A PRELIMINARY ASSESSMENT OF THE MONTRÉAL PROCESS INDICATORS OF AIR POLLUTION FOR THE UNITED STATES

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Abstract. Air pollutants pose a risk to forest health and vitality in the United States. Here we present the major findings from a national scale air pollution assessment that is part of the United States' 2003 Report on Sustainable Forests. We examine trends and the percent forest subjected to specific levels of ozone and wet deposition of sulfate, nitrate, and ammonium. Results are reported by Resource Planning Act (RPA) reporting region and integrated by forest type using multivariate clustering. Estimates of sulfate deposition for forested areas had decreasing trends (1994–2000) across RPA regions that were statistically significant for North and South RPA regions. Nitrate deposition rates were relatively constant for the 1994 to 2000 period, but the South RPA region had a statistically decreasing trend. The North and South RPA regions experienced the highest ammonium deposition rates and showed slightly decreasing trends. Ozone concentrations were highest in portions of the Pacific Coast RPA region and relatively high across much of the South RPA region. Both the South and Rocky Mountain RPA regions had an increasing trend in ozone exposure. Ozone-induced foliar injury to sensitive species was recorded in all regions except for the Rocky Mountain region. The multivariate analysis showed that the oak-hickory and loblolly-shortleaf pine forest types were generally exposed to more air pollution than other forest types, and the redwood, western white pine, and larch forest types were generally exposed to less. These findings offer a new approach to national air pollution assessments and are intended to help focus research and planning initiatives related to air pollution and forest health.

Keywords: acid deposition, forest health and vitality, multivariate clustering, ozone biomonitoring, spatial analysis, sustainable forests, tropospheric ozone

1. Introduction

Air pollutants are a global concern because they can have significant cumulative impacts on forest ecosystems by affecting regeneration, productivity, and species composition (Roundtable on Sustainable Forests, 2000). As part of the Montréal Process for the Conservation and Sustainable Management of Temperate and Boreal Forests (Montréal Process, 1995), air pollution is addressed as part of a set of Criteria and Indicators. Seven criteria and 67 indicators are guidelines for characterizing components of sustainable forest management such as biological diversity, productive capacity, maintenance of forest health and vitality, soil and

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water, and socio-economic conditions (Roundtable on Sustainable Forests, 2000). These guidelines are the framework for strategic forest planning (USDA Forest Service, 2000) as well as national assessments of forest sustainability (e.g., USDA Forest Service, 1997), resource management (e.g., USDA Forest Service, 2001), and forest health (e.g., Conkling *et al.*, In Press). For the United States' 2003 Report on Sustainable Forests, an assessment of the area and percent forest subjected to air pollutants that may cause negative impacts is incorporated under the criterion of maintenance of forest health and vitality. Here we present the major results of the air pollution assessment.

The term 'air pollution' encompasses a wide range of concerns, but acid deposition and ozone (O_3) are of primary concern for United States forests (Driscoll *et al.*, 2001; Hakkarienen, 1987). Acid deposition comes, in part, from gaseous emissions of sulfur dioxide (SO_2), nitrogen oxides (NO_x), and ammonia (NH_3) that are deposited in wet form as sulfate (SO_4^{2-}), nitrate (NO_3^-), and ammonium (NH_4^+) by rainfall, snowfall, and sleet. Inputs of sulfur and nitrogen can also come from dry deposition, or from clouds and fog in high elevation and coastal areas. Acid deposition impacts forests through indirect effects such as soil and water acidification (ESA, 1999, Driscoll *et al.*, 2001) and tree nutrition (Houston, 1999; ESA, 1999), and direct effects such as foliar injury (DeHayes *et al.*, 1999).

While there have been decreases in sulfur deposition, there is evidence that sulfur has accumulated in northeastern soils and trees. Watershed mass balances in the northeast have shown that loss of sulfur exceeds inputs from deposition, therefore suggesting that sulfur has accumulated in the soil (Driscoll *et al.*, 1998). Wood chemistry analysis of several northern tree species suggests that increased sulfur concentration in trees is associated with increased SO_4^{2-} deposition (Riitters *et al.*, 1991). Nitrogen saturation occurs when nitrogen input exceeds biological demand as has been observed in mixed conifer forests and chaparral stands near the California Los Angeles Basin (ESA, 1999). Nitrogen saturation leads to leaching of base cations, decreased plant function, loss of fine root biomass and decreases in symbiotic mycorrhizal fungi (ESA, 1999).

Tropospheric O_3 is produced from photochemical reactions between NO_x and volatile organic compounds. It is a gaseous air pollutant that causes foliar injury (Hakkarienen, 1987; Miller and Millecan, 1971; Skelly *et al.*, 1987; Treshow and Stewart, 1973), and is frequently measured at phytotoxic levels (Cleveland and Graedel, 1979; Lefohn and Pinkerton, 1988). Long-range transport of contaminated air masses contributes to elevated O_3 levels in forested areas. O_3 impacts trees and forests directly by causing visible foliar injury, reduced photosynthetic activity, and increased leaf senescence. Other effects from prolonged O_3 exposure can include altered carbon allocation, growth reduction, and predisposition to insects and pathogens (Chappelka and Samuelson, 1998, Cobb *et al.*, 1968). Because plant uptake of tropospheric O_3 occurs during gas exchange, potential effects are moderated by phenology and environmental conditions such as light, temperature, relative humidity, and soil moisture that ultimately determine O_3 uptake and plant response (McCool, 1998).

Forest landscapes have highly variable pollution sensitivities and there are complex interactions among pollutants. O_3 sensitivity is variable among and within species. Some tree species are very sensitive to O_3 exposure (e.g. black cherry: Krupa and Manning, 1988) but others (e.g. sugar maple; Renfro, 1992) are tolerant. Bennett *et al.* (1994) suggested that hypersensitive white pine genotypes have already been eliminated from some populations. In the case of acid deposition, high elevation areas with shallow soils and a low buffering capacity are particularly sensitive, and land use history influences sensitivity (ESA, 1999). Interactions among air pollutants are also important in identifying the overall impact of air pollution to forest ecosystems. For example, Takemoto *et al.* (2001) hypothesized that increased nitrogen supply from deposition could moderate the harmful effects of tropospheric O_3 on trees growing in nitrogen deficient soils in California mixed conifer forests. In other empirical studies, there is evidence of additive effects, synergistic effects, or no detectable changes in effects incited by multiple pollutants (e.g. Shan *et al.*, 1996; Izuta, 1998).

Pollutant interactions, inherent variability, and lack of understanding of the total ecosystem response to air pollution are significant barriers to large scale assessments that necessarily overlook many of the details understood to be important for particular species or sites at smaller spatial scales. The objective of this report is to assess air pollution as an ecological stressor that influences the maintenance of forest ecosystem health and vitality in the United States. To accomplish this, the percent of forestland that is exposed to wet SO_4^{2-}, NO_3^-, and NH_4^+ deposition and O_3 air pollution is summarized by Resource Planning Act (RPA) reporting region (Figure 1), and statistical composite summaries of those indicators are presented by forest type (Figure 1). This analysis does not identify specific negative impacts and does not attempt to establish cause-effect relationships.

2. Methods

Air quality and O_3 biomonitoring data were obtained from several sources (Table I). Annual wet deposition amounts (kg ha^{-1}) of NO_3^-, NH_4^+, and SO_4^{2-} from 1994 to 2000 for each National Atmospheric Deposition Program (NADP) monitoring station were used in this analysis. The Wisconsin Department of Natural Resources provided summarized U.S. Environmental Protection Agency data for 1994 through 2000 including the 24 hr SUM06 O_3 index for a 3-month growing season (June, July, August) nationwide. The SUM06 O_3 index is the sum of all average O_3 hourly concentrations greater than 0.06 ppm and will hereafter be referred to as 'O_3 index'. The U.S. Forest Service provided O_3 injury information from a network of biomonitoring plots that evaluated foliar injury response using O_3-sensitive bioindicator plants including tree, woody shrub, and herb species. On the biomonitoring

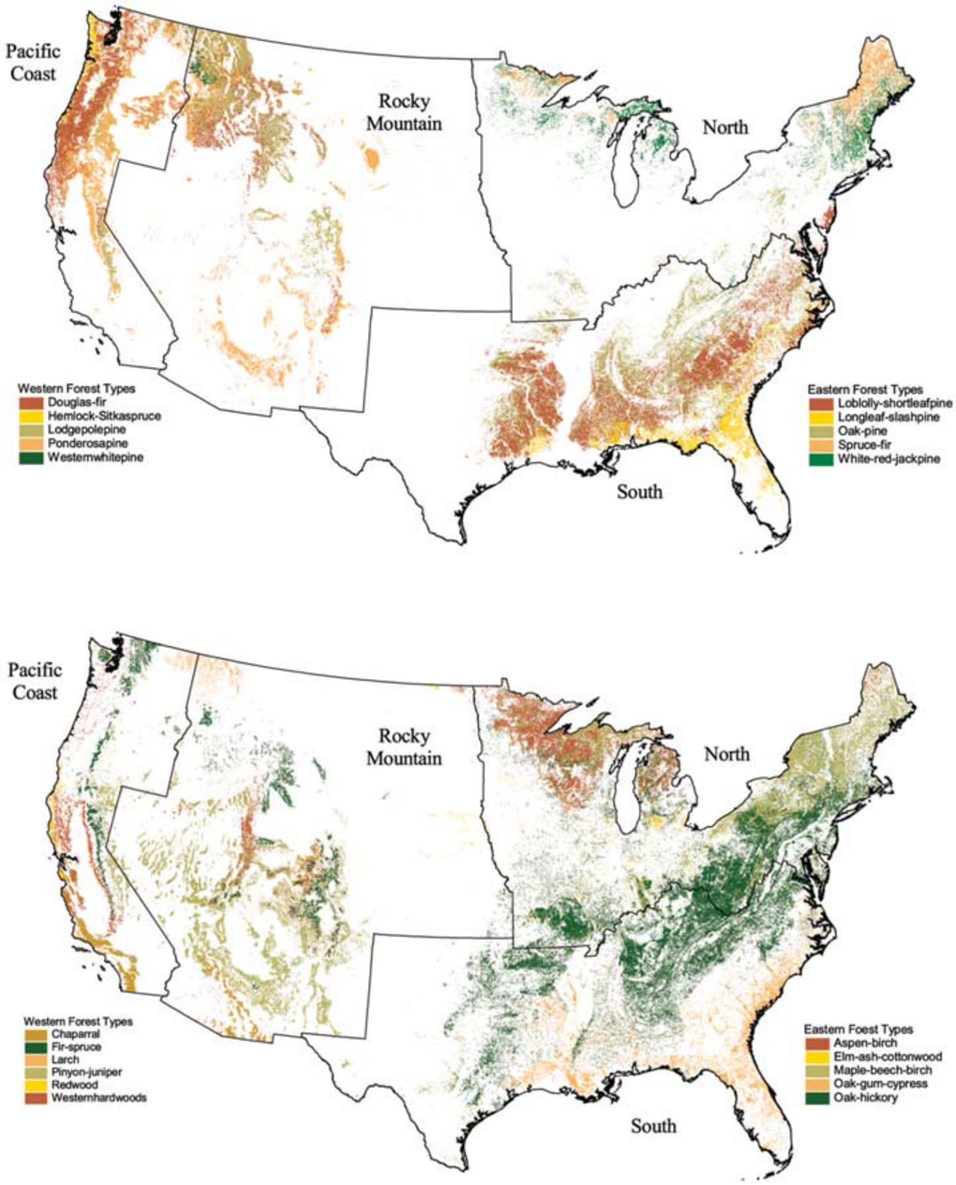


Figure 1. RPA regions and forest types in the coterminous United States.

plots, information is collected on specific bioindicator plants that respond to ambient levels of O_3 with distinct visible foliar symptoms that are easily diagnosed (USDA Forest Service, 1999; Smith *et al*., 2003). Data from biomonitoring plots were the only information about plant injury from air pollution.

We used three types of analyses to quantify air pollution exposure. The trends in exposure of forests to air pollution was assessed by RPA region for wet deposition

TABLE I

Characteristics of U.S. air pollution monitoring programs used in analysis

Pollutant	Monitoring program or agency	Spatial coverage and sampling intensity	Temporal coverage	Data issues
Wet Deposition (Sulfate, Nitrate, Ammonium)	National Atmospheric Deposition Program (NADP)	Coterminous United States, AK, HI, VI, PR; 220 monitors in Coterminous US in 2000	1978 to present	Pre 1994 data not comparable with post 1994 data (Lynch *et al.*, 1996)
Tropospheric ozone exposure	Environmental Protection Agency (EPA)	Coterminous United States, HI, VI, PR; 1114 monitors in Coterminous US in 2000	1993 to present	Monitoring stations concentrated in urban areas
Tropospheric Ozone biomonitoring	United States Department Of Agriculture Forest Service (USDAFS)	Coterminous United States; 918 plots in 2000	1994 to present	35 states currently implemented.

of NO_3^-, SO_4^{2-}, and NH_4^+, and for the O_3 index. The percent forest by RPA region exposed to specific levels of air pollution was estimated for NO_3^-, SO_4^{2-}, NH_4^+, O_3 index, and O_3 injury indicators (bioindicator data). We then used a cluster analysis to form composite air pollution indicators that were used to examine relative differences in air pollution exposure between forest types.

2.1. Trend estimates

We constructed maps of Thiessen polygons to identify the area of influence each NADP or EPA monitoring station had in each year (Isaaks and Srivastava, 1989). We then intersected each map of Thiessen polygons with a forest cover map (Zhu and Evans, 1994) to estimate the area of forest (km^2) represented by each monitoring station in each year. Weighted average annual (1994–2000) wet deposition of NO_3^-, NH_4^+, and SO_4^{2-} (kg ha^{-1} yr^{-1}), and O_3 index (ppm-hrs yr^{-1}) was calculated for each RPA region by $\Sigma(d_m w_m)/\Sigma w_m$ where d is the deposition rate for the indicator of interest (e.g. NO_3^-), and w is the weight (based on the km^2 of forest represented by each of m monitoring stations). Estimates of average annual change for forested areas were calculated for each RPA reporting region using the following general linear model: D = a + b(y) where b is the weighted estimate of annual change calculated by

$$[\Sigma w_m y_m d_m - ((\Sigma w_m y_m)(\Sigma w_m d_m)/\Sigma w_m)] / [\Sigma w_m y_m^2 - ((\Sigma w_m y_m)^2/\Sigma w_m)]$$

D is the weighted average annual indicator value and y is the year (Steel *et al.*, 1997). The probability that $b = 0$ was tested with the F-test and significance was assigned at the 0.05 level.

2.2. Estimating percent forest exposed to specific levels of air pollution

We used cumulative distribution functions (CDF) and frequency distributions to estimate the percent forest exposed to specific levels of air pollution. The Thiessen polygon method was used to estimate the CDF for wet deposition of NO_3^-, NH_4^+, and SO_4^{2-} (kg ha^{-1} yr^{-1}), and the O_3 index (ppm-hrs yr^{-1}) (Isaaks and Srivastava, 1989). For O_3 injury, a frequency distribution was calculated directly from the O_3 biomonitoring data.

For CDFs, the portion of forest below any cutoff value (c) in each RPA region was estimated. For each pollutant, the range of values across years and RPA regions was broken into 20 equal interval classes such that each class was identified by its upper limit c. The percent forest below each c in each RPA region across years was estimated by $(\Sigma A_c/7)/\Sigma A$ where A_c is the amount (km^2) of forest below c based on Thiessen polygons and A is the total amount of forest in the RPA region of interested based on Zhu and Evans (1994).

A different approach was needed for the O_3 biomonitoring data, where empirical frequency distributions of plot-level data were used to examine the percent of biomonitoring plots with O_3 injury. The proportion of leaves with O_3 injury and the mean severity of symptoms on injured foliage that were recorded for 10 to 30 plants of up to three species at each biomonitoring site (biosite) were used to calculate a biosite index (BI) (Smith 1995, Smith *et al.*, 2003). BI was defined as:

$$1000\left(m^{-1} \sum_{j=1}^{m} n_j^{-1} \sum_{i=1}^{n_j \geq 10} a_{ij}\, s_{ij}\right)$$

where m is the number of species evaluated, n_j is number of plants of the jth species, a_{ij} is the proportion of injured leaves on the ith plant of the jth species, and s_{ij} is the average severity of injury on the ith plant of the jth species. This index was classified into four risk categories that represent a relative measure of impacts from ambient O_3 exposure (Table II). These grouping were proposed by Smith (1995) and are based on the interpretation of preliminary field studies (1990–1994). They are designed to capture differences in plant damage to O_3 sensitive species in areas of low, moderate, and high O_3 exposure (Lewis and Conkling, 1994). While these categories represent injury to bioindicator plants there is an assumption of risk to other species associated with each category (Table II). The BI and associated groupings have been used in regional-scale risk assessments (Coulston *et al.*, 2003) and national-scale forest health reports (Conkling *et al.*, In Press).

TABLE II
Biosite index categories, risk assumption, and possible impact

Biosite index	Bioindicator response	Assumption of risk to forest resource	Possible impact
0 to < 5.0	Little or no foliar injury	None	Visible injury to isolated genotypes of sensitive species, e.g. common milkweed, black cherry
5.0 to < 15.0	Light to moderate foliar injury	Low	Visible injury to highly sensitive species, e.g. black cherry; effects noted primarily at the tree-level.
15.0 to < 25.0	Moderate to severe foliar injury	Moderate	Visible injury to moderately sensitive species, e.g. tulip poplar; effects noted primarily at the tree-level.
≥ 25	Severe foliar injury	High	Visible injury leading to changes in structure and function of the ecosystem.

TABLE III
Implementation of O_3 biomonitoring program in the United States

First year of monitoring	State
1994	CT, ME, MD, MA, MI, MN, NH, NJ, VT, WI
1995	DE, RI, WV
1996	IN
1997	AL, GA, IL, IA, OH, VA
1998	CA, CO, ID, OR, PA, WA, WY
1999	NY, NC, SC
2000	KY, MO, NV, TN, UT

The number of measurement years per biosite varied from 1 to 7. Some biosites in Massachusetts and Maine had seven annual measurements while Tennessee biosites were measured in 2000 only (Table III). We used the average BI for all measurements (1994 to 2000) in this analysis to estimate O_3 induced foliar injury for each RPA region.

2.3. Cluster analysis

Cluster analysis is a standard yet somewhat subjective multivariate statistical technique that is used for data reduction and interpretation (see, for example, Johnson and Wichern, 2002). It often reveals underlying statistical relationships and enables interpretations that would not otherwise be noticed. In this analysis, values for NO_3^-, SO_4^{2-}, NH_4^+ and O_3 index were based on interpolated maps created using inverse distance squared weighted interpolation (IDW) (Isaaks and Srivastava, 1989). Values were calculated by:

$$v'_p = \frac{\sum_{j=1}^{n \geq 12} d_j^2 (v_j)}{\sum_{j=1}^{n \geq 12} d_j^2}$$

where v'_p is the predicted indicator value at location p, p is any forested pixel identified by Zhu and Evans (1994), and d_j is the distance from the jth monitoring station to p. Monitoring stations greater than 500 km away from p were not used and a minimum of 12 monitoring stations were required to predict v'_p. For O_3 injury, BI values were used directly by giving all forested pixels within the hexagonal 65,000 ha area (White *et al.*, 1992) represented by each biomonitoring plot the BI value. For consistency among pollutants, each indicator was averaged across years and forest type within each hexagonal 65,000 ha area represented by each biomonitoring plot.

For the cluster analysis, only areas where estimates for all indicators were available were used. We used both the average and coefficient of variation (CV) for each air pollution indicator by forest type. The CV was included in the cluster analysis to account for variability of average values within forest types. The ten air pollution indicators thus included the average and CV of the BI (1994–2000), O_3 index (1994–2000), annual NO_3^- wet deposition (1994–2000), annual NH_4^+ wet deposition (1994–2000), and annual SO_4^{2-} wet deposition (1994–2000).

Each indicator was standardized to a mean of 0 (zero) and variance 1 (one) across forest types. A cluster analysis was then performed to reduce the dimensionality of air pollution indicators using the SAS VARCLUS procedure (SAS, 1999). The goal was to explain as much as possible of the original variance in the air pollution indicators with the fewest clusters. Clusters identified using the VARCLUS procedure are disjoint (non-overlapping). After the clusters were formed, the cluster score for each forest type was calculated (SAS 1999). The score shows the relative differences between forest types based on the linear combination of indicators identified using the VARCLUS procedure. This procedure was used to create a composite air pollution indicator by forest type for interpretive purposes.

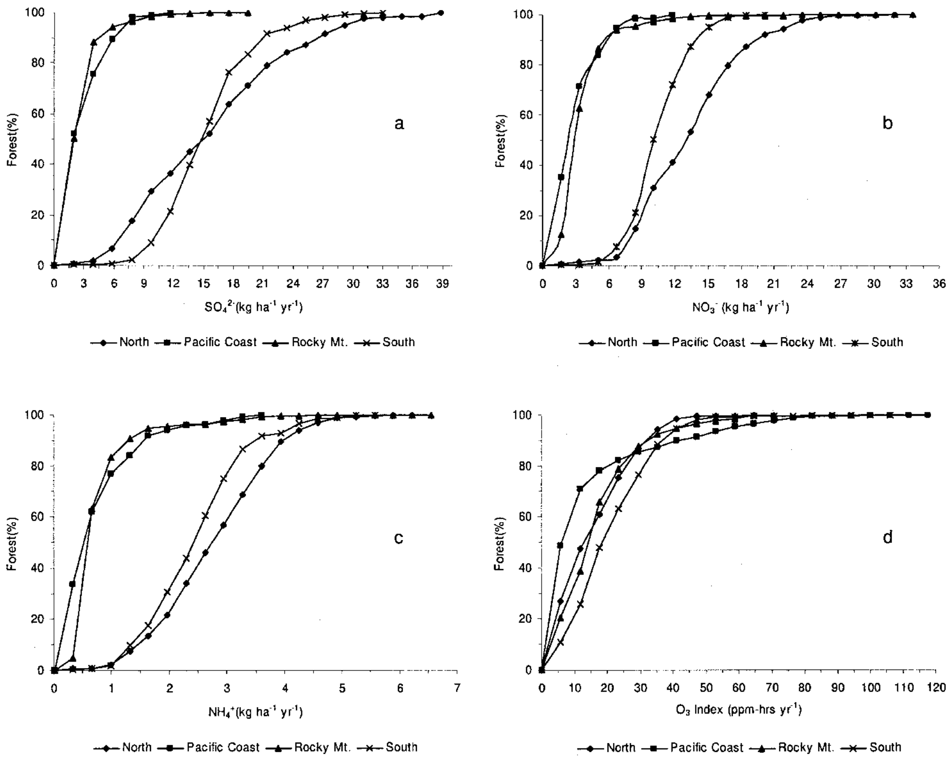


Figure 2. Cumulative distribution functions of average percent forest subjected to levels of (a) wet SO_4^{2-} deposition (1994–2000), (b) wet NO_3^- deposition (1994–2000), (c) wet NH_4^+ deposition (1994–2000), and (d) O_3 index (1994–2000).

3. Results

3.1. Forest ecosystem exposure

In the North and South RPA regions, approximately 50% of the forest was exposed to SO_4^{2-} deposition of more than 15 kg ha^{-1} yr^{-1} for the 1994 to 2000 period (Figure 2a). This was different from the Pacific Coast and Rocky Mountain RPA regions where approximately 50% of the forest received less than 2 kg ha^{-1} yr^{-1} for the 1994 to 2000 period. Less than 1% of the forest in the South and North RPA region received less than 2 kg ha^{-1} yr^{-1} (Figure 2a). The maximum amounts of SO_4^{2-} deposition for the North and South RPA regions were approximately 38 kg ha^{-1} yr^{-1} while maximums for the Pacific Coast and Rocky Mountain RPA regions were approximately 12 and 19 kg ha^{-1} yr^{-1}, respectively. Although average deposition rates were highest in the North and South, these regions had statistically significant decreasing trends of 0.471 and 0.348 kg ha^{-1} yr^{-1}, respectively (Table IV).

NO_3^- deposition was highest in the North RPA region where approximately 50% of the forest received an average annual input (1994 to 2000) of more than 13 kg

TABLE IV

Average and average annual change of wet SO_4^{2-}, NO_3^-, and NH_4^+ deposition (kg ha^{-1} yr^{-1}) and O_3 index (ppm-hrs yr^{-1}) by RPA region for the 1994 to 2000 period

	North	Pacific Coast	Rocky Mountain	South
SO_4^{2-} (kg ha^{-1} yr^{-1})	15.4 (–0.471*)	2.6 (–0.014)	2.5 (–0.094)	15.1 (–0.348*)
NO_3^- (kg ha^{-1} yr^{-1})	13.2 (–0.187)	2.9 (–0.025)	3.5 (0.006)	10.4 (–0.16*)
NH_4^+ (kg ha^{-1} yr^{-1})	2.8 (–0.049*)	0.7 (–0.024)	0.8 (–0.013)	2.4 (–0.066*)
O_3 index (ppm-hrs yr^{-1})	15.2 (–0.006)	13.9 (0.309)	16.0 (1.735*)	20.4 (2.648*)

ha^{-1} yr^{-1} (Figure 2b). Fifty percent of the forested area in the South RPA region received more than 10 kg ha^{-1} yr^{-1}. Deposition rates were lower in the Pacific coast and Rocky Mountain RPA regions. Approximately 50% of the forest in these areas received less than 2.5 kg ha^{-1} yr^{-1} for the 1994 to 2000 period (Figure 2b). In the North and South RPA regions, less than 3% of the forest received an average annual NO_3^- input of less than 5 kg ha^{-1} yr^{-1}. The South RPA region was the only region where the rate of NO_3^- deposition had a statistically significant decreasing trend of 0.16 kg ha^{-1} yr^{-1} (Table IV). The North RPA region also had a decreasing trend (0.187 kg ha^{-1} yr^{-1}) but this estimate was not significant at the 0.05 level.

Forests in the Pacific Coast and Rocky Mountain RPA regions were exposed to lower levels of NH_4^+ deposition than either the North or South RPA regions (Figure 2c). All forested areas in the Pacific Coast RPA region and 99% of the forest in the Rocky Mountain RPA region received inputs of less than 3.6 kg ha^{-1} yr^{-1}. In the North and South RPA regions, approximately 69 and 86% of the forested areas received less than 3.6 kg ha^{-1} yr^{-1} of NH_4^+ deposition, respectively (Figure 2c). Approximately 62% of the forest in the western U.S. (Pacific Coast, Rocky Mountain RPA regions) received less than 0.7 kg ha^{-1} yr^{-1} of NH_4^+ deposition during the 1994 to 2000 period. The North and South RPA regions had small but statistically significant decreasing trends in NH_4^+ deposition (Table IV).

On average, the growing season O_3 index was highest in the South RPA region (Table IV) where only 10% of the forests were exposed to O_3 index concentrations less than 6 ppm-hrs yr^{-1} (Figure 2d). The O_3 index was relatively low over most of the Pacific Coast region for the 1994 to 2000 period where 80% of the forest experienced exposures of less than 23 ppm-hrs yr^{-1}, however 10% of the forested area in this region experienced exposure between 41.2 and 117.8 ppm-hrs yr^{-1} (Figure 2d). Forests in the North RPA region were generally exposed to lower concentrations of O_3 (Table IV). Approximately 75% of the forest in this area had O_3 index exposures of less than 24 ppm-hrs yr^{-1} (Figure 2d). In the Rocky Mountain RPA region, approximately 50% of the forest had O_3 index exposure of less than 16 ppm-hrs yr^{-1}. Both the Rocky Mountain and South RPA region

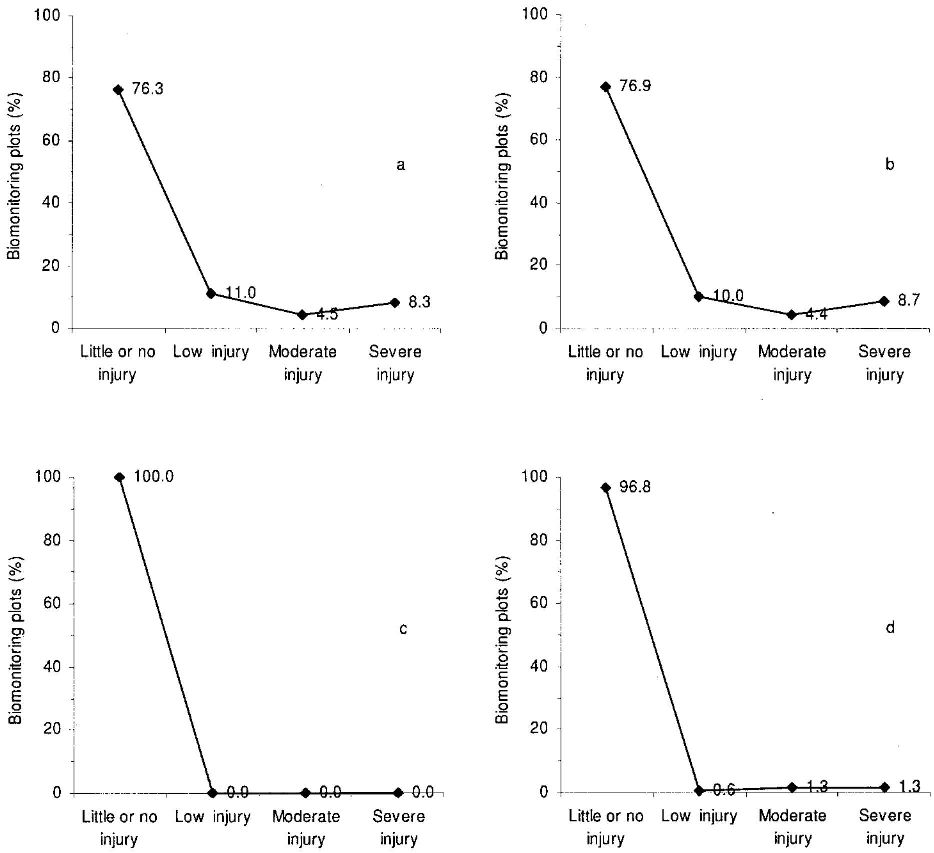


Figure 3. Frequency distribution of percent O_3 biomonitoring plots with levels of ozone injury in the (a) North, (b) South, (c) Rocky Mountain, and (d) Pacific Coast RPA region.

had significant increasing trends in O_3 index exposure for the 1994–2000 period (Table IV).

O_3 induced foliar injury to sensitive bioindicator species was recorded in all regions except for the Rocky Mountain region. In the North and South RPA regions, approximately 76% and 77%, respectively, of the biomonitoring plots received little or no O_3 injury (Figures 3a and b). In the Pacific Coast and Rocky Mountain RPA regions, 97% and 100%, respectively, of the biomonitoring plots had little or no injury from ambient levels of O_3 (Figures 3c and d). Approximately 8% of the biomonitoring plots in the North RPA region and 9% of the biomonitoring plots in the South RPA region had severe foliar injury (Figures 3a and b). Approximately 1% of the biomonitoring plots in the Pacific Coast RPA region were classified as having severe foliar injury.

3.2. Cluster analysis

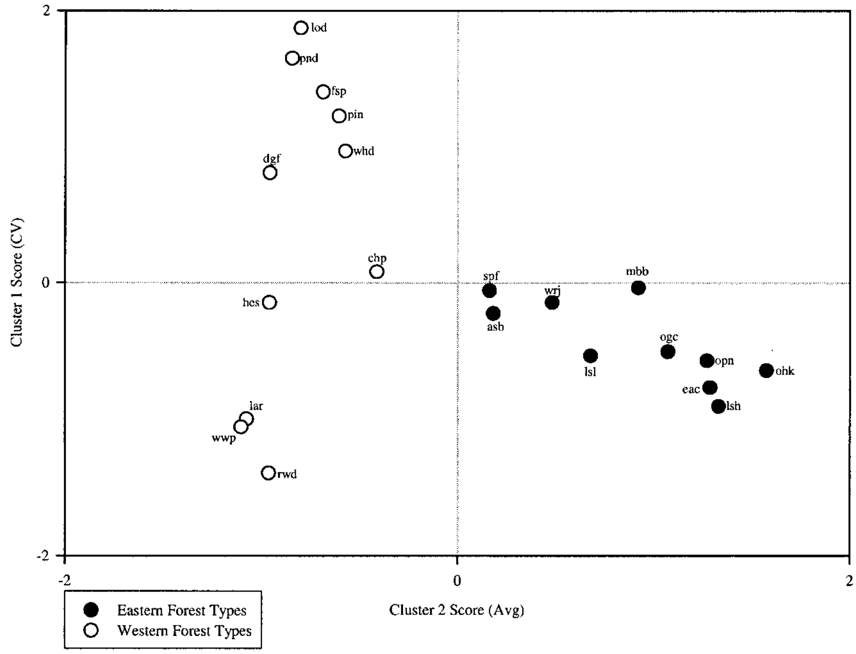
The initial ten air pollution indicators (average and CV of BI, O_3 index, NO_3^-, NH_4^+, and SO_4^{2-}) were reduced to two composite indicators, or clusters, using the VARCLUS procedure. Cluster 1 explained approximately 54% of the variance across forest types in the original 10 indicators and cluster 2 explained an additional 24.2% of the variance. No other clusters were used because their eigenvalues were less than 1. Cluster 1 was a linear combination of the within forest type CV of the air pollution indicators and cluster 2 was a linear combination of the average of air pollution indicators. These clusters were non-overlapping so variables included in calculating cluster 1 scores were not included in cluster 2 score calculations. Cluster 1 was interpreted as the relative within forest type CV of air pollution exposure while cluster 2 was interpreted as the relative air pollution exposure.

Cluster scores show relative differences between forest types based on the air pollution indicators. Forest types in the eastern United States had higher relative air pollution exposure scores (cluster 2 score) than western forest types (Figure 4). The oak-hickory forest type, which covers much of the North and South RPA regions (Figure 1), had the highest relative air pollution exposure score and had a relatively low within forest type CV score (cluster 1 score). Other eastern forest types with high relative air pollution exposure scores were loblolly-shortleaf pine, elm-ash-cottonwood, and oak-pine (Figure 1), each of which had relatively low within forest type CV scores (Figure 4).

Western forest types had lower air pollution exposure scores than eastern forest types but exhibited a wider range with regard to within forest type CV scores (Figure 4). Of the western forest types, chaparral had the highest relative air pollution score. Other western forest types such as western hardwoods, pinyon-juniper, fir-spruce, ponderosa pine, and lodgepole pine (Figure 1) had low relative air pollution exposure scores but relatively high within forest type CV scores, indicating that there may be pockets of high and/or low exposure. The larch, redwood, and western white pine forest types had both low relative air pollution scores and low within forest type CV scores. The larch and western white pine forest types have a small geographic range in northeastern Washington, northern Idaho, and northwestern Montana (Figure 1). The redwood forest type also has a small range and is limited to coastal areas of northern California.

4. Discussion

The goal of this assessment is to provide an overview of air pollution exposures in forested areas of the United States and to address the Montréal Process air pollution indicator to the extent possible. This analysis is experimental and there are several caveats. O_3 bioindicator data is the only information that directly addresses plant injury; however the data document the effects of ambient O_3 on bioindicator plants



Forest Type	Abbreviation
Chaparral	chp
Douglas fir	dgf
Fir-spruce	fsp
Hemlock-sitka spruce	hes
Larch	lar
Lodgepole pine	lod
Pinyon-juniper	pjn
Ponderosa pine	pnd
Redwood	rwd
Western hardwoods	whd
Western white pine	wwp

Forest Type	Abbreviation
Aspen-birch	asb
Elm-ash-cottonwood	eac
Loblolly-shortleaf pine	lsh
Longleaf-slash pine	lsl
Maple-beech-birch	mbb
Oak-gum-cypress	ogc
Oak-hickory	ohk
Oak-pine	opn
Spruce-fir	spf
White-red-jackpine	wrj

Figure 4. Relative air pollution exposure score (x-axis) and within forest type CV score (y-axis) for each forest type. Zero on either axis represents the average score across forest types. See Figure 1 for the geographic distribution of each forest type.

only. Furthermore, the temporal coverage of these data are inconsistent and this information is only available for 35 states (Table III). All other indicators are exposure estimates based on interpolated surfaces and forest area-weighted averages which is a particular problem with the O_3 index because monitoring stations tend to be near urban areas. For consistency we used a 3-month growing season O_3 index in this analysis however, other summary periods (e.g. 12-month) may be better suited for a particular location or species. No estimates of dry deposition are included in this report. This is problematic because dry deposition can contribute significantly to total deposition. Finally, the cluster analysis is restricted to areas in the 35 states where O_3 biomonitoring data were available and as a result, the relative air pollution exposure scores and within forest type CV scores are based on only the sampled portions of each forest type.

In the United States, NO_3^-, SO_4^{2-}, and NH_4^+ deposition and O_3 injury is highest in the North and South RPA regions. However, SO_4^{2-} and NH_4^+ deposition are decreasing in these areas. The oak-hickory forest type most often occurs in areas where high deposition rates and O_3 injury is found (Figure 4), but the regional influence of air pollution on this forest type is unknown. In the South, hardwood forests are considered less sensitive to nitrogen deposition because the soils have the capacity to retain deposited nitrogen and there are adequate base cation nutrients (NAPAP, 1998). The loblolly-shortleaf pine forest type also occurs in areas with relatively high air pollution (Figure 4) and O_3 may be of particular concern for this forest type (Taylor, 1994; Chappelka and Samuelson, 1998). Dougherty *et al*. (1992) suggested that at ambient levels of O_3 in the South, an average mature plantation grown loblolly pine tree has a 3% loss of gross primary production. In the North RPA region, sugar maple (*Acer saccharum*) and red spruce (*Picea rubra*) are considered sensitive to acid deposition and these two species are part of the maple-beech-birch and spruce-fir forest types, respectively (DeHayes *et al*., 1999; Horsely *et al*., 1999). Black cherry (*Prunus serotina*) is considered sensitive to O_3 (Krupa and Manning 1988; Chevone, 2001) and is considered likely to exhibit regional-scale O_3 impacts (Coulston *et al*., 2003). It is also a species found in the maple-beech-birch forest type group. The composite analysis in this report ranked the maple-beech-birch forest type sixth overall and the spruce-fir forest type tenth overall in relative air pollution exposure (Figure 4).

Based on the analyses in this report, western forest types (Pacific Coast and Rocky Mountain RPA regions) are generally exposed to less air pollution than eastern forest types (Figure 4). With respect to only western forest types, western hardwoods, pinyon-juniper, and chaparral had the highest relative air pollution exposure scores (Figure 4). The effects of air pollution on the western hardwood forest type are unknown, but pinyon-juniper forest types are considered O_3 tolerant. Nitrogen saturation was reported by Fenn *et al*. (1996) for chaparral watersheds in the San Bernardino Mountains of California. The forest types least likely to exhibit negative impacts due to high air pollution exposure were also found in the western United States. Based on this analysis, the redwood, western white pine and larch forest types are characterized by consistently low estimates of air pollution exposure (Figure 4).

There are several other concerns in the western United States. NAPAP (1998) identified Colorado Rockies Front Range ecosystems as nitrogen saturated. Takemoto *et al.* (2001) documented nitrogen saturation in selected areas of California's San Bernardino and San Gabriel mountains. Another major concern in California is O_3 (Miller *et al.*, 1996). California experienced the highest 3-month growing season O_3 index in the United States and mixed conifer forests in the Sierra Nevada have suffered stress from air pollution since the 1970's (Peterson and Arbaugh, 1992). Ponderosa and Jeffrey pine are particularly sensitive to O_3 in these areas. Increased susceptibility of Ponderosa pine to attacks by bark beetles resulting in reduced tree vigor due to long term exposure to O_3 was documented as early as 1968 (Cobb *et al.*, 1968). While the ponderosa pine forest type had a low relative air pollution exposure score it also had a high within forest type CV score (Figure 4). This implies that there are pockets of high air pollution exposure in this forest type. Other forest types such as lodgepole pine also fall under this scenario.

In this report, the O_3 bioindicator data are the only nationally consistent information on plant injury. O_3 injury to bioindicator plants was found, to varying degrees, in the North, South, and Pacific Coast RPA regions. We currently do not know if ambient levels of O_3 are reducing growth rates however, additional stress from ambient O_3 levels can open the door for several secondary stressors such as insects and pathogens as Smith (1974) suggests and Cobb *et al.* (1968) documents. The cumulative effects of air pollutants, such as O_3 and SO_4^{2-}, on forested ecosystems is not known but subtle changes create competitive advantages that in the long run may change the composition of forest ecosystems and their corresponding fauna (Kareiva *et al.*, 1993). There are several confounding factors such as interactions between pollutants that in some circumstances can mitigate the affects of the individual pollutants involved (Takemoto *et al.*, 2001) while in other cases impacts can be cumulative (Shan *et al.*, 1996).

We recognize that the information presented here represents only a portion of the influence that air pollution has on forested ecosystems. Indicators based solely on exposure information do not address the complexities of exposure-plant response relationships as influenced by exposure characteristics (e.g., turbulence), plant properties (e.g., stage of development), and external growth conditions. Further expansion of biomonitoring programs as well as the inclusion of biomonitoring data in models will improve the reporting of area and percent forest subjected to air pollutants that may cause negative impacts. Characterizing national air pollution trends provides an important contribution to the on-going assessment of forest health and vitality.

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