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Energy and Environment

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Executive Summary

Modern energy systems have been central to the development of human societies. They have perhaps been the single most important determinant of growth of our industrial societies and our modern economy. Unfortunately, they have also been a key driver of many of the negative environmental trends observed in the world today. For example, current energy systems are the predominant source of carbon dioxide (CO_2) emissions, accounting for 84% of total global CO_2 emissions and 64% of global greenhouse gas (GHG) emissions related to human activities. Past trends suggest that this percentage is likely to increase in the future if our energy needs continue to be met by fossil fuels.

The impact of GHG emissions on climate is arguably the most significant environmental impact associated with our energy systems, as the effects of such emissions are felt globally. However, these effects will not necessarily be equitable. Due to the realities of global and national economics, the areas that may suffer the greatest impacts from climate change may be those that have to date contributed the least in terms of GHG emissions. Our fossil fuel-based energy systems also emit substantial quantities of other atmospheric pollutants, for example sulphur dioxide (SO_2), nitrogen oxides (NO_x), primary particulate matter (PM), and non-methane volatile organic compounds (NMVOCs), which degrade air quality and cause damage to health and ecosystems through processes such as acidification, eutrophication, and the formation of ground-level ozone (O_3) and secondary PM. Biomass-based energy systems can also have substantial impacts on land and water resources.

Nevertheless, climate change and the reduction of CO_2 emissions are issues that need to be addressed immediately if we are to prevent irreversible environmental change on a planetary scale. This requires an international effort to develop alternative pathways that will enable our global energy systems to keep us within safe limits of environmental change whilst still providing the energy required for our human development. There has been much discussion of what such 'safe limits' might be for climate change, with scientists urging for constraining the temperature increase to 2°C, or even 1.5°C. The Global Energy Assessment (GEA) scenarios have been developed with a view towards achieving the former target by stabilizing concentrations of GHGs to less than 450 parts per million (ppm) CO_2 (equivalent). This target is considered to provide a reasonable chance of avoiding average global mean surface temperature (GMT) increases of above 2°C. However, it is important to realize that even if these alternative pathways are achieved in the future, there is still a significant risk of substantial environmental change, since it is not certain how the climate will respond to changes in GHG emissions. It is also important to note that although the focus of the GEA scenarios is on long-lived GHGs, short-lived climate forcers are always emitted along with CO_2 from combustion sources in energy systems, and have both warming (e.g., black carbon (BC)) as well as cooling properties (e.g., sulphur, particulate organic carbon (OC)). Therefore, energy policies that seek to mitigate climate change should also consider these other radiative forcers. Doing so could have co-benefits, since many of these short-lived climate forcers are air pollutants, impacting human health as well as the environment. Ideally, mitigation policy would target emission reductions that improve the situation for both climate and air quality, though such efforts may well be confounded by gaps in our scientific knowledge and by socioeconomic conditions and political priorities that vary by global region.

In view of such considerations, this chapter discusses the role of our energy systems in relation both to climate change and air quality. The latter is achieved by assessing current knowledge about air pollutants whose major sources are energy-related activities. The scale of the stresses placed on ecosystems from climate change and air quality vary by global region. This is in part due to the fact that regions are situated at different points along the energy-transition pathway, with transitional progress determining both their energy mix and associated emissions. For example, biomass energy may release carbon monoxide (CO), NMVOCs, and primary PM including BC and OC, but the net CO_2 emissions are typically small. Hence, economic regions that tend to rely on these more traditional types of energy (e.g., many African countries) will tend to contribute less to climate change and may suffer from more localized effects of poor indoor and outdoor air quality. In contrast, rapidly industrializing countries, such as those in Asia, rely heavily on fossil fuels (in particular coal), which leads to emissions of SO_2, NO_x, PM, and CO_2. As a result, such economies will be contributing to climate change as well as suffering the effects of eutrophication, acidification, elevated concentration of fine PM and ground-level O_3. Finally, those economies that have industrialized and are now putting efforts into renewable energy (including some economies in Europe) are seeing improvements in air quality, but still have a long way to go to achieve the reductions in CO_2 emissions that would be needed to achieve the 2°C limit to global warming.

Of course, changes in atmospheric composition are not the only way in which energy systems are causing environmental degradation. Energy systems, especially those reliant on biomass and hydropower, are also having substantial impacts on land and water resources. The potential threat posed by the trend towards increased biomass energy, in particular through the displacement of food crops and the added pressure this places on already scarce water resources, also needs to be considered when identifying future pathways to sustainable energy.

Finally, it is not enough to try to identify sustainability criteria that focus only on particular and individual environmental threats caused by energy systems. Rather, it is necessary to develop indicators that treat all threats in a holistic manner, recognizing the connections that exist between sources, processes, and impacts that result in environmental degradation. Only by understanding such connections will it be possible to identify pathways of change that will truly lead us towards energy systems that are able to meet demand while keeping within the safe limits of change of the Earth's biophysical processes. As such, an important conclusion of Chapter 3 is that a move towards redefining global sustainability criteria for energy following the 'planetary boundaries' approach may be advisable. Such a move could provide the holistic framework necessary to ensure that our global energy systems develop to achieve sustainability on a planetary scale.

3.1 Introduction

Growing evidence indicates that humanity has entered a global phase of development, in which anthropogenic pressures on Earth-system processes are at risk of reaching the limits of hardwired biophysical processes on a planetary scale (Crutzen, 2002; Steffen et al., 2007). The human pressures on the climate system, the oceans, the stratosphere, and the biosphere have now reached a point where the prospect of large-scale deleterious impacts on human development cannot be excluded (Lenton et al., 2008; Schaeffer et al., 2008; Gordon et al., 2008).

The Earth may be entering a new geological era, the Anthropocene, in which humanity constitutes the main driver of planetary change (Steffen et al., 2007; Zalasiewicz et al., 2010). This change may be threatening the environmental stability of the current geological era, the Holocene. The perception of these risks has been amplified by newly acquired knowledge that Earth systems may cross 'tipping points,' resulting in nonlinear, abrupt, and potentially irreversible change, such as the destabilization of the Greenland ice sheet or the tropical rainforest systems (Schellnhuber, 2009).

Energy systems play a critical role in determining our ability to achieve global sustainability in the short- and long-term, as they depend on natural resources and are among the most significant drivers of environmental impacts on the Earth's physical and living systems (Steffen et al., 2007). Easy access to energy was a prerequisite for the rapid improvements in human welfare and the rapid growth of the world's population, which began at the start of the industrial revolution. Empirical evidence shows that in the mid-1950s, humanity embarked on a 'great acceleration' (Steffen et al., 2007), characterized by an exponential exploitation of natural resources and ecosystems (Figure 3.1) and closely associated with a similar exponential growth in global population, economic growth, and energy use (see Chapter 1).

Increased energy use is also accompanied by heavy environmental costs. The true magnitude of these costs is only now being fully realized. Perhaps the most threatening of these environmental impacts is the marked increases in atmospheric concentrations of trace gases, in particular CO_2 (Keeling and Whorf, 2005), which have occurred in parallel with this rapid human development (Steffen et al., 2007). Observations of increases in global surface temperature (GMT) (IPCC, 2007) provide evidence that these increases in GHGs are causing climate change and that, when data are compared from the same source regions, this change has been rising as quickly in the past decade as in the previous two decades, and has been steadily increasing over the last century (Figure 3.2).

Other impacts associated with energy systems, historically viewed as acting at the local or regional level, are now also threatening at the global level. Increases in energy use per capita and exponentially increasing populations have led to an agglomeration of local impacts, which pose global threats (e.g., deforestation from the harvesting of wood fuel and limited availability of fresh water due to water extraction and pollution). Additionally, environmental impacts resulting from extreme events such as nuclear accidents and oil spills can also have global implications, both in terms of environmental consequences and in terms of changes in public perception of 'safe' energy that would make up our future energy systems.

This chapter addresses the environmental impacts of energy systems, focusing on those that place pressure on the atmosphere, the terrestrial biosphere and the hydrosphere while health related impacts are discussed in Chapter 4. The chapter considers how these impacts, in particular those associated with atmospheric emissions related to energy systems, vary across the globe in Europe, the Americas, Africa, Asia and polar regions. The chapter then considers all these impacts in relation to current environmental sustainability criteria used in energy systems, giving thought to how these criteria could be modified to capture the full range of environmental impacts. Such a holistic approach is appropriate for efforts that seek to ensure planetary-scale sustainability.

3.2 The Atmosphere and Energy Systems

Energy-related activities are responsible for a major share of anthropogenic emissions of GHGs, other radiative forcing substances, and air pollutants into the atmosphere. For example, energy-related GHG emissions, mainly from fossil fuel combustion for heat supply, electricity generation, and transport, account for approximately 64% of total emissions, including carbon dioxide (CO_2), methane (CH_4) and some traces of nitrous oxide (N_2O) (IPCC, 2007a). These atmospheric pollutants are primarily associated with impacts caused either by their radiative forcing (RF) properties leading to climate change, by their deposition to sensitive ecosystems causing damage through processes such as acidification and eutrophication, or by high PM concentrations leading to health impacts. In some cases, the atmospheric pollutants may have a role in all these types of impacts. For example, ground-level ozone (O_3), which is a secondary pollutant formed from a series of chemical reactions involving nitrogen oxides, (NO_x), volatile organic compounds (VOC) and carbon monoxide (CO), may cause direct damage to human health, and vegetation and is a powerful GHG.

First, we begin by describing the most important energy-related atmospheric pollutants, their major emission sources, and how emissions have varied over recent decades (see Section 3.2.1). Emission trends are derived from the Emissions Database for Global Atmospheric Research (EDGAR) which provides data on global annual emissions for 1990, 1995, and 2000 for direct GHG, precursor gases, and acidifying gases (van Aardenne et al., 2001; Olivier et al., 2005).[1] We then describe how these emissions lead to impacts on climate, both in the long-term and short-term (Section 3.2.2) and on air quality leading to impacts on terrestrial and aquatic ecosystems resulting from acidic deposition and eutrophication

1 The GEA scenarios are based on more recent emission inventories from the Reference Concentration Pathways developed for the IPCC Fifth Assessment Report, to be published in 2013 and 2014. See www.iiasa.ac.at/web-apps/tnt/RcpDb/.

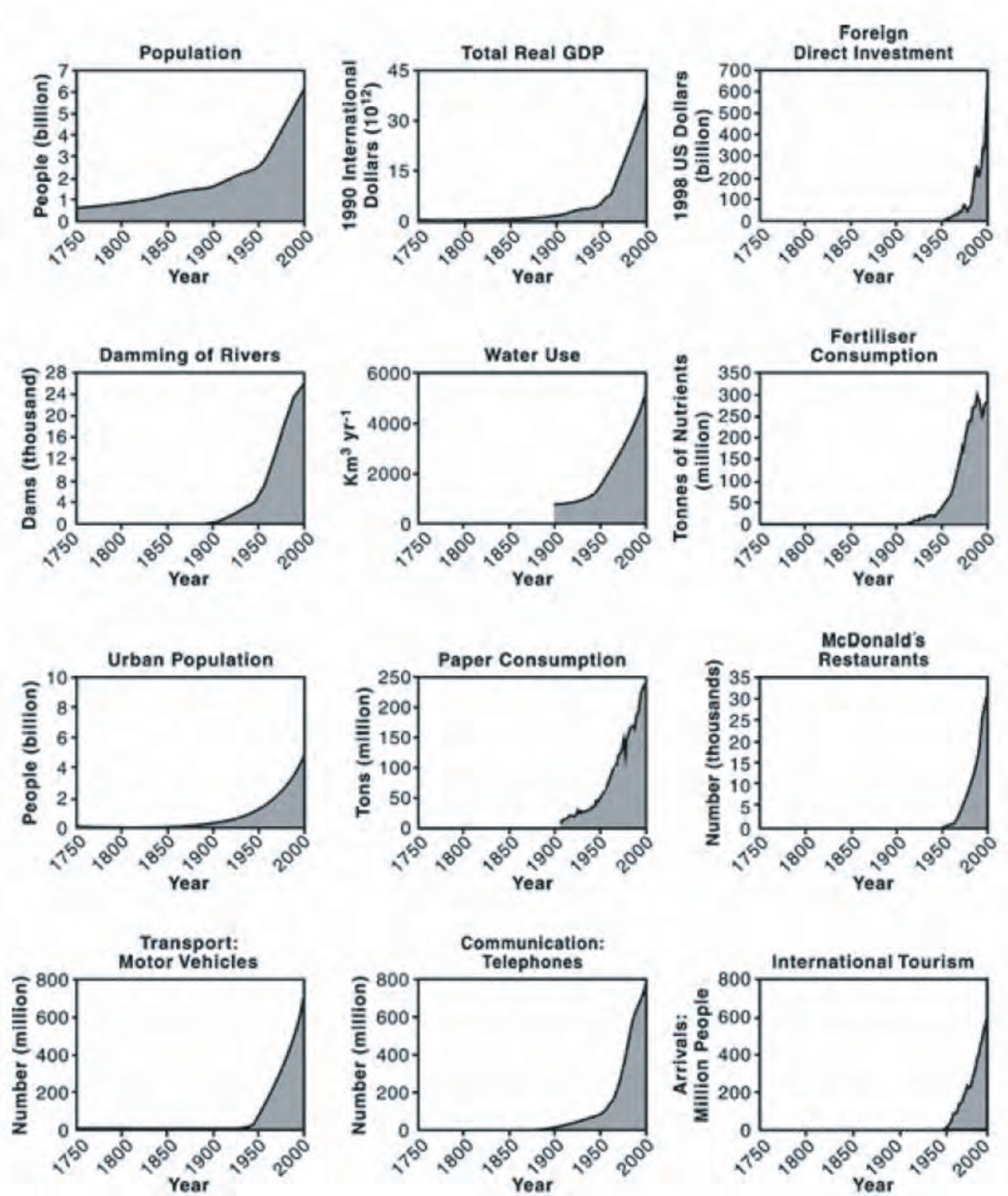

Figure 3.1 | Observed global trends of key environmental processes providing the evidence of the 'great acceleration' in human enterprise in the mid-1950s. Source: Steffen et al., 2007.

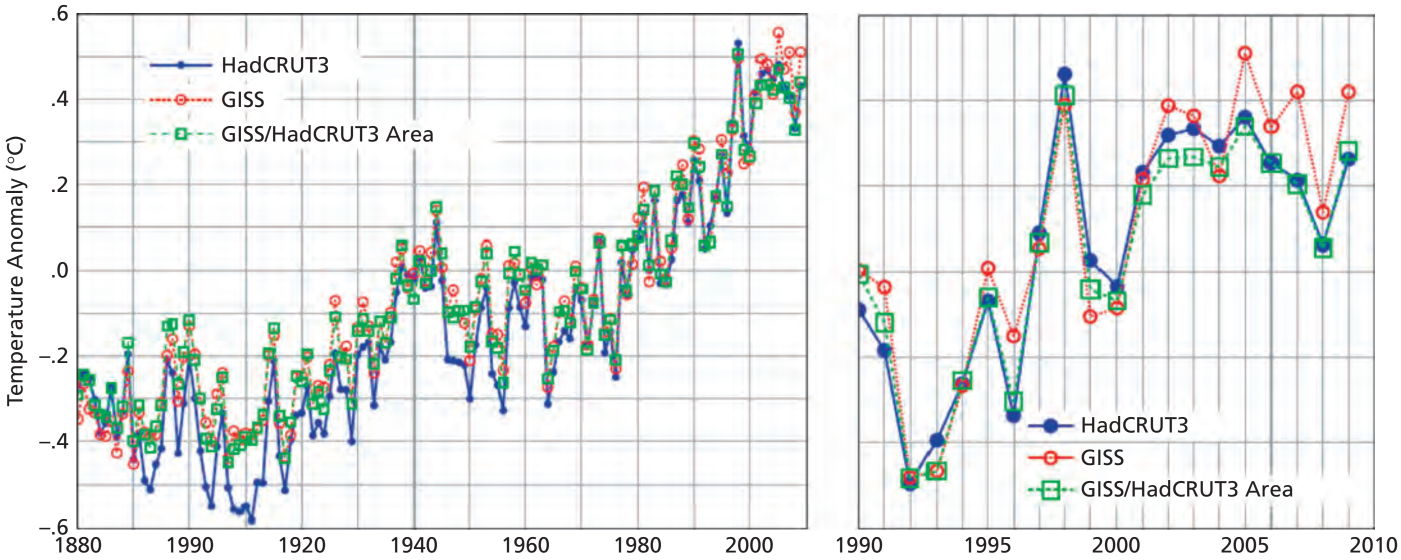

Figure 3.2 | Global surface temperature anomalies (°C) relative to 1961–1990 base period for three cases: HadCRUT, GISS, and GISS anomalies limited to the HadCRUT area. Source: Hansen et al., 2010.

(e.g., overfertilization with nitrogen), and through damage to ecosystems resulting from ground-level O_3 (Section 3.2.3); the health impacts from PM are discussed in Chapter 4. The regional variation in emission sources, climate change, and air quality impacts are discussed in Section 3.2.4.

3.2.1 Major Energy-related Sources of Atmospheric Pollution

3.2.1.1 Carbon Dioxide (CO_2)

Carbon dioxide released from anthropogenic sources has been identified as the key contributor to observed climate change (IPCC, 2007a). These sources include fossil fuel combustion, cement production, and emission flux from land-use changes. The largest sources of CO_2 emissions globally are related to energy, mainly from the combustion of fossil fuels such as coal, oil, and gas in power plants, vehicles, industrial facilities, and residential homes. In 2005, these energy-related activities accounted for 84% of total global anthropogenic CO_2 emissions (IEA, 2009).

Carbon dioxide emissions have increased continuously during the last century. The EDGAR database shows that global CO_2 emissions increased by 16% from 1990 to 2000 (Olivier et al., 2005), and a more recent analysis showed that global CO_2 emissions from fossil fuel combustion in 2008 have increased by 41% since 1990 (Le Quere et al., 2009). Although emissions increased in both developed and developing countries, the scale of the increase has varied between different global regions. For example, emissions of CO_2 in the United States increased by 16%, from 18.7 gigatonnes (Gt) of CO_2 in 1990 to 21.3 $GtCO_2$ in 2008 (US EPA, 2010a), while Chinese emissions increased by approximately 160%, from 9.2 $GtCO_2$ in 1990 to 23.8 $GtCO_2$ in 2007. However, when CO_2 emissions are expressed per capita, a different picture emerges. In 2007, the United States was the biggest per capita emitter at 18.7 tonnes, compared to China, which emitted only 4.6 tonnes per capita (IEA, 2009).

3.2.1.2 Methane (CH_4)

Methane is a potent GHG with a global warming potential (GWP) 25 times greater than that of CO_2 over a 100-year time horizon (IPCC 2007a). Importantly, CH_4 has an atmospheric lifetime of just over 10 years, which is fundamentally different from CO_2 and N_2O. Decreases in CH_4 emissions will therefore lead to a relatively rapid reduction in atmospheric concentrations of this GHG. Given the high RF of this pollutant, decreasing CH_4 emissions may be an attractive mitigation option.

Although CH_4 is relatively unreactive compared with other VOCs, it has a highly significant role in determining background O_3 concentrations on large geographic scales. Thus, higher emissions of CH_4 lead to elevated ground-level O_3 concentrations. The anthropogenic sources of CH_4 include rice agriculture, livestock, landfill, biomass burning, and fossil fuel combustion. Natural CH_4 is emitted from sources such as wetlands, oceans, forests, wildfires, termites, and geological sources. Total global pre-industrial emissions of CH_4 were dominated by natural sources (approximately 90%), with anthropogenic sources accounting for the rest (IPCC, 2007a). In contrast, anthropogenic emissions dominate present-day CH_4 budgets, accounting for more than 60% of the total global budget. Estimates of the share of anthropogenic CH_4 emissions due to fossil fuel extraction and combustion range from 20–30% (IPCC, 2007a).

3.2.1.3 Suphfur Dioxide (SO_2)

Sulphur dioxide and its atmospheric products (e.g., sulphate aerosols, sulphuric acid) cause a number of environmental problems. They can act as atmospheric radiative forcers (with implications for climate change), as air pollutants causing acidic deposition and acidification of terrestrial and aquatic ecosystems, and as air pollutants adding to the atmospheric PM load (with impacts on human health). These pollutants also act at a range of spatial levels, from local to global, depending on atmospheric circulation patterns. Sulphur dioxide is emitted into the atmosphere from both anthropogenic sources (e.g., fossil fuel combustion, industrial process) and natural sources (e.g., volcanic eruptions). It is estimated that anthropogenic sources account for more than 70% of SO_2 global emissions, 85% of which are from fossil fuel combustion (Olivier et al., 2005). The largest anthropogenic sources of SO_2 emissions are related to fossil fuel combustion at power plants and other industrial facilities. Smaller anthropogenic sources of SO_2 emissions include industrial processes such as extracting metal from ore, and the burning of high sulphur-containing fuels by locomotives, large ships, and non-road equipment.

The EDGAR database shows that global anthropogenic SO_2 emissions in 2000 were 150 megatonnes (Mt) of SO_2, coming mainly from power generation (54 $MtSO_2$, 35.6% of total), industrial fossil fuel combustion (24 $MtSO_2$, 16.2%), nonferrous metals melting (21 $MtSO_2$, 14.2%), other fossil fuel (10 $MtSO_2$, 6.8%), residential fossil fuel (8 $MtSO_2$, 5.4%), and international shipping (7 $MtSO_2$, 4.8%), according to Olivier et al. (2005).

Global SO_2 emissions have risen dramatically over the last century, approximately in parallel with increased fossil fuel use. Emission sources have changed considerably with time, both geographically and by sector. For example, emissions in most industrialized countries have fallen over the past two decades, due to the implementation of sulphur controls and a shift to lower sulphur fuels (Smith et al., 2011). Emissions in the United States have decreased from a high of 28 $MtSO_2$ in 1970 to 10 $MtSO_2$ in 2008 (US EPA, 2009). European anthropogenic SO_2 emissions have also decreased over the last two decades, from 55 $MtSO_2$ in 1980 to 15 $MtSO_2$ in 2004 (Vestreng et al., 2007). Conversely, anthropogenic SO_2 emissions in China are of increasing concern; they contributed about one quarter of the global total and more than 90% of East Asian emissions during the 1990s. A recent study showed that from 2000 to 2006, total SO_2 emissions in China increased by 53%, from 21.7 $MtSO_2$ to 33.2 $MtSO_2$, at an annual growth rate of 7.3% (Lu et al., 2010).

3.2.1.4 Nitrogen Oxides (NO_x)

Nitrogen oxides ($NO_x = NO + NO_2$) play a key role in tropospheric chemistry. Nitrogen oxides can either be deposited directly to ecosystems through dry deposition or through wet deposition, caused when nitrates (NO_3^-) form in cloud and rain. Both processes can cause eutrophication of ecosystems. Nitrogen oxides also play a role in the production of O_3 in the troposphere, where the abundance of O_3 is controlled by atmospheric concentrations of NO_x and VOCs. Nitrogen oxides can also contribute to fine particle pollution through the formation of nitrate aerosols. Nitrogen oxides are therefore linked to climate change (through contributions to RF atmospheric aerosols and tropospheric O_3 formation), to eutrophication (through wet and dry deposition), and ecosystem damage (again though tropospheric O_3 formation). Nitrogen oxides have also been linked to impacts on human health, as they can cause adverse effects on the respiratory system. They can also directly impact vegetation.

Like SO_2 emissions, global NO_x emissions have risen dramatically during the past century. It is estimated that global anthropogenic NO_x emissions increased five-fold between 1890 and 1990, from 6.9 MtN to 35.5 MtN (van Aardenne et al., 2001). After 1990, NO_x emissions from some industrialized regions began to decrease, mainly due to regulations in the transportation sector. Emissions in the United States decreased from 23 $MtNO_2$ in 1990 to 14.8 $MtNO_2$ in 2008, due to reductions in emissions from vehicles and power plants (US EPA, 2009). In Europe, road transport has been the dominant source of NOx emissions, accounting for 40% of the total emissions in 2005 (Vestreng et al., 2009). As a result of the combined control measures, the total NO_x emissions in Europe decreased by 32% between 1990 and 2005 (Vestreng et al., 2009). Nitrogen oxide emissions in China have continuously increased over the past two decades; the growth rate itself accelerated during this period. It is estimated that NOx emissions in China were 10.9 $MtNO_2$ in 1995 and 18.6 $MtNO_2$ in 2004, increasing by 70% during the period, at a 6.1% annual average growth rate (Zhang et al., 2007). More recent data estimates that nearly 21 $MtNO_2$ of NO_x have been emitted in China, compared to a value of 36.7 $MtNO_2$ of NO_x for the whole of Asia (Zhang et al., 2009).

3.2.1.5 Carbon Monoxide (CO)

Carbon monoxide is a significant air pollutant, capable of damaging human health as well as being an O_3 precursor. Carbon monoxide is emitted whenever fossil fuels and vegetation are incompletely combusted, whether in residential stoves, industrial boilers, vehicles, or through biomass burning. According to the EDGAR global inventory, in 2000 about 50% of CO emissions came from open biomass burning, 23% from biofuel combustion, and 22% from fossil fuel combustion, with the rest coming from industrial processes and waste treatment (Olivier et al., 2005).

Since the 1960s, great strides have been taken to reduce CO levels in the United States and Europe. However, in the developing world, few regulatory steps have been implemented. Emissions in the United States decreased from 185 MtCO in 1970 to 70 MtCO in 2008 (US EPA, 2009), while emissions in Asia increased from 207 MtCO in 1980 to 340 MtCO in 2003 (Ohara et al., 2007). A recent inventory study found that global CO emissions decreased by only 2–3% globally between 1988 and 1997, as increases in eastern Asia of 51% caused by rapid economic development were offset by declines in Europe and North America (Duncan et al., 2007). The largest decline, in Eastern Europe (45%), was largely caused by the economic contraction of the former Soviet Union (FSU). There were smaller declines in Western Europe (32%) and North America (17%), caused primarily by increasing levels of emissions control on vehicles (Duncan et al., 2007).

3.2.1.6 Non-methane Volatile Organic Compounds (NMVOCs)

Non-methane volatile organic compounds include a variety of chemicals that play an important role in atmospheric chemistry through tropospheric O_3 formation. There are many sources of NMVOCs in the atmosphere, including natural or 'biogenic' sources, such as trees and vegetation. Anthropogenic sources of NMVOCs include combustion of fossil and biofuels; biomass burning; the production, processing, and storage of liquid fuels; solvent use; and many other industrial processes. According to the EDGAR global inventory for 2000, about 42% of NMVOC emissions came from the combustion and processing of fossil fuels, 26% from open biomass burning, 16% from biofuel combustion, and 14% from solvent use and other industrial processes. Sectoral distribution might vary strongly between regions (e.g., Klimont et al., 2002; Wei Wei et al., 2008). Geographically, about 33% of the emissions came from Asia, 20% from Europe, 17% from Africa, 15% from Latin America, and 13% from North America (Olivier et al., 2005).

Global anthropogenic NMVOC emissions have risen dramatically during the past century. It is estimated that emissions increased from 24.8 MtNMVOC to 181 MtNMVOC between 1890 and 1990 (van Aardenne et al., 2001). After 1990, NMVOC emissions from some industrialized regions began to decrease. Emissions in the United States have decreased from 21.8 MtNMVOC in 1990 to 14.4 MtNMVOC in 2008 (US EPA, 2009), mainly due to emissions control on vehicles, as well as improved technologies related to solvent use. Meanwhile, emissions in Asia have increased from 21.9 MtNMVOC in 1980 to 45.5 MtNMVOC in 2003 (Ohara et al., 2007). As for NO_x, more recent estimates of 54 MtNMVOC for the whole of Asia for 2006 are provided by Zhang et al. (2009).

3.2.1.7 Black Carbon (BC) and Organic Carbon (OC)

Recent work has suggested that climate forcing by carbonaceous aerosols is probably a significant component of anthropogenic forcing. Forcing by BC from fossil fuel combustion ranges from about +0.1 to +0.3 W/m^2, hence having a warming influence on climate, and similar estimates for primary OC particles are −0.01 to −0.06 W/m^2 causing a cooling effect on climate. As global averages, these values, especially BC forcing, are significant relative to the average CO_2 forcing of about +1.5 W/m^2. Some studies suggest that regional aerosol forcings can be an order of magnitude greater than GHG forcings (Ramanathan and Carmichael, 2008). Carbonaceous aerosols are mainly produced during incomplete combustion of fossil fuels and biofuels, as well as open biomass burning. It has been estimated that global annual emissions in 1996 were 8.0 Mt for BC and 33.9 Mt for OC (Bond et al., 2004). The contributions of fossil fuel, biofuel, and open biomass burning are estimated at 38%, 20%, and 42% respectively for BC, and 7%, 19%, and 74% respectively for OC (Bond et al., 2004). Emissions of BC and OC have increased steadily over the past century. Global anthropogenic BC emissions increased from 2200 kilotonnes (kt) in 1900 to 4400 kt in 2000, and OC emissions increased from 5800 kt to 8700 kt during the same period (Bond et al., 2007).

Among 'contained' combustion sources (fossil fuel and biofuel), significant contributors to BC include the transportation, industry, and residential sectors, which account for 20%, 10%, and 25% respectively (Bond et al., 2004). Transportation is the most significant contributor to BC in developed regions, such as North America and Europe, with on-road and off-road diesels having approximately equal contributions. On the other hand, in developing regions like China, India, and Africa, the residential sector contributes the most to BC, though industry (e.g., coke making and brick kilns) and the transportation sector also makes significant contributions. For example, in China the residential and transport sector together comprise 63% and 82% of the total anthropogenic emissions respectively for BC and OC (Zhang et al., 2009). As a consequence of the poorer combustion in small devices, residential solid fuels (biofuel and coal) dominate 'contained' OC emissions in all regions except the Middle East and the Pacific. It is also estimated that residential solid fuels and transport contribute 20% and 4% to the global budget of OC respectively, if one considers all sources (e.g., open biomass burning (Bond et al., 2004)). The regional analyses for Asia presented in Zhang et al. (2009) and Klimont et al. (2009) highlight the dominant role of domestic combustion in OC emissions.

3.2.2 Climate Change and Energy Systems

The world's energy systems constitute an extremely important driver of climate change. This section reviews the current state of knowledge of climate change on consideration of energy system related emissions of GHGs and air pollutants.

3.2.2.1 Long-term Climate Change and Energy Systems

Key to understanding the physical mechanisms of climate change is the concept of energy balance and radiative forcing (RF) in the Earth's atmosphere (see Box 3.1). For the Earth's GMT to remain at an average of 15°C, the net incoming flux of solar radiation at the top of the atmosphere must equal the flux of long-wave radiation out to space. The chief physical mechanism by which the radiation imbalance arises as a consequence of human interference in the climate system is through increases in the atmospheric concentrations of long-lived GHGs. Long-lived GHGs, i.e., those gases that persist for periods of time ranging from decades to centuries, include CO_2, N_2O, and halocarbons. Other atmospheric trace gases are also crucial in determining the energy balance, including stratospheric O_3, which decreases RF (see Box 3.1); CH_4 and tropospheric O_3, which increase it; and aerosols, which on aggregate also decrease RF (IPCC, 2007a). The Intergovermental Panel on Climate Change (IPCC, 2007a) (see Box 3.2) estimates that the combined net RF

Box 3.1 | Radiative Forcing, Climate Sensitivity, and Carbon Dioxide Equivalent

Radiative Forcing (RF)

Radiative forcing (RF) can be defined as the net change in the energy balance between the Earth and space (i.e., the difference in incoming solar radiation less outgoing terrestrial or long-wave radiation) at the tropopause. It is quantified as the rate of energy change per unit area of the globe as measured at the top of the atmosphere and is expressed in units of 'watts per square meter' (W/m^2). Radiative forcing is used to assess and compare the anthropogenic and natural drivers of climate change (IPCC, 2007) and can be linearly related to the global mean equilibrium surface temperature (GMT) change (ΔGMTs); $\Delta GMT = \lambda RF$, where λ is the climate sensitivity parameter (e.g., Ramaswamy et al., 2001).

Climate Sensitivity (λ)

Climate sensitivity (λ) is a measure of the responsiveness of equilibrated global mean surface temperature (GMT) to a change in the radiative forcing equivalent to a doubling of the atmospheric equivalent CO_2 concentration (CO_2-eq) (IPCC, 2007a). Climate sensitivity (λ) is hard to predict, since it needs to incorporate various couplings, feedbacks (particularly those related to clouds, sea ice, and water vapor), and interactions that occur within the climate system in response to any changes within the system.

Carbon Dioxide Equivalent (CO_2-eq)

Carbon dioxide equivalent (CO_2-eq) is a universal unit of measurement used to indicate the GWP of one unit of CO_2 over a 100-year time horizon. It is used to evaluate the releasing of different GHGs against a common basis. Thus, for methane (CH_4) the GWP is 25, and for nitrous oxide (N_2O) the GWP is 296 (IPCC, 2007a).

for all anthropogenic agents is +1.6 W/m^2 (with a 0.6–2.4 W/m^2 90% confidence range) indicating that, since 1750, it is extremely likely that humans have exerted a substantial warming influence on climate.

The contributions from CO_2 and CH_4 to this RF are 1.66 W/m^2 (with a range of ±0.17 W/m^2) and 0.48 W/m^2 (with a range of ±0.05 W/m^2) respectively. The energy sector is important in determining emissions of both these GHGs. Energy systems are the predominant source of CO_2 emissions, accounting for 84% of total global CO_2 emissions in 2005 and for 64% of global GHG emissions related to human activities (IEA, 2009).

Observations of climate change in response to these anthropogenic increases in radiative forcers are now clearly being recorded (IPCC, 2007a). Observations show GMT to have risen by 0.74°C ± 0.18°C when estimated by a linear trend over the past 100 years (1906–2005), with the rate of warming over the past 50 years almost double that of the past 100 years (IPCC, 2007a; see also Figure 3.2). This has lead to changes in Earth system climate. For example, long-term trends in precipitation amounts from 1900 to 2005 have been observed across many large regions, with precipitation significantly increasing in the eastern parts of North and South America, northern Europe, and northern and central Asia. In contrast, drying has been observed in the Sahel, the Mediterranean, southern Africa, and parts of southern Asia. Substantial increases in heavy precipitation events have been observed (IPCC, 2007a). In addition, during the 1961–2003 period the average rate of global mean sea-level rise was estimated to be 1.8 ± 0.5 mm/yr (IPCC, 2007a).

The contemporary climate has moved out of the envelope of Holocene variability, sharply increasing the risk of dangerous climate change. Observations of a climate transition include a rapid retreat of summer sea ice in the Arctic Ocean (Johannessen, 2008), the retreat of mountain glaciers around the world (IPCC, 2007a), the loss of mass from the Greenland and West Antarctic ice sheets (Cazenave, 2006), an increased rate of sea-level rise in the last 10–15 years (Church and White, 2006), a 4° latitude pole-ward shift of subtropical regions (Seidel and Randel, 2006), increased bleaching and mortality in coral reefs (Bellwood et al., 2004; Stone, 2007), a rise in the number of large floods (Milly et al., 2002; MEA, 2005b), and the activation of slow feedback processes like the weakening of the oceanic carbon sink (Le Quéré et al., 2007).

Box 3.2 | Scientific Assessments of Climate Change and the IPCC

The main source of scientific knowledge on climate change is contained in the assessment reports of the Intergovernmental Panel on Climate Change (IPCC). The IPCC was established in 1988 by two United Nations Organizations, the World Meteorological Organization (WMO) and the United Nations Environment Programme (UNEP), to assess "the scientific, technical and socioeconomic information relevant for the understanding of the risk of human-induced climate change." The First Assessment Report, or FAR (IPCC, 1990), informed the intergovernmental negotiations that led to the United Nations Framework Convention on Climate Change (UNFCCC). The Second Assessment Report, or SAR (IPCC, 1995a), informed the negotiations leading to the Kyoto Protocol in 1997. The Third Assessment Report, or TAR (IPCC, 2001), and the Fourth Assessment Report, or AR4 (IPCC, 2007), informed the process leading up to the 2009 United Nations Climate Change Conference in Copenhagen (COP-15), which was intended to create an extended or new regime in anticipation of the 2012 expiration of the first commitment period of the Kyoto Protocol. Each Assessment Report consists of three volumes, from Working Group I on the science of climate change, Working Group II on impacts and adaptation, and Working Group III on mitigation.

IPCC has also produced a series of Special Reports, including the Special Report on Emissions Scenarios (SRES) in 2000 (IPCC, 2000a), the Special Report on Methodological and Technological Issues in Technology Transfer, also in 2000 (IPCC, 2000b), and the Special Report on Carbon Dioxide Capture and Storage (SRCCS) in 2005 (IPCC, 2005).

Each of the IPCC Reports is peer-reviewed and assesses a vast number of scientific publications, and is the most authoritative assessment available. IPCC has recently been subject to criticism due to a few mistakes in the AR4. These mistakes have not affected the overall conclusions or their soundness.

How climate will change in the future under anthropogenic pressures will depend to a large extent on future GHG emissions, changes to the biosphere, and feedbacks in the Earth system. Even if emissions of all anthropogenic RF agents were to remain constant at today's levels, the Earth's climate system would continue to change. This is often referred to as 'committed warming,' and is largely due to the thermal inertia of the oceans and ice sheets and their long time-scales for adjustment. For example, the IPCC (2007a) estimates that committed climate change due to atmospheric composition in the year 2000 corresponds to a warming trend of about 0.1°C per decade over the next two decades, in the absence of large changes in volcanic or solar forcing. By 2050, about a quarter of the 1.3–1.7°C warming relative to 1980–1999 estimated using Special Report on Emission Scenerios, often referred to as the IPCC SRES 'marker Scenarios' (Nakicenovic et al., 2000; see also Box 3.2), would be due to committed climate change if all RF agents were to be stabilized at today's concentration levels.

It is extremely unlikely that RF agents will be held constant, as evidenced by the continued rise in GHG emissions. In order to assess the likely future trends in our climate, the IPCC (2007a) has assessed global climate change projected from six SRES scenarios of emissions of RF agents (see Figure 3.3). These scenarios represent a range of plausible future trajectories of population, economic growth, and technology change, in the absence of policies to specifically reduce emissions in order to address climate change. The assessment of climate change under these scenarios was made using a number of climate models of varying levels of complexity (and hence capable of incorporating different aspects of climate sensitivity; see Box 3.1), from simple climate models to those that include ocean-atmosphere general circulation models and feedbacks between climate change and the carbon cycle (Betts et al., 2011). The IPCC (2007a) concluded that GMTs are likely to increase by between 1.1–6.4°C by the end of the 21st century relative to the 1980–1999 average. A key question is, What are these projected increases in GMT likely to mean for impacts associated with climate change? A related question is, How is this likely to affect our ability to stabilize GHG concentrations so as to "*prevent dangerous anthropogenic interference of the climate system*," as referred to in Article 2 of the United Nations Framework Convention on Climate Change (UNFCCC, 1992)? One of the core objectives of GEA is to answer these questions by assessing the implied constraints on future energy-related emissions of GHGs that would fulfill the stated objective of the UNFCCC (1992).

3.2.2.2 Impacts of Climate Change in the Future

How will climate change influence Earth systems, and what risks are involved? This has been continually assessed in the four IPCC Assessment Reports (IPCC, 1991a; 1995b; 2001a; 2007a) (see Box 3.2). Increases in GMT of the magnitude projected for 2100 as described in Figure 3.3 would be expected to have substantial global consequences both for near-term climate change and throughout the 21st century. Such consequences would include continued sea-level rise, changes in the cryosphere, decreases in snow cover, changes in global and regional patterns of temperature and precipitation, changes in extreme weather events such as heat-waves and drought, changes in the number and intensity of tropical cyclones, loss of genetic species and ecosystem

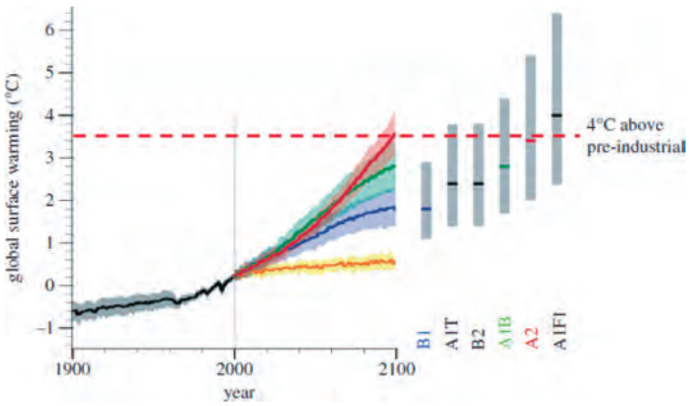

Figure 3.3 | Past changes in global mean surface temperature (GMT) (black curve), and projected future changes resulting from the IPCC SRES (Nakicenovic et al., 2000) 'marker scenarios' of GHG and aerosol emissions (colored curves and gray bars), relative to the 1980–1999 mean (Meehl et al., 2007). Climate changes under the A2, A1B, and B1 scenarios were projected with general circulation models (red, green, and blue lines, with plumes showing 5–95% range of model projections without uncertainties in climate-carbon cycle feedbacks). The full set of 'marker scenarios,' including a range of strengths of climate-carbon cycle feedbacks, were examined with simple climate models. Gray bars show the likely range of warming at 2090–2099 for each scenario, from expert assessments based on all available evidence from general circulation models, simple climate models, and observational constraints. The red dashed line marks warming of 3.5°C relative to 1980–1999, which represents 4°C relative to pre-industrial levels. Red line, A2; green line, A1B; blue line, B1; orange line, year 2000 constant concentrations; black line, 20th century. Source: IPCC, 2007a.

diversity, acidification of the oceans, and, perhaps most importantly, dangers of crossing tipping points that could lead to catastrophic ecological consequences (Schellnhuber et al., 2009). The uncertainties in assessments of these projections in climate change are many, and the IPCC process summarizes these uncertainties in its reports.

The IPCC (2001b) summarized an extensive analysis of the impacts of climate change. The IPCC identified 'reasons for concern' (RFCs), describing them in what has since become known as the 'burning embers' diagram. The diagram aimed to characterize the extent of the level of threat or risk associated with future projected anthropogenic climate change, defined as a change from 1990 levels of GMT.

The 'burning embers' diagram (IPCC, 2001b), as described in Smith et al. (2001), is shown in Figure 3.4, together with data updated by Smith et al. (2009). The comparison of the two diagrams suggests that the temperature range from which a consensus definition of dangerous anthropogenic interference might be drawn is getting lower, a result of advancements in our scientific insights regarding the functioning of the Earth system. There has been a growing consensus toward adopting a '2°C guardrail' approach, shown as a black dashed line in Figure 3.4 (Hare and Meinshausen, 2006). This approach has been adopted by the 'Copenhagen Accord' and the European Commission (United Nations Conference of the Parties, 2009). The 2°C barrier is based on recommendations by numerous scientific studies (Schneider and Mastrandrea, 2005; Fisher et al., 2007; Nakicenovic and Riahi, 2007; Hansen et al., 2008; Schellnhuber, 2008; Kriegler et al., 2009; Meinshausen et al., 2009; Rockström et al., 2009), which suggest that global warming in excess of 2°C from pre-industrial times could trigger several climate tipping elements and lead to unmanageable changes (Smith et al., 2009). This target represents a clear guiding principle for acceptable limits of climate change. However, it has been recognized that establishing how the target will be met is rather complicated, since uncertainties associated with our knowledge of climate sensitivity, particularly the carbon cycle and climate response (see Box 3.1), complicate efforts to estimate the GHG emission reductions that would be necessary to remain below this warming target. Figure 3.4 indicates that even if this rather ambitious target is met, three out of the five 'reasons for concern' would still be at high risk of manifestation.

It is also worth emphasizing that significant risks of adverse climate impacts for society and the environment will have to be faced even if the 2°C line can be held (see also IPCC, 2007b; Richardson et al., 2009; WGBU, 2009). In view of this fact, it is possible that the 2°C barrier will be revised to lower values; efforts to make the target more stringent may be renewed as our understanding of regional consequences of climate change improves (Schneider and Mastrandrea, 2005; Hansen et al., 2008; Kriegler et al., 2009; Rockström et al., 2009).

3.2.2.3 Emission Scenario Requirements to Remain Below the 2°C 'Guardrail'

The UNFCCC Conference of the Parties (COP) in Copenhagen (COP 15) in December 2009 and the UNFCCC COP 16 in Cancun in December 2010 did not arrive at a legally binding agreement on how to proceed after the first commitment period of the Kyoto Protocol ends in 2012. However, the three-page "Copenhagen Accord," which was offered by a subgroup of Parties and taken note of by the COP, provides a consensus, however limited, on defining a 2°C GMT increase as a global 'guardrail' for human-induced climate change. This is the nearest expression of how to interpret dangerous climate change and of the level of mitigation desired. It is therefore taken as the normative goal for energy systems development and used to define the global sustainability criteria used in this GEA assessment (see Chapter 17).

The IPCC (Fischer et al., 2007) addressed the question of what the GHG emission reductions might need to be in order to provide a chance of stabilizing GMT below the 2°C 'guardrail.' The left-hand graph in Figure 3.5 shows the emission paths that are consistent with various stabilization levels, and the right-hand graph indicates that staying below a 2ºC 'guardrail' with a 50% probability would require long-term GHG stabilization at around 440–450 ppm CO_2-eq. Figure 3.5 also indicates the uncertainties in climate-sensitivity estimates. For example, to increase the probability to around 90% would require stabilization below 400 ppm CO_2-eq, or essentially the maintenance of current concentrations throughout the century. Basically, global emissions need to decline almost immediately (within the next decade) to keep the goal of stabilizing at 2°C within reach. The higher the 'overshoot' of emissions,

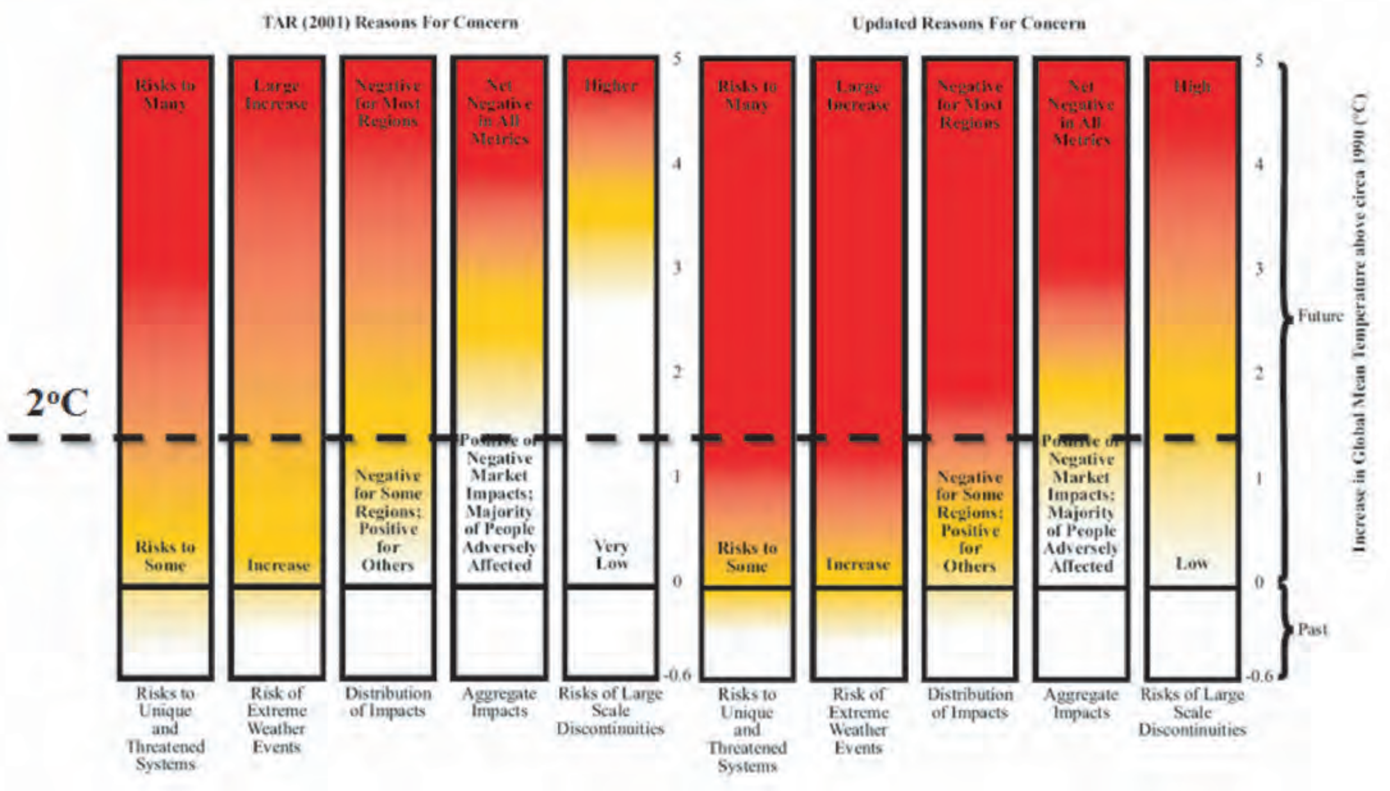

Figure 3.4 | Risks from climate change, by 'reason for concern' (Smith et al., 2001), compared with updated data (Smith et al., 2009). Climate change consequences are plotted against increases in GMT after 1990. The pre-industrial temperature level is also indicated. Each column corresponds to a specific reason for concern and represents additional outcomes associated with increasing GMT. The color scheme represents progressively increasing levels of risk. Both figures suggest that all stabilization levels, including the current atmospheric concentrations of GHGs, can be considered to be in principle dangerous, but it is important to note that the level of concern increases significantly with higher stabilization levels. Source: Smith et al., 2009.

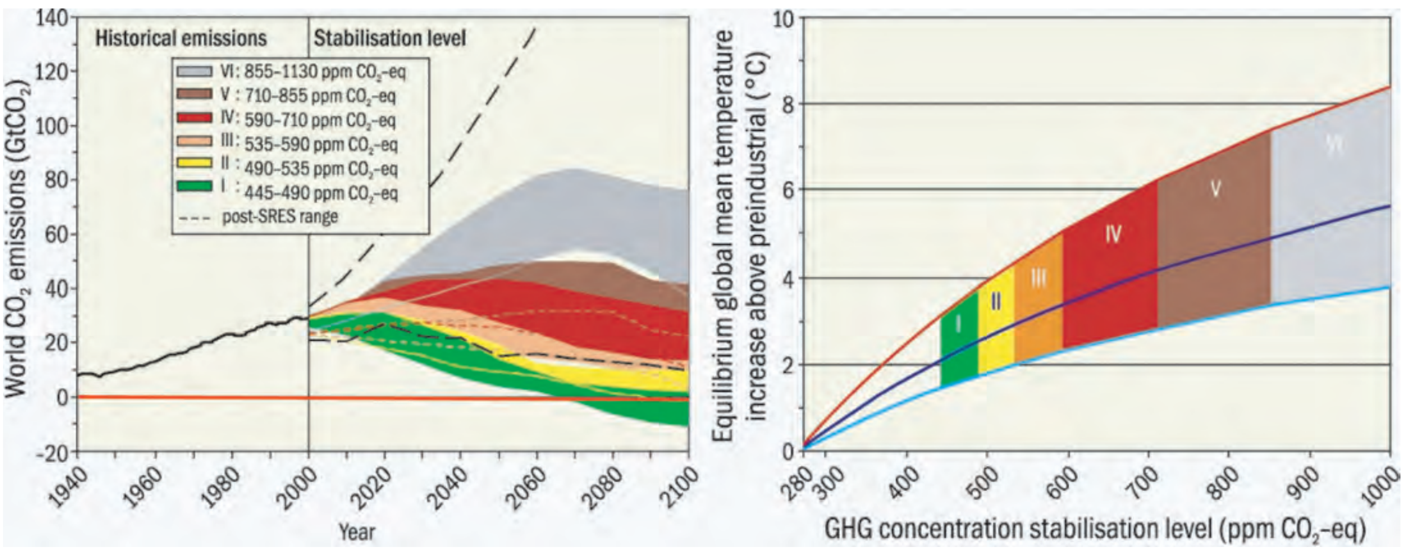

Figure 3.5 | The graph on the left shows the ranges of future emissions pathways for reaching different GHG emissions concentrations, expressed in terms of $GtCO_2$-eq. For example, the green range indicates the emissions trajectories that would lead to stabilization levels between 445–490 ppm $GtCO_2$-eq (as shown in the box within the figure). Note the need for net negative emissions post-2060 in case of the most stringent (green) trajectory. The graph on the right shows the equilibrium GMT increase above pre-industrial levels as a function of GHG stabilization level (ppm CO_2-eq). The middle black line indicates the most likely climate sensitivity, which is the most likely temperature increase at a certain GHG stabilization level. The red and blue lines indicate higher and lower climate sensitivity, that is higher or lower temperature increases for a given GHG stabilization level. Source: IPCC, 2007b.

the steeper the decline needs to be thereafter. At the moment, global GHG emissions are continuing to increase at close to historical rates without a sign of a reversal. A possible decline is being pushed further and further into the future with the recent failure to reach a 'global deal' in Copenhagen during the recent UNFCCC COP 15 (United Nations Conference of the Parties, 2009).

The temperature increase is in the first approximation a function of cumulative emissions. So far, humanity has emitted about 1000 $GtCO_2$-eq into the atmosphere, which has resulted in CO_2 concentrations increasing from about 280 to some 400 ppm today. In the case of lower stabilization levels, the remaining emissions 'endowment' is smaller than the cumulative historical emissions. Meinshausen et al. (2009) estimate that future cumulative emissions will be substantially lower than 1000 $GtCO_2$-eq. The exact amount will depend on the climate sensitivity to future emissions (which is not known with complete certainty) and the desired likelihood of not exceeding a particular stabilization level, say of 2°C. According to the German Advisory Council on Global Change (2009), the cumulative CO_2 emissions from 2010 to 2050 must not exceed 750 $GtCO_2$-eq in order to stay below a 2°C temperature increase with 67% probability. This assumes that there will not be any 'negative' emissions after 2050 to offset the excess emissions of the next several decades. Most of the 2°C stabilization scenarios do actually assume the possibility of negative emissions in the second half of the century (e.g., Fujino et al., 2006, Riahi et al., 2007; Van Vuuren et al., 2007; Wise et al., 2009).

The necessity of adopting such stringent emission reductions is evident, given the facts that a growing body of evidence suggests that the climate is changing more quickly than previously projected by the IPCC Assessment Reports (Jackson, 2009); that substantial climate impacts are occurring at lower GMTs (Smith et al., 2009); and that temperature changes may well be greater during this century than had been previously projected (Sokolov et al., 2009).

3.2.2.4 Near-term Changes of Radiative Forcing

Recent scientific studies indicate that short-term changes of RF play a significant role in climate change. For example, forcing of BC, a short-term radiative forcer, has been estimated to be 20–50% of CO_2 forcing, making it the second or third largest contributor to global warming (Wallack and Ramanathan, 2009). Black carbon and other short-term radiative forcers (e.g., O_3) will enhance warming, and therefore their mitigation would help prevent climate change. In the atmosphere, these short-term radiative forcers often co-occur with other short-lived pollutants such as sulphates, nitrates, OC, and other aerosols. These pollutants cool the climate through scattering and reflection of incoming solar radiation, and hence their mitigation would actually lead to a warming of the climate. These mixtures of anthropogenic particles and gases are sometimes referred to as atmospheric brown clouds (ABCs), especially when they occur in regions that particularly suffer from visible pollution. Such pollution may, for example, result from enhanced biomass burning, such as that occurring in south and southeast Asia (see also Box 3.3 and Ramanathan and Feng, 2008).

The realization of the substantial effect that these short-lived forcers can have on climate has led to a growing consensus regarding the need not only to mitigate those atmospheric agents responsible for long-term climate change, but also to manage the magnitude and rate of change of emissions of near-term radiative forcers and hence their RF (Ramanathan and Xu, 2010). Such mitigation of near-term climate change involves different pollutants, which often arise from different

Box 3.3 | Atmospheric Brown Clouds (ABCs)

What are ABCs

Atmospheric brown clouds (ABCs) are regional scale plumes of air pollution that consist of copious amounts of aerosols (tiny particles of BC, OC, sulphates, nitrates, fly ash) as well as many other pollutants including tropospheric O_3. The brownish color of ABCs is due to the absorption and scattering of solar radiation by BC, OC, fly ash, soil dust particles, and NO_2 gas. Typical background concentrations of aerosols are usually in the range of 100–300 particles/cm^3, in polluted continental regions suffering ABCs, aerosol concentrations are in the range of 1000–10,000 particles/cm^3.

ABCs start as indoor and outdoor air pollution consisting of particles (referred to as primary aerosols) and pollutant gases, such as NO_x, CO, SO_2, NH_3, and hundreds of organic gases and acids. These pollutants are emitted from anthropogenic sources, such as fossil fuel combustion, biofuel cooking, and biomass burning. Gases, such as NO_x, CO, and many VOCs, are important precursors of O_3 which is both an air pollutant and a strong GHG. Gases such as SO_2, NH_3, NOx, and referred to as aerosol precursor gases. These gases – over a period of a day or more – are converted to aerosols through the so-called gas-to-particle conversion process. Aerosols that are formed from gases through chemical changes (oxidation) in the air are referred to as secondary aerosols (Ramanathan et al., 2008).

Impacts of ABCs

Radiative forcing: Some components of ABCs, such as sulphate and nitrate aerosols, have a cooling effect on the climate system through reflection and scattering of incoming solar radiation. Others, such as BC, have a warming effect through absorption of solar radiation, which can lead to warming of the atmosphere or, where the BC is deposited on reflective snow- and ice-covered surfaces, can lead to surface warming and melting with implications for hydrological flows.

Glacial melting: ABCs solar heating (by BC) of the atmosphere is suggested to be as important as GHG warming in accounting for the anomalously larger warming trend observed in the elevated regions. In addition to the heating effect, deposition of BC on snow or ice can reduce the surface albedo and accelerate melting. Scientific studies suggest that ABC is one of the major contributing factors in glacier and sea-ice melting.

Water budget: ABCs change the cloud properties (cloud droplet numbers, size, albedo) and produce brighter clouds that are less efficient at releasing precipitation. Together, these effects can cause localised dimming (reduction of solar radiation reaching the Earth's surface) and lead to a alterations of the hydrological cycle.'

Human health: A large fraction of the aerosol particles that make up ABCs originate from emissions at the Earth's surface caused by the incomplete combustion of fossil fuels and biofuels. Humans are exposed to these particles both indoors and outdoors. Available information about the adverse health effects of airborne fine particles from studies conducted in many areas of the world suggests that ABC exposure is very likely associated with significant adverse health effects.

In summary, ABCs cause perturbations to regional climates, due to their comprising RF species. They also affect human health and agricultural productivity directly, through impacts resulting from the air pollution component (aerosols and ground-level O_3) of ABCs but also (particularly in the case of hydrology and agriculture) indirectly through their mediation of local climate. ABCs therefore represent a striking example of the interactions between climate change and air pollution, not only in relation to commonality in the atmospheric species causing both these environmental problems, but also in relation to the processes by which impacts are propagated. Hence, important lessons can be learned in relation to understanding, with a view to ultimately controlling, the adverse impacts associated with ABCs.

source activities compared with their long-term counterparts. This situation was eloquently described by Jackson (2009), who explored the contributions from near- and long-term climate forcers to climate change over a 20-year time frame, showing the relative contributions to RF from past emissions and from a variety of different pollutants (Figure 3.6). These pollutants are all sourced, albeit to varying extents, through energy-related processes.

Figure 3.6 highlights two important issues. The first is that positive RF resulting from the next 20 years of unrestrained human activity would exceed positive RF remaining from historical human activity after a couple of decades. The second issue is that short-lived pollutants (in particular BC, O_3, and CH_4) account for more than half (57–60%) of the positive RF generated in years 1 to 20.

Jackson also identified the 'top 10' pollutant-generating activities contributing to net RF, taking into account multiple pollutants from each source activity (Figure 3.7, see also Koch et al., 2007). From this information it is argued that the seven sources that appear on the left side (purple bars) would be overlooked by mitigation strategies focusing exclusively on long-lived pollutants. There is therefore an urgent need for integrated mitigation strategies that include both the long- and short-term changes of RF; of these, gas and coal production and residential biofuel combustion are the categories which Jackson believed could be addressed by changes in future energy use and supply.

Raes and Seinfeld (2009) describe the current policy conundrum associated with these short-lived climate forcers. This relates to the fact that in addition to being radiative forcers, these species also play a role in air pollution, causing impacts on both human health (see Chapter 4) and ecosystems (see Section 3.2.3). To prevent and control these air pollutants, particularly in relation to human health, policies are already in place to reduce some of these pollutants (especially those classified as PM which include BC, OC, and other aerosols). These policies do not distinguish between positive or negative radiative forcers. Therefore, they may not improve the situation for climate change, or even actually enhance RF by reducing atmospheric concentrations of the negative forcers more effectively than the positive forcing species. The reality of this situation has been recently investigated by Penner et al. (2010), who argue that the short-term climate forcers need to be brought under

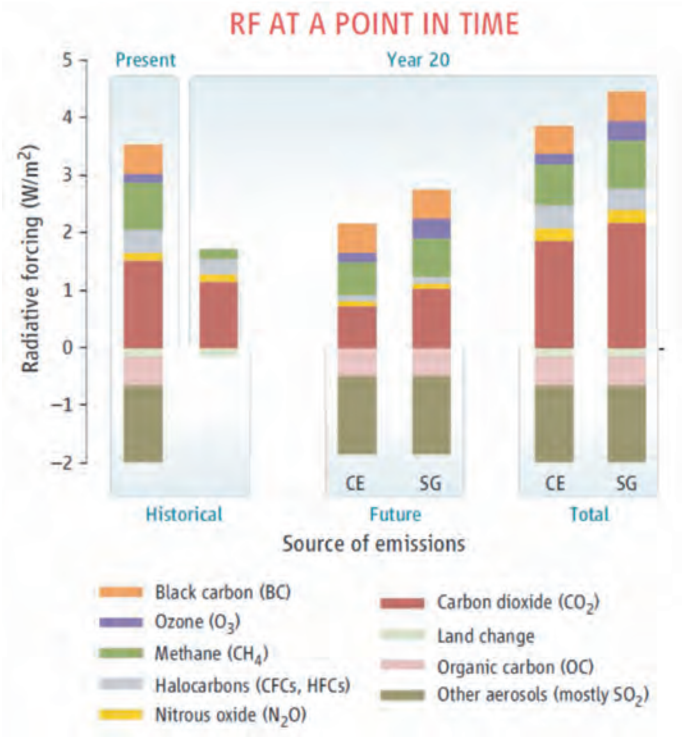

Figure 3.6 | Global radiative forcing (RF). The left-most bar shows RF attributable to historical human emissions (1750–2000), with the next bar representing historical RF that would remain after 20 years of atmospheric decay with zero additional human emissions. The next two bars represent 'future' RF in year 2020 resulting from human emissions. The two scenarios depicted are: emissions remain constant at year 2000 levels (CE), or emissions grow steadily at current rates (SG). The right-most columns show total RF experienced in year 2020 (historical + future emissions), again for both scenarios. For further details, see Jackson, 2009.

control within a few decades, and that the effect of this control on atmospheric composition and climate sensitivity needs to be monitored to provide an understanding of the warming and cooling contributions from CO_2 and short-lived air pollutants. Only with this information will it be possible to identify mitigation options that will afford the largest benefits in the alleviation of climate change while also addressing impacts on human health and ecosystems; a first attempt at identifying such options on a global scale has been achieved by a UNEP/WMO commissioned assessment on Black Carbon and Ozone (UNEP/WMO, 2011). This study found that considerable human health and crop productivity benefits could be realized through a number of technical and non-technical measures to limit emissions of BC and O_3 precursors and that these measures also provide a chance of constraining temperature increases below the 2°C, and even the 1.5°C, threshold if implemented in the very near future. This study helps to highlight that, especially in relation to energy systems, the most obvious policy option would seem to be identifying mitigation measures that would reduce both CO_2 and short-lived climate forcers that would otherwise lead to a warming of the climate systems.

3.2.3 Air Pollution and Energy Systems

As described previously, GHGs and RF agents that are produced from our energy systems can cause impacts other than climate change. Atmospheric emissions of SO_2, NO_x, and O_3 precursors are associated with eutrophication, acidification, and other types of direct ecosystem damage (Fowler et al., 2009). Much of the information describing these various air pollution impacts provided in the following sections is set within the context of an effects-based concept developed by the United Nations Economic Commission for Europe (UNECE) Convention

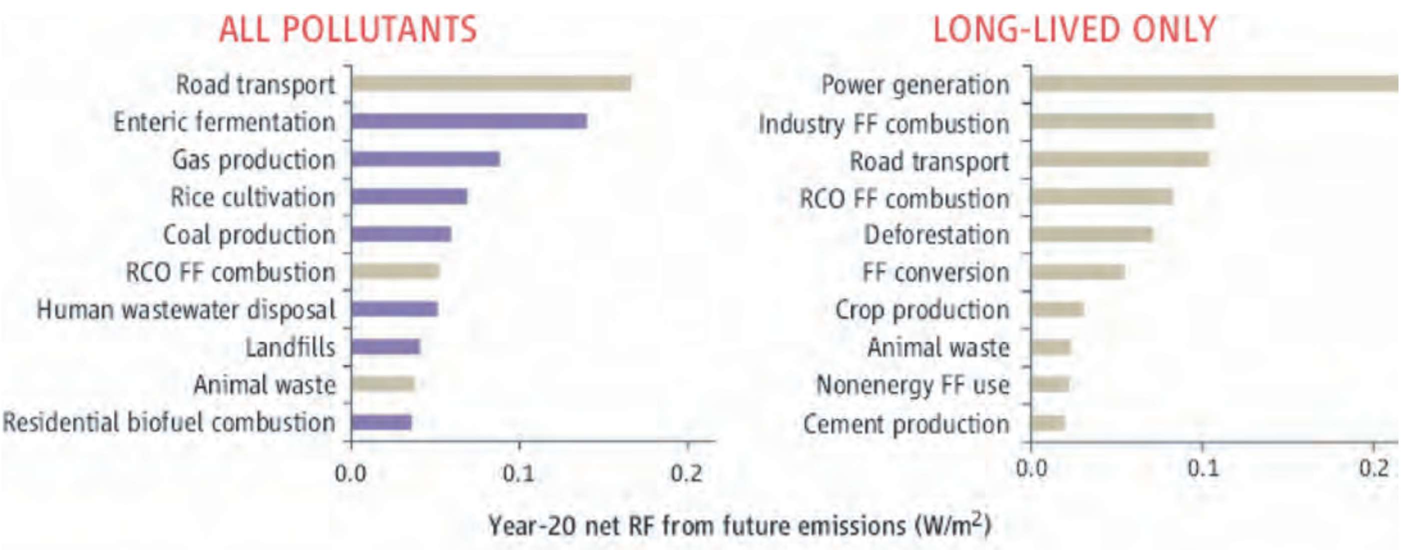

Figure 3.7 | Top-10 global sources of year 20 net radiative forcing (RF). Note: Long-lived pollutants (CO_2, N_2O) have only positive RF, whereas pollutants that are not long-lived have both positive RF (BC, O_3) and negative RF (OC, SO_2). Hence, a source may show a different RF on the left-hand versus the right-hand graph. See Jackson (2009) for further details.

on Long-Range Transboundary Air Pollutants (LRTAP). This approach has been successfully applied at the regional level in Europe to set critical limits below which ecosystems are unaffected by air pollution according to current knowledge (see Sliggers and Kakebeeke, 2004). These limits, which are referred to as critical loads and levels, have been established for different pollutants and impacts and have been used to define national and international air-quality guidelines (e.g., WHO, 2006a). They have also been used in European regional integrated assessment models to optimize air pollution mitigation policies, so that maximum benefits can be achieved at minimum cost (Schopp et al., 1999).

The following sections describe the processes by which these air pollutants impact ecosystems. The focus is on impacts by pollutants acting individually. However it is recognized, though poorly understood, that in reality these pollutants act together. For example, the potential for these pollutants to cause feedbacks on climate systems through perturbations to the terrestrial carbon sink strength has been identified as a major uncertainty which affects our ability to quantify the carbon cycle. Understanding interactions between pollutants is especially important here, given that impacts can both increase or decrease carbon sequestration.

3.2.3.1 Acidification of Soils and Freshwater

It is well documented in Europe and North America that sulphur(S) and nitrogen (N) deposition, predominantly from fossil fuel use, have caused widespread acidification of terrestrial and aquatic ecosystems. Acidic deposition has been linked to serious losses of fish stocks and other sensitive aquatic species, and it has been implicated as a potential cause of symptoms of forest decline (Rodhe et al., 1995; Menz and Seip, 2004) and effects on biodiversity (Bobbink et al., 1998). As emissions and deposition of S and N compounds increase in other parts of the world, particularly in Asia (see Section 3.2.4.4), there is a risk that acidification problems will become more widespread. However, the type and severity of acidification problems depend on many environmental factors, and evidence to date suggests that acidification problems in other parts of the world may not be as serious as they were in Europe (Hicks et al., 2008). It is also known that the reduction of atmospheric emissions of S and N to safe levels will not result in the immediate recovery of impacted ecosystems (e.g., Bishop and Hultberg, 1995), though there is evidence of recovery in European ecosystems (e.g., Vanguelova et al., 2010), where acidic deposition has been reduced significantly since the 1980s (Nakicenovic and Riahi, 2007; Vestreng et al., 2007).

3.2.3.2 Eutrophication

Increased release of reactive nitrogen (Nr) to the environment resulting from energy use is due mainly to increased NO_x emissions from fossil-fuel combustion processes that provide energy for transport, power generation, and industry. However, in terms of the total amount of Nr emitted to the atmosphere, an equal or even greater amount is emitted from agricultural sources in the form of ammonia. Furthermore, Nr losses to ground and surface waters, mainly as nitrates, are dominated by agricultural activity; for example, in Europe, agriculture contributes 60% of these ground and surface water Nr losses, with the remainder largely made up of discharges from sewage and waste water treatment systems. As such, although the atmospheric emission of NO_x from fossil fuel combustion contributes to the eutrophication effects described below, these emissions can be dwarfed by agricultural flows depending on the location of the sensitive systems (Galloway et al., 2008, ENA, 2011).

Nitrogen is an essential nutrient that limits growth in many ecosystems; however, when applied in excess, it can cause eutrophication of terrestrial and aquatic ecosystems. Eutrophication can result in excessive plant growth such as algal blooms in aquatic systems. It can also cause changes in biodiversity. The eutrophying effect of N pollution at the global level is of particular concern (Gruber and Galloway, 2008). Molecular nitrogen (N_2) in the atmosphere is not usable by most organisms; it is only when it is fixed into reactive compounds (i.e., Nr) that it causes environmental impacts. Before the 20th century, the fixation of Nr occurred predominantly via a limited group of microorganisms and by lightning. This was sufficient until the demand for food, driven by rapid population growth, led to new ways of converting nonreactive gaseous N_2 into reactive forms for agricultural purposes, mainly through industrial production of fertilizers. Such Nr fixation requires substantial amounts of energy to break the strong triple bond of N_2; hence, increased anthropogenic Nr fixation became possible through increased access to fossil fuels. This has led to an increase in the use of inorganic fertilizers in agriculture and associated increases in Nr leaching and eutrophication when fertilizer is applied in excess of agricultural system requirements (Smith et al., 1997). A recent study on energy use in the fertilizer industry also showed that despite significant energy-efficiency improvements in fertilizer manufacture, these improvements have not been sufficient to offset growing energy demand, due to rising fertilizer consumption (Ramírez and Worrell, 2006).

At the global level, current Nr emission scenarios project that most regions will have increased rates of atmospheric N deposition by 2030 (Dentener et al., 2006), which may cause significant impacts to global plant biodiversity in sensitive ecosystems (Vitousek et al., 1997; Sala et al., 2000; Phoenix et al., 2006). A significant proportion of these emissions will be related to air pollution from energy sources (Galloway et al., 2008). In addition to the acidification effects of atmospheric Nr loading described above, deposits of Nr compounds act as a fertilizer to increase the productivity of terrestrial and aquatic ecosystems. However, if the supply of Nr continues to increase, a complex series of alterations to soil and biogeochemistry may affect productivity, competition, and microbial community structure (Bobbink et al., 2010). Many of the European arctic, boreal, and temperate terrestrial ecosystems have already been allocated effect thresholds or empirical critical N loads under the LRTAP Convention in the UNECE region, but there is a lack of information on impacts in other parts of the world. Bobbink et al. (2010) conclude that reductions in plant diversity as a result of increased atmospheric N deposition may be more widespread than first thought. They show that vulnerable regions outside Europe and North America include eastern and southern Asia (China, India), an important part of the Mediterranean ecosystems (California, southern Europe), and several subtropical and tropical parts of Latin America and Africa. However, to date the effects of N deposition on biodiversity are mostly only quantified for plant richness and diversity, and the impacts on animals and other groups are barely studied. Freshwater ecosystems are also affected, for example, across most of Europe nitrate levels in freshwaters greatly exceed a threshold above which water bodies may suffer biodiversity loss (ENA, 2011).

Nitrogen pollution is now considered to be the biggest pollution problem in coastal waters (Howarth et al., 2000; NRC, 2000; Rabalais, 2002), due to algal blooms and hypoxia (Levin et al., 2009) predominantly resulting from Nr fertilizer runoff. Human activities have severely altered many coastal ecosystems by increasing the input of anthropogenic Nr through sources such as rivers and groundwater, direct discharges from wastewater treatment, and atmospheric deposition, resulting in increasing eutrophication (Duce et al. 2008). In addition to runoff from land, atmospheric anthropogenic fixed Nr can enter the open ocean. Duce et al. (2008) found that this could account for approximately one third of the ocean's external (non-recycled) Nr supply and up to 3% of the annual new marine biological production.

In addition to impacts on biodiversity, N deposition and eutrophication in general will also have significant impacts on ecosystem services. Potential impacts include human health impacts of high nitrate levels in drinking water, carbon sequestration, GHG fluxes from soils, pollination, and cultural aspects, such as the loss of treasured species (MEA, 2005; GBO-3, 2010; NEA, 2011). Studies on the significance of these aspects are only just beginning in earnest around the world.

3.2.3.3 Vegetation Damage from Ground-level Ozone (O_3)

Tropospheric O_3 is a naturally occurring atmospheric trace gas. Historically, pre-industrial levels of the pollutant were in the region of 15–20 parts per billion (ppb) (Vingarzan et al., 2004). However, concentrations increased following the advent of industrialization and the burning of fossil fuels for transport, industry, and power generation. This increase was largely the result of increasing NO_x emissions, coupled with increases in NMVOC emissions from industry; these precursor pollutants combine under the action of sunlight through photochemically driven processes to form O_3. Heavily polluted regions can experience frequent occurrences of O_3 concentrations of approximately 40–50 ppb, especially during the summer periods, when the photochemical activity that drives the O_3 formation reactions is high (Royal Society, 2008).

Ozone concentrations vary locally, regionally, and seasonally across the globe (see Figure 3.8). Ozone concentrations can be high in both urban and rural locations (downwind from precursor pollutant sources).

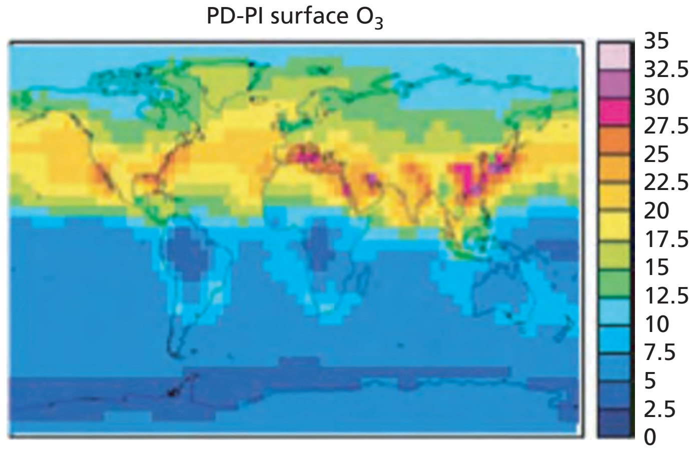

Figure 3.8 | Modelled global increase in surface ozone (O_3) concentrations between pre-industrial times and the present day (PD-PI). Source: Royal Society, 2008.

However, in urban areas, episodes tend to be short-lived, as O_3 reacts with other chemical pollutants, namely nitric oxide (NO). In contrast, rural episodes of elevated O_3 tend to have an enhanced longevity, as the pollutants that destroy O_3 are not so prevalent. This makes O_3 an extremely important rural pollutant, which is heightened by the fact that it is strongly phytotoxic and capable of causing a wide variety of damage to various ecosystems; most notable are decreases in agricultural crop yields (Fuhrer 2003; Fuhrer and Booker, 2003), reductions in forest biomass (Matyssek and Sandermann, 2003), and changes in species composition of seminatural vegetation communities (Davison and Barnes, 1998; Ashmore, 2005). Ground-level O_3 also impacts human health through damage to the respiratory system, with reduced lung function and lung irritation among the most commonly experienced respiratory impairments; these issues are discussed in more detail in Chapter 4.

Although tropospheric O_3 is a relatively short-lived pollutant, with an average tropospheric lifetime of 22 (±2) days (Stevenson et al., 2006), attention has been drawn recently to the phenomenon of hemispheric transport of this air pollutant. This phenomenon is important, since it means that geographical regions (e.g., Europe, Asia, North America) are not in complete control of the air pollution load they experience (UNECE Task Force on Hemispheric Transport of Air Pollution, 2007). This concern is a result of the complex atmospheric chemistry associated with O_3, which can result in formation from precursors long after they have been emitted (Stevenson et al., 2006).

The changes in mean global O_3 concentration between pre-industrial times to the present day, shown in Figure 3.8, emphasise the historical importance of fossil fuel-based energy use and supply as the major source of anthropogenic O_3 precursor emissions, both in industrialized and developing countries. This is reflected in the distribution of the global 'hotspots' of elevated O_3 concentrations (in the United States, Europe, south asia, and east Asia) in regions that have experienced rapid industrialization over the past 200 or so years.

3.2.3.4 Combined Effects of Air Pollutants

Since fossil fuel combustion is an important source of N and S pollution and O_3 precursors, these chemicals will have a tendency to occur as a mix of pollutants in the atmosphere. This has consequences for impacts on receptor ecosystems, since damage by a single pollutant can be altered by the presence of other pollutants, frequently with synergistic (i.e., more than additive) effects. However, calculating response functions for each pollutant in the presence of others is extremely complex (Bell, 1985; Bender and Weigel, 1993). Additionally, the impacts on net primary productivity caused by these pollutants will also affect terrestrial carbon sequestration and hence create feedbacks to climate change.

Perhaps the best-studied of such feedbacks are climate change interactions with the nitrogen cycle. There is growing evidence that the increase in N deposition since industrialization may have been responsible for maintaining at least part of the current terrestrial carbon sink (Magnani et al., 2007; de Vries et al., 2008; Reay et al., 2008; Janssen and Luyssaert, 2009). The natural nitrogen cycle may also have been accelerated by climate change thereby increasing N availability to ecosystems. Nitrogen deposition can also lead to increased emissions of the potent GHG N_2O from soils and can increase soil emissions of NO, one of the important chemical precursors for O_3 formation (Prather et al., 1995). Nitrogen deposition can also enhance the growth rates of Nr-limited forests (Hungate et al., 2009), resulting in enhanced uptake/sequestration of carbon in terrestrial ecosystems where Nr is the limiting nutrient. As atmospheric CO_2 concentrations increase, such Nr limitation may become more common. However, where N deposition exceeds critical loads, adverse effects on growth and carbon sequestration can occur.

There is some evidence that a long-term trend of increased productivity in European forests is associated with such environmental factors, that including N deposition, as well as increased CO_2, and climate change, (Nabuurs et al., 2003), and is not simply due to improved management as is sometimes suggested. These factors can also ameliorate the resilience of trees to O_3. However, O_3 itself is considered a factor potentially capable of reducing the 'benefits' of CO_2 and nitrogen fertilization (King et al., 2005; Magnani et al., 2007). Seminatural grasslands are often limited by nutrients such as Nr or phosphorous. Alleviating such constraints, for instance by the addition of Nr, could decrease the sensitivity of the plant community to O_3 through increasing biochemical detoxification capacity, or increase the sensitivity though increased stomatal conductance. However, to date, evidence of these effects is limited and contradictory. Research into the nitrogen cycle is relatively less well developed than for the carbon cycle. Several authors have evaluated the effect of including C-N coupling in carbon and or climate models (Sokolov et al. 2008; Xu-Ri and Prentice, 2008). These studies suggest that the likelihood of greatly enhanced global CO_2 sequestration resulting from future changes in N deposition is low (Dolman et al., 2010).

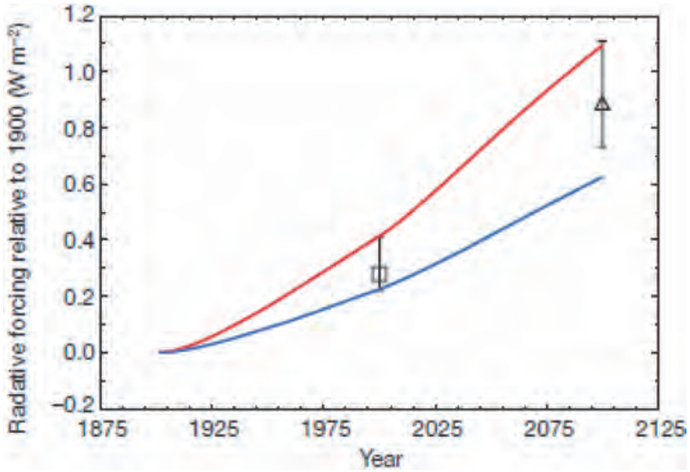

Figure 3.9 | Temporal changes in indirect radiative forcing (RF) due to O_3 for 'high' (red) and 'low' (blue) plant sensitivities to O_3. These results are diagnosed from model simulations using a fixed pre-industrial CO_2 concentration. For comparison, estimates of the direct RF forcing due to O_3 increases are shown by the bars. For further details, see Sitch et al. (2007).

The impact of tropospheric O_3 on ecosystem net primary productivity has also recently been used to estimate the indirect RF that would occur through O_3-induced alterations to carbon sequestration (see Figure 3.9). These estimates suggest that the indirect forcing by O_3 by 2100 would be 0.62 W/m^2 and 1.09 W/m^2 for 'low' and 'high' plant O_3 sensitivity respectively. This compares with a mean direct RF, estimated from 11 atmospheric chemistry models, of 0.89 W/m^2, highlighting the extreme importance of air pollution-dependant ecosystem feedbacks in relation to our understanding of how climate forcing is likely to change in the future.

3.2.4 Regional Impacts of Climate Change and Air Pollution

Energy systems are a significant contributor to climate change and atmospheric pollution in most parts of the world. In this section, the current state of environmental impacts that are related to atmospheric pollution across a number of regions (Europe, the Americas, Asia, Africa, and the polar regions) are discussed in detail. Climate change-related impacts have already been reviewed in great detail in the IPCC 2007 assessment, therefore only a short summary of these impacts is provided here. A more detailed account of relevant air quality impacts is given, since these tend to be less considered compared to climate change impacts, though they cause serious environmental degradation on a regional scale. Here we limit the discussion to environmental impacts; other critical issues, such as health-related impacts of outdoor and household air pollution, are discussed in other chapters, including Chapter 4 and Chapter 17.

3.2.4.1 Europe

Europe has a history of environmental policies that have targeted air quality, dating back to the 1950s. Following the region's increase in economic wealth, industrial activity, and use of fossil fuels, emissions of harmful air pollutants increased significantly in Europe, from the beginning of industrialization to the 1980s (see, for example, historical SO_2 emissions in Figure 3.10). However, the long-term growth trend of emissions reversed after the 1980s, triggered by public concern about the detrimental impacts of urban air pollution on public health, the dieback of forests in Central Europe, and the disappearance of fish and aquatic life in Scandinavia (UNECE, 2004). In numerous international agreements for harmonized emission reductions under the LRTAP Convention and the European Union, countries have agreed to substantially reduce their emissions. As of 2010, emissions of SO_2 and NO_x had declined by 70% and 50%, respectively, since their peaks, due to widespread application of dedicated end-of-pipe pollution-control equipment, as well as improved energy intensities of the European economies and changes in the composition of fuel consumption.

Polices to reduce emissions of GHGs to prevent climate change have only been recently established, even though the European Union (EU) prides itself on having been a driving force behind the international negotiations that led to the establishment of the UNFCCC in 1992 and the Kyoto Protocol in 1997. For the latter, the 15 EU member states signed up to reduce emissions in the 2008–2012 period to 8% below 1990 levels, a target that currently looks achievable. European Union leaders have also endorsed an integrated approach to climate and energy policy to make a transition to an energy-efficient, low-carbon economy. They have made a unilateral commitment in the 'Climate and Energy package', adopted in 2008. By adopting this strategy, the 27 nations of the EU-27 (Austria, Belgium, Bulgaria, Cyprus, Czech Republic, Denmark, Estonia, Finland, France, Germany, Greece, Hungary, Ireland, Italy, Latvia, Lithuania, Luxemburg, Malta, the Netherlands, Poland, Portugal, Romania, Slovakia, Slovenia, Spain, Sweden, and the United Kingdom) pledged to reduce overall emissions to at least 20% below their 1990 levels by 2020. The EU pledges that this emission reduction would be increased to 30% by 2020, if other major emitting countries in the developed and developing worlds take similar action.

However, there is ample and robust scientific evidence that even at present rates, Europe's emissions to the atmosphere pose a significant threat to human health, ecosystems, and the global climate (UNECE, 2007). Discussed below are some of the main environmental impacts associated with climate change and air pollution in Europe.

Climate change environmental impacts in Europe

Wide-ranging climate change impacts have been observed in Europe through warming trends (Jones and Moberg, 2003) and spatially variable changes in

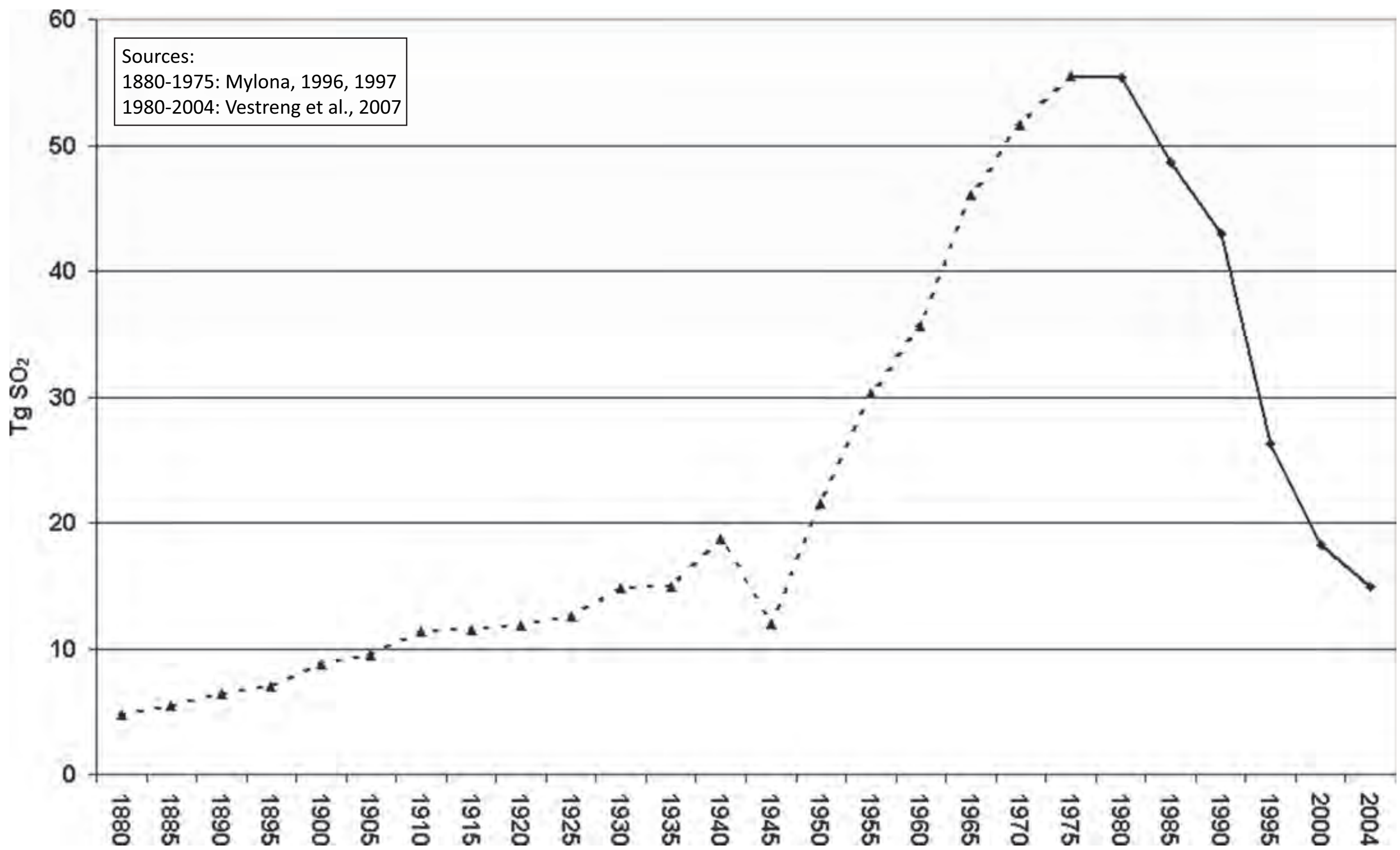

Figure 3.10 | Historical development of sulphur dioxide (SO_2) emissions in Europe (Unit: $MtSO_2$). Source: Mylona, 1996; 1998; Vestreng et al., 2007.

rainfall (Klein Tank et al., 2002). These impacts have affected the cryosphere, as seen in the retreat of glaciers (Hoelzl et al., 2003) and changes in the extent of permafrost (Frauenfeld et al., 2004). They have also impacted natural and managed ecosystems, as evidenced by changes in growing season length (Menzel et al., 2006) and species distribution (e.g., Walther et al., 2005), which have implications for biodiversity. Europe is considered most sensitive to climate change that causes extreme seasons, especially hot, dry summers and mild winters; the former would lead to more frequent and prolonged droughts (Schär et al., 2004) as well as to a longer fire season, especially in the Mediterranean (Moriondo et al.,2006). Climate-change projections indicate greater warming during winter in the north and during summer in the south. These projected changes have implications for crop suitability and production and for forest expansion and biomass growth, both of which are likely to increase in northern Europe and decrease in southern Europe (Olesen et al., 2007; Shiyatov et al., 2005; Metzger et al., 2004). The northward expansion of forests may reduce the extent of tundra regions (White et al., 2000). These changes may be accompanied by a shift in peak electricity demand from winter to summer, as demand for heating decreases and demand for cooling increases (Hanson et al., 2006). Water stress is projected to increase over central and southern Europe. This has important implications for energy supply, since the hydropower potential of Europe is expected to decline on average by 6% by the 2070s, and by 20–50% in the Mediterranean region (Lehner et al., 2005). The important role that water abstraction plays in energy supply is also evident from the fact that 31% of total water withdrawals from 30 European countries are used as cooling water in power stations (Flörke and Alcamo, 2005). Short-duration climate events such as windstorms and heavy rains may also increase in frequency, causing problems such as flooding (Christensen and Christensen, 2003). Longer-term changes in climate will also place considerable pressure on coastal areas through sea-level rise (Devoy et al., 2007). Further details of the observed and projected climate change impacts for Europe are provided by the IPCC in Alcamo et al. (2007).

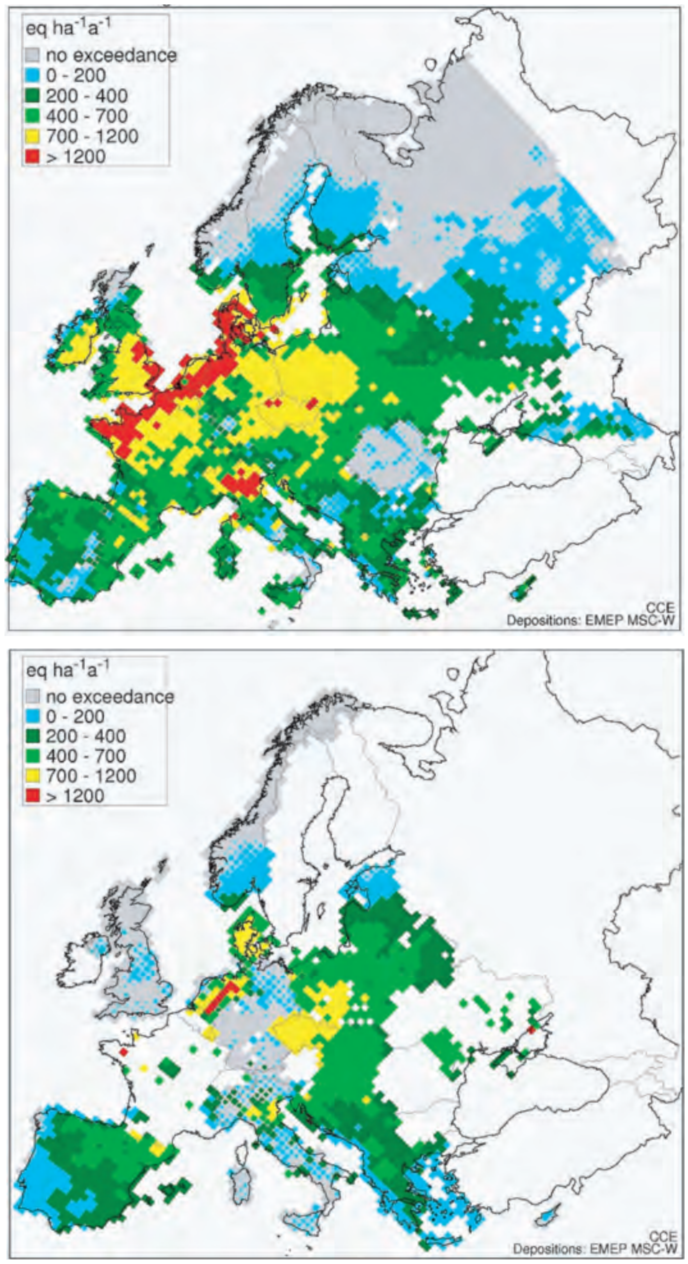

Figure 3.11 | Exceedance of critical loads for eutrophication for forest soils (top panel) and seminatural vegetation (grasslands, shrubs, etc., bottom panel) in the year 2000. The size of a colored grid cell is proportional to the fraction of the ecosystem area in the cell where critical loads are exceeded. Source: Hettelingh et al., 2008.

Air Quality Environmental Impacts in Europe

Eutrophication in Europe

Many plant species are endangered as a result of eutrophication in terrestrial ecosystems (WHO, 2006a). Ecosystems that include meadows, forests, and bogs that are characterized by low nutrient content and species-rich, slowly growing vegetation adapted to lower nutrient levels are overgrown by faster growing and more competitive species-poor vegetation, like tall grasses, that can take advantage of unnaturally elevated Nr levels. As a result, certain habitats may be changed beyond recognition, and vulnerable species may be lost (Hettelingh et al., 2007); for example, the majority of orchid species in Europe are considered at risk from eutrophication (WHO, 2006a). For the year 2000, it was estimated that N deposition had significantly exceeded thresholds that would guarantee ecological sustainability (i.e., critical loads) in most of the European forests and grassland areas (Figure 3.11). It should be noted that energy-related NOx emissions in Europe only account for about half of the total N deposition, with the rest coming from agricultural sources, in particular intensive livestock rearing (Stigliani and Shaw, 1990).

Methods have recently been developed that can be used to quantify the impacts, rather than merely identifying areas in exceedance, of excess N deposition on plant-species diversity (Hettelingh et al., 2008). These have estimated that current levels of Nr significantly degrade the species richness in many European ecosystems, leading to losses of up to 20% of the species in forests in northwest Europe (Figure 3.12).

Under the Natura 2000 program, the EU declared specific nature reserve areas to maintain and restore natural habitats. Among other stresses (e.g., from the fragmentation of habitats), ecosystems are under pressure

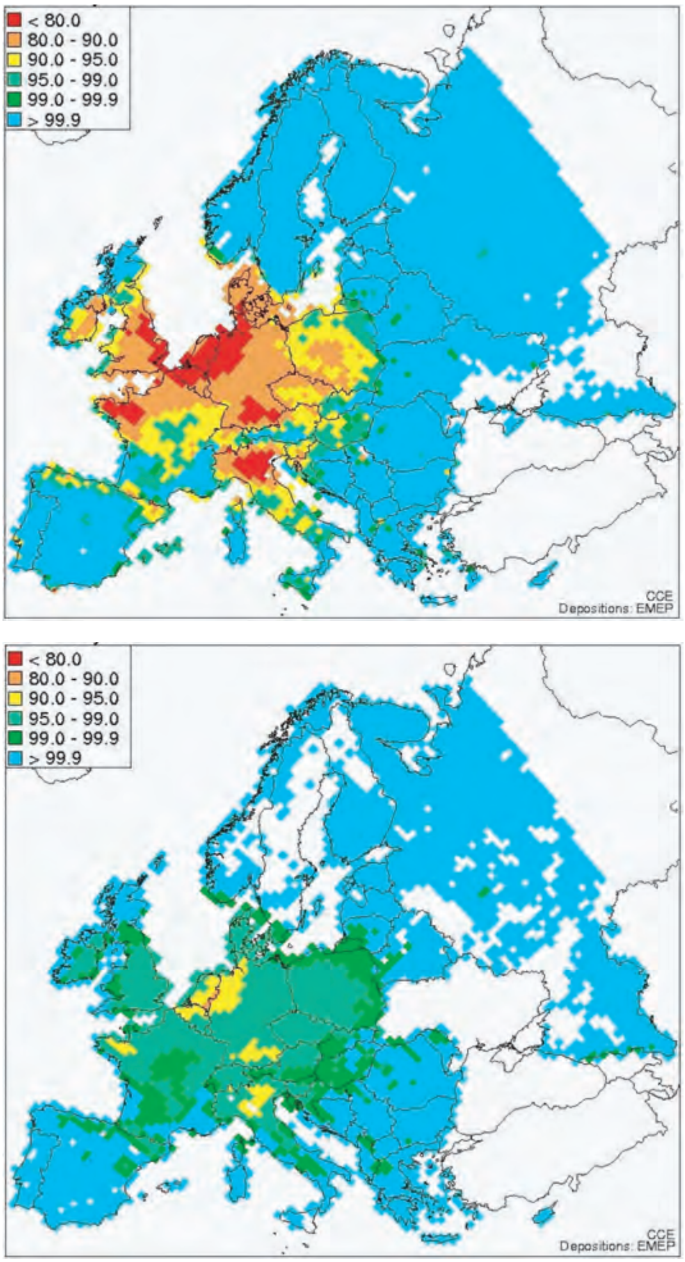

Figure 3.12 | Percentage of species richness in forests in Europe (top panel) and semi-natural vegetation (bottom panel) in the year 2000. Red-shaded areas indicate that the estimated biodiversity indicator percentages are lower than 80%, while green shadings indicate areas where this percentage is between 95–100%. Source: Hettelingh et al., 2008.

from excess deposition of atmospheric pollutants that affect plant species diversity and soil chemistry. For the year 2000, overly high deposition of N compounds constituted an important stress factor to Natura 2000 areas in Germany, the Netherlands, France, Poland, the Czech Republic, and Bulgaria, where deposition exceeded the tolerable levels of input (Figure 3.13).

Acidification in Europe

In the second half of the 20th century, the health of European forests, lakes, and rivers was heavily compromised by high acid deposition resulting from emissions of SO_2, NO_x, and NH_3. As a consequence, soils in forests and freshwater catchment areas, as well as lakes in Scandinavia, experienced strong acidification that led to plant damage and the disappearance of fish and other aquatic fauna. Since then, steep reductions in SO_2 and NO_x emissions have reduced deposition levels considerably, and many areas are now gradually recovering from past acidification. Nevertheless, current deposition rates are still exceeding sustainable levels for large forest areas in central Europe and freshwater catchments in Scandinavia and the United Kingdom (Figure 3.14). It is also clear that the full recovery of acidified soils will require deposition to be below critical-load thresholds for a substantial period of time in order to replenish the buffering capacity of soils, which has been depleted over the last decades. Thus, full recovery of acidified ecosystems in Europe would require SO_2 and NO_2 emissions to decline by 80–90% below current levels.

Ground-level Ozone (O_3)

In Europe, ground-level O_3 has been found to cause impacts to agricultural crops. These impacts include visible injury, (particularly important for leafy salad crop species (Emberson et al., 2003); declines in yields of arable crops (Mills et al., 2007); and alterations to quality of crop yields, for example the nitrogen content of harvestable products (Pleijel et al., 1999).

Exposure to ground-level O_3 also causes negative effects on sensitive forest trees, including reduced photosynthesis, premature leaf shedding, and growth reductions (Skarby et al., 1998). Ozone-sensitive forest tree species, including birch, beech, Norway spruce, Sessile oak, Holm oak, and Aleppo pine, are present across large areas of Europe (Karlsson et al., 2007). Ozone effects on these species have important negative consequences for carbon sequestration, biodiversity, and other ecosystem services that are provided by forest trees. Such services include reducing soil erosion and decreasing flooding and avalanches.

By impacting growth, seed production, and environmental stress tolerance, O_3 also affects the vitality and balance of seminatural vegetation ecosystems and the ecosystem services they provide. These services include carbon storage, water storage, and biodiversity (Fuhrer et al., 2009). The floral diversity of this vegetation type makes it more difficult to generalize about effects and to establish critical levels applicable across Europe. Widespread effects on *Trifolium* (clover species), an important component of productive pasture, have been found in O_3-exposure experiments (Mills et al., 2011). Effects include reductions in biomass, forage quality, and reproductive ability at ambient and near-ambient concentrations in many parts of Europe.

In recent years, research has focused on identifying appropriate indicators to quantify the risk of vegetation damage from O_3 (Emberson et al., 2007). Originally, risks for damage have been associated in Europe with the AOT40 (accumulated O_3 exposure over a threshold of 40 parts per billion) indicator, which measures O_3 concentrations during daylight hours that exceed a 40 ppb threshold, accumulated over the entire vegetation period (Fuhrer et al., 1997). This indicator suggests that for the year 2000 the largest risk to forest trees was in the Mediterranean countries (Figure 3.15).

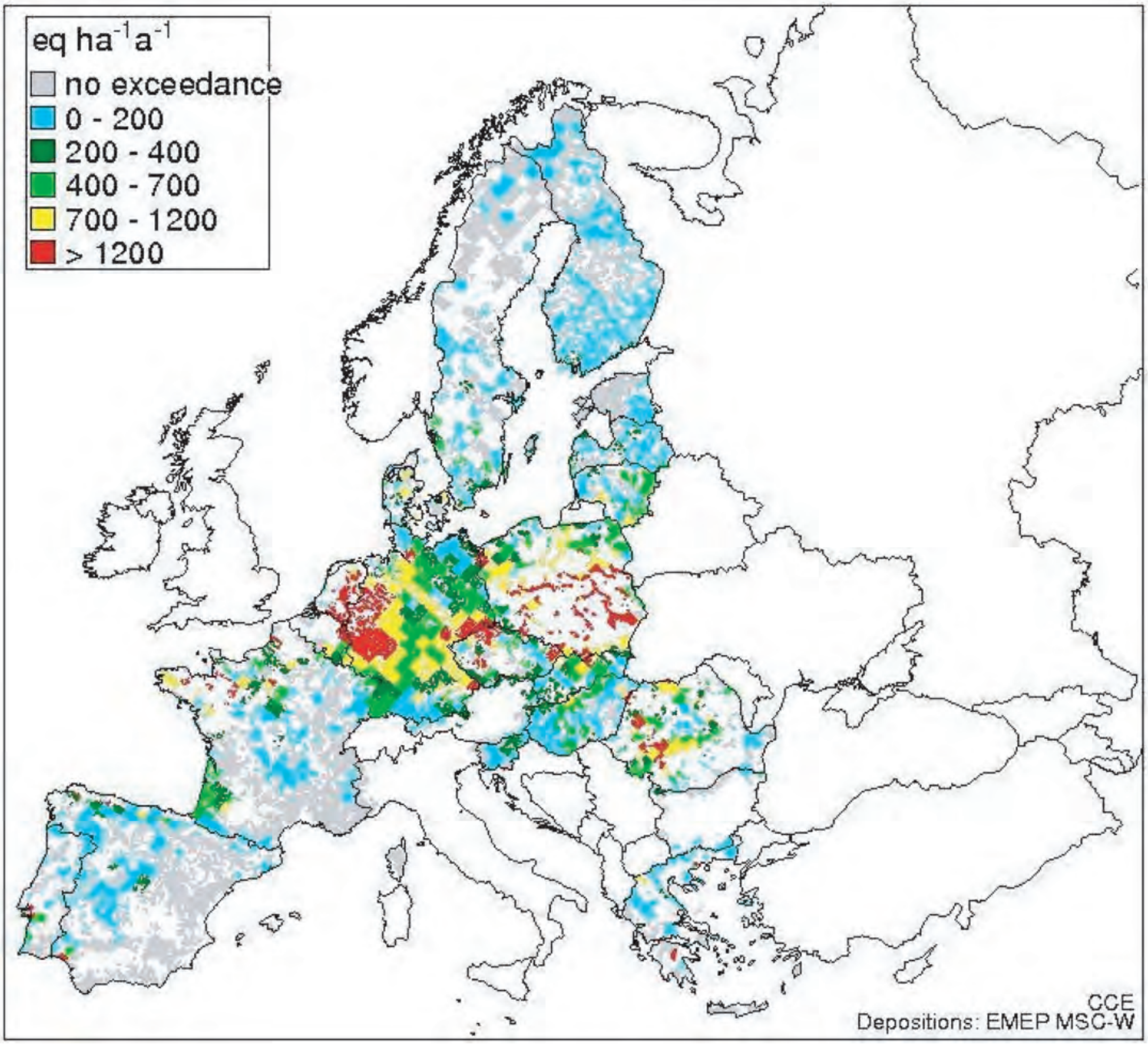

Figure 3.13 | Excess nitrogen deposition (Nr-eq/ha/yr) in Natura 2000 nature protection areas in 2000. Source: Hettelingh et al., 2008.

However, more recent work that associates actual vegetation damage with the O_3 dose that is absorbed by plants (often termed the 'ozone flux' approach) shows higher risks of O_3 in central Europe than indicated by AOT40. The areas at risk identified using this flux-based approach also bear a closer relation to those areas where vegetation damage has actually been found 'on the ground' (Mills et al., 2011).

Reductions in European precursor emissions of ground-level O_3 will certainly alleviate the pressure on vegetation. However, there is growing evidence of an increasing trend in hemispheric background concentrations of O_3 that could counteract the positive effects of measures within Europe (Royal Society, 2008). Effective response strategies that seek to eliminate the risk of vegetation damage from O_3 will therefore need to address not only sources in Europe, but also sources from other continents that may well be contributing to these increasing background concentrations.

3.2.4.2 The Americas

North, Central, and South American countries suffer the full range of environmental threats due to atmospheric emissions to which energy systems remain a major contributor. Problems exist in both rural and urban areas, including increased human morbidity and mortality plus agricultural, forest, water, visibility, and other welfare damage. Air-

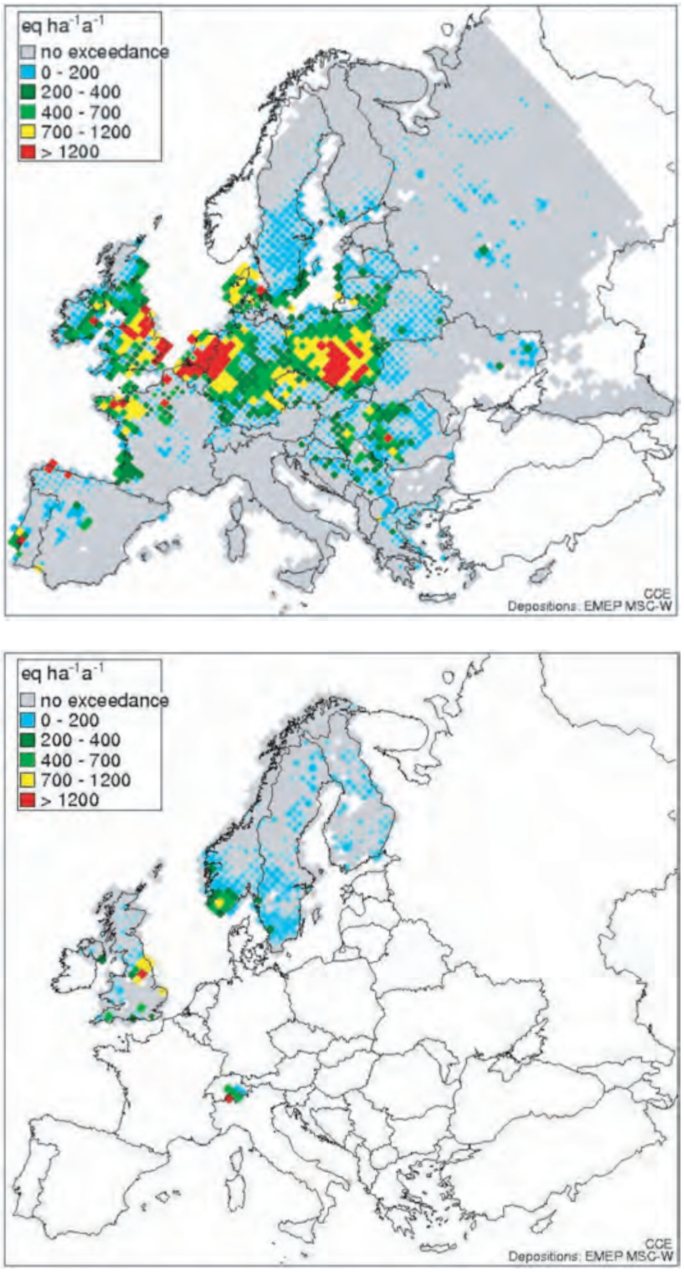

Figure 3.14 | Exceedance of critical loads of acidification (N-eq/ha/yr) in the year 2000 for forest soils (top panel) and freshwater catchment areas (bottom panel). The size of a colored grid cell is proportional to the fraction of the ecosystem area in the cell in which critical loads are exceeded. Source: Hettelingh et al., 2008.

quality deterioration in North American urban areas was initially noted in the first half of the 20th century, which resulted in the establishment of air-quality management programs in a number of the larger urban areas in the United States by 1950. Such programs were extended to Canada in the 1970s and to Mexico after 1980. Regional and global environmental threats were recognized in the late 1970s, when acid deposition was established as a significant problem in the northeastern part of the United States and in eastern Canada. By the 1980s, the potential threat of global climate change was recognized, along with the impacts from the release of toxic materials in all parts of North America. The Clean Air Act was established in 1970 to foster the growth of a strong American economy and industry while improving human health and the environment. Over the last 20 years, total emissions of the six principal air pollutants (also known as 'common' or 'criteria' pollutants) – PM, ground-level O_3, CO, sulphur oxides (SO_x), NO_x, and lead – have decreased by more than 41%. During the same period, GDP has increased by more than 64%.

Although significant progress has been made in improving the quality of the air in most US cities and communities, there is more to be done over the next 40 years. The development and use of energy in North America has been, and still remains, the prime source of environmental degradation. According to the Commission for Environmental Cooperation (CEC) in North America, there is a total of 3.17×10^{10} kilograms of criteria-related emissions emitted by the industrial sector in North America. Of these, 60% of the emissions are released by industrial sources in the United States, 26% by industrial sources in Mexico, and 14% by industrial sources in Canada (CEC, 2009).

Climate Change Environmental Impacts in the Americas

The Fourth Assessment Report of the IPCC states that there is high confidence that North America has experienced locally severe economic damage, plus substantial ecosystem, social, and cultural disruption from recent weather-related extremes, including hurricanes, other severe storms, floods, droughts, heat-waves, and wildfires (Field et al., 2007). There is also high confidence that climatic variability and extreme events have been severely affecting the Latin America region over recent years (Magrin et al., 2007).

Many coastal areas in North America are exposed to storm-surge flooding (Titus, 2005), especially those areas below sea-level. The breaching of New Orleans floodwalls following Hurricane Katrina in 2005 and storm-wave breaching of a dike in Delta, British Columbia, in 2006 demonstrate this vulnerability. Under El Niño conditions, high water levels combined with changes in winter storms along the Pacific coast have produced severe coastal flooding and storm impacts (e.g., Walker and Barrie, 2006). Significant impacts of projected climate change and sea-level rise are also expected for 2050–2080 on the Latin American coastal areas. With most of their population, economic activities, and infrastructure located at or near sea-level, coastal areas will be very likely to suffer floods and erosion, with high impacts on people, resources, and economic activities.

Changes in precipitation and increases in temperature are constraining over-allocated water resources, increasing competition among agricultural, municipal, industrial, and ecological uses across the Americas. In Latin America during the last decades, significant changes in precipitation and increases in temperature have been observed. Increases in rainfall in southeast Brazil, Paraguay, Uruguay, the Argentinean Pampas, and some parts of Bolivia have had impacts on land-use and crop yields, and have increased flood frequency and

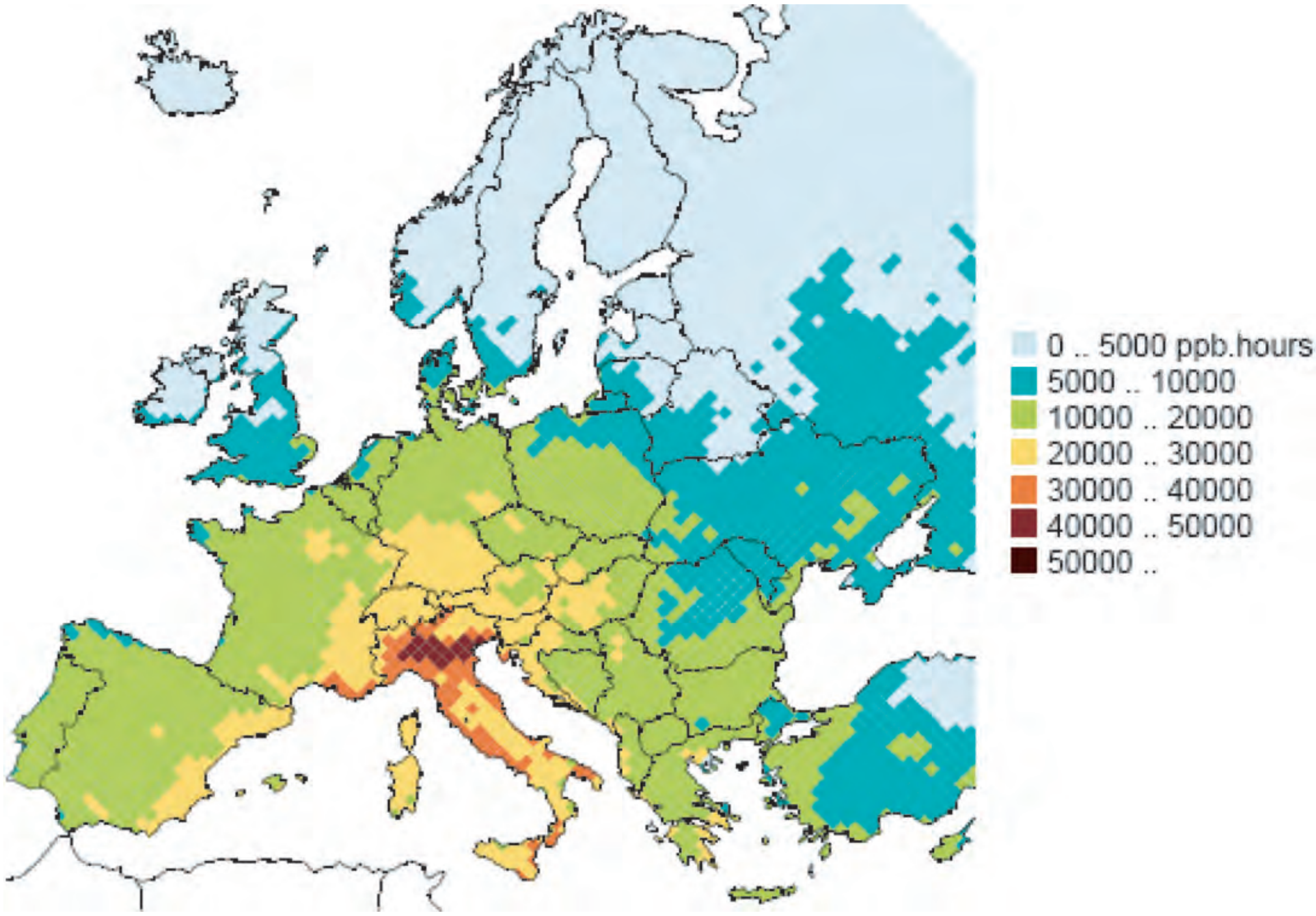

Figure 3.15 | An indicator of ozone damage to forest trees (AOT40) calculated for 2000. Source: Amann et al., 2011.

intensity. On the other hand, a declining trend in precipitation has been observed in southern Chile, southwest Argentina, southern Peru, and western Central America. As a consequence of temperature increases, the trend in glacier retreat is accelerating, with recent studies indicating that the volumes of most of the South American glaciers from Colombia to Chile and Argentina are decreasing at an accelerated rate (e.g., Leiva, 2006). During the next 15 years, intertropical glaciers are very likely to disappear, affecting water availability and hydropower generation. Hydropower production is known to be sensitive to total runoff, the timing of runoff, and to reservoir levels in North America. For example, during the 1990s, water levels in the Great Lakes fell as a result of a lengthy drought, and in 1999 hydropower production was down significantly both at Niagara and Sault St. Marie (CCME, 2003).

Climate change causes a risk of significant species extinctions in many areas of tropical Latin America. Up to 40% of the Amazonian forests could react drastically to even a slight reduction in precipitation. The tropical vegetation, hydrology, and climate system in South America could rapidly change to another steady state (Rowell and Moore, 2000). It is more probable that forests will be replaced by ecosystems that have more resistance to multiple stresses caused by temperature increases, droughts, and fires, such as tropical savannas. The replacement of tropical forest by savannas is expected in eastern Amazonia and the tropical forests of central and southern Mexico, along with the replacement of semiarid vegetation by arid vegetation in parts of northeast Brazil and most of central and northern Mexico, due to the synergistic effects of both land-use and climate changes (Magrin et al., 2007). By 2050, desertification and salinization will affect 50% of agricultural lands in Latin America and the Caribbean zone (FAO, 2004). Over the 21st century, pressure on species to shift north and to higher elevations will fundamentally rearrange North American ecosystems. Differential capacities for range shifts and constraints from development, habitat fragmentation, invasive species, and broken ecological connections will alter ecosystem structure, function, and services (Field et al., 2007).

Further details of the observed and projected climate change impacts for North America and Latin America and the Caribbean are provided by the IPCC in Field et al. (2007) and Magrin et al. (2007), respectively.

Air Quality Environmental Impacts in the Americas

Eutrophication

Although eutrophication was identified as an issue in North America in the 1970s, the problem is not ranked particularly highly within national environmental protection agencies. Nevertheless, the World Resource Institute, which carried out an assessment of eutrophication in North American coastal areas, found 131 areas that show symptoms of eutrophication (WRI et al., 2008).

Various studies in western North America (e.g., US EPA, 2008) demonstrate that some aquatic and terrestrial plant and microbial communities may be significantly altered by N deposition. For example, an accumulating weight of evidence has led some researchers to conclude that high-altitude watersheds in the Colorado Front Range show symptoms of ecological impacts, even at current N deposition levels. These effects include changes in alpine plant communities, elevated surface water nitrate concentrations, and changes in lake algal species communities. Further west, levels of stream water and groundwater nitrate in the San Gabriel and San Bernardino Mountains have been found to be strongly linked to the magnitude of N deposition in watersheds throughout the region. Stream water nitrate concentrations at Devil's Canyon in the San Bernardino Mountains and in chaparral watersheds with high smog exposure in the San Gabriel Mountains northeast of Los Angeles are the highest in North America for forested watersheds. Chronic N deposition and nitrate export from these watersheds contribute to the groundwater nitrate problems in the eastern San Gabriel Basin, where levels often exceed the federal drinking water standard.

A significant amount of research has been conducted since 1990 on the effects of N deposition on terrestrial ecosystems in the Los Angeles air basin in southern California (Fenn et al., 2003). Researchers have found that Nr enrichment, in combination with O_3 exposure, causes major changes in tree health by reducing fine root biomass and carbon allocation below ground and by greatly decreasing the life span of pine foliage. Nitrogen enrichment results in greater leaf growth, while O_3 causes premature leaf loss at the end of the growing season. The net result of these pollutant interactions is significant litter accumulation on the forest floor. Nitrogen cycling rates in soil are also stimulated by the high N inputs, resulting in large leachate losses of nitrate from these watersheds and elevated fluxes of NO gas from soil. Greenhouse and field studies indicate that N deposition may be one factor promoting the invasion of exotic annual grasses into coastal sage ecosystems occurring in low-elevation sites in the region.

One of the main adverse ecological effects resulting from N deposition, particularly in the Mid-Atlantic region of the United States, is the effect associated with nutrient enrichment in estuarine (Bricker et al., 2007) and coastal (Valigura et al., 2001) waters. Eutrophication in such ecosystems is associated with a range of adverse ecological effects, including low dissolved oxygen, harmful algal blooms, loss of submerged aquatic vegetation, and low water clarity. These changes disrupt aquatic habitats; cause stress to fish and shellfish (which in the short term can lead to episodic fish kills, and in the long-term can damage growth in fish and shellfish populations); and cause aesthetic impairments to estuaries. A recent assessment of 141 estuaries in the United States by the National Oceanic and Atmospheric Administration concluded that 64 estuaries (45%) suffered from moderately high or high levels of eutrophication due to excessive inputs of both Nr and phosphorus, with a majority of these estuaries being located in the coastal area from North Carolina to Massachusetts (Bricker et al., 2007). For estuaries in the Mid-Atlantic region, the contribution of atmospheric distribution to total Nr loads is estimated to range between 10% and 58% (Valigura et al., 2001).

Acidification and Visibility

Acid deposition and visibility reduction were noted as significant issues in the United States in the 1980s. In 1990, the US Clean Air Act was amended to specifically address these issues. The US program to address acid rain, which began in 1995, required US power plants to reduce SO_x emissions. A second phase of the SO_x reduction program began in 2000. Overall, the goal of the program has been to reduce SO_x emissions by 10 Mt/yr and cap emission levels from power plants at 8.95 Mt/yr in 2010. In addition to the requirement to reduce SO_x emissions, the US Environmental Protection Agency (US EPA) was required to adopt programs to reduce NO_x emissions by 2 Mt/yr beyond levels projected to occur without the program. Figure 3.16 indicates the emission reductions in NO_x and SO_2 in the United States between 1990 and 2009 (US EPA, 2010b) in relation to changes in electricity generation and retail price.

The success of the US program to address acid rain can be seen in monitoring data collected by the National Atmospheric Deposition Program/National Deposition Trends Network, or NADP/NTN (US EPA, 2010b). The data show significant improvements in the deposition of S and N across the United States (Figure 3.17). Reductions in N deposition recorded since the early 1990s have been less pronounced than those for sulphur.

Eastern Canada also suffers from acid deposition similar to that found in the northeastern United States. More than half of the acid rain in eastern Canada comes from emissions in the United States (Environmental Canada, 2010). It is estimated that 68% of the Canadian emissions are from industrial sources, while 67% of the US emissions are from electric power plants. In Mexico, there is concern that acid deposition is damaging the ancient Maya ruins that can be found in many parts of the country (Bravo et al., 2006).

Acidification Impacts on Aquatic Ecosystems

Acid deposition resulting from SO_2 and NO_x emissions is one of many large-scale anthropogenic impacts that negatively affect the health of lakes and streams in the United States (US EPA, 2010b). Surface-water chemistry provides direct indicators of the potential effects of acidic deposition on the overall health of aquatic ecosystems. Long-term surface water monitoring networks provide information on the chemistry of lakes and streams and on how water bodies are responding to changes in emissions. Since the 1980s, scientists measuring changes in a number

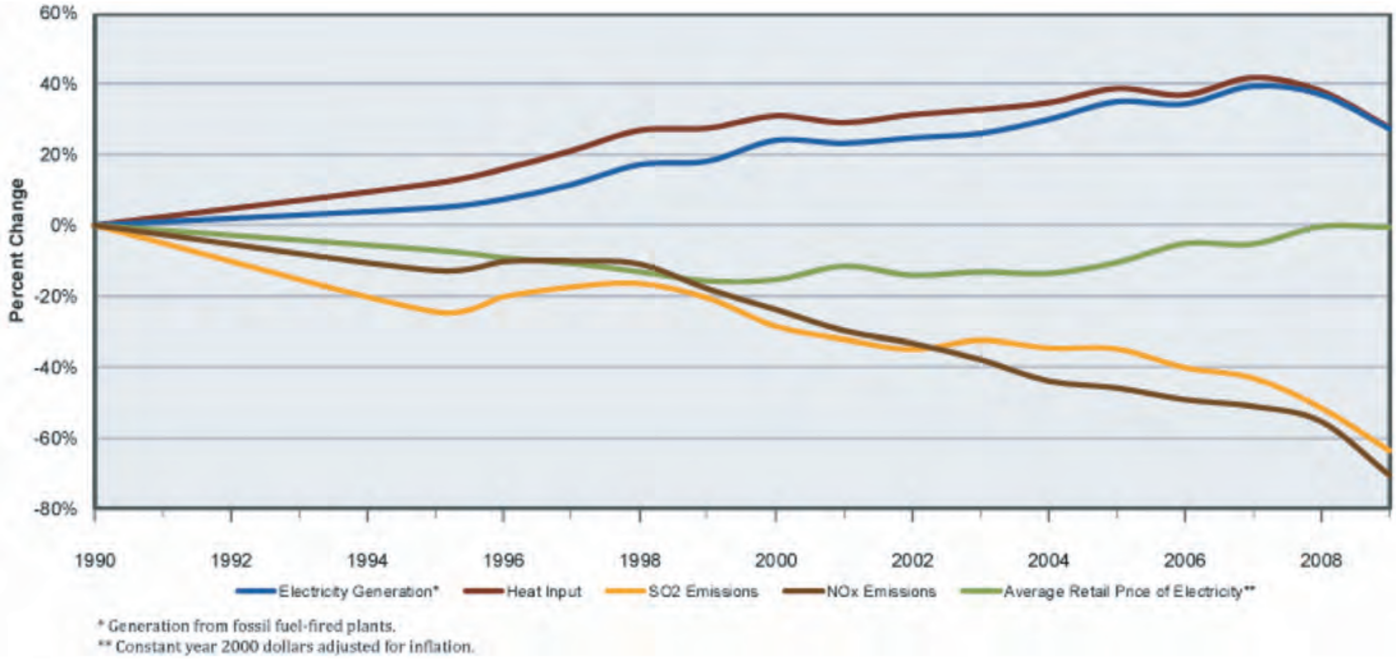

Figure 3.16 | Trends in SO_2 and NO_x emissions in relation to electricity generation and retail price from 1990 to 2009. Source: US EPA, 2010b.

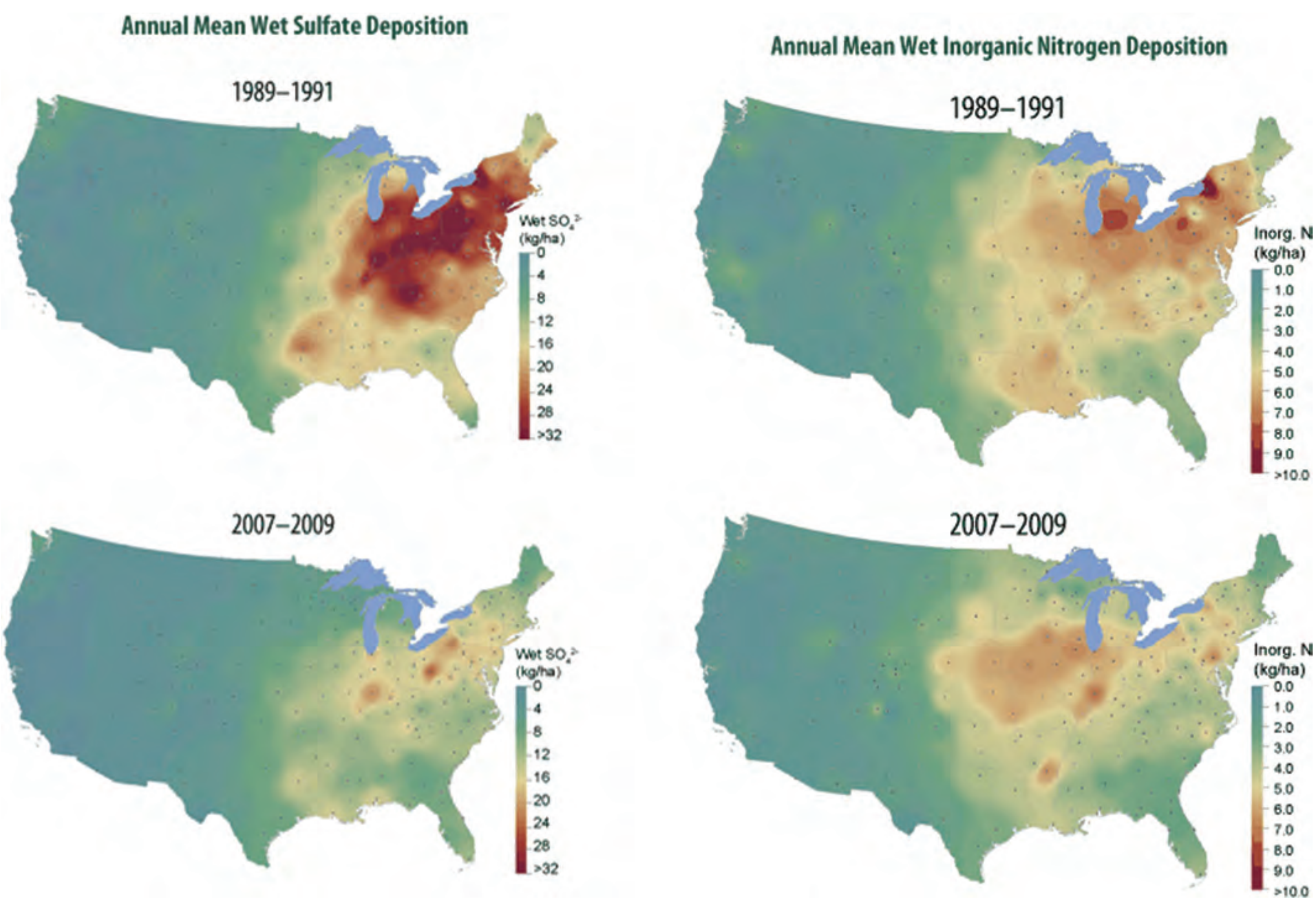

Figure 3.17 | Reductions in sulphate and nitrogen deposition in the United States between 1989 and 1991. Source: NADP/NTN monitoring data, US EPA, 2010b.

of lakes and streams in the eastern United States have found signs of recovery in many, but not all, of those bodies of water. These positive signs are a result of emission reductions that have occurred since the early 1990s (US EPA, 2010b).

The Long-Term Monitoring (LTM) program provides information on the effects of acid rain on aquatic systems. This program was designed to track the success of the 1990 Clean Air Act Amendments in reducing the acidity of surface waters in four regions: New England, the Adirondack Mountains, the Northern Appalachian Plateau, and the Central Appalachians (the Ridge and Valley and Blue Ridge Provinces). The surface-water chemistry trend data in the four regions monitored by the Long-Term Monitoring program (US EPA, 2010b) is essential for tracking the ecological response to emission reductions. One major finding of the program was that sulphate concentrations are declining at most sites in the Northeast (New England, Adirondacks, and the Catskills/Northern Appalachian Plateau). However, in the Central Appalachians, sulphate concentrations in some streams (21%) are increasing as a result of a decreasing proportion of the deposited sulphate being retained in the soil and an increasing proportion being exported to surface waters. Another important finding was that nitrate concentrations are decreasing in some of the sites in all four regions, but several lakes and streams indicate flat or slightly increasing nitrate trends. This trend does not appear to reflect changes in atmospheric pollutant deposition in these areas and is likely a result of ecosystem factors. A third finding was that acid-neutralizing capacity, as measured in surface waters, is increasing in many of the sites in the Adirondack and Catskills/Northern Appalachian Plateau regions, which can be attributed in part to declining sulphate deposition.

The critical load concept, which was first developed in Europe under the LRTAP Convention to assess the effects of acidification (see Section 3.2.3), has been used in Canada since the 1980s. The concept is currently being considered for use in the United States. The most recent analysis of the status of North American ecosystems with respect to acidification uses this critical load concept. This analysis is found in the Progress Report of the United States and Canada Air Quality Agreement, known as AQA (IJC, 2010). This report shows that in Canada, lakes that are highly acid-sensitive exist throughout northern Manitoba and Saskatchewan. Critical loads were set to protect 95% of the lake ecosystems. Critical-load exceedance occurs close to the base metal smelters in Manitoba and downwind of the oil sands operations in western Alberta. The exceedances were almost entirely due to sulphate deposition. Nitrogen inputs to the lakes, while significant, were virtually entirely retained within their catchments (lake water nitrate levels were below analytical detection in most cases), meaning that at present, N deposition is not an acidifying factor. It is estimated that lakes having critical loads as low as those observed in northern Manitoba and Saskatchewan will be threatened by long-term acid inputs. However, they do not presently exhibit obvious symptoms of chemical damage from anthropogenic acidic deposition (i.e., low pH and/or reduced alkalinity). Hence, there is still time to protect them from the acidification effects observed in many eastern Canadian lakes.

In the United States, the critical load approach is not yet officially accepted for ecosystem protection. For example, language specifically requiring a critical load approach does not exist in the Clean Air Act. Nevertheless, several federal agencies are now employing critical load approaches to protect and manage sensitive ecosystems. Modelling studies (US EPA, 2008) show that for the period 1989–1991, 56% of modelled lakes and streams received acid deposition greater than their estimated critical load, compared to approximately 36% in the 2006–2008 period. However, there are still regions where exceedances of critical loads occur. Areas with the greatest concentration of lakes in which acid deposition currently is greater than – or exceeds – estimated critical loads include the Adirondack Mountain region in New York; southern New Hampshire and Vermont; northern Massachusetts; northeast Pennsylvania; and the central Appalachian Mountains of Virginia and West Virginia (US EPA, 2008). Therefore, even though there has been improvement in acidic deposition rates over the past decade (US EPA, 2010b) (see Figure 3.17) application of the critical load concept and estimates from the scientific literature indicate that acid-sensitive ecosystems in the northeastern United States are still at risk of acidification at current deposition levels. As a result, additional reductions in acidic deposition from current levels might be necessary to protect these aquatic ecosystems (US EPA, 2008).

Acidification Impacts on Terrestrial Ecosystems

Certain ecosystems in the continental United States are potentially sensitive to terrestrial acidification, which is the greatest concern regarding N and S deposition (US EPA, 2008). Figure 3.18 depicts the areas across the United States that are potentially sensitive to terrestrial acidification. Current understanding of the effects of acid deposition on forest ecosystems has increasingly focused on the biogeochemical processes that affect plant uptake, retention, and cycling of nutrients within forested ecosystems. Research conducted during the 1990s indicated that decreases in base cations (e.g., calcium, magnesium, and potassium) from soils are at least partially attributable to acid deposition in the northeastern and southeastern United States (US EPA, 2008).

Ground-level Ozone (O_3)

Across North America, the approaches to establishing air quality guidelines for O_3 differ. In Canada, the federal government sets National Ambient Air Quality Objectives (NAAQOs) on the basis of recommendations from the National Advisory Committee and Working Group on Air Quality Objectives and Guidelines. Currently, 'desirable' and 'acceptable' NAAQOs for O_3 are defined as 50 ppb and 80 ppb respectively, expressed as an average O_3 concentration over a one-hour period. Provincial governments have the option of adopting these either as objectives or as enforceable standards, according to their legislation (Environment Canada, 2010).

In the United States, the US EPA is responsible for setting National Ambient Air Quality Standards (NAAQs), which are legally enforced. A revision of the NAAQs is currently ongoing as of 2011, and includes the possibility of introducing a more stringent secondary standard in addition to the primary standard that is designed to protect human health.

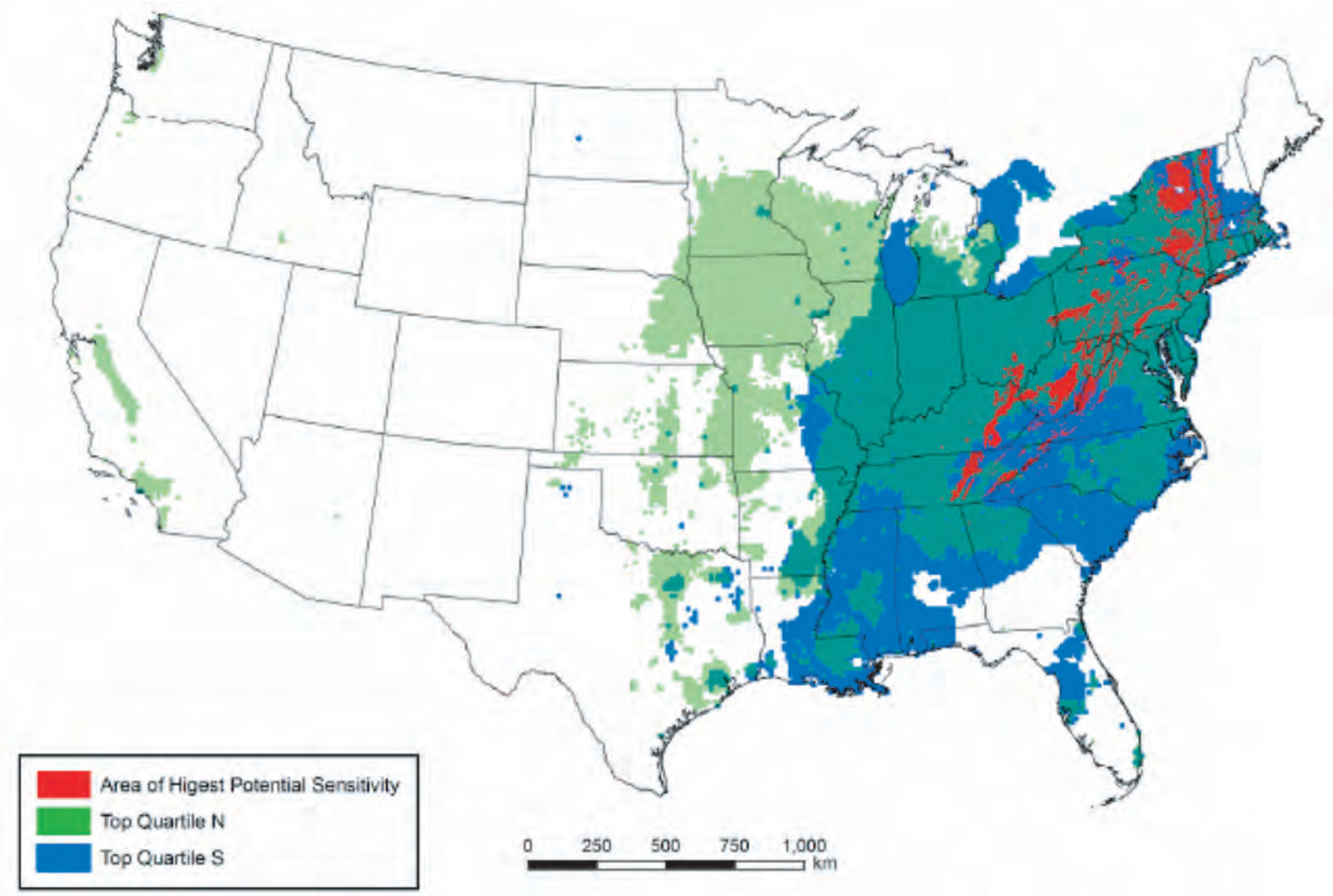

Figure 3.18 | Areas potentially sensitive to terrestrial acidification. Source: US EPA, 2011c.

The secondary standard protects against welfare effects (e.g., impacts on vegetation, crops, ecosystems, visibility, climate, man-made materials, etc.). Currently, the primary standard is 75 ppb averaged over an eight-hour period, or 120 ppb averaged over a one-hour period, with the secondary standard the same as the primary standard (US EPA, 2011a). Long-term trends in O_3 concentrations across the United States reflect notable decreases of approximately 29% in the second highest one-hour O_3 concentrations from 1980–2003, and of about 21% in the fourth highest eight-hour O_3 concentration during the same time period (see Figure 3.19). However, the effects of rising background O_3 concentrations (Vingarzen et al., 2004) and worsening chronic O_3 concentrations are uncertain in relation to ecosystem effects, since most experimental studies have used fumigation profiles that mimic episodic O_3 peaks.

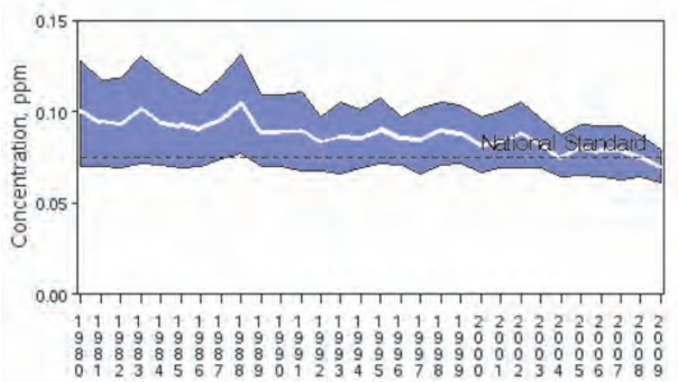

Figure 3.19 | Long-term trends in O_3 concentrations for the United States (1980–2003), based on 255 sites, represented as the fourth highest eight-hour O_3 concentration, with a 30% decrease in national average. Source: US EPA, 2011a.

The ecological effects of O_3 appear to be widespread across North America, based on biomonitoring studies and forest health surveys (US EPA, 2011b). The majority of existing experimental evidence comes from biomonitoring and Open Top Chamber studies that were conducted in North America under the National Crop Loss Assessment Network Programme (Heck et al., 1988). In recent decades, there has been a shift from chamber-based studies to field-based approaches to assess O_3 impacts. In the United States, such studies have investigated forest trees (aspen, maple, and birch) in Wisconsin (Karnosky et al., 2005) and crops (soybeans) in Illinois (Morgan et al., 2006). These studies tend to confirm the detrimental effects of O_3 exposure on vegetation that were found in chamber studies (King et al., 2005; Long et al., 2005).

Extrapolation of data provided by these site-specific studies to the entire United States suggests serious risk to crops and forests from O_3 pollution. The 'cornbelt' in the United States produces 40% of the world's corn and soybean crops, and this region is already potentially losing 10% of its soybean production to O_3 (Tong et al., 2007). In the United States as a whole, agronomic crop loss to O_3 is estimated to range from 5–15%, with an approximate cost of $3–5 billion annually (Fiscus et al., 2005; US EPA, 2006). Despite the overwhelming evidence that current O_3 concentrations are causing yield losses, new O_3-tolerant crop cultivars are not being developed for a future world that could be characterized by higher levels of O_3 (Ainsworth et al., 2008). In addition to regional-scale O_3 risk assessment based on site-specific experimental data, epidemiologic methods have also been used for crop-loss assessments (Fishman et al., 2010). These studies have found that the influence of O_3 can be detected in regional-level production statistics and field trial data, though damage estimates have been found to differ from those obtained from risk assessments performed using empirically derived dose-response relationships. This may be due to the fact that these methods are most effective in those regions characterized by higher average O_3 concentrations (Fishman et al., 2010), where the ozone signal is strong enough to overcome the influence of confounding variables affecting yield.

For forest trees, O_3 has been shown to cause visible foliar injury, accelerate leaf senescence, reduce photosynthesis, alter carbon allocation, and reduce growth and productivity (Karnosky et al., 2007). These effects vary by forest tree species and genotype (Karnosky and Steiner, 1981). Since the mid 1950s, the San Bernardino mixed coniferous forest in southern California has been exposed to some of the highest O_3 concentrations in North America. This exposure has affected such species as ponderosa and Jeffrey Pines. When concentrations were at their height in the 1970s, annual tree mortalities of between 2–2.5% were being reported (Miller and McBride, 1999). Extrapolation of site-specific experimental data in order to perform regional assessments in the United States provides estimates of annual yield losses equivalent to US$80 million (Muller and Mendelsohn, 2007). Finally, although few experimental studies have been conducted investigating the effects of O_3 on seminatural ecosystems in the United States, those that do exist have highlighted O_3 impacts on the nutritive quality of forage crops (Krupa et al., 2004).

Unlike in North America, in South America there have been relatively few studies on the impact of O_3 on vegetation. There is also a dearth of observational evidence on the subject. The most notable exceptions are studies conducted in Mexico in the vicinity of Mexico City. These studies have found evidence of visible foliar injury occurring on forest species, especially in the Valley of Mexico and in Desierto de los Leones National Park (Hernandez and de Bauer, 1984; Hernandez and de Bauer, 1986).

3.2.4.3 Africa

Energy supply and demand in Africa is extremely heterogeneous. For example, northern Africa is a major supplier of oil and gas, while in sub-Saharan Africa more than 70% of the energy demand is met by biomass, mostly from wood and charcoal. (see Chapters 19 and 23 for discussions on energy access issues in Africa). Wood, including charcoal, is the most environmentally detrimental biomass energy source, as it leads to considerable deforestation (Okello, 2001), which is one of the most pressing environmental problems faced by many African nations. Forests cover about 22% of the region, but they are disappearing faster than anywhere else in the world (FAO, 2005). Many sub-Saharan African countries have suffered depletion of more than three quarters of their forest cover (Energy Information Administration, 2000). Africa lost 10.5 % of its forests during the 1980s. It is estimated that if current trends continue, many areas, especially those in the Sudano-Sahelian belt, will experience a severe shortage of fuelwood by 2025 (Energy Information Administration, 2000). It should also be noted that producing and using charcoal is less efficient than using fuelwood directly and leads to forest depletion in rural areas providing fuel to cities (WHO, 2002). Deforestation also causes other problems, such as increased erosion and loss of biodiversity, and has contributed to desertification. Other environmental impacts directly associated with energy systems in Africa include atmospheric emissions from gas 'flaring' and impacts of large-scale hydropower.

Climate Change Environmental Impacts

Although Africa's contribution to global GHG emissions is only about 3%, the continent is more vulnerable than many of the world's regions to the impacts of climate change (Davidson et al., 2007). This vulnerability stems from the fact that, on average, Africa is hotter and drier than most other regions of the world and has less dependable rainfall. Moreover, Huang et al. (2009) show a large-scale effect of aerosols on precipitation in the West African monsoon region. They report a statistically significant precipitation reduction associated with high aerosol concentrations near the coast of the Gulf of Guinea from late boreal autumn to winter. Aerosols originate from various African sources. For example, large quantities of desert dust and biomass-burning smoke are emitted during much of the year across the African continent.

Climate change and variability have the potential to impose additional pressures on water availability, water accessibility, and water demand in Africa (Boko et al., 2007). The population at risk of increased water stress due to climate change is projected to be 75–250 million and 350–600 million people by the 2020s and 2050s, respectively (Arnell, 2004). The impact on water resources is likely to be greatest in northern and southern Africa. In eastern and western Africa, however, more people will be likely to experience a reduction in water stress (Arnell, 2006). In the future, climate change may become a contributing factor to conflicts, particularly those concerning water scarcity (Fiki and Lee, 2004).

It has been estimated by Mendelsohn et al. (2000) that by 2100, parts of the Sahara are likely to emerge as the most vulnerable to climate change, showing likely agricultural losses of between 2–7% of GDP. Western and central Africa are also vulnerable, with impacts ranging from 2–4% of GDP. By contrast, it's estimated that northern and southern Africa will experience losses of 0.4–1.3% GDP.

It is predicted that a significant decrease in the extent of suitable rain-fed land and production potentials for cereals will occur by the 2080s as a result of climate change. Furthermore, for the same projection and time horizon, the area of semiarid land in Africa could increase by 5–8%. southern Africa would be likely to experience a notable reduction in maize production under possible increased El Niño/La Niña-Southern Oscillation (ENSO) conditions (Stige et al., 2006). In some countries, additional risks that could be exacerbated by climate change include greater erosion, deficiencies in yields from rain-fed agriculture of up to 50% during the 2000–2020 period, and reductions in the length of the crop-growing season (Agoumi, 2003). Other agricultural activities could also be affected by climate change and variability, including changes in the onset of rainy days and the variability of dry spells (Reason et al., 2005).

It has been suggested that climate change will have a range of impacts on terrestrial and aquatic ecosystems. For example, two climatic-change scenarios used to assess the sensitivity of African mammals in 141 national parks in sub-Saharan Africa suggest that climate change will have an impact on species diversity. Assuming no migration of species, Boko et al. (2007) projected that 10–15% of the species will fall within the International Union for Conservation of Nature (IUCN) Critically Endangered or Extinct categories by 2050, a figure that will increase to 25–40% by 2080. Assuming unlimited species migration, the results were less extreme, with these percentages dropping to approximately 10–20% by 2080.

In Africa, highly productive ecosystems (mangroves, estuaries, deltas, lagoons, and coral reefs), which form the basis for important economic activities such as tourism and fisheries, are located in the coastal zone. The projected rise in sea-level will have significant impacts on the coastal megacities of Africa because of the concentration of poor populations in areas that are especially vulnerable to such changes (Klein et al., 2002; Nicholls, 2004). By 2080, across a range of SRES scenarios and climate change projections, three of the five regions shown to be at risk of flooding in coastal and deltaic areas of the world are located in Africa (Nicholls and Tol, 2006). Sea-level rise, combined with increases or decreases in rainfall, will alter the penetration of salt water into estuaries and could induce overtopping and even destruction of the low barrier beaches that limit the coastal lagoons, while changes in precipitation could affect the discharges of rivers feeding them. These changes could also affect lagoonal fisheries and aquaculture (République de Côte d'Ivoire, 2000).

The Indian Ocean islands could also be threatened by potential changes in the location, frequency, and intensity of cyclones, while East African coasts could be affected by potential changes in the frequency and intensity of ENSO events and coral bleaching (Klein et al., 2002). Coastal agriculture (e.g., plantations of palm oil and coconuts in Benin and Côte d'Ivoire, shallots in Ghana) could be at risk of inundation and soil salinization. In Kenya, losses for three crops (mangoes, cashew nuts, and coconuts) could cost almost US$500 million for a 1 meter sea-level rise (Republic of Kenya, 2002). In Guinea, between 130–235 km^2 of rice fields (17–30% of the existing rice field area) could be lost as a result of permanent flooding, depending on the inundation level considered (between 5–6 m) by 2050 (République de Guinée, 2002). In Eritrea, a 1 meter rise in sea-level could cost over US$250 million, as a result of the submergence of infrastructure and other economic installations in Massawa, one of the country's two port cities (State of Eritrea, 2001). These results confirm previous studies stressing the great socioeconomic and physical vulnerability of settlements located in marginal areas.

Initial assessments show that several regions in Africa may be affected by different impacts of climate change (Figure 3.20). Such impacts may further constrain attainment of the Millennium Development Goals in Africa.

Air Quality Environmental Impacts

Increased activities in key social and economic sectors (see Table 3.1) are contributing significantly to air pollution in Africa. Unsustainable patterns of use and supply of energy resources by industry, transport, and household sectors have been particularly important as sources of indoor and outdoor air pollutants. Generally, emission levels of NO_2 and SO_2 have increased significantly in many African countries due to the region's industrial activity. In addition, about 1.8 $MtSO_2$/yr are emitted from electricity generation alone (UNEP, 2006). However, Table 3.1 also shows the significant contribution to PM, NMVOCs, NO_x, and CH_4 pollution resulting from agricultural activities (i.e., not associated with energy systems).

Africa's rate of urbanization is the highest in the world. The rate is increasing rapidly, with urbanization rates in sub-Saharan Africa generally in excess of 4% (Clancy, 2008). This results in rapidly increasing energy use and pollutant emissions from industry, motor vehicles, and households, including SO_x, CO_2, and NO_x, as well as PM and other organic compounds (UNEP, 2006). The transportation sector is increasingly being recognized as the highest polluter in key African cities such as Cairo, Nairobi, Johannesburg, Cape Town, Lagos, and Dakar (UNEP, 2006). The transport systems in these cities are emitting tonnes of air pollutants, mainly NO_x, PM, and VOCs. Some of the challenges related to transportation are the use of old vehicles without emission controls; rapidly increasing fleets; an increase in two- and three-wheeler vehicles with 'dirtier' two-stroke engines; absent or improper vehicle maintenance; lack of cleaner burning fuels; and absent or poor regulatory frameworks for vehicle emissions and their enforcement (Schwela, et al., 2007). In addition, inadequate urban planning and poor road networks have led to traffic congestion in most African cities, with impacts on fuel wastage and air pollution. Traditional cooking is also a significant emitter of CO_2, CO, and NO_x (UNEP, 2006). Charcoal production emits significant levels of CH_4, CO, and other products of incomplete combustion and PM (UNEP, 2006).

However, most African countries do not have monitoring networks to measure air pollutants. Therefore, no comparisons of air quality can be made at the subregional level. Recognizing such limitations, the Air Pollution Information Network for Africa (APINA) has identified the lack of reliable emission inventories, the lack of experience in the use of atmospheric models, and the lack of data on measured impacts as important

Figure 3.20 | The vulnerability of different African regions to the multiple stresses likely to result from climate change. These impacts include desertification, sea-level rise, reduced freshwater availability, cyclones, coastal erosion, deforestation, loss of forest quality, woodland degradation, coral bleaching, the spread of malaria, and impacts on food security. Source: UNEP/GRID-Arendal, 2005. Courtesy of Delphine Digout, Revised by Hugo Ahlenius, UNEP/GRID-Arendal.

Table 3.1 | Contribution of key social and economic sectors to air pollution in Africa.

Pollutant	Commercial energy supply	Traditional energy supply	Agriculture	Manufacturing, other
Particulate Matter (PM)	35% (fossil fuel fires)	10% (traditional fuel fires)	40% (agriculture fires)	15% (smelting, land clearing, municipal, etc.)
Lead (Pb)	41% (gasoline burning)	Negligible	Negligible	59% (metal processing, manufacturing, municipal)
Cadmium (Cd)	13% (fossil fuel burning)	5% (traditional fuel burning)	12% (agriculture fires)	70% (metal processing)
Mercury (Hg)	20%	1% (traditional fuel burning)	2% (agriculture fires)	77%
Non-methane hydrocarbons (NMVOCs)	35% (fossil fuel burning)	5% (traditional fuel burning)	40% (agriculture fires)	20% (non-agricultural land clearing)
Sulphur dioxide (SO_2)	85% (fossil fuel burning)	0.5% (traditional fuel burning)	1% (agriculture fires)	13% (smelting, municipal)
Nitrogen oxides (NO_x)	30% (fossil fuel burning)	2% (traditional fuel burning)	67% (fertilizer, agriculture fires)	1% (municipal)
Carbon dioxide (CO_2)	75% (fossil fuel burning)	3% (net deforestation for fuelwood)	15% (net deforestation for land clearing)	7% (net deforestation for lumber, cement manufacturing)
Methane (CH_4)	18% (fossil fuel burning)	5% (traditional fuel burning)	65% (rice fields, animals, land clearing)	12% (landfills)
Nitrous oxide (NO)	12% (fossil fuel burning)	8% (traditional fuel burning)	80% (fertilizer, land clearing)	Negligible

Source: UNDP, 2000.

knowledge gaps to understanding the status of air quality across Africa (UNEP, 2006). Other challenges include weak national energy policies as well as the lack of local air-pollution exposure data. Such data would be necessary to establish air-quality standards that could be used to drive policy making targeted at reducing emissions. At the regional level, APINA and UNEP, together with other global air pollution institutions, have facilitated subregional agreements on air pollution that include the Lusaka Agreement (2008) – Southern African Development Community Regional Policy Framework on Air Pollution; the Eastern Africa Regional Framework Agreement on Air Pollution (Nairobi Agreement–2008); and the West and Central Africa Regional Framework Agreement on Air Pollution (Abidjan Agreement–2009). These agreements detail the actions that are required to reduce air pollution, including emissions from energy use, but the financial means and political will to implement them are still limited. Such policy processes will be important, given the variety of the proved energy potential that exists in Africa. At the global level, most African countries are signatories to the UNFCCC and produce national communications that are GHG inventories. However, they do not produce inventories for the other pollutants.

Acidification and Eutrophication

Africa has many ecosystems with a very high biodiversity value and which provide important ecosystems services. There is potential for sensitive ecosystems to be detrimentally affected by acidifying and eutrophying deposition. However, further research is required in this area to assess the full extent of ecosystem damage.

Acidification is potentially a problem for sensitive terrestrial and aquatic ecosystems downwind of major emission sources of SO_x and NO_x. One such sensitive ecosystem is in Mpumalanga on the Highveld of South Africa, where there are eight coal-fired power stations (Josipovic et al., 2010). A concentration, distribution, and critical level exceedance assessment of SO_2, NO_2, and O_3 in South Africa (Josipovic et al., 2010) shows that some critical levels for vegetation are only exceeded in the central area of the South African industrial Highveld where emission sources are high. Pollutant concentrations are below the critical thresholds for environmental damage in remote areas, including the sensitive forested regions of the Drakensberg escarpment. However, it is possible that critical loads for acidification may be exceeded in areas with sensitive soils and higher rates of total deposition (Fey and Dodds, 1998). There are also soils that are potentially sensitive to acidic deposition across western and central Africa, but deposition levels of S and N are still relatively low in these areas (Kuylenstierna et al. 2001), although they are projected to increase by 2030 (Dentener et al., 2006).

It is thought that between 2000 and 2030, deposition levels in Africa are likely to increase, in part due to increased intensity in the use of fossil fuel to meet increasing energy demand; NO_x emissions in sub-Saharan Africa resulting from fossil fuel combustion are already estimated to be 14 Mt/yr (Selman and Greenhalgh, 2009). Such enhanced deposition may seriously affect the integrity of protected areas, especially those in west and central Africa.

Ground-level Ozone (O_3)

Sunlight and heat stimulate O_3 formation. Therefore, the potential for elevated O_3 is high in many areas of Africa because of the combination of solar radiation and the high emission of precursors such as NO_x and VOCs, from both human activities and natural sources. There is a sharp increase of O_3 with altitude in the tropics, leading to a risk of vegetation damage in high mountainous regions (UNEP, 2006).

Studies in Africa have shown that current-day ambient air pollution concentrations can significantly damage human health, crops, local vegetation, and other materials. Ozone injury to turnips has been

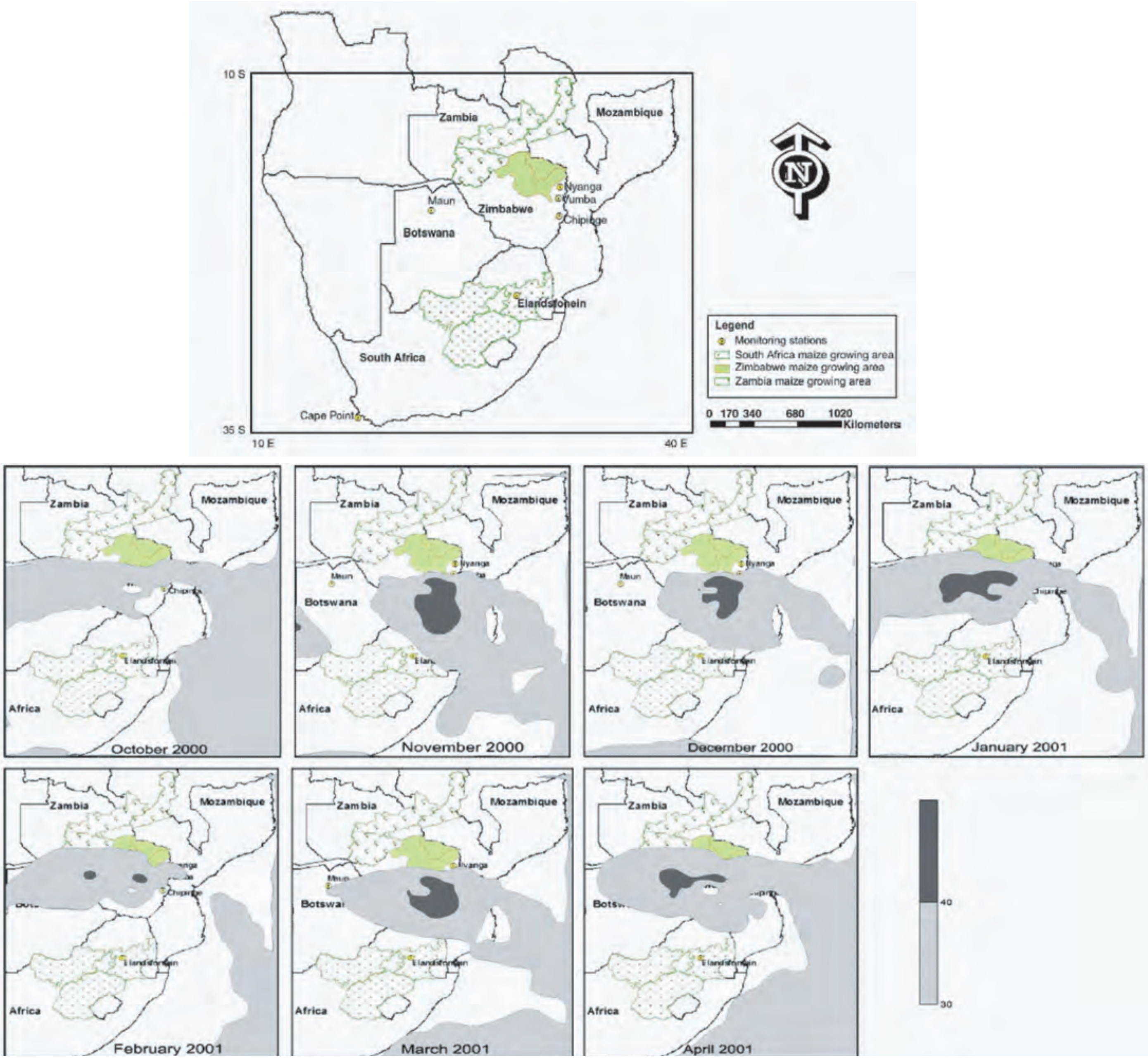

Figure 3.21 | Map of southern Africa showing the modelling domain, maize-growing regions, and locations of O_3-monitoring sites; the series of maps shows average daylight O_3 concentrations in ppb over the main maize-growing area in southern Africa for a five-day period (10th-14th) in each of the months from October 2000 to April 2001. Source: van Tienhoven et al., 2006.

reported in Egypt (Hassan et al., 1995). In parts of Africa, predicted increases in global background concentrations of O_3 combined with trends of increased emissions of O_3 precursors suggest that current and future O_3 impacts on crops and forests in these areas may be significant (Ashmore, 2005). Modelling studies based on 'current legislation' emission projections performed by the Royal Society (2008) suggest that O_3 concentrations are likely to increase in many regions by the year 2050, particularly over landmasses in the tropics, Asia, and Africa, largely because of climate change.

Even under current-day conditions, O_3 may be posing a threat to ecosystems across Africa. For example, photochemical modelling studies have suggested that levels of O_3 concentration that have resulted in impacts in Europe may be occurring in parts of Botswana, South Africa, and Zimbabwe in southern Africa (Figure 3.21). In addition, the maize-growing areas at risk from drought are similar to those modelled to be at risk from elevated O_3 concentrations (USAID, 2002). Therefore, maize may be suffering multiple stresses resulting from O_3 and drought, which may compromise crop productivity. Such stress may be magnified in the future, as increased air pollutant emissions lead to higher and more persistent O_3 concentrations. Additional studies are required to fully understand the potential threat posed by these combinations of stress.

3.2.4.4 Asia

Asia is experiencing rapid population growth. China and India alone account for approximately 2.5 billion of the global population; according to UN estimates (2008), this figure will have risen to over 3 billion by 2030. This population growth has been closely matched by strengthening economies in many of the most rapidly developing Asian nations. For example, over the past five years (2005–2010), China's annual GDP growth rate has been between 9–13%, while India's growth rate has fluctuated between 5–9 % (UN, 2011). This growth has in part been made possible by increased access and utilization of fossil fuels (van Ardenne et al., 1999), as evidenced by the present energy mix for the Centrally Planned Asia and South Asia regions. In these regions, approximately 85% of energy demand is met by fossil fuels, with coal being the principle energy source (summarized from the GEA database).

This heavy reliance on coal and other fossil fuels translates into substantial CO_2 emissions, with over 1600 MtC/year under current day (2005) conditions (as estimated from the GEA database). Such coal use also results in substantial emissions of SO_2. For example, in 2005 China emitted 25.5 $GtSO_2$, reversing a decreasing trend in SO_2 emissions that was evident prior to 2002 (Chan and Yao, 2008); this has implications for acidification across many parts of the country. In addition to increases in CO_2 and SO_2 emissions, NO_x emissions have also increased, due to combustion of fossil and biomass fuels, especially in the transport, industrial, and power-generation sectors (van Ardenne, 1999). The resulting increases in atmospheric NO_x concentrations will hasten environmental degradation caused by acid deposition, eutrophication, and ground-level O_3. The resulting high levels of air pollution across the Asian region are not a new phenomenon; atmospheric emissions have been steadily increasing over the past few decades, as countries in the region have experienced rapid industrialization and economic growth (Ohara et al., 2007). Even with this increase in economic growth in many Asian countries, large sectors of the population remain deprived of basic facilities and amenities. For example, around 500 million people still do not have access to electricity, and many more lack access to clean drinking water and modern energy fuels like liquid petroleum gas (LPG) and kerosene to meet their domestic cooking and lighting needs. This state of affairs has consequences for both indoor and outdoor pollution and human health (WHO, 2006b; for more information, see Chapter 4). Additionally, many poor people are reliant on agriculture for their livelihoods, and hence are also at risk from the threat of air pollutants that reduce the quality and quantity of crops as well as from droughts that are becoming more frequent with the onset of climate change.

Climate Change Environmental Impacts in Asia

The IPCC Fourth Assessment reported the increasing intensity and frequency of extreme weather events in Asia over the last century and into the 21st century (Cruz et al., 2007). It is likely that climate change will impinge on the sustainable development of most of the developing countries of Asia, as it compounds the pressures on natural resources and the environment associated with rapid urbanization, industrialization, and economic development.

Climate change is expected to affect forest expansion and migration, and exacerbate threats to biodiversity resulting from land-use, cover change, and population pressure in most of Asia (Cruz et al., 2007). Increased risk of extinction for many flora and fauna species in Asia is likely as a result of the synergistic effects of climate change and habitat fragmentation. In North Asia, forest growth and a northward shift in the extent of boreal forest is probable; the frequency and extent of forest fires in North Asia may well increase in the future. In southeast Asia, extreme weather events associated with El Niño were reported to be more frequent and intense in the past 20 years (Cruz et al., 2007). Significantly longer heat-wave duration has been observed in many countries of Asia, as indicated by pronounced warming trends and several cases of severe heat-waves. Generally, the frequency of occurrence of more intense rainfall events in many parts of Asia has increased, causing severe floods, landslides, and debris and mud flows, while the number of rainy days and the total annual amount of precipitation has decreased. However, there are reports that the frequency of extreme rainfall in some countries has exhibited a decreasing tendency. Increasing frequency and intensity of droughts in many parts of Asia are attributed largely to a rise in temperature, particularly during the summer and normally drier months, and during ENSO events. Recent studies indicate that the frequency and intensity of tropical cyclones originating in the Pacific have increased over the last few decades. In contrast, cyclones originating from the Bay of Bengal and Arabian Sea have decreased since 1970, but their intensity has increased. Damage caused by intense cyclones originating in both areas has risen significantly, particularly in India, China, Philippines, Japan, Vietnam, Cambodia, Iran, and the Tibetan Plateau.

Marine and coastal ecosystems in Asia may well be affected by sea-level rise and temperature increases (Cruz et al., 2007). Projected sea-level rise is very likely to result in significant losses of coastal ecosystems, and a million or so people along the coasts of south and southeast Asia may be at risk from flooding. Sea-water intrusion due to sea-level rise and declining river runoff is likely to increase the habitat of brackish water fisheries. Coastal inundation could seriously affect the aquaculture industry and infrastructure, particularly in heavily populated megadeltas. The stability of wetlands, mangroves, and coral reefs around Asia is likely to be increasingly threatened. Risk analysis of coral reefs suggests that 24–30% of the reefs in Asia may be lost during the next 10–30 years (Wilkinson, 2004), unless the stresses are removed and relatively large areas are protected.

Rapid thawing of permafrost and a decrease in the depths of frozen soils, due largely to rising temperatures, have threatened many cities and human settlements. They have led to an increase in the frequency of landslides, a degeneration of some forest ecosystems, and an increase in lake-water levels in the permafrost region of Asia (Cruz et al., 2007). In drier parts of Asia, melting glaciers account for over 10% of freshwater supplies. Glaciers in Asia are melting faster in recent years than before, as reported in Central Asia, Western Mongolia, and Northwest China, particularly the Zerafshan glacier, the Abramov glacier, and the glaciers on the Tibetan Plateau (Pu et al., 2004). Mudflows and avalanches have increased as a result of the rapid melting of glaciers, glacial runoff, and the frequency of glacial lake outbursts (WWF, 2005). A recent study in northern Pakistan, however, suggests that glaciers in the Indus Valley region may be expanding, due to increases in winter precipitation over western Himalayas over the past 40 years (Archer and Fowler, 2004).

The production of rice, maize, and wheat during the past few decades has declined in many parts of Asia, due to increasing water stress. This stress can be attributed in part to increasing temperatures, the increasing frequency of El Niño, and a reduction in the number of rainy days. In a study at the International Rice Research Institute, the yield of rice was observed to decrease by 10% for every 1°C increase in growing-season minimum temperature (Peng et al., 2004). A decline in potentially good agricultural land in East Asia and substantial increases in suitable areas and production potentials in currently cultivated land in Central Asia have also been reported (Fischer et al., 2002). Climate change could make it more difficult to step up agricultural production to meet the growing demands in Russia and other developing countries in Asia.

In Asia, water shortages have been attributed to rapid urbanization and industrialization, population growth, and inefficient water use, which are aggravated by changing climate and its adverse impacts on demand, supply, and water quality (Cruz et al., 2007). Overexploitation of groundwater in many countries of Asia has resulted in a drop in its level, leading to ingress of sea water in coastal areas, making the subsurface water saline. India, China, and Bangladesh are especially susceptible to increasing salinity of their groundwater as well as surface water resources, especially along the coast, due to increases in sea-level as a direct impact of global warming. Increasing sea-level by 0.4–1.0 m can induce saltwater intrusion 1–3 km farther inland in the Zhujiang estuary. Increasing frequency and intensity of droughts in the catchment area will lead to more serious and frequent salt-water intrusion in the estuary and thus deteriorate surface and groundwater quality.

Air Quality Environmental Impacts in Asia

Eutrophication

Recent global modeling studies suggest that N deposition related to NO_x and NH_3 emissions are now as high in some parts of south, southeast, and southern East Asia as they are in Europe and North America (Figure 3.22), with much of the NH_3 emission coming from agricultural sources such as cattle rearing. There is growing scientific consensus that eutrophication impacts will be considerable in Asia, although local evidence is only just starting to emerge (Phoenix et al., 2006). For example, in China, manipulation experiments suggest that N deposition has the potential to influence the species richness of the understory of temperate and tropical forests. The UN Convention on Biological Diversity has identified N deposition as an indicator of a threat to biodiversity in many Asian countries. In addition, effects on biodiversity have been linked to the type of forest decline that has occurred in some areas of China. This decline is the result of the direct effects of SO_2, via extremely acidic mist or rain events, and the effects of soil acidification, which is caused by S and N deposition as discussed in more detail in the following sectors.

Acidification

The extensive use of coal as a primary energy source across Asia has led to concerns over environmental degradation resulting from acidification. The main sectors contributing to SO_2 emissions are coal-fired electricity production and industry, which are responsible for approximately 45% and 36%, respectively, of SO_2 emissions (Hicks et al., 2008). However, other pollutants, especially NO_x and NH_3, are also involved in acidification via their transformations in the atmosphere and soil. Acid deposition is mainly a problem in the southern and southwestern regions of East Asia, where neutralization by alkaline desert dust in the atmosphere is relatively low. These acidifying emissions have resulted in parts of East Asia being identified as the third largest acid rain-prone region in the world. The variability in susceptibility to acid rain evident across this East Asian region would appear to exist across the whole of Asia, with modelling studies indicating that soil acidification effects are unlikely to be widespread, due to the insensitivity of soils and high concentrations of alkaline dust in the atmosphere. According to Hicks et al. (2008), areas at risk of acidification effects over the next 50 years are mainly restricted to areas of southern East and southeast Asia (Figure 3.23).

Various attempts have been made to estimate the societal costs of air pollution and acid rain in China. In 2004, the former Chinese State Environmental Protection Administration estimated the economic loss due to negative effects of acid rain on crops to be US$4.6 billion (World Bank, 2007). Although such estimates are highly uncertain, they make clear that the economic benefits of tackling acid rain may be considerable.

Policies to control SO_2 emissions in East Asia began in the early 1990s, when the concept of acid rain control zones was developed as the main framework for setting priorities on acid rain reduction policy (Shi et al., 2008). Mitigation actions focused on establishing emissions standards for coal- and oil-fired power plants, targeting the use of low-sulphur coals, the relocation of coal-fired power plants to rural areas, the installation of de-sulphurization technologies, increased chimney stack height, the closing of small coal-fired generating units and polluting industries, and the shifting to cleaner energy sources (Cofala and Syri, 1998). Intergovernmental cooperation has also been established through the Acid Deposition Monitoring Network in East Asia, or EANET (Tsunehiko et al., 2001). These measures, along with the economic recession of the late 1990s, resulted in significant decreases in SO_2 emissions. However, coal consumption continued to increase in East Asia, largely due to the continued expansion of electricity generation and heat provisioning coal-fired power plants (e.g., Fang et al., 2008).

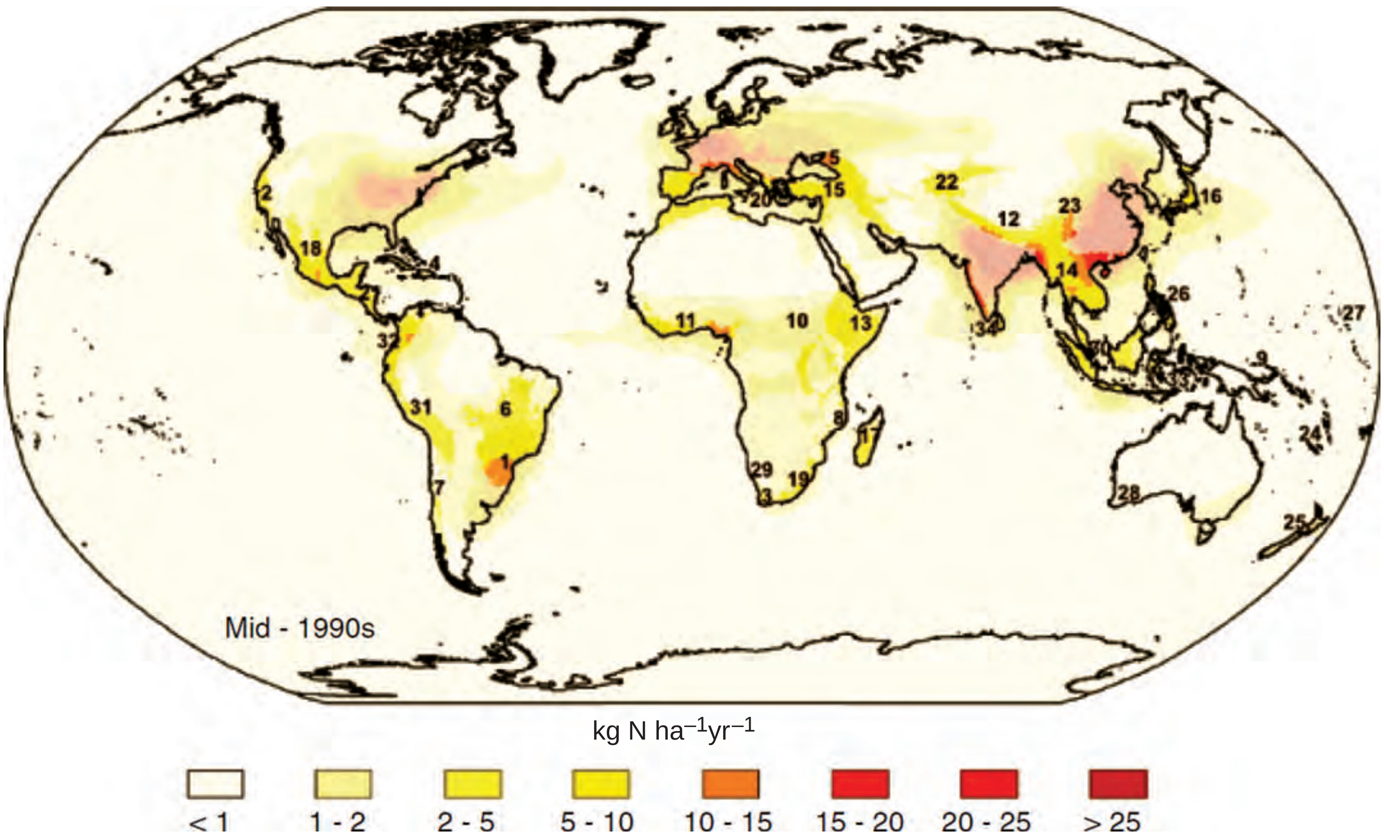

Figure 3.22 | Distribution of nitrogen deposition in relation to biodiversity hotspot areas in the mid-1990s showing high nitrogen deposition values in southern China and parts of south and southeast Asia. Numbers on map identify World biodiversity hotspots. Source: Phoenix et al., 2006.

Ground-level Ozone (O_3) and Food Security

The air pollutant that is arguably of greatest concern due to its occurrence at elevated concentrations across large parts of rural Asia is ground-level O_3 (Emberson et al., 2009). Experimental evidence collected in Asia over the past few decades provides a strong indication of the threat posed by O_3 to agricultural production, with staples such as wheat, rice, and beans commonly showing substantial yield losses of 10–30% under ambient pollution concentrations (Emberson et al., 2009). Modelling studies provide an indication of the magnitude and geographical extent of O_3 risk to crop production. These studies indicate that the Indo-Gangetic Plain, one of the most important agricultural regions in the world, is particularly at risk from high O_3 concentrations. This potentially has important implications for agricultural production (Van Dingenen et al., 2009). For example, it is possible that O_3 may be a significant contributing factor to the yield gap that currently exists across much of Asia. It is also possible that O_3 may have played a role in recent reductions in the growth rate in yield of key staple crops, which have recently been a cause for concern across much of south Asia (Emberson et al., 2009).

Economic loss estimates that have been conducted across Asia suggest substantial effects due to yield losses caused by O_3. Estimates from East Asia suggest annual losses of US$5 billion, based on four key crops – wheat, rice, soybean, and maize (Wang and Mauzerall, 2004). A global modelling study that translated production losses (see Figure 3.24, which shows global production loss estimates for wheat) into total global economic damage for the same four commodities, using world market prices for the year 2000, estimated an economic loss of US$16–30 billion per year. About 40% of this damage was found to occur in parts of China and India. For those Asian countries with economies largely based on agriculture, the O_3-induced damage was estimated to offset a significant portion (20–80%) of the increase in GDP in the year 2000 (Van Dingenen et al., 2009).

These modelling studies have relied on North American or European dose-response relationships to assess the yield losses caused by O_3, since equivalent Asian relationships do not exist. However, recent comparisons of Asian and North American O_3-response data strongly suggest that Asian crops (in this instance wheat, rice, and beans) and cultivars may well be more sensitive to O_3 concentrations when growing under Asian conditions (i.e., under Asian climates, on Asian soils, and under Asian management practices). This implies that current economic loss estimates may have substantially underestimated the yield, and subsequent production losses, in the region (Emberson et al., 2009).

The indirect effects that air pollutants may have on the climate system should also be considered within the context of agriculture. Work that

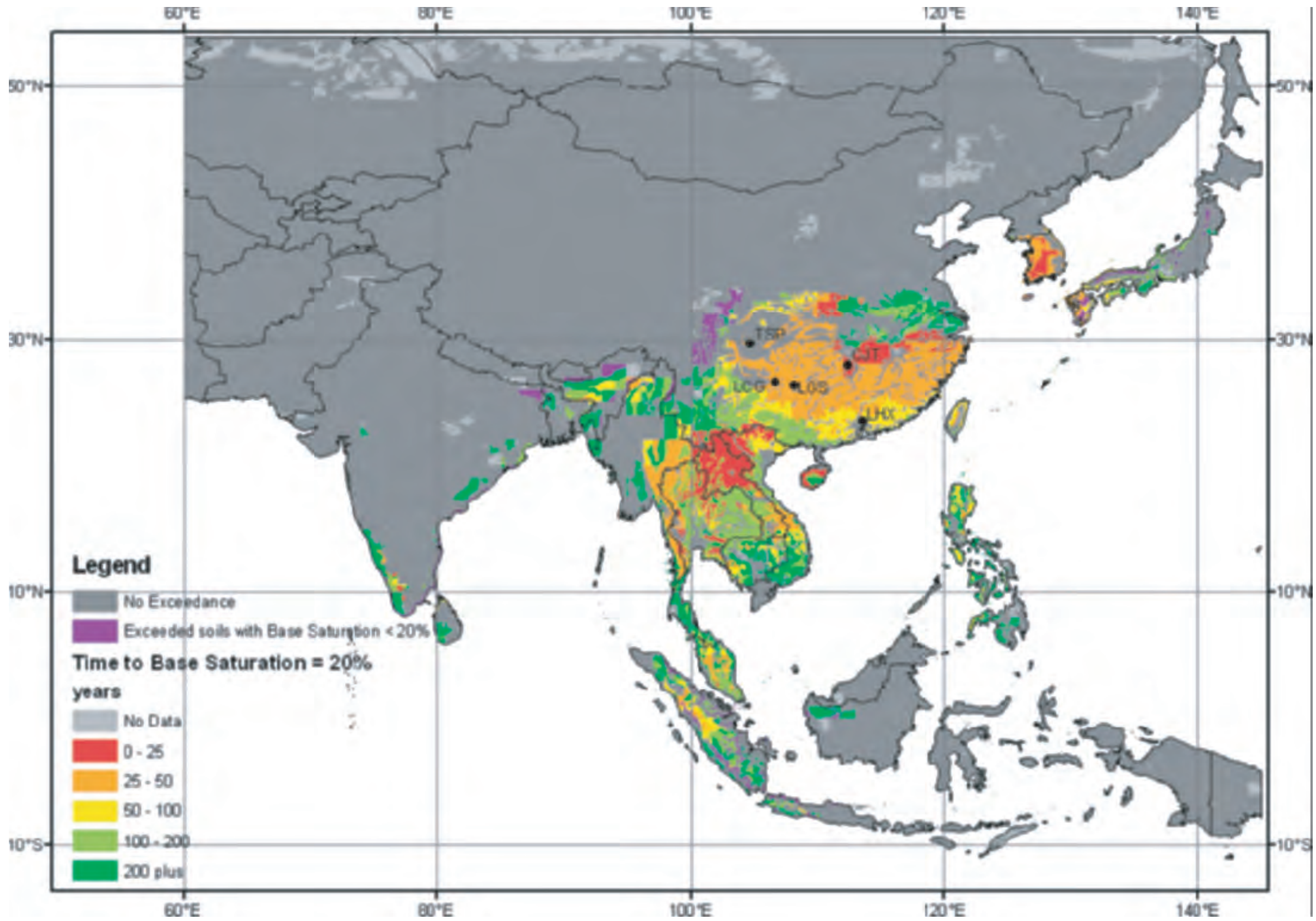

Figure 3.23 | Time development of soil acidification damage according to a modelling study for Asia using the best available data for soil and deposition parameters and deposition estimates obtained using a more pessimistic emission scenario (IPCC SRES A2) for 2030. The model calculates the time it takes for the neutralizing capacity of the soil (expressed as base saturation) to be reduced to a level where acidification effects are observed (i.e., approximately 20% base saturation). Source: Hicks et al., 2008.

has focused on understanding the role of ABCs on climate provides a particularly good example of such effects. The surface dimming (reduction in sunlight reaching the Earth's surface) associated with ABCs, described earlier in this chapter, is thought to impact agricultural yields in Asia, with estimates suggesting that 70% of the crops grown in China may have their yields suppressed by 5–30%. In addition, yields may be impacted through alterations to monsoon rainfall (UNEP, 2008). In particular, BC may play an important role in causing monsoonal shifts, as it dims the surface and warms the lower atmosphere, producing effects on the vertical atmospheric temperature profile, evaporation, atmospheric stability, and strength of convection. These changes in precipitation will also impact agricultural management and productivity across the region.

3.2.4.5 Polar Regions

The Earth's polar regions, the areas of the globe surrounding the North and South Pole, are dominated by polar ice caps. The fact that both of these polar regions are so remote from large-scale industrial activities might suggest that they would remain as pristine environmental strongholds. However, over recent decades the particular sensitivity of these regions to environmental degradation and global environmental change has been striking. Examples of such degradation have involved the global-scale transport and accumulation of persistent organic pollutants (POPs) within ecosystems, the depletion of the stratospheric O_3 layer, the transport of radionuclide material and, most recently, observations of the occurrence of enhanced climate change above the global mean. In addition, the lack of obvious sovereignty of the Arctic region, coupled with this region's wealth of natural resources, threatens to lead to further exploitation of a region that is unprotected by international treaty. Here, the potential of the Arctic as a supply of fossil fuels and the increased demand for energy is of particular relevance. These issues are discussed in the following sections to highlight the possible consequences of unrestrained energy supply and demand for these sensitive regions.

The Sensitivity of the Polar Regions

In recent decades, the particular sensitivity of the polar regions has become worryingly apparent. One of the best examples of this is the

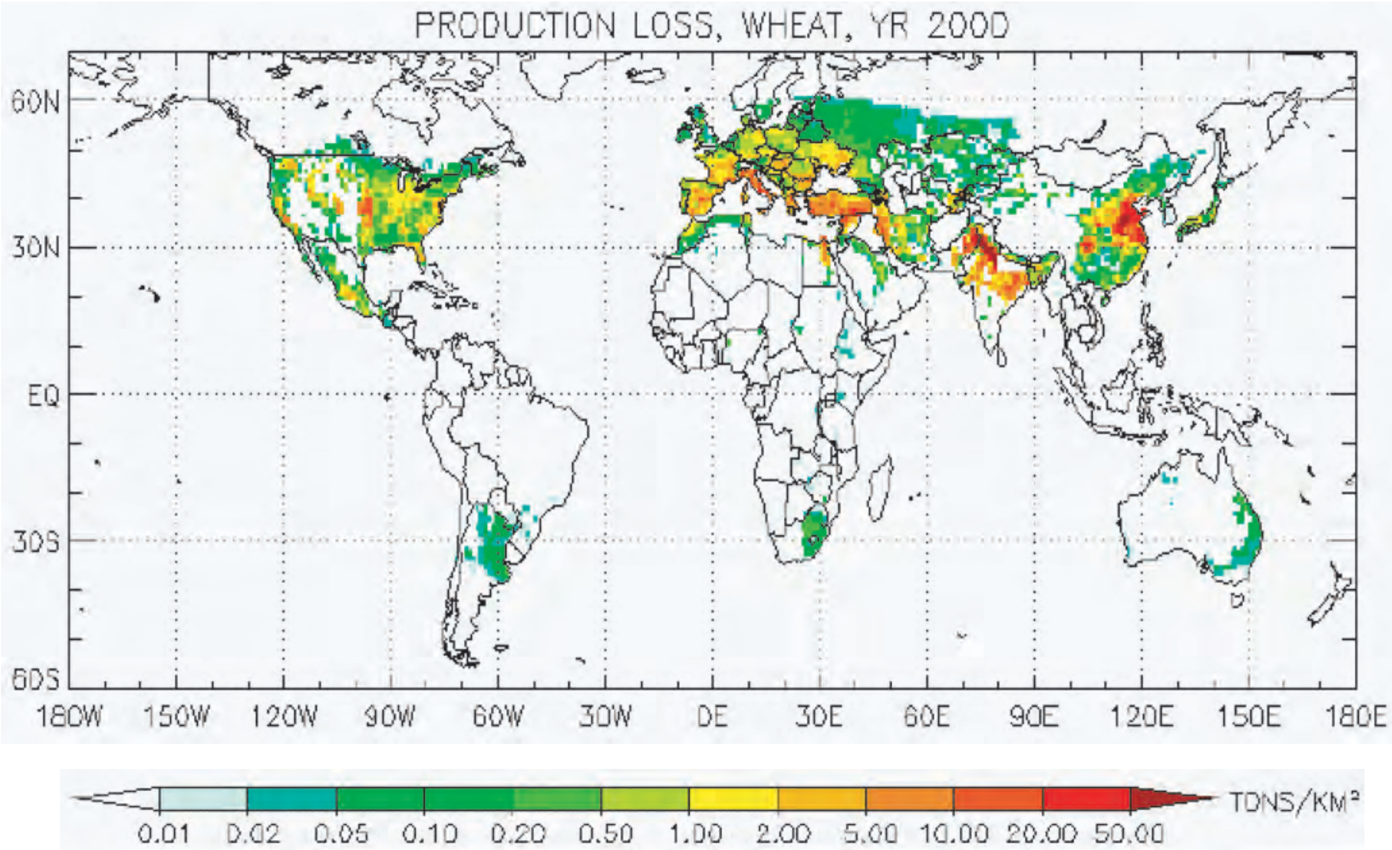

Figure 3.24 | Average wheat crop production losses due to O_3 estimated for the year 2000 using European and North American concentration-based exposure-response relationships. Source: Van Dingenen et al., 2009.

bioaccumulation and biomagnification of POPs in species and ecosystems of these regions. These processes refer to the accumulation of certain long-lived substances such as pesticides or heavy metals as they move up the food chain through a combination of ecosystem contamination and predation, concentrating the substances in tissues or internal organs. The first evidence of the occurrence of these processes was the measurement of the pesticide DDT and its derivatives in Adélie penguins in Antarctica in the 1960s (Sladen et al., 1964). This provided one of the first demonstrations of how human-made substances could spread and affect ecosystems on a global scale. The polar regions are particularly vulnerable to such processes, since their cold climate prevents the breakdown of contaminants, exacerbating their effects.

Another example of polar region sensitivity is that posed by the potential of ozone-depleting substances (ODS) to deplete the stratospheric O_3 layer, which was first recognized in 1974 (Molina and Rowland, 1974). Stratospheric O_3 depletion is important, because the O_3 layer in the stratosphere keeps 95–99% of the sun's ultraviolet-B radiation (UV-B) from striking the Earth. Consequences of increased levels of UV-B include genetic damage, eye damage, and damage to marine life. A decade after identification of the problem in the mid-1980s, the discovery of the O_3 hole over Antarctica (Farman et al., 1985) and the subsequent attribution to ODS (Soloman et al., 1986) further heightened concern for the O_3 layer. This scientific evidence resulted in the establishment of the Montreal Protocol in 1987, which has led to successful restrictions on ODS. These restrictions have meant that net ODS production, consumption, and subsequent emissions have decreased. However, the long atmospheric lifetimes of many ODSs, particularly the chlorofluoro carbons, mean that the mixing ratios of ODS in the stratosphere are decreasing relatively slowly (Velders et al., 2007). Once again, the particular vulnerability of the polar regions is a consequence of their climatic characteristics. Atmospheric circulation patterns, coupled with the absence of solar radiation during the polar winter, allow the buildup of reservoirs of chemical species in the atmosphere. The reintroduction of solar radiation and the concurrent increases in temperature kick-start the photochemical processes that are able to act on the large reservoir of chemicals, allowing rapid destruction of the stratospheric O_3 layer.

Another example of polar sensitivity is that of radionuclide contamination. The Arctic Monitoring and Assessment Programme (AMAP, 2010) reported radionuclide transport to the Arctic, which could be traced back to the nuclear reprocessing facility at Sellafield in the United Kingdom. Discharges of radioactive substances such as Technetium-99 (^{99}Tc), a long-lived product of uranium fission, were found to have been transported by the Atlantic currents to the Norwegian Sea, continuing northwards along the Norwegian coastline to the Barents Sea by the

Norwegian Coastal Current. The Sellafield-to-Barents Sea transport is estimated to take four to five years. In an attempt to avoid similar contamination in the southern polar region, the Antarctic Treaty has declared the region a nuclear-free zone, where nuclear explosions and radioactive waste disposal are forbidden.

The fact that contaminants in water, air, and ecosystems that originate from outside the polar regions can be observed and measured within the region is a critical reminder of the impossibility of confining emissions and their impacts to source regions, regardless of the remoteness of the impacted environment. This points to the need to consider environmental change, and the limits to such change, at the planetary level. In recent years the particular threat posed to the polar regions by climate change has been recognized, as once again these regions seem particularly sensitive when compared to the global mean. The following section describes climate change in the polar regions in more detail, highlighting the physical, socioeconomic, and political considerations that are particularly relevant when considering the sustainability of future global energy systems.

Climate Change in the Polar Regions

Arctic mean temperatures were found to be rising at almost twice the rate of the rest of the Earth. Many parts of the Arctic have seen warming of 2–3°C in the last 50 years alone. In comparison, over the last century GMTs have increased by only 0.6°C (ACIA, 2004).

By 2100, climatologists project an additional mean temperature increase of 5–7°C, with regions closest to the North Pole projected to reach 10–12°C above their current temperatures in the winter months (ACIA, 2004).

Similarly, a number of stations in the Antarctic Peninsula have experienced a major warming over the last 50 years, with temperatures at Faraday/Vernadsky station having increased at a statistically significant rate of 0.56°C/decade over the year and 1.09°C/decade during the winter. Overlapping 30-year-trends of annual mean temperatures also indicate a strong tendency for the warming trend to be greater during the 1961–1990 period compared with 1971–2000 (Turner et al., 2005).

This 'polar amplification,' whereby the rate of regional temperature change far outpaces global changes, is related to a powerful set of thermodynamic and dynamic factors. A driver of this phenomenon is the ice-albedo feedback mechanism, whereby the melting of sea ice in the Arctic caused by warming associated with climate change exposes dark open water or melt ponds on top of ice. This can decrease the albedo (reflectivity of the surface) to between 0.2 for dark melt ponds to 0.6–0.7 for melting multiyear ice, with such effects occurring during the summer, when warming is most acute. In comparison, the albedo of unaffected multiyear ice is high at 0.8–0.9 (Perovich, 2002). The mechanism is a 'positive feedback,' since the absorbed energy resulting from the lower albedo raises the sea and air temperatures, thereby melting even more ice and slowing down the winter refreeze. A phenomenon known as the Arctic haze (see next section) will also contribute to this enhanced warming of the North Polar Region.

Quantitative assessments of the extent of sea-ice melt suggest that sea-ice in September 2007 was half the typical ice extent in the same month between the 1950s-1970s (Stroeve et al., 2008). This area was 23% smaller than the previous ice extent minimum reached in 2005. Arctic ice also has an increased tendency to be younger and weaker. For example, while in 1987 ice older than five years comprised 57% of the central Arctic basin, now it has all but disappeared from this area (Maslanik et al., 2007). In addition, ice thickness has declined by 25% over the last two decades. This appears to be a critical parameter determining sea-ice extent, with evidence suggesting that a threshold may exist below which summer melt potential is greatly enhanced (Holland et al., 2006). The rapid melt of sea ice that has been observed in the Arctic has led some experts to predict that a 'tipping point' will be crossed, resulting in a seasonally ice-free Arctic. Based on observations of the 2007 ice melt, Stroeve et al. (2008) revised earlier projections, suggesting that rapid ice decays in Arctic sea-ice, caused by increases in ocean heat transport to the region, could, with a high degree of probability, realize a seasonally ice-free Arctic Ocean as early as 2030.

The Arctic Haze

The Arctic haze, a visible reddish-brown haze found in the atmosphere at high latitudes, results from the long-range transport of air pollution to the polar regions. This pollution remains in the polar atmosphere due to the slow rate of removal of pollutants in the cold polar air (Shaw, 1995). It was first observed in the 1950s by flight crews on weather reconnaissance missions, but only became public knowledge when Raatz (1984) reanalyzed and published data from these missions (known as the Ptarmigan flights). This reanalysis found that the most frequent reports of the Arctic haze occurred at about 75–80°N during late winter, under clear-sky conditions (Shaw, 1995).

Practically all pollution contributing to the Arctic haze originates from more southerly latitudes (Law and Stohl, 2007); with local pollution sources being small and limited to areas near the Arctic circle. These local pollution sources include volcanic emissions; anthropogenic emissions from urban centers, e.g., Murmansk; and industrial emissions, particularly in the northern parts of Russia, and emissions from the oil industry and shipping (AMAP, 2006). In terms of the long-range transported pollutants, Eurasian emissions are more important than those from North America, due to the closer proximity of Eurasian sources, atmospheric circulation patterns, and the extended distribution of the haze over Eurasia (Law and Stohl, 2007). Some studies have even suggested that Asian dust may be contributing to the Alaskan Arctic haze (Rahn et al., 1981), though this claim has been disputed (Stohl, 2006).

The Arctic haze is important, since it changes the short- and long-term radiation balance in the Arctic. Because large areas are affected, this can have significant effects, even though the concentrations of aerosols found in the Arctic are approximately an order of magnitude lower than those found in polluted and industrial locations (AMAP, 2006). These aerosols can also affect the physical properties of clouds, which will affect the radiation balance (Garrett et al., 2002). Black carbon will also cause heating in the haze layers (Quinn et al., 2007) and, on deposition to snow and ice, will result in a reduction in surface albedo. The resulting warming may lead to the melting of ice and may be contributing to earlier snowmelts on tundra in Siberia, Alaska, Canada, and Scandinavia (Foster et al., 1992).

In terms of future trends for this pollutant haze, there is evidence that ship traffic is already affecting the summertime Arctic atmosphere; increases have been seen in the deposition of BC from increased shipping following reductions in summer sea-ice (Corbett et al., 2010). The warming of northern latitudes could also cause boreal forest fires to become more frequent, thus increasing pollution transport to the Arctic (Stocks et al., 1998). Finally, the polar dome, which currently presents a barrier to pollution transport in the Arctic, may weaken in the future, as the Arctic continues to warm relatively faster than lower latitudes, allowing more efficient pollution import into the Arctic (Eckhardt et al., 2003).

Energy Issues in the Arctic

The physical changes in the Arctic environment will cause substantial impacts on the regions human populations and ecosystems. However, perhaps more worrying, at least from the point of view of enhancement of the level and rate with which impacts are occurring in the Arctic, are the economic and political implications of these physical changes. Importantly, the physical feedback of ice-melt has realized the potential for transport routes (with the apparent opening of the North West Passage for ice-free navigation which would substantially reduce shipping distance from Europe to Asia) and hence improved accessibility to energy sources (Powell, 2008). Johnson (2010) describes this situation as 'accumulation by degradation' where the previous combustion of fossil fuels generates geophysical changes that are exploited in order to identify, extract, and combust additional hydrocarbons that fuel development and lead to future CO_2 emissions and additional climate change; this process will lead to self-amplifying feedbacks of the geophysical world.

Fuelling the likelihood of such a feedback process are the claims of undiscovered hydrocarbon reserves in the Arctic region. There are estimates that as much as 25% of the World's undiscovered oil and gas lie in the Arctic (AMAP, 2007) though this figure has been contested (Powell, 2010). The US Geological Survey recently completed its first probabilistic assessment of all undiscovered oil and gas deposits of the Arctic Circle. It estimated that of the world's undiscovered resources the region holds 30% of natural gas, 20% of natural gas liquids and 13% of oil (US GS, 2008).

The potential hydrocarbon gains, coupled with climate change-driven ice melt, could substantially reduce the costs associated with hydrocarbon exploration and exploitation, including deep-water exploration and the transport of cargo, drilling equipment, and petroleum, and extend the seasonal period during which such operations are feasible. These activities themselves will cause environmental degradation of the region by increasing the likelihood of oil spills and pollution (Casper et al., 2009).

3.3 Land-use and Energy Systems

Renewable energy sources often require more land than the fossil energy sources that they replace, which means that making energy systems more sustainable can place additional pressures on land-use. The transition towards sustainable energy thus tends to increase land pressures due in part to the fact that current energy systems to some extent substitute nonrenewable resources – fossil and nuclear – for land. It is clear that a critical component of a sustainable energy path – and indeed the sustainable use of natural resources in general – is to use land more effectively in the future, with the recognition that some of that land will have to substitute for the fossil resources that will no longer be available at acceptable levels of economic and environmental costs. Renewable resources that occur on water (e.g., tidal power) or land that is difficult to use for other human needs (e.g., mountain tops) will naturally also become more valuable in the sustainability transition.

In this section, we provide a brief overview on the *quantity* of land associated with energy systems, while of course recognizing that the quality of land-use is just as important, if not more important, than the quantity *per se.* The reason for focusing on the quantity of land required is to provide some rough idea of how priorities might be set where land-use pressures are a key concern. Some explanation and/or examples are also given here, in instances where there are significant qualitative land-use impacts that are obscured by simple quantitative comparisons.

3.3.1 Land-use Changes

Land is essentially always transformed in the initial establishment of any energy system. This transformation can be significant, in that the effects are not easily reversed. In the case of coal, hundreds of years may be needed for recovery or reclamation from the toxic effects of coal mining. In the case of nuclear power, the consequences of a serious accident result in land-use transformation that requires tens of thousands of years for recovery. Consequently, changes in land-use from toxic, nonrenewable resource uses should not be directly compared to land-use changes associated with biological or natural resources.

Indeed, the term 'land-use change' has a special meaning within the context of natural resources and climate, in that it refers to the transition from one type of land-use such as forestry to another, such as agriculture. Although also representing a transformation and involving energy use and GHG emissions, these land-use changes are qualitatively quite different from fossil or nuclear land-use impacts, in that there are generally no toxic effects and the land could potentially be converted back

Table 3.2 | Land-use intensity (LUIs) for various electric power production systems.

Type	Reference capacity (for energy conversion)	Land-use intensity for installed power (km^2/GW)		Land-use intensity for electricity generation (km^2/TWh/yr)		Assumptions and/or type of impact
		low	high	low	high	
Coal	85%	18.6	126.6	2.5	17	Area disturbed by mining
Hydropower	44%	62.2	354.8	16.1	92.1	Area submerged by lake
Biomass electricity	75%	2844	4294	432.9	653.6	Area for growing woody feedstock (willow)
Geothermal	85%	7.5	103.6	1.0	13.9	Area covered by plant and access infrastructure, fragmented habitat
Solar thermal	29%	25.9	51.8	10.2	20.4	Area covered by plant and access infrastructure, fragmented habitat
Solar photovoltaic (PV)	28%	51.8	129.5	21.1	52.8	Area covered by plant and access infrastructure, fragmented habitat
Onshore wind	35%	199.4	242.8	65.0	79.2	Area covered by turbine and access infrastructure, fragmented habitat
Nuclear	91%	3.02	4.78	1.9	2.8	Area covered by plant, as well as area for uranium mining and waste storage

Source: adapted from McDonald et al., 2009.

to its previous use or to some other use, especially if the previous uses were agricultural. In this respect, the impacts of the sheer quantity of land-use change associated with agriculture or bioenergy are tempered somewhat by the fact that they can be reversed in some cases without extensive damage. However, there can also be serious and irreversible losses when land of high carbon stock or land with high biodiversity value is converted for food, fiber, energy, or other commercial uses. The complexity and dynamic nature of land-use change places it beyond the scope of this brief review, which focuses mainly on the quantity of land required by energy systems.

3.3.2 Land-use Intensity

Land-use intensity (LUI) is a static measure of the land needed for continuous operation and that which is unavailable or otherwise impacted with respect to other uses. The LUIs of energy systems vary by several orders of magnitude. Nuclear and fossil fuels have fairly low land-use intensities, while renewable energy systems exhibit much more variation, ranging from modest requirements for geothermal and solar photovoltaics (PVs) to extremely high requirements for some types of biomass to electricity systems. The high value for biomass electricity is also partly due to the fact that biomass is most efficiently used and optimized for heat production in combination with electricity.

The land-use intensity of energy systems provides some first-order indication of the types of land-use pressures associated with the transition to sustainable energy systems. Land-use impacts are based on the amount of land that is rendered unavailable for other uses for at least the duration of the lifetime of the energy systems. It is important to note that such calculations include only the energy system in use and do not include land-use impacts due to accidents, which could be quite serious in the case of nuclear power and could contaminate land for thousands of years. Accidents in most renewable energy facilities would generally not be serious, as there are very few potentially toxic elements and those that are used could not spread over large areas.

A representative comparison of LUIs with respect to installed power and electricity production in the United States is given in Table 3.2. The systems with the lowest LUI are nuclear, coal, and geothermal. Solar and wind have LUIs that are 1 order of magnitude higher, while biomass has LUIs that are 2 orders of magnitude higher. However, since biomass energy systems normally have coproducts and since the land can often accommodate other uses, the effect of the land impacts can be significantly less. In the case of geothermal energy, there are underground impacts that are not included. There are also some similarities in the land impacts of geothermal energy with those of oil and natural gas. In the case of geothermal energy supply that involve wells, only 5% of the associated impacts are from direct land-use effects; the remaining 95% are due to habitat fragmentation and species-avoidance behavior. Onshore wind has a comparable breakdown, in that 95–97% of the impacts are not due to the land occupied, but rather to fragmentation and the effects on birdlife (McDonald et al., 2009).

3.3.3 Power Density

Another way to compare energy use in relation to area is to use the power density, expressed in watts per square meter (W/m^2) and representing the incident energy that can be delivered. Incident solar energy is typically in the range of 100–200 W/m^2 in Europe, but is much higher in the lower latitudes. The highest delivery of high-quality (electric) renewable energy per unit of land is likely to be from solar PV,

Table 3.3 | Land-use intensity and nitrogen intensity of various energy crops used for biofuel production.

	Land-use intensity (LUI)		Nitrogen intensity		Combined weighted ranking
	(ha/1000 GJ)	rank	(kg N/1000 GJ)	rank	
Sugarcane	2.3	2	110	2	0.02
Willow	5.3	6	90	1	0.03
Miscanthus	4.2	5	210	5	0.03
Sugar beet	1.9	2	460	8	0.03
Oil palm	3.0	4	440	8	0.04
Birch	6.8	8	160	3	0.04
Poplar	7.2	8	160	3	0.04
Switchgrass	6.5	8	300	6	0.05
Corn	4.9	6	490	8	0.05
Sweet sorghum	6.1	8	390	7	0.05
Algae	0.3	1	1100	11	0.05
Grain sorghum	16.2	12	1000	11	0.13
Rapeseed	16.5	12	1400	12	0.15
Soybean	20.2	13	3900	13	0.30

Source: Miller, 2009.

which could capture in the order of 10–20% or roughly 10–40 W/m^2 for European conditions when placed on south-facing surfaces. This might be compared with other energy systems in terms of how much electricity they can deliver, such as wind (2 W/m^2) or tidal (3 W/m^2), again based on conditions common in Europe (MacKay, 2008). The calculation of power density is somewhat more location-specific than the LUI measure used in Table 3.2, because it is based mainly on the energy that can be instantaneously extracted, whereas the LUI includes various land impacts that are then averaged across the power or energy that is available.

3.3.4 Biofuels, Land-use, and Nutrient Demand

The land-use associated with energy crops that are used to make liquid biofuels differ qualitatively from other types of energy supply in that they involve agricultural operations, and therefore other types of land issues arise. A particularly useful metric that can be compared alongside LUI is the nitrogen requirement; by combining this measure with LUI, one obtains a much better indicator for sustainability. Table 3.3 provides an example of such analysis, in which a combined ranking for two criteria was made, with equal weighting given to LUI and nitrogen-use intensity (Miller, 2009). An obvious conclusion from this comparison is that soybean, rapeseed, and grain sorghum are highly inefficient crops for biofuel production compared to almost any other option. Sugarcane scores highest in the initial ranking, and this held true even when sensitivity analysis was conducted for key parameters such as higher heating value (HHV), nitrogen and harvestable yield (Miller, 2009).

3.3.5 Multiple Uses, Hybrid Systems, and Land-use Efficiency

As the demand for food, feed, fiber, energy, and other products requiring significant land-use increases, greater land-use competition will be expected. Furthermore, since land is also valued for recreation, conservation, biodiversity, and many other uses, the 'productive' capacity of land must also be weighed against ecosystem services that are in fact highly productive, but whose value is poorly reflected in our socioeconomic systems. Consequently, the rise in importance of bioenergy has raised concerns that other valuable land-uses will suffer as a result of bioenergy expansion. One approach to reducing the LUI, while also improving the system design, can be hybrid systems that combine biomass with intermittent renewables such as wind. One example is a hybrid wind/biomass system that uses compressed air storage, which was estimated to require 70 m^2/MWh, a LUI that is just a fraction of the range given for biomass in Table 3.2 (Denholm, 2006).

The attention paid to bioenergy systems has also led to increased scrutiny of the land-use efficiency of agricultural and livestock practices. Greater land-use efficiency in agriculture, forestry, and other land-uses could free up some land for the renewable energy options that will be ultimately required to replace fossil and nuclear fuels. Current agricultural practices have tended to be land-intensive, as the low price of rural land in combination with fossil fuel infrastructure led to continual expansion. With precision farming methods and low tillage methods, land can be used more effectively while also reducing GHG impacts from fertilizer use, N_2O, and soil carbon release; this is true regardless of whether the land is used for food, feed, or fuel (Fischer et al., 2010).

Table 3.4 | Principal water uses for various energy systems.

Process requiring water	Type of energy produced	Environmental impacts	Energy eq. produced
Biomass growth	Traditional heat/cooking	Consumptive: less quantity of water for other uses, ecosystem services	36 EJ (traditional)
	Commercial biomass: heat, electricity biofuels	Non-consumptive: water quantity used released back into flow system, sometimes with degraded quality	9 EJ (commercial)
Direct energy generation	Hydropower	Non-consumptive, but can have large impacts on upstream and downstream habitats, depending on water flows affected by hydropower dams Consumptive: (unproductive) evaporation losses from dams, reservoirs	20% of total global electricity
Direct energy generation	Wave and tidal power (sea water)	Non-consumptive	
Energy processing	Thermal power	Non-consumptive	
Energy processing	Fossil fuel and nuclear power stations	Non-consumptive: for cooling; release of cooling water increases local habitat water temperatures	

Source: IEA, 2008.

3.4 Water Resources and Energy Systems

Water resources and the flow of freshwater through the biosphere are fundamental to supplying benefits to society, including various forms of energy. Fresh water is a finite global resource, with no substitute in biological processes. On a planetary scale, water stress and water pollution are already reaching alarming levels, though currently concentrated in local and regional 'hot spots.' With increasing demand for energy, the competition for appropriating more water is expected to increase, in particular for hydropower and for the production of bioenergy crops. Mining activities also have implications for water resources, as they can cause pollution and change local water flows with implications for groundwater reservoirs. Impacts on the quality and quantity of water resources often relate to local social, economic, and environmental settings. The decisions on how best to develop energy system strategies and investments therefore need to be made at all levels, from local to national, in a transparent manner and according to best-informed practices.

3.4.1 Water Resources in Energy Systems: Demand and Impacts

Unlike the atmospheric impacts of different energy systems, which can be considered globally or regionally, the impacts of energy systems on water resources are more meaningfully considered at smaller spatial scales, from local to national levels. This is due to the nature of water resources and the boundaries of water flows. The impacts of various energy systems on water resources must be considered in terms of quality as well as quantity, at local and regional scales, in landscapes, catchments, or river basins. There are large differences in the availability of water resources among regions in the world, with ample freshwater in temperate cold regions and humid tropical regions contrasted by water scarcity in arid, semiarid, and sub-humid tropical regions. As a result, assessments of aggregate water resource availability at the global level may exaggerate actual access to water resources in societies and for specific ecosystem purposes.

Water has three key uses in energy systems (see Table 3.4). The first use is biomass production (rainfed and irrigated), including crops, plantations, and seminatural vegetation types, including forests and grasslands. The second use is processing (e.g., sugarcane, coal washing) and cooling to produce energy (e.g., fossil fuel and nuclear power stations). The third use is generating energy (e.g., hydropower, wave power, and tidal power).

3.4.2 Water Embedded in Biomass Production for Energy

A large part of the energy stemming from biomass is used for primary human energy needs such as heating and cooking. Only a small part of biomass is used for secondary energy supply, involving substantial post-harvest processing, sometimes including additional water uses.

All biomass production, be it for crops or primary biomass production for energy, is associated with large consumptive use (water that does not return to the river or groundwater) of freshwater. Biomass production is the single largest freshwater-consuming process on Earth, with large volumes of water used to produce energy (e.g., a range of 60–110 m^3 per gigajoule (GJ), for major bioenergy crops), according to Geerben-Leenes et al. (2009).

The use of wood fuel for basic human needs of heating and cooking will continue to remain dependent on local resources, and thus the water input will depend on the local hydrological conditions prevailing at the source of production, which in this instance will be the same as the site of use and energy supply. Change in energy use, for example cooking facilities with better wood fuel energy conversion, can be translated into a 'water saving,' as less embedded water is used to

Table 3.5 | Water productivity (as actual evapotranspiration per GJ output of electricity for biofuel).

Crop	Biofuel	Energy water footprint (as tonne water per GJ output)
Rapeseed	Biodiesel	100–175
Sugarcane	Ethanol	37–155
Sugar beet	Ethanol	71–188
Corn	Ethanol	73–346
Wheat	Ethanol	40–351
Lignocellulosic crops, including salix, eucalyptus, switchgrass, miscanthus	Ethanol	11–171
Lignocellulosic crops, including salix, eucalyptus, switchgrass, miscanthus	Methanol	10–137
Lignocellulosic crops, including salix, eucalyptus, switchgrass, miscanthus	Hydrogen	10–124
Lignocellulosic crops, including salix, eucalyptus, switchgrass, miscanthus	Electricity	13–195

Source: adapted after Berndes, 2008.

produce the same amount of energy. Although 'water footprints' are a common way of accounting for water use embedded in biomass produce, care should be taken, as water-use efficiency values vary globally and locally, depending on production systems and agroclimatic conditions for the same species. Also, the calculations vary in regard to how the water consumption is related to the supply of energy. For example, is the full crop used for energy supply, or are only crop by-products used for energy supply? A synthesis by Berndes (2008) show large variations in embedded water in different types of energy systems (Table 3.5). Thus, good planning both in terms of location and type of energy can have large local impacts on landscape and river basin water use efficiency.

Practices that will cause changes in land-use, e.g., shifting biomass production systems, are likely to lead to changes in landscape hydrology, which in turn can affect the downstream availability of water. Bioenergy production upstream may thus impact on water-dependent provisioning and supporting or regulating ecosystem services downstream in a catchment or river basin. A meta-analysis of catchment land-use changes shows that transitions from annual crop and/or grassland to more water-demanding woody species can reduce stream flow (e.g., Jackson et al., 2005; Locatelli and Vignola, 2009). Other crops and habitats that depend on certain volumes and the timeliness of river flows may be impacted, with subsequent implications for livelihoods and economies depending on these ecosystem services. These changes in water flow, caused by changes in crop production for bioenergy or other land-use changes, may not be evident at the outset and may be difficult to anticipate (e.g., Gordon et al., 2008).

3.4.3 Impacts on Water Resources by Post-harvest Processing Biomass for Biofuel

Crops (annual or perennial) that are produced for commercial purposes and eventually processed into energy as heat, electricity, or biofuels, may have a production and processing location that is different from the location of consumption. Thus, over extraction or pollution of water resources due to such commercial biomass production or to post-harvest processing is not necessarily co-located with the place of consumption of the end product. Two water-related issues, in addition to the previous discussion about water footprints, have also been documented: degradation of surface and groundwater by agrochemicals, and post-processing of biomass for bioenergy.

Degradation of surface and groundwater by agrochemicals: As with other intensive crop-production systems, biofuel crops have also been associated with pollution due to the intensive use of agrochemicals such as fertilizers, pesticides, and herbicides. These different chemical components have been shown to cause harm to both surface and groundwater resources in areas of intensive, large-scale use, such as maize (corn) in the midwestern United States for ethanol (e.g., Donner and Kucharik, 2008), and sugarcane in various parts of Brazil (e.g., Goldemberg et al., 2008).

A second pressure on water resources can be associated with post-processing of biomass for bioenergy. This water use is non-consumptive, and the water used will eventually be released back into the landscape. However, as has been seen, for example in association with the production of large amounts of sugarcane, the release can be associated with degraded quality, thus affecting downstream users and habitats.

3.4.4 Impacts on Water Resources by Utilization of Water for Cooling

In various energy systems, such as fossil fuelled power stations and nuclear power stations, water is used for cooling purposes. This use is non-consumptive, and water can be used downstream for other purposes. However, local impacts on water-related habitats have been identified, as the returned cooling water has a higher temperature than natural temperature cycles, especially during cold seasons. This can alter local habitats and species (e.g., Teixeira et al., 2009; Yi-Li et al., 2009; Svensson and Wigren-Svensson, 1992).

3.4.5 Impacts on Water Resources by Hydropower Generation

A major impact on global freshwater flows is the construction of dams and reservoirs for hydropower generation. Thus the environmental impacts, in terms of the hydropower energy chain, are almost entirely due to infrastructure construction activities. Although the energy supply is non-consumptive in terms of landscape water flows, several water-related impacts are associated with hydropower generation. The key ecosystem services that are affected by dams are: the change of flow patterns in time and space at site and downstream; the change of aquatic and riparian habitats; and river-system fragmentation.

The most important impact is the fragmentation of river systems (e. g., Nilsson et al., 2005). The construction of large dams and reservoirs change temporal water flow distribution in river systems, affecting downstream habitats and water quality. This has far-reaching impacts on various ecosystem services that benefit human activities, economies, and development. Although the World Commission on Dams (2000) estimated 40–80 million people to be affected by dam constructions, the numbers of people impacted downstream of dams are in the order of 500 million (Richter et al., 2010). The loss of functioning habitats such as wetlands and floodplains often means that benefits, both valued and non-valued, are irreversibly lost. A recent global assessment indicates that of 292 rivers, 172 are already seriously affected and fragmented by dams (Nilsson et al., 2005). The most fragmented rivers systems are also some of the most habitat-rich, biodiverse, and populated river systems of the world. On a global scale, dams and reservoirs have already had a significant impact on freshwater flow-related ecosystem services, and the societies, economies, and population relying on these ecosystem services (Meybeck, 2003). In addition, natural recreational areas may be destroyed by large-scale dam construction.

Sometimes the regulating function of dams is desired. For example, dams may protect valuable infrastructure and settlements from recurrent flooding events. Dams and reservoirs can also ensure the provision of a regular water supply to society and agriculture. However, the net outcome of a holistic assessment of dams and reservoirs is, more often than not, the loss of provisioning, supporting, and regulating ecosystem services that would directly benefit humans in terms of energy, water supply, and water regulation. In some cases, large dams are being deconstructed to restore natural river flows (e.g., Gosnell and Kelly, 2010). In view of the development demands of the world, it is expected that a number of large dams for hydropower purposes will be constructed in many countries in Latin America, Africa, and south Asia in the near future. However, even the World Bank recognizes the limited operational capacity that exists to ensure that ecosystem values are truly accounted for, despite the increased recognition of these values highlighted by the groundbreaking report of the World Commission of Dams in 2000 (World Bank, 2009).

GHG release from large dams and reservoirs: A recently recognized added impact of large manmade dams is the release of CH_4 to the atmosphere. Methane is released as a by-product by anaerobic breakdown of inundated biomass. Methane has GWP 25 times greater than that of CO_2 and thus will have a significant contribution to total effective GHG emissions from large hydropower energy systems (e.g., Chen et al., 2009; Guerin et al., 2006). According to International Rivers (2011), 23% of all manmade CH_4 emissions can be attributed to large dams.

The trapping of sediments in dams: a serious impact of reservoirs and dams is the reduced levels of sediments transported to various parts of the river systems. An earlier estimate suggested that globally, dams in river systems trap 4–5 Gt/yr (Vorosmarty et al., 2003), or approximately 25% of total sediment transport. The environmental issue related to the sediment trapping is the concentration of harmful compounds that are stored in these sediments. Sometimes this has unintended benefits, such as dams acting as storage for harmful substances. Some authors are warning that a restoration or unintended break in these dams would pose serious threats downstream due to the years of built-up polluted sediments.

3.5 Environmental Sustainability in Energy Systems

It is a major challenge, and an absolute necessity, to understand the demands that will be placed on future energy systems to develop means of ensuring that energy can be provided in a sustainable manner. One approach to achieving this is to develop energy systems within globally defined sustainability criteria for critical Earth-system processes.

Defining these criteria requires the inclusion of living and nonliving systems (ecosystems and natural resources) and the consideration of risks of abrupt nonlinear changes, or tipping points/thresholds, as a result of environmental overshoot. Attempts have been made in the past to develop global indicators for sustainability, most famously in the Limits to Growth World3 scenarios in the 1972 Club of Rome report (Meadows et al., 1972). These scenarios, later revised twice (Meadows et al., 1992; 2004), excluded ecosystems and focused on nonrenewable resources and persistent pollution. Later global assessments, such as the Millennium Ecosystem Assessment (MEA, 2005) and the IPCC (2007a), include analyses of environmental states and trends, with partial attempts to also define global sustainability criteria. The IPCC stabilization scenarios come closest to defining such criteria at the global scale for the climate system. These scenarios involve targets of maximum atmospheric CO_2 concentration, but the CO_2 concentrations are not linked to specific thresholds or other features of the climate system. Other atmospheric pollutants that are targeted for their impacts on poor air quality rather than climate change have been addressed at the regional level. Methods such as the effects-based approach developed by the UNECE Convention on Long-Range Transboundary Air Pollution aim to optimize improvements in air quality (see Sliggers and Kakebeeke, 2004) and have defined critical loads and levels for different pollutants to protect human health, materials, and a range of ecosystems from the impacts of air pollution impacts.

Other approaches to defining global sustainability criteria include the World Wildlife Fund Living Planet Index, which is an indicator of global biodiversity, and the global ecological footprint assessment, which attempts to compare humanity's demand for renewable provisioning ecosystem services (crops, fish, livestock, timber) in relation to the capacity of the planet to regenerate these services (Living Planet Report, 2010). Regulating services from ecosystems, such as climate stability and freshwater supplies, are only partially represented, through estimates of carbon footprints.

Perhaps coming closest to defining appropriate global sustainability criteria for the Anthropocene era are the scientific advancements made in assessing the risks of tipping elements in the climate system (Schellnhuber, 2009), and the 'tolerable windows approach' that has been developed to provide a climate policy guidance framework. In the tolerable windows approach, which was first proposed by the German Advisory Council on Global Change (WBGU, 2009), limits designed to avoid dangerous anthropogenic climate change are set against assessments of tolerability. A key feature of this approach, and most other attempts to develop sustainability criteria, is the compromise between limits set by nature on the one hand, and the demands and the 'tolerable' costs set by society on the other.

A number of modeling tools have also been developed to specifically represent the sustainability impacts of energy systems. These include life-cycle analysis tools that provide quantitative assessments that compare the potential for environmental impacts (GHG emissions, SO_2 emissions, NO_x emissions, direct land requirements) associated with different energy systems at different points along the energy chain (Evans et al., 2009). Other tools include natural resource management; ecologically sustainable development tools, and integrated assessment models. The latter have been specifically developed for research on climate change and air quality (e.g., IPCC, 2007) and incorporate, for example, the full cycle of anthropogenic GHG emissions, the options and costs of their mitigation, the resulting climate change, the impacts of climate change, and the related options and costs of adaptation (Toth, 2003). There is also increasingly a focus in models on the 'triple bottom line,' or sustainability in terms of ecological, sociological, and economic factors (Harris, 2002).

3.5.1 Environmental Sustainability in GEA

The GEA transition pathways (see Chapter 17) build on existing frameworks and use a range of modeling tools. These pathways propose the development of future energy systems under a number of sustainability criteria. Key environmental targets are formulated with respect to climate change and air pollution. These include limiting global mean surface temperature change to 2°C above pre-industrial levels with a likelihood of more than 50% and achieving global compliance with WHO air-quality standards (PM2.5 < 35 μg/m^3) by 2030 (see Chapter 17 for more detail). The climate-change target is discussed in Section 3.2.2, while the air-quality target relates to human health and is considered in Chapter 4. Both targets consider current knowledge and policy in developing stringent environmental goals for energy systems.

The stringency of such goals implies significant environmental outcomes of these pathways measured with respect to atmospheric pollutants. As discussed in Chapter 17, the GEA transitions are most comparable to the most stringent IPCC scenarios. In these low pathways, GHG emissions may continue to increase only for a very short period of time (peaking between 2020 and 2030) before they must decline to reach levels at about zero or even negative over the long-term (beyond 2060). These scenarios are compatible with long-term atmospheric CO_2 concentrations stabilizing at below 400 ppm (see Figure 17.35 in Chapter 17).

The GEA pathways indicate that the nature and magnitude of future environmental impacts will greatly depend on the evolution of energy systems and other anthropogenic activities in the coming decades. A major global energy transition in the future as described by the GEA pathways will require on the one hand conditional convergence in incomes across regions, and on the other hand a combination of policies at the local and regional levels, with global availability of energy-efficient technologies and devices (see Chapter 17 for a detailed discussion). Future energy systems are expected to evolve differently across regions under such a major energy transition. In general, one can expect increases in fossil-energy systems equipped with carbon capture and storage (CCS) and increases in renewable energy (wind, solar, bioenergy) across all regions. The exact extent of such changes in energy systems will depend on availability of energy sources and other constraints (see Chapter 17, Section 17.3.4.4 for more regional discussion of energy systems). The changes in the energy system will have a major impact on GHG emissions and air pollutants in the future. In addition to global climate change policies, regional and local policies will play a major role in influencing future environmental impacts. Policies will need to be stringent in order to significantly control future environmental and other impacts. Ensuring universal access to clean cooking fuel and controlling levels of air pollution across all regions through stringent legislation are seen as being essential to ensuring improvements in environmental quality.

As summarized in Figure 3.25, the GEA 'Counterfactual' scenario indicates that in the absence of stringent policies in all regions, GHG emissions can be expected to increase across all regions, while the regional evolution of air pollutants (shown here are SO_2, NO_x, and $PM_{2.5}$) will depend on the effectiveness of currently legislated air-quality controls. While currently planned air quality legislation, as in the GEA Counterfactual scenario, will bring reductions in outdoor air pollution levels across Organisation for Economic Co-operation and Development (OECD) regions, for other regions, in particular Asia and sub-Saharan Africa, increasing populations and rapid expansions in energy systems

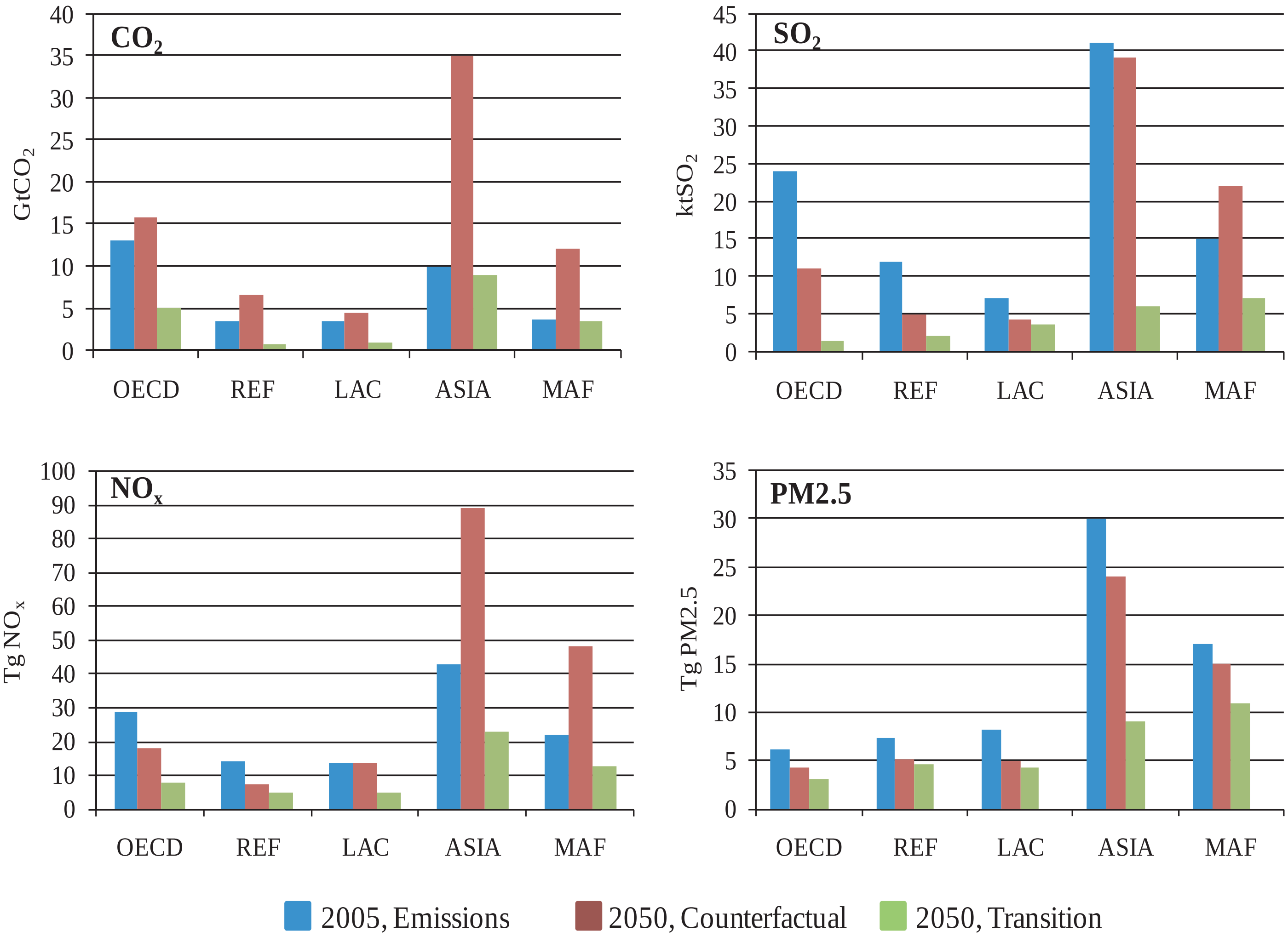

Figure 3.25 | CO_2, SO_2, NO_x, and $PM_{2.5}$ emissions in 2005 and 2050 in GEA Counterfactual and GEA Mix (Transition) Scenarios. Regional definition in Chapter 17.

will mean that emissions continue to increase. The inclusion of sustainability criteria, as in the GEA transition pathways (shown here is the GEA Mix; see Chapter 17 for a full description of pathways), and the subsequent increase in the stringency of policies, lead to significant declines in both GHG and pollution levels across regions.

These emission reductions from anthropogenic systems can be expected to bring significant environmental benefits across regions compared to current levels. While a detailed description of regional environmental impacts is beyond the scope of this discussion, such impacts will extend to a wide range of environmental concerns, as discussed earlier in the regional descriptions. These would include, for example, decreased sea-level rise; decreased frequency of droughts and flooding; decreased impacts of BC-related climate effects, decreased crop losses from O_3-related effects; and decreased acidification and eutrophication. In addition to environmental benefits, significant health-related benefits can be expected as a result of stringent pollution energy access policies (see Chapters 4 and 17 for discussions on the health-related impacts of air pollution).

3.5.2 Proposal for Alternative Framework for Environmental Sustainability of Energy Systems

Our assessment is that global sustainability criteria for the global energy systems in the Anthropocene era must be comprehensive, including all relevant environmental processes (e.g., carbon and nitrogen cycles) and systems on Earth that are affected by our energy systems (e.g., the climate system, the ocean system, and terrestrial and aquatic systems). Such criteria must consider the interactions and risks of abrupt, nonlinear change (tipping points and thresholds). They must also address the ability of systems on Earth to maintain their resilience, and thereby their capacity to remain in desired states conducive to human development in an era of rapid global change. The GEA sustainability indicators provide a systematic approach to defining specific quantitative goals for environmental sustainability with respect to the atmospheric impacts associated with particular scenarios involving air pollution and climate change (see Chapters 4 and 17). This approach will also need to extend to encompass the full range of Earth-system considerations.

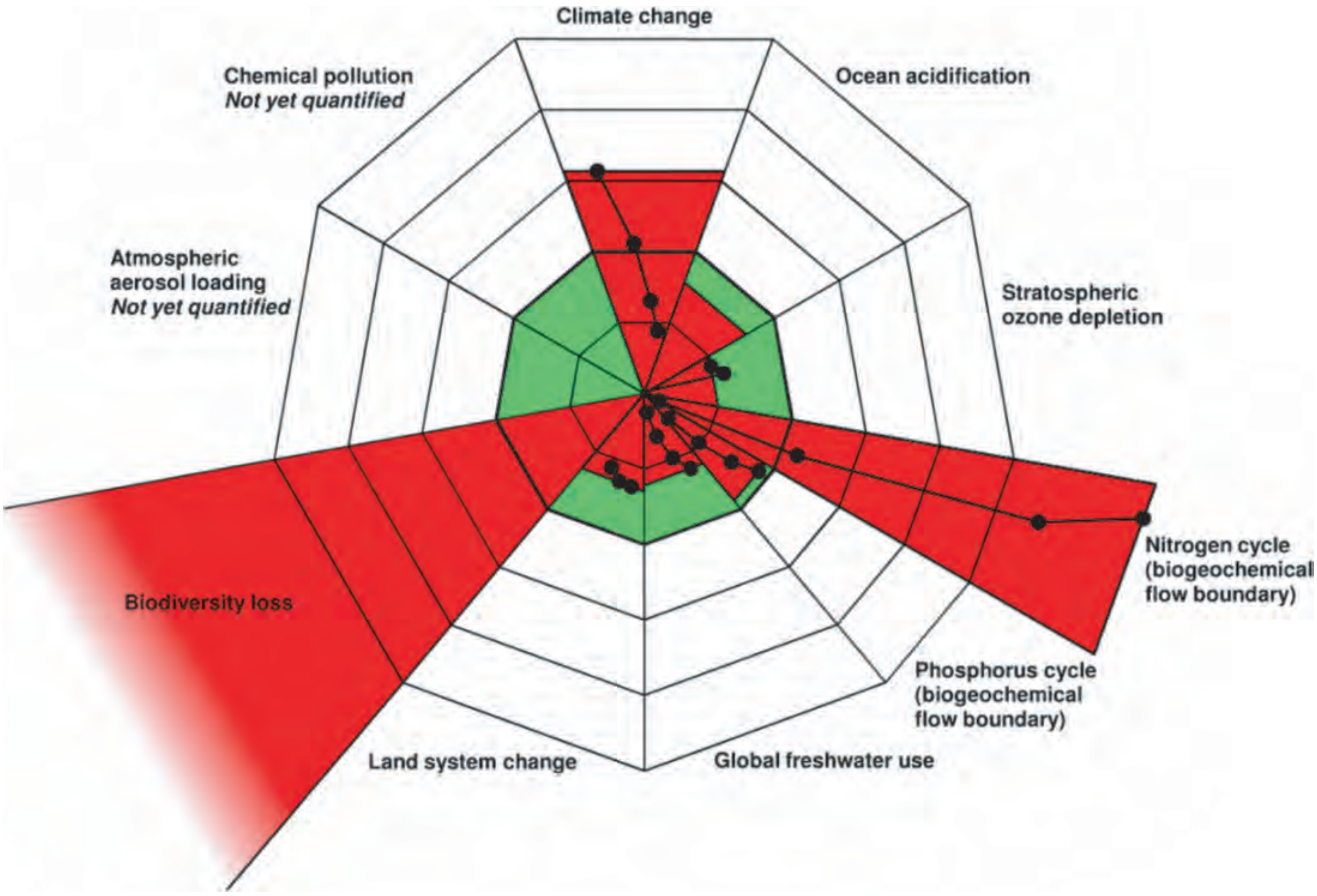

Figure 3.26 | Current global state of the world for the ten proposed planetary boundaries. The green areas denotes a "safe operating space" for human development, and red indicates the current position for each boundary process. The dots indicate evolution by decade from the 1950s.

Based on this conclusion and on the assessment of previous approaches to defining global sustainability criteria, the recently advanced planetary boundaries concept (Rockström et al., 2009) is considered here as a complementary way of framing the assessment of sustainability criteria for global energy systems. This framework, which builds on previous global sustainability approaches, advances in Earth system science, and resilience research, was developed as a means of addressing and establishing global sustainability criteria that recognize the global pressures from human enterprise and the risk of undermining the resilience of major biophysical systems on Earth. It complements existing environmental indicators, such as the critical loads and levels approach, global footprint analysis, and the 'tolerable windows approach,' by addressing the biophysical prerequisites for avoiding abrupt and undesired global change. It is an expansion of the 'guardrail' approach (Hare and Meinshausen, 2006) used to define safe climate-mitigation targets by also addressing other global environmental change processes. The approach is presented in Figure 3.26, which shows the planetary boundaries for nine Earth-system processes, which together define a safe operating space for humanity (indicated by the green area), within which human development stands a good chance of proceeding without large-scale deleterious change. Estimates indicate that the safe levels are approached or in some cases surpassed. Energy systems contribute to humanity's approach to all the planetary boundaries, but in particular climate change, biodiversity loss, land-system change, atmospheric aerosol loading, and chemical pollution.

So how can the planetary boundary framework be redefined to be appropriate for an assessment of our energy systems? First this requires that the main Earth-system processes affected by our energy systems are defined (as already discussed in previous sections) and related to impacts that act at the planetary level. The planetary boundary framework already includes proposed boundary levels appropriate for the prevention of Earth system-threatening climate change (see Rockström et al., 2009). Conceptually, the framework may also facilitate efforts to understand the impacts of the more 'traditional' atmospheric pollutants on ecosystems, since a similar critical limits threshold-driven concept has long been used in scientific research to define loads and levels below which pollution would not threaten perturbations to environmental systems (Sliggers and

Table 3.6 | Summary of proposed global sustainability indicators and target levels for energy systems based on planetary boundaries.

Pressure	Receptor system	Planetary boundary	Indicator target or orientation value
Emission of radiative forcers (e.g., CO_2, PM, aerosols), leading to changes in global and regional climate	Atmosphere	Climate change; Rate of biodiversity loss; Ocean acidification; Atmospheric aerosol loading	350 ppm CO_2 or climate forcing less than 1 W/m^2 above the pre-industrial level, or less than 1000 GtCO_2-eq released in the period 2000–2050
			No more than 2°C above pre-industrial of human-induced global warming, with a probability greater than 50%
			Aim for 50% GHG emission reduction by 2030 and subsequent reductions to achieve the target
			Reduce by 2030 and eliminate by 2050 emissions of BC, OC, nitrogen and sulphur, and other PM that contribute to atmospheric aerosol loading, so that radiative forcing remains less than 1 W/m^2.
Emission of air pollutants (e.g., SO_2, No_x, O_3)	Atmosphere	Climate change, Rate of biodiversity loss, Nitrogen cycle, Chemical pollution; Atmospheric aerosol loading	Reduce by 2030 and eliminate by 2050 emissions of BC, OC, nitrogen and sulphur species, and other PM that contribute to atmospheric aerosol loading, so that radiative forcing remains less than 1 W/m^2.
			Reduce by 2030 and eliminate by 2050 emissions of air pollutants that contribute to human health and ecosystem damage – *further research required to define such limits*
Land requirement and degradation	Terrestrial biosphere	Change in land-use; Rate of biodiversity loss; Chemical pollution	No more than 15% of global land cover should be converted to cropland
Water resource requirement and impaired water quality	Freshwater and marine systems	Global freshwater use; Rate of biodiversity loss; Chemical pollution	Limit global freshwater use to no more than 4, 000–6, 000 km^3/yr of consumptive use of accessible river flow

Kakabeeke, 2004). Therefore, for many of the environmental threats related to atmospheric emissions from energy, it might be considered relatively straightforward to use existing indicators and sustainability criteria to assess whether the planetary boundary has been crossed. The actual reality of using these indicators is discussed in more detail below. For the other environmental impacts, which are not directly related to atmospheric emissions (for example, land-use and water resources), different threshold criteria will have to be defined according to a full understanding of the potential impacts, their causes in relation to energy use and supply, and trends and variations within geographical regions.

Discussed below are proposed definitions for planetary boundaries for energy systems that include a range of environmental impacts. They build on earlier analysis by Rockström et al. (2009), and also include boundaries for air pollution, land-use, and water use, based on the summary of current impacts in Sections 3.2, 3.3, and 3.4. The proposal is summarized in Table 3.6.

3.5.2.1 Defining Planetary Boundaries for Climate Change

The planetary boundary GHG concentration-based target proposed by Rockström et al. (2009) calls for limiting concentrations to 350 ppm CO_2, or approximately 400 ppm CO_2-eq, with an uncertainty range of 350–500 ppm CO_2. The climate-change boundary proposed here aims at minimizing the risk of highly nonlinear, possibly abrupt and irreversible, Earth-system responses (National Research Council, or NRC, 2002; IPCC, 2007c). These responses may be related to one or more thresholds, the crossing of which could lead to the disruption of regional climates (Lenton et al., 2008), trigger the collapse of major climate-dynamic patterns such as the thermohaline circulation (Clark et al., 2002), and drive other impacts that would be difficult for society to address, such as rapid sea-level rise. The risk of crossing such thresholds will rise sharply with further anthropogenically driven deviation from the natural variability of the Holocene climate.

This dual approach to defining the planetary boundary for climate change uses both atmospheric CO_2 concentration and RF as global-scale control variables. Boundary values of 350 ppm CO_2 and 1 W/m^2 above pre-industrial levels are suggested. The boundary is based on: an analysis of the equilibrium sensitivity of the climate system to GHG forcing; the behavior of the large polar ice sheets under climates warmer than those of the Holocene (Hansen et al., 2008); and the observed behavior of the climate system at a current CO_2 concentration of about 387 ppm and +1.6 W/m^2 (with a 0.6–2.4 W/m^2 90% confidence range) net RF (IPCC 2007a).

3.5.2.2 Defining Planetary Boundaries for Air Pollution

How do the critical loads and levels currently defined by the UNECE LRTAP Convention relate to the planetary boundary concept? Firstly, it is useful to identify which boundaries presented in Rockström et al. (2009) are relevant for air pollution. Perhaps the most obvious is that for the nitrogen cycle, which proposes a boundary of 35 Mt/year of N_2 removed from the atmosphere for human use. How does this compare with LRTAP Convention critical loads? Current empirical critical loads for nutrient nitrogen (which protect against changes in plant growth, interspecific relationships, and soil-based processes) are provided for a wide variety of different natural and seminatural ecosystem habitats. These include forest habitats, heathlands, scrubland, tundra, grasslands, mires, bogs, fens, and inland surface water and coastal and marine habitats. The critical loads vary from 5–40 kg N/ha/year, with the range indicating the variability in ecosystem sensitivity to excess Nr input. Considering that there are similar variations in ecosystem sensitivity to acidifying nitrate and to ammonium deposition, which causes soil acidification, the difficulty of defining a single, planetary-scale boundary for N deposition becomes apparent.

The planetary boundary limit on N_2 fixation attempts to reduce Nr at the source, rather than at different points along the nitrogen cascade that will lead to varying types of ecosystem damage defined by Galloway et al. (2003). However, the difficulty comes in using scientific or expert judgment (the latter is necessary where there are gaps in our scientific understanding) to define a threshold for damage for the different processes through which excess Nr causes eutrophication and acidification. This is the case, since unless N_2 fixation rates equal denitrification rates, there will always be excess Nr in the system, which may cause damage. The geographical spatial heterogeneity of atmospheric Nr pollution and deposition as well as ecosystem sensitivity will mean that some ecosystems will be unprotected by a single planetary-scale boundary that is set to avoid stepping outside the resilience of the system as a whole. It is also not possible to exclude the possibility of cumulative impacts caused by steady increases in N deposition that may result in systems crossing tipping points, even under low pollution loads, if they persist for substantial periods of time. Therefore, we propose that the planetary boundary for N_2 fixation provides a useful guide to encourage more efficient use of Nr in our agricultural systems and more efficient combustion of fossil fuels, but that it should be used in concert with regionally based indicators that employ an effects-based approach to limit pollution impacts. Only then will control be sufficient to ensure protection of the most vulnerable systems. Similar arguments can be made for each of the regional-scale air pollutants discussed in this Chapter.

3.5.2.3 Defining Planetary Boundaries for Land-use

The planetary boundary for land-use proposes that no more than 15% of global land cover should be converted to cropland. Given the current level of malnourishment in the world and the expected 50% increase in world population by 2050, the demand for food and animal feed is expected to require a 70% increase in agricultural production by 2050 (FAO, 2006). Even if most of the increase comes through intensification and yield improvements, there remains a chance that land converted to agriculture could surpass 15% by 2050; this chance is increased by the likelihood that an increasing share of agricultural land will also be devoted to production of biofuels. However, with major investments in agricultural research and an emphasis on high-efficiency agricultural and integrated food-energy systems, expansion of land under agriculture could be significantly constrained (Johnson and Virgin, 2010). The conclusion is that expansion of land used for biofuels must accompanied by much greater investment in 'smart agriculture,' both for the sake of reducing land pressures and for improving food and energy security simultaneously.

3.5.2.4 Defining Planetary Boundaries for Water Resources

Actual freshwater availability is manifested at the local catchment or river-basin level. At the same time, there is increasing evidence that humanity faces global freshwater constraints due to the finite nature of freshwater resources, and the coupling of local water balances with the global hydrological cycle. Currently some 30% of the world's population faces water stress, and approximately 25% of the world's rivers dry out before reaching the ocean (Molden, 2007). The global freshwater cycle has entered the Anthropocene era (Meybeck, 2003), where humans now constitute the dominant driving force, altering river flows at the global level (Shiklimanov and Rodda, 2003) and the spatial patterns and seasonal timing of vapor flows (Gordon et al., 2005).

Global freshwater assessments show that the accessible global volume of runoff water (accessible base flow) is in the order of 12,500–15,000 km^3/yr (Postel et al., 1998; deFraiture et al., 2001). Several analyses show that severe water scarcity is experienced on the regional scale when withdrawals of runoff exceed 40–60% of this stable freshwater resource. This provides an uncertainty range of sustainable global freshwater withdrawals of 5000–9000 km^3/yr, beyond which negative implications for human societies are expected. However, not all of these withdrawals constitute consumptive use. Current withdrawals of approximately 4000 km^3/yr (World Water Development Report, 2009), of which less than 3000 km^3/yr is consumptive use. Based on these indicators of sustainability thresholds for freshwater use, a planetary boundary range for global freshwater use has been proposed at 4000–6000 km^3/yr of consumptive use of accessible river flow. Evidence indicates that transgressing this boundary range leads to an overuse of freshwater in catchments and river basins where water-induced thresholds, e.g., the collapse of freshwater dependent ecosystems, can no longer be excluded (Rockström et al., 2009). This freshwater boundary is highly tentative, given the uncertainties associated with aggregating sustainable freshwater use at a global level, but it does provide an indicator of the magnitude of freshwater that can be used for bioenergy and other purposes before serious water-related problems occur.

3.6 Conclusion

The assessment in this chapter confirms earlier scientific findings that a global energy transformation is needed to address the growing risks associated with accelerated global environmental change. Anthropogenic pressures on the planet have reached a level where large-scale deleterious impacts, or even catastrophic ones, can no longer be excluded. Such impacts have the potential to undermine human development. This new global social-environmental predicament is closely associated with energy.

Atmospheric emissions from energy use contribute to multiple environmental impacts. In addition to climate change, atmospheric pollutants may limit net primary productivity of ecosystems, and lead to the acidification and eutrophication of land and seascapes. These impacts interact, reinforcing impacts on social and environmental systems, in complex ways that are not always well understood. This chapter confirms the necessity for the global energy system, which is the largest source of GHG emissions, to – as a minimum requirement – operate within the 2°C climate guardrail. In fact, based on the latest science, this chapter concludes that 1.5°C may be a more appropriate guardrail. This conclusion is based on the high likelihood that even small increases in global mean surface temperature will have extensive negative impacts on societies and ecosystems. This chapter also concludes that immediate action on reducing BC and tropospheric O_3 should be a high priority for short-term climate mitigation with associated benefits for human health. Action to control BC will particularly aid improvements in indoor air quality in the poor households of the world (see also Chapter 4).

This chapter also confirms the interconnectedness among all regions of the world, in terms of high dependency on fossil-energy sources and negative environmental impacts from the current energy mix. This energy mix generates impacts at the local, regional, and global levels. This chapter also confirms that there are winners and losers on the global energy scene – with the poorest tropical regions in the world being most vulnerable to the environmental impacts of unsustainable energy use, and the lowest energy-using regions, including the poorest developing countries and the polar regions, being highly affected by negative impacts originating from energy use in other regions.

This chapter confirms the intricate link between land and energy. Land is affected through loss or damage to ecosystems from land-use change and contamination from energy-related waste arising from activities such as mining, drilling, and the transport of fossil fuel raw materials. The alternatives to fossil fuel-based energy systems (e.g., nuclear power, hydropower, biomass-derived fuels and solar power) also lead to a variety of adverse environmental impacts on air, land, and water at various stages in the energy chain. In particular, the intensive use of land and fresh water by bioenergy systems has implications for meeting increased global food demands, as assessments increasingly indicate the existence of regional and global limitations to the expansion of agricultural land and water use for biomass production. It is concluded that the expansion of land used for biofuels must be accompanied by greater investment in 'smart agriculture,' both for the sake of reducing land pressures and for improving food and energy security simultaneously.

Water resources and aquatic ecosystems are also adversely impacted by various types of energy systems. Water may be diverted from other uses by biomass crop production or hydropower schemes. Aquatic ecosystems may be damaged by the interruption of hydrological flows (e.g., from dam construction or open-cast mining operations) as well as by contamination during coal and uranium ore mining, oil and gas drilling, fossil fuel-processing and transportation, or thermal pollution from power stations.

Drawing upon the latest science, this chapter confirms earlier assessments (particularly within climate science) that atmospheric emissions, of both GHGs and air pollutants, constitute the core and most immediate environmental challenges within the energy sector. However, it is equally clear that energy impacts the biosphere; this calls for immediate attention to reduce the impacts of energy systems on land and water resources, including the use and flow of nitrogen; biodiversity loss; the toxic effects of tropospheric O_3 and other toxic chemical pollution.

Chapter 3 therefore concludes that there is a need for an integrated approach, in which all environmental impacts from energy use are considered, both in terms of climate and ecosystem change. An energy transformation would bring multiple benefits and would help humanity tunnel out of the current era of rapid global environmental change. Such a transformation would have benefits ranging from averting global climate change to reducing the burden of air pollution and ecosystem degradation. It would also require the integration of policy and development action on climate change, air pollution, and ecosystem management, from local to regional to global levels. Energy systems, climate change, and air pollution are strongly connected, in such a way that integrated decision making, coordinated at an international level, will be absolutely crucial to the development of viable options for mitigating the adverse environmental impacts of our energy needs.

A new framework is therefore needed to guide a global energy transformation. This chapter concludes that there is an urgent need for global sustainability criteria, within which the global energy system can operate and identifies the 'planetary boundary' approach as one means of defining global sustainability criteria that could help establish future sustainable energy pathways.

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References

ACIA, 2004: *Impacts of a Warming Arctic: Arctic Climate Impact Assessment*. Cambridge University Press, Cambridge, UK and New York, NY.

Agoumi, A., 2003: *Vulnerability of North African Countries to Climatic Changes: Adaptation and Implementation Strategies for Climate Change*. International Institute for Sustainable Development.

Ainsworth, E. A., et al., 2008: Targets for crop biotechnology in a future high-CO_2 and high-O_3 world. *Plant Physiology*, **147**: 13–19.

Alcamo, J., J. M. Moreno, B. Nováky, M. Bindi, R. Corobov, R. J. N. Devoy, C. Giannakopoulos, E. Martin, J. E. Olesen, and A. Shvidenko, 2007: *Climate Change 2007: Impacts, Adaptation and Vulnerability. Chapter 12: Europe. Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change*, M. L. Parry, O. F. Canziani, J. P. Palutikof, P. J. van der Linden, and C. E. Hanson (eds.), Cambridge University Press, Cambridge, UK and New York, NY, pp.541–580.

AMAP, 2006: *AMAP Assessment 2006: Acidifying Pollutants, Arctic Haze and Acidification in the Arctic*. Arctic Monitoring and Assessment Programme (AMAP), Oslo, Norway.

AMAP, 2007: *Arctic Oil and Gas 2007 Oslo. Arctic Monitoring and Assessment Programme (AMAP), Oslo, Norway.AMAP, 2010: AMAP Assessment 2009: Radioactivity in the Arctic*. Arctic Monitoring and Assessment Programme (AMAP), Oslo, Norway.

Amann, M., I. Bertok, J. Borken-Kleefeld, J. Cofala, C. Heyes, L. Höglund-Isaksson, Z. Klimont, B. Nguyen, M. Posch, P. Rafaj, R. Sandler, W. Schöpp, F. Wagner, and W. Winiwarter, 2011: Cost-effective control of air quality and greenhouse gases in Europe: modeling and policy applications. *Environmental Modelling and Software*, *in press* doi 10.1016/j.envsoft.2011.07.012

Archer, D. R. and H. J. Fowler., 2004: Spatial and temporal variations in precipitation in the Upper Indus Basin, global teleconnections and hydrological implications. *Hydrology and Earth System Sciences*, 8: 47–61.

Arnell, N.W., 2004: Climate change and global water resources: SRES emissions and socio-economic scenarios. *Global Environmental Change,* 14: 31–52.

Arnell, N. W., 2006: *Global impacts of abrupt climate change: an initial assessment*. Working Paper 99, Tyndall Centre for Climate Change Research, University of East Anglia, Norwich, UK, pp.37.

Ashmore, M., 2005: Assessing the future global impacts of ozone on vegetation. *Plant, Cell & Environment*, 28(8): 949–964.

Bell, J. N. B., 1985: *SO2 effects on productivity of grass species, in Effects of SO2 on Plant Productivity*, Mooney and A. H. Goldstein (eds.), Stanford University Press, Palo Alto, CA, pp.209–266.

Bellwood, D., T. Hughes, C. Folke, and M. Nystrom, 2004: Confronting the coral reef crisis. *Nature*, 429: 827–833.

Bender, J. and H. J. Weigel, 1993: *Crop responses to mixtures of air pollutants, in Air Pollution and Crop Responses in Europe*, H. J. Jager et al. (eds.), CEC, Brussels, Belgium, pp.445–453.

Berndes, G., 2008: Future biomass energy supply: the consumptive water use perspective. *International Journal of Water Resources Development.* 24: 235–245.

Betts, R. A., M. Collins, D. L. Hemming, C. D. Jones, J. A. Lowe, and M. G. Sanderson, 2011: When could global warming reach 4°C? *Philosophical Transactions of the Royal Society A*. 369: 67–84.

Bishop, K. and H. Hultberg, 1995: Reversing acidifcation in a forest ecosystem – The Gardsjon covered catchment. *Ambio*, 24(2): 85–91.

Bleeker, A., W. K. Hicks, F. Dentener, J. Galloway, and J. W. Erisman, 2011: N deposition as a threat to the World's protected areas under the Convention on Biological Diversity. *Environmental Pollution* (in press).

Bobbink, R., K. Hicks, J. Galloway, T. Spranger, R. Alkemade, M. Ashmore, M. Bustamante, S. Cinderby, E. Davidson, F. Dentener, B. Emmett, J.W. Erisman, M. Fenn, F. Gilliam, A. Nordin, L. Pardo, and W. De Vries, 2010: Global assessment of nitrogen deposition effects on terrestrial plant diversity: a synthesis. *Ecological Applications*, 20(1): 30–59.

Bobbink, R., M. Hornung, and J. G. M. Roelofs, 1998: The effects of air-borne nitrogen pollutants on species diversity in natural and semi-natural European vegetation. *Journal of Ecology*, 86: 717–738.

Boko, M., I. Niang, A. Nyong, C. Vogel, A. Githeko, M. Medany, B. Osman-Elasha, R. Tabo, and P. Yanda, 2007: *Climate Change 2007: Impacts, Adaptation and Vulnerability. Chapter 9: Africa. Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change*, M. L. Parry, O. F. Canziani, J. P. Palutikof, P. J. van der Linden, and C. E. Hanson (eds.), Cambridge University Press, Cambridge, UK and New York, NY, pp.433–467.

Bond, T., E. Bhardwaj, R. Dong, R. Jogani, S. Jung, C. Roden, D. Streets, and N. Trautmann, 2007: Historical emissions of black and organic carbon aerosol from energy-related combustion, 1850–2000. *Global Biogeochemical Cycles*, 21: GB2018.

Bond, T., D. Streets, K. Yarber, S. Nelson, J-H. Woo, and Z. Kilmont, 2004: A technology-based global inventory of black and organic carbon emissions from combustion. *Journal of Geophysical Research*, 109: D14203.

Bravo, H., A. R. Soto, R. Sosa, P. Sánchez, A. L. Alarcón, J. Kahl, and J. Ruíz, 2006: Effect of acid rain on building material of the El Tajín archaeological zone in Veracruz, Mexico. *Environmental Pollution*, 144: 655–660.

Bricker, S., B. Longstaff, W. Dennison, A. Jones, K. Boicourt, C. Wicks, and J. Woerner, 2007: *Effects of Nutrient Enrichment In the Nation's Estuaries: A Decade of Change. NOAA Coastal Ocean Program Decision Analysis Series No. 26*. National Centers for Coastal Ocean Science, Silver Spring, MD, USA.

Brohan, P., J. J. Kennedy, I. Harris, S. F. B. Tett, and P. D. Jones, 2006: Uncertainty estimates in regional and global observed temperature changes: A new data set from 1850. *Journal of Geophysical Research*, 111: D12106.

Casper, K. N., 2009: Oil and gas development in the Arctic: Softening of Ice demands hardening of international law. *Natural Resources Journal*, 49: 825–881.

Cazenave, A., 2006: How fast are the ice sheets melting? *Science*, 314, 1250–1252.

CCME, 2003: *Climate, Nature, People: Indicators of Canada's Changing Climate. Climate Change Indicators Task Group of the Canadian Council of Ministers of the Environment, Canadian Council of Ministers of the Environment Inc*.,Winnipeg, Canada.

CEC, 2009: *Taking Stock – 2005 North American Pollutant Releases and Transfers,* Commission for Environmental Cooperation, Montreal, Canada.

Chan, C. K. and X. Yao, 2008: Air pollution in mega cities in China. *Atmospheric Environment*, 42: 1–42.

Chen, H., Y. Wu, X. Yuan, Y. Gao, N. Wu, and D. Zhu, 2009: Methane emissions from newly created marshes in the drawdown area of the Three Gorges Reservoir, *Journal of Geophysical Research*, 114: D18301.

Christensen, J. H. and O. B. Christensen, 2003: Severe summertime flooding in Europe. *Nature*, 421, 805–806.

Church, J. A. and N. J. White, 2006. A 20th century acceleration in global sea-level rise. *Geophysical Research Letters,* **33**: LO1602.

Clancy, J. S., 2008: Urban ecological footprints in Africa. *African Journal of Ecology*, 46(4): 463–470.

Clark, P., N. Pisias, T. Stocker, and A. Weaver, 2002: The role of thermohaline circulation in abrupt climate change. *Nature*, 415: 863–869.

Cofala, J. and S. Syri, 1998: *Sulfur Emissions, Abatement Technologies and Related Costs for Europe in the RAINS Model Database*. IR-98–035. IIASA, Laxenburg, Austria.

Corbett, J. J., D.A. Lack. J. J. Winebrake, S. Harder, J. A. Silberman, and M. Gold, 2010: Arctic shipping emissions inventories and future scenarios. *Atmospheric Chemistry and Physics*, 10: 10271–10311.

Crutzen, P. J., 2002: Geology of mankind the anthropocene. *Nature*, 415: 23.

Cruz, R. V., H. Harasawa, M. Lal, S. Wu, Y. Anokhin, B. Punsalmaa, Y. Honda, M. Jafari, C. Li, and N. Huu Ninh, 2007: *Climate Change 2007: Impacts, Adaptation and Vulnerability, Chapter 10: Asia. Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change*, M. L. Parry, O. F. Canziani, J. P. Palutikof, P. J. van der Linden, and C. E. Hanson, (eds.), Cambridge University Press, Cambridge, UK and New York, NY, pp.469–506.

Davidson, O., M. Chenene, E. Kituyi, J. Nkomo, C. Turner, and B. Sebitosi B., 2007: *Sustainable Energy in sub-Saharan Africa*. Draft Science/Work plan, International Council for Science, Regional Office for Africa.

Davison A.W. and J. D. Barnes, 1998: Effects of ozone on wild plants. *New Phytologist*, 139(1): 135–151.

deFraiture, C., D. Molden, U. Amarasinghe, and I. Makin, 2001: PODIUM: Projecting water supply and demand for food production in 2025. *Physics and Chemistry of the Earth, Part B: Hydrology, Oceans and Atmosphere,* 26(11–12), 869–876.

de Vries, W., S. Solberg, M. Dobbertin, H. Sterba, D. Laubhahn, G. J. Reinds, G-J. Nabuurs, P. Gundersen, M. A. Sutton, 2007: Ecologically implausible carbon response? *Nature*, 451: E1-E3.

Denholm, P., 2006: Improving the technical, environmental and social performance of wind energy systems using biomass-based energy storage. *Renewable Energy*, 1: 1355–1370.

Dentener, F., S. Kinne, T. Bond, O. Boucher, J. Cofala, S. Generoso, P. Ginoux, S. Gong, J. J. Hoelzemann, A. Ito, L. Marelli, J. E. Penner, J. P. Putaud, C. Textor, M. Schulz, G. R. van der Werf, and J. Wilson, 2006: Emissions of primary aerosol and precursor gases in the years 2000 and 1750 prescribed data-sets for AeroCom. *Atmospheric Chemistry and Science*, 6: 4321–4344.

Devoy, R. J. N., 2007: Coastal vulnerability and the implications of sea-level rise for Ireland. *Journal of Coastal Research,* 24(2): 325–341.

Di Iorio, L. and C. W. Clark, 2010: Exposure to seismic survey alters blue whale acoustic communication. *Biology Letters*, 6(1): 51–54.

Dolman, A. J., G. R. van der Werf, M. K. van der Molen, G. Ganssen, G., J. W. Erisman, and B. Strengers, 2010: A carbon cycle science update since IPCC AR-4. *Ambio*, 39: 402–412.

Donner, S. D., and C. J. Kucharik, 2008: Corn-based ethanol production compromises goal of reducing nitrogen export by the Mississippi River. *Proceedings of the National Academy of Sciences of the United States of America*, 105: 4513–4518.

Duce, R. A., J. LaRoche, K. Altieri, et al., 2008: Impacts of atmospheric anthropogenic nitrogen on the open ocean. *Science*, 320(5878): 893–897.

Duncan, B., J. Logan, I. Bey, I. Megretskaia, R. Yantosca, P. Novelli, N. Jones, and C. Rinsland, 2007: Global budget of CO, 1988–1997: Source estimates and validation with a global model. *Journal of Geophysical Research*, 112: D22301.

Eckhardt, S., A. Stohl, S. Beirle, N. Spichtinger, P. James, et al., 2003: The north Atlantic Oscillation controls air pollution transport to the Arctic. *Atmospheric Chemistry and Physics*, 3: 1769–1778.

EEA, 2008: *Energy and environment indicator fact sheet, EN13 Nuclear waste production*. European Environment Agency (EEA), Copenhagen, Denmark.

Emberson, L. D., et al., 2003: *Air Pollution Impacts on Crops and Forests-A Global Assessment*, edited, Imperial College Press, London, UK.

Emberson, L. D., P. Bueker, and M. R. Ashmore, 2007: Assessing the risk caused by ground level ozone to European forest trees: A case study in pine, beech and oak across different climate regions. *Environmental Pollution,* 147: 454–466.

Emberson L. D., P. Büker, M. R. Ashmore, et al., 2009: A comparison of North American and Asian exposure-response data for ozone effects on crop yields. *Atmospheric Environment*, 43: 1945–1953.

ENA, 2011: *The European Nitrogen Assessment: Sources, Effects and Policy Perspectives*. Eds. Sutton. M.A., Howard, C.M., Erisman, J.W., Billen, G., Bleeker, A., Grennfelt, P., van Grinsven, H. and Grizzetti, B. Cambridge University Press. Avaialble at: http://www.nine-esf.org/ENA-Book

Energy Information Administration, 2000: Sub-Saharan Africa: Environmental Issues, November 2000.

Environment Canada, 2008: Canadian Environmental Sustainability Indicators. ec.gc.ca/indicateurs-indicators/default.asp?lang=en&n=7D7BDF1E-1 (accessed 23 March 2010).

Environment Canada 2010: www.ec.gc.ca/acidrain/acidfact.html (accessed 29 March 2010).

Environment Canada 2010: National Ambient Air Quality Objectives. www.ec.gc.ca (accessed 27 March 2010).

Evans, A., V. Strezov, and T. J. Evans, 2009: Assessment of sustainability indicators for renewable energy technologies. *Renewable and Sustainable Energy Reviews*, 13: 1082–1088.

Fang, Y. P., Y. Zeng, and S. M. Li, 2008: Technological influences and abatement strategies for industrial sulphur dioxide in China. *International Journal of Sustainable Development and World Ecology*, 15: 122–131.

FAO, 2004: *Seguridad alimentaria como estrategia de Desarrollo rural. Proc. 28ava Conferencia Regional de la FAO para América Latina y el Caribe*, Ciudad de Guatemala, Guatemala.

FAO, 2005: *State of the World's Forests 2005. Food and Agriculture Organization of the United Nations (FAO)*, Rome.

FAO, 2006: *World agriculture: towards 2030/2050. Food and Agriculture Organization of the United Nations (FAO)*, Rome.

Farman, J. C., R. J. Murgatroyd, A. M. Silnickas, and B. A.Thrush, 1985: Ozone photochemistry in the antarctic stratosphere in summer. *Quarterly Journal of the Royal Meteorological Society,* 111: 1013–1025.

Fenn, M. E., J. S. Baron, E. B. Allen, H. M. Reuth, K. R. Nydick, L. Geiser, W. D. Bowman, J. O. Sickman, T. Meixner, D. W. Johnson, and P. Neitlich, 2003: Ecological Effects of Nitrogen Deposition in the Western United States. *Bioscience*, 53(4): 404–420.

Fey, M. V. and H.A. Dodds, 1998: Classifying the sensitivity of soils of the South African highveld to acidification. *South African Journal of Plant and Soil*, 15: 99–103.

Field, C. B., L. D. Mortsch, M. Brklacich, D. L. Forbes, P. Kovacs, J. A. Patz, S. W. Running, and M. J. Scott., 2007: *Climate Change 2007: Impacts, Adaptation and Vulnerability. Chapter 14: North America. Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change*, M.L. Parry, O.F. Canziani, J.P. Palutikof, P.J. van der Linden, and C.E. Hanson, (eds.), Cambridge University Press, Cambridge, UK and New York, NY, pp.617–652.

Fiki, C., and B. Lee, 2004: Conflict generation, conflict management and self-organizing capabilities in drought-prone rural communities in north-eastern Nigeria: A case study. *Journal of Social Development in Africa*, 19(2):25–48.

Fischer, G., S. Prielera, H. van Velthuizena, S. M. Lensink, M. Londo, and M. de Wit, 2010: Biofuel production potentials in Europe: Sustainable use of cultivated land and pastures, Part I: Land productivity potentials. *Biomass and Bioenergy*, 34: 159–172.

Fischer, G., M. Shah, and H. vanVelthuizen, 2002: *Climate change and agricultural vulnerability. Preprints, World Summit on Sustainable Development*, Johannesburg, pp.160.

Fiscus, E. L., et al., 2005: Crop responses to ozone: uptake, modes of action, carbon assimilation and partitioning. *Plant Cell Environment*, 28: 997–1011.

Fisher, B., N. Nakicenovic, K. Alfsen, J. Corfee Morlot, F. de la Chesnaye, J.-C. Hourcade, K. Jiang, M. Kainuma, E. La Rovere, A. Matysek, A. Rana, K. Riahi, R. Richels, S. Rose, D.P. van Vuuren, R. Warren, P. Ambrosi, F. Birol, D. Bouille, C. Clapp, B. Eickhout, T. Hanaoka, M.D. Mastrandrea, Y. Matsuoko, B. O'Neill, H. Pitcher, S. Rao, and F. Toth, 2007: Issues related to mitigation in the longterm context. In: B. Metz, O. Davidson, P. Bosch, R. Dave, L. Meyer (eds.), *Climate change 2007: Mitigation of climate change. Contribution of working group III to the fourth assessment report of the intergovernmental panel on climate change*. Cambridge University Press, Cambridge, UK and New York, NY, pp.169–250.

Fishman, J., et al., 2010: An investigation of widespread ozone damage to the soybean crop in the upper Midwest determined from ground-based and satellite measurements. *Atmospheric Environment,* 44(2010): 2248–2256.

Flörke, M., and J. Alcamo, 2005: European Outlook On Water Use. *Prepared for the European Environment Agency*, pp.83.

Foster, J. S., J. W. Winchester, and E. G. Dutton, 1992: The date of snow disappearance on the Arctic tundra as determined from satellite, meteorological station and radiometric in situ observations. *IEEE Transactions on Geoscience and Remote Sensing,* 30(4): 793–798.

Fowler, et al., 2009: Atmospheric composition change: Ecosystesm-Atmosphere interactions. *Atmospheric Environment,* 43: 5193–5267.

Frauenfeld, O. W., T. Zhang, R. G. Barry, and D. Gilichinsky, 2004: Interdecadal changes in seasonal freeze and thaw depths in Russia. *Journal of Geophysical Research,* 109: D05101.

Fuhrer, J., 2003: Agroecosystern responses to combinations of elevated CO2, ozone, and global climate change. *Agriculture, Ecosystems and Environment,* 97: 1–20.

Fuhrer, J., 2009: Ozone risk for crops and pastures in present and future climates. *Naturwissenschaften,* 96: 173–194.

Fuhrer, J. and F. Booker, 2003: Ecological Issues related to ozone: agricultural issues. *Environment International,* 29(2–3): 141–154.

Fuhrer, J., L. Skärby, and M. R. Ashmore, 1997. Critical levels for ozone effects on vegetation in Europe. *Environmental Pollution,* 97: 91–106.

Fujino, J., R. Nair, M. Kainuma, T. Masui, and Y. Matsuoka, 2006: Multigas mitigation analysis on stabilization scenarios using AIM global model. *Energy Journal,* 27: 343–354.

Galloway J., A. Townsend, J. W. Erisman, Z. Cai, F. Martinelli, S. Seitzinger, and M. Sutton, 2008: Transformation of the nitrogen cycle: Recent trends, questions and potential solutions. *Science*, 320: 889–892.

Galloway, J. N., J. D. Aber, J. W. Erisman, S. P. Seitzinger, R. W. Howarth, E. B. Cowling, and B. J. Cosby, 2003: The Nitrogen Cascade. *BioScience,* 53(4): 341–356.

Garrett, T. J., L. F. Radke, and P. V. Hobbs, 2002: Aerosol effects on the cloud emissivity and surface longwave heating in the Arctic. *Journal of the Atmospheric Sciences,* 59: 769–778.

Geerben-Leenes, W., A. Y. Hoekstra, and T. H. van der Meer, 2009: The water footprint of bioenergy. *Proceedings of the National Academy of Sciences of the United States of America,* 106(25): 10219–10223.

Global Biodiversity Outlook 3, 2010: Secretariat of the Convention on Biological Diversity, Montreal. www.cbd.int/gbo3/ebook/ (accessed 14 March 2010).

Gordon, L. J., G. D. Peterson, and E. B. Bennett, 2008: Agricultural modifications of hydrological flows create ecological surprises. *Trends in Ecology and Evolution,* 23(4): 211–219.

Gordon, L., W. Steffen, B. F. Jönsson, C. Folke, M. Falkenmark, and Å. Johannessen, 2005: Human modification of global water vapour flows from the land surface. *PNAS,* 105: 7612–7617.

Gosnell, H., Kelly, K., 2010: Peace on the river? Social-ecological restoration and large dam removal in the Klamath basin, USA. *Water Alternatives,* 3(2): 362–383.

Grennfelt, P. and Ø. Hov, 1995: Regional Air Pollution at a turning point. *Ambio,* 34(1): 2–10.

Gruber, N. and J. N. Galloway, 2008: An Earth-System perspective of the global nitrogen cycle. *Nature,* 451: 293–296.

Guerin, F., G. Abril, S. Richard, B. B. Burban, C. Reynouard, P. Seyler, and R. Delmas, 2006: Methane and carbon dioxide emissions from tropical reservoirs: significance of downstream rivers. *Geophysical Research Letters,* 33: L21407.

Hansen, J., R. Ruedy, M. Sato, M. Imhoff, W. Lawrence, D. Easterling, T. Peterson, and T. Karl, 2001: A closer look at United States and global surface temperature changes. *Journal of Geophysical Research,* 106: 947–963.

Hansen, J., M. Sato, P. Kharecha, D. Beerling, R. Berner, V. Masson-Delmotte, M. Pagani, M. Raymo, D. L. Royer, and J. S. Zachos, 2008: Target atmospheric CO2: Where should humanity aim? *Open Atmospheric Science Journal,* 2: 217–231.

Hansen, J., R. Reudy, M. Sato, and K. Lo, 2010: Global surface temperature change. *Reviews of Geophysics*, 48: RG4004.

Hanson, C. E, J. P. Palutikof, A. Dlugolecki, and C. Giannakopoulos, 2006: Bridging the gap between science and the stakeholder: the case of climate change research. *Climate Research,* 13: 121–133.

Hare, B. And M. Meinshausen, 2006: How much warming are we committed to and how much can be avoided? *Climatic Change,* 75(1–2): 111–149.

Harris, G., 2002: Integrated assessment and modelling: an essential way of doing science. *Environmental Modelling & Software*, 17(3): 201–207.

Hassan, I., M. Ashmore, and J. Bell, 1995: Effect of ozone on radish and turnip under Egyptian field conditions. *Environmental Pollution,* 89(1): 107–114.

Heck, W. W, O. C. Taylor, and D. T. Tingey (eds.), 1988: *Assessment of Crop Loss from Air Pollutants.* Barking, UK: Elsvier Applied Science.

Hernandez, T., and L. I. de Bauer, 1984: Evolucion del daño por gases oxidantes en Pinus hartwegii P. montezumae var. Lindleyi en el Ajusco. *D. F. Agrociencia*, 56:183–194.

Hernandez, T., and L. I. de Bauer 1986: Photochemical oxidant damage on Pinus hartwegii at the Desierto de los Leones, *D. F. Phytopathology,* 76(3): 377.

Hettelingh, F.-P., M. Posch, J. Slootweg, G. J. Reinds, T. Spranger, and L. Tarrason, 2007: Critical loads and dynamic modelling to assess Eruopean areas at risk of acidification and eutrophication. *Water, Air and Soil Pollution Focus,* 7: 379–384.

Hettelingh, J. P., M. Posch, and J. Slootweg, 2008: *Critical Load, Dynamic Modelling and Impact Assessment in Europe: CCE Status Report 2008. Coordination Centre for Effects, Netherlands Environmental Assessment Agency*, Bilthoven, Netherlands.

Hicks, W. K., J. C. I. Kuylenstierna, A. Owen, F. Dentener, H. M. Seip, and H. Rodhe, 2008: Soil sensitivity to acidification in Asia: Status and prospects. *Ambio,* 37: 295–303.

Hoelzle, M., W. Haeberli, M. Dischl, and W. Peschke, 2003: Secular glacier mass balances derived from cumulative glacier length changes. *Global and Planetary Change,* 36: 295–306.

Holland, M, C. Bitz, and B. Tremblay, 2006: Future abrupt reductions in the summer Arctic sea ice. *Geophysical Research Letters,* 33(L23503): 1–5.

Howarth, R., D. Anderson, J. Cloern, C. Elfring, C. Hopkinson, B. Lapointe, T. Malone, N. Marcus, K. McGlathery, A. Sharpley, and D. Walker, 2000: Nutrient pollution of coastal rivers, bays, and seas. *Ecological Society of America,* 7: 2–15.

Huang, J., C. Zhang, and J. M. Prospero, 2009: Large-scale effect of aerosols on precipitation in the West African Monsoon region. *Quarterly Journal of the Royal Meteorological Society,* 135: 581–594.

Hungate, B. A., et al., 2009: Nitrogen and climate change. *Science,* 203: 1512–1513.

IEA, 2009: *World energy outlook 2009. International Energy Agency, OECD Publication Service*, OECD, Paris.

IJC, 2010: *US and Canada Air Quality Agreement Progress Report*. International Joint Commission (IJC).

International Rivers, 2011: www.internationalrivers.org/ (accessed 24 March 2011).

IPCC, 1990: *First Assessment Report (FAR) – Climate Change 1990*. Cambridge University Press, Cambridge, UK and New York, NY.

IPCC, 1995a: *Second Assessment Report (SAR) – Climate Change 1995*. Cambridge University Press, Cambridge, UK and New York, NY.

IPCC, 1995b: *The Science of climate change. Contribution of Working Group I to the Second Assessment Report of the Intergovernmental Panel on Climate Change*. J. T. Houghton, L. G. Meira Filho, B. A. Callander, N. Harris, A. Kattenberg and K. Maskell (eds.), Cambridge University Press, Cambridge, UK, New York, NY, and Melbourne, Australia, pp.531.

IPCC, 2000a: Emission scenarios. N. Nakicenovic, and R. Swart, (eds.). *The Science of climate change. Contribution of Working Group I to the Second Assessment Report of the Intergovernmental Panel on Climate Change*, pp.570. Cambridge University Press, Cambridge, UK and New York, NY.

IPCC, 2000b: Methodological and technological issues in technology transfer. Metz, B., Davidson, O., Martens, J.-W., Van Rooijen, S. and Van Wie Mcgrory, L. (eds.), *The Science of climate change. Contribution of Working Group I to the Second Assessment Report of the Intergovernmental Panel on Climate Change*, Cambridge University Press, Cambridge, UK and New York, NY, pp.432.

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

IPCC, 2001b: *Impacts, Adaptation, and Vulnerability. Contribution of Working Group II to the Third Assessment Report of the Intergovernmental Panel on Climate Change*. J. J. McCarthy, O. F. Canziana, N. A. Leary, D. J. Dokken, K. S. White, (eds.), Cambridge University Press, Cambridge, UK and New York, NY, pp.976.

IPCC, 2005: *Carbon dioxide capture and storage*, B. Metz, O. Davidson, H. de Coninck, M. Loos, L. Meyer, (eds.), Cambridge University Press, Cambridge, UK and New York, NY. pp.431.

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

IPCC, 2007b: *Impacts, adaptation and vulnerability. Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change*, M. L. Parry, O. F. Canziani, J. P. Palutikof, P. J. van der Linden, and C. E. Hanson (eds.), Cambridge University Press, Cambridge, UK and New York, NY.

IPCC, 2007c: *Mitigation of climate change. Contribution of Working Group III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change*, B. Metz, O. Davidson, P. Bosch, L. A. Meyer, (eds.), Cambridge University Press, Cambridge, UK and New York, NY, USA.

Jackson, R. B, G. J. Esteban, A. Roni, S. B. Roy, D. J. Barrett, C. W. Cook, K. A. Farley, D. C. le Maitre, B. A. McCarl, and B. C. Murray, 2005: Trading water for carbon with biological carbon sequestration. *Science,* 310(5756): 1944–1947.

Jackson, C., 2009: Parallel pursuit of near-term and long-term climate mitigation. *Science,* 326(5952): 526–527.

Janssens, I. A. and S. Luyssaert, 2009: Carbon cycle: Nitrogen's carbon bonus. *Nature Geoscience,* 2: 318–319.

Johannessen, O. M., 2008: Decreasing Artic Sea Ice mirrors increasing CO2 on Decadal time scale. *Atmospheric and Oceanic Science Letters,* 1: 51–56.

Johnson, F. X. and I. Virgin, 2010: Future Trends in biomass resources for food and fuel. In *Food versus Fuel: an informed introduction*. F. Rosillo-Calle and F. X. Johnson (eds.), ZED books, London.

Jones, P. D., and A. Moberg, 2003: Hemispheric and large scale surface air temperature variations: an extensive revision and an update to 2001. *Journal of Climate,* 16, 206–223.

Josipovic, M., H. J. Annegarn, M. A. Kneen, J. J. Pienaar, and S. J. Piketh, 2010: Concentrations, distributions and critical level exceedance assessment of SO2, NO2 and O3 in South Africa. *Environmental Monitoring and Assessment,* 171: 181–196.

Karlsson, P. E., et al., 2007: Risk assessments for forest trees: The performance of the ozone flux versus the AOT concepts. *Environmental Pollution,* 146, 608–616.

Karnosky, D. F., and K. C. Steiner, 1981: Provenance variation in response of Fraxinus americana and Fraxinus pennsylvanica to sulfur dioxide and ozone. *Phytopathology,* 71: 804–807.

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

Karnosky, D. F., et al., 2007: Perspectives regarding 50 years of research on effects of tropospheric ozone air pollution on US forests. *Environmental Pollution,* 147: 489–506.

Keeling, C. D. and T. P. Whorf, 2005: *Atmospheric CO^2 records from sites in the SIO air sampling network. In Trends: A compendium of data on global change. Carbon Dioxide Information Analysis Centre*, Oak Ridge National Laboratory, US Dept. of Energy, Oak Ridge, TN.

King, J. S., et al., 2005: Tropospheric O3 compromises net primary production in young stands of trembling aspen, paper birch and sugar maple in response to elevated atmospheric CO_2. *New Phytologist,* 168: 623–636.

Klein Tank, A. M. G., J. B. Wijngaard, G. P. Konnen, R. Bohm, G. Demaree, A. Gocheva, M. Mileta, S. Pashiardis, L. Hejkrlik, C. Kern-Hansen, R. Heino, P. Bessemoulin, G. Muller-Westermeier, M. Tzanakou, S. Szalai, T. Palsdottir, D. Fitzgerald, S. Rubin, M. Capaldo, M. Maugeri, A. Leitass, A. Bukantis, R. Aberfeld, A.F.V. VanEngelen, E. Forland, M. Mietus, F. Coelho, C. Mares, V. Razuvaev, E. Nieplova, T. Cegnar, J.A. López, B. Dahlstrom, A. Moberg, W. Kirchhofer, A. Ceylan, O. Pachaliuk, L.V. Alexander, and P. Petrovic, 2002: Daily dataset of 20th-century surface air temperature and precipitation series for the European Climate Assessment. *International Journal of Climatology,* 22: 1441–1453.

Klein, R. J. T., R. J. Nicholls, and F. Thomalla., 2002: *The resilience of coastal megacities to weather-related hazards. Building Safer Cities: The Future of Disaster Risk*, A. Kreimer, M. Arnold and A. Carlin, Eds., The World Bank Disaster Management Facility, Washington, DC, 101–120.

Klimont, Z. Z., Cofala, J., Xing, J., Wei, W., Zhang, C., Wang, S., Bhandari, P., Mathur, R., Purohit, P., Rafaj, P., Chambers, A., Amann, M., Hao, J., 2009: Projections of SO2, NOx and carbonaceaous aerosols emissions in Asia. *Tellus,* 61: 602–617.

Klimont, Z., Streets, D. G., Gupta, S., Cofala, J., Fu, L., Ichikawa, Y., 2002: Anthropogenic emissions of non-methane volatile organic compounds in China. *Atmospheric Environment*, 36(8): 1309–1322

Koch, D., T. C. Bond, D. Streets, N. Unger, 2007: Linking future aerosol radiative forcing to shifts in source activities. *Geophysical Research Letters,* 34, L05821

Kriegler, E., J. W. Hall, H. Held, R. Dawson, H. J. Schellnhuber, 2009: Imprecise probability assessment of tipping points in the climate system. *Proceedings of the National Academy of Sciences of the United States of America, USA,* 106: 5041–5046.

Krupa, S. V., et al., 2004: Effects of ozone on plant nutritive quality characteristics for ruminant animals. *The Botanica,* 54: 1–12.

Kuylenstierna J. C. I, H. Rodhe, S. Cinderby, and K. Hicks, 2001: Acidification in developing countries: Ecosystem sensitivity and the critical load approach on a global scale. *Ambio,* 30: 20–28.

Law, K. S., A. Stohl, 2007: Arctic air pollution: Origins and impacts. *Science,* 315: 1537–1540.

Le Quéré, C., M. Raupach, J. Canadell, G. Marland, et al., 2009: Trends in the sources and sinks of carbon dioxide. *Nature Geoscience,* 2: 831–836.

Le Quéré, C., C. Rodenbeck, E. Buitenhaus, T. Conway, R. Lagenfelds, A. Gomez, C. Labuschagne, M. Ramonet, T. Nakazawa, N. Metzl, N. Gillett, M. Heimann, 2007: Saturation of the southern ocean CO2 sink due to recent climate change. *Science,* 316: 1735–1738.

Lehner, B., G. Czisch, and S. Vassolo, 2005: The impact of global change on the hydropower potential of Europe: a model-based analysis. *Energy Policy,* 33: 839–855.

Leiva, J. C., 2006: *Assesment climate change impacts on the water resources at the northern oases of Mendoza province, Argentina. Global Change in Mountain Regions*, M. F. Price (ed.), Sapiens Publishing, Kirkmahoe, Dumfriesshire, 81–83.

Lenton, T. M., H. Held, E. Kriegler, J. W. Hall, W. Lucht, S. Rahmstorf, and H. J. Schellnhuber, 2008: Tipping elements in the Earth's climate system. *Proc Natl Acad Sci USA,* 105: 1786–1793.

Levin, L. A., W. Ekau, A. J. Gooday, F. Jorissen, J. J. Middelburg, S. W. Naqvi, C. Neira, N. N. Rabalais, and J. Zhang, 2009: Effects of natural and human-induced hypoxia on coastal benthos. *Biogeosciences,* 6(10): 2063–2098.

Living Planet Report, 2010: Biodiversity, biocapacity and development, WWF. wwf.panda.org/about_our_earth/all_publications/living_planet_report/ (accessed 26 March 2011).

Locatelli, B. and R. Vignola, 2009: Managing watershed services of tropical forests and plantations: can meta-analyses help? *Forest Ecology and Management,* 258: 1664–1870.

Long, S. P., E. A. Ainsworth, A. D. B. Leakey, and P. B. Morgan, 2005: Global food insecurity. Treatment of major food crops with elevated carbon dioxide or ozone under large-scale fully open-air conditions suggests recent models may have overestimated future yields. *Philosophical Transactions of the Royal Society B,* 360: 2011–2020.

Lu, Z., D. G. Streets, Q. Zhang, S. Wang, G. R. Carmichael, Y. F. Cheng, C. Wei, M. Chin, T. Diehl, and Q. Tan, 2010: Sulfur dioxide emissions in China and sulphur trends in East Asia since 2000. *Atmospheric Chemistry and Physics,* 10: 6311–6331.

Lugina, K. M., et al., 2005: Monthly surface air temperature time series area-averaged over the 30-degree latitudinal belts of the globe, 1881–2004. In: *Trends: A Compendium of Data on Global Change. Carbon Dioxide Information Analysis Center*, Oak Ridge National Laboratory, US Department of Energy, Oak Ridge, TN, cdiac.esd.ornl.gov/trends/temp/lugina/lugina.html (accessed 3 April 2011).

MacKay, 2008: Sustainable Energy – without the hot air. UIT, Cambridge, UK. www.withouthotair.com/ (accessed 22 January 2011).

Magnani, F., M. Mencuccini, P. Borghetti, P. Berbigier, F. Berninger, S. Delzon, et al., 2007: The human footprint in the carbon cycle of temperate and boreal forests. *Nature,* 447: 849–851.

Magrin, G., C. Gay García, D. Cruz Choque, J. C. Giménez, A. R. Moreno, G. J. Nagy, C. Nobre, and A. Villamizar, 2007: *Climate Change 2007: Impacts, Adaptation and Vulnerability. Chapter 13: Latin America. Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change*, M. L. Parry, O. F. Canziani, J. P. Palutikof, P. J. van der Linden, and C. E. Hanson (eds.), Cambridge University Press, Cambridge, UK and New York, NY, 581–615.

Maslanik, J., C. Fowler, J. Stroeve, S. Drobot, J. Zwally, D. Yi, and W. Emery, 2007: A younger, thinner Arctic ice cover: increased potential for rapid, extensive sea-ice loss. *Geophysical Research Letters,* 34: L24501.

Matyssek, R., and H. J. Sandermann, 2003: Impact of ozone on trees: An ecophysiological perspective. *Progress in Botany,* 64: 349–404.

McDonald, R. I., J. Fargione, J. Kiesecker, W. M. Miller, and J. Powell, 2009: Energy Sprawl or Energy Efficiency: Climate Policy Impacts on Natural Habitat for the United States of America. *PLoS ONE,* 4(8): e6802.

Meadows, D. H., and D. L. Meadows, 1972: *The Limits to Growth.* New American Library, New York, NY.

Meadows, D. H., D. L. Meadows, and J. Randers, 1992: *Beyond the limits: Confronting global collapse-envisioning a sustainable future*. Chelsea Green Publishing Company, Post Mills, VT.

Meadows, D. H., J. Randers, and D. L. Meadows, 2004: *Limits to Growth – the 30 year update*. Earthscan, London.

Meehl, G. A. et al., 2007: *Global climate projections. In Climate change 2007: the physical science basis. Contribution of Working Group I to the 4th Assessment Report of the Intergovernmental Panel on Climate Change*. S. Solomon, D. Qin,

M. Manning, Z. Chen, M. Marquis, K. B. Averyt, M. Tignor, and H. L. Miller (eds.), Cambridge University Press, Cambridge, UK and New York, NY.

Meinshausen, M., N. Meinshausen, W. Hare, S. C. B. Raper, K. Frieler, R. Knutti, D. J. Frmae, M. R. Allen, 2009: Greenhouse-gas emission targets for limiting global warming to 2°C. *Nature,* 458: 1158–1162.

Mendelsohn, R., A. Dinar, and A. Dalfelt, 2000: *Climate change impacts on African agriculture*. Preliminary analysis prepared for the World Bank, Washington, DC.

Menz, F. C. and H. M. Seip, 2004: Acid rain in Europe and the United states: an update. *Environmental Science and Policy,* 7(4): 253–265.

Menzel, A., T. H. Sparks, N. Estrella, E. Koch, A. Aasa, R. Ahas, K. Alm-Kübler, P. Bissoli, O. Braslavska, A. Briede, F. M. Chmielewski, Z. Crepinsek, Y. Curnel, Å. Dalh, C. Defila, A. Donnelly, Y. Filella, K. Jatczak, F. Måge, A. Mestre, Ø. Nordli, J. Peñuelas, P. Pirinen, V. Remišová, H. Scheifinger, M. Striz, A. Susnik, A. VanVliet, F.-E. Wielgolaski, S. Zach, and A. Zust, 2006: European phenological response to climate change matches the warming pattern. *Glob. Change Biol.,* 12: 1969–1976.

Metzger, M. J., R. Leemans, D. Schröter, W. Cramer, and the ATEAM consortium, 2004: *The ATEAM Vulnerability Mapping Tool*. Quantitative Approaches in System Analysis No. 27. Wageningen, C.T. de Witt Graduate School for Production Ecology and Resource Conservation, Wageningen, CD ROM.

Meybeck, 2003: M. Global analysis of river systems: from earth system controls to Anthropocene controls. *Philosophical Transactions of the Royal Academy, London B,* 358: 1935–1955.

MEA (Millennium Ecosystem Assessment) 2005: *Millennium Ecosystem Assessment* 2005, Synthesis, Island Press, Washington, DC, USA.

Miller, P. and J. McBride (eds.), 1999: *Oxidant air pollution impacts in the Montane forests of Southern California: The San Bernardino Mountain Case Study*. Springer-Verlag, New York, USA.

Miller, S., 2009: Minimizing Land Use and Nitrogen Intensity of Bio-energy. *Environmental Science & Technology,* 44: 3932–3939.

Mills, G., et al., 2007: A synthesis of AOT40-based response functions and critical levels of ozone for agricultural and horticultural crops, *Atmospheric Environment,* 41: 2630–2643.

Mills, G., et al., 2011: Evidence of widespread effects of ozone on crops and (semi-) natural vegetation in Europe (1990–2006) in relation to AOT40 and flux-based risk maps, *Global Change Biology,* 17: 592–613.

Milly, P. C. D., R. T. Wetherald, K. A. Dunne, and T. L. Delworth, 2002: Increasing risk of great floods in a changing climate. *Nature,* 415: 514–517.

Molden, D. (ed.), 2007. *Water for food, water for life: A Comprehensive Assessment of Water Management in Agriculture, Earthscan*, London, and IWMI, Colombo.

Molina, M. J. and F. S. Rowland, 1974: Stratospheric sink for chlorofluoromethanes: chlorine atomc-atalysed destruction of ozone. *Nature,* 249(5460): 810–812.

Morgan, P. B., et al., 2006, Season-long elevation of ozone concentration to projected 2050 levels under fully open-air conditions substantially decreases the growth and production of soybean. *New Phytologist,* 170: 11.

Moriondo, M., P. Good, R. Durao, M. Bindi, C. Gianakopoulos, and J. Corte-Real, 2006: Potential impact of climate change on fire risk in the Mediterranean area. *Climate Research,* 31: 85–95.

Muller, N. Z. and R. Mendelsohn, 2007: Measuring the damages of air pollution in the United States. *Journal of Environmnetal Economics and Management,* 54: 1–14.

Mylona, S., 1996: Sulphur dioxide emissions in Europe 1880–1991, *Tellus,* 48B: 662–689.

Mylona, S., 1997: Corrigendum to Sulphur dioxide emissions in Europe 1880–1991, *Tellus,* 49B: 447–448.

Nabuurs, G. J., et al., 2003: Temporal evolution of the European forest sector carbon sink from 1950 to 1999, *Global Change Biology,* 9: 152–160.

Nakicenovic, N. and R. Swart (eds.), 2000: *Emissions Scenarios: A Special Report of Working Group III of the Intergovernmental Panel on Climate Change*. Cambridge University Press, Cambridge, UK and New York, NY.

Nakicenovic, N. and K. Riahi (eds.), 2007: Integrated assessment of uncertainties in greenhouse gas emissions and their mitigation, *Technological Forecasting and Social Change,* 74(2007): 873–886.

NRC (National Research Council), 2002: Defining the mandate of proteomics in the post-genomics era: Workshop report. *Molecular and Cellular Proteomics,* 1(10): 763–780.

NRC (National Research Council), 2000: *Clean Coastal Waters: Understanding and Reducing the Effects of Nutrient Pollution*. National Academy Press, Washington, DC.

NEA (National Ecosystem Assessment), 2011: *The UK National Ecosystem Assessment: Synthesis of the Key Findings*. UNEP-WCMC, Cambridge, UK.

Nicholls, R., and R Tol, 2006: Impacts and responses to sea-level rise: a global analysis of the SRES scenarios over the twenty-first century. *Philosophical Transactions of the Royal Society A,* 364: 1073–1095.

Nicholls, R. J., 2004: Coastal flooding and wetland loss in the 21st century: changes under the SRES climate and socio-economic scenarios. *Global Environmental Change-Human and Policy Dimensions,* 14(1): 69–86.

Nilsson, C., C. A. Reidy, M. Dynesius, and C. Revenga, 2005: Fragmentation and flow regulation of the world's large river systems. *Science,* 308(5720): 405–408.

Ohara, T., H. Akimoto, J. Kurokawa, N. Horii, K. Yamaji, X. Yan, and T. Hayasaka, 2007: An Asian emission inventory of anthropogenic emission sources for the period 1980–2020. *Atmospheric Chemistry and Physics,* 7: 4419–4444.

Okello, B. D., T. G. O'Connor, and T. P. Young, 2001: Growth, biomass estimates, and charcoal production of Acacia drepanolobium in Laikipia, Kenya. *Forest Ecology and Management,* 142: 143–153.

Olesen, J. E., T. R. Carter, C. H. Díaz-Ambrona, S. Fronzek, T. Heidmann, T. Hickler, T. Holt, M. I. Mínguez, P. Morales, J. Palutikof, M. Quemada, M. Ruiz- Ramos, G. Rubæk, F. Sau, B. Smith, and M. Sykes, 2007: Uncertainties in projected impacts of climate change on European agriculture and terrestrial ecosystems based on scenarios from regional climate models. *Climatic Change,* 81: S123-S143.

Olivier, J., J. Van Aardenne, F. Dentener, L. Ganzeveld, and J. Peters, 2005: *Recent trends in global greenhouse gas emissions: regional trends and spatial distribution of key sources, in: Non-CO2 Greenhouse Gases (NCGG-4)*. A. van Amstel (ed.), Millpress, Rotterdam, the Netherlands. pp.325–330.

Pacnya, E., and J. Pacnya, 2002: Global emission of mercury from anthropogenic sources in 1995. *Water, Air & Soil Pollution,* 137(1–4): 149–165.

Pacnya, E., J. Pacyna, F. Steenhuisen, and S. Wilson, 2006: Global anthropogenic mercury emission inventory for 2000. *Atmospheric Environment,* 40(22): 4048–4063.

Peng, S., J. Huang, J. E. Sheehy, R. E. Laza, R. M. Visperas, X. Zhong, G. S. Centeno, G. S. Khush, and K. G. Cassman, 2004: Rice yields decline with higher night temperature from global warming. *Proceedings of the National Academy of Science of the United States of America,* 101: 9971–9975.

Penner, J. E., M. J. Prather, I. S. A. Isaksen, J. S. Fuglestvedt, Z. Klimont, and D. S. Stevenson, 2010: Short-lived uncertainty? *Nature Geoscience,* 3(2010): 587–588.

Perovich, D., T. Grenfell, and P. Hobbs, 2002: Seasonal evolution of the albedo of multi-year Arctic sea-ice. *Journal of Geophysical Research-Oceans,* 107: SHE20–21.

Phoenix, G. K., W. K. Hicks, S. Cinderby, J. C. I. Kuylenstierna, W. D. Stock, F. J. Dentener, K. E. Giller, A. T. Austin, R. D. B. Lefroy, B. S. Gimeno, M. R. Ashmore, and P. Ineson, 2006: Atmospheric nitrogen deposition in world biodiversity hotspots: the need for a greater global perspective in assessing N deposition impacts. *Global Change Biology,* 12: 470–476.

Pirrone, N., and R. Mason, 2008: *Mercury Fate and Transport in the Global Atmosphere, Measurements, Models and Policy Implications, Interim Report of the UNEP Global Mercury Partnership*, Mercury Air Transport and Fate Research Partnership Area.

Pleijel, H. et al., 1999: Grain protein accumulation in relation to grain yield of spring wheat (Triticum aestivum L.) grown in open-top chambers with different concentrations of ozone, carbon dioxide and water availability. *Agriculture, Ecosystems & Environment,* 72(3): 265–270.

Postel, S. L., 1998: Water for food production: will there be enough in 2025? *BioScience,* 48: 629–638.

Prather, M., R. Derwent, D. Ehhalt, P. Fraser, E. Sanheuza, and X. Zhou, 1995: *Other trace gases and atmospheric chemistry. In: Climate change 1994: Radiative forcing and climate change and an evaluation of the IPCC IS92 Emission Scenarios*. J. T. Houghton, et al. (eds.) Cambridge University Press, Cambridge, UK and New York, NY, pp.73–126.

Pu, J. C., T. D.Yao, N. L.Wang, Z. Su, and Y. P. Shen, 2004: Fluctuations of the glaciers on the Qinghai-Tibetan Plateau during the past century. *Journal of Glaciology and Geocryology,* 26: 517–522.

Quinn, P. K., Shaw, G., Andrews, E., Dutton, E. G., Ruoho-Airola, T., Gong, L., 2007: Arctic Haze: Current trends and knowledge gaps. *Tellus,* 59B: 99–114.

Raatz, K. A., 1984: Observations of Arctic haze during the Ptarmigan weather reconnaissance flights 1948–1961. *Tellus,* 36B: 126–136.

Rabalais, N. N., 2002: Nitrogen in aquatic ecosystems. *Ambio,* 31(2): 102–112.

Raes, F., and J. H. Seinfeld, 2009: New Directions: Climate change and air pollution abatement: a bumpy road. *Atmospheric Environment,* 43: 5132–5133.

Rahn, K. A., R. D. Borys, and G. E. Shaw, 1981: Asian desert dust over Alaska: anatomy of an Arctic haze Episode, Desert Dust. In: *Origin, Characteristic and Effect on Man,* T. Pewe (ed.), Special paper No. 186, The Geological Society of America, Boulder, Colorado, pp.37–70.

Ramanathan, V. and G. Carmichael, 2008: Global and regional climate changes due to black carbon. *Nature Geoscience,* 1: 221–227.

Ramanathan, V. and Y. Feng, 2008: On avoiding dangerous anthropogenic interference with the climate system: Formidable challenges ahead. *Proceedings of the National Academy of Sciences of the United States of America,* 105(38): 14245–14250.

Ramanathan, V. and Y. Xu, 2010: The Copenhagen Accord for limiting global warming: Criteria, constraints and available avenues. *Proceedings of the National Academy of Sciences of the United States of America,* 107(18): 8055–8062.

Ramaswamy, V., et al., 2001: *Radiative forcing of climate change. In: Climate Change 2001: The Scientific Basis. Contribution of Working Group I to the Third Assessment Report of the Intergovernmental Panel on Climate Change*. J. T. Houghton, et al., (eds.), Cambridge University Press, Cambridge, UK and New York, NY, pp.349–416.

Ramírez C. A. and E. Worrell, 2006: Feeding fossil fuels to the soil: An analysis of energy embedded and technological learning in the fertilizer industry. *Resources, Conservation and Recycling,* 46: 75–93.

Reason, C. J. C., and M. Rouault, 2005: Links between the Antarctic Oscillation and winter rainfall over western South Africa. *Geophysical Research Letters,* 32(L07705).

Reay, D., F. Dentener, P. Smith, J. Grace, and R. Feely, R., 2008: Global nitrogen deposition and carbon sinks. *Nature Geoscience,* 1: 430–437.

Republic of Kenya, 2002: *First National Communication of Kenya to the Conference of Parties to the United Nations Framework Convention on Climate Change*. Ministry of Environment and Natural Resources, Nairobi.

République de Côte d'Ivoire, 2000: *Communication Initiale de la Côte d'Ivoire*. Ministère de l'Environnement, de l'Eau et de la Forêt, Abidjan.

République de Guinée, 2002: *Communication Initiale de la Guinée à la Convention Cadre des Nations Unies sur les Changements Climatiques*. Ministère des Mines, de la Géologie et de l'Environnement, Conakry.

Riahi, K., A. Grübler, and N. Nakicenovic, 2007: Scenarios of long-term socio-economic and environmental development under climate stabilization. *Technological Forecasting and Social Change,* 74: 887–935.

Richardson, K., W. Steffen, H. J. Schellnhuber, J. Alcamo, T. Barker, D. M. Kammen, R. Leemans, D. Liverman, M. Munasinghe, B. Osman-Elasha, N. Stern, and O. Wæve, 2009: *Synthesis Report: Climate Change- Global Risks, Challenges & Decisions*. International Alliance of Research Universities. www.climatecongress.ku.dk (accessed 14 April 2011).

Richter, B., S. Postel, C. Revenga, T. Scudder, B. Lehner, A. Churchill, and M. Chow, 2010: Lost in development's shadow: The downstream human consequences of dams. *Water Alternatives,* 3(2): 14–42.

Rockström, J., W. Steffen, K. Noone, A. Persson, F. S. Chapin, E. F. Lambin, T. M. Lenton, M. Scheffer, C. Folke, H. Schellnhuber, B. Nykvist, C. A. De Wit, T. Hughes, S. Van der Leeuw, H, Rodhe, S. Sörlin, P. K. Snyder, R. Costanza, U. Svedin, M. Falkenmark, L. Kalberg, R. W. Corell, V. J. Fabry, J. Hansen, B. Walker, D. Liverman, K. Richardson, P. Crutzen, and J.A. Foley, 2009: A safe operating space for humanity. *Nature,* 461: 472–475.

Rodhe, H., J. Langner, L. Gallardo, and E. Kjellstrom, 1995. Global scale transport of acidifying pollutants. *Water, Air, & Soil Pollution,* **85(1)**: 37–50.

Rowell, A., and P. F. Moore, 2000: *Global Review of Forest Fires*. WWF/IUCN, Gland, Switzerland.

Royal Society, 2008: *Ground-level ozone in the 21st century: future trends, impacts and policy implications*. Science Policy Report 15/08, The Royal Society, London, UK.

Schaeffer, S. M., J. B. Miller, B. H. Vaughn, J. W. C. White, and D. R. Bowling, 2008: Long-term field performance of a tunable diode laser absorption spectrometer for analysis of carbon isotopes of CO2 in forest air. *Atmospheric Chemistry and Physics,* 8(17): 5263–5277.

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

Schellnhuber, H. J., 2008: Global warming: Stop worrying, start panicking? *Proceedings of the National Academy of Sciences of the United States of America,* 105: 14239–14240.

Schellnhuber, H.J., 2009: Tipping elements in the Earth System. *Proceedings of the National Academy of Sciences of the United States of America Special Issue,* 106(49): 20561–20563.

Schneider, S. H. and M. D. Mastrandrea, 2005: Probabalistic assessment of 'dangerous' climate change and emissions pathways. *Proceedings of the National Academy of Sciences of the United States of America,* 102: 15728–15735.

Schopp, W., M. Amann, J. Cofala, C. Heyes, and Z. Klimont, 1999: Integrated assessment of European air pollution emission control strategies. *Environmental Modelling and Software,* 14(1): 1–9.

Schwela, D., G. Haq, C. Huizenga, W.-J. Han, H. Fabian, and M. Ajero, 2006: *Urban Air pollution in Asian Cities: Status, Challenges and Management*. Earthscan, London.

Seidel, D. J., and W. J. Randel, 2006: Variability and trends in the global tropopause estimated from radiosonde data. *Journal of Geophysical Research Atmospheres,* 111(D21101).

Selman, M., and S. Greenhalgh, 2009: *Eutrophication: Sources and drivers of nutrient pollution*. WRI Policy Note. Water Quality: Eutrophication Hypoxia No. 2. World Policy Institute, Washington, DC.

Shaw, G. E., 1995: The Arctic Haze Phenomenon. *American Meteorological Society,* 76: 2403–2413.

Shi, L., L. Xing, G. F. Lu, and J. Zou, 2008: Evaluation of rational sulphur dioxide abatement in China. *International Journal of Environmental Pollution,* 35: 42–57.

Shiklomanov, I. A., and J. C. Rodda, 2003: *World Water Resources at the Beginning of the 21st Century*. United Nations Educational, Scientific and Cultural Organization (UNESCO) and Cambridge University Press, Cambridge, UK and New York, NY.

Shiyatov, S. G., M. M. Terent'ev, and V. V. Fomin, 2005: Spatiotemporal dynamics of forest-tundra communities in the polar Urals. *Russian Journal of Ecology,* 36, 69–75.

Sitch, S., P. M. Cox, W. J. Collins, and C. Huntingford, 2007: Indirect radiative forcing of climate change through ozone effects on the land-carbon sink. *Nature,* 448: 791–794.

Skarby, L., et al., 1998: Impacts of ozone on forests: A European perspective. *New Phytologist,* 139: 109–122.

Sladen, W. J. L., C. M. Menzie, and W. L. Reichel, 1966: DDT residues in Adélie penguins and a crabeater seal from Antarctica. *Nature,* 210: 670–673.

Sliggers, J. and W. Kakebeeke, 2004: *Clearing the Air: 25 years of the Convention on Long-Range Transboundary Air Pollution*. United Nations ECE/EB.AIR/84.

Smith, J. B. et al., 2001: Impacts, Adaptation, and vulnerability. In *Climate Change 2001*. J. McCarthy, O. Canziana, N. Leary, D. Dokken, K. White (eds.), Cambridge University Press, Cambridge, UK and New York, NY, pp.913–967.

Smith, J. B., H. Schneider, M. Oppenheimer, G.W. Yohe, W. Hare, M. D. Mastrandrea, A. Patwardhan, I. Burton, J. Corfee-Morlot, C. H. D. Magadza, H.-M. Füssel, A. B. Pittock, A. Rahman, A. Suarez, and J.-P. van Ypersele, 2009: Assessing dangerous climate change through an update of the Intergovernmental Panel on Climate change (IPCC) "reasons for concern." *PNAS,* 106: 4133–4137.

Smith, K., 1997: The potential for feedback effects induced by global warming on emissions of nitrous oxide form soils. *Global Change Biology,* 3: 327–338.

Smith, T. M. and R. W. Reynolds, 2005: A global merged land and sea surface temperature reconstruction based on historical observations (1880–1997). *Journal of Climate,* 18: 2021–2036.

Smith, S. J., van Aardenne, J., Klimont, Z., Andres, R., Volke, A., Delgado Arias, S., 2011: Anthropogenic sulfur dioxide emissions: 1850–2005. *Atm. Chem. & Phys*., 11(3): 1101–1116

Sokolov, A., D. Kicklighter, J. Melillo, B. Felzer, C. Schlosser, and T. Cronin, 2008: Consequences of considering carbon-nitrogen interactions on the feedbacks between climate and the terrestrial carbon cycle. *Journal of Climate,* 21(15): 3776–3796.

Sokolov, A., P. Stone, C. Forest, C. Prinn, M. Sarofim, M. Webster, S. Paltsev, and C. Schlosser, 2009: Probabilistic Forecast for Twenty-First-Century Climate Based on Uncertainties in Emissions (Without Policy) and Climate Parameters. *Journal of Climate*, 22: 5175–5204.

Solomon S., R.R. Garcia, F.S. Rowland, and D. J. Wuebbles, 1986: On the Depletion of Antarctic Ozone. *Nature,* 321(6072): 755–758.

State of Eriteria, 2001: *Eritrea's Initial National Communication under the United Nations Framework Convention on Climate Change (UNFCCC)*. Ministry of Land, Water, and Environment. Asmara.

Steffen, W., J. Crutzen, and J. R. McNeill, 2007: The Anthropocene: are humans now overwhelming the great forces of Nature? *Ambio,* 36(8): 614–621.

Stevenson, D. S., F. J. Dentener, M. G. Schultz, K. Ellingsen, T. P. C. van Noije, O. Wild, G. Zeng, M. Amann, C. S. Atherton, N. Bell, D. J. Bergmann, I. Bey, T. Butler, J. Cofala, W. J. Collins, R. G. Derwent, R. M. Doherty, J. Drevet, H. J. Eskes, A. M. Fiore, M. Gauss, D. A. Hauglustaine, L. W. Horowitz, I. S. A. Isaksen, M. C. Krol, J. F. Lamarque, M. G. Lawrence, V. Montanaro, J. F. Muller, G. Pitari, M. J. Prather, J. A. Pyle, S. Rast, J. M. Rodriguez, M. G. Sanderson, N. H. Savage, D. T. Shindell, S. E. Strahan, K. Sudo, and S. Szopa, 2006: Multimodel ensemble simulations of present-day and near-future troposheric ozone. *Journal of geophysical research-atmospheres,* 111(8): D08301.

Stieb et al., 2002: Meta-analysis of time-series studies of air pollution and mortality: Effects of gases and particles and the influence of cause of death, age and season. *Journal of the Air and Waste management association,* 52(4): 470–484.

Stige, L., J. Stave, K. Chan, L. Ciannelli, N. Pettorelli, M. Glantz, H. Herren, and N. Stenseth, 2006: The effect of climate variation on agro-pastoral production in Africa. *PNAS,* 103(9): 3049–3053.

Stigiiani, W. M., and R. W. Shaw, 1990: Energy use and acid deposition: The view from Europe. *Annual Review of Energy,* 15: 201–216.

Stocks, B. J. et al., 1998: Climate change and forest fire potential in Russian and Canadian boreal forests. *Climatic Change,* 38: 1–13.

Stohl, A., 2006, Characteristics of atmospheric transport into the Arctic troposphere. *Journal of Geophysical Research-Atmospheres,* **111**: D11306.

Stone, R., 2007: A world without corals? *Science,* 316(5825): 678–681.

Stroeve, J., M. Serreze, S. Drobot, S. Gearheard, M. Holland, J. Maslanik, W. Meier, and T. Scambos, 2008: Arctic sea ice extent plummets in 2007. *EOS Transactions of the American Geophysical Union,* 89: 13–14.

Svensson, R., and M. Wigren-Svensson, 1992: Effects of cooling water discharge on the vegetation in the Forsmark Biotest Basin, Sweden. *Aquatic Botany,* 42(2): 121–141.

Teixeira, T., L. Neves, and F. Araújo, 2009: Effects of a nuclear power plant thermal discharge on habitat complexity and fish community structure in Ilha Grande Bay, Brazil. *Marine Environmental Research,* 68(4): 188–195.

Titus, J. G., 2005: *Sea-level rise effect. Encyclopaedia of Coastal Science*. M. L. Schwartz (ed.), Springer, Dordrecht, pp.838–846.

Tong, D., et al., 2007: The use of air quality forecasts to assess impacts of air pollution on crops: methodology and case study. *Atmospheric Environment,* 41: 8772–8784.

Toth, F., 2003: Climatic policy in light of climate science: The iclips project. *Climatic Change,* 56: 7–36.

Tsunehiko, O., F. Norio, L. Hu, H. Hiroshi, S. Hiroyuki, S. Massahi, and S. Katsunori, 2001: Quality control and its constraints during the preparatoryphase activities of the acid deposition monitoring network in East Asia. *Water, Soil and Air Pollution,* 131: 1613–1618.

Turner, J., S. R. Colwell, G. J. Marshall, T. A. Lachlan-Cope, A. M. Carleton, P. D. Jones, V. Lagun, P. A. Reid, and S. Lagovkina, 2005: Antarctic climate change during the last 50 years. *International Journal of Climatology,* 25: 279–294.

UN, 2008: World Population Prospects, the 2008 revision. United Nations Department of Economic and Social Affairs. esa.un.org/unpd/wpp2008/index.htm (accessed 10 June 2011).

UN,2011: National Accounts database. United Nations Department of Economic and Social Affairs unstats.un.org/unsd/snaama/introduction.asp (accessed 9 September 2010).

UNDP, 2000: *World Energy Assessment: Energy and the Challenge of Sustainability*. UNDP, New York.

UNECE, 2004: *25 Years of the Convention on Long-Range Transboundary Air Pollution*. United Nations Economic Commission for Europe (UNECE), New York.

UNECE 2007: *Strategies and Policies for Air Pollution Abatement. 2006 Review prepared under The Convention on Long-range Transboundary Air Pollution*. United Nations Economic Commission for Europe (UNECE), Geneva.

UNECE Task Force on Hemispheric Transport of Air Pollution, 2007: Hemispheric Transport of Air Pollution 2007, Air pollution studies No. 16. United Nations Economic Commission for Europe (UNECE), Geneva.

UNEP, 2006: Report on Atmosphere and Air Pollution. Africa Regional Implementation Review for the 14th Session of the Commission on Sustainable Development (CDS-14). United Nations Environmental Programme (UNEP), Nairobi.

UNEP/GRID-Arendal, 2005: Climate change vulnerability in Africa. In UNEP/GRID-Arendal Maps and Graphics Library. maps.grida.no/go/graphic/climate_change_vulnerability_in_africa (accessed 22 July 2011).

UNFCCC, 1992: *United Nations Framework Convention on Climate Change*. FCCC/INFORMAL/84. United Nations (UN), New York and Geneva.

USAID, 2002: USAID response to drought in southern Africa. www.usaid.gov/press/releases/2002/fs020510.html (accessed 12 March 2011).

US EPA, 2006: *Air quality criteria for ozone and related photochemical oxidants (final)*. US Environmental Protection Agency (US EPA). Washington, DC.

USEPA 2008. Integrated Science Assessment for Oxides of Nitrogen and Sulfur – Ecological Criteria National (Final Report). National Center for Environmental Assessment, Research Triangle Park, NC. US Environmental Protection Agency (US EPA). EPA/600/R-08/139. cfpub.epa.gov/ncea/cfm/recordisplay.cfm?deid=201485 (accessed 22 November 2010).

USEPA 2010a: AirNow web site. US Environmental Protection Agency (US EPA). www.epa.gov/aircompare/compare-trip.htm (accessed 22 November 2010).

USEPA 2010b: Acid Rain and Related Programs: 2009 Environmental Results. US Environmental Protection Agency (US EPA). Washington, DC. www.epa.gov/airmarkets/progress/ARP09_3.html (accessed 20 November 2010).

USEPA 2011a: National Ambient Air Quality Standards. US Environmental Protection Agency (US EPA). www.epa.gov/air/criteria.html (accessed 20 November 2011).

USEPA 2011b: Ozone National Ambient Air Quality Standards: Scope and Methods Plan for Welfare Risk and Exposure Assessment. Office of Air Quality Planning and Standards. US Environmental Protection Agency (US EPA).

USEPA 2011c: Regulatory Impact Analysis for the Federal Implementation Plans to Reduce Interstate Transport of Fine Particulate Matter and Ozone in 27 States; Correction of SIP Approvals for 22 States. Office of Air and Radiation (US EPA). http://www.epa.gov/crossstaterule/pdfs/FinalRIA.pdf (accessed 20 November 2011).

Ramanathan, V., M. Agrawal, H. Akimoto, M. Auffhammer, H. Autrup, L. Barregard, P. Bonasoni, M. Brauer, B. Brunekreef, G. Carmichael, W.-C. Chang, U. K. Chopra, C. E. Chung, S. Devotta, J. Duffus, L. Emberson, Y. Feng, S. Fuzzi, T. Gordon, A.K. Gosain, S. I. Hasnain, N. Htun, M. Iyngararasan, A. Jayaraman, D. Jiang, Y. Jin, N. Kalra, J. Kim, M. G. Lawrence, S. Mourato, L. Naeher, T. Nakajima, P. Navasumrit, T. Oki, B. Ostro, Trilok S. Panwar, M.R. Rahman, M. V. Ramana, H. Rodhe, M. Ruchirawat, M. Rupakheti, D. Settachan, A. K. Singh, G. St. Helen, P. V. Tan, S.K. Tan, P. H. Viet, J. Vincent, J. Y. Wang, X. Wang, S. Weidemann, D. Yang, S. C. Yoon, J. Zelikoff, Y. H. Zhang, and A. Zhu, 2008: *Atmospheric Brown Clouds: Regional Assessment Report with Focus on Asia*. Published by the United Nations Environment Programme, Nairobi, Kenya.

Valigura, R. A., R. B. Alexander, M. S. Castro, T. P. Meyers, H. W. Paerl, P. E. Stacy, and R. E. Turner, 2001: *Nitrogen Loading in Coastal Water Bodies: An Atmospheric Perspective*. American Geophysical Union, Washington, DC.

van Aardenne, J. A., F. J. Dentener, J. G. J. Olivier, C. G. M. K. Goldewijk, and J. Lelieveld, 2001: A 1 degrees x 1 degrees resolution data set of historical anthropogenic trace gas emissions for the period 1890–1990. *Global Biogeochemical Cycles,* 15 (4): 909–928.

van Aardenne, J.A., G. R. Carmichael, H. Hiram Levy II, D. Streets, and L. Hordijk, 1999. Anthropogenic NOx emissions in Asia in the period 1990–2020. *Atmospheric Environment,* **33**: 633–646.

van Dingenen, R., F. J. Dentener, F. Raes, M. C. Krol, L. D. Emberson, and J. Cofala, 2009: The global impact of ozone on agricultural crop yields under current and future air quality legislation. *Atmospheric Environment,* 43: 604–618.

van Tienhoven, A. M., M. Zunckel, L. Emberson, A. Koosailee, and L. Otter, 2006: Preliminary assessment of risk of ozone impacts to maize (Zea mays) in southern Africa. *Environmental Pollution,* 140: 220–230.

van Vuuren, D. P., M. den Elzen, P. Lucas, B. Eickhout, B. Strengers, B. van Ruijven, S. Wonink, and R. van Houdt, 2007: Stabilizing greenhouse gas concentrations at low levels: an assessment of reduction strategies and costs. *Climatic Change,* 81: 119–159.

Vanguelova, E. I., S. Benham, R. Pitman, A. J. Moffat, M. Broadmeadow, T. Nisbet, D. Durrant, N. Barsoum, M. Wilkinson, F. Bochereau, T. Hutchings, S. Broadmeadow, P. Crow, P. Taylor, and T. D. Houston, 2010: Chemical fluxes in time through forest ecosystems in the UK – Soil response to pollution recovery. *Environmental Pollution,* 158(5): 1857–1869.

Velders, G. J. M., S. O. Andersen, J. S. Daniel, D. W. Fahey, and M. McFarland, 2007: in *The Montreal Protocol, Celebrating 20 years of environmental progress*, D. Kaniaru (ed.), Cameron May, London, UK, pp.285–290.

Vestreng, V., G. Myhre, H. Fagerli, S. Reis, and L. Tarrason, 2007: Twenty-five years of continuous sulphur dioxide emission reduction in Europe. *Atmospheric chemistry and physics,* 7(13): 3663–3681.

Vestreng, V., L. Ntziachristos, A. Semb, S. Reis, I.S.A. Isaksen, and L. Tarrason, 2009: Evolution of NO*x* emissions in Europe with focus on road transport control measures. *Atmospheric Chemistry and Physics,* 9: 1503–1520.

Vingarzan, R., 2004: A review of surface ozone background levels and trends. *Atmospheric Environment,* 38: 3431–3442.

Vitousek, P. M., H. A. Mooney, J. Lubchenco, and J.M. Melillo, 1997: Human domination of Earth's ecosystems. *Science,* 277: 494–499.

Vörösmarty, C., M. Meybeck, B. Fekete, K. Sharma, P. Green, and J. Syvitski, 2003: Anthropogenic sediment retention: major global impact from registered river impoundments. *Global and Planetary Change*, 39(1–2): 169–190.

Wafula, E., M. Kinyanjui, L. Nyabola, and E. Tenambergen, 2000: Effect of improved stoves on prevalence of acute respiration infection and conjunctivitis among children and women in a rural community in Kenya. *East African Medical Journal,* 77(1): 37–41.

Walker, I. J., and J. V. Barrie, 2006: Geomorphology and sea-level rise on one of Canada's most 'sensitive'coasts: northeast Graham Island, British Columbia, Canada. *Journal of Coastal Research SI,* 39: 220–226.

Wallack, J. S., and V. Ramanathan, 2009: The other Climate Changers: Why black carbon and ozone also matter. *Foreign Affairs,* 88(5): 105–113.

Walther, G.-R., S. Beissner and C.A. Burga, 2005: Trends in upward shift of alpine plants. *Journal of Vegetation Science,* 16, 541–548.

Wang X., and D. L. Mauzerall, 2004: Characterizing distributions of surface ozone and its impact on grain production in China, Japan and South Korea: 1990 and 2020. *Atmospheric Environment,* 38: 4383–4402.

Wei Wei., Wang, S., Chatani, S., Klimont, Z., Cofala, J., Hao, J., 2008: Emission and speciation of non-methane volatile organic compounds from anthropogenic sources in China. *Atmospheric Environment* 42(20): 4976–4988

WGBU (German Advisory Council on Global Change), 2009: *Solving the Climate Dilemma: The Budget Approach*, WGBU, Berlin, Germany.

White, A., M. G. R. Cannel, and A. D. Friend, 2000: The high-latitude terrestrial carbon sink: a model analysis. *Global Change Biology,* 6: 227–246.

WHO, 2002: *Addressing the links between indoor air pollution, household energy and human health*. World Health Organization (WHO), Switzerland.

WHO 2006a: *Air quality guidelines: global update 2005, particulate matter, ozone, nitrogen dioxide and sulphur dioxide*. World Health Organization (WHO) Regional Office for Europe: Copenhagen, Denmark.

WHO 2006b: *Health risks of particulate matter from long-range transboundary air pollution. European Centre for Environment and Health, Report E88189*, World Health Organization (WHO), Switzerland.

Wilkinson C. R. (ed.), 2000: *Global Coral Reef Monitoring Network: Status of Coral Reefs of the World in 2000*. Townsville, Queensland: Australian Institute of Marine Science.

Wise, M., A. Calvin, A.Thomson, L. Clarke, B. Bond-Lamberty, R. Sands, S. J. Smith, A. Janetos, and J. Edmonds, 2009: Implications of limiting CO2 concetrations for land use and energy. *Science,* 324: 1183–1186.

World Commisssion on Dams, 2000: *Dams and development: a new framework for decision-making*. Report of the World Commission on Dams, London: Earthscan.

World Resources Institute (WRI), United Nations Development Programme, United Nations Environment Programme, and World Bank, 2008. World Resources 2008: Roots of Resilience – Growing the Wealth of the Poor. Washington, DC.

World Bank, 2007: Cost of pollution in China: Economic estimates of physical damages. The World Bank, State Environmental Protection Administration. P.R. China. www.worldbank.org/eapenvironment (accessed 20 February 2011).

World Bank Group, 2009: *Directions in hydropower*. The World Bank, Washington, DC.

World Water Assessment Programme, 2009: *The United Nations World Water Development Report 3: Water in a Changing World*. United Nations Educational, Scientific and Cultural Organization (UNESCO), Paris, and Earthscan, London.

WWF, 2005: *An overview of glaciers, glacier retreat, and subsequent impacts in Nepal, India and China*. World Wildlife Fund (WWF), Nepal Programme.

Xu-Ri, and I. Prentice, 2008: Terrestrial nitrogen cycle simulation with a dynamic global vegetation model. *Global Change Biology,* 14(8): 1745–1764.

Yi-Li, C., Y. Hsiao-Hui, and L. Hsing-Juh, 2009: Effects of a thermal discharge from a nuclear power plant on phytoplankton and periphyton in subtropical coastal waters. *Journal of Sea Research,* 61(4):197–205.

Zalasiewicz, J., M. Williams, W. Steffen, and P. Crutzen, 2010: The new world of the Anthropocene. *Environmental Science & Technology,* 44: 2228–2231.

Zhang, Q., D. G. Streets, G.R. Carmichael, K. He, H. Huo, H., A. Kannari, Z. Klimont, I. S. Park, S. Reddy, J. S. Fu, D. Chen, L. Duan, Y. Lei, L. T. Wang, and Z. L. Yao, 2009: Asian emissions in 2006 for the NASA INTEX-B mission. *Atmospheric Chemistry and Physics,* 9: 5131–5153.

Zhang, Q., D. Streets, K. He, Y. Wang, A Richter, J. Burrows, I. Uno, C. Jang, D. Chen, Z. Yao, and Y. Lei, 2007: NOx emission trends for China, 1995–2004: The view from the ground and the view from space. *Journal of Geophysical Research,* 112(D22306).