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# Integrated effects of air pollution and climate change on forests: A northern hemisphere perspective

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*Simultaneous addressing air pollution and climate change effects on forests is an opportunity for capturing synergies in future research and monitoring.*

**Abstract**

Many air pollutants and greenhouse gases have common sources, contribute to radiative balance, interact in the atmosphere, and affect ecosystems. The impacts on forest ecosystems have been traditionally treated separately for air pollution and climate change. However, the combined effects may significantly differ from a sum of separate effects. We review the links between air pollution and climate change and their interactive effects on northern hemisphere forests. A simultaneous addressing of the air pollution and climate change effects on forests may result in more effective research, management and monitoring as well as better integration of local, national and global environmental policies.


*Keywords:* Acidification; Climate change; Eutrophication; Forest monitoring; Tropospheric ozone

## 1. Introduction

Climate change and air pollution are closely linked, although in applied scientific research and even more in political negotiations they have been largely separated. Many of the traditional air pollutants (APs) and greenhouse gases (GHGs) have not only common sources, but may also interact physically and chemically in the atmosphere causing a variety of environmental impacts on the local, regional and global scales. The impacts on forest ecosystems have been traditionally treated separately for air pollution and climate change. However, the combined effects of numerous climate change and air pollution factors may significantly differ from a sum of separate effects due to an array of various synergistic or antagonistic interactions. The net effect varies for different ecosystem types and geographic regions, and depends on magnitude of climate or AP drivers, and types of interactions between them (Bazzaz and Sombroek, 1996).

In this review, we will discuss links between air pollution and climate change and their interactive effects on northern hemisphere forests from a perspective of the key ecological, economic and societal values of forests. Our aim is to stimulate a simultaneous addressing of the air pollution and climate change effects on forests. This is an opportunity for capturing synergies and avoiding overlaps between two lines of traditional research (Swart et al., 2004). This could result in more effective research, monitoring and management as well as better integration of local, national and global environmental policies.

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## 2. Links between air pollution and climate change in the atmosphere

Many traditional APs and GHGs have common sources and interact in the atmosphere. The most important GHG, carbon dioxide ($CO_2$), is mainly produced by burning fossil fuels, which is also an important source of several APs. Elevated $CO_2$ itself can be considered a pollutant, depending on the terminology.

Some APs contribute to radiative forcing (Fig. 1). $CO_2$ is a main driver in that regard, and is followed by methane ($CH_4$), halocarbons, and nitrous oxide ($N_2O$). Aerosols and particulate matter (PM) affect climate depending on their composition. Soot enhances warming, while other aerosol constituents (like the APs S and N species) generally have a cooling effect (Houghton et al., 2001). Climate change, especially high radiation and temperature, promote increases in tropospheric ozone ($O_3$), the secondary pollutant generated from the GHGs non-methane volatile organic compounds (VOCs), carbon monoxide (CO) and nitrogen oxides ($NO_x$). $O_3$ is a potent GHG itself and indirectly influences lifetimes of other GHGs such as $CH_4$ (Fiore et al., 2002). Methane has not been considered an $O_3$ precursor, because of its long lifetime, until intercontinental transport was recognized to contribute to $O_3$ pollution (Derwent et al., 2004).

Acidic deposition and eutrophication affect natural emissions of the GHGs $CH_4$ and $N_2O$ from soils (Brink et al., 2001). For the oxidized N species, climate change increases the amount of nitric acid ($HNO_3$) (Sanderson et al., 2006). For reduced N species, both climate change and AP increases act to convert more ammonia ($NH_3$) to ammonium sulphate. This change occurs via increased aqueous phase oxidation of $SO_2$, indicating that the S and reduced N cycles are closely linked (Sanderson et al., 2006).

Climate change may affect distribution patterns and mixture of APs. Such changes are caused by changing wind patterns, and amount and intensity of precipitation. The intensity of precipitation determines the atmospheric concentration and deposition of acidifying compounds. This may also change frequency and extent of pollution episodes (e.g., $O_3$). However, AP emission is more important than the effect of climate change on the dispersion and chemical transformation of APs (Mayerhofer et al., 2002), while regional air pollution (in the form of sulfate aerosols) was not found to have a large effect on climate change in Europe (Alcamo et al., 2002).

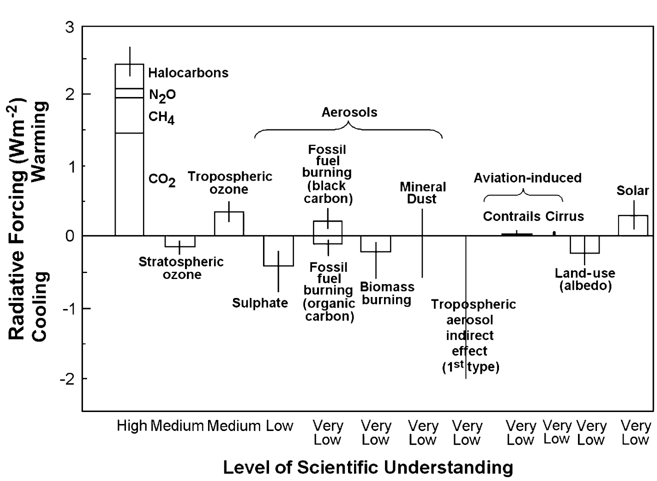


Fig. 1. The global mean radiative forcing of the climate system for the year 2000, relative to 1750 (Houghton et al., 2001). The rectangular bars represent estimates of the contributions of these forcings, some of which yield warming, and some cooling.

## 3. Integrated effects of air pollution and climate change on forests

Climate change and air pollution interact in affecting forests by changes in soil processes, tree growth, species composition and distribution, increased plant susceptibility to stressors, increased fuel built-up and fire danger, water resources, recreation value, etc. Climate change can alter the effects of APs on ecosystems, and vice versa, APs can modify responses of ecosystems to specific climatic change impacts.

### *3.1. Soil processes*

Climate change (especially temperature) alters many soil processes, having consequences for the entire ecosystem. Higher temperatures, changed precipitation patterns and modified net primary production (NPP) increase the weathering rate, resulting in higher critical loads (i.e., lower sensitivity of ecosystems to APs) (Posch, 2002). The increased mineralization increases N availability and leaching (Mol-Dijkstra and Kros, 2001). Deposition of various APs also increases N availability. Climate change acts to worsen the problem of acidification by increasing the production—and deposition to soils—of $HNO_3$ from NO, and the proportion of $NH_3$ converted into ammonium sulfate, which in turn may result in further acidification of soils (Sanderson et al., 2006). Higher $CO_2$ levels can increase soil moisture due to changes in evaporation (Eguchi et al., 2005). Natural ability of forest soils to take up $CH_4$ decreases due to N deposition (http://www.bae.uky.edu/IFAFS/FAQS.htm).

### *3.2. Tree health*

APs may affect forests (Table 1). $SO_2$ is probably the best known example (Legge et al., 1999). $NO_x$ and $NH_3$ as well as $HNO_3$ vapor may have direct phytotoxic effects but only at high concentrations (Bytnerowicz et al., 1999). Gaseous N pollutants and the dissolved in water N compounds contribute to N deposition that can have various effects on forests (Fenn et al., 1998). $O_3$ has the highest phytotoxic potential and is predicted that by 2100 half of the World's forest will be exposed to phytotoxic $O_3$ levels (Fowler et al., 1999). However, high levels of APs (mainly $O_3$) do not necessarily translate into substantial negative effects on forests (e.g., Paoletti, 2006). In contrast, an increase in forest growth has been demonstrated for several European countries (Spiecker et al., 1996). The presumed reasons are increased N deposition,

Table 1
Effects of primary air pollutants on forests can be seen as a multi-pollutant, multi-effect problem extended towards radiative forcing (EEA, 2004b).

| | $SO_2$ | $NO_x$ | $NH_3$ | VOC | CO | PM | $CH_4$ | $CO_2$ + GHGs |
|---|---|---|---|---|---|---|---|---|
| Ecosystems | | | | | | | | |
| Acidification | × | × | × | | | | | |
| Eutrophication | | × | × | | | | | |
| Surface ozone | | × | | × | × | | × | |
| Radiative forcing | | | | | | | | |
| Direct | | × | | | | | × | × |
| Via aerosols | × | × | × | × | | × | | |
| Via OH | | × | | × | × | | × | |

temperature, and availability of $CO_2$ as well as changes in silvicultural practices. The inherent aspecificity of tree response indicators of AP effects is another likely reason of the observed discrepancies between high pollution and low effects on forests (e.g., Ferretti et al., 2007).

Climate change parameters that trigger stomata opening (e.g., increasing temperature) increase the sensitivity of plants to APs like $SO_2$ and $O_3$. Parameters that lead to stomata closure (e.g., water stress, increased $CO_2$) help to protect the plant from APs. A complication is that $O_3$ slows the stomatal response to reduced water availability (Paoletti, 2005). Climate change parameters that lead to a longer growing season (e.g., warming) increase the exposure of plants to APs like $SO_2$ and $O_3$, whereas parameters that shorten the growing season (e.g., water stress) reduce the exposure and damage (Guardans, 2002). A simulation for Norway spruce and European beech showed that climate change sensitivity increases in boreal areas and decreases in temperate areas due to temperature and water stress (Guardans, 2002). The AP impacts were less severe in 2000 than in 1990, but the cumulative stress was still significant.

### *3.3. Growth of trees and carbon sequestration*

Climate warming, increased mineralization in soils, and higher N availability increase plant growth and thus the C sink especially if N is the limiting nutrient (Bazzaz and Sombroek, 1996). Increased NPP has been hypothesized to be due to increases in N deposition (Nadelhoffer et al., 1999), atmospheric $CO_2$ concentrations (Friedlingstein et al., 1995), temperature, and longer growing season (Myneni et al., 1997). The contribution of N deposition to the increase in C in standing biomass is approximately 10 to 20 Mton $y^{-1}$, i.e., 3.5–7% of the estimated forest growth (de Vries et al., 2003). By far the largest amount of C stored in northern hemisphere forests is in the soil (de Vries et al., 2003). Increased N deposition causes an increased rate of soil organic matter accumulation due to an increased leaf/needle biomass and litter production, and a reduced decomposition of organic matter (Schulze, 2000). An increase in $CO_2$ concentrations favors both growth and water use efficiency of plants. However, trees may adapt and such an effect may diminish soon (Tognetti et al., 2000). Using a modeling approach, temperature has been claimed to be relatively unimportant, whereas the combination of elevated $CO_2$ and N may account for a 15–20% increase in forest NPP (Rehfuess et al., 1999). In this context, N deposition is claimed to be the most important factor of the increased forest growth. The remaining explanation would be the impact of forest management.

The AP effects on forests may provide an important control on the C cycle that has not yet been widely considered. Prolonged exposure to $O_3$ may suppress gains in C sequestration of trembling aspen as it has been shown in a free-air $CO_2 + O_3$ experiment (Karnosky et al., 2003). Including $O_3$ in a forest ecosystem model applied to the north-eastern United States, offsets a substantial portion of the growth increases caused by $CO_2$ and N deposition (Ollinger et al., 2002). Felzer et al. (2004) have incorporated empirical equations derived for trees (hardwoods and pines) and crops into the Terrestrial Ecosystem Model to explore the effects of $O_3$ on NPP and C sequestration across the conterminous US. The results show that C sequestration since the 1950s has been reduced by 18–38 Tg C $y^{-1}$ with the presence of $O_3$.

More frequent extreme drought events may counteract the effects of the anticipated mean warming and lengthening of the growing season, and erode the health and productivity of ecosystems, reversing sinks to sources (see Section 3.5).

### *3.4. Biodiversity changes*

Species composition is mainly driven by climate, soil and forest type, but N and S deposition has also a significant impact (de Vries et al., 2003). Some crops and trees need low temperatures in winter to trigger bud bursting in spring. These species can no longer grow in areas where winter temperatures are becoming too high. An overload with N leads to eutrophication and changed ecosystem composition in Europe (Bakkenes et al., 2002). Since critical loads of AP are species-specific, the critical loads in a region may change if ecosystem composition changes, meaning that the sensitivity to AP is also altered. The changes in composition may also change the sensitivity of an ecosystem to climate change. Changes in biodiversity have been observed under historic climate change (Prentice et al., 1998) and are also projected for the future (Bakkenes et al., 2002). Assessment of climate change impacts on terrestrial ecosystems in Japan suggests that the average NPP increase could change the vegetation type (Shimizu et al., 2005). In North America, modeling efforts suggest substantial changes in potential habitats of several species and communities (Bachelet et al., 2001; Iverson and Prasad, 2001). The forest area in the USA is projected to decrease by an average of 11%, with the lost forest replaced by savanna and arid woodland biome. Community types predicted to increase include oak/hickory and oak/pine in the East and ponderosa pine and arid hardwoods in the West. Seven of the 80 eastern species are predicted to be reduced in suitable habitat by at least 90%. Potential habitats for several subalpine conifers are simulated to contract substantially. The potential habitat for big sagebrush (*Artemisia tridentata*) will shift from the US into Canada (Hansen and Dale, 2001).

### 3.5. Susceptibility to natural disturbances

Natural disturbances having the greatest effects on forests include insects, disease, drought, fires, or wind storms. Chronic excess input of N to forest ecosystems causes nutrient imbalances (i.e., deficiencies of the macro-nutrients K, P, Mg and Ca) which, in turn, increase the sensitivity of plants to climatic factors, such as frost or drought, and susceptibility to parasite attacks (UNECE, 2005). Ozone also decreases winter hardiness increasing the risk of frost damage (Bazzaz and Sombroek, 1996).

The seasonality of fire hazards in the US is projected to increase about 10% over the next century, or even about 30% in Alaska and south-eastern US (NSST, 2000). Examples of catastrophic fires are those in southern California in fall 2003. Years of fire suppression have lead to severe overstocking of the mixed-conifer mountain forests. Several years of severe drought as well as chronic exposure to elevated $O_3$ concentrations and N deposition have contributed to fuel buildup at the forest floor. Additionally, drought and AP weakened the trees promoting bark beetle infestation (Miller and McBride, 1999) and massive tree dieback. Hot weather in the 2003 summer and fall, strong winds, and fires started by arsonists resulted in catastrophic fires (Keeley et al., 2004). Potential for such events is high in semi-arid Mediterranean type ecosystems. The catastrophic fires in Spain and Portugal in summer 2005 are additional examples of that increasingly important problem.

Due to the decreased proportion between roots and above ground portions caused by N deposition, susceptibility of trees to windthrow can increase (Bytnerowicz, 2002). Also, elevated $O_3$ levels reduce the supply of carbohydrates to roots reducing their biomass (Bytnerowicz and Grulke, 1992). In Switzerland, after the 1999 windstorm, 1600 trees were analyzed (UNECE, 2005). Broken trees showed wider annual growth rings—and thus decreased mechanical resistance—in the last decade compared to the unbroken trees. The presumed reasons are increased N deposition, temperature, and availability of $CO_2$. Higher N concentrations were measured in the wood of broken trees that most likely reflects higher N supply. Over the past decades storm events and damage severity to forest have increased over Europe (UNECE, 2005). In Slovakia, a storm in November 2004 destroyed 24,000 ha of forest stands (1.2% of the total forest area) mostly in the Tatra National Park (UNECE, 2005). The southern part of the Park has been characterized by elevated levels of $O_3$, S and N deposition (Bytnerowicz et al., 2004). The destroyed stands are also mostly monocultures of mature Norway spruce trees. Lower resistance of monocultures to windbreaks results in large areas of Norway spruce stands felled by wind storms in Central Europe (Spiecker, 2000).

In the mountains of the western US and Canada, unprecedented outbreaks of bark beetles killing pine forests at a large scale have occurred. Recent increase of temperatures, especially at high elevations, allow for shortening the life cycle of bark beetles and shifting from typical one-year cycle to two-year cycle. In New Mexico and Arizona, bark beetles killed about 70 million pines in 2002 and 2003. These forests, similarly as the southern California forest described above, had experienced severe drought—the year 2002 was the driest year in the last 1400 years. Overstocking of trees caused by almost 100 years of fire suppression led to tripled or even quadrupled stand densities and severe limitations of available water to individual trees (Nijhuis, 2004). Although $O_3$ was not mentioned, one should ask if elevated $O_3$ levels have not played a role in that pine dieback.

The positive effects of rising temperature on plant growth may be offset by an increased risk of water shortage. Europe experienced a particularly extreme climate anomaly during 2003, with July temperatures even 6 °C above long-term means, and annual precipitation deficits 50% below the average. Forest sites experienced a significant reduction of GPP (gross primary productivity) and net C uptake, with some forests even temporarily becoming net C sources in August (Ciais et al., 2005). GPP did not entirely recover from the summer stress during the remainder of the year. Several Mediterranean sites showed a smaller decrease in C uptake, largely dominated by less respiration. Pronounced soil water deficits compensated for the effect of warmer temperatures in affecting soil respiration (Ciais et al., 2005). Although 2003 was not the driest year on record, the impact of drought was amplified by high summer temperatures and soil water deficits carried over from the previous spring. The GPP anomalies correlated better with rainfall than with summer air temperature, indicating the dominant role of water limitations (Ciais et al., 2005). Despite climatic conditions that favor drought and heat also increase $O_3$ concentrations, the ambient $O_3$ was not mentioned in the above article.

## 4. Perspectives and recommendations

International policy-making on climate change still takes place independently from air pollution. Opportunities for synergies and avoiding trade-offs have neither been integrated into international air quality policies in Europe, such as the sixth environmental action program and the Convention on Long-range Transboundary Air Pollution, nor are they addressed in current climate negotiations under the United Nations Framework Convention on Climate Change. At the technical assessment level, however, the opportunities are increasingly noted. For example, the Intergovernmental Panel on Climate Change addresses multiple pollutants and sources of GHGs (Metz et al., 2001), and the UNECE Task Force on Integrated Assessment Modelling integrates climate change considerations into regional air pollution policies as a logical next step. The complex atmospheric interactions between air pollution and climate change, however, are presently not understood sufficiently well to allow their quantitative incorporation in integrated assessment modeling (EEA, 2004b). For example, more precise determination of the chemical composition of aerosols is necessary in order to assess the overall net warming or cooling effect, as this is the largest source of uncertainty in the radiative balance (Houghton et al., 2001).

Assuming a reduction of $CO_2$ emissions to comply with the Kyoto protocol, there will be significant ancillary benefits in terms of additional reduced emissions of APs and reduced costs of AP abatement (Metz et al., 2001). The indirect effects of climate policies in Europe targeting the stabilization of GHG concentrations in the atmosphere would cut the costs of abating $SO_2$ emissions by 50–70% and $NO_x$ by about 50% (Van Harmelen et al., 2002). For the shorter-term implementation of the Kyoto Protocol, savings of 10% of the costs of controlling acidification and ground-level $O_3$ were found (Syri et al., 2001). Put another way, there are strong financial arguments for developing joint policies to reduce AP and GHG emissions (Alcamo et al., 2002). Outside Europe, the potential synergies between GHG mitigation and abatement of local air pollution have received attention especially in developing countries (Chile, Brazil, Mexico; Cifuentes et al., 2001) and in the US, where NGOs such as the World Resources Institute (MacKenzie et al., 1992) and Resources for the Future (Burtraw and Toman, 2000) have promoted an integrated approach, primarily on the basis of the advantages for human health. In the US, harmonized options for joint abatement of APs and GHGs are being promoted at the State level (STAPPA/ALAPCO, 2000) and are recommended by the National Research Council recent report (NRC, 2004).

Dynamic industrial growth without an effective air pollution control and rapid increase of a number of combustion engine vehicles in Asia, especially in China, will be the main drivers for increasing background $O_3$ concentrations in the northern hemisphere in coming years (Klimont et al., 2001). There is a need for improved understanding of $O_3$ distribution patterns and trends in forests. Ground-based monitoring networks, remote sensing, integration of data generated at different scales (forest stand, landscape, region, continent, hemisphere) through various modeling efforts should be implemented. The ongoing monitoring of incidences of $O_3$ damage to forests may provide evidence of changes in effects, provided that the better indicators are selected. Ozone visible injury is regarded as a result of oxidative stress, leading to a cascade of adverse effects resulting in a reduced vitality of forest species and increasing predisposition to the climatic, edaphic and biotic factors (de Vries et al., 2003), but the ultimate link between $O_3$ visible injury and plant functionality and growth is still to be ascertained in a wide range of species. Understanding responses of key forest species to ambient $O_3$ is essential for developing secondary $O_3$ biologically based models. Until today, $O_3$ responses of only a few tree species in North America and Europe are known and very little is known about the responses of Asian forests. Therefore a development of improved $O_3$ risk assessment models is recommended (as an improvement from simple exposure indices such as AOT40 and promotion of new indices such as those based on stomatal $O_3$ flux and plant defense mechanisms). The flux-based approach provides a more biologically realistic representation of the $O_3$ exposure of plants than indices based on concentration only (UN, 2004). Work is needed to validate flux-based indices for more species and to verify the applicability of the flux-approach to large-scale surveys under the routine condition of monitoring programs (Ferretti et al., 2007). Eddy correlation may provide more robust results and fluxes of $O_3$ since inert gases, such as $CO_2$, can be easily monitored. Nevertheless, ozone flux from dry soils should be carefully evaluated (Omasa et al., 2000). In addition, a compliance with possible flux-based air quality standards might be difficult to be assessed and monitored (Ferretti et al., 2007). Increased knowledge on ecosystem-specific $O_3$ sensitivity may help to set up different critical levels across Europe, analogously to the EMEP effects-based critical loads, instead of the present flat-approach critical levels.

Climate change acts to worsen the problem of acidification by increasing the production of $HNO_3$ and ammonium sulfate (Sanderson et al., 2006). Implementation of existing technology to reduce emissions of precursor gases will help alleviate the problem. Our knowledge of the long-term effects of increased N deposition on a representative range of terrestrial ecosystems should be improved. While wet N deposition is well characterized, contribution of dry N deposition to forest ecosystems, especially in arid and semi-arid zones, should be better understood. There is also a need to better understand effects of atmospheric N deposition on tree growth as well as the interactions between N deposition, forest stand, ecosystem stability, and biodiversity. Model development is still needed regarding several key processes, particularly N dynamics and relations to climate change. Understanding the N cycle in semi-natural ecosystems may be the key to understand the long-term source or sink strength of soils for C (de Vries et al., 2003). Since N often is the limiting nutrient in forests, N deposition may increase wood production and accumulation of soil organic matter, thus increasing C sequestration into the forest. Earlier estimates suggested that this mechanism could take up one third of the global $CO_2$ emission from fossil fuel if most of the N deposition was taken up by trees and used to form new woody biomass (Holland et al., 1997). Recent data, however, suggest that the increase in N deposition may cause a 10-times smaller additional $CO_2$ sequestration in forests (Nadelhoffer et al., 1999). When the large uptake is mainly due to elevated growth, it is likely that this is a transitory phenomenon, whereas it could be a C sink for a long period if soil accumulation is the main cause since below ground C has much lower turnover times than above ground C. The EUROFLUX project provided measurements of C fluxes above a range of forests across Europe (Tenhunen et al., 1998), but extrapolation of these results to a continental scale is still prone to large uncertainties (de Vries et al., 2003). The aboveground $CO_2$ sequestration in trees can also be estimated from yield tables and models on tree growth or can be based on repeated forest surveys. Recent EU projects have increased the understanding of controls in the N and C cycle in forests (e.g. Dise et al., 1998; Gundersen et al., 1998). A comprehensive evaluation of the effect of eutrophication on C sequestration in the entire ecosystem needs to trace C and N jointly through the ecosystem and needs to include the response of the aboveground biomass, the input of C to the soil (aboveground and belowground), the aquatic phase and the response of soil microorganisms (emission of $CO_2$ and $NO_x$; formation of labile soil organic matter).

In addition, more research, such as that conducted by Karnosky et al. (2003) and Marinari et al. 2007, is needed for understanding how ambient $O_3$ or N deposition may modify responses of trees to elevated $CO_2$. The effects of $O_3$ on NPP and C sequestration should be factored into future calculations of C budget (Felzer et al., 2004).

Achieving stabilization of atmospheric GHG concentrations would require substantial (ca. 70%) reductions in global GHG emissions (Houghton et al., 2001), i.e., much larger global emission reductions than agreed in the Kyoto Protocol. Even if industrialized countries substantially reduce their GHG emissions over the next few decades, the climate system is expected to continue changing over the coming centuries (EEA, 2004a). This is due to the long time delay before emission reduction policies have an effect on GHG concentrations and, in turn, on climate. Therefore, in addition to reducing emissions, understanding mechanisms of adaptation of forests and other ecosystems to climate change is increasingly needed.

To quantify the long-term consequences of extreme climate conditions on forest productivity, we need to understand better the consequences of xylem embolism; the effects of reduced carbohydrate pool sizes on subsequent leaf and fine root production and turnover, and on the ability of plants to resist pathogen attacks; the impacts on soil microbial dynamics, decomposition and nutrient-supply processes, shifting competitive abilities between plant species (Ciais et al., 2005), and the relative contribution of concurrent pollution. In this regard, a much better knowledge on the confounding factors that affect chemical and biological recovery of ecosystems is needed.

Better understanding of the effect of air pollution and climate change interactive effects on catastrophic fires is needed, as well as improved understanding of mechanisms leading to catastrophic windthrows. This should lead to development of management strategies to improve the resistance of forest stands to these stressors.

The data collected at intensive APs monitoring sites may be useful also in relation to other environmental problems than air pollution, such as climate change and changes in biodiversity (Manual for Integrated Monitoring, 1998). Data collected at ICP Integrated Monitoring sites have already been used in global change research framework for calculations of, e.g., C and N pools and fluxes in Finnish forests (Ilvesniemi et al., 2002). Clearly, there is a great need for better utilization of the existing monitoring and modeling efforts. This could be done through the improved cooperation between various ICP activities in the EU, coordination of various monitoring programs in the US (CASTNET, NADP, IMPROVE, FHM, FIA, LTER, etc.), and more effective cooperation between US, Canada, EU and Asia in developing the coordinated methodologies for evaluation of critical loads for N, S and acidity.

Experiments at different scales (branch, seedling, mature tree) are needed to better understand mechanisms of the effects of single factors or simple interactions. In this regard long-term, large-scale experiments such as the free-air $CO_2$ and/or $O_3$ and N enrichment studies or gradient air pollution studies (e.g., FACE study in Wisconsin and Italy, Kranzberg Forest in Germany, San Bernardino Mountains in southern California) should be encouraged and funded.

Collaboration of the forestry research scientists within the IUFRO working parties, research groups, scientific divisions and across them is needed and may greatly help in exchange of information, technology transfer or starting new multidisciplinary and multinational cooperative projects.

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