CHAPTER 2

AIR POLLUTION IMPACTS IN NORTH AMERICA

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1. Introduction

There is a long record of air pollution damage to forests and crops in North America. Regional levels of air pollution have been positively linked to decline in at least four diverse and widely separated forest types (McLaughlin and Percy, 1999). Pollutants of concern to forests at the regional scale include wet- (not discussed in this paper) and dry-deposited acids and ozone (O_3). In the past, sulphur dioxide (SO_2) emissions from industrial processes such as sulphide-ore smelting and fossil fuel combustion (petroleum/coal) have been sufficient to induce acute injury to foliage on a large number of tree/crop species across North America (cf. Legge *et al.*, 1998). Plate 1 show examples of SO_2 injury to two tree species. The best known case history (SO_2, particulate, Cu, Ni) in North America occurred around the smelter complex in Sudbury, Ontario, Canada. By 1979, the areal extent of damage due to long-term exposure (early 1900s onwards) of the surrounding boreal forest ecosystem was manifested in no forest communities within 3 km SSE of the smelters, only pockets of remnant forest up to 8 km, and bare hilltops as far as 15 km (Freedman and Hutchinson, 1980a). Following installation of a 380 m smokestack emitting over three million tons of SO_2 in 1973, only 3% of sulphur

Plate 1. Field injury to Sumac (*Rhus* species) in the U.S., caused by exposure to SO_2 (source: L. Weinstein).

was calculated to be deposited within a 60 km radius and high levels of Ni and Cu were found in soils up to 70 km away (Freedman and Hutchinson, 1980b). With significant reduction in SO_2 and metal emissions from the mid-1980s onwards, rehabilitation of 10,000 ha of barren land and 36,000 ha of stunted, open birch-maple woodland is underway (Winterhalder, 1996).

With successful implementation of SO_2 emission control strategies in North America, vegetation injury is now restricted to localised geographic areas (Legge *et al.*, 1998) around smaller point sources. Legge *et al.* (1996) summarised boreal forest response to sour gas emissions in a 25-year study and demonstrated a relationship between ambient SO_2 exposure and boreal tree growth. The pollutant of most concern to crops remains O_3 with large economic impacts occurring over wide geographic areas in both the United States (EPA, 1996) and southern Ontario, Canada.

1.1. *Recent Emission and Concentration Trends*

Emissions of SO_2 in eastern North America have declined from over 20 million tons in 1980 to under 17 million tons in 1993 (EC, 1997a). U.S. air concentrations of SO_2 decreased 37% between 1985 and 1995 and particulate SO_4 concentration reductions have been widespread, except at several mid- to high-elevation, forested areas (NAPAP, 1998). U.S. emissions of anthropogenic NO_x decreased 6.5% from 23.3 million tons in 1980 to 21.8 million tons in 1995 (NAPAP, 1998). Trends in nitrogen species (HNO_3, NO_3) were generally more variable than for sulphur. Inputs of dry deposition may add 8–37% more S, and 15–65% more N, to that received via wet deposition (rain, cloud, fog), the dominant form of S and N deposition, depending on the region (EC, 1997a).

Composite national daily maximum 1h O_3 concentrations in the U.S. decreased 15% between 1987 and 1996. The highest national 1h maximum was in 1988. Ozone levels have declined 10% since 1987 at 194 rural monitoring sites (EPA, 1998). In Canada, time series analysis (Dann and Summers, 1997) identified a significant declining trend in daily maximum O_3 concentrations ranging from −0.05% to −0.08% yr^{-1}. Average days per year (1986–1993) exceeding the pre-2000 one-hour National Ambient Air Quality Objective (NAAQO) ranged from 18 in southern Ontario to 3 in the Southern Atlantic Region and 2 in the Lower Fraser Valley, British Columbia.

In the U.S., particulate matter less than 10 µm diameter (PM_{10}) trends were published for the first time in 1997. Composite average (845 sites) PM_{10} concentrations decreased 26% between 1988 and 1997 (EPA, 1999). In Canada, total suspended particulate decreased 54% during 1974–1992 (EC, 1999). Both countries have recently revised standards for particulate matter less than 2.5 µm in diameter ($PM_{2.5}$), of particular concern to human health (Tables 1 and 2).

1.2. *Air Quality Criteria*

Canada and the U.S. differ in approach to air pollution regulation. In Canada, the federal government sets National Ambient Air Quality Objectives (NAAQO) on the basis of recommendations from the Ca-

Table 1. Canadian Ambient Air Quality Objectives and Canada-Wide Standards.

Canada-Wide Standards (CWS)			
Ozone*	65 ppb		
$PM_{2.5}$†	30 µg m^{-3}		
National Ambient Air Quality Objectives (NAAQO)‡			
	Desirable	Acceptable	Tolerable
Sulphur dioxide (ppb)			
1 Hour	172	334	
24 Hours	57	115	306
Annual	11	23	
Nitrogen dioxide (ppb)			
1 Hour		231	532
24 Hours		106	160
Annual	32	53	

Sources: Environment Canada (1999): http://www.ec.gc.ca/pdb/uaqt/obj_e.html
Environment Canada (2001): http://www.ec.gc.ca/air/gov-efforts_e.shtml

*A Canada-Wide Standard (CWS) of 65 ppb, 8-hour averaging time by 2010. Calculated as the 4th highest measurement annually, averaged over three consecutive years.

†A CWS of 30 µg/m^{-3}, 24-hour averaging time by year 2010. Achievement to be based on the 98th percentile ambient measurement annually, averaged over three consecutive years.

‡The maximum desirable level defines the long-term goal for air quality and provides the basis for an anti-degradation policy in unpolluted areas of the country. The maximum acceptable level is intended to provide adequate protection against adverse effects on humans, animals, vegetation, soil, water, materials and visibility. The maximum tolerable level is determined by time-based concentrations of air contaminants.

nadian Environmental Protection Act/Federal-Provincial Advisory Working Group on Air Quality Objectives and Guidelines. The NAAQO are listed as maximum desirable, acceptable and tolerable concentrations (Table 1). Provincial governments had the option of adopting these as objectives or enforceable standards. NAAQO are designed to provide protection to human health, vegetation, animals and materials. Recently, Canada-Wide Standards (CWS) for particulate matter less than 2.5 µm diameter ($PM_{2.5}$) and O_3 have been developed to protect human health under auspices of the Canadian Council of

Table 2. United States National Ambient Air Quality Standards (NAAQS).

Pollutant	Standard value	Standard type
Ozone*+		
1-Hour average	120 ppb	Primary and Secondary
8-Hour average	80 ppb	Primary and Secondary
$PM_{2.5}$+		
Annual arithmetic mean†	15 $\mu g\ m^{-3}$	Primary and Secondary
24-Hour concentration‡	65 $\mu g\ m^{-3}$	Primary and Secondary
PM_{10}		
Annual arithmetic mean§	50 $\mu g\ m^{-3}$	Primary and Secondary
24-Hour concentration¶	150 $\mu g\ m^{-3}$	Primary and Secondary
Sulphur dioxide (SO_2)		
Annual arithmetic mean	80 $\mu g\ m^{-3}$	Primary
24-Hour average	365 $\mu g\ m^{-3}$	Primary
3-Hour average	1300 $\mu g\ m^{-3}$	Secondary
Nitrogen dioxide		
Annual arithmetic mean	100 $\mu g\ m^{-3}$	Primary and Secondary

Source: EPA (2000) http://www.epa.gov/airs/criteria.html

+The ozone 8-hour standard and the $PM_{2.5}$ standard are pending U.S. Supreme Court reconsideration of a 1999 Federal Court ruling blocking implementation of the 1997 standards proposed by the U.S. EPA.

PM_{10} = particles with diameters ≤ 10 μm; $PM_{2.5}$ = particles with diameters ≤ 2.5 μm.

*As the three-year average of the 4th-highest daily maximum 8-hour average of continuous ambient air monitoring data over each year.

†As the three-year average of the annual arithmetic mean of the 24-hour concentrations from single or multiple population oriented monitors.

‡As the 98th percentile of the distribution of the 24-hour concentrations for a period of one year, averaged over three years, at each monitor within an area.

§As the arithmetic average of the 24-hour samples for a period of one year, averaged over three consecutive years.

¶As the 99th percentile of the distribution of the 24-hour concentrations for a period of one year, averaged over three years, at each monitor within an area.

Ministers of Environment (Table 1). The CWS of 65 ppb O_3, with an 8-hour averaging time by 2010 (Table 1), was adopted in 2000. Achievement is based on the 4th highest measurement annually, averaged over three consecutive years. The revised U.S. standard for ozone (Table 2) differs from the CWS only in attainment being based on an 8-hour 80 ppb level (EPA, 2001).

In the U.S., the United States Environmental Protection Agency (EPA) is responsible for setting National Ambient Air Quality Standards (NAAQS) for criteria pollutants (Table 2). Unlike the Canadian NAAQO, the NAAQS are legally enforced. Primary standards are designed to protect human health; secondary standards are designed to protect welfare including vegetation. NAAQS are set following a process during which a "criteria document" is developed comprising a compilation and scientific assessment of all health and welfare information available for the pollutant. EPA also develops a "staff paper" compiled by technical staff, to translate science into terms that can be used in making policy decisions. Both "papers" are based upon extensive consultation and scientific review, and are examined by the Congressionally-appointed Clean Air Scientific Advisory Committee prior to recommendations being made to the EPA Administrator for consideration in proposing revisions to the standards.

Both countries continue to make progress under the Ozone Annex to the 1991 Canada-United States Air Quality Agreement negotiated to reduce the transboundary movement of smog-causing pollutants. However, when the CWS is calculated for the 1996–1998 period using Canadian and U.S. data for eastern North America (Dann, 2001), only small portions in the extreme northeast are seen to meet the CWS. Highest 4th highest daily maximum 8-hour O_3 occurred in mid-coastal and mid-western areas, as well as the Lakes Michigan, Erie and Ontario basins with concentrations typically decreasing in a northeasterly direction (Plate 2).

1.3. *Vegetation*

North American forests are immense in range and ecological diversity. The 326 M ha of forested land in the U.S. (Powell *et al.*, 1992) and 417 M ha in Canada (NRCAN, 1998) represent a highly valuable

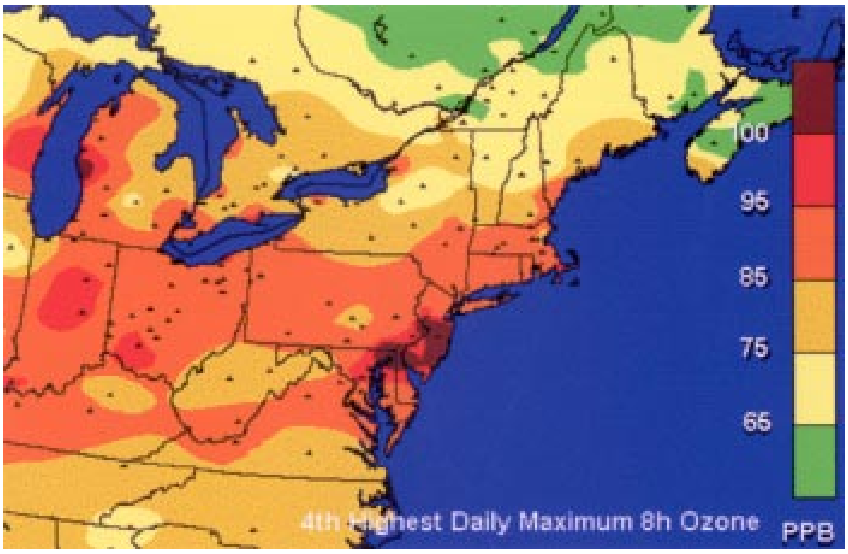


Plate 2. Northeast North American regional levels of 4th highest daily maximum O_3 calculated for the period 1996–1998 using the Canada-Wide Standard of 65 ppb, 8-hour averaging time. Triangles represent locations of Canadian and U.S. ground-level O_3 monitoring stations contributing data. Reproduced from Dann (2001).

economic resource for which maintenance of long-term productivity is a very high priority. Both actual and perceived potential responses of North American forests and crops to atmospheric pollution during recent decades have figured strongly in policy decisions on air quality regulation. During 1998 in Canada, 10.1 M ha of grains and oilseeds, along with 0.87 M ha speciality crops, were harvested (AAFC, 1998). Crops harvested and classified O_3- (Krupa *et al.*, 1998) and SO_2- (Legge *et al.*, 1998) sensitive included wheat, soybean, dry peas, dry beans and potato. SO_2-sensitive crops harvested included barley and oats. In the U.S. during 1997, 345 M ha were farmed (NASS, 1997). Crops harvested and classified as O_3- and SO_2-sensitive included wheat (23.8 M ha), soybean (26.8 M ha), cotton (5.3 M ha), potatoes (0.53 M ha) and tomato (0.15 M ha). O_3-sensitive crops harvested included edible dry beans (0.69 M ha), tobacco (0.32 M ha), rye (0.12 M ha) and red clover (0.02 M ha). SO_2-sensitive crops harvested included barley

(2.18 M ha), oats (1.09 M ha), sugar beets (0.57 M ha), and lettuce/romaine (0.12 M ha).

2. Impacts of Air Pollution

2.1. *Field Evidence of Effects on Forests*

Regional air pollution is an important component of the changing atmospheric environment in which North American forests are growing. The areal extent within which air pollutants are affecting forests at the regional level is shown in Fig. 1. High levels of mortality in northeastern hardwood forests since the early 1980s have been directly linked to air pollution. While sugar maple condition is generally improved in the U.S. (Stoyenhoff *et al.*, 1998) and Canada (Hall *et al.*, 1998), regional analysis of soil buffering in Ontario has shown that areas where critical loads of S/N were exceeded had higher levels of branch

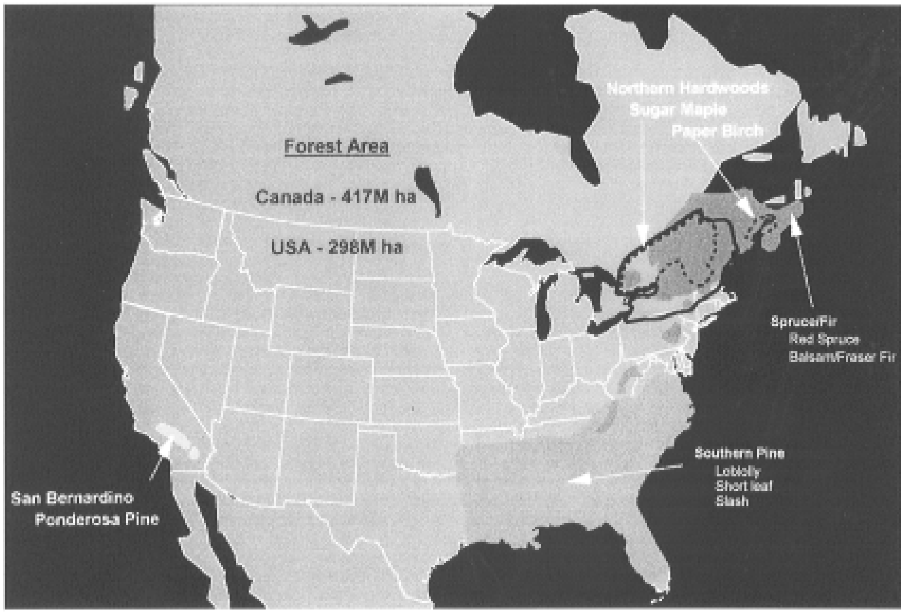


Fig. 1. Locations and area extent of four multi-disciplinary forest case studies used to evaluate process-level response to ambient levels of air pollution in regional forest types. Reproduced from McLaughlin and Percy (1999).

dieback (Arp *et al.*, 1996). Tree response has been primarily linked to soil sensitivity to acidic deposition, especially in the northern Appalachians. Accelerated loss of base cations from soils resulted in reduced soil Ca, Mg and K concentrations in sugar maple stands (Foster *et al.*, 1992). N saturation may also be partly responsible for nutrient imbalances and reduced growth on some sites (Yin *et al.*, 1994). However, O_3 cannot be excluded, although rigorous analysis of the circumstantial relationship between increased seasonal O_3 exposure and increased crown transparency has not been conducted. Interaction of cation leaching with extreme climate events may have been ultimately responsible for sugar maple decline, given the role of deep freezing of fine roots on canopy processes (Robitaille *et al.*, 1995). In eastern coastal birch forests, deterioration of white birches was first observed in the early 1980s (Magasi, 1989). Field studies subsequently documented a correlation between H^+ ion and NO_3 in fog and leaf browning (Cox *et al.*, 1996). The area also receives long-range transported O_3 from urban conurbations to the southwest and has the second highest annual mean O_3 concentration (34 ppb) in Canada (EC, 1997b).

Eastern-spruce forests occupy 10 M ha in eastern North America. In northern Appalachian spruce-fir forests, "winter injury" was first observed in the late 1950s on red spruce growing at high elevations. Recurring injury resulted in decreased radial growth and increased mortality in both cloud-/O_3-exposed, mountain (Johnson and Fernandez 1992) forests and fog-/O_3-exposed, coastal (Jagels *et al.*, 1989; Percy *et al.* 1993) red spruce forests. Growth decline of high elevation red spruce in southern Appalachian spruce-fir forests began later (1965). Canopy condition at low- and high-elevation sites declined during 1985–1989 (Peart *et al.*, 1992), while red spruce mortality increased with elevation and ranged from 3–4% yr^{-1} (Nicholas, 1992). A summary of research on decline of red spruce conducted under the Spruce-Fir Research Cooperative is available (Eagar and Adams, 1992). Dendro-climatological analysis (Cook and Johnson, 1989) has shown that growth decline was due to increased sensitivity of trees to winter (northern) and warmer late summer (southern) temperatures. Field studies have further shown that deposition of strong anions has reduced cation availability, reduced net carbohydrate production in foliage (Schaberg *et al.*, 1997), reduced photosynthesis and increased respiration (McLaughlin *et al.*, 1993). Acid deposition impacts on Ca availability (McLaughlin and Wimmer, 1999), causing reduced cold hardiness of foliage (DeHayes *et al.*, 1997) in the north, and acid deposition-induced Ca deficiency, leading to increased dark respiration, in the south are possible explanations.

Covering an area of 25 M ha in North America, significant growth declines in un-managed southern pines reported in the early 1980s (Sheffield and Cost, 1987) have been a major concern. The results of eight years of coordinated work under the Southern Commercial Forest Research Cooperative have been compiled by Fox and Mickler (1996). O_3 is known to significantly increase effects of soil moisture stress on stem growth of loblolly pine (McLaughlin and Downing, 1996). Somers *et al.* (1998) were recently able to correlate visible O_3 injury with radial growth of individual yellow-poplar trees in the Great Smokey Mountains National Park, but cautioned that this did not imply cause-effect. Acidic deposition effects in the short term are not expected to be significant (Teskey, 1995) and the main effects are expected to occur through primary impact on nutrient cycles. While 10–15% of commercial pine forest was believed to be limited by low cation supply (Binkley *et al.*, 1989), it has been shown that approximately 80% of exchangeable calcium has been lost over 30 years in a reference watershed, while base saturation has declined from > 55% in 1962 to 10% in 1990 (Richter *et al.*, 1994).

Since at least the mid-1950s, much of the 16 M ha of mixed conifer forest in Southern California has been exposed to the highest concentrations of O_3 in North America, including nighttime concentrations > 50 ppb at higher elevation sites. The detrimental role of N deposition is also well established, and is unique in that most occurs in dry form as acidic vapour, gaseous and particulate species. Deposition as high as 25–45 kg N ha^{-1} yr^{-1} has resulted in localised N saturation (Bytnerowicz and Fenn, 1996). Species most affected include ponderosa and Jeffrey pines. The complete history of the extensive, multidisciplinary case study in the San Bernardino Forest has recently been published (Miller and McBride, 1999). Effects from foliar level to successional stage have been documented. Early work by Miller (1973), when O_3 levels were highest, reported annual mortality between

2–2.5% per year. Concentrations ranging from 50–60 ppb O_3 induced foliar injury and early needle loss, decreased nutrient availability in stressed trees, reduced carbohydrate production with lessened tree vigour resulting in decreased height/diameter growth and increased susceptibility to bark beetles (Miller *et al.*, 1982). With gradually diminishing O_3 stress during 1976–1991, Miller *et al.* (1989) have reported an improvement (1974–1988) in the foliar injury index except at the most exposed plots.

2.2. *Field Evidence of Effects on Crops*

Ozone remains the most important air pollutant for crops in North America. Foliar injury from O_3 in Canada has been reported in a number of sensitive species; potato in New Brunswick, dry bean, soybean and tobacco in Quebec; dry bean, soybean, tomato, onion, tobacco, cucumber, radish, grape and peanut in Ontario; pea and potato in British Columbia (Pearson and Percy, 1997). Yield loss in some important field/horticultural crops is documented as well. In Ontario, crops considered at greatest risk include dry bean, potato, onion, hay, turnip, winter wheat, soybean, spinach, green bean, tobacco, tomato and sweet corn. Crop loss estimates range from 4% of the total U.S. $1.9 billion annual Ontario crop sales to U.S. $9 million (1986) in the Lower Fraser Valley, British Columbia.

In the U.S., foliar injury was definitively attributed to O_3 in 1959 (Heggestad and Middleton, 1959) and crop yield loss is now widespread. Adams *et al.* (1988) reported that the impacts of O_3 on crops may amount to U.S. $3 billion per year. Using annual-weighted yield reductions for four major crop species and all U.S. crops in the National Crop Loss Assessment Network (NCLAN) with estimated 1988 and 1989 O_3 exposure, Tingey *et al.* (1993) estimated yield reduction. Reductions were highly variable, depending upon species and year. Variation in corn yield loss due to O_3 was least variable, ranging from 1–20% per year while wheat was most variable, ranging from 2–80% yield reduction per year. Median yield losses across all crop growing areas in 1988, an extreme O_3 year, were 18.9% and only 3.1% in 1989, a lower O_3 year.

2.3. *Experimental Research*

There is a large body of scientific literature since the early 1970s from North American experimental research. The majority of this work was conducted with O_3 exposure of crops for the purposes of defining dose-response relationships and better defining risk at the species/varietal levels. These data were summarised during the air quality criteria review process in the U.S. (EPA, 1996) and the 1996 Canadian NOx/VOC Science Assessment (Pearson and Percy, 1997). Considerable research is also available on tree seedling exposure to wet-deposited acids (mainly simulated rain and fog) and O_3. Recent reviews of this body of work can be found in forest case study summaries above and in state of science reviews (Chappelka and Chevonne, 1992; EPA, 1996; Chappelka and Samuelson, 1998).

In North America, O_3 exposures of crops and trees have occurred in chambered laboratory and greenhouse, chambered-field, and open-air systems. Generally, methodologies became more sophisticated with time as new technology became available and new concepts forcing more "natural" exposure were developed. Portable cuvettes (Legge *et al.*, 1978) allow the controlled exposure of intact leaves/small branches while branch chambers (Teskey *et al.*, 1991) have been fitted around complete age classes of needles for controlled exposure of a combination of gases including ^{14}C tracer studies in mature pine canopies. Earlier use (Percy *et al.*, 1992) of continuous stirred tank reactor (CSTR) systems in laboratories/greenhouses designed for controlled, mass balance studies of O_3 flux, have largely gone out of use except in targeted applications, such as exposure of pine/hardwood seedlings to the extremely phytotoxic dry N species, HNO_3 (Bytnerowicz *et al.*, 1998) where excess vapour must be captured and destroyed.

Most experimental research, however, has used the chambered-field approach as open-top chambers (OTC) modified for field use with crops by Heagle *et al.* (1979). The OTCs conferred the advantage of allowing controlled gas application, statistical power through a randomised block allocation of up to five treatments (ambient-open, charcoal-filtered chamber, non-filtered chamber, $1 \times O_3$, $2 \times O_3$) with three replicate OTC's per treatment in a field setting. OTC's were used in the largest experimental O_3 fumigation project under NCLAN

(1980–1986) and data were summarised in EPA (1996). Data from NCLAN exposure-crop response regression analyses indicated that at least 50% of the species/cultivars tested were predicted to exhibit 10% yield loss at 7-hour seasonal mean O_3 concentrations of < 50 ppb (EPA, 1996). Effects were found to occur following only a few hours at > 80 ppb O_3. Ambient levels in many parts of the U.S. were considered sufficient to impair plant growth and yield. Retrospective analysis of NCLAN crop-response by Legge *et al.* (1993) used statistical techniques to compare ambient air versus non-filtered OTC treatment response and remove studies with a chamber effect. With the reduced data set, they reported a crop growth threshold as low as 35 ppb O_3. Best fit was achieved between air quality and impacts on crop yield using cumulative frequency of mid-range concentrations between 50–87 ppb O_3 (Krupa *et al.*, 1994; Legge *et al.*, 1995). While OTC's remain in use for certain applications such as relating foliar injury to local O_3 exposure (Skelly *et al.*, 1999), significant differences in micro-meteorology created within the chambers has seen their use decline. State of science at this time, while recognising the continued importance of high, short-duration O_3 concentrations, points to the increasing importance of mid-range levels in growth effects and yield loss, especially given the cumulative nature of plant response to O_3, particularly in long-lived forms such as trees.

In the case of cuvettes, branch chambers, CSTR's and OTC's, extrapolation of data to the field situation was found to be difficult due to chamber effect and age/size of plant material fumigated. Since the early 1990s, open-air fumigation systems have been favoured. Predominant are the zonal air pollution system (ZAPS) used in exposure of tree seedlings (Runeckles and Wright, 1996) and Free Air Carbon Dioxide Enrichment (FACE) (Hendrey and Kimball, 1994). The Aspen FACE in Wisconsin, U.S. is exposing hardwood species to O_3 in a randomised block design comprising 12, 30 m diameter rings including four treatments [ambient, enhanced CO_2 (+200 ppm), $1.5 \times O_3$, $CO_2 + O_3$], with three replicate rings per treatment set across a 32 ha site. While expensive to establish and operate, FACE has many advantages including: open-air exposure of plants to controlled fumigations; no chamber effects; and multi-year exposures of dynamic ecosystems under the influence of yearly climate variability. Earlier results from the aspen FACE reported highly significant effects of O_3 in aspen and birch clones such as decreased height growth, reduced radial growth, early leaf abscission, reduced below-ground growth and increased insect feeding and disease incidence (Karnosky *et al.*, 1999a). Now, after integrating four years of above-ground growth data, it is clear that O_3 offsets productivity gain in a CO_2-enriched atmosphere in rapidly growing aspen and birch, and that effects of O_3 on foliar chemistry are having a consequential effect on some insects and diseases (Isebrands *et al.*, 2001; Karnosky *et al.*, 2001; Karnosky *et al.*, 2002). Interestingly, Aspen FACE is providing confirmation in direction and magnitude of effects reported from OTC (Karnosky *et al.*, 1996) and "natural" field exposures along O_3 gradients (Karnosky *et al.*, 1999b).

3. Current Dose-Response Relationships and Risk Assessment Methods

Risk assessment involves a quantification of the likelihood of adverse effects resulting from exposure to a stress, or complex of stressors. Current approaches to defining risk to forests and crops from air pollution in North America are focused on the spatial characterisation of risk from O_3 in the U.S., and on mapping critical loads for forest soils in Canada. Methods used to assess forest risk O_3 include use of physiological models with species distribution and O_3 levels to estimate geographical extent for a range of growth responses. For mature loblolly pine and selected hardwoods in the southeast, regional application of physiological growth models with O_3 exposure data indicates an expected growth reduction of 0–35% depending upon species and O_3 level (McLaughlin and Percy, 1999). Others used experimentally-derived response data, usually from OTC's, to assess risk in the eastern U.S. (Flagler *et al.*, 1997; Taylor, 1994; Lefohn *et al.*, 1997). Outputs of these and other studies have been summarised by Chappelka and Samuelson (1998). They conclude that O_3 is affecting forest growth in the eastern U.S., but that genotype, edaphic, climatic factors, and assumptions used in the assessments, affect the assessment of risk. They also conclude that seedling stage studies are not suitable for scaling-up purposes (i.e. predictive risk assessment).

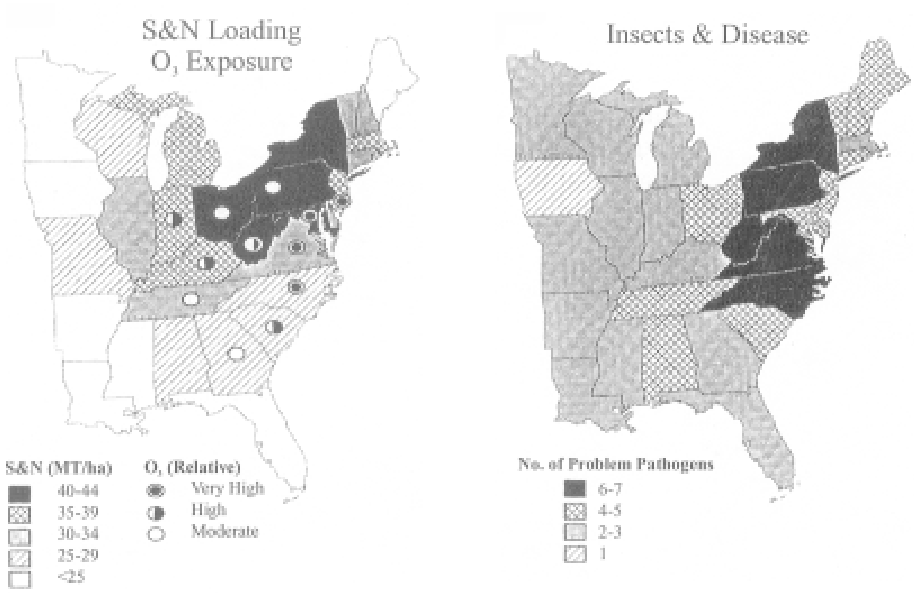
Spatial characterisation of risk using a GIS resource base is a promising tool. Hogsett *et al.* (1997) used GIS to integrate: (1) estimated O_3 exposures over forested regions; (2) measures of O_3 effects on species' and stand growth; and (3) spatially-distributed environmental, genetic and exposure influences on species' response to O_3 to characterise risk to eastern U.S. forests. Area-weighted response of annual seedling biomass loss was reported by sensitivity class: sensitive — aspen/black cherry (14–33% biomass loss over 50% of their distribution); moderately sensitive — tulip polar, loblolly pine, eastern white pine and sugar maple (5–13% biomass loss); and insensitive — Virginia pine and red maple (0–1% loss). For crops, Tingey *et al.* (1991) estimated relative yield loss for all crops using the NCLAN database across crop-growing regions in the eastern U.S., on a 20 km cell, based on estimated three-month SUM06 O_3 exposure and Weibull parameters for species' response functions. Predicted relative yield loss ranged from 0–> 30% in 1988 and 1989. Areal extent of largest loss (> 30%) was limited to portions of the mid-west, east-central and southeast in 1989; it covered the majority of crop ranges in 1988, a year of record high O_3 levels. No comparable risk assessment has been completed for Canada, although O_3 is known to affect crop yield in Ontario (Pearson, 1989) and has been circumstantially related to sugar maple crown dieback in eastern Canada (McLaughlin and Percy, 1999).

According to Nilsson and Grennfelt (1988), critical loads are quantitative estimates of an exposure to one or more pollutants below which significant harmful effects on specified sensitive elements of the environment do not occur. Regional estimates have been published, including those of Richter and Markewitz (1995) who estimated that 60% of southeastern pine soils are susceptible to accelerated cation leaching from acidic deposition. Lately, Arp *et al.* (1996) have developed critical loads for S and N deposition, by superimposing atmospheric deposition rates on soil acidification potential using a steady-state model to calculate exceedance or non-exceedance for Acid Rain National Early Warning System (ARNEWS) plots. Exceedance of > 500 eq ha^{-1} yr^{-1} is associated with an annual productivity loss of 10%. Most risk assessment has focused on the S and N deposition. However, recent compelling evidence put forward by McLaughlin and

Wimmer (1999) points to a central role for Ca physiology in terrestrial ecosystem response to air pollution.

4. Conclusions

Regional levels of O_3 are significantly impacting yield of a number of commercially significant crop species in the U.S. and in certain Canadian provinces, with important economic consequences. National SO_2 concentrations are no longer considered a threat to crops and forests, but damage continues to occur in localised areas where strong point sources exist. Acid deposition has not been shown to be a significant threat to crop yield given the annual cycle of growth. There is no conclusive evidence to date linking degree of visible, foliar injury with productivity loss in crops or trees. It is clear from process-level studies that regional levels of air pollution measured in the four diverse and widely-distributed, regionally important North American case study forests are reducing carbon reserves, increasing water stress and reducing nutrient availability (McLaughlin and Percy, 1999). Carbon reserves in particular, are critical in plant defense against a variety of biotic/abiotic stressors, which may partly explain the coincidence in frequency of insects/disease with rates of air pollution exposure in the eastern U.S. (Fig. 2). Monitoring systems established to detect change with their utilitarian assumptions, however, have not been designed to link cause and effect. Although countries do not always attribute damage to air pollutants, absence of a record does not imply that the particular cause of injury (air pollution) is not present (Innes, 1998). New, integrated monitoring and research programmes are required to link air quality and forest health-productivity (Percy *et al.*, 2000; Percy 2001). Process-level modelling should be an essential ingredient (Chappelka and Samuelson, 1998) for future risk assessment. While forest case studies in Europe and North America have documented important air pollutant impacts, patterns of decline are distinct and reflect different underlying forest processes/disturbance histories, and, therefore, cannot be directly compared (Percy *et al.*, 1999). However, it is clear from North American forest-air pollution case histories that improvement in air quality does lead to improvement in forest health (McLaughlin and Percy, 1999).



(**A**) (**B**)

Fig. 2. Areas of the eastern U.S. having the greatest frequency of problems from forest insects and disease (**B**), related to areas receiving largest atmospheric S/N deposition and highest annual exposure to O_3 (**A**). Reproduced from McLaughlin and Percy (1999).

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