4 Air Pollution Impacts on Forests in a Changing Climate

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Abstract: Growing awareness of air pollution effects on forests has, from the early 1980s on, led to intensive forest damage research and monitoring. This has fostered air pollution control, especially in Europe and North America, and to a smaller extent also in other parts of the world. At several forest sites in these regions, there are first indications of a recovery of forest soil and tree conditions that may be attributed to improved air quality. This caused a decrease in the attention paid by politicians and the public to air pollution effects on forests. But air pollution continues to affect the structure and functioning of forest ecosystems not only in Europe and North America but even more so in parts of Russia, Asia, Latin America, and Africa. At the political level, however, attention to climate change is focussed on questions of CO_2 emission and carbon sequestration. But ecological interactions between air pollution including CO_2 and O_3 concentrations, extreme temperatures, drought, insects, pathogens, and fire, as well as the impact of ecosystem management practices, are still poorly understood. Future research should focus on the interacting impacts on forest trees and ecosystems. The integrative effects of air pollution and climatic change, in particular elevated O_3, altered nutrient, temperature, water availability, and elevated CO_2, will be key issues for impact research. An important improvement in our understanding might be obtained by the combination of long-term multidisciplinary experiments with ecosystem-level monitoring, and the integration of the results with ecosystem modelling within a multiple-constraint framework.

Keywords: pollution, climate change, monitoring, sulphur, nitrogen, ozone, carbon dioxide

4.1 Introduction

Air pollution impacts on forests have been known for about two millennia. The ancient Romans and Greeks described typical symptoms of air pollution damage in the immediate vicinity of their foundries. Reports on forest damage, even dieback, became more frequent in the medieval era, and subsequently, even more so during the Industrial Revolution. For more than a century, acid precipitation and other forms of localised air pollution had been identified as causing "classical smoke damage," i.e. damage to forests that could be explained by the influence of air pollution released by nearby identifiable sources.

In the late 1970s and early 1980s, increasing forest decline was observed in many parts of Europe; in many cases, it was attributed to the impact of long-range transboundary air pollution (Schütt 1979, Manion 1981, Ulrich 1981). Long-term forest monitoring in Europe has shown that forest conditions there deteriorated far less dramatically than was feared two decades ago. However, three decades of research on forest damage, including the results of recent forest monitoring, has revealed that earlier hypotheses developed by forest researchers have held true in many forest ecosystems across Europe. For example, critical loads of sulphur and nitrogen were exceeded on the majority of the forest monitoring sites (Lorenz et al. 2008). These findings are supported by monitoring and research results in Russia, Asia, and North America. At the same time, however, public, political, and scientific debates are focussing on climate change while attention to air pollution effects on forests is decreasing. Against this background, the aims of the present chapter are to provide the following:

- an assessment of the status of, and the trends in, air pollution in industrialised regions of the world;
- a demonstration that, in many regions of the world, air pollution continues to affect forest health;
- a description of important cause-effect relationships;
- an analysis of the relationships and synergies between air pollution and climate change;
- examples of the mitigation of air pollution and its effects;
- conclusions regarding gaps in knowledge and awareness of the role of air pollution in climate change.

For this purpose, Section 4.2, as a starting point, refers to the most important air pollutants and their sources, with emphasis on spatial differences and temporal developments. Special attention is paid to ozone (O_3) because of its importance in climate change. Section 4.3 describes the main impacts on forests, giving examples from different regions of the world. Interactions between air pollution and climate change are reported in Section 4.4. Good examples for successful mitigation of air pollution and its effects are given in Section 4.5. Section 4.6 draws conclusions on gaps in knowledge and on the awareness of the public, politicians, and scientists on the role of air pollution in climate change.

4.2 Air Pollutants Affecting Forest Health

4.2.1 Main Pollutants and Sources of Their Emissions

Air pollutants may impact trees as both wet and dry deposition. Wet deposition comprises rain, hail, and snow, and is largely determined by atmospheric processes. Dry deposition consists of gases, aerosols, and dust, and is largely influenced by physical and chemical properties of the receptor surface. Forests receive higher deposition loads than open fields, depending on the tree species and canopy structure. A higher roughness of the canopy causes higher air turbulences and more intensive interactions between the air and the foliage. The interception of pollutants by the foliage in turn is determined by such factors as leaf area index, leaf shape, leaf surface roughness, and stomata size. Dry deposition accumulated on the foliage is washed off by precipitation and enhances the deposition under the canopy (throughfall) in comparison to deposition in an open field (bulk deposition). Moreover, throughfall is influenced by two components of canopy exchange: canopy leaching and canopy uptake of elements. The main air pollutants involved in forest damage are sulphur compounds, nitrogen compounds, ozone, and heavy metals.

Sulphur dioxide (SO_2) was the first air pollutant found to cause damage to trees (Stöckhardt 1871). Its air concentrations increased rapidly in central Europe when it was released into the atmosphere by the combustion of fossil fuels during the course of industrialisation occurring at the end of the 19th century. While damaging trees directly via their foliage, SO_2 also reacts with water in the atmosphere to form sulphurous acid (H_2SO_3) and sulphuric acid (H_2SO_4), thus contributing to the formation of acid precipitation and hence to indirect damage of trees (see below). The detection of widespread forest decline in the late 1970s and early 1980s, also in European forest areas remote from industries, raised concerns that this decline might be caused by long-range atmospheric transport of pollutants, mainly SO_2. Throughout the middle of the 20th century, growing awareness of long-range transboundary air pollution threats of acidification to aquatic and terrestrial ecosystems, triggered air pollution control policies from the late 1960s on. This lead to the establishment of the Convention on Long-range Transboundary Air Pollution (CLRTAP) under the United Nations Economic Commission for Europe (UNECE) in 1979. CLRTAP, the European Union (EU), and the laws in many nations started regulating the reduction of sulphur emissions as early as three decades ago. As a result, sulphur emissions in many industrialised countries today are considerably lower than they were 30 years ago. In Russia, SO_2 emissions dropped by 38% from 1980 to 1990. In the same period, they decreased by 9% in the United States of America (USA), and by 17% in Europe. In Europe, there was a further decrease to 60% between 1980 and 2000 (UNECE 2004).

Nitrogen oxides (NOx) are released into the atmosphere in the course of various combustion processes in which nitrogen (N) in the air is oxidised mainly to nitrogen monoxide (NO), with a small admixture of nitrogen dioxide (NO_2). In daylight, NO is easily converted to NO_2 by photochemical reactions involving hydrocarbons present in the air. Both gases, especially NO, are also produced biologically by soil bacteria during nitrification, denitrification, and decomposition of nitrite (NO_2^-) (Finlayson-Pitts and Pitts 2000). These emissions can be quite substantial when they come from highly fertilised agricultural soils and forests exposed to high levels of nitrogen deposition (Erisman et al. 2008). Emissions from transportation are important sources of NOx. For instance, in the USA, out of 23.19 Tg total NOx emitted in 2002, 12.58 Tg could be attributed to transportation emissions (EPA 2008). Large amounts of NOx are also emitted from chemical factories, e.g., during the production of fertilisers (Finlayson-Pitts and Pitts 2000). Fertiliser production, excessive N fertilisation, and livestock farming are also important sources of nitrogen-containing ammonia (NH_3) emissions.

In spite of this, emissions and concentrations of NOx in many industrialised countries have recently significantly decreased. For example, in the European Union (EU), emissions of NO_2 decreased by 32.6% between 1980 and 2000 (Agren 2003), and during the same period by 15% in the USA (EPA 2000). However, reduction of NH_3 emissions are less pronounced. While they decreased by 27.3% from 1980 to 2000 in the EU (Agren 2003), they increased by 3% (EPA 2000) in the USA. Rapid economic development has the potential to significantly increase the emissions of N compounds in parts of Asia. Increasing NH_3 emissions are predicted for Asia, and will be seen mostly in China (Klimont et al. 2001). The world's highest tropospheric NO_2 concentrations have been recorded over Beijing and in the northeast of China (Richter et al. 2005).

Most of the above-mentioned substances are gaseous and act on trees as dry deposition directly via the foliage. Some of them are acidifying and lead – by means of chemical reactions with water in the atmosphere – to acid precipitation. Rain is slightly acidic (pH 5.6), even in the absence of acidifying air pollutants because of the presence of carbon dioxide (CO_2) in the air, which, with water, forms carbonic acid (H_2CO_3). Acidifying compounds such as SO_2, NOx, and NH_3, however, enhance the concentrations of protons and form sulphuric acid, nitric acid (HNO_3), ammonium (NH_4), and nitrate (NO_3).

Heavy metals result from most combustion processes and from many industrial production processes. They are released into the atmosphere by means of dust and, at high temperatures, also as gases. The main heavy metals considered to be detrimental to forest health are cadmium (Cd), lead (Pb), mercury (Hg), cobalt (Co), chromium (Cr), copper (Cu), nickel (Ni), and zinc (Zn). However, largely because of their impacts on human health, heavy metal emissions have been reduced greatly within the last three decades in many industrialised countries. For instance, lead emissions in Europe decreased by about 85% from the years1980 to 2000.

A large group of air pollutants is constituted by volatile organic compounds (VOCs). VOCs include hydrocarbons and organic atmospheric trace gases other than CO_2 and carbon monoxide (CO). VOCs are generated from both man-made sources (anthropogenic) and natural sources (biogenic). Vegetation emits biogenic VOCs that include the isoprenoids (isoprene and monoterpenes), as well as alkanes, carbonyls, alcohols, esters, ethers, and acids. Isoprene is the most abundant hydrocarbon emitted by terrestrial vegetation (530 Tg C/year), and may contribute to increased O_3 to +8–12 ppb in the mid-latitude land areas (Wang and Shallcross 2000).

CO_2, methane (CH_4), and nitrous oxide (N_2O) affect forest health through their action as greenhouse gases. All are produced both naturally and by anthropogenic activities. Although it is debatable to what extent these gases should be considered pollutants, we will here consider the excess production over the natural level as an effect of human activity as pollution. For example, CO_2 is generated by the combustion of fossil fuels or the burning of biomass, CH_4 by anaerobic decomposition of organic matter (e.g., in landfills and rice paddies), and N_2O by agricultural fertilisation and biomass burning.

Tropospheric O_3 is an important phytotoxic air pollutant and a significant greenhouse gas (Bytnerowicz et al. 2007). The impact of O_3 as a greenhouse gas is minimal at temperatures encountered at the Earth's surface, while it is much more effective at the border of the troposphere and stratosphere, where temperatures of –60 to –80°C are encountered (Slanina and Hanson 2006). The troposphere extends above the surface of the Earth and consists of many layers. O_3 is more concentrated above the ground layer and forms a protective "shield" from harmful UV-B and UV-C radiation in the stratosphere. While O_3 is disappearing in the stratosphere, ground-level O_3 concentrations are increasing all over the world. Around 90% of total O_3 is in the stratosphere and just 10% is in the troposphere. Hence, the increase in tropospheric O_3 cannot compensate for the loss in stratospheric O_3. In addition, O_3 in the lower and higher troposphere damages human and ecosystem health, and amplifies the greenhouse effect, respectively.

The formation of O_3 in the troposphere is the result of reactions between solar light and precursors, mainly NOx and VOCs, as well as CH_4 and CO. The overall reactions, however, are complex and non-linear (Stockwell et al. 1997). As O_3 formation is modulated by solar light, daily peaks usually occur around midday. The annual cycle at background temperate sites in the northern hemisphere is characterised by a spring maximum in the month of May (Vingarzan 2004), while at Mediterranean sites, the highest concentrations occur June to August (Paoletti 2006).

Because of anthropogenic emissions of precursors, background O_3 levels have been rising since the first measurements in 1874 (Marenco et al.

1994). Although there are uncertainties about the measurement technique at that time, the mean O_3 concentration was ~10 ppb (Anfossi and Sandroni 1994). Annual average concentrations over the mid latitudes of the northern hemisphere currently range between 20 and 45 ppb, and are expected to be 42 to 84 ppb by the year 2100 (Vingarzan 2004). In the last decades, control strategies have limited the emission of precursors so that O_3 peaks have decreased in North America, Europe, and Asia, but background levels continue to increase (Midgley et al. 2002, EEA 2007). Despite cleaning strategies designed to reduce local emissions of O_3 precursors, global emissions of NOx and VOCs have increased (Royal Society 2008). Transcontinental transport of O_3-enriched air masses from Asia to North America, from North America to Europe, and from Europe to Asia has been demonstrated (Derwent et al. 2004).

Over the past three decades, background O_3 levels in the Northern Hemisphere have increased 0.5–2% per year (Vingarzan 2004). Projections of the International Panel on Climate Change (IPCC) based upon scenarios with high emissions indicate further increase of 20 to 25% (2015 through 2050) and 40 to 60% (through 2100) (Meehl et al. 2007). Increased temperature and reduced humidity will increase O_3 production in already polluted environments, while decreasing it in clean (lower NOx) environments (Royal Society 2008).

4.2.2 Regional Trends in Emissions and Depositions

East Asia

In East Asia, the growth of the population and of economical prosperity have greatly increased air pollution within the past decades. The Acid Deposition Monitoring Network in East Asia (EANET) published its first Periodic Report based on data collected from 2000 to 2004 (EANET 2006). The report indicates that wet depositions of nitrogen and sulphur, corrected for sea salt (nss-sulphur), in East Asia, especially in big cities and their surrounding areas, are significantly larger than those in Europe and the United States. For example, the depositions of nitrogen and nss-sulphur per hectare per year were 29.7 kg N and 52.4 kg sulphur (S) at "Guanyinqiao," an urban site in Chongqing of China. Similarly, they were 25.3 kg N and 20.9 kg S at "Petaling Jaya," an urban site of Malaysia. At "Weishuiyuan," a rural site in China, the respective figures were 20.9 kg N and 54.1 kg S.

In Japan, the wet deposition data collected by the wet-only precipitation samplers are available for more than 80 sites, based on surveys by the Ministry of the Environment of Japan (MOEJ) and Japan Environmental Laboratories Association (JELA). The values measured at these sites were relatively high compared with those measured by the Co-operative Programme for Monitoring and Evaluation of the Long-range Transmission of Air Pollutants in Europe (EMEP), and by the National Atmospheric Deposition Program (NADP) in the USA. The wet deposition of nitrogen in Japan was one and half times larger than in Europe and two times larger than in the United States on the median (7.86 kg N/ha/year in Japan, 5.19 kg S/ha/year in EMEP, and 3.61 kg S/ha/year in NADP). The wet deposition of nss-sulphur in Japan was three times larger than in Europe and in the United States on the median (8.01 kg S/ha/year in Japan, 2.57 kg S/ha/year in EMEP, and 2.85 kg S/ha/year in NADP), although the sulphur deposition in Europe and the United States has decreased in the last decades (Matsubara et al. 2009).

High O_3 concentration and its effects on ecosystems is a recent hot topic in the East Asian region. Several case studies on measurements of ozone concentrations in forest areas have been implemented using passive samplers in several types of forests: in a tropical rainforest in Danum Valley, Sabah, Malaysia; in a tropical dry-evergreen forest in Sakaerat Silvicultural Research Station (SRS), Nakhon Ratchasima Province, Thailand; and in a Japanese cedar forest in Kajikawa Catchment, Niigata Prefecture, Japan. Ozone concentrations at Sakaerat SRS and Kajikawa Catchment were relatively high, even in 15-day mean values, while the concentrations were mostly below 10 ppb at Danum Valley. The annual mean and highest value among the 15-day mean values were 33.7 ppb and 66.7 ppb, respectively, at Sakaerat SRS (Sase et al. 2009); and 42.3 ppb and 59.9 ppb, respectively, at Kajikawa Catchment (Take et al. unpublished data). Especially in Sakaerat SRS, the concentrations were significantly higher in dry seasons from December to March compared to wet seasons.

Europe

In Europe, emissions of the main air pollutants decreased clearly as a consequence of the above-mentioned air pollution control policies under UNECE protocols and EU legislation, but to a different extent with regard to the substances and regions considered. Emission reductions were largest for sulphur, at about 70% from 1980 to 2000, but regional differences were high. In Austria, Germany, Switzerland, and Scandinavia, sulphur emissions decreased by nearly 90%, whereas in southeastern Europe, the respective decrease was only about 40%. These reductions are largely reflected in decreasing air concentrations of SO_2, as well as decreasing SO_4 concentrations in

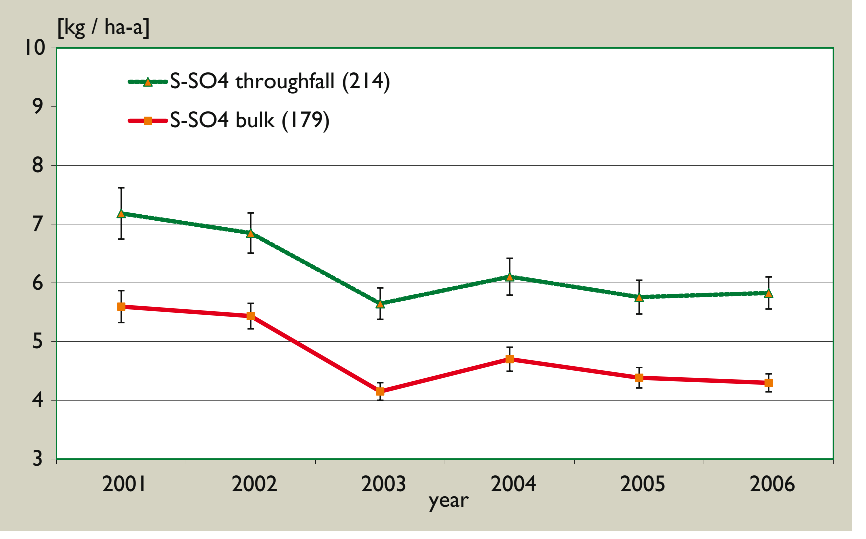

Figure 4.1 Decrease in SO_4 throughfall and bulk deposition as measured by ICP Forests from 2001 to 2006 (Lorenz et al. 2009).

precipitation (EMEP 2004).

While sulphur was mainly emitted by stationary combustion sources, increasing emissions of NOx from traffic also came to be considered a threat to human health and terrestrial ecosystems in the 1980. Despite emission control policies under UNECE protocols and EU legislation, however, emissions of NOx have so far decreased less than emissions of sulphur. For all of Europe, the reduction is estimated to be 25–30% within the last three decades. However, Germany and Switzerland reduced NOx emissions by about 50%. Reductions in some countries of eastern Europe reached between 40% and 50% as a result of the restructuring of their economies. Reductions of NH_4 concentrations in precipitation are comparable to those of NO_3 (EMEP 2004).

The reduction of emissions in Europe reveal themselves in decreasing deposition, as is shown in the annual reports on forest conditions in Europe by the International Co-operative Programme on Assessment and Monitoring of Air Pollution Effects on Forests (ICP Forests) under UNECE. ICP Forests has been monitoring forest conditions in Europe since 1986, and aims, in particular, to identify and quantify the effects of air pollution (e.g., Lorenz et al. 2009). Its deposition measurements from up to 308 intensive monitoring plots show that SO_4 bulk deposition was larger than 8 kg/ha/year on 13.3% of the plots. When corrected for sea-salt deposition, this share is 8.8%. SO_4 throughfall is higher than 8 kg/ha/year on 25.4% of the plots. This value is reduced to 17.8% after sea-salt correction. Figure 4.1 shows throughfall and bulk deposition of SO_4 from 2001 until 2006. Throughfall is higher than bulk deposition, but the slopes of the decrease of both substances are approximately the same. The low in 2003 reflects the drought in the hot summer of that year, as deposition loads are strongly dependent on the amount of precipitation. For NH_4 and NO_3, the decreases in throughfall and bulk deposition were much less pronounced (Lorenz et al. 2009).

A reduction of the formation of O_3 is revealed by a decreasing number of O_3 peak concentrations in the 1990s. But O_3 concentrations still exceed thresholds set for human health and vegetation in many regions of Europe, mostly in central and southern Europe (EMEP 2004). In the USA, a primary standard protects public health, including the health of "sensitive" populations, such as asthmatics, children, and the elderly; while a secondary standard protects public welfare, including protection against decreased visibility, damage to animals, crops, vegetation, and buildings. Primary and secondary standards for O_3 are the same, under the assumption that the primary standard will protect public welfare. The Environmental Protection Agency (EPA) set a primary O_3 National Ambient Air Quality Standard of 75 ppb (EPA 2008). The form is calculated as the 3-year average of the annual fourth highest value in the array of the highest daily maximum 8-hour averages measured at each monitor within an area. At the European Community level (Daughter Directive 2008/50/EC), the target value to protect all types of vegetation is 9 ppm h AOT40, calculated from 1-hour O_3 concentrations from May to July (8:00–20:00) and averaged over five years. AOT40 is the sum of the daylight hourly concentrations >40 ppb. The long-term target value is 3 ppm h AOT40, calculated as above. More stringent, provisional exposure-based critical levels have been set by the UNECE for particular types of European vegetation, under the framework of the Convention on Long-Range Transboundary Air Pollution (ICP Modelling and Mapping 2004). For forest trees, two approaches are used: cumulated

Box 4.1 Ozone impact and risk assessment

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Ozone (O_3) can cause injury to vegetation that may have both ecological and economic impacts, resulting in the need for standards to protect plants. Criteria for protecting forest trees from O_3 have been developed (see a review in Paoletti and Manning 2007). Exposure-based indices, including those at present in use in Europe and the USA, assume that plant injury positively correlates with O_3 exposure. Exposure may have a linear effect, or weigh higher ozone concentrations (Lefohn et al. 1988). A list of 34 of the most common exposure-based indices has recently been reviewed (Paoletti et al. 2007). One of the most frequently used threshold-based indices is AOT40, being the sum of the daylight hourly concentrations >40 ppb, assuming that plants have adapted to low, pre-industrial, naturally occurring O_3 concentrations. These concepts are useful for regulatory purposes, but lack a mechanistic basis. Researchers now recognise that O_3 in the air does not characterise the real potential for plant injury (Matyssek et al. 2007). Injury arises when O_3 enters the leaf and reaches the apoplast and plasmalemma membranes in the mesophyll (Matyssek et al. 2007). At present, only the stomatal O_3 flux (the rate of entry of O_3 into the leaf via the stomatal pores on the leaf surface) has been modelled (Emberson et al. 2000). The stomatal O_3 flux uses algorithms that describe the species-specific effects of temperature, light, soil water pressure, vapour pressure deficit, and plant growth stage on stomatal functioning, but not the effect of the O_3 concentration itself (see Grulke et al. 2007). The hourly mean stomatal flux of O_3 based on the projected leaf area (PLA), F_{st} (in nmol/m^{2}PLA/s), is accumulated over a stomatal flux threshold of Y nmol/m^2/s ($AF_{st}Y$). A flux-based critical level is then the cumulative stomatal flux of O_3, $AF_{st}Y$, above which direct adverse effects may occur, according to present knowledge. An AF_{st} 1.6 of 4 mmo/m^{2}PLA over one growing season has been provisionally identified for sensitive forest trees represented by birch and beech (ICP Modelling and Mapping 2004). Compared to $AF_{st}Y$, AOT40, both overestimates and underestimates O_3 risk in southern and northern Europe, respectively (Simpson et al. 2007).

Whichever the approach for assessing O_3 impacts on forests, there is evidence that ambient ozone levels can cause a range of effects, including visible foliar injury. Ozone, unlike fluoride or sulphur dioxide, does not leave elemental residue that can be detected by analytical techniques. However, O_3 uptake does leave a "signature" of antioxidant upregulation that can be differentiated from photosynthetic pigment oxidation (Grulke et al. 2003). However, visible injury on foliage is the only direct and observable O_3 effect in the field and is regarded as a result of oxidative stress that leads to a cascade of adverse effects. Observation of typical symptoms has turned out to be a valuable tool for the assessment of the impact of ambient O_3 exposures on sensitive species in Europe (Bussotti et al. 2003). Within-species sensitivity may vary, and other factors (e.g., leaf senescence, drought stress, nutritional deficiencies) may mimic symptoms, thus rendering the assessment subject to many constrains. These symptoms can be diagnosed in the field only after adequate training. Ozone visible injury atlases are available on the web at http://www.ozoneinjury.org/. Within the ICP Forests Program for the Assessment of Ozone Injury on European Forest Ecosystems, three countries (Spain, Switzerland, and Italy) were investigated in the period 2002–2004 (Lorenz et al. 2008). On average, 7% of species showed O_3 visible injury on the leaves.

O_3 concentrations (5 ppm h AOT40 over the growing season), and stomatal fluxes (provisionally for sensitive forest trees). For details see the Box 4.1 and Paoletti and Manning (2007).

Among the heavy metals, Cd, Pb, and Hg are of greatest concern in Europe. Their emission sources are mining, foundries, smelters, combustion, and traffic. The introduction of unleaded petrol in Europe decreased Pb depositions by 60–70% from 1990 to 2000. In the same period, emissions of Cd decreased by 30–40%, and of Hg by 50% (EMEP 2004).

Russia

In Russia, non-ferrous metal smelters, power plants, oil and gas processing, and the pulp and paper industry are considered as the main sources of acidifying compounds, such as sulphur dioxide (SO_2), sulphate (SO_4), nitrogen oxides (NOx), and ammonium (NH_4). Ammonium and NOx emissions derive also from diffuse sources, such as agriculture and vehicular traffic. There are three main anthropogenic sources of heavy metal emissions to the atmosphere: fossil fuel combustion, non-ferrous metal production, and waste incineration.


Erkki Oksanen

Photo 4.1 A visible symptom of air pollution damage is defoliation.

The major anthropogenic sources of SO_2 and heavy metal emissions are the non-ferrous metal smelter complexes located in Krasnoyarsk (Norilsk), Murmansk (Monchegorsk, Nikel, Zapolyarnyy) oblasts, and the Ural. The sharp increase in the number of private vehicles has resulted in a clear increase in NOx concentrations in Russian urban cities since 1990. The sources of acidifying gases include the pulp and paper industry in the Republic of Karelia and Archangelsk Oblast.

About 80% of the oil and 99% of the gas produced in the arctic currently comes from Russia (the Nenets Autonomous Okrug and Komi Republic). Oil and gas production involves emissions into the air, including exhaust gases containing CO_2, NOx, SO_x (sulphur oxides), CH_4, and non-methane volatile organic compounds (nmVOCs) (AMAP 2002). Environmental pollution originating from oil and gas exploration and extraction activities is expected to increase considerably in the Eurasian Arctic.

Fires are a major source of SO_2, sulphur aerosols, black carbon, heavy metals, and organic pollutants. It has been forecasted that a warming climate will result in a considerable increase in the area burned annually in boreal forest fires (Stocks et al. 1998). In Russia, 0.5 to 5.5 million ha of managed forests burn annually, and about 80% of these areas are located in Siberia and the Russian Far East (Isaev and Korovin 2003).

4.3 Air Pollution as a Global Risk for Sustainable Forest Development

4.3.1 Cause-Effect Mechanisms

Air pollutants may damage forests directly via the foliage, and indirectly via the soil. The direct effects of O_3, SO_2, NO_2, and NH_3 include visible leaf damage, a decrease in the number of needle age classes in conifers, and elevated pollutant concentrations in plant tissues. Indirect damage is provoked by the negative impacts of deposition of air pollutants via soil-mediated processes. Indirect effects include soil acidification, which results in leaching of base cations, thereby releasing toxic species of aluminium (Al). Air pollution causes water and nutrient imbalances and higher sensitivity to frost, droughts, insect pest attacks, and fungal diseases.

At present, even at the highest concentrations, phytotoxic effects of NOx are very unlikely in forests of Europe and North America (Bytnerowicz et al. 1998). In the areas of rapidly increasing air pollution in China and other parts of Asia, such negative effects should not be excluded. However, elevated levels of N-compounds contribute to eutrophication. Examples are grasslands in the Netherlands and coastal sage communities in southern California (Allen et al. 1998). Nitrogen deposition may have serious ecological effects on ecosystems (Fenn et al. 2005). While N is usually the growth-limiting nutrient in forest and semi-natural terrestrial ecosystems, chronic excess of N can lead to saturation manifested by increased leaching of inorganic N (generally nitrate). Increased leaching of nitrate enhances acidification of soils and surface waters, and the risk of eutrophication of coastal marine areas and groundwater quality. While in Europe and North America the critical loads for nutrient N are set generally between 5 and 15 kg N/ha/year, such values have been set much higher for sensitive Japanese evergreen broad-leaved species. Models suggest that N critical loads of 50 kg/ha/year have already been exceeded in parts of southern China (Kohno et al. 2005).

Increased atmospheric N deposition is well known to reduce plant diversity in natural and semi-natural ecosystems. Examples of such effects are numerous, such as on mixed conifer forests or coastal sage ecosystems in California (Allen et al. 1998, 2007). Negative effects also occur in many other parts of the northern hemisphere and have been studied mostly in Europe and North America. Based on the outputs from global chemistry transport models, Phoenix et al. (2006) predicted that by 2050 the number of biodiversity hotspots will double compared to the mid-1990s, significantly exacerbating global threat of N deposition to world floristic diversity.

The effects of elevated N deposition on plant disease have not been studied in detail, although N enrichment has been shown to increase fungal and bacterial diseases of foliage (Snoeijers et al. 2000). Nitrogen fertilisation has also been shown to increase root rot of eucalyptus caused by *Phytophthora* species (Marks et al. 1973).

Ozone has strong biodiversity effects, such as species-specific and individual-specific effects on resource acquisition and root/crown architecture, and thus space sequestration (Matyssek and Sandermann 2003). In general, poor competitors have an added disadvantage in elevated O_3 conditions (McDonald et al. 2002). Ambient O_3 can cause significant effects on photosynthesis of broadleaved species (–10%) and has less to no effect on conifers (Wittig et al. 2007), suggesting that O_3 gives conifers an advantage in mixed deciduous forests that can lead to changes in community composition. Fast-growing pioneer species, such as birch, aspen, and poplar, have been shown to be more O_3-sensitive than climax species, such as beech and oak (Matyssek et al. in press). This has implications for future O_3 effects if climax species are replaced by fast-growing plantations for agro-forestry and bio-fuel production (Royal Society 2008). Ozone may feed biodiversity changes by feedbacks on VOCs emissions (Lerdau 2007). Biogenic VOCs can either ameliorate or aggravate O_3 pollution, depending on whether the concentrations of NOx are low or high (Lerdau 2007). When plants emit isoprene into the air with high NOx concentrations, O_3 levels can increase. Recent advances, however, suggest that the damaging effects of O_3 are ameliorated by isoprene (Velikova and Loreto 2005, Loreto and Fares 2007). Isoprene-emitting species (e.g., oaks) will thus be better protected against O_3 damage than those that do not produce isoprene (such as maples, birches, or hickories). Most ecosystems consist of both VOC-emitter and non-emitter species. Trees with VOCs emissions will change the atmospheric composition in a way that causes more damage to non-emitters than to emitters. Over time, isoprene-producing taxa may become more abundant because of their greater resistance to O_3. This feedback could lead to changes in forest ecosystems, with decreased plant diversity and a cascade of effects on higher trophic levels and the atmosphere (Lerdau 2007).

The global impacts of O_3 on biodiversity are difficult to predict because of gaps in knowledge. To date, O_3 responses of only a few dozen species in North America and Europe are known, and little is known about the responses of tropical forests, grasslands, and savannah. Olson and Dinerstein (2002) suggested that the areas of greatest risk are in eastern North America, the Alps and other Central European mountains, the northern half of South America, Central Africa, and Southeast Asia. In total, O_3 decreases global plant productivity (GPP) by more than 20% in 17 of the world's priority-conservation ecoregions (G200), covering an area of 1.4 million km^2. In addition, some of the hotspots at high risk from O_3 effects coincide with those at high risk from N deposition, including the forests of Southeast Asia, southwestern China, and the Cerrado of Brazil (Phoenix et al. 2006).

Elevated levels of heavy metals derived from anthropogenic sources can have adverse effects on forest ecosystems, and Hg, Cd, and Pb are currently of the greatest toxicological concern. Biological soil processes that are extremely important for the establishment, growth, and reproduction of the vegetation cover may be indirectly affected by the adverse effects of elevated metal concentrations on soil micro-organisms and invertebrates (AMAP 2006). Heavy metals are associated with several environmental risks to mammals, such as estrogenic effects, disruption of endocrine functioning, impairment of immune system functioning, functional and physiological effects on reproduction, and reduced survival and growth of offspring (AMAP 1998, 2002). In the case of smelters, it is difficult to differentiate the effects of heavy metals from those caused by the high SO_2 emissions.

4.3.2 Regional Trends in Air Pollution

Observations of tree decline have been carried out for forest vegetation monitoring of EANET (EANET 2006). In the first Periodic Report in 2006, several declining symptoms were observed in 16 plots of four countries, but declining levels were mostly Level 1 or Level 2 (based on five classifications ranging from Level 0 [healthy] to Level 4 [dead]). Possible causes of the declining symptoms were mostly natural environmental factors, such as strong winds, topography, soil condition, and suppression of other trees. Insect attacks and disease by fungal infections are also major causes. However, tree decline by unknown factors can be seen in several plots, especially in the plot at Irkutsk, Russia. All the trees within this plot showed some form of declining symptoms. It was surmised, in the National Assessment of Russia, that the effects of local air pollution had an influence on the region's tree decline (EANET 2006). Concentrations of sulphur, fluorine, and heavy metals, such as lead and mercury, in needles of pine (*Pinus sylvestris*), were significantly higher in the Irkutsk area than in reference areas (Mikhailova et al. 2005).

Effects of local air pollution on forests were also suggested by the EANET joint research project in Mongolia. Decline of larch trees (*Larix sibirica*) at the Bogdkhan Mountain near Ulaanbaatar City may be caused by air pollution derived from a thermal power plant or mobile homes. Sulphur contents of larch needles were mostly two times higher on the slope facing the power plant than in reference areas (Sase et al. 2005).

So far, clear evidence of soil acidification is very limited in eastern Asia. The EANET regular-phase monitoring started in 2001, and most monitoring plots have been surveyed only once or twice during the last 8 years. However, at the Lake Ijira Catchment in Japan, where the "Ijira" station is located, soil monitoring has been carried out since 1988. Both surface soil and subsoil showed decreasing trends of soil pH. The mean pH of surface soil decreased from 4.5 in 1990 to 3.9 in 2004 (Nakahara et al. 2009). Exchangeable Al in the soil slightly increased with decreased pH levels of the soil. At present, effects of soil acidification on tree health have not been observed in the Lake Ijira Catchment.

The impact of air pollution on forests in Europe has been a subject of intensive forest damage research and monitoring since the detection of widespread forest decline in the late 1970s and early 1980s. Three decades of forest damage research and 25 years of monitoring by ICP Forests have, meanwhile, shown that forest conditions in Europe deteriorated far less dramatically than originally feared. However, hypotheses developed by forest researchers in the 1980s hold true in many forest ecosystems across Europe.

As widespread forest damage in Europe was identified by means of a thinning of tree crowns, ICP Forests launched an annual Europe-wide annual defoliation assessment in 1986. Since then, defoliation has been highest in the Czech Republic, Germany, Poland, and the Slovak Republic; i.e., in the region of central Europe showing the severest damage due to local and long-range transboundary air pollution at that time. Moreover, during the first years of the assessments, defoliation was found to increase strongly in Belarus, Bulgaria, Croatia, Denmark, France, Italy, and Ukraine. But recuperation was also observed in several regions. At the Europe-wide scale, the development of defoliation is mainly influenced by tree age, weather conditions (mainly heat and drought), and biotic factors (insects and fungi). Statistical relationships between defoliation and air pollution are weak. The annual defoliation assessments have been retained because they are a low-cost tool for the timely detection of detrimental developments of forest conditions. For specific assessments of air pollution effects on forests, other approaches have been implemented that involve deposition measurements and soil analyses.

An approach for the assessment of forest damage by air pollution is the calculation of critical loads of deposition (Nilsson and Grennfelt 1988) and the determination of their exceedances. The critical load constitutes the tolerable deposition of a given pollutant per unit of time that will not cause any long-term adverse effects to the structure and functioning of a forest ecosystem. It can be derived from parameters of the chemical composition of the dry and wet phases of the soil. Damage can be expected when certain chemical parameters (= critical limits) of the soil and soil solution are violated and, therefore, will lead to destabilisation of soil processes (ICP Modelling and Mapping 2004).

Results show that critical loads for N and acidity in forest soils differ greatly across Europe. Their spatial variation is largest in geologically variable regions, such as central Europe and the Alps. Geological uniform regions, such as northern Europe, with their mainly acidic parent material, have only slightly varying critical loads. Critical loads for N show high spatial variation. The lowest critical loads for N are found mainly on plots in Norway, Finland, northern Germany, The Netherlands, Belgium, Spain, and Greece. They characterise forest ecosystems that are sensitive to high nitrogen inputs. Critical loads for acidity and their spatial variation are higher than those for N. Forest sites with lowest critical loads for acidity are most frequent in Finland, Norway, The Netherlands, and various parts of Germany, indicating forest ecosystems most susceptible to acid deposition. High critical loads prevail on forest sites in the United Kingdom, in Spain and Greece, as well as in several parts of central Europe. Nitrogen throughfall, measured by ICP Forests in 2004, exceeded the critical loads on about two-thirds of the assessed sites (Figure 4.2). This shows that N deposition and the resulting accumulation of N in forest soils remain a widespread risk. Exceedances are highest at forest sites in The Netherlands, Belgium, and parts of Germany. Critical loads for N are not exceeded at most sites in Finland, Norway, the United Kingdom, and in the Lower Alps in Switzerland, Germany, and Austria. In the Lower Alps, however, it must be taken into account that high precipitation causes high N leaching and, consequently, little N accumulation in the soil, and hence high critical loads. This means that critical loads for N are not exceeded even if N deposition is high, but that high deposition causes nitrate leaching, which is detrimental to groundwater quality. Critical loads for acidity were exceeded on less than a quarter of the sites. Exceedances are particularly high in The Netherlands and in southern Sweden, several parts of Germany, and in Hungary (Lorenz et al. 2008).

Forest damage research in Europe has revealed numerous direct and indirect effects of air pollution on such factors as the nutritional status, crown condi-

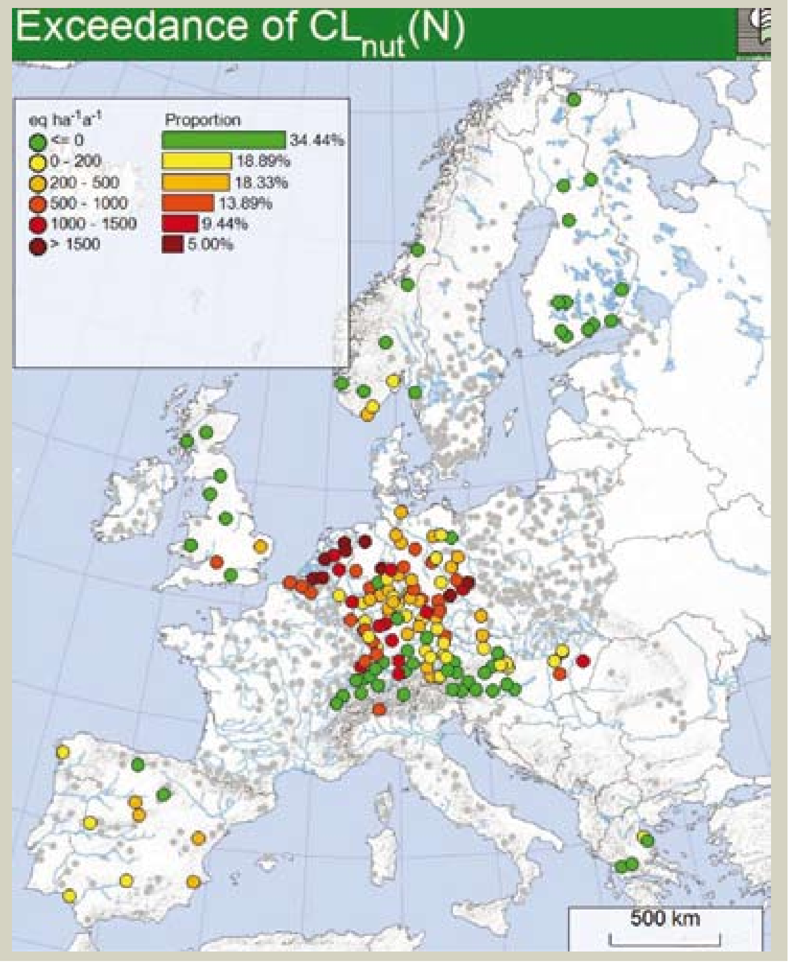

Figure 4.2 Exeedance of critical loads of nitrogen (Lorenz et al. 2008).

tion, and growth of forest trees at many individual sites. On ICP Forests' intensive monitoring plots in Germany, with exceedances of critical loads for acidity, for instance, Augustin et al. (2005) found high S contents in needles and leaves that were weakly correlated with defoliation. Nitrogen deposition at ICP Forests sites was related to species composition of ground vegetation. Nitrogen-indicator plants occurred more frequently at sites with high nitrogen deposition (Lorenz et al. 2007). Nitrogen deposition was also shown to enhance tree growth.

In Russia, the visible effects of air pollution have been observed in the surroundings of non-ferrous metal smelters. In Norilsk, the total area of barren and semi-barren land currently exceeds 400 000 ha (Kozlov and Zvereva 2007). An area of about 2 million ha of taiga-tundra forests are dead or severely damaged by pollution from the Norilsk smelters in NW Siberia. The area of industrial barren and semi-barren land, caused by pollution from the Cu-Ni smelters on the Kola peninsula in northwestern European Russia, totalled 75 000 ha at the beginning of the 1990s. The total area on the Kola Peninsula where the critical deposition of sulphur (0.3 $gS/m^2/g$) is exceeded is more than 90 000 km^2 (Tuovinen 1993). There is visible damage to forest ecosystems over an area of 39 000 km^2 (Rigina and Kozlov 1998). Effects of air pollution combined with other disturbances, such as fires (because of larger amounts of combustible materials on polluted areas), fungal diseases, and insect attacks, may cause an increase in forest damage over significantly larger areas.

4.4 Interactions Between Air Pollution and Climate Change

Air pollution may interact with climate change in ways that tend to enhance warming (positive feedbacks) and through effects that tend to mitigate warming (negative feedbacks) (Derome and Lukina, in print). The major mechanisms (feedbacks) by which environmental pollution can contribute to climate change are changes in land cover (albedo) and changes in the amount of greenhouse gases emitted from the land to the atmosphere.

There is still little clear evidence to suggest how the dual threats of air pollution and climate change may interact. In certain areas of northern Europe, rainfall has been predicted to increase as a result of climate change. In this case, even with a decrease in pollutant emissions, total pollutant deposition may not decrease substantially. It has also been suggested that, along with increased temperatures, soil nitrification processes may be speeded up (Tickle et al. 1995). This may result in increased nitrate leaching from forest ecosystems, resulting in enhanced soil acidification. Climate change may also have an effect on contaminant pathways in woodlands.

Climate change may aggravate the effects of pollutants. The melting of permafrost will result in the release of many contaminants currently encapsulated in the soil; for instance, in areas subjected to air pollution from Norilsk. The combined effects of SO_2 and heavy metal pollution and fire result in the replacement of coniferous forests by birch forests, which have a different albedo and carbon cycle. Even though tropospheric O_3 is an unstable pollutant that tends to form local hotspots of high concentrations, its radiative properties as a warming agent are global. IPCC estimates tropospheric O_3 radiative forcing at 0.39 W/m^2 (Forster et al. 2007), thus ranking O_3 as the third most important anthropogenic greenhouse gas after CO_2 and CH_4. Ozone will also have an indirect effect on climate change by reducing the capacity of vegetation to sequester carbon (C). This indirect radiative effect could increase the total radiative forcing due to O_3 over the period 1900–2100 by at least 70% (Sitch et al. 2007).

Projected changes in climate are likely to increase regional and local O_3 concentrations where emissions are high (Royal Society 2008). An increase in the production of O_3 is influenced by increased temperature, increased sunlight, decreased humidity (all components of climate change), and the increase in long-range transport of pollutants. Interactions between climate and the terrestrial biosphere have an important effect on surface O_3 levels because vegetation is both a source and a sink for O_3. Also, biogenic VOC emissions are sensitive to environmental factors such as temperature and light. One of the most important feedbacks between O_3 and climate is through temperature-increased emissions of O_3 precursors, including biogenic VOCs, as well as NO from soils, and CH_4 from wetlands (Royal Society 2008). The expected increase in spells of excessively high temperatures in the northern hemisphere will increase the frequency of high O_3 episodes. Ozone-induced reductions in stomatal conductance and transpiration imply reduced transfer of water to the atmosphere, decreasing humidity, and potentially altering regional rainfall patterns in temperate and boreal forests (Wittig et al. 2009).

Because O_3 reduces carbon assimilation (Novak et al. 2005, Wittig et al. 2007), there is great concern about its impact on the reduced capacity of forest ecosystems to sequester C, although there are few long-term studies to fully assess this effect (Manning 2005). A recent meta-analytic review of 263 peer-reviewed articles reporting O_3 impacts on northern hemisphere tree biomass showed that current ambient O_3 concentrations (40 ppb on average) significantly reduced the total biomass of trees by 7% compared with control trees in charcoal-filtered air, which approximates pre-industrial concentrations (Wittig et al. 2009). When elevated O_3 concentrations (64 ppb on average, close to the 68 ppb projected by 2050; Ehhalt et al. 2001) were examined, total biomass was reduced by 11% compared with trees at present ambient O_3, while 97 ppb (projected by the end of 2100) reduced total biomass by 17% relative to pre-industrial levels, which is a further 10% reduction relative to today. These results demonstrate that O_3 has been reducing the C-sink strength of northern hemisphere forests since the pre-industrial age, and will further reduce it in the future (Wittig et al. 2009).

These conclusions were based on controlled experiments with young trees. These kinds of results cannot simply be translated to adult trees in the field (Kolb and Matyssek 2001, Schaub et al. 2005), although negative correlations between stem growth and ambient O_3 exposure in mature trees in the field have been reported (Braun et al. 1999, Karlsson et al. 2006, McLaughlin et al. 2007). By comparing biomass reductions in aspen (*Populus tremuloides*) from the Free Air CO_2 Enrichment Experiment (FACE) in Rhinelander, Wisconsin in the USA (King et al. 2005), and in *Populus* species from the meta-analysis, similar values were obtained (–21% and –22%, respectively), although O_3 exposure was lower in the former than in the latter (50 ppb and 60 ppb, respectively) (Wittig et al. 2009). As most of the studies in the meta-analysis were short-term experiments in artificial chamber environments, this suggests a greater impact of O_3 when applied in open-air and is consistent with findings about soybean biomass in growth chambers (Morgan et al. 2003, 2005). Concern thus arises that the large losses projected across the chamber studies may be underestimates of what will occur in the real world and over longer, more realistic growth periods (Wittig et al. 2009). Further uncertainties arise from gaps in information in the literature, which did not allow assessing the magnitude or significance of the interactions between O_3 and drought, or O_3 and CO_2 (Wittig et al. 2009).

Moderately high to high O_3 exposure imposes a loss in annual carbon balance of trees, with consequences for the atmospheric CO_2 balance. Tree seedlings need more light to achieve a positive C balance, and larger gaps are required. An increase in temperature is currently observed (0.5 to 0.8°C increases, van Mantgem and Stephenson 2007), and further increases are expected over the next century due to climate change. Leaving some pole-sized trees within large gaps will mitigate the increased temperatures associated with larger forest gaps. Moderately high to high O_3 exposure increases tree susceptibility to drought stress, with subsequent deleterious effects on forest stand susceptibility to insect and pathogen outbreaks (Grulke et al. 2009). Forest management strategies to reduce individual tree-level drought stress can mitigate the effects of tree susceptibility to O_3 through thinning, not just at the stand level but to ensure adequate spacing between trees. To adapt forests for the future, larger gaps with sparse pole-sized trees left standing, as well as forest patches that have been thinned, will help mitigate the expected increase in drought with climate change (including the chemical environment). For afforestation efforts, drought-tolerant ecotypes could be used if there are not other limiting factors (e.g., contamination of local gene pool), or more drought-tolerant species appropriate to the ecological zone.

Modelled estimates of C sequestration by forests in the Great Smoky Mountains National Park (southeast-central USA) attribute a 50% loss to ambient O_3 between 1971 and 2001 (Zhang et al. 2007). Between 2001 and 2003, a maximum of 31% loss in productivity of aspen in parts of its North American range was estimated to be caused by O_3 (Percy et al. 2007). Models, however, should include O_3 effects on both photosynthesis and plant avoidance/defence ability (Matyssek et al. 2007), as well as feedbacks and susceptibility resulting from reduced C allocation to below-ground tissues (–15% of root-to-shoot ratio in Wittig et al. 2009). Ozone alters source–sink balance, initially resulting in C retention in shoots and decreased C allocation below ground to roots and mycorrhizas (Andersen 2003). Decreased allocation below ground alters C flux to soil and soil processes. In parallel, above-ground allocation is also affected. Compensatory growth of new leaves may occur by using nutrient and C from declining leaves or reserve storage (Matyssek and Sandermann 2003). Reduced branching and leaf size, along with premature leaf loss may limit biomass production more than the decline in leaf photosynthesis (Matyssek 2001). Trees that allocate less C to fine-root system production are competitively disadvantaged for exploiting soil resources. Elevated O_3 and N deposition can alter root function and health by decreasing the degree of mycorrhizal colonisation or altering community structure (Grulke et al. 2009). Ozone exposure increases lignification of above-ground tissues, decreases decomposability, and thus a more recalcitrant N pool develops (Treseder and Allen 2000).

The complex range of effects on trees caused by O_3 also highlights the importance of understanding the combined effects of O_3 and other environmental factors, such as elevated CO_2. The predicted benefits of increased CO_2 concentrations on C storage may be offset at high O_3 concentrations. The total C incorporated into soil under aspen and aspen–birch components was reduced by about 50% over 4 years in the elevated O_3 + CO_2 FACE exposure compared with the elevated CO_2 treatment (Loya et al. 2003). Using the A2 scenario, Sitch et al. (2007) predicted a reduction in terrestrial C storage over the period 1900–2100 of 143 Pg C, or 17% of the C storage projected to occur due to increasing CO_2 concentrations over this period. Biogeochemical models suggest that O_3 may offset about 7% of the gains in productivity projected with increasing atmospheric CO_2 and N deposition (Ollinger et al. 2002, Felzer et al. 2004, Sitch et al. 2007). This type of large-scale model provides new insights into important global feedbacks, even though there are still significant uncertainties because parameterisation is based on a small number of long-term experiments that do not represent a range of global plant functional types, and neglect the combined effects of O_3, CO_2, N deposition and climate (Royal Society 2008). Additional uncertainties arise from O_3 effects on other greenhouse gases, such as CH_4 (Fiore et al. 2002, Rinnan et al. 2003).

Although most of the research on O_3 effects focus on the C cycle, O_3 has a number of other effects on forest ecosystems that indirectly impact C sequestration, but are not yet included in models. Ozone may weaken frost-hardening and predispose trees to frost injury (Skärby et al. 1998; Ranford and Reiling 2007). Ozone may also affect plant/disease interactions (Manning and von Tiedemann 1995). Ozone injury has been shown to predispose ponderosa pine (*Pinus ponderosa* Dougl.) to annosus root disease (*Heterobasidion annosum*; James et al. 1980) and black stain root rot disease (*Leptographium wageneri*; Fenn et al. 1989). Although long-term O_3 exposure reduces stomatal conductance, peaks affect stomatal control and may predispose trees to drought stress (Paoletti and Grulke 2005). Mature deciduous trees from a mixed forest were shown to amplify diurnal water loss because of O_3-induced increases in sap-flow (McLaughlin et al. 2007). Ozone may alter tree leaf chemistry and insect herbivore performance (Percy et al. 2002, Valkama et al. 2007). In the San Bernardino Mountains of south-central California, bark beetle activity and mortality of ponderosa and Jeffrey pine (*P. jeffreyi*) were positively related to O_3 injury and N deposition or experimentally amended N level (Stark et al. 1968, Eatough-Jones et al. 2004). Environmental factors that reduce stomatal aperture, such as drought and O_3, and the root-to-shoot ratio, such as O_3 and N, consequently reduce both C acquisition in the short term (daily, cumulative) and nutrient flow in the transpirational stream to the foliage over the long term (months). When attacked, resin production provides both a physical and chemical impedance to bark beetle attack. Oleoresin pressure, related to the turgor potential of cells lining the resin ducts, forces pre-formed resin to the site of invasion. The cell turgor is derived from the transpirational stream; thus, if the tree is under O_3, N, or drought stress, the resin pressure is reduced (Vité 1961). Once mature bark beetles emerge from the galleries, they begin searching for a host. Interestingly, conifers under moderate drought stress produce jasmonate, a plant hormone that stimulates resin production (Zeneli et al. 2006). Dense stands of pine, overgrown from excess N deposition and historic land management practices, may produce ample signal (terpene production), which could alter non-olfactory aspects of short-range host selection behaviour (Grulke et al. 2009). Warmer springs are anticipating the start of the second generation of bark beetles in the southeastern Alps (Faccoli 2009), suggesting that climate change increases bark beetle pressure also by increasing the number of generations in a year.

4.5 Mitigation of Air Pollution and Its Effects on Forests

4.5.1 Examples of Regional Air Pollution Control

In East Asia, EANET has started a process to discuss an appropriate instrument and its legal status to provide a sound basis for financial contribution to EANET. Among the activities being implemented by EANET are capacity building, information sharing, and raising public awareness on acid deposition and related air pollution issues. There are other initiatives promoting regional environmental cooperation in East Asia, such as the north-east Asian Sub-regional Programme for Environmental Cooperation (NEASPEC), the ASEAN Transboundary Haze Agreement in Southeast Asia, as well as programs implemented under UNEP. It is hoped that the current efforts and pollution abatement measures of the countries can effectively conserve the vast and varied forest ecosystems of East Asia.

Most countries in Europe, the European Community, and several countries in North America and Asia have signed CLRTAP under UNECE (Section 4.2.1). CLRTAP is meeting its aim to control air pollution by means of a series of legally binding protocols on the reduction of emissions of S, N, O_3, heavy metals, VOCs, and persistent organic pollutants (POPs). Within the last three decades, CLRTAP and related legislation of the EC have succeeded in improving air quality and reducing deposition of air pollutants. Between 1980 and 2000, S deposition decreased by 70%, and N deposition decreased by 30% in Europe (Section 4.2.2).

The success in air pollution control is not only reflected by decreasing deposition in the forests in Europe (Section 4.2.2), there are also first indications of the start of recovery of forest soils and trees. There is a correlation between the decrease in defoliation of *Pinus sylvestris* in Europe since 1994 and the decrease in S deposition. This holds true in particular in regions of previously high sulphur deposition and defoliation in parts of Poland, the Czech Republic, the Slovak Republic, and the Baltic States (Figure 4.3) (Lorenz 2004). De Vries et al. (2001) reported the recovery of forest soils from S deposition. Lorenz et al. (2007) concluded that the acidity of forest soils was the highest around 1990, and decreased within the last decade. Dynamic modelling approaches predict that the acidity of soil solution will continue to decrease if air pollution control continues to be implemented, as specified by the CLRTAP protocols.

In Russia, sulphur dioxide emissions from the largest non-ferrous smelter complexes located on the Kola peninsula and Norilsk have decreased substantially during the last couple of decades due to changes in production and better technology for controlling emissions. The main reductions have occurred since 1995, and have been considerably greater on the Kola Peninsula than at Norilsk. SO_2 emissions from the Pechenganickel (Zapolyarnyy/Nikel) and Severonickel (Monchegorsk) smelters decreased by 1.7 and 5 times, and emissions of heavy metals from the Severonikel smelter on the Kola Peninsula decreased considerably (as much as 2–3 times). A further decrease in the emissions of pollutants is needed.

4.5.2 Possibilities of Adaptation to Air Pollution

According to a meta-analysis of peer-reviewed studies (De Schrijver et al. 2007b), atmospheric deposition of acidifying and eutrophying compounds and the leaching of nitrate, sulphate, and base cations (e.g., calcium [Ca] and magnesium [Mg]) from the soil were significantly higher in coniferous forests than in deciduous forests at comparable sites. Deciduous forests annually received less N and S via throughfall (+ stemflow) deposition on the forest floor than adjacent coniferous forests. The mean ratio of throughfall (+ stemflow) deposition flux under coniferous and deciduous canopies was 1.72 for both ammonium (NH_4^+) and nitrate (NO_3^-), and 1.74 for sulphate (SO_4^{2-}). The deposition of base cations was also significantly higher for coniferous than deciduous forests, but to a smaller extent than for N and S. Coniferous forests thus annually intercept more atmospheric pollutants than deciduous forests in comparable site conditions, most probably due to the generally higher height and stand density and

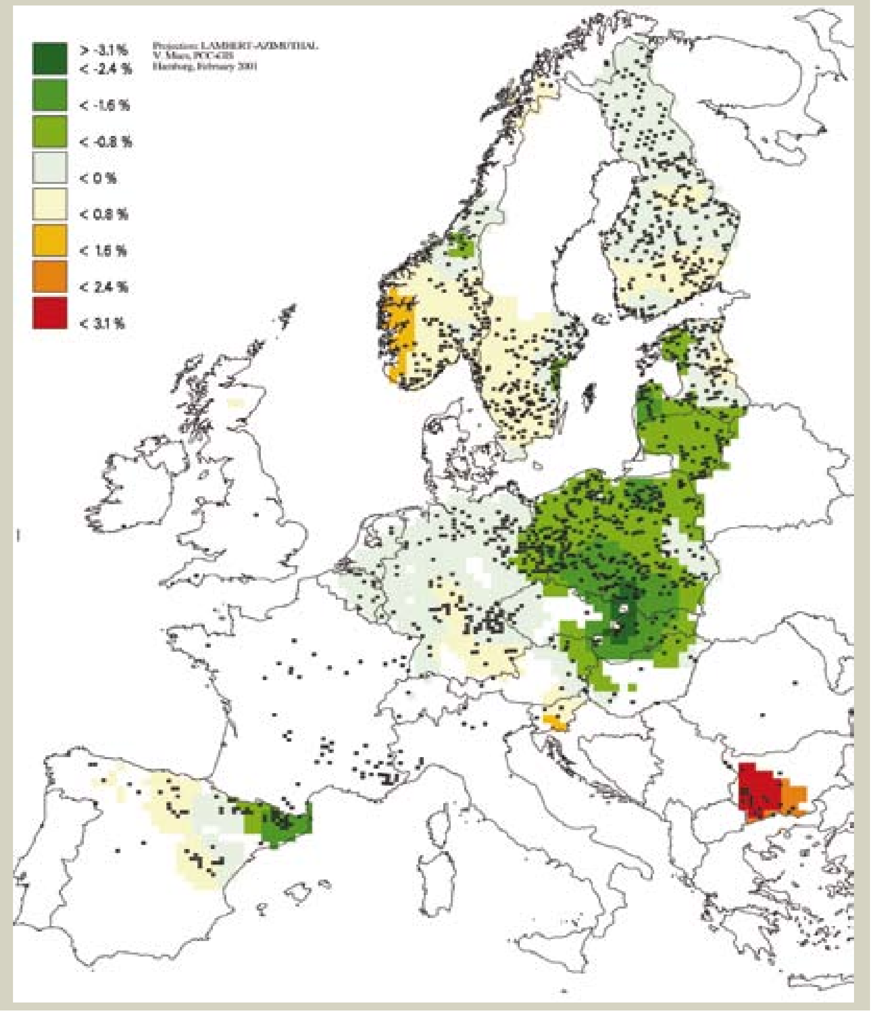

Figure 4.3 Changes in defoliation of *Pinus sylvestris* from 1994 to 2000 in percent of defoliation (Lorenz 2004).

the evergreen character of coniferous forests and the long narrow shape of their needles. The significantly higher throughfall (+ stemflow) flux of inorganic N in coniferous forests is clearly reflected in the higher leaching of nutrients from the soil. Leaching was, on average, almost twelve times higher under coniferous than deciduous forest stands for NO_3^-, and ten times higher for inorganic N. Sulphate leaching was, on average, 1.6 times higher under coniferous soils. The leaching of NO_3^- and SO_4^{2-} was closely related to the leaching of base cations and aluminium.

Consequently, the conversion of high-density pine plantations into deciduous forest types can reduce the atmospheric input of eutrophying and potentially acidifying deposition to forest ecosystems and, in the long term, decrease the loss of essential nutrients from forest soils. Combined with the generally higher litter quality of deciduous forests, and the consequently faster decomposition and internal nutrient cycling, effects of air pollution can be diminished by conversion to deciduous forests.

The effect of deciduous vs. coniferous forest type on acidifying and eutrophying deposition is even more enhanced in forest edges than in the interior of forest stands. Hence, tree species choice is even more important in landscapes with fragmented forests. Forest fragmentation occurs when continuous forests are divided into smaller isolated patches by activities such as timber harvesting, road construction, clearing for agricultural expansion, urbanisation, or other human development. Increasing forest fragmentation increases the proportion of forest area exposed to forest edge effects, so that forest edges can become dominant features of the landscape (De Schrijver et al. 2007a, Echeverria et al. 2008). The edge between forested and non-forested area can have important effects on microclimate, forest regeneration, and biodiversity, and also affects nutrient fluxes. Many nutrients and pollutants are delivered to ecosystems via atmospheric transport and wet, dry, and occult deposition. Because of the steep transition in vegetation height at most forest edges, air flow is disrupted, and canopy wind speed and air turbulence are enhanced at the edge, thus enhancing dry deposition of particles and gases. Thus, forest edges have been shown to act as hotspots of deposition, showing up to a fourfold increase in the rate of atmospheric delivery compared with nearby areas without edges (Weathers et al. 2001; see De Schrijver et al. 2007 for a review). In addition to these atmospheric processes, edge gradients in forest structure, soil, microclimate, and precipitation can alter nutrient fluxes in forest edges.

Adaptive forest management is of great importance in Russia. A forest monitoring system providing reliable, internationally harmonised data is being developed in Russia to assist in decision making. ICP Forests monitoring activities are currently carried out by the Federal Agency of Forestry in the Leningrad, Pskov, Novgorod, Kaliningrad, and Murmansk oblasts, and in the Republic of Karelia, in the 500 km-wide zone along Russia's western borders. Because fires are a major source of many pollutants, more efficient forest protection from fires will result in a significant decrease in air pollution. The remote sensing monitoring information system for fire detection and propagation prevention is now operating at the Federal Agency of Forestry. Possible technologies for the rehabilitation of land subjected to air pollution have been suggested, and first results on their implementation by local forestry enterprises have recently been received. For supporting the vitality of coniferous forests at a distance from the smelters, the approaches to their nutritional status improvement by treatment of the soil with appropriate ameliorative substances and fertilisers have also been suggested.

Mitigating tropospheric O_3 means reducing emissions of precursor chemicals, particularly NOx and VOCs. In the USA, control options for O_3 include tailpipe NOx filters in vehicles and regulation of oil tankers and gasoline stations to reduce leakage into the environment (USCCTP 2005). Because the chemical reactions that produce O_3 increase at higher temperatures, simple measures that reduce the urban heat-island effect (such as white or green roofs, solar panels covering roofs, or shade trees) can also lower precursors to O_3 production (USCCTP 2005). Concentrating air pollution control on tropospheric O_3 is likely to yield significant cross-cutting benefits in terms of climate, human health, forests, and agriculture. It is also likely to be highly cost-effective because much of the technology to control emissions already exists and has been deployed with effect in developed countries (Moore 2009). Working Group 3 of the IPCC Fourth Assessment Report discusses several climate change mitigation policies that also have air pollution co-benefits. Most decarbonisation strategies in the electricity and transport sectors, either by increasing energy efficiency or switching to renewable technologies, will reduce emissions of air pollutants, including O_3 precursors and black carbon. Depending on the valuation of these health benefits, they could amount to as much as three to four times the cost of mitigation (Barker et al. 2007).

The technologies available for the restoration of land impacted by air pollution have been elaborated and have recently been applied in the surroundings of the most significant sources of air pollution in Northern Europe – the "Severonikel" and "Pechenganikel" smelters on the Kola peninsula. Today, soil properties are a more critical factor in the suppression of vegetation cover development than air quality. Limestone application and treatment of the soil with fertilisers promote colonisation by native plant species: mosses, herb plants, and willows in the polluted areas. The introduction of grass species in the barren land gives some advantages: rapid establishment of plant cover and improvement of soil nutrient status for succeeding colonisation by native plants species, including woody plants, such as willows. Seeding with grasses in the zone of semi-barren woodland contributes to the development of an understorey layer, which is currently almost absent. This allows improvement of the soil conditions for succeeding colonisation by native plants species. The limiting factors for plant cover development in the barren areas are not only unfavourable soil conditions, such as a deficiency of nutrients and toxicity of heavy metals, but also seed availability. Because there is a small resident bank of seeds, birch and willow planting has been conducted in order to contribute to the rapid development of a tree layer.

4.6 Conclusions

Growing awareness of air pollution effects on forests has, from the early 1980s on, led to intensive forest damage research and monitoring. Results of forest damage research and monitoring have fostered air pollution control, especially in Europe and North America, and to a smaller extent also in other parts of the world. In these areas, air pollution control has already succeeded in improving air quality and decreasing atmospheric deposition. Clean air policies under CLRTAP in Europe have reduced S emissions within the last three decades by 80% and N emissions by 30%. The frequency of O_3 peak concentrations has decreased. At several sites there are first indications of a recovery of forest soil and tree conditions that may be attributed to improved air quality.

The fact that exaggerated predictions of large-scale forest dieback did not prove true, and that forest condition shows first – even if modest – indications of improvement, caused a decrease in the attention paid by politicians and the public to air pollution effects on forests. But, despite undeniable success of clean air policies, air pollution continues to affect the structure and functioning of forest ecosystems in many regions of the world. Even after three decades of air pollution control under CLRTAP in Europe, critical loads of acidity and of nitrogen, in particular, are exceeded on the majority of forest sites. Constituting not only a most phytotoxic air pollutant but also a greenhouse gas, O_3 shows even rising air concentrations at the global scale. While for these reasons air pollution control will have to be continued in Europe and North America, efforts in clean air politics have to be fostered, or even initiated, in other parts of the world. In Russia and Asia, air pollution has long been recognised as a problem for forest health. Systematic monitoring of air pollution effects on forests, as well air pollution control, have been launched. In Latin America and Africa, however, there are fears that air pollution must also affect forest condition – at least in the vicinity of big cities and in certain regions of growing industrialisation. But systematic monitoring and research on air pollution effects on forests aimed at a conversion of scientific findings into clean air policies are only little developed in those parts of the world.

While at the political level, attention to climate change is focussed on questions of CO_2 emission and carbon sequestration, relationships between air pollution and climate change are hardly recognised. These relationships should be increasingly emphasised and explored by scientists.

At present, there are well-established relationships between the exceedance of critical level/loads of pollutants and ecosystem degradation. Nevertheless, ecological interactions between critical loads and other environmental factors, such as the impacts of increased concentrations of CO_2 and O_3, insects, pathogens, fire, drought, flooding, wind, and extreme temperatures, as well as ecosystem management practices, are still poorly understood (McNulty et al. submitted). There is a need to develop dynamic models for calculating critical loads, i.e., addressing changing levels of deposition and their effects, to define suitable indicators and damage thresholds, and to evaluate the combination of critical loads with other indicators of environmental stress.

A new approach based on stomatal O_3 flux is under consideration for establishing O_3 uptake-based critical levels for the protection of vegetation (Paoletti and Manning 2007). Such an approach should be tested on a wide range of species of conservation importance. Most observations of O_3 effects on trees have been made on northern temperate species (Wittig et al. 2009). More and longer duration open-air studies on mature trees in different forest types are critical if we are going to understand future changes to forest productivity.

Current process models are parameterised with data collected under steady-state conditions, yet environmental conditions in natural forest systems are highly dynamic. Focused research and model development incorporating stomatal responses and C balance under dynamic environmental conditions (e.g., rapid changes in light and medium term changes in humidity) will improve predictive capabilities of models.

New long-term forest monitoring approaches must be developed in order to translate the atmospheric changes in climate and pollution into biological effects on forests (Fischer 2008). Large-scale monitoring data should be available to the whole scientific community.

Future research should focus on the interacting impacts on forest trees and ecosystems. The integrative effects of air pollution and climatic change, in particular elevated O_3, altered nutrient, temperature, water availability, and elevated CO_2, will be key issues for impact research. An important improvement in our understanding might be obtained by the combination of long-term multidisciplinary experiments with ecosystem-level monitoring, and the integration of the results with ecosystem modelling within a multiple-constraint framework.

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