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Assessing Ozone Effects on Plants Native to the Southeastern United States

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Abstract

For the past six years, the U.S. National Park Service, U.S. Environmental Protection Agency, and University researchers have been documenting the effects of ozone on a large number of woody and herbaceous species native to the southeastern United States. In Great Smoky Mountains National Park (GRSM), ozone levels exhibit diel patterns at low elevations, where concentrations are low in the morning and high in the afternoon. At high elevations (> 800 m), morning concentrations are much higher, and the total daily exposure is approximately twice that at the lower elevations. Putative ozone injury has been observed in the field in GRSM on 90 species, representing approximately 6% of the known flora in the Park. Surveys of foliar injury on several tree species show a general pattern of increasing frequency and amount of stipple with increasing elevation in GRSM, and in nearby Shenandoah National Park. Tall milkweed (*Asclepias exaltata* L.) and black cherry (*Prunus serotina* Ehrh.) appear to be among the most sensitive herb and woody species, respectively, in GRSM. Exposure-response studies were carried out in open-top chambers for six years with 46 species. Foliar symptoms seen in the field, were reproduced on 30 species, providing evidence that the foliar injury found in the field was probably due to ozone exposure. The exposure-response studies indicated that early-successional and shade intolerant species were less resistant to high ozone levels than late-successional species and dry-site adapted conifers. Geographic information systems (GIS) techniques, coupled with growth models, were used to scale-up responses from the seedling to the geographic level. Exposure-response data were coupled with species' distributions and geographic patterns of ozone exposure to predict growth losses for several woody species. These data suggest that species such as black cherry may suffer yearly growth losses of up to 32%, whereas less sensitive species, such as red maple (*Acer rubrum* L.) may exhibit losses of only 2-4%, even though they share a similar geographic range. In addition, some species, such as trembling aspen (*Populus tremuloides* Michx.) show genetic variation in tolerance to ozone. When growth data for species are projected over a typical 70 year life span, small yearly losses can be compounded, resulting in over 50% growth losses, as is the case predicted for loblolly pine (*Pinus taeda* L.). Because the modeling work is totally dependent on the quality of the exposure-response data, future work must concentrate on more realistic exposure systems (i.e. chamberless systems, plants rooted in the ground), genetic variation within species, and refinement of the modeling procedures used in scaling up.

[1]Introduction

Ozone is regarded as the most widespread phytotoxic air pollutant in the United States, and an assessment of its impacts on plant growth and ecosystem functioning is necessary for setting air quality standards. Currently, the National Ambient Air Quality Standard (NAAQS), as set by the Clean Air Act of 1977 (U.S. EPA 1986), mandates that hourly ozone levels in an airshed not exceed 120 ppb more than three times in three years. Many urban areas often exceed this level, and must take measures to bring the levels down. However, recent observations in natural areas throughout North America, where ozone concentrations are below NAAQS, are showing many plants exhibiting foliar symptoms suggestive of ozone injury (Neufeld *et al.* 1992). This suggests that the primary NAAQS may be inadequate for protecting certain plant species, and that perhaps either the primary or secondary standards could be modified. Currently, the secondary standard is being modified to reflect recent studies of the effects of ozone on woody plants (W. Hogsett, U.S. Environmental Protection Agency, pers. comm.).

Nearly two-thirds of the population of the United States lives east of the Mississippi River (U.S. Bureau of the Census, 1991). A night view of North America shows the high density of lights in the eastern half of the nation, which is highly correlated with population estimates (Figure 1). Not only do these lights reflect population density, they also act as surrogate indicators of energy consumption. Since fossil fuels are the primary energy source, these lights are also good visual estimators of the production of air pollutants, particularly mobile sources, such as cars, and includes nitrogen oxides, SO_x and volatile organic carbon compounds.

Most Federal natural areas in the United States are located in the western half of the nation (Figure 2). The Clean Air Act (CAA) amendents of 1977 define wilderness areas and national parks as Class I areas if they were formed prior to the act and are over 6,000 acres in size. The CAA specifies that the ozone in these areas must remain below the 120 ppb standard to maintain a high degree of air quality. These areas are afforded the greatest degree of protection under the CAA. Although relatively few key Class I areas are located in the eastern United States, they are potentially at great risk due to high pollutant loadings. The most notable areas are Acadia National Park in Maine, Shenandoah National Park in Virginia, Great Smoky Mountains National Park (GRSM) in Tennessee and North Carolina, Okefenokee Wildlife Refuge in Georgia, and Everglades National Park in southern Florida. Great Smoky Mountains National Park is the most visited park in the nation, drawing some 8-10 million visits each year (Peine and Renfro 1988), and has some of the highest levels of biological diversity in the southeast (White, 1982). In addition, there are numerous rare and endangered species that occur only within the boundaries of

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[1]Some of the research described herein was developed by Deborah Mangis while an employee of the National Park Service. It was conducted independent of EPA employment and has not been subjected to the agency's peer and administrative review. Therefore, the conclusions and opinions drawn are solely those of the author and should not be construed to reflect the views of the EPA.

GRSM which might be at risk either to pollutants directly, or secondarily because of ecosystem degradation due to pollutants. These Federal areas are of concern to the U.S. National Park Service and U.S. Fish and Wildlife Service because of their potential vulnerability to air pollutants.

The success of any air quality program depends not only on determining pollutant loadings in critical areas, but also on determining the source locations for these pollutants. Identification of pollution sources enables regulators to apply control measures specified in the Clean Air Act for maintaining good air quality. Air pollutants may drift hundreds of miles before they reach natural areas. For example, when ozone levels in GRSM are high, the air containing that ozone can be shown to have originated either over the industrial Ohio Valley, or to have traveled northward from industrial areas in Louisiana, Alabama, Georgia and Tennessee (Dattore *et al.* 1991). Even in fairly remote areas, it is common to see pollutants accumulate at the inversion layer, such as the one shown in Figure 3, which is on Whitetop Mountain in southwestern Virginia, one of the most rural areas in the eastern United States.

This paper deals primarily with the effects of ozone on native plants in GRSM, but because many of these species are geographically widespread, has important implications for plants throughout the southeastern United States. We concentrate on this particular geographic area because it is subject to chronically high ozone levels (60-100 ppb) during the growing season, and because of the importance of forests and native plants to this region. The southeast is prone to high ozone because of several reasons. First, as mentioned above, polluted air drifts into the region from other areas. Second, the region has abundant sunshine, high temperatures, and a large number of days with stagnating air masses. Together, these factors contribute to the rapid and abundant formation of ozone (Lefohn 1992). Thus, despite relatively low population figures in comparison to the more densely settled northeast, the southeast is at a relatively greater threat from ozone than would be predicted from population estimates alone.

For the last six years, the Air Quality Division of the U.S. National Park Service has been funding research into the effects of ozone pollution on plants native to GRSM. This research had several goals: 1) to survey plants in the field for signs of putative ozone injury, 2) to determine relative sensitivities of plants to ozone injury, based on both foliar symptoms and growth reductions, and 3) to document whether the injury observed in the field was indeed due to ozone exposure. For this, an ozone exposure facilty was set up at the Uplands Field Research Laboratory, located at Twin Creeks in GRSM. Plants were exposed to varying levels of ozone and foliar symptoms, if any, were noted. Additionally, the facility was used to generate an extensive set of exposure response curves for biomass accumulation, which are currently being used in conjunction with the foliar data to rate species' sensitivities to ozone. Finally, the exposure data, along with the field data, are being used to estimate the potential threat of ozone to the plants and ecosystems of the Park.

Models are being developed to predict ozone in remote locations, where monitoring equipment is not available, and then used in conjunction with plant growth models to scale up responses from seedling level to large-scale geographic patterns.

The intent of this paper is to present a synopsis of our work in the southeastern United States regarding ozone effects on native plants, and to suggest possible future avenues of research in this area. In addition, we outline a preliminary attempt to scale up responses from seedlings, to mature trees, to stands, and finally to the regional geographic level.

Methodologies

Plants in GRSM have been observed during the summers of 1987-1993 for foliar symptoms consistent with those known to be caused by ozone. For the past three years, additional effort has been made using trend plots to survey dominant canopy trees at high and low elevations. Trend plots have no fixed boundaries, rather, the criteria are that sample trees be located within 152 meters elevation and 3.3 km of stationary ozone monitors. These monitors are located at Cove Mountain (1264 m elevation), Look Rock (863 m elevation), and Twin Creeks (594 m elevation). Sampling large trees has entailed the use of tree climbers to collect foliage from the crown. Data from these plots allows researchers to identify geographic spatial variation in ozone injury within GRSM. In addition to the trend plots, observers have walked or driven along trails and roads looking for injury symptoms. These visual surveys are always carried out from mid- to late-summer when symptoms of ozone injury are most likely to be seen. Observers look for any species that shows symptoms that might be due to ozone, and only presence or absence of injury is recorded. Slightly greater than 50% of the trail system in GRSM has been surveyed. Because observers change from year to year, extensive training sessions and quality control exercises are done to minimize human error in the assessments. Voucher specimens showing putative injury are located in the herbarium at the Park.

The ozone exposure system started out as a nine chamber system in 1987, and with additional funding from the U.S. Environmental Protection Agency (U.S. EPA), expanded to 15 chambers in 1990. Frustums and raincaps were also added in 1990 to better distribute the ozone within the chambers, and to prevent rain damage from thunderstorms.

Ozone was generated by an electric spark discharge generator. Ambient air was used to generate ozone in 1987 and 1988, but liquid oxygen was used thereafter to prevent formation of oxides of nitrogen (Brown and Roberts 1988). Ozone was dispensed 24 hours per day, seven days per week. This was done because high ozone levels have occassionally been observed at night, and without a 24 hour fumigation, these peak events would have been missed.

During 1987 and 1988, the ozone treatment levels consisted of 7-day exposure profiles (developed from data at the Look Rock site), with one simulating ambient conditions at Look Rock, and the other 2.0x ambient. Control chambers had charcoal filters to reduce the ozone levels below ambient. In 1989, modified ambient exposures were used, and the treatments were: charcoal-filtered, 1.0x, 1.5x, and 2.0x ambient, as well as open ambient plots. In 1990, a 0.5x treatment was added. Ozone levels were continually adjusted by a datalogger throughout the day approximately five times per hour so as to more closely track diurnal patterns. There were three replicate chambers for each ozone treatment for 1987 and 1988. In 1989 treatment replications were reduced to two, except for the 2.0x and open plots. Beginning in 1990, all treatments were replicated three times. Generally, about 10 plants per species were placed in each chamber, but for difficult-to-grow species, there were sometimes fewer than this number. Plants were arranged within the chambers to minimize shading by taller species.

Ozone concentrations in the chambers were monitored by TECO Model 49 analyzers. Teflon tubing and filters were used throughout the system, and line losses were consistently less than 5%. All monitoring equipment was checked quarterly by state and national auditors, and in all cases the monitors were within compliance for precision and accuracy. Data were stored in a Campbell 21x datalogger, and downloaded to a computer for final storage twice daily.

Plants were raised from seed collected in the field within GRSM, and were generally first year or one year old seedlings at the time of exposure. Seedlings were grown in pots of varying sizes, depending on species, in Pro-mix soil-less media, watered to excess daily, and fertilized once weekly with a 20-20-20 water soluble fertilizer. For certain tree species, slow-release Osmocote fertilizer was used instead.

Where possible, height, diameter, biomass, leaf count and area were obtained at the end of the exposure period. Most species were exposed for a single growing season (typically from some time in May to late August or September), but some woody and herbaceous species were exposed for more than one growing season. Table mountain pine *(Pinus pungens* Lambert), and Canadian hemlock (*Tsuga canadensis* L. Carr.) for example, were exposed for three consecutive seasons.

The data from the exposure experiments were subjected to regression analysis, covariance analysis, and Chi-square analysis depending on the variable being analyzed. The Weibull function was used to model responses to ozone exposure if the analysis of variance indicated a significant non-linear trend. Details of the statistical treatments can be found in Neufeld *et al.* (1992) and Neufeld and Renfro (1993).

Results and Discussion

Ozone Dynamics

Diel (day and night) concentrations at low elevation sites show a typical pattern of low concentrations in the morning hours, with higher concentrations in the afternoon (Figure 4). This happens because ozone is formed through the interactions of light, nitrogen oxides, and volatile organic compounds (Chameides and Lodge, 1992). Because of the diurnal patterns of light, and the time constants involved in the reactions, it takes several hours for the ozone to build up each day. During the night, ozone concentrations decrease because the inherently unstable molecule breaks down and is scavenged by various sources, mainly NO_x. In contrast, at high elevations, early morning ozone concentrations are high, and there is much less contrast between afternoon maxima and morning minima. This results when low elevation inversions trap ozone at higher elevations, when there is a lack of NO_x, and if stratospheric intrusions occur (Wolff *et al.* 1987). In GRSM, on average, ozone exposures at high elevations (> 800 m) (obtained by integrating under the daily concentration curves) run about twice that at low elevations ($\leq$ 800 m) in the Park. Thus plants at high elevations are potentially at greater risk due to the higher exposures.

In addition to the different diel patterns, higher ozone concentrations are more frequent at the upper elevations in the Park. The majority of ozone concentrations are between 40-70 ppb at high elevations, whereas at low elevations, they are between 20-40 ppb (Figure 4). Much of the difference between the two elevations is due to the low nighttime values at lower elevations.

Because ozone is highly reactive, it can be scavenged out of the air by coming into contact with surfaces, such as soil, tree trunks and leaves. Fully 75% of GRSM is closed canopy hardwood forest. Using a portable ozone monitoring station (Neufeld *et al.* 1992), we found that at the ridge-tops, where the stationary ozone monitors are located, and where forests are of smaller stature and lower basal area, depletion of ozone through the canopy is at most only about 20% at 1 meter from the forest floor (Figure 5). In contrast, ozone 1 meter above the ground in closed-canopy cove hardwood forests can be depleted to less than 50% of that above the canopy. Thus, even though the upper canopy trees in cove hardwood forests may be experiencing high ozone, plants in the understory are relatively protected. These microsite and topographic influences make it difficult to model ozone distributions in areas with complex relief and diverse vegetation types.

Foliar Symptoms

To date, 90 species of plants have been observed to exhibit putative ozone symptoms in the field in GRSM. A table of species and their foliar sensitivities can be found in the appendix at the end of this paper (from Neufeld *et al.* 1992). This represents approximately 6% of the total known flora in GRSM, and given the small amount of land area covered by the surveys, suggests that there may remain other species which are sensitive to ozone. Tall milkweed, (*Asclepias exaltata* L.) appears to be the most sensitive herbaceous species. Complete defoliation has been observed in the field, and has occurred in areas where the maximum concentrations have not exceeded 70 ppb. The symptomatology prior to defoliation is suggestive of ozone injury. The most sensitive tree species appears to be black cherry (*Prunus serotina* Ehrh.), and foliar symptoms on this species have been found in the field throughout the Park (Chappelka *et al.* 1992).

In our surveys of mature trees, using the tree climbers, foliar injury in 1991 and 1992 followed a general pattern of increasing frequency and amount with increasing elevation in GRSM (Chappelka *et al.* 1992). A similar pattern was noted in Shenandoah National Park (J.M. Skelly, Dept. of Plant Pathology, Pennsylvania State University, pers. comm.), which is about 500 km to the north. Since higher elevation sites were sampled earlier than low elevation ones, this trend seems due more to ozone than to seasonality.

Exposure-Response Results

We have fumigated a total of 46 species to date in the chambers. A list of the species tested can be found in Neufeld *et al.* (1992). Of these 46, 35 had shown foliar injury in the field. Among these 35, 30 showed symptoms in the chambers like those found in the field, lending credence to our hypothesis that the injury symptoms seen in the field were due to ozone.

As mentioned earlier, tall milkweed appeared hypersensitive in the field, and proved to be so in the exposure chambers as well. After four weeks in the 2.0x treatment, there was premature sensescence of the leaves, and after six weeks, many of the plants had died. Only plants grown in charcoal-filtered air retained all their leaves throughout the exposure period (Figure 6). The large loss of leaves in the open plots relative to the 1.0x chambers is most likely a result of strong winds and rain in the former.

Black cherry exhibited large, statistically significant biomass responses to increasing ozone. As with tall milkweed, the most obvious response was discoloration, and stipple on older leaves, followed by premature senescence. This led to more than 50% losses in leaf number and area in the 2.0x treatment (Figure 7).

These losses in leaf material contributed significantly to large biomass losses, particularly for leaves and roots. At 2.0x, total biomass was reduced by up to 32% in comparison to the charcoal-filtered treatment, as estimated by the Weibull function (Figure 7). For sycamore, (*Platanus occidentalis* L.) there was severe defoliation of the lower leaves at 2.0x (Figure 8), but interestingly, no change in final leaf count, presumably because this species compensated for the lower leaf loss by producing more new leaves. In addition, there were no effects on height or diameter, and only a marginal effect on total biomass. Finally, the most resistant species appeared to be late successional species, and dry-site adapted conifers. Table mountain pine (*Pinus pungens* Lambert), which was exposed for three consecutive seasons, showed severe foliar injury on two year old needles at 2.0x, but did not exhibit a significant biomass response. This illustrates an important point, namely, that foliar injury and biomass responses are not highly correlated. Growth reductions can occur in the absence of injury (Hogsett *et al.* 1985), and vice-versa (Neufeld *et al.* 1992). In addition, it points to an important modifier of ozone susceptibility, namely drought. Because ozone enters leaves primarily through the stomata, any environmental stress that causes stomatal closure is likely to reduce ozone uptake and injury. For plants growing in the field, drought is the most likely cause of reduced stomatal aperture. Thus to accurately rate the sensitivity of a species requires documentation of both foliar and growth responses, as well as the modifications induced by environmental change, particularly drought.

Spatial Risk Characterization[2]

Characterizing the spatial risk that ozone poses for forested areas requires knowledge of the ozone exposure characteristics over the appropriate geographic area (watershed, region, national), quantification of the phytotoxic effects at the appropriate biological level (species, population, community), and the types of uncertainty associated with these effects. It also includes recognizing the perspectives by which cultural and societal needs dictate the definition of "risk". For the purposes of this paper, risk is associated only with development of damage, where damage means an impairment of growth. It is implicitly assumed that damage is correlated with altered community and ecosystem functioning, and that such changes represent a cost to society as a whole. Therefore, such risk assessments can be useful in setting guidelines for secondary air quality guidelines within the criteria of the Clean Air Act for protecting forest resources.

A first attempt was made several years ago to estimate ozone risks for crops under the NCLAN program (National Crop Loss Assessment Program) and the results

[2]The spatial characterization section in this paper is adapted from Hogsett *et al.* (1993) and appropriate authorship should be attributed to them for the content contained therein.

suggest that current ambient levels of ozone cause anywhere from 11-14% reductions in crop yields (Tingey *et al.* 1993). However, similar risk assessments for natural areas, and forests in particular, are lacking due to the difficulty of working with large trees, and organisms with long lifespans.

The risk characterization presented in this paper uses GIS (Geographic Information Systems) to integrate empirical growth data from ozone-exposure response studies and model simulations of long-term growth trends with spatial estimates of ozone exposure and species distributions in order to generate a spatial assessment of ozone impact. In this preliminary assessment, we use the eastern United States to illustrate the approach, mainly because there are more data available for species in this geographic region, and the ozone exposures are higher. The eastern United States contains 10 different forest types (Eyre 1980) and a substantially large number of species (over 100 in GRSM alone). The assessment is based on the adverse effect on productivity and species assemblages. We have used only limited empirical data on seedling exposure-responses of eight different species.

The spatial distribution of ozone exposures was interpolated from stationary monitoring sites located in the eastern United States, using a model for the formation and degradation of ozone (Hogsett *et al.* 1993). The model uses environnmental variables such as temperature, wind direction, NO_x formation, stagnation and radiation values to estimate ozone formation, drift, and degradation. This results in a contour map showing regions of similar ozone exposure. For model simulations, the exposures are routinely standardized to a three month summer exposure period. GIS technology is used to generate the maps, using grids 20 km on a side. A total of 140 north-south cells and 135 east-west cells (18,900 total) were used (Figure 9).

Seedling ozone exposure response data are then used to generate species specific responses to different ozone exposures. These can be overlain on the GIS maps of ozone exposure to generate spatial patterns in predicted biomass responses. A tree growth model, in this case TREGRO (Weinstein *et al.* 1992) is used to "grow" species over standardized lengths of time to generate the necessary data for the spatial modeling effort. Eventually, it should be possible to pass these data on to a stand growth model, i.e., ZELIG (Urban 1990), to simulate biomass responses at the next higher level of organization and to estimate multispecies responses. Finally, the stands can be "grown" over long time intervals to estimate the potential effects of ozone on forests themselves. The decreases in biomass over these time intervals can then be used as one of several inputs for spatial risk characterization. Details of the modeling efforts can be obtained from Hogsett *et al.* (1993).

Results of the Modeling Efforts

Exposure-response data were obtained for eight different tree species (see Figure 12 for the list). The SUM06 exposure index was used to calculate the response curves. This is simply the sum of the concentrations for those hours when the ozone exceeded 60 ppb (Hogsett *et al.* 1988). The Weibull function was used to model the growth responses (Rawlings and Cure 1985), and maps of the spatial distribution of each species were generated using the GIS. The maps of ozone exposure and distribution were then superimposed, and the response functions used to predict percent biomass losses relative to charcoal-filtered air at each point on the map. Interpolation techniques allowed predictions for areas where there were no ozone data available.

Black cherry is predicted to have the greatest losses due to its high sensitivity (Figure 10). In contrast, red maple (*Acer rubrum* L.), which has a very similar distribution, is predicted to have losses of less than 2% in 1988, a high ozone year (Hogsett *et al.* 1993). This is because it is inherently less sensitive to ozone than black cherry (Neufeld *et al.* 1992). There are both year to year variations in losses, due to different amounts of ozone each year, and there are genotypic variations within species. For example, losses of greater than 30% are predicted for black cherry over 90% of its range in 1988 (Figure 10), but in the low ozone year of 1989, less than 10% of the area was predicted to have losses over 20%. In Michigan, studies were carried out in a similar fashion to those in GRSM, and for one species of aspen (*Populus tremuloides* Michx.) different genotypes were compared for their response to ozone. Based on the results of these studies, one clone is predicted to have much greater losses than another clone, even though they can be found growing over the same geographic range (Figure 11). Looking at all the species surveyed, the predicted area-weighted biomass losses vary widely, ranging from 0-35% depending on species and year (Figure 12).

When NCLAN data are used to predict annual agricultural crop losses over a standardized three month growing season, and using the SUM06 as the index, 50% of the crops would suffer little or no loss up to 26.4 ppm*hrs exposure (Tingey *et al.* 1991). For trees, the data suggest that at 20 ppm*hrs, 50% of the species would suffer annual losses of 4%. This is a small number, but if the losses are compounded over a portion of the lifespan of a tree (40-80 years), the cumulative losses become substantial. In fact, over this time span, losses may exceed 50%, even in relatively insensitive species such as loblolly pine (*Pinus taeda* L.) (Figure 13).

Conclusions

This risk assessment characterization is still in the preliminary stages. More data are needed on environmental interactions, and how they affect exposure response curves. We need to study temperature, drought, light, and humidity interactions, as well as soil factors such as fertility and type (Pye 1988). Furthermore, we must ascertain whether exposure response curves from studies using potted seedlings are useful for scaling up to sapling and mature tree responses. Increasingly, global climate change must be factored in, particularly the potential influence of rising CO_2 levels. And finally, the genetic and physiological bases of resistance to ozone require much more study (Roose 1991).

Taken in context, the data suggest potential risk at current levels of ozone to plants in natural areas of the southeastern United States. Only further intensive study can determine what the ecological, sociological, economic and cultural risks are to the United States.

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Figure Legends

1. A night view of North America, showing the lights from major urban areas. Photo courtesy of NASA.

2. Mandantory Class 1 Areas in the United States.

3. View from Whitetop Mountain, Virginia, showing accumulation of pollutants in the inversion layer. Photo by H. Neufeld.

4. Diel ozone patterns for one day in Great Smoky Mountains National Park. Twin Creeks is a low elevation site (594 m), while Look Rock (823 m) and Cove Mountain (1264 m) are ridge top sites.

5. Mean ozone concentrations at 3 heights above the forest floor for several sites in GRSM expressed as a percentage of the value above the canopy at Look Rock, a mid-elevation (823 m), ridge-top site. The first three stations on the left side of the graph are ridge sites, and the remaining three stations are closed canopy forests.

6. Final leaf number at the end of the season for tall milkweed (*Asclepias exaltata* L.) as a function of ozone treatment. Treatments (along with their SUM0 exposures in ppm*hrs) are: CF (charcoal-filtered, 7.4), 1.0x ambient (20.5), 1.5x ambient (30.2), 2.0x ambient (44.0), and open plots (non-chambered, 26.3).

7. Response of black cherry seedlings to ozone fumigation in 1989: a) leaf area, b) height (○) and diameter (●), and c) dry weight accumulation for various plant parts: (▽) - total dry weight, (●) - root dry weight, (○) - leaf dry weight. Solid lines are best fits obtained using a Weibull function. Points are chamber means. Ozone exposure calculated as a 24 hour seasonal sum.

8. Sycamore seedlings (*Platanus occidentalis* L.) grown in (A) charcoal-filtered chambers, and (B) in the 2.0x chambers. Notice the lack of leaves on the lower stem of the plants from the 2.0x chambers.

9. Ozone monitoring site locations for 1988 and calculated 3-month SUM06 at each site (A). Ozone exposure potential surface "EPS" (B). Increasing potential for high ozone exposure is indicated with increasing degree of shading. EPS derived from factors given in Table 1 of Hogsett *et al.* (1993).

10. Predicted biomass loss (PRBL) for black cherry (*Prunus serotina* Ehrh.) (A) and red maple (*Acer rubrum* L.) (B) with 1988 ozone exposure. PRBL calculated for each 20 km cell based on estimated ozone exposure value (3-month SUM06) and Weibull parameters for each species response function. Note the different scales for each species. The PRBL ranges from <20 to >30% in black cherry, but in red maple, PRBL is <2 to >4% over its entire range.

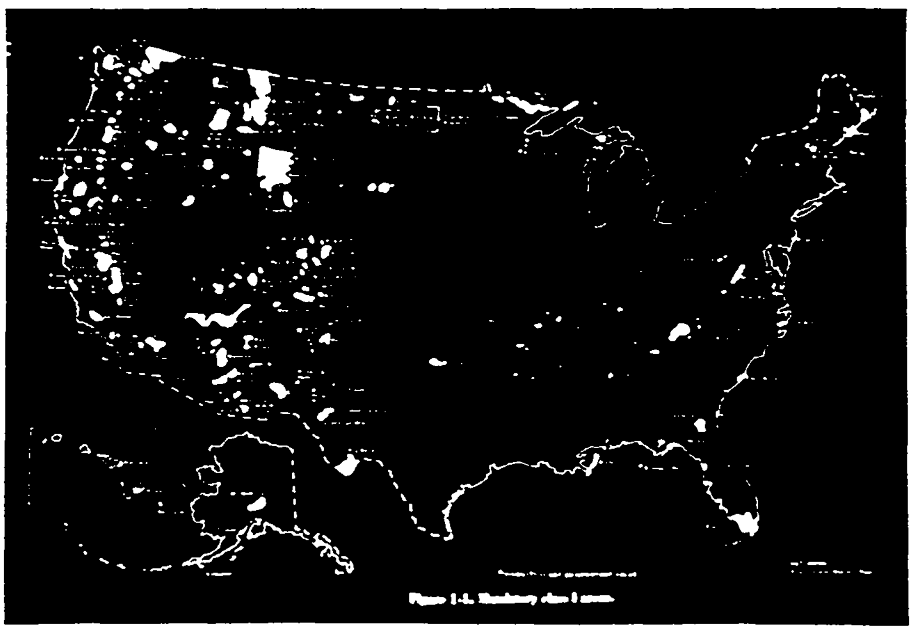
11. Variation in biomass loss with genotype with estimated 1988 ozone exposure. Aspen (*Populus tremuloides* Michx.) clone 259 (A) and clone 271 (B). PRBL calculated for each 20 km cell based on estimated ozone exposure value (3-month SUM06) and Weibull parameters for each genotype response function.

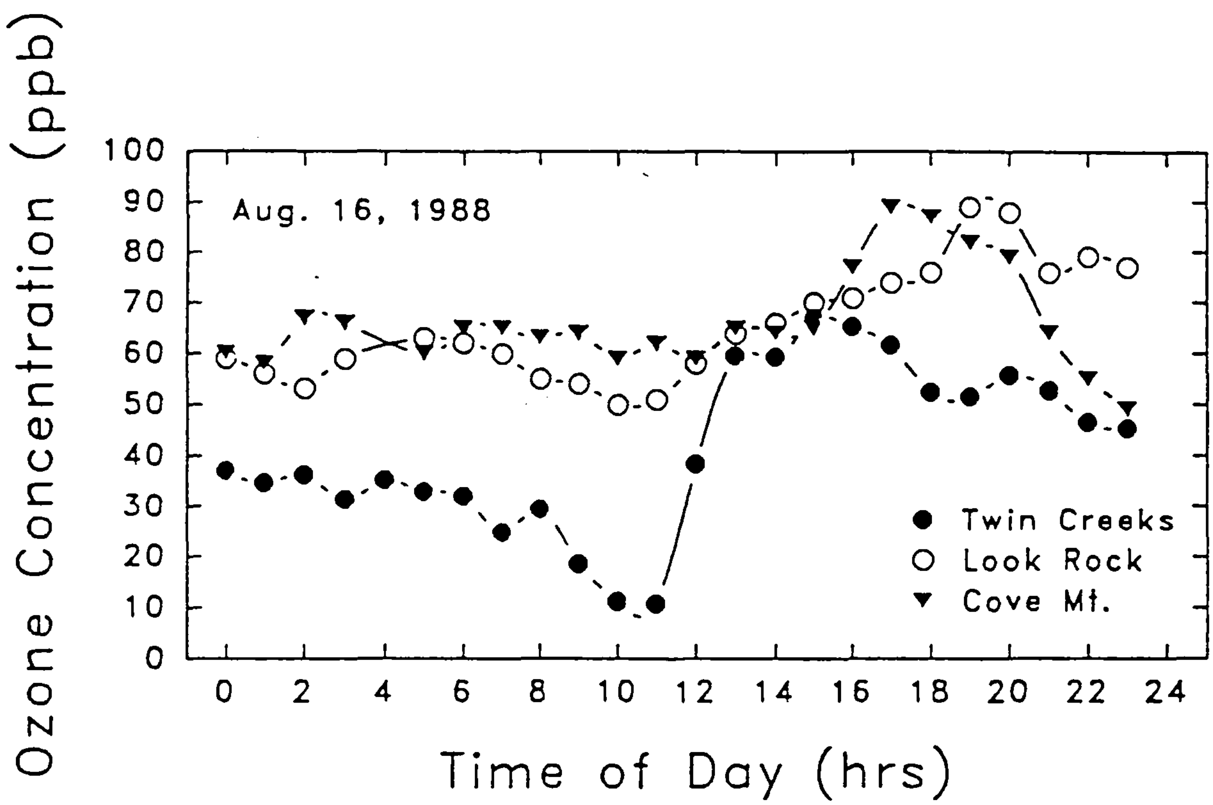
12. Box-plots of annual area-weighted biomass loss for the 8 tree species with estimated ozone exposure in 1988 (A) and 1989 (B). The predicted biomass loss is taken from each 20 km cell in the species' distribution, weighted for the area and the distribution plotted showing the 10th (bracket), 25th (lower shaded box), 50th (clear bar in shaded box), 75th (upper shaded box), and 90th (bracket) percentile. The single values outside these percentiles are also plotted (●). The percentiles represent the area of the species exhibiting that level or less of biomass loss. OR = Oregon site, MI = Michigan site, SMNP = Smoky Mountain National Park, and AL = Alabama.

13. Effect of ozone exposure on productivity of aspen (*Populus tremuloides* Michx.) (A) and loblolly pine (*Pinus taeda* L.) (B). Predictions of loss were made for aspen using allometric equations from Cooper (1981) and for loblolly using equations of Schumacher and Coil (1960). The effect of ozone was incorporated by multiplying the yearly incremental growth times experimentally determined reductions in growth from Table 3 in Hogsett *et al.* (1993). The reductions were further modified by assuming either a simple interest or compound interest model for the reduction in growth over years.

Figure 1

Figure 2





Ozone Concentration (ppb)
100
90
80
70
60
50
40
30
20
10
0
Aug. 16, 1988
Twin Creeks
Look Rock
Cove Mt.
0
2
4
6
8
10
12
14
16
18
20
22
24
Time of Day (hrs)

FIGURE 4.

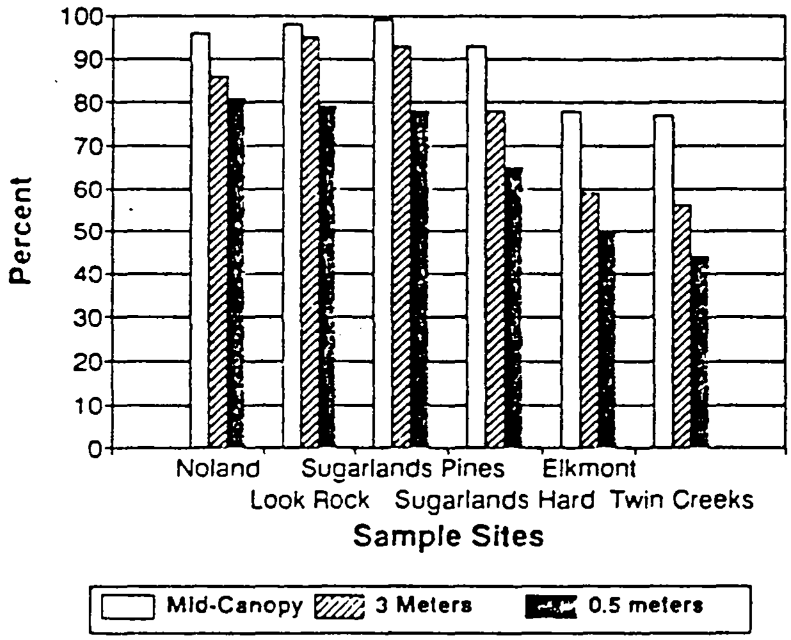
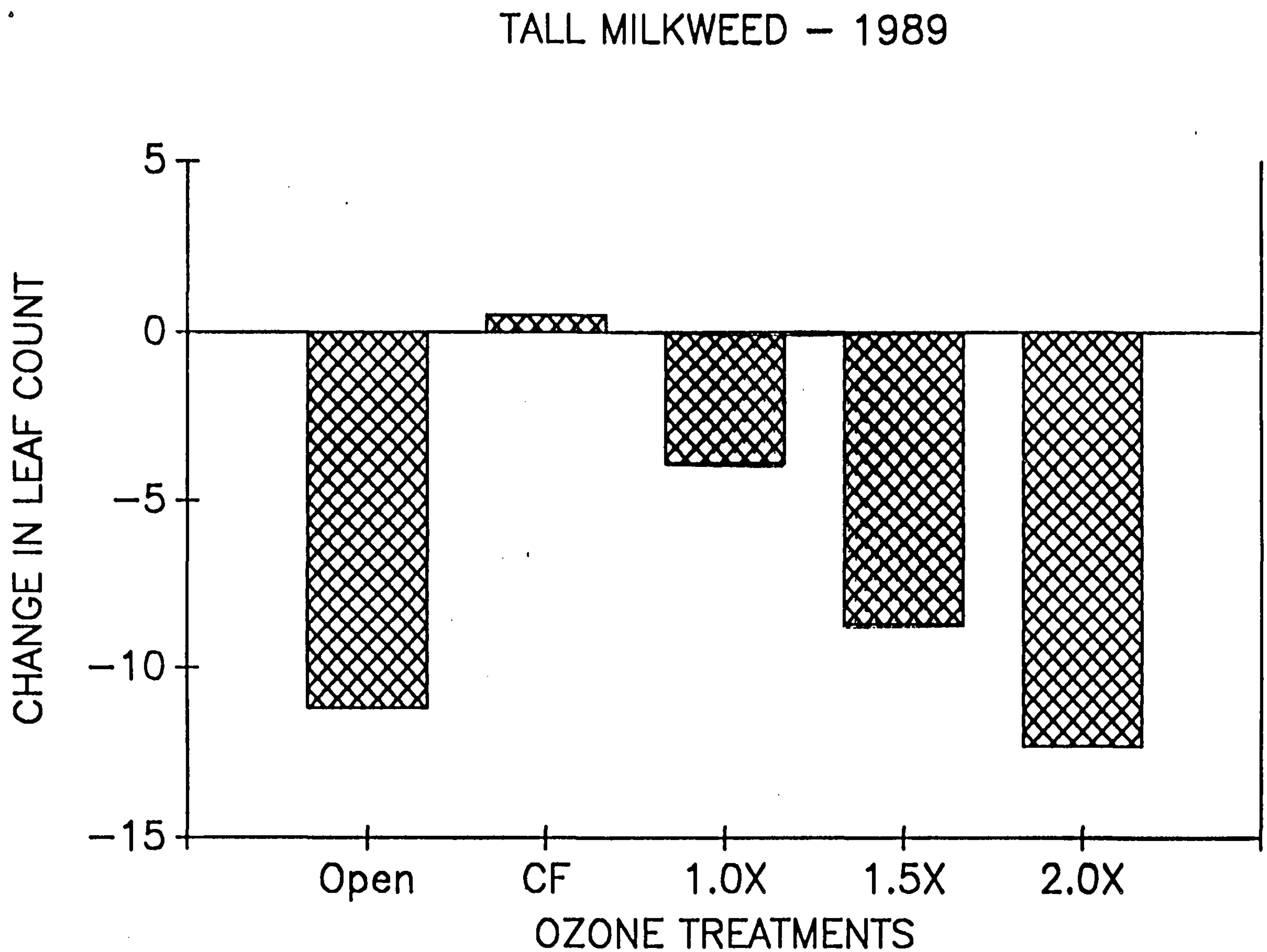
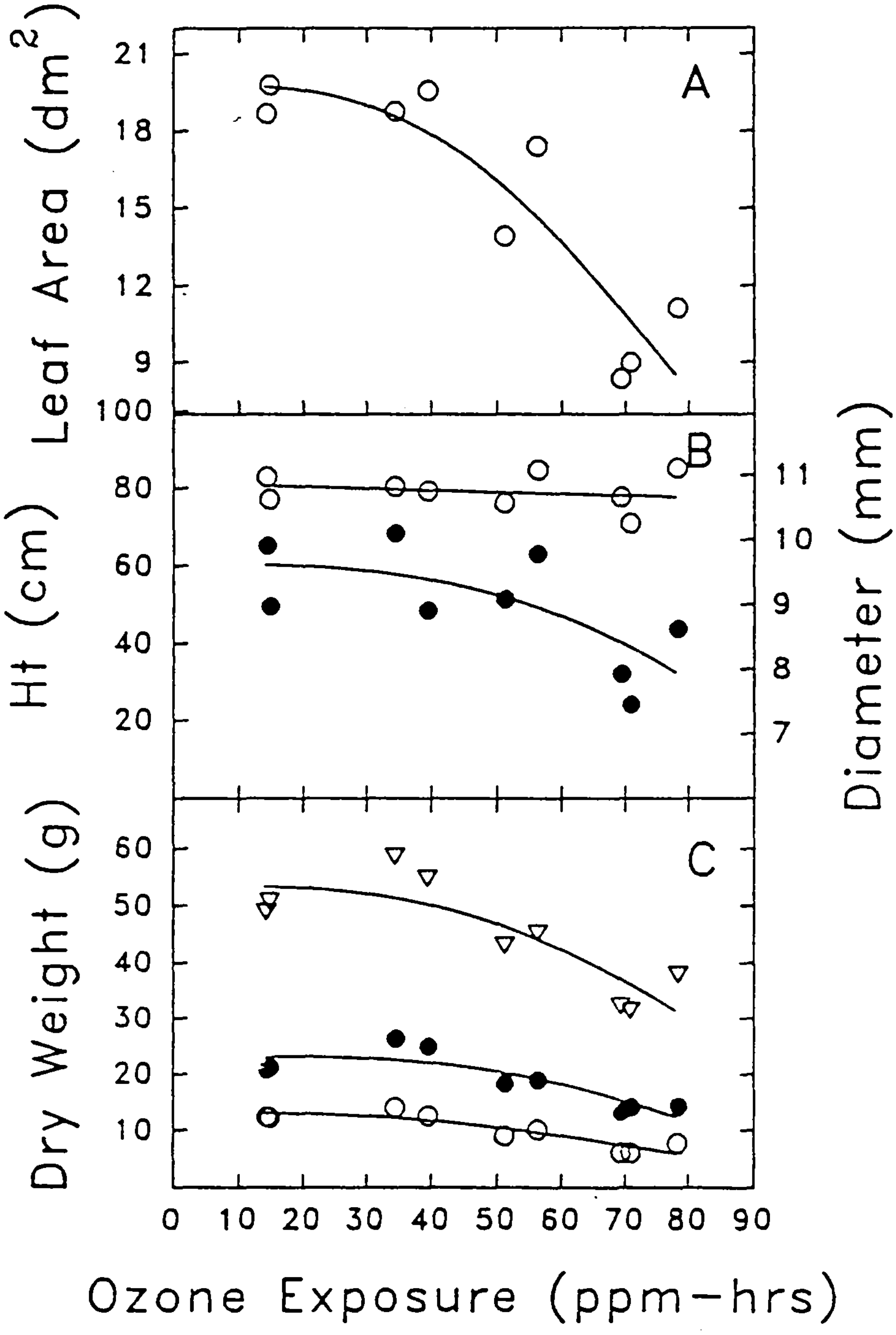

100
90
80
70
60
50
40
30
20
10
0
Percent
Noland
Look Rock
Sugarlands Pines
Sugarlands Hard
Elkmont
Twin Creeks
Sample Sites
Mid-Canopy
3 Meters
0.5 meters

FIGURE 5.


TALL MILKWEED — 1989
CHANGE IN LEAF COUNT
5
0
−5
−10
−15
Open
CF
1.0X
1.5X
2.0X
OZONE TREATMENTS



Leaf Area (dm^2)
21
18
15
12
9
A
100
80
60
40
20
Ht (cm)
B
11
10
9
8
7
Diameter (mm)
Dry Weight (g)
60
50
40
30
20
10
C
0 10 20 30 40 50 60 70 80 90
Ozone Exposure (ppm−hrs)

FIGURE 7

Figure 8

TECHNICAL REPORT DATA
(Please read Instructions on the reverse before completing)

1. REPORT NO.: EPA/600/A-94/073
2.
3. RECIPIENT'S ACCESSION NO.:

4. TITLE AND SUBTITLE: Assessing Ozone Effects on Plants Native to the Southeastern United States
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7. AUTHOR(S): [1]H.S.Neufeld, [2]J.R.Renfro, [3]S.Huang, [4]W.D.Hacker, [5]D.Mangis, [6]A.H.Chappelka, [7]W.E.Hogsett, [8]A.A.Herstrom, [9]J.A.Laurence, [8]E.H.Lee, [7]J.E.Weber, [7]D.T.Tingey

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1993 In: Proceedings of "International Association of Botanical Gardens"

16. ABSTRACT

For the last six years, the U.S.National Park Service, U.S.Environmental Protection Agency, and University researchers have been documenting the effects of ozone on a large number of woody and herbaceous species native to the southeastern United States. In Great Smoky Mountains National Park, (GRSM), ozone levels exhibit diel patterns at low elevations, where concentrations are low in the morning and high in the afternoon. At high elevations (>800 m), morning concentrations are much higher, and the total daily exposure is approximately twice that at the lower elevations. Putative ozone injury has been observed in the field in GRSM on 90 species, representing approximately 6% of the known flora in the Park. Surveys of foliar injury on several tree species show a general pattern of increasing frequency and amount of stipple with increasing elevation in GRSM, and in nearby Shenandoah National Park. Exposure-response studies were carried out in opentop chambers for six years with 46 species. Foliar symptoms seen in the field, were reproduced on 30 species, providing evidence that the foliar injury found in the field was probably due to ozone exposure. The exposure-response studies indicated that early-successional and shade intolerant species were less resistant to high ozone levels than late-successional species and dry-site adapted conifers. Geographic informations systems (GIS) techniques, coupled with growth models, were used to scale-up responses from the seedling to the geographic level. Because the modeling work is totally dependent on the quality of the exposure-response data, future work must concentrate on more realistic exposure systems, genetic variation within species, and refinement of the modeling procedures used in scaling up.

17. KEY WORDS AND DOCUMENT ANALYSIS

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ozone, plants, southwestern U.S.		

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