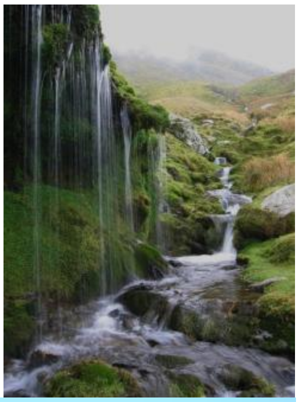
LRTAP
Long-range Transboundary Air Pollution

wge
Working Group on Effects
of the
Convention on Long-range Transboundary Air Pollution

Benefits of air pollution control for biodiversity and ecosystem services

REPORT BY THE WORKING GROUP ON EFFECTS





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Executive summary

The Earth's ecosystems provide an array of services upon which humans depend for food, fresh water, timber production, disease management, air and climate regulation, aesthetic enjoyment and spiritual fulfilment. Such 'Ecosystem Services' are currently grouped according to the benefits they provide to humans, distinguishing between provisioning (e.g. food, fresh water, fuel, wood), regulating (e.g. water purification, water and climate regulation, pollination), supporting (e.g. biomass production, soil formation, nutrient and water cycling) and cultural services (e.g. education, recreation, aesthetic). The role of biodiversity in ecosystem services is often rather unclearly stated – biodiversity is sometimes considered as a separate service and yet is implicit in most ecosystem services. Although humans are an integral part of ecosystems, the increased global population along with increased standards of living and other socio-political, economic, technological and societal changes, mean that our interventions can have profound negative effects on the quality of the services provided by ecosystems, hence affecting human well-being. The concept of ecosystem services has arisen in response to an increased need for making visible human dependency on nature and ecosystems, in order to ensure sustainable management and avoid irreversible damage to the ecosystems that ultimately will damage human well-being. Ecosystem services can capture a wider set of costs and benefits, not traditionally valued in economic analysis.

In this report we provide some examples of data, available from several International Cooperative Programmes (ICPs) under the Working Group on Effects of the Convention on Long-range Transboundary Air Pollution (LRTAP), on how air pollution abatement policies provide benefits to ecosystem services and biodiversity and how further benefits can be achieved in the future. The report is not an exhaustive review of the literature but more a compilation of the present knowledge used to provide policy-relevant information by the WGE. The advantages and disadvantages of valuation in monetary and non-monetary terms were also discussed.

Biodiversity

Deposition of reactive nitrogen currently is a threat for plant diversity and remains a threat in the foreseeable future. Particularly so as the effects of excessive nitrogen deposition on the structure and functioning of ecosystems and its biodiversity may not occur instantly, in some instances it may take several decades over which the resilience of soils and vegetation is weakened and impacts become apparent. Large areas in Europe still show exceedance of the nutrient nitrogen critical load and in acids grasslands a reduction in plant diversity due to elevated nitrogen deposition has been shown. So far, little is known about the recovery from historic nitrogen pollution; full recovery might not occur in the future, especially in areas where nitrogen-sensitive plant species have disappeared and where other drivers such as climate change have modified the environment. Assessments should be extended to other ecosystems and biodiversity indicators (e.g. presence of red list species, soil organisms) for a comprehensive analysis of impacts of excessive nitrogen deposition on biodiversity. Impacts of other atmospheric pollutants also need to be considered. For example, there is a trend towards an increase in the number of benthic invertebrates since the beginning of the 1980's that might be related to a recovery from acidification in fresh water systems across Europe. Also, experiments at different scales have shown that a shift in plant species composition can occur due to ozone exposure. Ozone-sensitive plant species might be outcompeted by more ozone-resistant plant species in areas where the 'uptake' of ozone by vegetation is high (i.e. high phytotoxic ozone dose). However, these observations need to be confirmed by further field-based evidence for impacts of ozone on plant species diversity.

Ecosystem services

Although elevated nitrogen deposition stimulates tree growth in areas where nitrogen is currently the limiting factor for growth, thereby enhancing timber production and the potential for carbon sequestration in forests ecosystems, forest health and vitality may be at risk when organic matter and nutrient cycling is disturbed due to nitrogen enrichment of forest soils. Soils play an important role in storage of air pollutants such as reactive nitrogen and heavy metals, thereby mitigating leaching of these pollutants to water ways and maintaining good water quality. However, the stored pollutants may adversely affect soil functioning (e.g. microbes and invertebrates) and create problems when the retention capacity is reached or disturbed, and pollutants start leaching to surface and drinking water, and coastal zones. Nitrogen leaches from forest soil at a carbon to nitrogen ratio below 23 in the organic layer and when the critical load is exceeded; excessive nitrogen input in lakes will enhance algal growth.

In contrast to nitrogen, current atmospheric ozone concentrations reduce tree growth, resulting in a decline in timber production and the potential for carbon sequestration in forests ecosystems. Hence, emission abatement policies that reduce the atmospheric concentrations of ozone precursors will be beneficial for forest growth and health. Vegetation is an important sink for ozone and therefore plays an important role in improving air quality and mitigating climate change. Ozone is the third most important greenhouse gas and the deposition of ozone to vegetation contributes significantly to a reduction in global warming. In addition, ozone has shown to be a threat to food security by reducing both yield quantity and quality of ozone sensitive species (e.g. wheat and soybean). Such impacts have been valued in monetary terms. In addition, ozone might adversely affect the pollination of flowers by for example affecting the synchronization of the time of flowering with the presence of pollinators or floral scent trails in plant-insect interactions. Current ambient ozone concentrations significantly reduce seed number, fruit number and fruit weight compared to pre-industrial ozone levels. Ozone has also been shown to affect water cycling via its impacts on the opening of leaf pores.

A good example of how air pollution abatement benefits ecosystem services has been the decline in sulphur deposition since the establishment of the LRTAP Convention in 1979. Acidification of surface waters in northern Europe due to sulphuric acid deposition had resulted in a loss of fish population and other organisms in many rivers and lakes. However, chemical conditions in many surface waters have improved since the mid-1980s and after a long lag period, biological recovery has started during the last decade. Fish species such as brown trout and salmon have returned, as well as other species such a mayfly and zooplankton. This is of huge benefit to recreational fishing in these areas. However, another problem for fishing is the high level of mercury that has accumulated in fish through the food chain. For example, in over half of the lakes in Sweden, the content of mercury in fish is higher than the recommended values for human consumption.

Conclusions

Based on this report, we draw the following conclusions:

- Awareness of ecosystem services, including biodiversity, in both monetary and non-monetary terms helps to assess the real benefits of air pollution control;
- It is very encouraging that there are signs of chemical and biological recovery from acidification. It remains uncertain whether full recovery of biodiversity from adverse effects of historic air pollution will be possible;
- Further air pollution abatement will continue to reduce the threat to loss of biodiversity, however, "no net loss of biodiversity" will not be achieved by 2020 under the revised Gothenburg Protocol;
- With full implementation of the revised Gothenburg Protocol, further benefits are expected for ecosystem services such as air, soil and water quality and crop production;
- Further air pollution abatement policies will enhance the resilience of biodiversity and ecosystem services to climate change.

Policy recommendations

Based on this report, we make the following policy recommendations:

- To halt biodiversity loss and adverse impacts of air pollution on human well-being, policy negotiations should take into account the benefits of air pollution control for ecosystem services in addition to the direct benefits for human health;
- More stringent air pollution abatement measures beyond the revised Gothenburg Protocol are required to achieve "no net loss of biodiversity";
- The full benefits of air pollution abatement for ecosystem services (and hence human well-being) have to be assessed and weighed against the costs of more stringent air pollution controls;
- The effects-based integrated assessment of policies that address driving forces of environmental issues could be further balanced by including "no net loss of biodiversity and ecosystem services" in air, waters, soils and vegetation as an explicit endpoint.

Content

1. Introduction

1.1 Ecosystem services – an introduction

The Earth's ecosystems provide an array of services upon which humans depend for food, fresh water, timber production, disease management, air and climate regulation, aesthetic enjoyment and spiritual fulfilment (Millennium Ecosystem Assessment, 2005). Such 'Ecosystem Services' are currently grouped according to the benefits they provide to humans, distinguishing between provisioning, regulating, supporting and cultural services (**Figure 1.1**). Provisioning services are the products obtained from ecosystems, such as food, fibre and wood/fuel. Regulating services refer to the regulation of e.g. climate, water quantity and quality. Cultural services are the non-material benefits people obtain from ecosystems through spiritual enrichment, cognitive development, reflection, recreation, and aesthetic experiences. Supporting services are those that are necessary for the production of all other ecosystem services. The role of biodiversity in ecosystem services is often rather unclearly stated – biodiversity is sometimes considered as a separate service and yet is implicit in most ecosystem services. The concept of ecosystem services has arisen in response to an increased need for making visible human dependency on nature and ecosystems, in order to ensure sustainable management and avoid irreversible damage to the ecosystems that ultimately will damage human well-being. Ecosystem services can capture a wider set of costs and benefits, not traditionally valued in economic analysis.

Although humans are an integral part of ecosystems, the increased global population along with increased standards of living and other socio-political, economic, technological and societal changes, mean that our interventions can have profound negative effects on the quality of the services provided by ecosystems, hence affecting human well-being. Because ecosystems are complex systems comprising animal, plant and microorganism communities together with the non-living environment (Millennium Ecosystem Assessment, 2005), these systems are inherently dynamic whilst maintaining some intrinsic resilience to natural disturbances. However, human-driven changes have become increasingly worrying, and thus many of the World's ecosystems and the services they provide are now degraded, or vulnerable to degradation. At a global level, it is estimated that nearly two thirds of ecosystem services have been degraded in just fifty years (Millennium Ecosystem Assessment, 2005).

The additional stresses imposed by climate change will require extraordinary adaptation (Mooney et al., 2009; Fu et al., 2013). Climate change is predicted to have both positive and negative effects on key ecosystem services, the results being sector and scenario specific (e.g. Forsius et al., 2013). For example, in Finland food and timber production would largely benefit from increasing temperatures and prolongation of the growing season in the cool Finnish conditions, although increasing occurrence of factors such as fungal diseases and insect outbreaks were estimated to cause increasing risks. On the other hand, climate change was predicted to pose a major threat to several endangered and valuable species, water and air quality, and tourism services dependent on present climate conditions. Goal conflicts between maximising service production and meeting environmental quality objectives were also identified. Controlled and spontaneous adaptation can, however, reduce the vulnerability of the different ecosystem services and sectors to climate change

(Fu et al., 2013). The need for unifying concepts, indicator development, and observation schemes for global change monitoring and analysis have also been identified (Vihervaara et al., 2013).

Global toxification (including air pollution) is one of the “savage sextet” (Aguirre, 2009) of direct drivers of ecosystem degradation, with the others being over-exploitation of species, introduction of novel exotic species, land use changes (principally habitat destruction, fragmentation and degradation), pathogen pollution and global warming (Mantyka-Pringle et al., 2012). Indirect drivers of ecosystem change are associated with demographic, economic, socio-political and cultural or religious changes, and advancements in science and technology. Stressed or degraded ecosystems do not have the resilience or re-bound capacity of pristine/unstressed systems (Rapport and Maffi, 2009). Furthermore, there is often a substantial time-lag between a change in a driver and the time taken to realize the full consequences of that change in any given system. Even more worrying is that once a threshold is crossed, a system may alter to a distinctly changed and sometimes irreversible new state. Careful management of our ecosystems and the benefits and services we derive from them are therefore vital for future prosperity and general human well-being.

Human influence extends into even the remotest landscapes and more often than not has a pervasive influence on the ecosystems they support, frequently irreversibly changing biodiversity. Whilst extinction rates of species are now estimated to be 1,000 times greater than historical background levels (Millennium Ecosystem Assessment, 2005; Mantyka-Pringle et al., 2012), recent studies have identified linkages between changes in biodiversity and ecosystem functioning, highlighting the importance of adopting a multi-sectoral approach to policy and decision making (e.g. Maestre et al., 2012; Mace et al., 2012). Such an approach fully evaluates changes in ecosystem services and their impacts on humans and examines the supply and condition of each ecosystem service, as well as the interactions among them. Society needs to make difficult decisions regarding its use of biological resources and environmental valuation techniques provide useful evidence to support polices by quantifying both the monetary and non-monetary value associated with the protection of resources. To support this drive, the Intergovernmental Platform on Biodiversity and Ecosystem Services (IPBES) was established in April 2012 by 90 governments and acts as a global mechanism for gathering, analyzing and synthesizing information to advise decision-making on biodiversity and ecosystem services (Redford et al., 2012). Further, possibilities for introducing human manufactured substitutions are limited for many ecosystem services, especially for supporting services. Therefore, preservation of functioning, and restoration of degraded systems is paramount at this time in history.

As shown in **Figure 1.1**, ecosystem services can be classified into provisioning, regulating, supporting and cultural services. When considering impacts of one driver of change (in this case air pollution), it immediately becomes clear that impacts on one service are linked to several and sometimes all of the other services. Complex interactions have been identified between the different ecosystem processes as well as trade-offs between the ecosystem services (Forsius et al., 2013; Smith et al., 2013). Because of such complexities and the growing desire to add an economic value to ecosystem services, the final ecosystem services that provide goods of value to humans can be considered to be linked by “stocks and flows” to the underpinning ecological processes (Mace et al., 2012).

The links between nature and the economy are often described using the concept of ecosystem services, or flows of value to human societies as a result of the state and quantity of natural capital (Millennium Ecosystem Assessment, 2005; TEEB, 2010). The objective of the Millennium Ecosystem Assessment (2005) conducted under the auspices of the United Nations was to assess the consequences of ecosystem changes for human well-being and the scientific basis for actions needed to enhance the conservation and sustainable use of those systems and their contributions to human well-being.

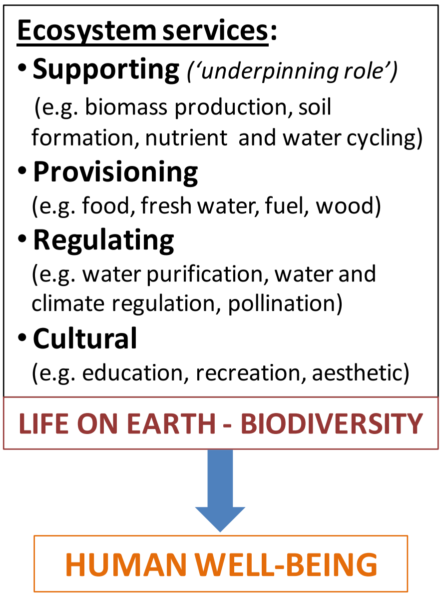

Figure 1.1 Ecosystems services are the benefits people obtain from ecosystems. These include provisioning, regulating, and cultural services that directly affect people and supporting services needed to maintain the other services (Modified after Millennium Ecosystem Assessment, 2005).

1.2 Biodiversity as an ecosystem service

Mace et al. (2012) showed how biodiversity is involved throughout the ecosystem hierarchy: "as a regulator of underpinning ecosystem processes, as a final ecosystem service and as a good that is subject to valuation." They described biodiversity contributions as being from both an "ecosystem services perspective", measured in simplest terms by ecosystem service flows, and from a "conservation perspective", where higher value is given to conserving charismatic species. There are many drivers of loss in biodiversity, with the increase in human population, especially in the last century, having a profound influence by, for example, increasing the need for biomass for fuel and construction, changes in land-use towards food and fodder production, industrial and residential developments, introduction of invasive species, pollution and climate change. Species losses are currently outpacing background rates calculated from fossil records (Millennium Ecosystem Assessment, 2005) and it is widely recognised that the earth is facing its sixth mass extinction (Barnosky et al., 2011). Some ecosystems are more resilient to change than others, with for example, primary forests being more resistant to change than modified natural forests or plantations (Thompson et al., 2009). Recently, many have emphasised the importance of biodiversity for ecosystem services, for example "biodiversity enhances the ability of ecosystems to maintain multiple functions" (Maestre et al., 2012), "species richness has positive impacts on ecosystem services" (Gamfeldt et al., 2013), "biodiversity decreases the occurrence of diseases through predictable changes in host community competence" (Johnson et al., 2013), "increased biodiversity enhances ecosystems services such as pollination and provide an opportunity to increase agricultural yields whilst also benefitting wildlife" (Brittain et al., 2013). It has been emphasized that many ecosystem services ultimately depend on the variety of life forms that comprise an ecosystem and that control the ecological processes that underlie all services. Therefore, a solid understanding of the linkages between biodiversity, ecosystem functioning and the production of ecosystem services is paramount (Cardinale et al., 2012).

1.3 Examples of the global significance of ecosystem services

1.3.1 Carbon cycle and primary productivity

Annual net primary productivity (NPP) is the net amount of carbon (C) captured by land plants through photosynthesis. It is of fundamental importance to humans because the largest proportion of our food supply is from plant productivity. Recent estimates of the global NPP range from 19.6 g C m^{-2} yr^{-1} to 43.5 g C m^{-2} yr^{-1} (Prieto-Blanco et al., 2009). Total global CO_2 emissions were estimated to be approximately 8.7 ± 0.5 Gt C yr^{-1} in 2008 and were shown to have increased at a rate of 3.4% per year between 2000 to 2008 (Le Quere et al., 2009). Most of the CO_2 emissions increase is from developing countries (non-Annex B countries) where emissions have more than doubled over the last decade. Shockingly, tropical deforestation is estimated to have released between 1-2 Gt (billion tonnes) of CO_2 per year during the 1990s (i.e. 15 – 25% of annual global emissions) (Gibbs et al., 2007). Despite the pressing need to reduce CO_2 emissions, Le Quere et al. (2009) report a rapid increase in fossil fuel CO_2 emissions since the 1990s and a dramatic increase in per-capita emissions since the early 2000s. Although around 55% of all anthropogenic CO_2 emissions are absorbed by land and ocean sinks (Friedlingstein and Prentice, 2010), a large quantity remains in the atmosphere.

World forests are a vital component in the global carbon cycle as they sequester and store more carbon than any other terrestrial ecosystem and are therefore a major natural sink for anthropogenic emissions (Gibbs et al., 2007). For example, total global forests sequester 1.4 Gt CO_2 annually, with temperate and boreal ecosystems sequestering 0.5 Gt CO_2 of this amount (Pan et al., 2011). European forests contribute around 10% of the global sequestration of carbon with Norway, Finland, Germany and Sweden having the greatest potential for CO_2 capture due to the large forested areas. Further, managed forests generally sequester carbon at a faster rate than natural forests (Pingoud et al., 2010). Any factor that increases primary productivity in temperate and boreal forests is likely to increase forest carbon sequestration; conversely any factor that negatively affects primary productivity will reduce CO_2 sequestration.

The CO_2 taken up by vegetation will be sequestered in the shorter or longer term in plant material or soils. Soils are the largest carbon reservoir of the terrestrial carbon cycle. Worldwide, they contain three to four times more organic carbon (1500 Gt to 1m, 2500 Gt to 2m depth) than vegetation (610 Gt) and twice or three times as much carbon as the atmosphere (750 Gt; Batjes and Sombroek, 1997). Carbon storage in soils is the balance between the input of dead plant material (leaf and root litter, decaying wood) and losses from decomposition and mineralization of organic matter (heterotrophic respiration). Under aerobic conditions, most of the carbon entering the soil returns to the atmosphere by autotrophic root respiration and heterotrophic respiration (together called soil respiration). Under anaerobic conditions, resulting from constantly high water levels, part of the carbon entering the soil is not fully mineralized and accumulates as peat.

1.3.2 Water cycling

Water is essential for life on Earth and supports all other ecosystem processes. Human water use has increased drastically over the last 50 years and is now double pre-1960 values. Most of this water (70% worldwide) is used for irrigation of crops. Estimated mean annual global land-surface evapotranspiration from vegetation is approximately 65 ± 3 x 10^3 km^3 per year, with forests, grasslands and crops accounting for 29 x 10^3 km^3, 21 x 10^3 km^3, and 7.6 x 10^3 km^3 respectively (Jung et al., 2010; Oki and Kanae, 2006). Any factor that acts to alter evapotranspiration will have potential effects on local/regional microclimate/climate and soil water status/hydrology (Blyth and Harding, 2011). Most of the water transpired by plants passes through the leaf (stomatal) pores, the diameter of which is in turn modified by external climatic and edaphic conditions such as light, temperature, soil moisture, and carbon dioxide (CO_2). Consequently, transpiration processes impact on the global hydrological cycle (Lombardozzi et al., 2012). Effects of air pollutants such as ozone on transpiration can be either positive or negative depending on species, episodic/background ozone characteristics and soil water availability (Mills et al., 2013).

1.3.3 Nutrient cycling

Nitrogen is a vital element determining the diversity, dynamics and functioning of many ecosystems. Numerous natural ecosystems have relatively low levels of nitrogen availability, for example, nitrogen deposition in the absence of human influence is typically about 0.5 kg N ha^{-1} yr^{-1}, whereas in many areas of the world nitrogen deposition rates now exceed 10 kg N ha^{-1} yr^{-1}, and are often higher. Alarmingly, by 2050 nitrogen deposition rates could reach 50 kg N ha^{-1} yr^{-1} in some regions (Galloway et al., 2008). The two main anthropogenic drivers of nitrogen loading into natural (eco)systems are agriculture practices and combustion of fossil fuels. Estimations surmise that more than half of all synthetic nitrogen fertilizer ever used on the planet has been used since 1985, and as such, humans have doubled the flow of reactive nitrogen within natural and man-made ecosystems. Worryingly, this nitrogen burden is anticipated to increase by a further 66% by 2050 (Millennium Ecosystem Assessment, 2005). Oxidized nitrogen concentrations in the atmosphere have also increased dramatically during the last 100 years, largely arising from combustion sources. Total reactive nitrogen is now estimated to be greater than 0.187 Mt yr^{-1} (formally 0.015 Mt yr^{-1} in the late 1800's), with about 70% arising from food production (fertilizers) (Galloway et al., 2003; Galloway et al., 2008). Unsurprisingly, both these anthropogenic sources have increased the cycling of fixed/reactive nitrogen through ecosystems and changed species composition and ecosystem dynamics globally.

1.4 Working Group on Effects and International Cooperative Programmes

Established in 1980 as one of the working bodies of the Convention on Long-range Transboundary Air Pollution (LRTAP), the Working Group on Effects of Sulphur Compounds, later the Working Group on Effects (WGE), started its activities with the first meeting in 1981 in Geneva. Over the last 30 years, the WGE has contributed to the demonstrable improvements the Convention has achieved, e.g. in reducing acidification of ecosystems, reducing the highest peak levels of ozone and the albeit considerably smaller reduction of emissions of nitrogen compounds. Six International Cooperative Programmes (ICPs) and a Task Force on Health Effects of Air Pollution (Task Force Health) form the WGE. Their work covers a variety of receptors (forests, surface waters, vegetation, materials and people) and activities (monitoring, modelling, mapping, scenario analysis and policy advice). The WGE addresses many interlinking environmental issues: nitrogen enrichment ('eutrophication'), acidification, ground-level ozone pollution, particulate matter impacts, health effects, corrosion, contamination by heavy metals and persistent organic pollutants. Consequences for biodiversity and interactions with climate change are also high on the agenda. A Joint Expert Group on Dynamic Modelling supports exchange of research between dynamic modelling efforts of the ICPs.

The following ICPs have contributed to the current report:

- ICP Vegetation (http://icpvegetation.ceh.ac.uk)
- ICP Modelling and Mapping (http://www.icpmapping.org) and the Coordination Centre for Effects (http://www.wge-cce.org)
- ICP Waters (http://www.icp-waters.no)
- ICP Integrated Monitoring (http://www.syke.fi/nature/icpim)
- ICP Forests (http://icp-forests.net)

The Joint Expert Group on Dynamic Modelling also contributed to the discussions.

Current activities and future challenges of the WGE include:

- Perform long-term monitoring of air pollution impacts in widespread networks across the UNECE region and case studies at plots and catchments with intensive measurements;
- Provide information on the degree and geographic extent of impacts of air pollution on human health and the environment;
- Demonstrate relationships between concentrations of air pollutants and effects on human health and the environment using policy relevant indicators;
- Conduct scientific research on dose-response functions to establish acceptable thresholds of air pollution for ecosystems ('critical loads and levels');
- Apply models to evaluate the success of air pollution abatement policies in terms of benefits for the environment and human health and assess the impacts of future emission scenarios.

1.5 Aims and structure of this report

The concept of ecosystem services has arisen in response to an increased need for making visible human dependency on nature and ecosystems, in order to ensure sustainable management and avoid irreversible damage to the ecosystems that ultimately will damage human well-being. The aim of this report is to provide examples of how air pollution control is of benefit to ecosystem services and biodiversity. It is not an exhaustive review of the literature but more a compilation of the present knowledge used to provide policy-relevant information by the WGE. The benefits of reducing nitrogen enrichment of the environment and the formation of ground-level ozone for biodiversity, particularly plant diversity, are being explored in Chapter 2. Subsequently, examples of the benefits of air pollution control for ecosystem services are described in Chapter 3, followed by a discussion on the valuation of ecosystem services in Chapter 4. Conclusions and policy recommendations are provided in Chapter 5.

2. Impacts on biodiversity

2.1 The revised Gothenburg Protocol contributes to reduction of harmful nitrogen effects

Terrestrial eutrophication continues to be a serious threat to European ecosystems. In 1980, critical loads of nutrient nitrogen were exceeded in about 67% of the European area (80% in the EU27), which is expected to decrease to around 42% (62% in the EU27) in 2020 under the Revised Gothenburg Protocol[1] (RGP2020; **Figure 2.1a**). Whilst the area at risk is remaining high, the average accumulated exceedance (AAE) shows a significant reduction between 1980 and 2020 (**Figure 2.1b**). This reduction may delay effects on biodiversity, but will stand in the way of full recovery.

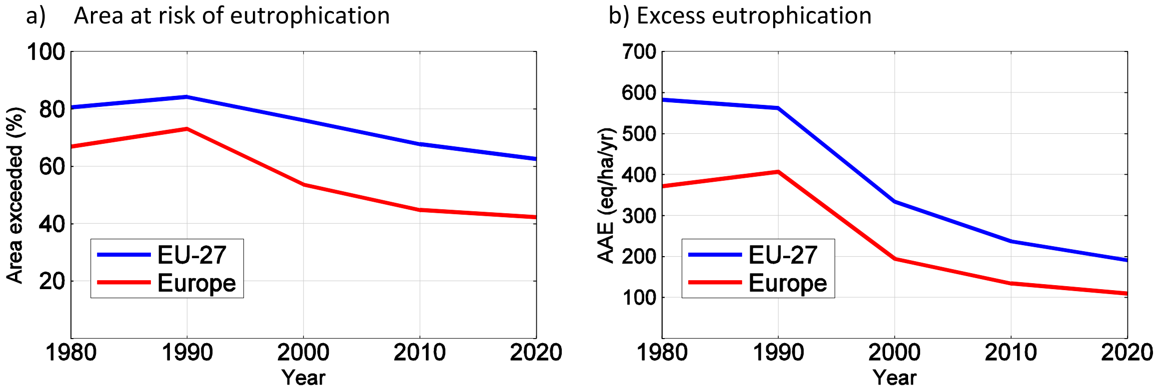

Figure 2.1 Trend between 1980 and 2020 (Revised Gothenburg Protocol) of a) the area where critical loads of nutrient nitrogen are exceeded and b) the Average Accumulated Exceedance (AAE) for eutrophication in the EU-27 and in Europe.

The trend between 1980 and 2020 of the distribution over Europe of areas where critical loads for eutrophication are exceeded confirms the continued stress to European ecosystems, in Central Europe in particular (**Figure 2.2**). The broad Central European area of high exceedances in 1980 (red shading) is markedly reduced in 2020, but still occurs in western France and the border areas between the Netherlands, Belgium and Germany, as well as in northern Italy. The country-specific trend since 1980 of the area at risk of eutrophication is summarized in Annex 1. The (hypothetical) implementation of maximum technically feasible reduction (MFR) of emissions of acidifying and eutrophying pollutants would yield a further increase of areas that are protected, whilst areas with high exceedances of critical loads would further decrease (**Figure 2.3**). However, even under maximum (technically) feasible reductions of nitrogen emissions, the deposition of nitrogen continues to put a large area at risk, implying that the potential of technical measures alone is not sufficient to achieve non-exceedance of critical loads for eutrophication.

[1] The Gothenburg protocol has been revised in 2012 under the LRTAP Convention. “RGP2020” refers to a scenario where pollutants emissions are decreased from 2020 according to the conditions agreed under the revised Gothenburg Protocol.

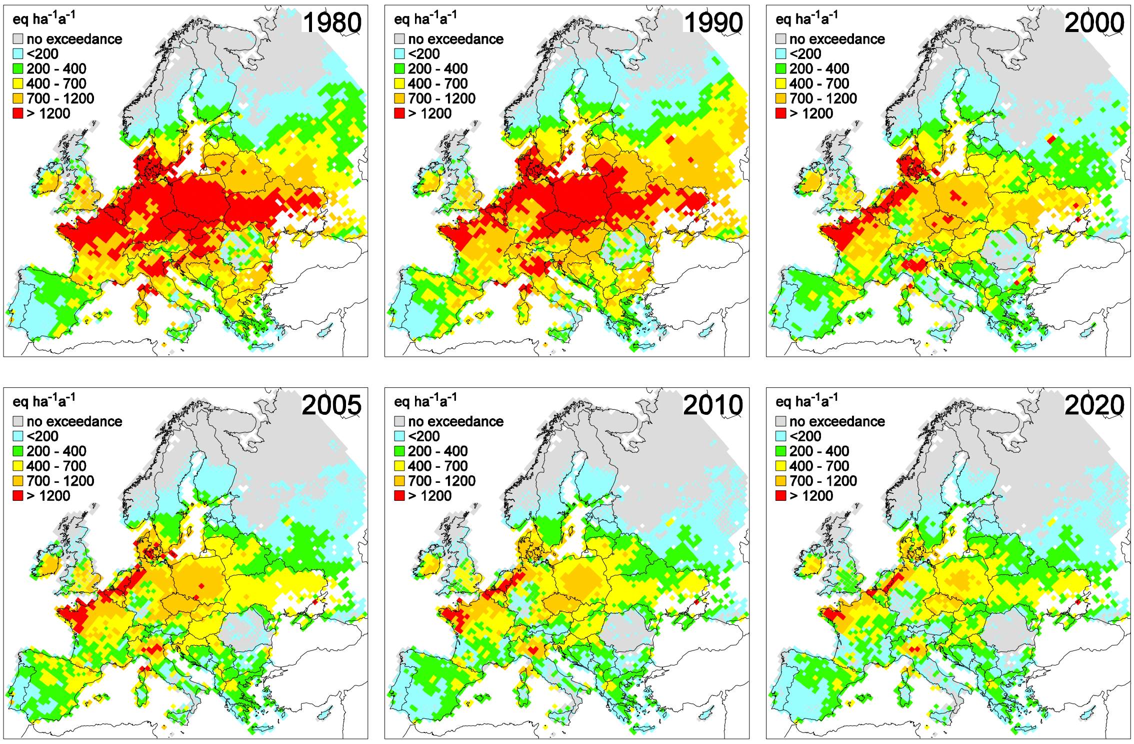

Figure 2.2 Areas where critical loads for eutrophication are exceeded by nutrient nitrogen depositions caused by emissions between 1980 and 2020, the last projected under the Revised Gothenburg Protocol (RGP).

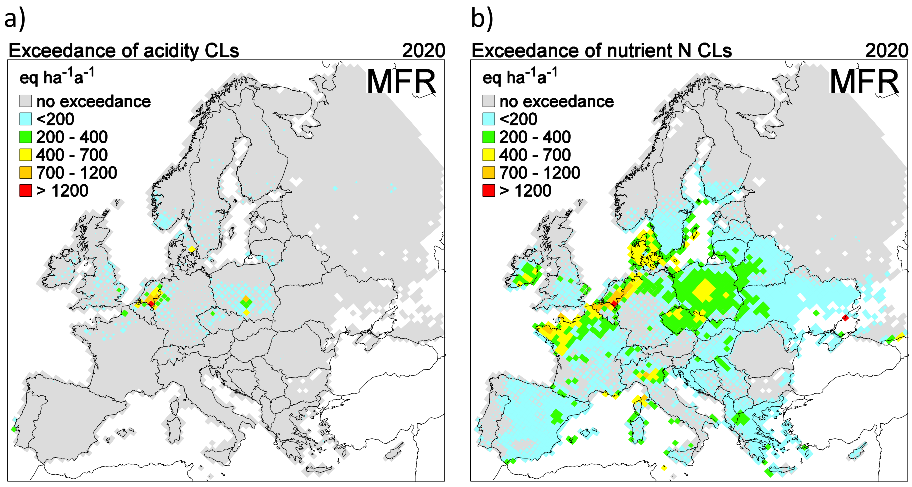

Figure 2.3 Areas where critical loads for a) acidification and b) eutrophication are exceeded by sulphur and nitrogen depositions under the maximum technically feasible reduction (MFR) emission scenario.

2.2 Low nitrogen deposition enhances plant species diversity

This section provides two examples of a tentative assessment on broad natural scales in Europe of adverse effects of nitrogen deposition on plant species diversity. The change of species richness has been assessed by applying computed European nitrogen deposition on a European scale to available dose response relationships for selected habitat classes. These relationships have been taken from experimental nitrogen-addition studies (for an overview see Bobbink and Hettelingh, 2011), as well as from a European gradient study (Stevens et al., 2010a,b)

In the first example, dose-response relationships were based on a literature survey prepared for the review and revision of empirical nitrogen critical loads (Bobbink and Hettelingh, 2011). The assessment of changes in plant diversity using these dose-response functions on a regional scale is based on the extrapolation of the functions for the EUNIS classes E, F2 and G3 (Bobbink, 2008). In 1990, the area where more than 5% of plant diversity is at risk is clearly larger than in 2020 (**Figure 2.4**). In 1990, the area covers 288,000 km^2 in the EU27 (24% of the EUNIS areas E+F2+G3). In 2020, under the Revised Gothenburg Protocol (RGP) scenario this area is reduced to 68,400 km^2 (about 6% of these EUNIS classes).

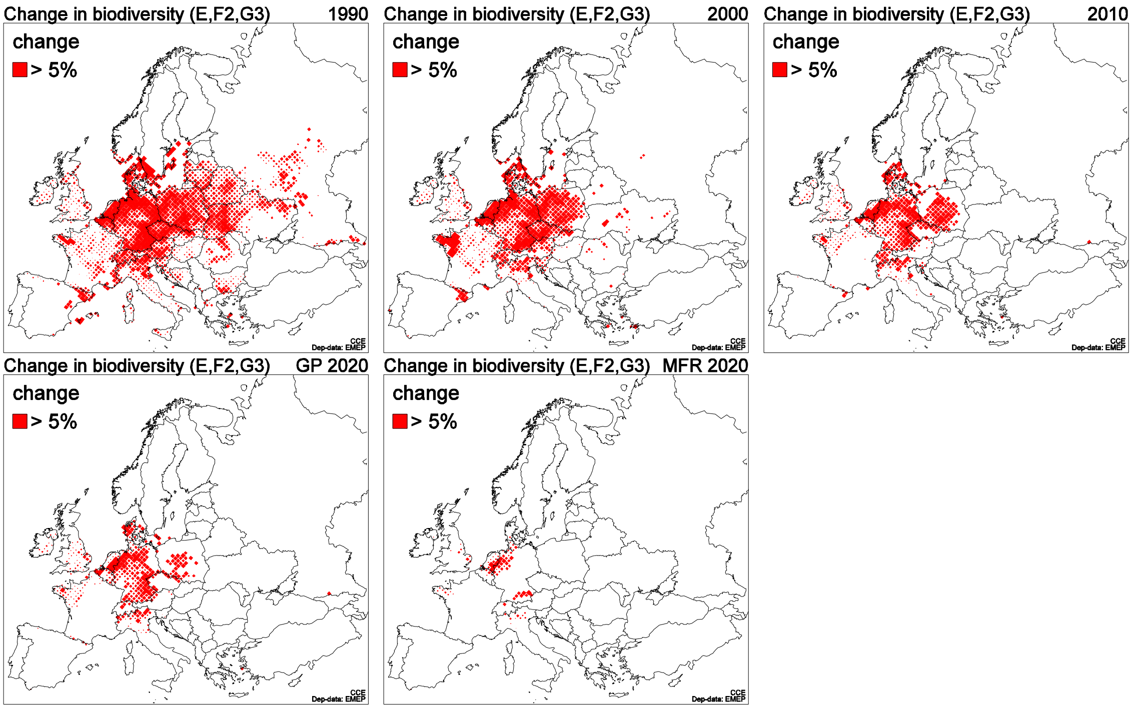

Figure 2.4 Changes (red shading) by more than 5% in plant species diversity in EUNIS classes E and F2 and in plant species similarity in EUNIS class G3 in 1990 (top left), 2000 (top middle), 2010 (top right) and in 2020 under the Revised Gothenburg Protocol (bottom left) and maximum technically feasible reduction scenario (bottom right).

In the second example, a dose-response relationship was derived from gradient studies on selected Natura2000 areas across Europe. Stevens et al. (2010a,b) surveyed 153 semi-natural acid grasslands on a transect across the Atlantic biogeographic zone of Europe with a total atmospheric nitrogen deposition ranging from 2.4 to 43.5 kg N ha^{-1} yr^{-1}, covering much of the range of deposition found in the industrialised world. The surveyed grasslands were dominated by species such as *Agrostis capillaris*, *Festuca ovina* and *F. rubra*, *Potentilla erecta* and *Galium saxatile*. The survey consisted of: nine grasslands in Belgium, three grasslands in Denmark, twenty-five grasslands in France, twelve grasslands in Germany, eleven grasslands in Ireland, Northern Ireland and the Isle of Man, seven grasslands in the Netherlands, nine grasslands in Norway, four grasslands in Sweden and seventy-seven grasslands in Great Britain. The large number of sites surveyed in Great Britain derives from the intensive national survey of the earlier work and from the fact that *Violion caninae* grasslands cover a much larger area in Great Britain than in other countries in the study (Stevens et al., 2004). For all of the sites, well documented deposition models were used for estimating the deposition of nitrogen and sulphur, resulting in some variation in the models used. National models were used for

Germany (Gauger et al., 2002), the Netherlands (van Jaarsveld, 1995, 2004; Asman and Van Jaarsveld, 2002) and Great Britain (NEGTAP, 2001). For all other countries the EMEP-based IDEM (Pieterse et al., 2007) models were used. The relationship between nitrogen deposition and species richness was fitted with a negative exponential curve. The harmonized European land-cover map was used in a similar way as with the study in example 1. To increase the applicability of the dose-response function, the analysis was restricted to locations with precipitation between 490 and 1971 mm yr^{-1}, altitude below 800 m and a soil pH < 5.5. These restrictions were applied to ensure that only grasslands with precipitation and soil pH within the range of conditions found in the original dataset were considered. The limitation of available precipitation data led to the analysis being restricted to E1 (including E1.7 and E1.9), E2 and E3 grasslands areas located west of 32 ºE. This resulted in an area covering about 446.000 km^2. Using depositions on Natura2000 areas since 1980 computed with the EMEP model on a 50 x 50 km^2 grid, the estimated relative grassland species richness is shown in **Figure 2.5**. The extent of the area with a computed species richness of less than 70% (red shading) in 1980 clearly diminishes in 2020, particularly under the maximum technically feasible reduction scenario.

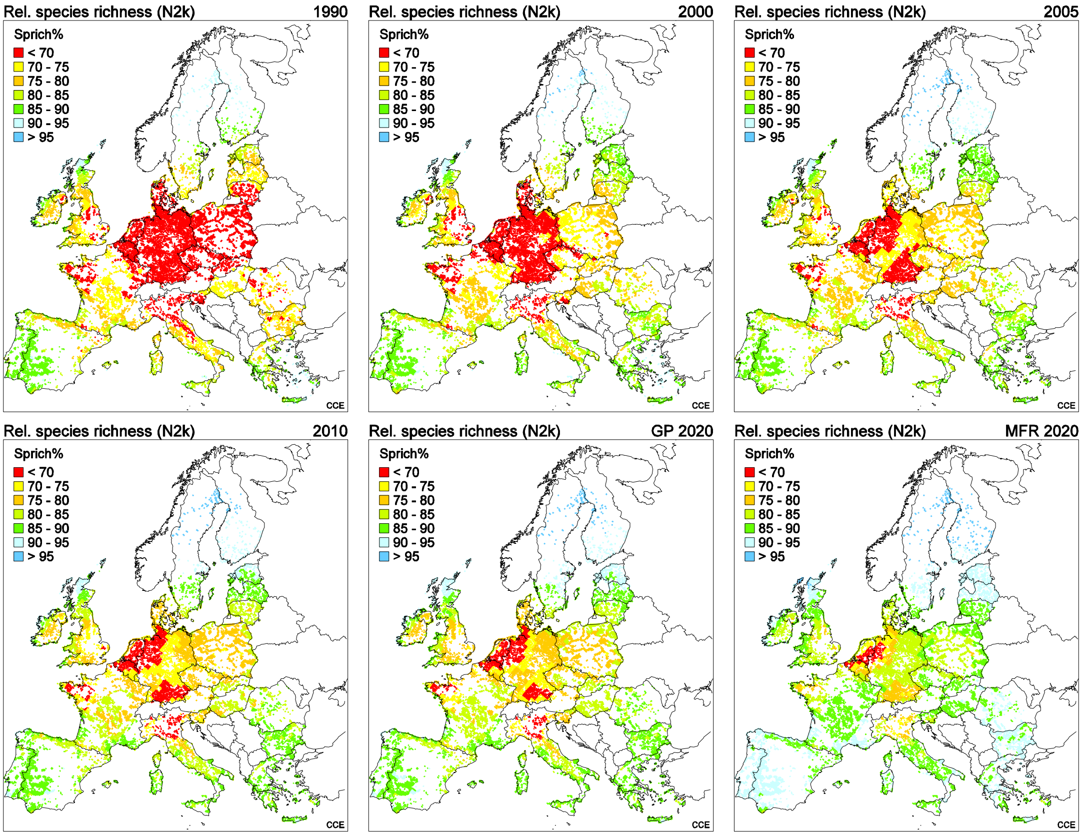

Figure 2.5 Relative plant species richness in EUNIS classes E1, E2 and E3 grasslands in 1980 (top left), 2000 (top middle), 2005 (top right), in 2010 (bottom left), 2020 under the Revised Gothenburg Protocol (bottom middle) and maximum technically feasible reductions scenario (bottom right).

Annex 2 shows the average species richness per country and in the EU27. It turns out that the average species richness in EUNIS classes E1, E2 and E3 grasslands in the EU27 in 1990 (high nitrogen deposition) is computed to be lower than in 2020 under the Revised Gothenburg Protocol, i.e. 72 % and 81 % respectively.

Although the above examples suggest that the impact of atmospheric nitrogen deposition on plant species diversity is expected to decline in the future, there still will be a net loss of plant diversity, even under the maximum feasible reduction scenario. The above assessments should be extended to other ecosystems and biodiversity indicators (e.g. presence of red list species, soil organisms) for a comprehensive analysis of impacts of excessive nitrogen deposition on biodiversity. It should also be noted that effects of excessive nitrogen deposition on the structure and functioning of ecosystems and its biodiversity may not occur instantly, it may take several decades over which the resilience of soils and vegetation is weakened and impacts become progressively apparent. In addition, little is known about the recovery from historic nitrogen pollution, which is unlikely to follow the same dose-response relationship.

2.3 Excessive nitrogen deposition reduces the occurrence of plant species adapted to low nitrogen availability

A study on 224 forest plots in the central region and the southern boreal region of Europe revealed a statistically significant relationship between deposition of nitrogen and acidity on the one hand and the composition of ground vegetation species on the other hand. **Figure 2.6** shows that plots with a large share of nitrogen-indicating species are located in regions with high nitrogen deposition, such as in The Netherlands, Flanders, Denmark, northern Germany, southern Poland, Slovakia and Hungary.

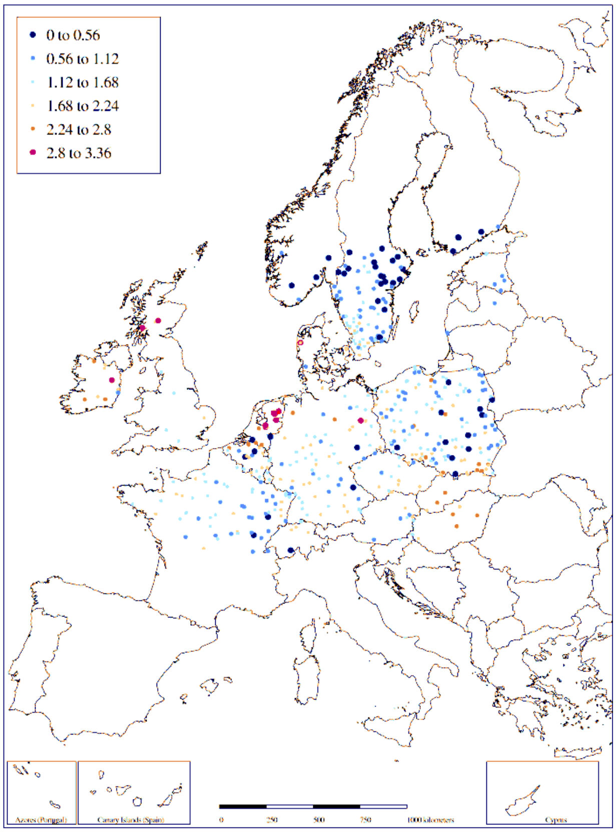

Figure 2.6 Forest ecosystem monitoring plots (n = 224) grouped according to the occurrence of nitrogen-indicating plant species. Plots with a stronger occurrence of nitrogen indicators (orange and red dots) are located in regions with high nitrogen deposition. On the plots in Scotland and Ireland, species that are typical for the Atlantic climate prevail.

There is a significant relationship between the occurrence of nitrogen-indicating species and nitrogen deposition in forests across Europe. The presence of nitrogen-loving plant species increases with increasing nitrogen deposition up to a maximum level (**Figure 2.7a**). In addition, forest plots where sulphur and nitrogen deposition were high had a relatively low number of epiphytic lichen species. The share of lichen species adapted to low nitrogen availability decreased below a 40% threshold when the nitrogen deposition measured below the forest canopy (throughfall) exceeded 3.8 kg ha^{-1} yr^{-1} (**Figure 2.7b**). This shows that even a relatively low nitrogen deposition has a clear influence on the species composition of epiphytic lichens. The critical load for nitrogen of 3.8 kg ha^{-1} yr^{-1} was exceeded on 80% of forest plots. Effects of nitrogen deposition on epiphytic lichens are much less evident in coniferous than in broadleaved forest types. Effects on other groups of vegetation only occur at higher nitrogen inputs.

a) b)

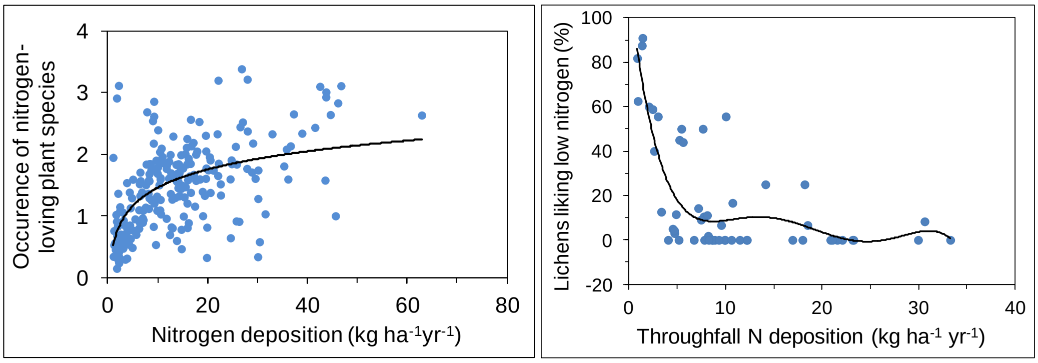

Figure 2.7 a) Relationship between the occurrence of nitrogen-loving plant species and nitrogen deposition for 224 forest plots and b) the percentage of lichens adapted to low nitrogen, as a function of the total throughfall nitrogen deposition in 87 forest plots.

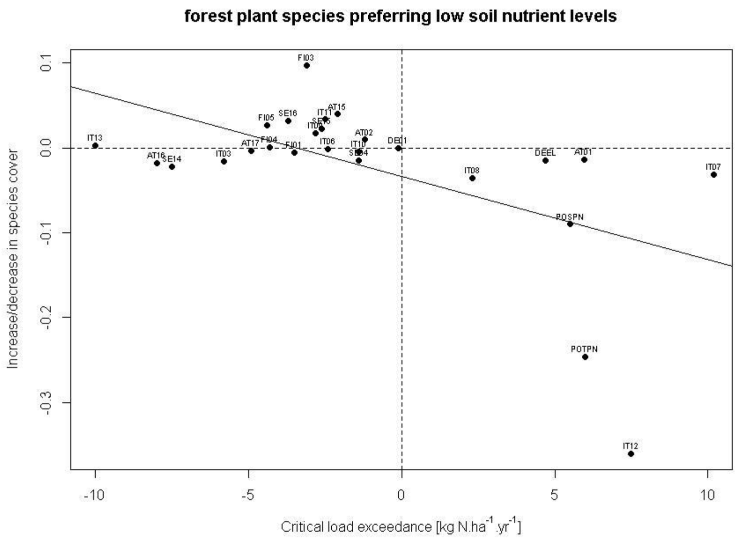

Figure 2.8 Forest plant species that prefer low soil nutrient levels have decreased during the last 10-50 years in 28 sites across Europe (from northern Finland to southern Italy) owing to the exceedance of the nitrogen critical loads. The Y-axis indicates the strength of the cover change of these oligotrophic species in the study site (negative values indicate a decrease, positive values an increase). The critical load exceedances are shown as the difference between nitrogen deposition and empirical critical load (negative values indicate no exceedance, positive values an exceedance) (modified from Dirnböck et al., submitted).

A recent study using long-term ICP Integrated Monitoring and other observation data from 28 forest sites from northern Sweden to southern Italy together with their nitrogen critical load exceedances, indicated that the higher a forest was exposed to nitrogen deposition the lower was the cover of species adapted to low nitrogen availability, and also the more sensitive a forest was with regard to nitrogen effects (**Figure 2.8**; Dirnböck et al., submitted). Nitrogen-loving species reacted the opposite way, though showing a much weaker relationship with nitrogen deposition. This might affect the benefits of for example harvesting berries and mushrooms and might therefore affect the recreational value and appreciation of the forest.

Nuisance growth of the aquatic macrophyte *Juncus bulbosus* has been observed in an increasing number of lakes and rivers in Europe. Among the consequences of such nuisance growth are reduced biodiversity, reduced suitability of the ecosystems for fish spawning, clogging of hydropower inlet screens and reduced suitability of the ecosystems for recreational use such as fishing, boating and bathing (Moe et al., 2013). For rivers an enhanced supply of nitrogen appears to be a trigger for the enhanced growth (Schneider et al., 2013).

2.4 Regeneration ability of forest species

Deposition of sulphur and nitrogen has a continuous influence on soil-chemical properties and nutrient availability of soils. This affects the vitality of single plants and whole ecosystems. Since the vitality and functionality of ecosystems are crucial for the protection of biodiversity, ICP Forests studied the biological response of plant species and plant communities by means of dynamic modelling of soil chemistry with the VSD+ model (Bonten et al., 2009) and by means of the BERN model (Bio indication of Ecosystem Regeneration potentials towards Natural conditions; Schlutow and Huebener, 2004). The BERN model allows an evaluation of the current plant composition and an outlook to the future development of regeneration abilities of plant communities.

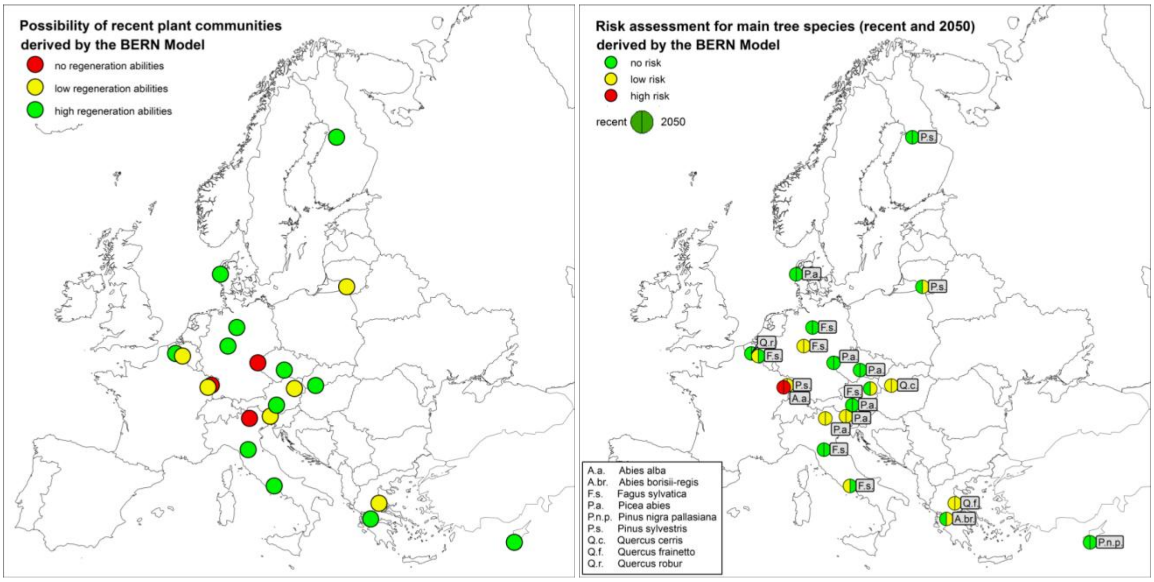

Figure 2.9 a) Regeneration abilities of current plant communities and b) the change in regeneration ability of main tree species currently and in 2050 under Cost Optimized Baseline deposition scenario.

The 21 forest ecosystem monitoring plots studied show a wide variety of plant species communities and presently occurring main tree species. Of the analysed plots, 12 were found to have "high regeneration abilities" of the presently occurring plant species composition, indicating that species composition was rated as adapted to the presently occurring geo-chemical site conditions. At six sites the regeneration ability was rated as low, and on the remaining three sites there was "no regeneration ability" in the long term, indicating that the currently occurring vegetation composition is not well adapted to present site conditions (**Figure 2.9a**).

The currently occurring main tree species were found to have full regeneration abilities and therefore no elevated risk of suffering from natural diseases on 12 of the plots. On 8 plots the current main tree species were found to have "low regeneration ability", corresponding with a low risk. For one plot the model indicates "no regeneration ability", which means an elevated risk. When relating the presently occurring main tree species to geo-chemical site conditions predicted by the VSD+ model assuming the baseline deposition scenario for the year 2050 (Cost Optimized Baseline, COB), regeneration abilities changed on several plots. On two plots the potential risk for natural diseases increased while on three other plots the risk decreased to a level of "no risk" (**Figure 2.9b**).

2.5 Ozone impacts on plant diversity

Recent attempts to predict the sensitivity of ecosystems to biodiversity loss as a result of ozone exposure have concentrated on compiling data from experiments involving exposure of plants to ozone pollution in solardomes, open top chambers or open field exposure systems. In three related studies, data were collated for 83 species from over 60 papers (Hayes et al., 2007; Jones et al., 2007; Mills et al., 2007). Hayes et al. (2007) found that species with a therophytic life form (a plant that overwinters as a seed) were quite sensitive to ozone as were those from the *Fabaceae* family (plants bearing bean pods). Comparison of relative sensitivity to ozone with Ellenberg ecological values (Ellenberg et al., 1988) showed that light-loving plants tend to be more sensitive to ozone than plants that normally occur in the shade (Jones et al., 2007), although species representing the most shade-tolerant Ellenberg values (1 - 4) were not represented in the database. Plants of Ellenberg moisture value 3 (dry site indicator) tended to be more sensitive to ozone than those found in more moist soils. Plants which can tolerate moderately saline conditions (Ellenberg salt value of 1) are more sensitive to ozone than those of non-saline habitats. There were no relationships between Ellenberg nutrient, 'reaction' (pH) or temperature value and ozone sensitivity. Jones et al. (2007) developed a method of identifying ozone sensitive species and communities from the Ellenberg Light and Salinity values and successfully applied this to predictions of ozone sensitive communities in the UK. Mills et al. (2007) used the same data to determine the habitats with the highest proportion of ozone-sensitive communities using the EUNIS (European Nature Information System) habitat classification system. These habitats were found to be: Dry grasslands (E1), Mesic grasslands (E2), Seasonally-wet and wet grasslands (E3) and Woodland fringes (E5). Alpine and subalpine grasslands (E4) and Temperate shrub heathland (F4) were also considered to be potentially ozone-sensitive. However, a long-term field exposure study in Switzerland did not confirm the sensitivity of alpine and subalpine grasslands to ozone (Bassin et al., in press). Wittig et al. (2009) conducted a meta-analysis of tree biomass and growth responses and found that angiosperms such as needle-leaf trees were more sensitive to ozone than gymnosperms such as broadleaf trees.

However, there is currently a lack of field-based evidence for the impacts of ozone on plant species diversity. Results from European grassland field exposure experiments have been rather mixed regarding the impacts of ozone on plant growth and species composition (see Mills et al., 2013). Although field exposure studies with trees generally confirm the impact of ozone on tree growth, it is not known how this affects tree species diversity (see Mills et al., 2013). There is however substantial evidence that ozone effects on trees affect associated organisms and ecological processes such as the growth rate of insects (Valkema et al., 2007), tree-fungi associations (Karnosky et al., 2002; Olbrich et al., 2010) and the soil microbial community composition (see Mills et al., 2013). So far, only a few field surveys have shown an impact of ozone on plant species composition. For example, in the highly polluted San Barnardino National Forest in California, the more ozone-sensitive tree species ponderosa pine had in part been replaced by the more ozone-tolerant species white fir in the 1970s (Miller et al., 1973). In the UK, Payne et al. (2011) identified ozone as the third strongest driver of plant community composition in calcifuges grasslands, behind inorganic nitrogen deposition and mean annual potential evapotranspiration. Very limited information is available on the ozone sensitivity of individual species and communities in the Mediterranean, considering that this area is a biodiversity hotspot in Europe (see Mills et al., 2013). More studies are needed in the Mediterranean to determine the impact of relatively high ozone concentrations on plant diversity.

3. Impacts on ecosystem services

3.1 Impacts of nitrogen deposition on ecosystem services

3.1.1 Forest productivity (timber production) and carbon sequestration

Sustainably managed forests constitute a renewable resource providing a wealth of socio-economic benefits. Among the most important of these services is timber supply as a basis for timber industries which employ 2.2 million people throughout the EU. Further important services are water supply, improving air quality, regulation of weather conditions, protection against erosion, landslides and avalanches, as well as the recreational and cultural value of forests for the society. These socio-economic benefits are jeopardised by threats to forest ecosystem functioning posed by e.g. air pollution and climate change. Timber products are an important part of several countries' economy but the harvesting exert detrimental influences on forest ecosystems. While in production phase, the trees takes up soil nutrients and transfers protons to the soils and waters, causing potential acidification of the systems. Enhanced production for energy purposes accelerates the extraction of base cations and soil acidification (e.g. Aherne et al., 2012). In the natural habitats with return of biomass to the soil in a decomposition phase, base elements are preserved in the system at the same time as carbon is furnished to the microbial society and other decomposers, keeping up a rich biodiversity for sustainable life in the biological system.

Natural habitats provide complementary values and possibilities to harvest natural products without comprehensive negative impacts. Included products are berries, mushrooms, medical plants, herbs and lichens (e.g. Vihervaara et al., 2010). Such products also provide food for higher animals such as reindeer, sheep and other herbivores. Other recreational benefits such as bird watching, safaris, hunting and single people's visits to nature reserves gain from a natural habitat. In addition, there seem to be a range of existing and novel possibilities related to different bioinnovations (so called "bioeconomy" such as identification and production of pharmaceuticals or development of water pollution remediation techniques; Kettunen et al., 2012).

Natural habitats such as forests, mires, grasslands, wetlands, waterways are most important reference areas for managed land and are for science purposes invaluable. They provide the background situation and allow the assessment of natural trends, when evaluation of man-made impacts have to be separated from natural fluctuations and long-term changes. The natural capture of carbon as CO_2 in plant biomass stores carbon over long periods and add to soil carbon storage, a highly desirable sink for CO_2 from a climate change perspective. For example, Matero et al. (2007) estimated the value of carbon sequestration of Finnish forest trees to be 1876 million EUR, and the value of change in mineral soil carbon stock to be 136 million EUR. In Sweden, Gren and Svensson (2004) calculated the annual carbon sequestering value of Swedish forest to be between about 3300 - 5200 million EUR. A special large storage is in the organic soils of peatlands and mires with ongoing carbon accumulation in a natural peat development, and as such belonging to the preserved environments of the forest landscape. Mires and peatlands store the same amount of carbon as stored in all mineral soils together or as the content in the atmosphere.

Monitoring of tree growth across Europe has shown that enhanced nitrogen deposition increases tree growth. An annual nitrogen deposition of 1 kg ha^{-1} yr^{-1} resulted in an average increase in basal area increment (BAI) of about 1% (**Figure 3.1**). This corresponds to an average carbon fixation in tree stems of about 20 kg ha^{-1} yr^{-1}. Hence, enhanced nitrogen deposition is likely to enhance carbon sequestration in the living biomass of trees, at least temporarily, but only if nitrogen is the limiting factor for tree growth. On soils that were already well supplied with nitrogen the effect of enhanced nitrogen deposition on tree growth was smaller. Decline of nitrogen deposition in recent years and in the future as a result of the successful implementation of air pollution abatement policies in Europe is likely to result in a reduction in tree growth in years to come (De Vries and Posch, 2011).

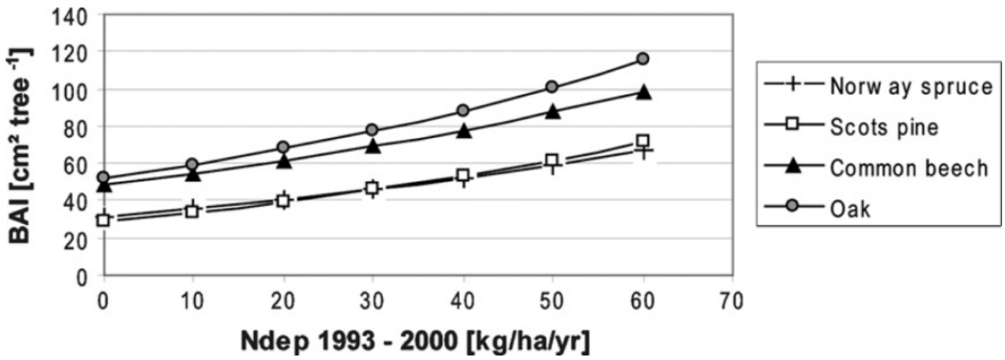

Figure 3.1 Basal area increment (BAI) of trees as a function of nitrogen deposition (Ndep).

3.1.2 Soil characteristics and nitrogen retention

Soils store air pollutants temporarily and therefore play an important role in water purification as water to inland watercourses and coastal marine habitats originate to a large extent from the catchment terrestrial landscape. However, excessive storage of sulphur, nitrogen and metal compounds will adversely affect soil functioning (e.g. microbes and invertebrates). When the retention capacity of soils is reached or disturbed, pollutants may start leaching to surface water and coastal zones, threatening the availability of clean water for multiple purposes (e.g. drinking, bathing, fishing).

Nitrogen retention and exceedance of critical loads for nitrogen

Currently and for a long-time period nitrogen deposition has been high and greatly exceeding the critical loads on many places in large parts of Europe (Posch et al., 2012). Input by deposition also highly exceeds the catchment outflows to surface waters, resulting in accumulation in forest ecosystems (**Figure 3.2a**). A critical deposition threshold of about 8-10 kg N ha^{-1} yr^{-1} was confirmed by the input-output calculations with integrated monitoring data (Forsius et al., 2001). The output flux of nitrogen was strongly correlated with key ecosystem variables like nitrogen deposition, nitrogen concentration in soil organic matter and current year needles, and nitrogen flux in litterfall. The carbon to nitrogen (C:N) ratio in soil organic matter has been identified as a key indicator for estimating nitrogen retention and the risk for nitrogen leaching in forested ecosystems (**Figure 3.2b**; Gundersen et al. 2006). Nitrogen leaches from the forest soil at a C:N below 23 in the organic layer.

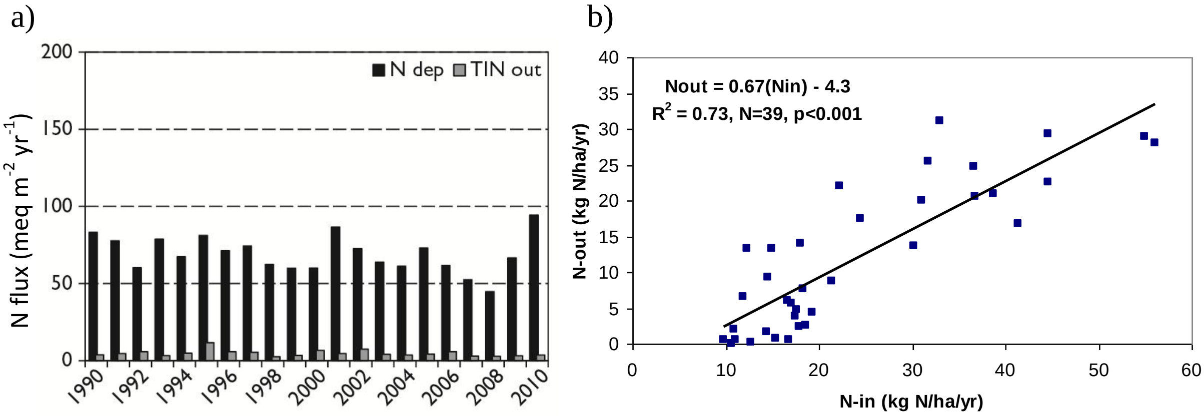

Figure 3.2 a) Nitrogen flux at the ICP Integrated monitoring site CZ01 the Czech Republic between 1990 and 2010 (N dep = nitrogen deposition as input, TIN out = total inorganic nitrogen leaching as output; Vuorenmaa et al., 2012) and b) nitrogen leaching flux (N-out) in water (kg N ha^{-1} yr^{-1}) against nitrogen input in throughfall deposition (N-in kg N ha^{-1} yr^{-1}) for forest sites with a C:N ratio < 23 in the organic soil layer. The data indicates that nitrogen deposition clearly increases nitrogen leaching below the C:N threshold value of 23 in soil (from Gundersen et al., 2006).

The storage in the terrestrial system mitigates leaching and enhances possibilities for healthy waters (Forsius et al., 2001; Gundersen et al., 2006; Holmberg et al., 2013). When critical thresholds and critical loads are exceeded, elevated outflow of nitrogen to surface water may occur (**Figure 3.3**; Holmberg et al., 2013) with eutrophication effects on inland and coastal waters. The ecosystem functions in natural systems are better preserved as compared to managed land with fertilization and harvesting. Such land exerts higher loads on the water courses which results in eutrophication. Costs will arise to mitigate such influences and currently the EU is applying financial penalties to some member states because of exceeded nitrogen loads to surface waters. The critical load and level calculations form the basis for the European policy work to reduce the emissions of harmful pollutants (Amann et al., 2011; Posch et al., 2012). It is essential that these concepts are developed and validated using empirical observations from ICP monitoring sites.

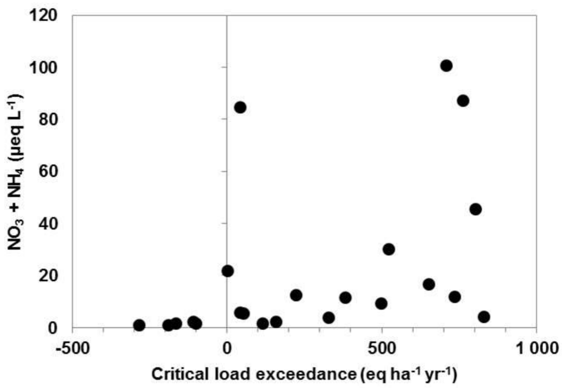

Figure 3.3 NO_3+NH_4 concentration in water (µeq l^{-1}) plotted against the exceedance of empirical critical loads for nitrogen (eq ha^{-1} yr^{-1}) at ICP Integrated Monitoring sites in Europe (negative values indicate no exceedance, positive values an exceedance). Exceedance of the critical loads increases nitrogen leaching and thus detrimental effects on groundwater and surface water quality (from Holmberg et al., 2013).

In healthy forests the carbon-to-nitrogen concentration (C:N) of the forest floor is distinctly higher than in the mineral soil. However, in areas with a high nitrogen deposition load, the C/N ratio of the forest floor (C/N_{FF}) may become smaller than the C/N ratio of the mineral topsoil (C/N_{MIN}). Hence, the proportion of the C/N_{FF} over C/N_{MIN}, referred to as the C/N-index, is a useful indicator for the imbalance induced by excess nitrogen input. If this index is less than 1, the organic matter and nutrient cycling is most likely disturbed and forest health and vitality may be at risk. Areas with C/N indices between 0 and 1 are mainly situated in central-western Europe, in parts of central-eastern Europe, and in the Baltic States (**Figure 3.4**). Forest growth is strongly stimulated by nitrogen deposition and by smaller C/N ratios in the forest floor (see above). However, if the forest soil cannot supply other nutrients (especially base cations like calcium and magnesium) in a balanced and sustainable way, impaired tree health is likely to occur. Furthermore, when the C/N ratio in the forest floor is small and nitrogen deposition is high (> 20 kg N ha^{-1} yr^{-1}), nitrates leach from the soil into ground and surface waters, leading to eutrophication and ground water contamination. Nitrogen concentrations in the soil solution exceeding the critical limit for elevated nitrogen leaching occurs on two thirds out of 171 investigated forest plots.

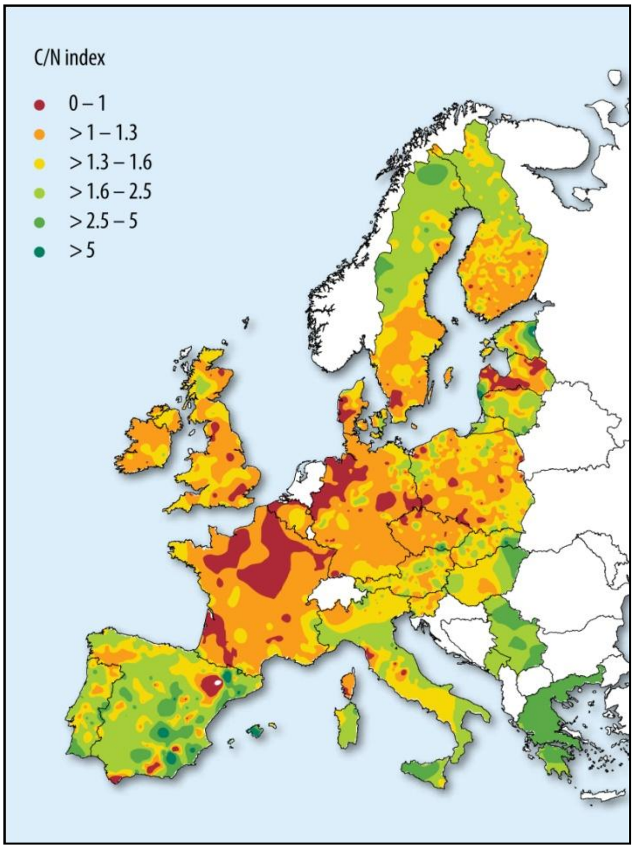

Figure 3.4 Soil C/N-indices in forests across Europe.

By applying dynamic models to selected nitrogen deposition scenarios, quantitative links between nitrogen deposition and ecosystem services can be established with currently available methodologies (see De Vries et al., 2009). In a case study, the Very Simple Dynamic (VSD) model (Posch et al., 2003) was run over 2000-2050 using EMEP deposition data for reduced and oxidized nitrogen and sulphur dioxide. The European background database of critical loads at the Coordination Centre for Effects was used for the assessment of the comparative static assessment of exceedances and the dynamic analysis of relevant soil chemical indicators. Three scenarios were analyzed, i.e. (A) depositions in 2000-2050 are equal to the deposition in 1980 (Ndep1980), (B) Depositions of 2010 as agreed under the 1999 Gothenburg Protocol are kept constant until 2050

(Current Legislation; CLE2020), and (C) Nitrogen depositions in 2000 were linearly reduced to reach critical load values for nutrient nitrogen (and related maximum critical load of sulphur) in 2010 and kept these values constant thereafter (CLnut2020). The results show that in the present situation (CLE2020), aluminium depletion from soils is particularly an issue in central Europe. The depletion of base cations is more widespread over Europe, although the rate of depletion has strongly diminished due to emission reductions since 1980. Areas with excessive concentrations of nitrate in 2050 diminish under CLE2020 compared to Ndep1980, and recover when CLnut2020 is applied. However, the concentration of aluminium and cadmium cannot sufficiently be diminished everywhere in Europe under scenarios that focus on the reduction of nitrogen deposition.

3.2 Impacts of ozone on ecosystem services

3.2.1 Impacts of ozone on food security

In 2012, the ICP Vegetation reviewed the hidden threat of ozone pollution to food security (Mills and Harmens, 2011). The key components of the food system that ozone pollution interferes with are the productivity of crops, the nutritional value and the stability of food supplies. Ozone is absorbed into plants via the thousands of microscopic pores (stomata) on the leaf which normally open during the day to allow CO_2 absorption for photosynthesis and evaporation of water. The more open the pores are, the more ozone will enter the plant. Once inside the plant, the reactive oxygen species that are formed damage cell walls and membranes, leading to cell death and/or reductions in key processes such as photosynthesis. The results of these damaging effects depend on both the concentration and duration of ozone exposure. Plants are able to detoxify low concentrations of ozone but only to a certain threshold level. Above the detoxification level, ozone pollution damages crop plants by, for example, causing a yellowing of leaves and premature leaf loss, decreased seed production and reduced root growth, resulting in reduced yield quantity and/or quality and reduced resilience to other stress such as drought. Ozone-induced damage on the leaves of salad crops (**Figure 3.5**) will reduce the market value of these crops.

Figure 3.5 Ozone-induced leaf damage on salad onion (left; source: J. Bender), spinach (top right; source J. Bender) and lettuce (bottom right; source D. Velissariou).

Two of the world's most important staple food crops, wheat and soybean are sensitive to ozone with yield being reduced by 18% at a 7h mean ozone concentration of 60 ppb compared to 30 ppb (**Table 3.1**). Rice, maize and potato are moderately sensitive, having a ca. 10% yield reduction at 60 ppb ozone. In terms of economic value, eight of the nine crops with the highest production in Europe are sensitive or moderately sensitive to ozone. Sensitivity to ozone varies between cultivars, which mean that there is scope for exploiting ozone resistance within breeding programmes. In general, modern cultivars of crops such as wheat seem to be more ozone sensitive than older, traditional cultivars, suggesting that breeding for high crop productivity might have resulted unintentionally in breeding more ozone-sensitive cultivars. Compared to the impact on yield quantity, considerably less information exists on the impacts of ozone on food and feed quality and few dose-response relationships have been derived. So far, impacts have been found on important parameters for food security such as the protein yield of wheat, sugar content of potato, and oil quality in oilseed rape (Mills and Harmens, 2013).

Table 3.1 Grouping of crops by sensitivity of yield to ozone. Values in brackets represent the percentage decrease in yield at a 7h mean ozone concentration of 60 ppb compared to that at 30 ppb (from Mills and Harmens, 2011).

Sensitive	**Moderately sensitive**	**Tolerant**
Peas and beans (including peanut) (30) Sweet potato (28) Orange (27) Onion (23) Turnip (22) Plum (22) Lettuce (19) Wheat (18) Soybean (18)	Alfalfa (14) Water melon (14) Tomato (13) Olive (13) Field mustard (12) Sugar beet (11) Oilseed rape (11) Maize (10) Rice (9) Potato (9) Barley (6) Grape (5)	Strawberry (1) Oat (-3) Broccoli (-5)

The quantification of global impacts of ozone pollution on food security currently relies on the use of concentration-based ozone metrics such as AOT40[2] and 7h mean ozone concentration. All such studies have highlighted the potential for ozone to impact on yield by between ca. 3 and 20% depending on crop (see Mills and Harmens, 2011). Current global yield losses are estimated to be between 4 - 15% for wheat, 6 - 16% for soybean, 3 - 4% for rice and 2.2 - 5.5% for maize, with global economic losses estimated to be in the range $11 - $26 billion (Van Dingenen et al., 2009). Under the IPCC SRES[3] A2 Scenario, global yield losses for the year 2030 due to ozone are predicted to range from 5.4 - 26% for wheat, 15 - 19% of soybean, and 4.4 - 8.7% for maize, with total global agricultural losses in the range $17 - $35 billion annually (Avnery et al., 2011a,b). Even under the lower emission scenario B1, less severe impacts will nevertheless be in the range $12 - $21 billion annually. In areas of the world where demand already outweighs supply, the "hidden" threat from ozone impacts on crop production will add to the many threats to food security. So far no global evaluation is available

[2] The accumulated hourly mean ozone concentration above 40 ppb, during daylight hours
[3] The Intergovernmental Panel on Climate Change Special Report on Emissions Scenarios

on the impacts of ozone on food and feed quality, thus the total impacts of ozone on food security might be even higher than those described here.

Mills and Harmens (2011) quantified for the first time ozone impacts on wheat and tomato yield in Europe using the flux-based methodology which incorporates the effects of climate, soil moisture, ozone concentration and plant growth stage on the hourly uptake of ozone through the stomatal pores in the leaf surface (ozone flux or stomatal flux). This method is biologically more relevant than the AOT40-based method which only takes into the account the amount of ozone in the air above the plant. They have shown that using the national emissions projections scenario for 2000, ozone pollution in EU27 (+ Norway and Switzerland) was predicted to be causing an average of 13.7 % yield loss for wheat, with an economic loss of €3.2 billion predicted if soil moisture is not limiting (**Table 3.2**). Economic losses per grid square in 2000 were greatest for wheat in the highest producing areas in France, Germany, Belgium, Denmark and the UK, indicating that ozone flux was high enough in these central and northern areas to have an impact on wheat production (**Figure 3.6a**). Considerable effects were also predicted for more southern countries such as Italy and Bulgaria. Impacts on tomato, a moderately ozone sensitive crop, were investigated as a representative horticultural crop for southern Europe. Using the flux-based method, economic losses of €1.02 billion representing 9.4% of production value were estimated for 2000, with the highest total losses predicted for Italy, Spain and Greece, but also for the Netherlands due to its high tomato production (**Figure 3.6b**). Implementation of current legislation (based on scenarios available before the revision of the Gothenburg Protocol) will be of benefit for food security in the future as predicted ozone effects on crops for 2020 were generally lower than those in 2000. For both wheat and tomato, economic impacts were predicted to decrease by 38% to €1.96 billion and €0.63 billion respectively. However, for wheat, critical level exceedance remained high at 82.2% for the wheat growing areas. Critical level exceedance reduced from 77.8% of tomato growing areas in 2000 to 51.3% in 2020 (Mills and Harmens, 2011).

a) b)

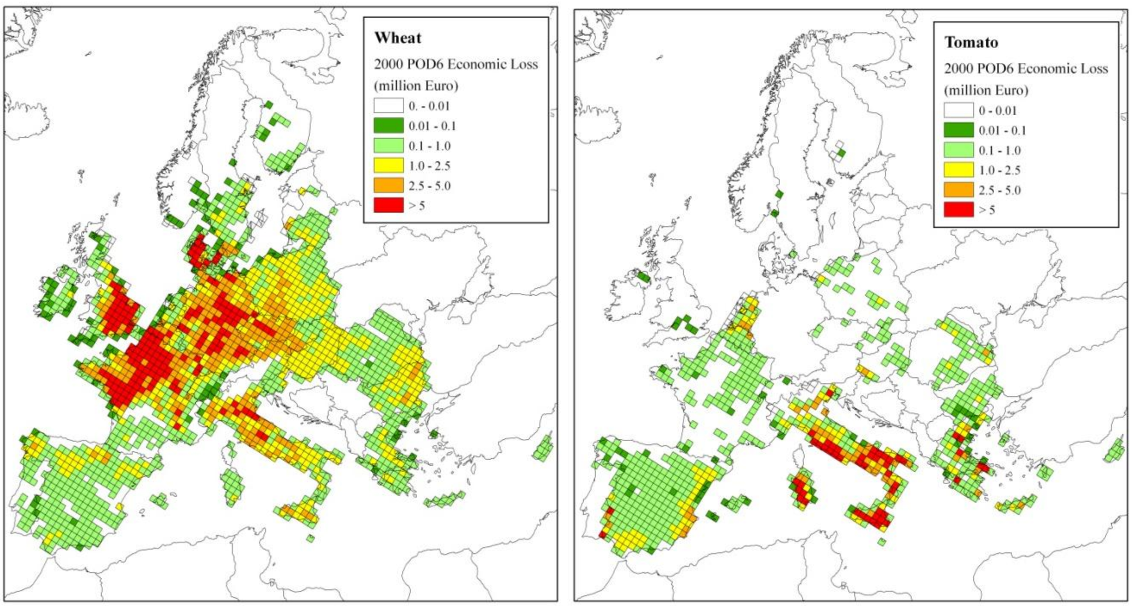

Figure 3.6 Predicted economic losses for ozone effects on a) wheat and b) tomato in million Euro per 50 x 50 km grid square in 2000 for these crop growing areas in EU27+Switzerland+Norway, based on the flux-based methodology (from Mills and Harmens, 2011).

Table 3.2 Predicted impacts of ozone pollution on wheat and tomato yield and economic value, together with critical level exceedance in EU27+Switzerland+Norway in 2000 and 2020 under the current legislation scenario (NAT scenario). Analysis was conducted on a 50 x 50 km EMEP grid square using crop values in 2000 and a flux-based risk assessment (modified from Mills and Harmens, 2011).

	Wheat		Tomato	
	2000	**2020**	**2000**	**2020**
Total economic value of wheat in 2000, billion Euro	15.87		6.85	
Mean % yield loss per grid square	13.7[1]	9.07[1]	9.4[2]	5.7[2]
Total production loss, million t	26.89	16.45	2.64	1.62
Total economic value loss, billion Euro	3.20	1.96	1.02	0.63
Percentage of EMEP grid squares exceeding critical level	84.8[1]	82.2[1]	77.8[2]	51.3[2]

[1]based on all grid squares with wheat production, [2] based on grid squares with > 1 tonne of production.

This study has highlighted the contrasting concerns in northern and southern Europe. Despite experiencing lower atmospheric ozone concentrations, yield losses for crops such as wheat are predicted to be as high in northern Europe as in central areas due to favourable climatic conditions for ozone uptake. There is concern that the risk of crop losses might increase for northern Europe in a future, warmer climate when spring peak ozone concentrations might start to overlap with earlier growing seasons. In contrast, in Mediterranean areas climatic conditions (such as drought, low air humidity) do not necessarily result in high ozone uptake in rain-fed crops despite generally high atmospheric ozone concentrations. However, significant effects of ozone are likely in Mediterranean areas where crops are irrigated, inducing stomatal opening, increasing ozone uptake and increasing impact. Prediction of ozone effects on crops in the Mediterranean part of Europe are more uncertain than those for central and northern Europe as flux models and dose-response functions are still being developed in Mediterranean countries.

3.2.2 Impacts of ozone on timber production and carbon sequestration

Terrestrial vegetation is an important sink for the greenhouse gases CO_2 (fixed by vegetation in the photosynthesis process) and ozone. Globally, it has been estimated that ozone deposition to vegetation reduces trophospheric ozone concentrations by as much as 20% (Royal Society, 2008). However, if ozone concentrations are high enough to reduce photosynthesis (i.e. CO_2 fixation) and/or above-ground plant growth, then less CO_2 and ozone will be taken up by the vegetation, leading to a positive feedback to atmospheric CO_2 and ozone concentrations and therefore global warming (Sitch et al., 2007), in addition to poorer air quality (e.g. higher ozone concentrations). Within the terrestrial biosphere, forest ecosystems have the greatest carbon sink capacity of any vegetation type (Janssens et al., 2003; Luyssaert et al., 2010) and indeed hold the largest amount of biomass carbon, totalling 50% of all terrestrial carbon (Körner et al., 2005). Many experimental studies have shown that current baseline levels of tropospheric ozone induce biomass reductions in trees, with deciduous trees generally being more sensitive to ozone then coniferous trees (Wittig et al., 2009).This has major implications for timber production and also major repercussions for the global carbon cycle and climate change policy as the terrestrial biosphere removes approximately a third of all present day anthropogenic CO_2 emissions (Felzer et al., 2004; Canadell et al., 2007; Royal

Society, 2008). European forests are predicted to currently sequester 0.11 Gt C yr^{-1}, which is approximately 10% of the European emissions (De Vries and Posch, 2011).

The ICP Vegetation recently conducted the first flux-based assessment of ozone effects on carbon sequestration in the living biomass of trees (Harmens and Mills, 2012). These analyses showed that the spatial pattern of impacts of ozone on carbon sequestration is quite different for the concentration-based (AOT40) approach compared with the flux-based (POD = phytotoxic ozone dose) approach. The concentration-based approach identified parts of southern Europe to be at highest risk of ozone impacts, while the flux-based approach identified large parts of central Europe to be at highest risk, with also parts of northern Europe being at risk of a reduction in carbon sequestration in the living biomass of trees, especially when applying regional parameterisations for the flux-model for tree species present in northern Europe (Harmens and Mills, 2012). Compared to pre-industrial ozone levels, it was estimated that current ambient ozone stomatal fluxes reduce the potential carbon sequestration in trees by ca. 14%. This is slightly higher than the estimated ozone impact based on the concentration-based approach (ca. 8 – 10%). By 2040, the negative impact of ozone on carbon sequestration in the living biomass of trees is estimated to be ca. 22% less than in 2000, primarily due to predicted reductions on atmospheric ozone concentrations (rather than due to changes in climate).

3.2.3 Examples of impacts of ozone on other ecosystem services

For a detailed review of ozone impacts on ecosystem services we refer to Mills et al. (2013). Below we provide some examples of impacts that ozone has on other ecosystem services than food and timber production and carbon sequestration.

Water cycling

Tropospheric ozone is known to alter stomatal responses to environmental stimuli and in the short term (at higher concentrations) can cause stomata (leaf pores) to close, however, under prolonged chronic exposure (at lower concentrations) many reports document ozone-induced stomatal opening or loss of stomatal sensitivity to closing stimuli, such as drought, light and humidity. Although a review of 49 papers covering 68 species showed no clear patterns in the stomatal response of plants to ozone, there was a tendency for stomatal opening to occur at lower concentrations than stomatal closure (Mills et al., 2013). Hence, the general assumed response that stomata will close due to ozone exposure is a simplification of the variation of responses observed. Ozone-induced stomatal closure will preserve water within soils whilst ozone-induced stomatal opening will increase water loss from vegetation and soils. Extensive measurements of a Southern Appalachian forest in the USA have indicated an almost linear increase in average daily sap flows and enhancement of the amplitude of daily water–loss from native trees with increasing ambient ozone exposure (Sun et al., 2012). These results support the concept of ozone-induced increases rather than decreases in transpiration, resulting in a reduction in stream flow. Sun et al. (2012) suggested that loss of stomatal sensitivity will not only increase drought frequency and severity in the region, thus affecting ecosystem hydrology and productivity, but it will also have negative implications for flow-dependent aquatic biota.

Flowering, pollination, reproduction and insect signalling

Studies conducted during recent decades have demonstrated that various stages of the reproductive development plants are clearly sensitive to ozone. A recent meta-analysis of ozone effects on plant reproductive growth and development indicated that current ambient ozone concentrations significantly reduced seed number, fruit number and fruit weight, while there was a trend towards increasing flower number and flower weight at elevated ozone (Leisner and Ainsworth, 2012). Negative effects on the reproductive performance in response to ozone may result from a reduction in plant growth, a decreased reproductive allocation, or from direct effects on reproductive structures (Black et al., 2000). Bergmann et al. (1996) observed contrasting effects on resource allocation to the vegetative and reproductive organs of 17 herbaceous species that were exposed to different ozone regimes from the seedling stage to the flowering stage. Although ozone caused comparable reductions in both vegetative and reproductive growth in the majority of the investigated species, three species (*Chenopodium album, Matricaria discoidea, Stellaria media*) showed a greater vegetative growth and reduced reproductive allocation. The germination ability of the seeds was affected by ozone such that germination rate was up to 30% lower in ozone-treated plants compared to control plants (Bergmann et al., 1996).

Any impact of ozone exposure on the timing of flowering may also play an important role in reproductive success, particularly for species in which flowering is closely synchronized with pollinating species (Black et al., 2000; Hayes et al., 2012). However, the impact of ozone on the timing of flowering varies markedly between species. For example, ozone exposure has been reported to delay flowering in two species (*Campanula rotundifolia* and *Vicia cracca*) of simulated meadow community mesocosms (Rämö et al., 2007). In mesocosms representing 'calcareous grassland', ozone has been found to accelerate the timing of the maximum number of flowers in *Lotus corniculatus* (**Figure 3.7**; Hayes et al., 2012). By contrast, Bergmann et al. (1996) showed that the timing of flowering and seed set in 17 wild plant species were not significantly influenced by season-long exposure to 1.5 x ambient ozone concentration in open top chambers.

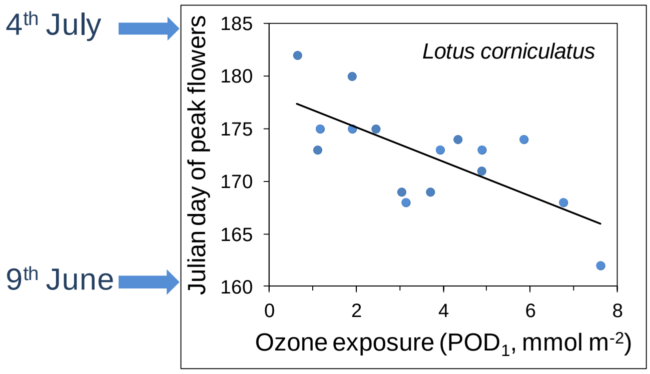

Figure 3.7 Julian day of maximum flower number for *Lotus corniculatus* in reponse to stomatal ozone flux (POD_1) (modified after Hayes et al., 2012).

Floral volatile hydrocarbons play an important role in pollinator attraction and, additionally, serve as indirect plant defenses against herbivorous insects. These floral scent trails in plant-insect interactions can be destroyed or transformed by ozone (McFrederick et al., 2008). Signals may travel shorter distances before being destroyed by chemical reactions with ozone, thus losing their specificity. Pollinators that rely on scents to orient to flowers are likely to spend more time searching for forage, which could affect their reproductive fitness, but could also affect plant reproduction as reduced pollinator efficiency could result in greater pollinator limitation (McFrederic et al., 2008). The implications of a loss or modification of scent signals by ozone pollution for both pollinators and signaling plants may be even greater in patchy or fragmented habitats because pollinators may be spending more time searching for flowers.

3.3 Impacts of heavy metals on ecosystem services

3.3.1 Heavy metal accumulation in soils

The deposition of heavy metals to ecosystems is also a concern for ecosystem services, with lead (Pb), cadmium (Cd) and mercury (Hg) being identified as priority metals under the LRTAP Convention. High loads existed in the past and have resulted in accumulation in the soil. Although heavy metal deposition in Europe has declined significantly in recent decades (Travnikov et al., 2012), accumulation in soils is ongoing with very high observed input:output ratios (**Figure 3.8**; Bringmark et al., 2011, 2013). The terrestrial catchment ecosystem stores metals in the soil, mitigating outflow to surface waters.

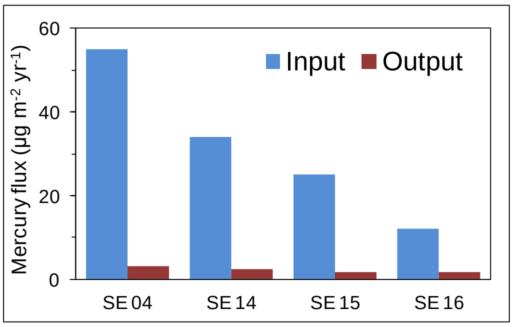

Figure 3.8 Input and output fluxes of mercury at four Swedish ICP Integrated Monitoring sites.

The input/output budgets and catchment retention for cadmium, lead and mercury in the years 1997–2011 were recently determined for 14 ICP Integrated Monitoring catchments across Europe (Bringmark et al., 2013). Metal inputs were considered to derive from bulk deposition, throughfall and litterfall, outputs were estimated from run-off values. Litterfall plus throughfall was taken as a measure of the total deposition of lead and mercury (wet + dry) on the basis of evidence suggesting that, for these metals, internal circulation is negligible. The same is not true for cadmium. Excluding a few sites with high discharge, between 74 and 94 % of the input lead was retained within the catchments; significant cadmium retention was also observed. High losses of lead (>1.4 mg m^{-2} yr^{-1}) and cadmium (>0.15 mg m^{-2} yr^{-1}) were observed in two mountainous Central European sites with high water discharge. All other sites had outputs below or equal to 0.36 and 0.06 mg m^{-2} yr^{-1}, respectively, for the two metals. Almost complete retention of mercury, 86–99% of input, was reported for the Swedish sites. These high levels of metal retention were maintained despite the recent reductions in pollutant loads.

3.3.2 Mercury accumulation in fresh water fish

The soils cope better with elevated metal concentrations compared with surface waters, but the microbial activity of the soils may still be affected, in turn having effects on biodiversity (Åkerblom et al., 2010, Tipping et al., 2010). However, accumulation of heavy metals in the soil could create a chemical time bomb for the future. Occasionally, elevated unnatural outflow of metals to surface waters may occur. This is especially evident for disturbed sites (e.g. with intensive forestry practices) with mercury methylation, releasing one of the most hazardous elements for freshwater biota - methylmercury - that is taken up by organisms in the water, among them fish, and accumulates in the food chain. In over half of the lakes in Sweden, the content of mercury in fish is higher than the recommended values for human consumption (**Figure 3.9;** Munthe et al., 2007).

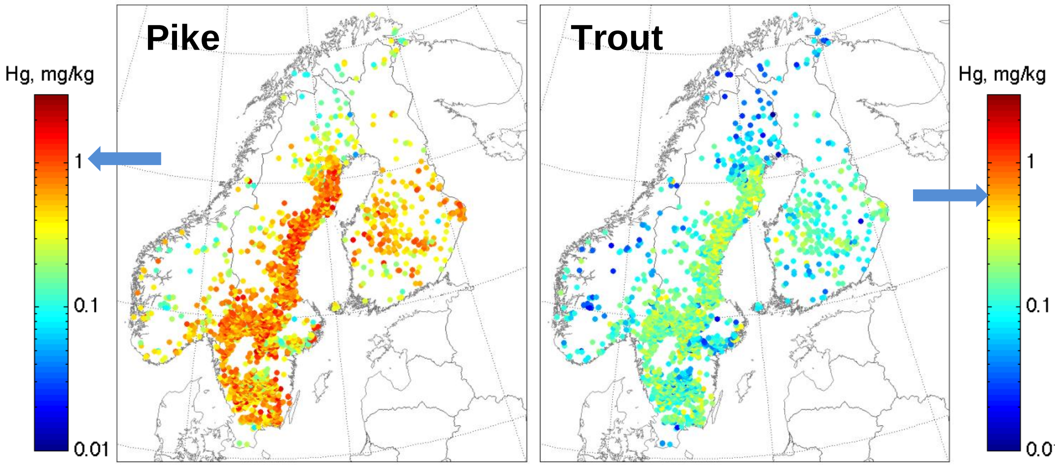

Figure 3.9 Mercury concentrations in pike and trout in lakes in Fennoscandia. The recommended mercury concentration in pike and trout for human consumption is 0.5 and 1 mg kg^{-1} respectively (from Munthe et al., 2007).

3.4 Recovery from acidification in freshwaters

In the 1970s, the suspected link between widespread acidification of ecosystems (e.g. freshwaters and forests) and the damage to for example fish populations and tree health ('forest dieback') in Europe prompted calls for reductions in emissions of air pollutants. More recently acid deposition has also become a problem in other parts of the world, such as eastern China, Japan and other highly-industrialised regions. Acidification of freshwaters requires two factors: acid deposition and acid-sensitive catchment ecosystems. Because most precipitation falls on the terrestrial parts of the catchment, lake and streamwater chemistry is strongly affected by soil properties. Acid sensitive waters are typically located in catchments with highly siliceous soils with low acid neutralising capacity. In Europe, water acidification has been most widespread in Fennoscandia, where thousands of freshwaters have been affected (Skjelkvåle et al., 2006). Upland areas elsewhere in Europe such as the UK, central and eastern Europe and the Alps have also been affected. In North America, widespread acidification occurs in southeastern Canada (Ontario, Quebec, and the Atlantic provinces) (Jeffries, 1997), and eastern United States (upland areas of New York, the New England states and the southern Appalachian Mountains) (Driscoll et al., 2001).

Acidification causes major changes in aquatic ecosystems. Most prominent is the damage and loss of fish populations. In Norway, acidic deposition caused the loss of brown trout populations in thousands of lakes and the extinction of native salmon populations from seven major rivers

(Hesthagen et al., 1999; Jensen and Snekvik, 1972). A survey conducted in the 1990s in Fenno-Scandia showed that acidification has affected fish populations in more than 10,000 lakes (Tammi et al., 2003). Acidification affects all trophic levels in aquatic ecosystems, including benthic invertebrates, planktonic fauna, planktonic and attached algae. Acid sensitive species decline in abundance and disappear, while acid tolerant species increase in abundance and invade. Adverse effects on aquatic ecosystems have been reported across Europe and in parts of Canada and the United States of America.

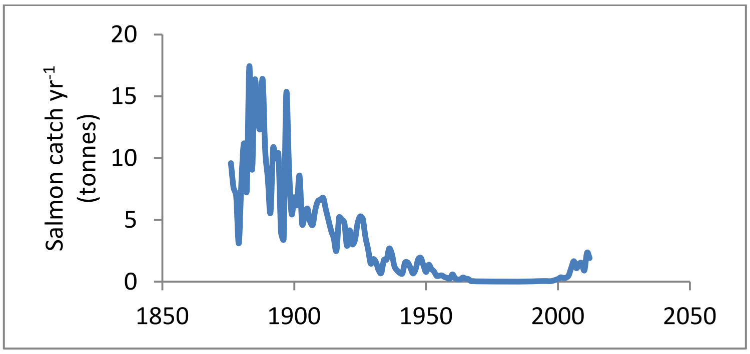

Figure 3.10 Salmon catch statistics for the Tovdal River (1876 – 2012). Liming started in late 1995. Source: Statistics Norway.

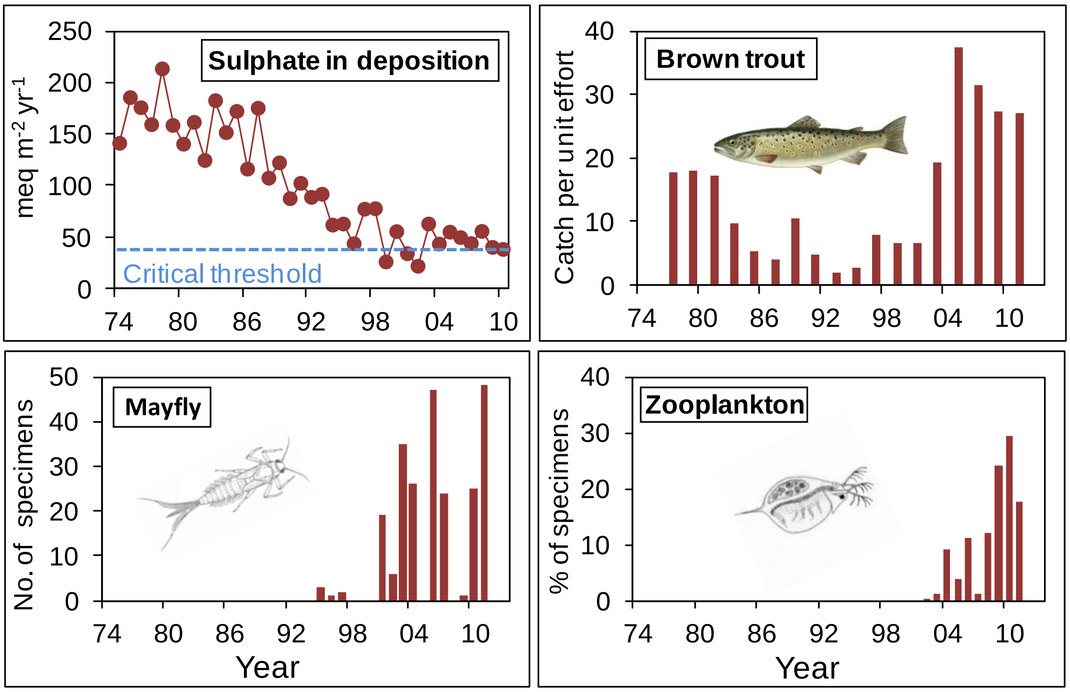

Figure 3.11 Biological recovery from acidification at Lake Saudlandsvatn, a typical non-limed lake in southern Norway, has started in the last decade. There has been a delay between reduced pollution and biological recovery (modified after Hesthagen et al., 2011).

Freshwaters in Europe and North America have begun to recover in response to the reduced acid deposition. Chemical conditions have improved since the mid-1980s in acidified lakes and streams. These changes have been extensively documented by ICP-Waters and national monitoring programmes (Evans and Monteith, 2001; Hruška et al., 2002; Jeffries et al., 2003; Kernan et al., 2010; Skjelkvåle et al., 2007; Skjelkvåle et al., 2003; Stoddard et al., 1999; Wright et al., 2005). In many of these waters the biota also shows signs of recovery, but often lagging many years behind the chemical recovery (**Figure 3.10 and 3.11**). Examples include the Killarney lakes near Sudbury in southern Ontario Canada (Gunn and Sandøy, 2003), rivers (Kroglund et al., 2001) and lakes (Hesthagen et al., 2011) in Norway, and acidified waters in the UK (Monteith et al., 2005). There is a trend towards an increase in the number of benthic invertebrates since the beginning of the 1980's that might be related to improved fresh water quality across Europe.

The loss of sport fisheries is by far the most important ecosystem service affected by acidification of freshwaters. Both inland fisheries (such as brown trout) and coastal anadromous fish (such as Atlantic salmon) are sensitive to acidification. For example, in the 1800's and up to about 1950 the Tovdal River was one of the best salmon rivers in southern Norway. In the period 1876-1885 the annual catch was 9.5 tonnes. But just like several other major salmon rivers in southernmost Norway, the catch began to decline in the early 1900's and by the 1950's only a few hundred kg were caught. By 1970 the entire population was wiped out. The loss of the salmon meant the loss of all the activities dealing with salmon fisheries. The English lords no longer came to fish, the local landowners and fishermen lost a valuable source of income, and the local community was affected. Since late 1995 the river has been limed to raise the pH so that water quality is adequate for salmon. Liming costs were about 420 k€ yr^{-1} at the end of the 20th century. Salmon have now begun to return to the Tovdal River. Liming has also benefited the brown trout population in the river, as well as macroinvertebrates, macrophytes and other organisms (DN, 2010). Cost–benefit analysis in Norway (Navrud, 1993a,b,c) and Sweden (Bengtsson and Bogelius, 1995) showed that liming is socioeconomically profitable in certain water bodies.

Damage and loss of fish populations has large ramifications on other ecosystem services such as tourism, biodiversity, aesthetic value and cultural value. In Sweden, recreational fishing exceeds the economic value of commercial fishing. While effects on aquatic and terrestrial ecosystems due to acidification have been widely studied for decades, they have seldom been evaluated in economic terms. However, valuation research in the context of air pollution and freshwaters has focused largely on damage to fish populations, particularly popular angling species such as trout and salmon. Water acidification also increases corrosivity, and thus increases corrosion of turbines for hydropower production and adversely affects other industrial uses of water. Further details on the impacts of air pollution on freshwater ecosystem services can be found in Holen et al. (2013).

4. Valuation of ecosystem services

The low visibility of biodiversity values has often encouraged inefficient use or even destruction of the natural capital that is the foundation of our economies (TEEB, 2010). The concept of ecosystem services has arisen in response to an increased need for making visible human dependency on nature and ecosystems, in order to ensure sustainable management and avoid irreversible damage to the ecosystems that ultimately will damage human welfare. There is not any straightforward link between the concept of ecosystem services and consequences of ecosystem change for human well-being and the monetary valuation of these changes. However, there is an increasing focus on the relationship between ecosystem services and their economic value and especially the intrinsic value of nature.

While the Millennium Ecosystem Assessment (2005) identifies ecosystem services from a human point of view with a focus on how the ecosystems contribute to human welfare, The Economics of Ecosystems and Biodiversity (TEEB, 2010) is a global initiative focused on drawing attention to the economic benefits of biodiversity. The TEEB study was launched by Germany and the European Commission in response to a proposal by the G8+5 Environment Ministers in 2007, to develop a global study on the economics of biodiversity loss. The study highlights the cost of biodiversity loss and ecosystem degradation and brings together expertise from ecology, economics and development to support the mainstreaming of biodiversity and ecosystem considerations into policy making. The TEEB approach is about recognizing, demonstrating and capturing value (**Table 4.1**). The values of nature vary according to local biophysical and ecological circumstances and the social, economic and cultural context. Intangible values, which may be reflected in society’s willingness to pay to conserve particular species, ecosystems, landscapes or to protect common resources, must be considered alongside more tangible values like food or timber to provide a complete economic picture. However, intangible values are far more complex to determine and considerably less certain than tangible ones.

Table 4.1 The TEEB approach: recognizing, demonstrating and capturing value (modified from TEEB, 2010).

The Economics of Ecosystems and Biodiversity (TEEB)		
Recognizing value	**Demonstrating value**	**Capturing value**
Recognizing value in ecosystems, landscapes, species and other aspects of biodiversity is a feature of all human societies and communities, and is sometimes sufficient to ensure conservation and sustainable use. This may be the case especially where the spiritual or cultural values of nature are strong.	Demonstrating value in qualitative and quantitative terms is, nevertheless, often useful for policymakers and others, such as businesses, in reaching decisions that consider the full costs and benefits of a proposed use of an ecosystem, rather than just those costs or values that enter markets in the form of private goods.	Capturing value involves the introduction of mechanisms that incorporate the values of ecosystems into decision making, through policy incentives and price signals.

The value of natural resources is often considered within the framework of Total Economic Value (TEV; Pearce and Moran, 1994), and this framework can be used also in the economic valuation of ecosystem services. TEV acknowledges that environmental resources have value beyond their direct consumption. Failure to consider all sources of value underestimates the benefits of pollution abatement and inhibits sustainable development planning. The TEEB (2010) study uses the Total Economic Value (TEV) setup to systematize the valuation of economic benefits. Bateman et al. (2003) added the concept of bequest value. This modifies the value of an environmental good to include the value to those alive now of leaving the good for future generations. This then shows up as both a use value, and as a non-use value, on the basis that the future generations will get both kinds of use from the asset (**Figure 4.1**).

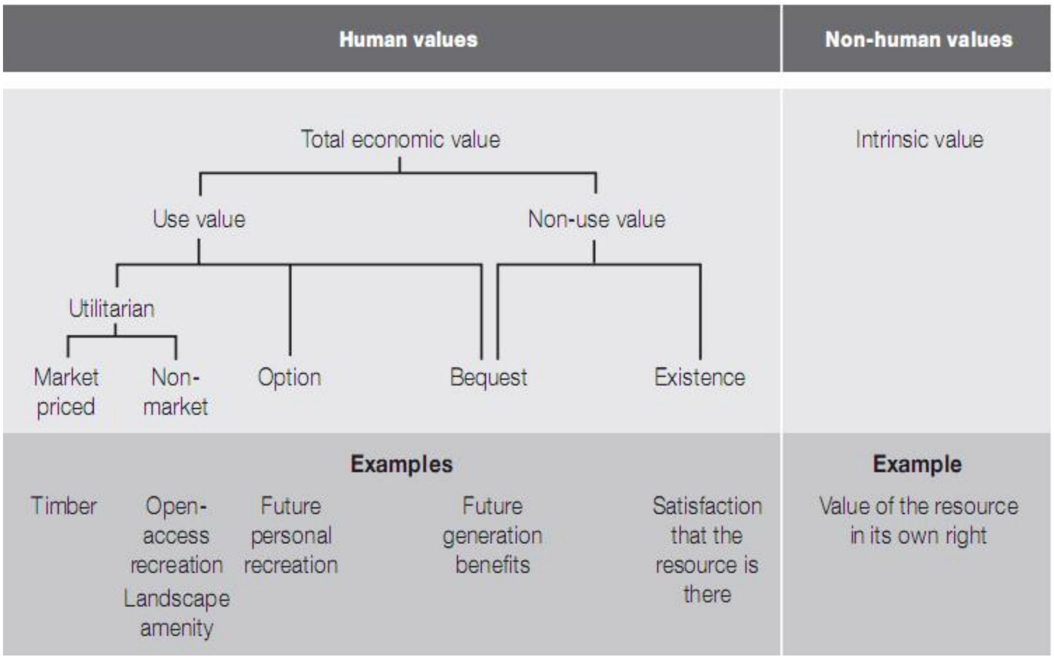

Figure 4.1 Various components of environmental value (from Bateman et al., 2003).

The sum of all categories equals the TEV. However, these are the "economic" values done in an anthropocentric calculation. There is a category of non-economic values as well, often called intrinsic values. These values do not depend on human willingness to pay for them, but are intrinsic to the animal, ecosystem or other part of nature. Although TEV is a useful tool for illustrating the many sources of value for ecosystem services, it is not used directly in cost-benefit analysis calculations.

While environmental economic research has come a long way and improvements in modelling and valuation increase understanding of the socioeconomic impacts of air pollution abatement policies (e.g. Holen et al., 2013; Kettunen et al., 2012; Mills et al., 2013), there are still large gaps in the understanding of the nature and value of air pollution impacts. Despite the fact that some economic-evaluation modelling capacity currently exists, economic-evaluation models for air pollution do not adequately account for all environmental benefits resulting from abatement. It appears that there has never been a thorough analysis of the total costs and benefits of reducing air pollution in terms of ecosystem services. It should also be noted that some of the impacts of air pollution are not instantaneous and thus there is a need to aggregate the costs of impacts over time. This is the present value of the impacts over time, discounted at the social discount rate. The social discount rate is the economic measure of how timing affects values (Birol et al., 2010). The discount rate adjusts downward costs and benefits occurring in the distant future.

5. Conclusions and recommendations

In this report we have provided some examples of data available from several ICPs under the Working Group on Effects of the LRTAP Convention on how air pollution abatement policies provide benefits to ecosystem services and biodiversity and how further benefits can be achieved in the future. We have considered impacts on biodiversity separately whilst acknowledging that biodiversity is an integral part to many ecosystem services and is also subject to valuation. The advantages and disadvantages of valuation in monetary and non-monetary terms have also been discussed.

Biodiversity

We have shown that deposition of reactive nitrogen remains a threat for plant diversity in the future. Particularly so as the effects of excessive nitrogen deposition on the structure and functioning of ecosystems and its biodiversity may not occur instantly, in some instances it may take several decades over which the resilience of soils and vegetation is weakened and impacts become apparent. Large areas in Europe still show exceedance of the nutrient nitrogen critical load and in acids grasslands a reduction in plant diversity due to elevated nitrogen deposition has been shown. So far, little is known about the recovery from historic nitrogen pollution; full recovery might not occur in the future, especially in areas where nitrogen-sensitive plant species have disappeared. Assessments should be extended to other ecosystems and biodiversity indicators (e.g. presence of red list species, soil organisms) for a comprehensive analysis of impacts of excessive nitrogen deposition on biodiversity. Impacts of other atmospheric pollutants also need to be considered. For example, there is a trend towards an increase in the number of benthic invertebrates since the beginning of the 1980's that might be related to a recovery from acidification in fresh water systems across Europe. Also, experiments at different scales have shown that a shift in plant species composition can occur due to ozone exposure. Ozone-sensitive plant species might be outcompeted by more ozone-resistant plant species in areas where the 'uptake' of ozone by vegetation is high (i.e. high phytotoxic ozone dose). However, these observations need to be confirmed by further field-based evidence for impacts of ozone on plant species diversity.

Ecosystem services

The Earth's ecosystems provide an array of services upon which humans depend for food, fresh water, timber production, disease management, air and climate regulation, aesthetic enjoyment and spiritual fulfilment. Such 'Ecosystem Services' are currently grouped according to the benefits they provide to humans, distinguishing between provisioning (e.g. food, fresh water, fuel, wood), regulating (e.g. water purification, water and climate regulation, pollination), supporting (e.g. biomass production, soil formation, nutrient and water cycling) and cultural services (e.g. education, recreation, aesthetic).

Although elevated nitrogen deposition stimulates tree growth in areas where nitrogen is currently the limiting factor for growth, thereby enhancing timber production and the potential for carbon sequestration in forests ecosystems, forest health and vitality may be at risk when organic matter and nutrient cycling is disturbed due to nitrogen enrichment of forest soils. Soils play an important role in storage of air pollutants such as reactive nitrogen and heavy metals, thereby mitigating leaching of these pollutants to water ways and maintaining good water quality. However, the stored pollutants may adversely affect soil functioning (e.g. microbes and invertebrates) and create problems when the retention capacity is reached or disturbed, and pollutants start leaching to surface and drinking water, and coastal zones. Nitrogen also leaches from forest soil at a carbon to nitrogen ratio below 23 in the organic layer and when the critical load is exceeded. This may lead to enhanced algal growth where nitrogen input becomes excessive.

In contrast to nitrogen, current atmospheric ozone concentrations reduce tree growth, resulting in a decline in timber production and the potential for carbon sequestration in forests ecosystems. Hence, emission abatement policies that reduce the atmospheric concentrations of ozone precursors will be beneficial for forest growth and health. Vegetation is an important sink for ozone and therefore plays an important role in improving air quality and mitigating climate change. Ozone is the third most important greenhouse gas and the deposition of ozone to vegetation contributes significantly to a reduction in global warming. In addition, ozone has shown to be a threat to food security by reducing both yield quantity and quality of ozone sensitive species (e.g. wheat and soybean). Such impacts have been valued in monetary terms. For example economic losses of wheat yield were estimated to be €3.2 billion for EU27 + Norway + Switzerland. In addition, ozone might adversely affect the pollination of flowers by for example affecting the synchronization of the time of flowering with the presence of pollinators or floral scent trails in plant-insect interactions. Current ambient ozone concentrations significantly reduce seed number, fruit number and fruit weight compared to pre-industrial ozone levels. Ozone has also been shown to affect water cycling via its impacts on the opening of leaf pores.

A good example of how air pollution abatement benefits ecosystem services has been the decline in sulphur deposition since the establishment of the LRTAP Convention in 1979. Acidification of surface waters in northern Europe due to sulphuric acid deposition had resulted in a loss of fish population and other organisms in many rivers and lakes. However, chemical conditions in many surface waters have improved since the mid-1980s and after a long lag period, biological recovery has started during the last decade. Fish species such as brown trout and salmon have returned, as well as other species such a mayfly and zooplankton. This is of huge benefit to recreational fishing in these areas. However, another remaining problem for fishing is the high level of mercury that has accumulated in fish through the food chain. For example, in over half of the lakes in Sweden, the content of mercury in fish is higher than the recommended values for human consumption.

Conclusions

Based on this report, we draw the following conclusions:

- Awareness of ecosystem services, including biodiversity, in both monetary and non-monetary terms helps to assess the real benefits of air pollution control;
- It is very encouraging that there are signs of chemical and biological recovery from acidification. It remains uncertain whether full recovery of biodiversity from adverse effects of historic air pollution will be possible;
- Further air pollution abatement will continue to reduce the threat to loss of biodiversity, however, “no net loss of biodiversity” will not be achieved by 2020 under the revised Gothenburg Protocol;

- With full implementation of the revised Gothenburg Protocol, further benefits are expected for ecosystem services such as air, soil and water quality and crop production;
- Further air pollution abatement policies will enhance the resilience of biodiversity and ecosystem services to climate change.

Policy recommendations

Based on this report, we make the following policy recommendations:

- To halt biodiversity loss and adverse impacts of air pollution on human well-being, policy negotiations should take into account the benefits of air pollution control for ecosystem services in addition to the direct benefits for human health;
- More stringent air pollution abatement measures beyond the revised Gothenburg Protocol are required to achieve “no net loss of biodiversity”;
- The full benefits of air pollution abatement for ecosystem services (and hence human well-being) have to be assessed and weighed against the costs of more stringent air pollution controls;
- The effects-based integrated assessment of policies that address driving forces of environmental issues could be further balanced by including “no net loss of biodiversity and ecosystem services” in air, waters, soils and vegetation as an explicit endpoint.

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Annex 1 Area at risk (all EUNIS classes) of nutrient nitrogen and the Average Accumulated Exceedance (AAE) from 1980 until 2020 (%). The exceedance in 2020 is computed with nitrogen deposition that follows from the implementation of the Revised Gothenburg Protocol.

	1980		1990		2000		2005		2010		RGP2020	
COUNTRY	Area at risk (%)	AAE eq/ha/a	Area at risk (%)	AAE eq/ha/a	Area at risk (%)	AAE eq/ha/a	Area at risk (%)	AAE eq/ha/a	Area at risk (%)	AAE eq/ha/a	Area at risk (%)	AAE eq/ha/a
Albania	100	453	100	444	99	331	99	298	99	267	99	223
Austria	100	881	100	795	100	444	99	371	92	250	79	166
Bosnia and Herzegovina	97	514	99	551	87	240	88	258	78	197	77	162
Belgium	100	1606	100	1267	100	1011	100	823	99	623	89	419
Bulgaria	100	826	100	767	93	258	94	235	80	133	70	106
Belarus	100	834	100	1065	100	451	100	386	99	358	95	272
Switzerland	100	1119	100	910	99	573	99	607	98	519	95	342
Cyprus	49	62	58	112	49	103	66	120	66	123	66	127
Czech Republic	100	1875	100	1737	100	1129	100	995	100	838	100	728
Germany	95	1385	92	1096	84	672	82	553	73	406	65	305
Denmark	100	1679	100	1559	100	1268	100	1000	100	784	100	647
Estonia	98	253	100	363	78	120	69	89	53	51	42	37
Spain	91	238	95	334	93	258	95	329	92	236	91	208
Finland	56	89	71	117	56	76	46	52	40	36	31	23
France	98	891	98	769	97	584	98	514	95	396	89	302
United Kingdom	30	197	27	154	23	121	25	133	21	94	17	57
Greece	98	447	99	397	100	288	100	290	99	231	99	210
Croatia	100	961	100	829	100	513	100	512	99	425	99	345
Hungary	100	1267	100	976	100	538	100	537	100	429	100	354
Ireland	87	546	85	501	84	560	87	572	83	471	80	415
Italy	76	552	77	551	61	237	68	316	60	235	55	179
Lithuania	100	1001	100	1179	100	502	100	516	100	466	100	369
Luxembourg	100	1617	100	1352	100	1161	100	1025	100	856	100	718
Latvia	100	575	100	730	99	264	99	268	97	216	94	166
Moldova, republic of	100	1040	100	793	100	513	96	314	92	251	92	246
Macedonia, the former Yugoslav republic	100	546	100	481	100	321	100	304	100	251	100	217
Netherlands	96	2500	96	2251	94	1525	91	1284	87	1099	85	844
Norway	24	37	34	74	25	40	19	25	14	15	10	8
Poland	100	1491	100	1523	100	795	100	751	100	633	99	532
Portugal	84	103	90	110	86	110	97	190	83	99	76	80
Romania	66	296	55	248	31	67	20	22	8	5	3	2
Russian Federation	53	135	62	224	29	43	27	31	19	20	19	20
Sweden	66	194	88	271	71	193	54	126	46	93	40	68
Slovenia	100	887	99	801	99	394	96	326	91	207	79	117
Slovakia	100	1341	100	1356	100	707	100	640	100	521	100	431
Ukraine	100	1195	100	1206	100	608	100	481	100	402	100	371
Serbia + Montenegro	99	586	100	519	98	248	97	292	92	215	90	206
EU-27	**80**	**582**	**84**	**562**	**76**	**335**	**73**	**313**	**68**	**237**	**62**	**188**
All	**67**	**371**	**73**	**408**	**54**	**194**	**51**	**176**	**45**	**134**	**42**	**109**

Annex 2 Species diversity per country in EUNIS classes E1, E2 and E3 grasslands as affected by atmospheric nitrogen deposition. RGP = Revised Gothenburg Protocol; MFR = Maximum Feasible Reduction.

Country	Area (Km^2)	1990 (%)	2000 (%)	2005 (%)	2010 (%)	RGP2020 (%)	MFR2020 (%)
Austria	85,2	66,22	71,57	74,4	76,43	78,08	**84,93**
Belgium	614,9	57,57	59,51	64,53	66,66	69,54	**74,46**
Bulgaria	1144,4	75,36	83,68	84,98	86,66	87,13	**90,87**
Cyprus	N/A	N/A	N/A	N/A	N/A	N/A	**N/A**
Czech Republic	845,4	58,71	67,87	72,17	74,68	76,54	**83,38**
Germany	6103,8	59,15	64,09	68,97	71,41	73,45	**80,33**
Denmark	341	66,5	68,42	73,95	77,15	79,29	**83,19**
Estonia	336,5	81,14	85,46	88,79	89,92	90,68	**93,37**
Spain	5912,4	86,51	86,63	85,58	87,24	87,68	**91,69**
Finland	29,3	90,37	90,98	93,44	94,1	94,74	**96,1**
France	7804,2	73,08	74,86	77,7	79,38	80,83	**87,07**
United Kingdom	3198,3	76,88	77,43	78,34	80,17	82,33	**85,95**
Greece	2113,3	86,21	87,18	86,31	87,28	87,41	**89,73**
Hungary	1257,2	72,46	78,87	78,81	80,77	82,11	**88,44**
Ireland	343,2	83,58	82,4	82,47	83,69	84,43	**88,1**
Italy	2586,3	77,14	81,54	81,4	82,99	84,13	**88,03**
Lithuania	320,3	68,66	79,91	80,98	81,9	83,83	**88,33**
Luxembourg	42,9	62,38	63,93	68,97	71,54	73,74	**80,34**
Latvia	728,2	74,46	84,79	86,38	87,32	88,53	**91,69**
Netherlands	648,6	51,79	56,76	61,1	63,58	67,17	**70,84**
Poland	6284,3	64,59	74,21	76,54	78,06	79,56	**84,86**
Portugal	1045	87,77	87,93	86,59	88,12	88,59	**92,19**
Romania	1025,4	71,11	78,02	81,8	83,74	84,75	**89,9**
Sweden	294,1	78,66	80,69	83,8	85,5	86,95	**89,86**
Slovenia	259,6	68,25	73,46	76,25	78,05	79,65	**86,01**
Slovakia	206,2	63,46	73,77	78,11	80,22	81,91	**87,9**
All	**43570,2**	**72,45**	**76,47**	**78,32**	**80,09**	**81,46**	**86,49**

Annex 3 Major relationships between N deposition and ecosystem services as distinguished in the Millennium Ecosystem Assessment (after De Vries et al., 2009).

Ecosystem services	Examples of nitrogen effects	Causal link with nitrogen deposition
Provisioning services		
Food/fibre, including		
- Crops	Increase in crop production	N deposition increases crop growth in N limited systems (low N fertilizer inputs)
- Wild plants and animal products	Impacts on biodiversity (based products)	N induced eutrophication and soil acidification affects soil, plant and faunal species diversity and thereby biodiversity-based products
Timber/wood fuel	Increase in wood production	In N-limited systems, nitrogen increases forest growth and wood production; in N saturated forests, N can induce mortality.
Natural medicines	Impacts on medicinal plants	N induced eutrophication and soil acidification affects plant species, but linkage to medicinal plants is largely unknown.
Fresh water	Impacts on ground water recharge and drainage	N induced impacts on growth and plant species diversity also affect water uptake and thereby freshwater supply (see also water quantity regulation).
Regulating services		
Air quality regulation	Decline in air quality	Nitrogen deposition is correlated with increased concentrations of ammonia (NH_3), nitrogen oxides (NO_x), ozone (O_3) and particulate matter (PM_{10} and $PM_{2.5}$), all affecting human health and ecosystems
Climate regulation Green house gas balance	Increased carbon sequestration in forests	In N limited systems, N deposition increases forest growth and related tree carbon sequestration, but can enhance mortality in some species. It also can cause an increased litterfall and reduced decomposition, leading to soil carbon sequestration
	Increased/decreased carbon sequestration in peat lands	At low N deposition, additional atmospheric N deposition may stimulate net primary productivity. At high rates of N deposition, species composition changes lead to loss of peat land forming species and changed microbial activity causing degradation of peat lands
	Increased N_2O production	Ecosystem losses as N_2O increase with increasing N loading
	Decreased CH_4 consumption	Soil microbes decrease CH_4 consumption in response to increased NH_4 availability
	Increased O_3 production	Increased production in tropospheric O_3 from interactions with NO_x and VOC emitted from ecosystems, which serves as GHG and can also inhibit CO_2 uptake through plant damage
Water quantity regulation	Increased/decreased runoff and ground water recharge	Excess N may cause decreased runoff and ground water recharge due to increased water uptake (elevated growth) but also the reverse because of lower leaf area index due to defoliation caused by either pests or diseases. Recharge may in the long term also be affected by impacts on soil carbon content and soil biodiversity, affecting water retention in soil
	Increased drought stress	Excess N causes an increased need for water by an increased growth and an increased sensitivity for drought stress by an increase in the ratio of above versus below ground biomass
Water quality regulation (water purification)	Decline in ground water and surface water (drinking water) quality	Nitrogen eutrophication and N induced soil acidification increases NO3, Cd and Al availability, leading to: - NO3, Cd and Al concentrations in groundwater and surface water exceeding drinking water quality criteria in view of human health effects - Increased Al concentrations in acid sensitive surface waters resulting in the reduction or loss of fish (salmonid) populations and reduction of aquatic diversity at several trophic levels (acidification) - Fish dieback by algal blooms and anoxic zones (eutrophication). Eutrophication is also affected by silica and phosphorus in estuaries

Annex 3 continued. Major relationships between N deposition and ecosystem services as distinguished in the Millennium Ecosystem Assessment (after De Vries et al., 2009)

Ecosystem services	Examples of nitrogen effects	Causal link with nitrogen deposition
Soil quality regulation	Decrease in acidity buffer; change in soil structure	N induced soil acidification decreases the exchangeable pool of base cations, potentially causing reduced forest growth, and decreases the readily available Al pool, affecting soil structure
Pest/disease regulation	Increased human allergic diseases	Increasing N availability can stimulate greater pollen production, causing human allergic responses, such as hay fever, rhinitis and asthma
	Increase in forest pests	Increase in bark or foliar N concentrations can attract higher infestation rates, such as beech bark disease
Supporting services		
Nutrient cycling and primary production	Increases N inputs by litterfall influences soil biodiversity	N induced impacts on growth/litterfall and on soil biodiversity (soil mesofauna and bacteria composition) affects decomposition, nutrient mineralization and N immobilization, and thereby impacts nutrient cycling and primary production
Cultural services		
Cultural heritage values	Impacts on culturally significant species in historically important landscapes.	N deposition may change heathlands into grasslands, affecting historically important landscapes
Recreation and ecotourism	Impacts on recreation due to impacts on ecosystems	Nitrogen induces the increase in nitrophilic species like stinging nettles and algal blooms reducing recreational and aesthetic values of nature. Examples include closed beaches due to algal blooms resulting from N-induced eutrophication in estuaries and coastal ecosystems.