

Air pollution and forest health: toward new monitoring concepts

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“Capsule”: *There is a need for reappraisal of monitoring methods used to assess air pollution effects on forest health.*

Abstract

It is estimated that 49% of forests (17 million km²) will be exposed to damaging concentrations of tropospheric O₃ by 2100. Global forest area at risk from S deposition may reach 5.9 million km² by 2050, despite SO₂ emission reductions of 48% in North America and 25% in Europe. Although SO₂ levels have decreased, emissions of NO_x are little changed, or have increased slightly. In some regions, the molar SO₄/NO₃ ratio in precipitation has switched from 2/1 to near 1/1 during the past two decades. Coincidentally, pattern shifts in precipitation and temperature are evident. A number of reports suggest that forests are being affected by air pollution. Yet, the extent to which such effects occur is uncertain, despite the efforts dedicated to monitoring forests. Routine monitoring programmes provide a huge amount of data. Yet in many cases, these data do not fit the conceptual and statistical requirements for detecting status and trends of forest health, nor for cause–effect research. There is a clear need for a re-thinking of monitoring strategies.

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1. Introduction

Air pollutants affecting forest health at national and multinational scales include increasing tropospheric, or surface level, ozone (O₃) concentrations, increasing atmospheric carbon dioxide (CO₂) concentrations, and acidic precipitation. At a regional to local scale, emissions of sulphur dioxide (SO₂), oxides of nitrogen (NO_x) and a number of other pollutants such as ammonia (NH₃) emissions from animal feeding operations affect forests downwind of point or urban sources, particularly in rapidly industrializing regions of the world. Increasing levels of ultraviolet-B radiation from stratospheric O₃ depletion at a global scale are possibly a threat to forest health (Percy and Gordon, 1998). There is evidence as well for an increasing emission of persistent organic pollutants (POPs) (EMEP, 2000). Coincidentally, the world's climate is changing and anthropogenic influences are strongly implicated (Houghton et al., 2001).

Air pollution and climate change are two key factors comprising the global change threat to forest health and sustainability. Considerable scientific effort in northern hemisphere countries has been devoted to the enhancement of our understanding of forest responses to global change at the process, organ, system, stand and ecosystem levels (see reviews by McLaughlin and Percy 1999; Innes and Oleksyn, 2000; Percy et al., 2000; Innes and Haron, 2001). Here we draw upon recent reports and retrospective analyses in order to provide a short summary of trends in air pollution and forest health as it is routinely assessed.

2. Air pollutant trends

2.1. Ozone

Surface level O₃ is a secondary air pollutant formed in the atmosphere under bright sunlight from the oxidation of the primary pollutants NO_x and volatile organic compounds (VOC). Ozone is the most prominent of secondary pollutants formed and is well understood

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(Jenkin and Clemitshaw, 2000). Ozone is the most pervasive of all air pollutants affecting forest health. Levels were about 10–15 ppb a century ago, compared with 30–40 ppb measured as background around the world today (Finlayson-Pitts and Pitts, 2000). There are three trends apparent in global O₃: (1) an increase in extent of O₃ and forest area at risk; (2) a decrease in maximum 1-h O₃ concentrations in northern hemisphere countries; and (3) an increase in background O₃ concentrations over much of the world (Percy et al., in press). The latter trend is particularly important from the policy perspective as background O₃ levels may approach concentrations used to calculate some critical levels (Lefohn et al., 2001).

In the United States during the past 20 years (1982–2002), national ambient O₃ levels decreased 18% based on 1-h data and 11% based on 8-h data (US EPA, 2002). For the period 1981–2001, the downward trend (>10%) in 1-h maximum O₃ levels occurred in every geographic area. Nearly all regions experienced improvement in 8-h O₃ levels between 1981 and 2001 except the north central region, which showed little change. However, between 1992 and 2002, the average 8-h O₃ level in 33 national parks actually increased nearly 4%!

In Canada, national ambient O₃ levels (as fourth highest daily maximum 8 h O₃) showed a decrease from 70 ppb in 1991 to 62 ppb in 1993 (Dann, 2001). Since 1993, the trend, except for 1996 and 2000, has shown increasing levels up to 70 ppb in 2001. Parallel trends have been observed for locations in southern Ontario. Levels decreased from 88 ppb in 1991 to 74 ppb in 1993, followed by an increase to 88 ppb again in 1999 and 87 ppb in 2001.

Monitoring in Europe shows that episodes of O₃ occur over the continent every summer. There are no clear trends in alpine regions with episodes in northern and central regions occurring downwind of industrialized areas. In 2000, surface level O₃ concentrations were monitored at 124 ground locations in 26 countries (Hjellbrekke and Solberg, 2002). The 1-h critical level (75 ppb) for O₃ was exceeded at 95 (77%) of the 124 locations, especially in central Europe. Highest hourly average concentrations (126, 128 ppb) were reported for two Italian locations. Hourly concentrations >100 ppb O₃ were also recorded at locations in Austria, Denmark, Germany, Poland, Slovakia, Sweden and Switzerland during a large-scale episode around June 20. In theory, O₃ concentrations in many regions of Europe are sufficient to adversely affect tree growth (Matyssek and Innes, 1999).

The underlying processes responsible for the formation and deposition of O₃ are reasonably well understood. Despite some remaining uncertainties, the rate of O₃ deposition onto a given forest area can be calculated once ambient concentrations are known and inform-

ation on key environmental variables such as wind velocity, air temperature, solar radiation and degree of leaf surface wetness is available (Percy et al., in press). Indeed, recent progress in scientific understanding has led to the development of O₃ budgets at the country level (Coyle et al., 2003). Yet, true risk assessment remains problematic due to scientific uncertainty around the magnitude of O₃ flux into the plant. Indeed, standards and critical levels currently used by regulators remain based upon some index of O₃ exposure alone (Percy et al., in press).

2.2. Acidic precipitation

Acidic precipitation arises from the oxidation of sulphur dioxide and nitrogen dioxide (NO₂) in the atmosphere to form H₂SO₄ and HNO₃ acids then deposited onto forests via rain, fog, cloud or by dry deposition of gases and vapour. Organic acids play an important role (Finlayson-Pitts and Pitts, 2000) and deposition may occur close to the point or area source as well as over long distances of 1000 km or more. Emissions of SO₂ have declined in North America (25%) and Europe (48%) since the early 1980s (Fowler et al., 1999). However, emissions of SO₂ in rapidly developing economies of south and east Asia, as well as Africa, South and Central America have been increasing quickly, to the point where southeast Asia now emits more S into the atmosphere than either Europe or North America. In CEE (Central and Eastern Europe) and ECE (Economic Commission for Europe) countries, SO₂ emissions have been reduced significantly between 1980 and 1998. Many countries (Austria, Czech Republic, Germany, Slovakia) have decreased their SO₂ emissions during that period by over 75%. Other countries such as Bulgaria and Romania have reduced less, by 39 and 14% respectively. As a whole the average reduction was 62.6% compared with 40% in Canada and 21% in the US (EMEP, 2000).

While the contribution of SO₂ to acidification of precipitation has lessened, NO_x emissions have not been reduced by the same level. For the countries reporting 1980–1998 data, the average reduction in NO₂ emissions was 38%. Ukraine (60%), Czech Republic (56%) and Germany (47%) reported the largest decreases. Reductions of 2% in the US were reported, although NO₂ emissions in Canada increased by 4% during the same period (EMEP, 2000).

2.3. Ammonia

Ammonia (NH₃) is the most important reduced form of anthropogenic nitrogen emitted, with France, the Russian Federation, Germany, and the UK being the largest NH₃ emitters in 1998. Emission of NH₃ in 1998 in the USA totalled 4,477 thousand tonnes (EMEP,

2000). Emission reductions (1990–1998) in the ECE ranged from 70% in Ukraine to 10% in Austria. Average degree of decrease was 39% (EMEP, 2000).

2.4. Heavy metals

A trend of declining heavy metal emissions has been reported for ECE countries and the US. Significant reductions (1990–1998) in lead (Pb) emissions ranged from 37% in Czech Republic to 82% in Austria and Hungary (EMEP, 2000). With one exception (Ukraine +15%), mercury (Hg) emissions also decreased within the ECE. Reductions were greatest in Bulgaria (65%) and Austria (40%) (EMEP, 2000).

2.5. Organic air pollutants

Emissions of persistent organic pollutants (POPs) are several orders of magnitude lower than most inorganic pollutants. However they are transported over long distances. Polycyclic aromatic hydrocarbons (PAHs) are now detected in air over most of the industrialized world, particularly in urban air sheds. Of note are the reactions of certain PAHs in VOC-NO_x atmospheres to form biologically active compounds with major toxicological implications (Finlayson-Pitts and Pitts, 2000).

2.6. The special case of carbon dioxide

Atmospheric CO₂ concentrations have risen from about 280 ppm in the pre-industrial period to >365 ppm today. Doubling to 560 ppm is anticipated between 2050 and 2100. Forests have considerable potential to reduce the anthropogenically driven increase in CO₂, being generally carbon limited at current atmospheric concentrations (Drake et al., 1997). The case of CO₂ is, therefore, a special one in that it is essential for plant growth, yet remains a serious pollutant of the upper troposphere and the largest contributor to radiative forcing (Rawmaswamy et al., 2001).

New evidence makes it clear that O₃ has the capacity to reduce productivity gains in fast-growing species growing under enriched CO₂ atmospheres (Karnosky et al., 1999; Isebrands et al., 2001; Percy et al., 2002; Karnosky et al., 2003). For a detailed discussion of sinks potential, the reader is referred to recent reviews on forest response to enhanced CO₂ contained in Karnosky et al. (2001).

3. Defining forest health and condition

Unfortunately, the terms forest health and forest condition are frequently used interchangeably. According to Percy (2002), forest health has been used to denote the degree to which insects and diseases were

disrupting normal tree processes. On the other hand, forest condition has been used in relation to the descriptive indicators used in routine forest assessments. However, adopting a “forest health” or a “forest condition” viewpoint should not be done lightly as selection of one or the other will ultimately drive the operational steps of any monitoring programme (Ferretti, 1997). For example, if one considers forest health to be defined only by the crown condition of trees, one will proceed to assess spatial and temporal variation of defoliation and foliar symptoms. On the other hand, if the ecosystem as a whole is considered, many different components and indicators will be taken into account (e.g., soil nutrients, soil biota, ecosystem productivity) (Innes and Karnosky, 2001). It is obvious that the two visions of forest status will produce different research concepts and monitoring systems yielding differing levels of insight.

Regrettably, there is no universally accepted definition of forest health. However, in the context of forest response to air pollution, the sustaining of ecosystem function is especially important because process and pattern-oriented considerations have been shown to underpin any definition of forest health. McLaughlin and Percy (1999) have accordingly defined forest ecosystem health as “...the capacity to supply and allocate water, nutrients and energy in ways that increase or maintain productivity while maintaining resistance to biotic and abiotic stresses.” This definition fits quite well within new forest health concepts built around issues such as long-term sustainability, resilience, maintenance of structure and functions and multiple benefits and products (Kolb et al., 1994).

The most complete review of international forest health and assessment was compiled by Innes (1993). The reader is referred there for a detailed presentation of cases of forest decline throughout the world (see also Ciesla and Donaubaer, 1994) and of international forest health assessment in the early 1990s. However, to build upon Innes (1993) we must recognize that forests respond to air pollution in a dynamic and complex manner. Accordingly, Percy (2002) has listed many of the attributes of a forest that contribute to its health (Table 1). Ecosystem processes that contribute include net primary productivity, biogeochemical cycles, water flux, organic matter cycling, insect cycles and disease incidence. These processes of course are often the product of past and present influences/characteristics of ecosystem structure including stand structure, species life history, genetic diversity, soil quality, site history and management practices. Historical/cultural differences in a number of these attributes are one reason why etiologies of European and North American forest declines attributed to air pollution in the 1980s cannot be directly compared (Percy et al., 1999).

Table 1
Internal and external factors that contribute to forest health (after Percy, 2002)

Ecosystem processes	Ecosystem structure	Growth environment
Net primary productivity	Stand structure	Solar radiation
Biogeochemical cycles	Species' life history	Temperature
Water cycles	Genetic diversity	Precipitation
Organic matter cycles	Soil quality	Growing season length
Insect population cycles	Site edaphics	Air pollution
Disease incidence and severity	Site history	Extreme events
	Management practices	

4. Trends in air pollution and forest health

4.1. North American case studies

Earlier, McLaughlin and Percy (1999) presented a retrospective analysis of the four most-prominent North American air pollution–forest case studies. They concluded that O₃ and/or nitrogen (N) deposition (single or co-exposure) induced changes in depth and vigour of root systems, shifts in pool sizes and allocation patterns of carbon, and changes in supply rates of nitrogen and calcium represent important shifts in ecological function currently occurring in diverse forest types across a large geographic area in North America. They predicted that the influence of these process level changes of future health of North American forests could be greatly increased due to climate change. Chappelka and Samuelson (1998) reported that O₃ induces foliar injury on a number of tree species throughout much of the eastern US. However, as symptom expression is influenced by a complex of factors, it has proven difficult to confirm cause–effect in routine monitoring.

Growth declines in unmanaged southeastern pines were reported in the early 1980s by Sheffield and Cost (1987). Process research conducted through the Southern Commercial Forest Research Cooperative has been summarized in Fox and Mickler (1996). They have compiled a summary of forest characteristics, biotic and abiotic stresses along with the results from a large number of integrated field and experimental research projects. Studies at ambient levels early on provided evidence that moderate O₃ levels can increase water stress and reduce growth in larger trees. Seasonal growth patterns of mature loblolly pine (*Pinus taeda* L.) trees were evaluated by McLaughlin and Downing (1996) over 5 years to determine the effect of O₃ on large trees. Although levels of O₃, rainfall and temperature varied widely over the period, regression analysis was able to identify significant influences of O₃ on stem growth patterns. Ozone exposures interacted with soil moisture and high air temperatures to reduce short-term rates of stem expansion (McLaughlin and Downing, 1995). Observed O₃ responses were rapid, occurring within 1–3 days of exposure to O₃ at 40 ppb. These data indicate that low

O₃ levels can reduce growth of mature forest trees and that O₃–climate interactions are likely important modifiers of southeastern US forest growth into the future.

Since the mid 1950s, much of the mixed conifer forest in southern California has been exposed to some of the highest concentrations of O₃ and highest N deposition in North America. Lately, there is new evidence for the more widespread occurrence and effects of these pollutants in the Sierra Nevada of the southwestern US, and the reader is referred to this volume (see Bytnerowicz et al., 2003). Long-term exposure to high levels of O₃ and oxidants in the San Bernardino Forest has produced the classic example of hierarchical forest response to O₃ (see Miller and McBride, 1999). Effects from foliar level to successional stage have been documented. Seasonal mean 24-h concentrations 50–60 ppb O₃, along with peak 1-h averages frequently exceeding 200 ppb, were sufficient to cause foliar injury, early needle loss, decreased nutrient availability, reduced carbohydrate production, lower vigour, decreased height/diameter growth and increased susceptibility to bark beetles. The concurrence of drought, long-term reduction in precipitation and high O₃ (Arbaugh et al., 1999) have contributed to a period of growth decline for ponderosa (*Pinus ponderosa* Laws) and Jeffrey (*Pinus jeffreyii* Grev. and Balf.) pines. Decreases in radial basal area growth rates during 1950–1975 were 25–45%, (ponderosa and Jeffrey pines) and 28% (bigcone Douglas fir). Other factors such as high levels of nitric acid (HNO₃) vapour (Bytnerowicz et al., 1999) and stand development changes due to fires suppression have also possibly contributed. Older trees are most vulnerable and additional stress imposed by high O₃ may make these trees more vulnerable to bark beetle attack. Unfortunately, only scattered dominant trees remain in much of the lower elevation bigcone Douglas fir stands (Arbaugh et al., 1999).

It is interesting that under diminishing annual average O₃ concentrations, Miller et al. (1989) reported an improvement (1974–1988) in the foliar injury index. This recovery is not expected to prevail indefinitely due to changing precipitation patterns, continuously high O₃ levels and N deposition as well as increased area affected by air pollution in this mountain range. Due to urban sprawl extension, high O₃ levels are now

monitored in the eastern San Bernardino, which in the 1970s was characterized by low, near background O₃ levels (Alonso et al., 2002).

4.1.1. Experimental stand-level pollutant fumigation: Aspen FACE

The Aspen Free Air Carbon Dioxide Enrichment (FACE) experiment in Rhinelander, Wisconsin, USA is the first open-air facility to examine the responses of forest trees to interacting CO₂ and O₃ (see Dickson et al., 2000). The multidisciplinary, multinational project examines how elevated CO₂ and O₃ affect carbon/nitrogen cycles and ecological interactions of forests. The impacts of these co-occurring greenhouse gases are being studied in terms of carbon sequestration, physiological processes, growth and productivity, competitive interactions and stand dynamics, interactions with pests and ecosystem processes, such as foliar decomposition, mineral weathering, and nutrient cycling. Average O₃ concentrations (12-h daytime mean during the growing season) in fumigation rings during the first 4 years were 46–55 ppb.

Results have been consistent across functional groups, from leaf level biochemistry, gene expression and gas exchange through to ecosystem level, and across trophic levels, in that elevated CO₂ (560 ppm) and O₃ exert opposite effects (Karnosky et al., 2003). When the two gases co-occur, low levels of ambient O₃ offset or moderate the responses attributable to elevated CO₂. Percy et al. (2002) reported that in the case of the most widely distributed North American tree species, trembling aspen (*Populus tremuloides* Michx.), CO₂ and O₃ singly or in combination affected productivity, physical and chemical leaf defenses, with stimulating effects from O₃ on performance of key defoliators like the forest tent caterpillar (Figs. 1 and 2). Because of these changes in plant quality, insect and disease populations were altered. What Aspen FACE is demonstrating is that “bottom up” and “top down” effects leading to increased (CO₂) or decreased (O₃) plant growth are possible. It is noteworthy that such effects cascading through the ecosystem would not have been documented by studying individual processes or trophic levels in isolation and, therefore, changes in pest dynamics could not have been linked with changes in plant quality (Percy et al., 2002).

4.2. Forest condition in Europe

4.2.1. The European monitoring system

With the establishment of the United Nations Economic Commission for Europe (UN/ECE) International Cooperative Programme on Assessment and Monitoring of Air Pollutant Effects on Forests (ICP Forests) and a similar programme within the European Union (the European Union Scheme on the Protection of Forests Against Atmospheric Pollution), forest con-

dition has been monitored using harmonized methods and criteria since 1986/1987 (UN/ECE and EC, 2000).

The two programmes, which merged in 1991, are structured on different monitoring intensities. Level I monitoring has been carried out on a large number (5942 in 2001) of plots selected (UN/ECE, 1998) according to a systematic sampling design (16×16 km), but with a low intensity of observations on a fixed number of trees per plot (Lorenz et al., 2002). Crown condition (defoliation and discoloration) has been assessed annually since 1987 in all plots. In most recent years, the presence-absence of identifiable damage such as biotic or climatic stress was assessed as well, although not on all plots. A portion of these Level I plots has also been sampled for soil condition and foliar element concentrations.

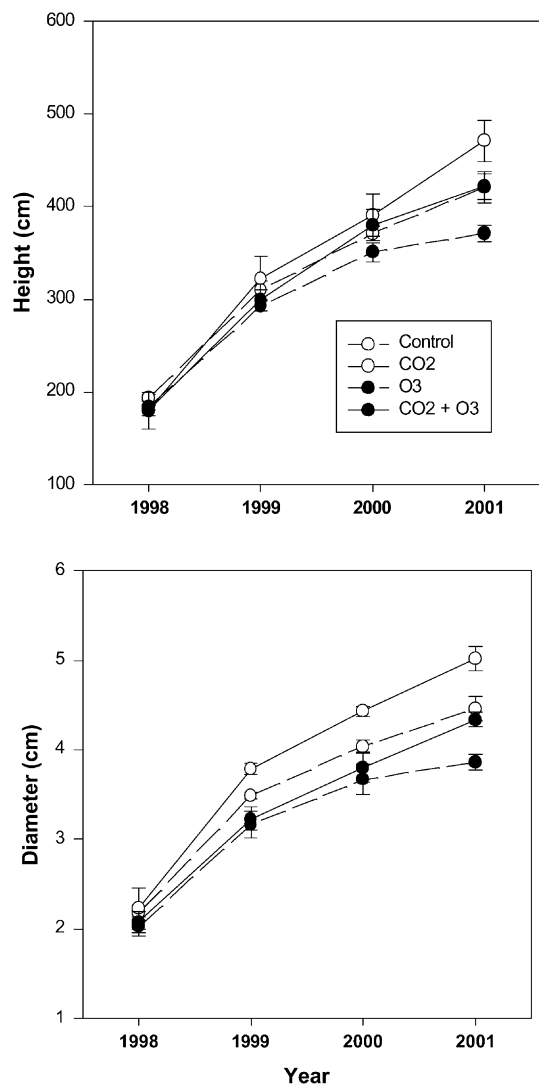


Fig. 1. The effects of elevated CO₂ and O₃ on growth of trembling aspen. Data are means ± 1 SE averaged over five clones covering a range of sensitivities to O₃. Note the significant decline in height and diameter due to O₃ over time and the ability of O₃ to offset gains due to elevated CO₂. (Reproduced from Percy et al., 2002.)

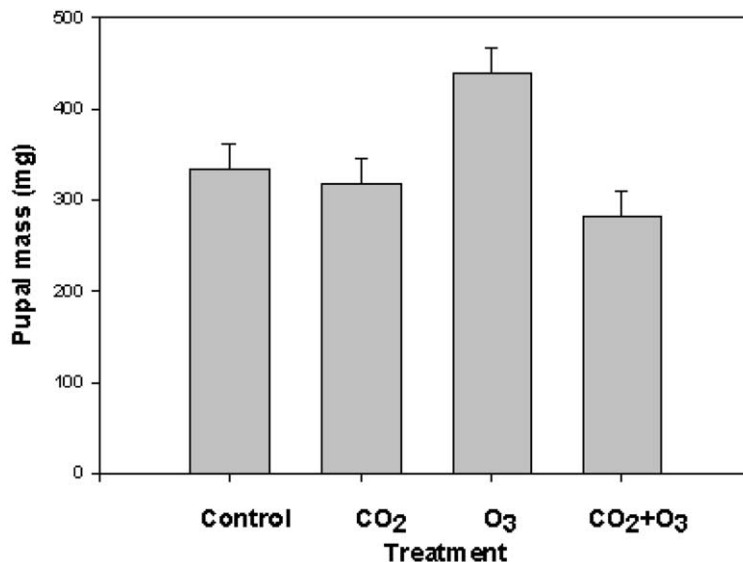


Fig. 2. The effects of elevated CO₂ and O₃ on forest tent caterpillar performance. Note the significant increase in pupal mass due to changing plant quality due to O₃. (Reproduced from Percy et al., 2002.)

Level II monitoring consists of more intensive investigations on a reduced, less randomized number of sites. It was started in 1994 and now has more than 860 plots (EC-UN/ECE, 2002). Level II features monitoring of both stressor and response indicators. Crown condition (on the basis of Level I assessment), soil and foliar chemistry, tree growth and ground vegetation are assessed on most plots. Open field and throughfall deposition, atmospheric chemistry, ozone-like foliar symptoms, soil solution chemistry, tree phenology and meteorological parameters are assessed on a part of the plots.

4.2.2. European forest condition

In its most recently published assessment of European forest condition, 22.4% of 132,350 Level I trees assessed in 2001 were classified as having defoliation >25% (Lorenz et al., 2002). Discoloration was infrequent (7.6% of the observed trees), with attribution of damaging agent completed for 43–55% of trees. Insects were the most frequently observed damaging agent (9.9%); effects due to “known air pollution” were scored on 2.4% of trees, all located in non-European Union (EU) countries.

A *t*-test was subsequently carried out to investigate differences between 2000 and 2001. Compared with the year 2000, 2001 defoliation increased significantly in 13% of the plots and significantly decreased in 8.1% (Lorenz et al., 2002). In 78.9% of the plots, defoliation remained unchanged. With few exceptions (e.g., a cluster of plots showing significant decrease in south Spain), there was no pattern to changes between 2000 and 2001.

In the UN/ECE report on 10 years of monitoring (1986–1996), a deterioration in forest condition was

identified for many of the most frequent tree species, despite distinct regional differences at the European scale (Muller-Edzards et al., 1997). However, when comparing defoliation between 1989 and 2001 (Fig. 3), no obvious trend was detected for Scots pine (*Pinus sylvestris* L.), Norway spruce [*Picea abies* (L.) Karst.], European beech (*Fagus sylvatica* L.) or sessile and pedunculate oaks (*Quercus petraea* L. and *Q. robur* L.) when considering the detection limit (5%) of the scoring system adopted (Fig. 3). Data for maritime pine (*Pinus pinaster* Ait.) and evergreen oaks (*Quercus ilex* L., *Q. rotundifolia* Lam.) displayed a distinct increase. A more formal comparison was attempted with the data series 1994–2001, for which plot-wise estimates of trends were made (Lorenz et al., 2002). The results were reported in terms of proportion of plots for which both significant ($P < 0.05$) slope of the linear regression of defoliation vs. year of observation and changes exceeding the 5% threshold were recorded. In most plots, defoliation remains unchanged; when a change occurred, the direction of change differed between species (Table 2).

4.2.3. Forest condition in the Austrian Alps and Carpathian Mountains

Forest condition and related aspects of forest health in the Carpathian Mountain region, ranging mostly from Slovakia to Romania and across the western-most part of Ukraine, have recently been summarized at an international workshop (see Szaro et al., 2002). Tree condition data are reported for countries within the Alpine system through the EU and UN/ECE (Lorenz et al., 2002). A range of tree conditions was evident, with trees in Austria apparently in much better condition than elsewhere. Air pollution is considered the most

important anthropogenic factor affecting central and eastern European forests (EC/PHARE, 2000). However, caution is required when comparing between countries. Although species assemblage is per se a factor, differences in assessment methods have been suspected for some time (Innes et al., 1993) and these have been recently confirmed (Cozzi et al., 2002).

Species-specific maps published in the EU and UN/ECE provide additional information (Lorenz et al.,

2002). Data were reported for Norway spruce (Austria, Slovakia, Romania), Scots pine (Slovakia), beech and sessile and European oak (Slovakia and Romania). In most species, trends were not significant. In the case of Norway spruce, instances of significant increases in defoliation were reported for southern Austria and the southern/western Carpathians. For beech, a significant increase in defoliation occurred only at plots in the latter region. On the other hand, significant decreases in

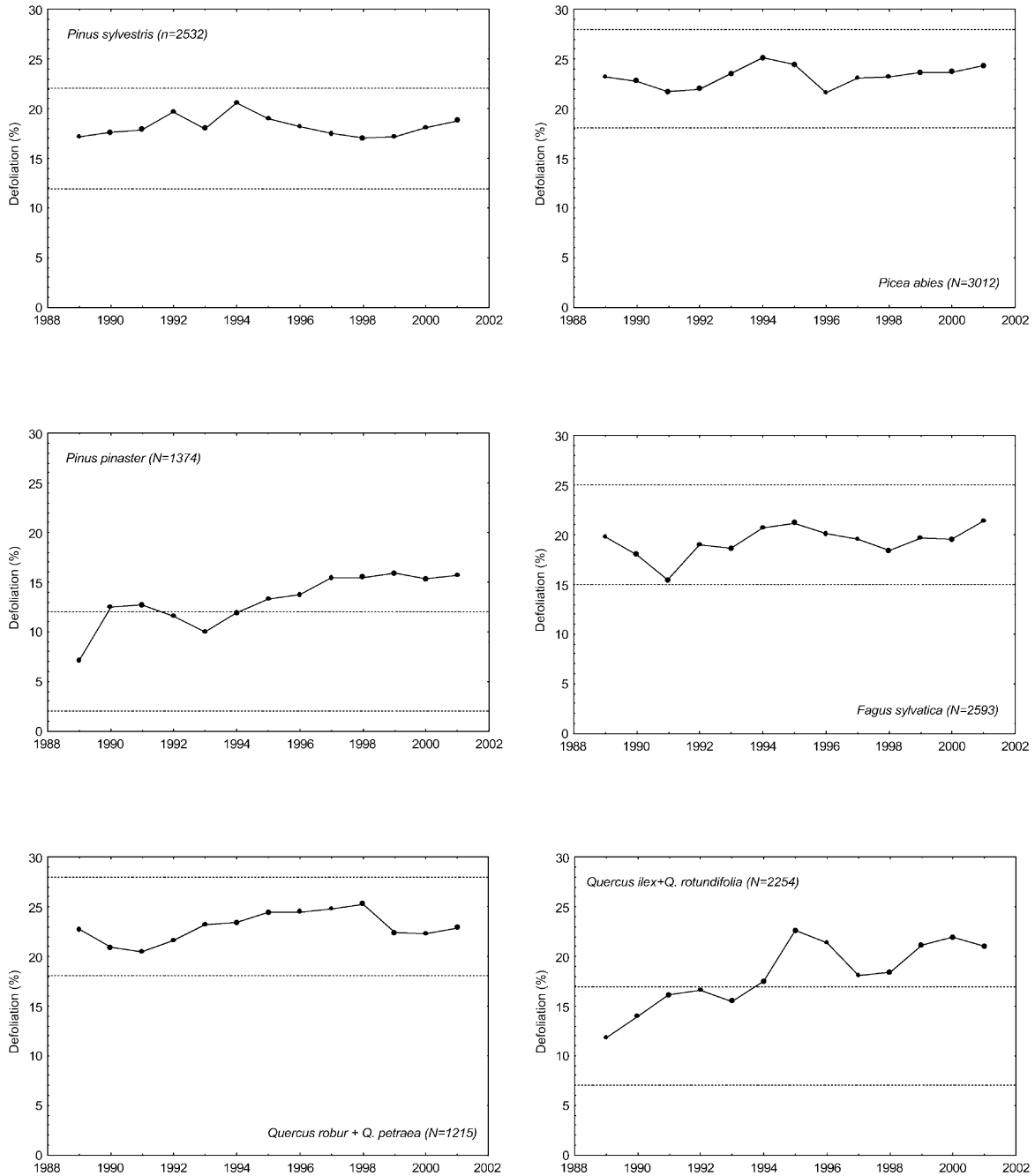


Fig. 3. 1989–2001 Trend of mean defoliation for six forest species in Europe. The dashed lines indicate the $\pm 5\%$ interval corresponding to the detection limit of the assessment method in the first survey year. Data source: Lorenz et al. (2002).

Table 2

Frequency (%) of plots of the EC and UN/ECE Level I network with different trends in defoliation (1994–2001). Plots are separated according to the species. Plots in France, Italy and Sweden are not considered because of changes in assessment or sampling methods. See text for further explanations (data source: Lorenz et al., 2002)

Species	Significant decrease	Unchanged	Significant increase
<i>Pinus sylvestris</i>	15.0	77.1	7.0
<i>Picea abies</i>	6.4	78.4	15.2
<i>Fagus sylvatica</i>	10.1	73.6	16.3
<i>Quercus robur</i> and <i>Q. petraea</i>	11.4	76.6	12
<i>Quercus ilex</i> and <i>Q. rotundifolia</i>	6.7	80.4	12.9

defoliation were also reported for Romania and Slovakia. Little change was reported for Scots pine and the oaks.

4.2.4. Interpretation of tree condition data

Difficulties have been encountered in interpreting the results of the European forest condition survey due to a number of confounding factors (Muller-Edzards et al., 1997). Recently, analysis of Level I data was carried out by Lorenz et al. (2002) for beech and Scots pine. Using multiple linear regression and deposition data obtained from the Co-operative Programme for Monitoring and Evaluation of the Long-Range Transmission of Air Pollutants in Europe (EMEP) model and precipitation data from the Global Precipitation Climatology Centre (GPCC), they attempted to explain spatial and temporal variation over the period 1994–1999. In Scots pine, the various models explained 6–44% of the variance for the temporal trends and about 61% for spatial variation. Sulphur deposition was found to be a significant predictor for both spatial and temporal changes, while insect, age and country have been reported to be statistically significant predictors only for the spatial variation. Improvements of defoliation for Scots pine in Poland were related to decreased S deposition. On the other hand, the model adopted to explain time trends in beech was able to explain 37–38% of the variance, but none of the predictors (precipitation April–September, insect, fungi, deposition of S, NH₃, NO_x and year) was found to be statistically significant. Where the spatial pattern was considered, the model adopted was able to explain 38–40% of the variance and precipitation, fungi, country (used as a proxy of method's differences between countries) and age (but not deposition of S, NH₃, NO_x) were determined significant predictors.

De Vries et al. (2000) also conducted another multiple regression analysis of Level II data. They considered 1997 defoliation data from four groups of species in 262 plots for which site, soil, meteorology and deposition data were available: pine (mostly Scots pine), spruce (mostly Norway spruce), oaks (mostly sessile and European oak) and beech (mostly European beech). Plots used in the analysis were clustered in central and northern Europe (mostly in Germany, Austria, Sweden)

(EC-UN/ECE, 2000). Varying with the group of species being considered, stand characteristics, age, meteorology, throughfall of N and S and, to a lesser extent, foliar nutrients explained 21–48% of the variance in defoliation. At most Level II plots, deposition of N now exceeds that of S, and 55% of sampled plots had N inputs greater than 14 kg ha⁻¹ year⁻¹ (UN/ECE and EC, 2000). The influence of individual factors like N was stated to depend on tree species, site type and geographic region, confounded by data quality at the plot level (Ferretti et al., 1999). Interestingly, no relationship was found between defoliation and measured sulphur deposition in pines, contradicting the results of the analysis carried out with Level I using modelled deposition data.

5. Issues related to monitoring networks

One of the outstanding questions, then, is to what extent current monitoring programmes can actually provide scientifically defensible data on status and trends in European forests. There is no question that the EU and UN/ECE pan-European forest monitoring system is an unprecedented effort aimed at describing changes in forest condition. Yet, the programme may be constrained by limitations rooted in its design. These include issues of indicator development (especially response indicators), sampling design, data quality and data processing (Ferretti, 1998; Ferretti and Chiarucci, 2003).

5.1. Indicators

At the beginning of the monitoring programme, air pollution effects on forests were mostly understood in terms of tree defoliation. This assumption was based on the reports of heavy defoliation occurring at sites for which severe air pollution effects were reported. Since then, defoliation was adopted as the main response indicator within the EU and UN/ECE monitoring programme. However, defoliation fails to meet the requirements for an effective indicator (Ferretti, 1997).

Developing an effective indicator requires attention to several key constraints (Hunsaker, 1993). Firstly, clear

objectives and endpoints must be elaborated. Secondly, a conceptual model identifying the linkage between the issue examined and responses expected is needed. In this respect, defoliation could in fact be considered as an indicator of overall tree condition. However, it should not be considered a specific indicator of air pollution effects on trees nor used as a diagnostic tool to detect these effects.

5.2. Network design

Given the systematic nature of the network, Level I is intended to provide a representative sample of the European forests. It is based on a two-stage sampling design; (1) site selection; and (2), sample tree selection within the site. However, the variation in national definition of a forest (Köhl et al., 2000) may have resulted in different attributes being ascribed to the target population. In addition, it is unclear whether the origin of the 16×16 km² grid was randomized, normally a prerequisite for the application of both parametric and non-parametric statistics.

For this reason, the procedure adopted (selection of a fixed number of trees on pre-defined and permanent subplots located in four cardinal directions) can be considered as a standard protocol that does not incorporate any element of randomness (Elzinga et al., 2001). As population is undefined, neither unbiased estimates of population parameters (e.g., mean defoliation) nor the statistical testing of hypotheses (e.g., changes in defoliation across sites or years assessed by a *t*-test) are possible at the site level. In Level II, site selection occurred on a preferential basis. This makes it difficult to make inferences at the European scale. Within sites, problems may occur with allocation of measurements that may make it difficult to obtain site-related unbiased estimates of the indicator(s) (Ferretti and Chiarucci, 2003).

5.3. Quality and use of defoliation data

Data quality has been an issue of concern in crown condition assessment for some time (Innes et al., 1993). The spatial and temporal comparability of data have been questioned and there is evidence that there are several sources of uncertainty in crown condition data generated by the EU and UN/ECE programme (Cozzi et al., 2002). For example, data reported in Fig. 1 and Table 3 do not include France, Italy or Sweden, as methodological changes occurred in these countries (Lorenz et al., 2002). Although change is sometimes formally reported, it is also likely that more subtle changes (e.g. turnover of the field crews, subtle deviations from reference standards) may have occurred. Defoliation assessment in Europe is based on the work of hundreds of field crews (Cozzi et al., 2002). In most

Table 3

Retrospective analysis of degree of success in documenting the role of air pollution as an important factor in forest health (after Percy, 2002)

Successful when:	Unsuccessful when:
Network-level monitoring is succeeded by process-oriented research across spatial and temporal scales of stressors	Systematic monitoring is disconnected from process-oriented research
Appropriate indicators of ecosystem function are investigated at an intensity/ extent appropriate to stressors and ecosystem	System protocols are not developed for single agents Systematic monitoring is not stratified on stressor gradients
Investigations in essential processes/cycles are integrated with investigations in ecosystem resilience through pests, genetics, succession	Endpoints measured are inappropriate Hierarchical nature of forest response is not recognized
There is continuity of investigation	Dominant role in predisposition is not recognized

cases, crews are trained at the national level, but consistency has yet to be proven. The EU and UN/ECE are currently addressing this issue.

Defoliation data are collected in categories of 5% (UN/ECE, 1998) and, therefore, can be analyzed as a continuous measurement only assuming an even distribution of observations within each 5% category. Secondly, defoliation data are aggregated with a positively skewed distribution (Lorenz et al., 2002). Without transformation, a parametric analysis like the *t*-test cannot be used. No transformation of the original positively skewed data has been reported by Lorenz et al. (2002). Thirdly, a plot was attributed to a given species when at least three individuals of that species were found on the plot. The extent to which three trees can provide an indication of whole plot status is questionable.

6. A new approach to forest health monitoring

6.1. General guidelines for successful monitoring programmes

One of the inherent features of national and multi-national monitoring schemes is the averaging of tree condition across regions. Earlier, McLaughlin and Percy (1999) had proposed two guidelines to better design effects-oriented monitoring: (1) monitoring and understanding the relative roles of natural and anthropogenic

stress in influencing forest health will require programs that are structured to evaluate responses at appropriate frequencies across gradients of forest resources that sustain them; (2) such programs must be accompanied by supplemental process-oriented investigations that more thoroughly test cause-and-effect relationships among stresses and responses of both forests and the biogeochemical processes that sustain them.

With those two guidelines as a basis, Percy (2002) subsequently conducted a follow-up analysis of when monitoring and research had in fact demonstrated air pollution to be an important factor in forested areas where air quality was an issue. From that analysis, it is clear that the link was made (Table 3) when several conditions were met: (1) when scales of stressors have been considered and monitoring has been succeeded by process-oriented research; (2) when appropriate indicators and endpoints were measured; (3) when investigations on physical/chemical cycles were coupled with biological cycles; (4) when there was continuity in investigation.

In contrast, the linkage was not made (Table 3) when: (1) systematic monitoring at larger scales was disconnected from process-oriented research; (2) systematic monitoring was not stratified along stressor gradients; (3) system protocols were developed around a single class of damaging agents; (4) endpoints measured (defoliation) were insensitive or inappropriate to the stressors investigated; and (5) the hierarchical nature of forest health response was ignored when protocols were designed.

6.2. Recommendations for improved design of monitoring systems

It is clear from the above analysis that future monitoring designs must consider ecosystem function and stressor–indicator relationships within the framework of an appropriate statistical design. This is only way in which questions posed by policy makers, which are by definition generic (e.g., “What is the effect of air pollution on forests?”) can be translated into an operational assessment endpoint. The latter is by definition more stringent (e.g., “What is the proportion of forest area affected by defoliation due to ozone?”). Monitoring programmes must be designed to provide information at a stated probability (e.g., $P < 0.05$) level about the status and degree of change in the indicator being monitored as well as about the relationships between these changes and the stressor(s) of concern. This goal can be achieved only if the monitoring is properly designed. In the following sub-sections we discuss: (1) the need for a straightforward and conceptual approach to a structured monitoring system; (2) the technical requirements essential to the success of a monitoring programme.

6.2.1. Structure of the monitoring system

The need for a conceptual approach linking large-scale monitoring to process-oriented studies has been highlighted (e.g., McLaughlin and Percy, 1999; Bricker and Ruggiero, 1998). Yet, intensive study sites are often selected on a preferential basis, and, in statistical terms, their connection with the extensive network is weak (Ferretti and Chiarucci, 2003). This creates problems in extrapolation of results although deterministic models created around empirical relationships between attributes may help somewhat (Overton et al., 1993). As a consequence, we recommend that monitoring systems based on sites having different levels of investigation should be conceptually linked, ensuring formal connections between the sites (at every intensity level of monitoring) and the target population on which inferences are requested (see below).

6.2.2. The technical steps required

There are clearly some key steps that the designers of monitoring programmes should follow, and these have been discussed in part elsewhere (Legendre and Legendre, 1998; Stohlgren et al., 1995; Ferretti and Ehrardt, 2002; Vos et al., 2001; Elzinga et al., 2001). The four steps required are to: (1) understand the nature of the study needed; (2) clearly express the objectives; (3) define a sampling design tailored to address the objectives; (4) implement Quality Assurance (QA) procedures. These steps are discussed in more detail below.

(1) *Understand the nature of the study needed.* In the absence of acute exposure leading rapidly to tree mortality [air pollutants are most often present at sub-lethal levels (Eberhardt and Thomas, 1991)], monitoring of air pollution effects on the forest can be regarded essentially as a mensurational effort (Hurlbert, 1984). There are several categories (analytical, descriptive, observational) of field studies, which “...all depend on sampling, and may be characterized by the way samples are distributed (allocated) over prospective sampling units in the (target) population as a whole” (Eberhardt and Thomas, 1991). A decision on which study approach to adopt should be made after taking into account the nature of the output desired. For example, analytical sampling allows for comparison “...among different subgroups of the population, in order to discover whether differences exist among them, and to form or verify hypotheses about the reasons for these differences” (Cochran, 1977). Inferences from sampling over the entire population of interest are possible. On the other hand, descriptive sampling is designed “to obtain certain information about large groups,” for instance, the proportion of defoliated trees in a given area (Cochran, 1977). With descriptive sampling, efficient estimation of means and totals is possible, but differences among subgroups within a population remain unexplored. Observational studies aim to compare the effects of a

given stressor on groups of individuals subjected to different levels of the stressor, and are usually conducted on limited portions of the domain of interest. In this way they are similar to experiments, although an actual treatment is not feasible. On the other hand, sampling for pattern has much to do with geostatistics, and it is mainly concerned with description of spatial patterns and map production.

(2) *Avoid ambiguity—define clear objectives.* The definition of clear objectives involves the explicit identification of the precise assessment question being posed. Definition also involves the related assessment and measurement endpoints, the target population and the geographical coverage of the investigation (Hunsaker, 1993; Eberhardt and Thomas, 1991). The assessment endpoint is an operational expression of the policy-relevant question of interest, and the measurement endpoint is a measurable ecological attribute related to the chosen assessment endpoint. An unambiguous definition of the objective will surely provide input to the definition of the indicators. When defining the assessment endpoint it is essential to establish at least the following attributes: a time frame for detecting change; the extent of change considered acceptable; and the probability level at which a change should be “accepted”.

Olsen et al. (1999) have defined the target population as “...the aggregate of units whose characteristics define the desired scope of inference...” or, in other words, the totality of situations to which the conclusion can be applied. Recognition of the target population is important as it drives the identification of the sampled population. The relationship between target and sampled population is important in order to assess the validity of the inferences and the geographical coverage (e.g., the area to be considered by the investigation).

(3) *Define a sampling design tailored to address the objectives.* A sampling strategy is essential if one is to reach conclusions with a known level of confidence. In general, monitoring programmes are asked to provide data to accomplish three tasks: (a) estimate population parameters (e.g., mean, totals, proportion for a given attribute: mean S concentration in the foliage, total annual N deposition, proportion of defoliated trees) at a given geographical scale; (b) test differences between two (or more) sites and/or sampling periods; (c) identify statistical relationships between stressors and responses. It is worth noting that elements of randomness in sampling are essential for each of the above objectives. In the first two cases, it is a requirement for the application of both parametric and non-parametric statistics. In the third case, randomness is essential because it protects against biased estimates being used as dependant/independent modelling variables. Probabilistic sampling can only be carried out if there is an explicit definition of the target population (see above).

(4) *Implement proper QA procedures.* The relevance of each of the above-listed three technical steps becomes obvious as one thinks in terms of data quality. The adoption of QA/QC principles in the early stages of programme design through preparation a QA plan (Shampine, 1993) forces managers to identify and evaluate the majority of monitoring programme factors. The main benefits derived from of a QA programme are consistency, reliability and cost-effectiveness over time. For example, long-term monitoring programmes may last for decades, and the time factor can have a strong impact on the implementation of the work by the personnel involved. In addition, the continuous assessment of data quality permits mathematical assessment of uncertainty, which can result in a more appropriate presentation and use of the data (Ferretti, 1997). The reader is referred to Cline and Burkman (1989) and to Innes (1993) for definition and examples of QA procedures.

7. Air pollution and future forest health

The most pervasive air pollutant now and into the future affecting forests is O₃. Fowler et al. (1999) calculated the global forested area at risk to concentrations > 60 ppb. In 1950, O₃ exceedance was largely restricted to the temperate latitudes, some 9.2% of the temperate and subpolar forest (Fig. 4). By 1990, almost 25% of the world's forest was exposed with a significant increase in the tropical and subtropical forest. In 2100, fully 49.8% of world forests (17.0 million km²) will be exposed to damaging O₃ concentrations including 11 million km² (Fig. 4) in temperate and subpolar regions (Fowler et al., 1999).

Fowler et al. (1999) also estimated the area of global forests at risk from acidification (>2 keq H⁺ ha⁻¹ year⁻¹ as S). They predicted a 624% increase in area of global forest at risk between 1985 (0.28 million km²) and 2050 (5.9 million km²), with the majority of the increase in sub-tropical and tropical forest regions.

Unless there is a strong downturn in global population growth and industrialization, forests will continue to be exposed to a deteriorating atmospheric environment. Area of forest at risk from O₃, S and acidification is expanding under current economic and social trends. Modelling of future S and N scenarios for North America and Europe indicates that although the driver of acidification is changing (molar ratios in rain are now almost equal for N and S), acidification potential in many areas remains high. It is unclear at present whether climate change will lessen or enhance air pollution effects on forest health (McLaughlin and Percy, 1999). However, there is good evidence that O₃ has the ability to offset or negate expected increases in growth due to increasing concentrations of atmospheric CO₂

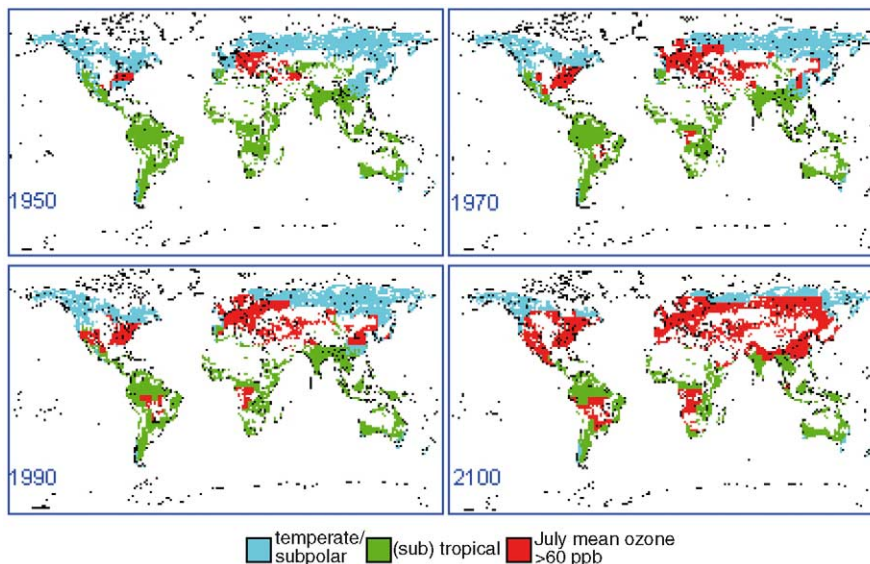


Fig. 4. Global areas of forest cover where July peak surface ozone concentration exceeds 60 ppb. (Reproduced from Fowler et al., 1999.)

(Karnosky et al., 2003). Of note also, is the increasingly important (in a warming climate) reported interaction of two major greenhouse gases (O_3 , CO_2) on insect populations and pathogens (Percy et al., 2002). For European forest growth, increasing CO_2 and climate change are expected to become more important over time as nitrogen effects diminish (European Forest Institute, 2002).

Approaches commonly used to assess forest health are in most cases likely inadequate either for quantitative evaluation of status and trends and for the detection of future change and elucidation of the roles of natural and anthropogenic stressors. Integrated approaches linking into a consistent vision, such as process-oriented empirical studies with pattern-oriented monitoring along defined pollution gradients using clonal plantations (Karnosky et al., 1999), genetically screened tree pairs (Mueller-Starck et al., 2000), ecological analogues (Krupa and Legge, 1998), ecosystem-based research on essential cycles (FACE) with better characterization of physical and chemical environment yielding new approaches toward a statistically and conceptually sound monitoring system, will be required if the interactive effects of global change (air pollution + climate change) on forest health and sustainability are to be understood in the 21st century. In this respect, long-term ecological research is essential to provide the basis for understanding status and trends of components, and structure and processes within forest ecosystems. If past experience is at all instructive, the dominant role of air pollutants in predisposition must form a central point around which new developments coalesce. The interactive effects of pollutants and other factors must also be understood.

In summary, we believe that scientifically defensible forest health monitoring can be achieved only if it is properly designed using a conceptual approach founded upon the four technical steps essential to its success. Within the context of the policy question(s) posed it is critical to: (1) understand the nature of the study needed; (2) clearly express the objectives; (3) define a sampling design tailored to address the objectives; and (4) implement Quality Assurance (QA) procedures.

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