

Indicator 16: Area and percent forest land subject to specific levels of air pollutants (e.g., sulfates, nitrates, ozone) or ultraviolet B that may cause negative impacts on the forest ecosystem

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Abstract

There is clear evidence that a variety of 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 as part of the United States' 2003 Report on Sustainable Forests. We examined 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. However, ozone stress characterized in terms of ozone-induced foliar injury to sensitive species was recorded in only a small percentage of forested areas across all RPA regions. 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. Conversely, the redwood, western white pine, and larch forest types were generally exposed to less air pollution than all other forest types. These findings offer a new approach to national air pollution assessments and are intended to help direct research and planning initiatives related to air pollution and forest health.

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 through 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 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 and others, 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. The objective of this paper is to communicate the major results of the air pollution assessment.

The term “air pollution” encompasses a wide range of concerns, but acid deposition and ozone (O₃) are of primary concern for United States forests (Driscoll and others, 2001; Hakkarienen, 1987). Acid deposition comes, in part, from gaseous emissions of sulfur dioxide (SO₂), nitrogen oxides (NO_x), and ammonia (NH₃) that are deposited in wet form as sulfate (SO₄²⁻), nitrate (NO₃⁻), and ammonium (NH₄⁺) 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 and others, 2001) and tree nutrition (Houston, 1999; ESA, 1999), and direct effects such as foliar injury (DeHayes and others, 1999).

Tropospheric O₃ is produced from photochemical reactions between oxides of nitrogen (NO_x) and volatile organic compounds (VOC). It is a gaseous air pollutant that causes foliar injury (Hakkarienen, 1987; Miller and Millecan, 1971; Skelly and others, 1987; Treshow and Stewart, 1973), and is frequently measured at phytotoxic levels (Cleveland and Graedel, 1979; Lefohn and Pinkerton, 1988; Chappelka and Samuelson, 1998). Because plant uptake of tropospheric O₃ 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 O₃ uptake and plant response (McCool, 1998).

While there have been decreases in sulfur deposition because of the Clean Air Act, 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 and others, 1998). Wood chemistry analysis of several northern tree species suggests that increased sulfur concentration in trees is associated with increased SO₄²⁻ deposition (Riitters and others, 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).

Forest landscapes have highly variable pollution sensitivity and there are complex interactions among pollutants. O₃ sensitivity is variable among and within species. Some tree species are very sensitive to O₃ exposure (e.g. Prunus serotina: Krupa and Manning, 1988) but others (e.g. Acer saccharum; Renfro, 1992) are tolerant. Bennett and others (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 and others (2001) hypothesized that increased nitrogen supply from deposition could moderate the harmful effects of tropospheric O₃ 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 interaction between pollutants (e.g. Shan and others, 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 (fig. 1), and statistical composite summaries of those indicators are presented by forest type. This analysis does not identify specific negative impacts and does not attempt to establish cause-effect relationships.

Methods

Air quality and biomonitoring data were obtained from several sources (table 1). 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-hour SUM06 index for a 3-month growing season (June, July, August) nationwide. SUM06 was the sum of all average O_3 hourly concentration greater than 0.06 ppm. The U.S. Forest Service provided ozone injury information from a network of biomonitoring plots that evaluated field impacts using ozone-sensitive bioindicator plants including tree, woody shrub, and herb species. Data from biomonitoring plots was the only information about plant injury from air pollution.

We used three types of analysis to quantify air pollution exposure. The trends in exposure of forests to air pollution was assessed by RPA region for wet deposition of NO_3^- , SO_4^{2-} , NH_4^+ , and SUM06. The percent forest by RPA region exposed to specific levels of air pollution was estimated for NO_3^- , SO_4^{2-} , NH_4^+ , SUM06, and O_3 injury indicators (bioindicator data). A cluster analysis was then used to form composite air pollution indicators that were used to examine differences between forest types.

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-} ($\text{kg ha}^{-1} \text{ yr}^{-1}$), and SUM06 (ppm-hrs yr^{-1}) was calculated for each RPA region by $\Sigma(\underline{d}_m \underline{w}_m) / \Sigma \underline{w}_m$ where \underline{d} is the indicator of interest (e.g. NO_3^-), and \underline{w} is the weight (based on the km^2 of forest represented by each of \underline{m} monitoring stations). Estimates of average annual change for forested areas was calculated for each RPA reporting region using the

following general linear model: $D = a + b(y)$ where \underline{b} is the weighted estimate of annual change calculated by:

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

where D is the weighted average annual indicator value and y is the year (Steel and others, 1997). The probability that $\underline{b}=0$ was tested with the F-statistic and significance was assigned at the 0.05 level.

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-} ($\text{kg ha}^{-1} \text{ yr}^{-1}$), and SUM06 (ppm-hrs yr^{-1}) (Isaaks and Srivastava, 1989). For ozone injury, a frequency distribution was calculated directly from the ozone biomonitoring data.

For CDFs, the portion of forest below any cutoff value (\underline{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 \underline{c} . The percent forest below each \underline{c} in each RPA region across years was estimated by $(\sum A_c / 7) / \sum A$ where A_c is the amount (km^2) of forest below \underline{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 ozone biomonitoring data, where empirical frequency distributions of plot-level data were used to estimate the percent forest with ozone injury. The proportion of leaves with ozone 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, Coulston and others, 2003). BI was defined as:

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

where \underline{m} is the number of species evaluated, \underline{n}_j is number of plants of the j^{th} species, \underline{a}_{ij} is the proportion of injured leaves on the i^{th} plant of the j^{th} species, and \underline{s}_{ij} is the average severity of injury on the i^{th} plant of the j^{th} species. This index was classified into four risk categories that represent a relative measure of impacts from ambient ozone exposure (table 2).

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 3). We used the average BI for all measurements (1994 to 2000) in this analysis because it served as our best estimate of ozone induced foliar injury for each RPA region. It is justifiable to calculate frequency distributions of ozone injury for forested areas by RPA region directly from the biosite data because the sampling grid is a systematic sample of forested areas (White and others, 1992).

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, 1982). 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 SUM06 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), d_j is the distance from the j^{th} 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 ozone injury, BI values were used directly by giving all forested pixels within the hexagonal 65,000-ha area (White and others, 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. These maps represented our best estimate of air pollution exposure and foliar injury.

For the cluster analysis, only areas where estimates for all indicators were available were used. Both the average and coefficient of variation (CV) for each air pollution indicator by forest type were used. 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 index (1994 - 2000), SUM06 (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.

Results

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 $15 \text{ kg ha}^{-1} \text{ yr}^{-1}$ for the 1994 to 2000 period (fig. 2a). This was different from the Pacific Coast and Rocky Mountain RPA regions where

approximately 50% of the forest received less than $2 \text{ kg ha}^{-1} \text{ yr}^{-1}$ for the 1994 to 2000 period. Less than 1% of the forest in the South and North RPA region received less than $2 \text{ kg ha}^{-1} \text{ yr}^{-1}$ (fig. 2a). The maximum amounts of SO_4^{2-} deposition for the North and South RPA regions were approximately $38 \text{ kg ha}^{-1} \text{ yr}^{-1}$ while maximums for the Pacific Coast and Rocky Mountain RPA regions were approximately 12 and $19 \text{ kg ha}^{-1} \text{ yr}^{-1}$ respectively. Although average deposition rates were highest in the North and South, these regions had significant decreasing trends of 0.471 and $0.348 \text{ kg ha}^{-1} \text{ yr}^{-1}$, respectively (table 4).

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 $13 \text{ kg ha}^{-1} \text{ yr}^{-1}$ (fig. 2b). Fifty percent of the forested area in the South RPA region received more than $10 \text{ kg ha}^{-1} \text{ yr}^{-1}$. Deposition rates were lower in the Pacific coast and Rocky Mountain RPA regions. Approximately 50% of the forest in these areas received less than $2.5 \text{ kg ha}^{-1} \text{ yr}^{-1}$ for the 1994 to 2000 period (fig. 2b). In the North and South RPA regions, less than 3% of the forest received an average annual NO_3^- input of less than $5 \text{ kg ha}^{-1} \text{ yr}^{-1}$. The South RPA region was the only area where the rate of NO_3^- deposition had a significant decreasing trend of $0.16 \text{ kg ha}^{-1} \text{ yr}^{-1}$ (table 4). The North RPA region also had a decreasing trend ($0.187 \text{ kg ha}^{-1} \text{ yr}^{-1}$) 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 (fig. 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 $3.6 \text{ kg ha}^{-1} \text{ yr}^{-1}$. In the North and South RPA regions, approximately 69% and 86% of the forested areas received less than $3.6 \text{ kg ha}^{-1} \text{ yr}^{-1}$ of NH_4^+ deposition respectively (fig. 2c). Approximately 62% of the forest in the western U.S (Pacific Coast, Rocky Mountain RPA regions) received less than $0.7 \text{ kg ha}^{-1} \text{ yr}^{-1}$ 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 4).

On average, growing season SUM06 ozone was highest in the South RPA region (table 4) where only 10% of the forests were exposed to SUM06 concentrations less than $6 \text{ ppm-hrs yr}^{-1}$ (fig. 2d). SUM06 ozone was also high in certain forested areas of the Pacific Coast RPA region for the 1994 to 2000 period where 10% of the forest experienced exposures of greater than $41 \text{ ppm-hrs yr}^{-1}$ (fig. 2d). Forests in the North RPA region were generally exposed to lower concentrations of SUM06 (table 4). Approximately 75% of the forest in this area had SUM06 exposures of less than $24 \text{ ppm-hrs yr}^{-1}$ (fig. 2d) In the Rocky Mountain RPA region, approximately 50% of the forest had SUM06 exposure of less than $16 \text{ ppm-hrs yr}^{-1}$. Both the Rocky Mountain and South RPA region had significant increasing trends in SUM06 ozone exposure for the 1994-2000 period (table 4).

Little or no O_3 injury to plants was recorded on most biomonitoring plots across RPA regions. In the North and South RPA regions, approximately 77% of the biomonitoring plots received little or no ozone injury (fig. 3). 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 O₃. Only a small portion of plots had severe foliar injury and most of this was recorded in the North and South RPA regions (fig. 3). Approximately 1% of the biomonitoring plots in the Pacific Coast RPA region were classified as having severe foliar injury.

Cluster analysis

The initial ten air pollution indicators (average and CV of BI, SUM06, NO₃⁻, NH₄⁺, and SO₄²⁻) were reduced to two composite indicators, or clusters, using the VARCLUS procedure. Cluster 1 explained approximately 54.6% 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 significant (i.e. all other 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.

Estimated 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 (fig. 4). The oak-hickory forest type, which covers much of the North and South RPA regions (fig. 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, each of which had relatively low within forest type CV scores (fig. 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 (fig. 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 had low relative air pollution exposure scores but relatively high within forest type CV scores, indicating that there may be pockets 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 (fig. 1). The redwood forest type also has a small range and is limited to coastal areas of northern California.

Discussion

The goal of this assessment was 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. O₃ bioindicator data is the only information that directly addresses plant injury from air pollution, however this information is only available for 35 states and does not represent urban areas. Furthermore, the temporal coverage of these data are inconsistent (table 3). All other indicators are exposure estimates based on interpolated surfaces and forest area-weighted averages which is a particular problem with the SUM06 indicator because monitoring stations tend to be near urban areas. 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 ozone biomonitoring data were available and as a result, the relative air pollution exposure scores and within forest type CV scores are based on sampled portions of each forest type.

In the United States, NO₃⁻, SO₄²⁻, and NH₄⁺ deposition and O₃ injury is highest in the North and South RPA regions. However, SO₄²⁻ and NH₄⁺ deposition are decreasing in these areas. The oak-hickory forest type most often occurs in areas where high deposition rates and O₃ injury is found, but the regional influence air pollution has 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 and O₃ may be of particular concern for this forest type (Taylor, 1994; Chappelka and Samuelson, 1998). Dougherty and others (1992) suggested that at ambient levels of O₃ in the South, an average mature plantation grown loblolly pine tree has a 3% loss of gross primary production.

In the Eastern United States there are documented effects of air pollution to red spruce (*Picea rubra*) and sugar maple (*Acer saccharum*). DeHayes and others (1999) hypothesized that acid deposition has decreased the cold hardiness of red spruce by leaching cellular Ca from foliage which has led to mortality of canopy trees. Horsely and others (1999) found that the depletion of Ca²⁺ and Mg²⁺ cations has contributed to sugar maple mortality in Pennsylvania. It was interesting that the composite analysis in this report ranked the maple-beech-birch forest type sixth, and spruce-fir forests tenth overall in the relative air pollution exposure score. However, our analysis examined forest types and did not take into account elevation effects, soil buffering capacity, or species-specific responses.

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 (fig. 4). With respect to only western forest types, western hardwoods, pinyon-juniper, and chaparral had the highest relative air pollution exposure scores (fig. 4). The effects of air pollution on the western hardwood forest type are unknown, but pinyon-juniper forest types are considered ozone tolerant. Nitrogen saturation was reported by Fenn and others (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. These forest

types are redwood, western white pine and larch. Based on this analysis, each of these forest types are characterized by consistently low estimates of air pollution exposure.

There are several other concerns in the western United States. NAPAP (1998) identified Colorado Rockies Front Range ecosystems as nitrogen saturated. Takemoto and others (2001) documented nitrogen saturation in selected areas of California's San Bernardino and San Gabriel mountains. Another major concern in California is O₃ (Miller and others, 1996). California experienced the highest 3-month growing season SUM60 ozone concentrations in the United States and mixed conifer forests in the Sierra Nevada has suffered stress from air pollution since the 1970's (Peterson and Arbaugh, 1992). Ponderosa and Jeffrey pine are particularly sensitive to O₃ in these areas. Increased sensitivity of Ponderosa pine to bark beetles resulting from air pollution was documented as early as 1968 (Cobb and others, 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. 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.

O₃ bioindicator data are the only nationally consistent information on plant injury from air pollution in the field and this analysis shows that plant injury is not observed everywhere. It is well documented that sulfur, nitrogen, and O₃ can be important stressors to forest ecosystems in particular areas and there are numerous reports describing these impacts (e.g. Takemoto and others, 2001; Bennett and others, 1994). While tree growth is not negatively affected in most cases, the additional stress can open the door for several secondary stressors such as insects and pathogens as Smith (1974) suggests and Cobb and others (1968) documents. Furthermore, the cumulative effects of air pollution on forested ecosystems are not known. Subtle changes can create competitive advantages that in the long run may change the composition of forest ecosystems and their corresponding fauna (Kareiva and others, 1993). In addition, there are interactions between pollutants that in some circumstances can mitigate the affects of the individual pollutants involved (Takemoto and others, 2001) while in other cases impacts can be cumulative (Shan and others, 1996).

Ultra-violet beta (UV-B) radiation can damage plants but is not included in this report because the USDA UV-B monitoring program is relatively new and most of the current efforts are on data calibration and quality control. UV-B occupies wavelengths between 290 and 320 nanometers in the electromagnetic spectrum and when plants absorb photons in this range, damage to DNA and the photosynthetic system can occur (Sullivan and Rozema, 1999). Secondary effects, such as changes in chlorophyll content, stomatal density, and reduction in leaf area may also occur. Sullivan and Rozema (1999) also suggest that UV-B will not cause substantial damage to the photosynthetic system or loss of productivity in most species but the secondary effects have major consequences on ecosystem process. Future reports will include information on UV-B as the monitoring program and indicators develop.

There are several other topics associated with air pollution as a forest stressor. Drought plays a major role in the effect air pollution has on trees. Under droughty conditions,

plants close their stomates to reduce transpiration and water loss. This limits the amount of O₃ injury in dry years, an effect that probably influenced the BI values used in this report (Smith and others, 2003). Forest fragmentation can also influence the effect that air pollution has on forest trees (Weathers and others, 2000). The canopy in a forest ecosystem can serve as an air filter to trees in the lower canopy or understory (Treshow and Stewart, 1973; Skelly and others, 1996). High forest fragmentation increases the amount of tree surface area possibly exposed to ambient O₃ levels because less forest area is protected by the canopy. Sulfur and nitrogen deposition have obvious relationships with soil properties and forest productivity. Sulfur and nitrogen inputs can change the chemical properties of forest soils through cation leaching, accumulation of nitrogen and sulfur in the soil, and can increase aluminum mobility. However, the depletion of base cations and release of acid cations ultimately depends on the soil's ability to neutralize acid inputs.

Finally, we recognize that the information presented here represents only a portion of the influence 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. Here we use a statistical approach to address air pollution, however, there are also opportunities to develop national scale mechanistic models and examine critical load approaches as more information becomes available. Future efforts to address the Montréal Process indicators of air pollution will integrate information on UV-B and dry deposition. Characterizing national air pollution trends provides an important contribution to the on-going assessment of forest health and vitality. It remains a challenge to report and integrate national level exposure information with a biological interface such that results are interpretable and useful to the applied concepts of sustainable forest management.

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Table 1. 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.

Table 2. Biosite index categories, risk assumption, and possible impact.

Biosite Index Category	Assumption of Risk	Possible Impact
1. Biosite index < 5 Little or no foliar injury	None	Tree-level response Visible injury to leaves and needles
2. Biosite index 5.0 to <15 Low foliar injury	Low	Tree-level response Visible and invisible injury
3. Biosite index = 15 to < 25 Moderate foliar injury	Moderate	Tree-level response Visible and invisible injury
4. Biosite index >25 Severe foliar injury	High	Structural and functional changes Visible and invisible injury

Table 3. Implementation of ozone biomonitoring program in the United States.

<u>First year of monitoring</u>	<u>State</u>
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

Table 4. Average and Average annual change of wet sulfate, nitrate, and ammonium deposition ($\text{kg ha}^{-1} \text{ yr}^{-1}$) and SUM06 (ppm-hrs yr^{-1}) by RPA region for the 1994 to 2000 period.

	North	Pacific Coast	Rocky Mountain	South
Sulfate ($\text{kg ha}^{-1} \text{ yr}^{-1}$)	15.4 (-0.471 *)	2.6 (-0.014)	2.5 (-0.094)	15.1 (-0.348 *)
Nitrate ($\text{kg ha}^{-1} \text{ yr}^{-1}$)	13.2 (-0.187)	2.9 (-0.025)	3.5 (0.006)	10.4 (-0.16 *)
Ammonium ($\text{kg ha}^{-1} \text{ yr}^{-1}$)	2.8 (-0.049 *)	0.7 (-0.024)	0.8 (-0.013)	2.4 (-0.066 *)
SUM06 (ppm-hrs yr^{-1})	15.2 (-0.006)	13.9 (0.309)	16.0 (1.735 *)	20.4 (2.648 *)

Figure Captions

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

Figure 2. Cumulative distribution functions of average percent forest subjected to levels of (a) wet sulfate deposition (1994-2000), (b) wet nitrate deposition (1994-2000), (c) wet ammonium deposition (1994-2000), and (d) SUM06 ozone (1994-2000).

Figure 3. Frequency distribution of percent ozone biomonitoring plots with levels of ozone injury.

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.