar, R., Suresh, H.S., Sun, I.F., Tan, S., Yamakura, T. & Zimmerman, E. 2006. Nonrandom processes maintain diversity in tropical forests. Science 311(5760): 527–531.

Wilson, K., Newton, A., Echeverria, C., Weston, C. & Burgman, M. 2005. A vulnerability analysis of the temperate forests of south central Chile. Biol. Conserv. 122(1): 9–21.

Wingfield, M.J. & Robison, D.J. 2004. Diseases and insect pests of Gmelina arborea: real threats and real opportunities. 28(2–3): 227–243.

Wotton, B.M. & Flannigan, M.D. 1993. Length of the fire season in a changing climate. Forestry Chron. 69(2): 187–192.

Wright, S.J., Calderón, O., Hernandéz, A. & Paton, S. 2004. Are lianas increasing in importance in tropical forests? A 17-year record from Panama. Ecology 85(2): 484–489.

Ympäristöministeriö 2007. Luonnon puolesta – ihmisen hyväksi. Suomen luonnon monimuotoisuuden suojelun ja kestävän käytön strategia ja toimintaohjelma 2006–2016. Suomen Ympäristö 35, Edita Prima, Helsinki, Finland. 162 p.

Zackrisson, O. 1977. Influence of forest fires on north Swedish boreal forest. OIKOS 29(1): 22–32.

Zhao, Y.X., Wang, C.Y., Wang, S.L. & Tibig, L. 2005. Impacts of present and future climate variability on agriculture and forestry in the humid and sub-humid tropics. Clim. Chang. 70(1–2): 73–116.

Zhou, L., Tucker, C.J., Kaufmann, R.K., Slayback, D., Shabanov, N.V. & Myneni, R.B. 2001. In northern vegetation activity inferred from satellite data of vegetation index during 1981 to 1999. J. Geophys. Res. D 106(D17): 20069–20084.