5-20-2005

Ozone Damage Potential to Loblolly Pine Ecosystems in Metropolitan Atlanta, Georgia

Diane Marie Styers

Georgia State University

OZONE DAMAGE POTENTIAL TO LOBLOLLY PINE ECOSYSTEMS

IN METROPOLITAN ATLANTA, GEORGIA

by

DIANE MARIE STYERS

Under the Direction of Jeremy E. Diem

ABSTRACT

Atlanta is one of the largest metropolitan areas in the southeastern United States and is the only area in the region currently listed in “serious” 1-hour ozone nonattainment. Despite its exceedance history, impacts on Atlanta’s urban forests have not been the focus of any major studies. The purpose of this study was to examine air pollution damage to vegetation using a foliar-injury survey on Stone Mountain. The objectives of this project included 1) establishing that pollution transport from Atlanta to Stone Mountain occurs, 2) determining the magnitude of ozone concentrations near Stone Mountain and 3) assessing sensitive plant species on Stone Mountain for foliar injury. Results from this study confirm that Stone Mountain is located downwind from Atlanta. Ozone concentrations were sufficiently high to damage vegetation and these consistently peaked in July. Foliar injury was present on understory species on Stone Mountain, but was not observed on loblolly pine species.

INDEX WORDS: air pollution, Atlanta, foliar injury, loblolly pine ecosystem, ozone, Stone Mountain, transport, urban, vegetation

OZONE DAMAGE POTENTIAL TO LOBLOLLY PINE ECOSYSTEMS

IN METROPOLITAN ATLANTA, GEORGIA

by

DIANE MARIE STYERS

A Thesis Submitted in Partial Fulfillment of the Requirements for the Degree of

Master of Arts

Georgia State University

2005

OZONE DAMAGE POTENTIAL TO LOBLOLLY PINE ECOSYSTEMS

IN METROPOLITAN ATLANTA, GEORGIA

by

DIANE MARIE STYERS

Major Professor:	Jeremy E. Diem
Committee:	Paul A. Knapp
	Truman A. Hartshorn

Electronic Version Approved:

Office of Graduate Studies
College of Arts and Sciences
Georgia State University
May 2005

TABLE OF CONTENTS

Page

ACKNOWLEDGEMENTS

The author wishes to acknowledge the following people for their support throughout this endeavor: Laura E. Sanders, for providing me with the moral support needed to see this through; Jeremy E. Diem, for providing guidance and content input and for editing and proofing the manuscript; Paul A. Knapp and Truman A. Hartshorn, for providing additional comments and project support; Department of Anthropology & Geography and Georgia State University for providing the financial support to make the pursuit of this Masters Degree possible; Stone Mountain Memorial Association, for allowing this research to be conducted at the Park; Arthur H. Chappelka, for assisting with fieldwork and for providing additional comments; and to my parents, grandparents, brother, family and friends, for supporting me throughout this task.

LIST OF TABLES

LIST OF FIGURES

Page

Chapter 1. Introduction

Tropospheric ozone has been recognized as the most widespread phytotoxic air pollutant in eastern North America (U.S. Environmental Protection Agency 1996). The prevalence of high ozone concentrations was once thought to be a problem only in urban areas, due to industrial and vehicle emissions of nitrogen oxides (NO_X), a precursor compound in ozone formation (Aneja et al. 2000; U.S. National Research Council 1992). Studies on the role of atmospheric transport of ozone and its precursors have indicated that rural and forested areas located downwind from metropolitan regions are also susceptible to above-average concentrations of ozone (Mueller et al. 1996; Gregg et al. 2003).

This study explores possible air pollution damage to vegetation in metropolitan Atlanta. Specifically, the impacts of ozone exposure to vegetation on Stone Mountain will be examined in late summer of 2004, where loblolly pine (*Pinus taeda*) and associated understory species such as black cherry (*Prunus serotina*), sumac (*Rhus spp.*) and blackberry (*Rubus spp.*) dominate. Within the United States, previous studies have focused on the impacts of ozone on vegetation throughout the western states (e.g., Miller et al. 1995 and Arbaugh et al. 1998), as well as the Appalachian Mountains in the east (e.g., McLaughlin and Downing 1995 and Chappelka et al. 1999b). This project aims to supplement previous research by examining the impact of ozone on vegetation downwind of an urban area in the southeastern United States. Particularly, how do ambient ozone concentrations in Atlanta affect native plant species on Stone Mountain? If ozone-induced foliar injury is present, to what extent have these species been damaged? These questions will be addressed in detail following a review of current literature on these issues.

Formation of Tropospheric Ozone

Ozone is formed by a complex chain of chemical reactions. Ultraviolet radiation (0.37-0.42 μm) breaks up natural nitrogen dioxide (NO_2) into nitric oxide (NO) and monatomic oxygen (O) which then combines with natural oxygen (O_2) to produce ozone (O_3) (Krupa et al. 2000). Ozone can then react with the NO, which is also emitted naturally from soils, fires and lightning and anthropogenically through fossil fuel combustion to produce NO_2 and O_2, completing the photolytic cycle (Atkinson 2000).

Besides NO_X, the emission of volatile organic compounds (VOCs) is needed to produce ozone. Non-methane organic compounds (NMOC) such as isoprene, monoterpenes, sesquiterpenes and oxygenated VOCs are naturally emitted from vegetation (Geron et al. 1995; Atkinson 2000; Pompe and Veber 2001) and are commonly referred to as 'biogenic VOCs' or BVOCs. In urban areas the majority of NMOC and NO_X emissions are from anthropogenic sources, while in rural areas NMOC and NO_X of biogenic origins prevail (Atkinson 2000). For example, isoprene dominates over anthropogenic NMOC production in the southeastern US where the presence of oak species (*Quercus spp.*) is numerous (Geron et al. 1995).

A complication in ozone photochemistry is that of NO_X- and VOC-limitation. In natural environments NO_X are much less abundant than VOCs rendering that environment "limited" to the amount of NO_X available to produce ozone (Chameides 1992). These conditions also generally increase the rate of ozone destruction as VOCs can react with ozone and break down the ozone compounds (Chameides 1992). In contrast, urban environments typically have high NO_X levels from vehicle and industrial emissions but low VOC levels causing these environments to be limited to the amount of VOCs available to produce ozone (Chameides 1992). However, in areas with extremely high NO_X concentrations the rate of ozone production

can actually decline due to ozone break-down associated with NO_X interactions causing ozone production to be limited solely by NO_X (Chameides 1992).

Climatic variables such as solar radiation and temperature can cause certain areas to produce higher levels of ozone (McLaughlin and Downing 1995; Krupa et al. 2000). Solar radiation is responsible for initiating the photolytic cycle and its intensity is largely a function of season and latitude. Temperature has been shown to induce ozone formation as well and a linear relationship between temperature and ozone concentration has been established (Kelly and Gunst 1990), particularly when temperatures are greater than 30º Centigrade (U.S. EPA 1996). Environments in which temperatures exceed 30º C generally produce the potential maximum ozone concentration under favorable conditions; that is, in the absence of climatic variables that can decrease ozone production, such as degree of cloudiness, precipitation and high wind speeds (U.S. EPA 1996). Stagnant air, generally resulting from stalled high-pressure cells, is the ideal environment for the production of ozone (U.S. EPA 1996). The most favorable conditions for ozone formation resulting from these climatic variables occur in the coterminous United States during summertime (June, July and August). In the southeastern United States (Köppen Climate Class Cfa), ozone production occurs mainly from March to October (U.S. EPA 1996). Thus, ozone production is usually maximized during the growing season of vegetation when the intensity of the sun and temperature are highest and cloudiness and winds are lowest (Sillman 1999; Krupa et al. 2000).

Ozone Transport

Horizontal transport of ozone can occur within the troposphere due to synoptic and mesoscale climatic patterns. As a result, peak concentrations of ozone generally occur

downwind from emission sources (Seinfeld 1989; Sillman 1999). On a synoptic scale ozone can accumulate within high-pressure cells and be transported long distances, up to hundreds of kilometers, to rural and forested sites (Seinfeld 1989; Comrie 1990, 1994; Boehm et al. 1994; Diem 2004). At the mesoscale ozone transport is influenced by local wind systems traveling over varying topography (McKendry and Lundgren 2000). Consequently, ozone concentrations at the top of a mountain can be higher than at the base of the mountain due to upslope winds (Boehm et al. 1994; Diem and Comrie 2001).

Typically ozone transport from urban emission sites result in higher ozone concentrations at downwind rural locations than from *in situ* production of ozone in those rural areas (Meuller et al. 1996). Rural sources of NO_X can combine with BVOCs such as isoprene (emitted form trees) and eventually become ozone (Geron et al. 1994; Meuller et al. 1996). However, the combination of biogenic and anthropogenic sources of NO_X and VOCs in urban areas usually result in higher ozone production and thus, higher ozone levels (Meuller et al. 1996). Consequently, horizontal transportation of ozone from urban areas is an important component of ozone exposure potential at rural and forested sites downwind.

Ozone can also be transported vertically from the stratosphere to the troposphere through the folding of the tropopause associated with mid-latitude cyclonic systems (Elbern et al. 1997; Roelofs et al. 2003; Diem 2004) and diurnal boundary layer differences associated with local summertime convection (Zaveri et al. 1995; Zhang and Rao 1999; Aneja et al. 2000). In the southeastern United States there is potential for stratospheric intrusion at any time during the year since tropospheric folding events in the region can occur at any time (Viezee et al. 1983; U.S. EPA 1996). However, these stratospheric intrusions generally contribute to ground-level ozone concentrations less than 1% of the time (Viezee et al. 1983; U.S. EPA 1996). Local

convection constitutes the majority of vertical ozone transportation in the Southeast as ozone and NO_X trapped in the nocturnal boundary layer can enhance ground-level ozone concentrations the following morning as the inversion begins to break up (Zaveri et al. 1995). Air masses above the nocturnal boundary layer can also transport trapped ozone to mountain-top and high-elevation locations (Zaveri et al. 1995; Zhang and Rao 1999; Aneja et al. 2000).

Potential Effects of Ozone Exposure

The U.S. Environmental Protection Agency (U.S. EPA) considers ozone a major air pollutant as it involves the welfare of both humans and vegetation (1999). Low levels of ozone have been shown to disturb human health causing skin and eye irritation, shortness of breath, chest pain and decreased lung function to sensitive individuals; high levels of ozone can cause symptoms in anyone of the general population (U.S. EPA 1999). It is for this reason that most ozone studies in the U.S. are conducted. However, ozone is equally detrimental to the health of vegetation and research has proven this much more solidly. Trees that have been adversely affected by ozone commonly exhibit reduced photosynthesis rates (Wiselogel et al. 1991; Richardson et al. 1992), reduced height and/or diameter growth (Pye 1988; Taylor 1994), biomass loss (Shafer and Heagle 1989; Hogsett et al. 1997) and/or foliar injury (Horton et al. 1990; Kuehler and Flagler 1999). If damage is great enough an entire forest ecosystem can be significantly altered.

The mixed pine-hardwood forests of the San Bernardino Mountains in southern California have been exposed to chronic doses of ozone for over 50 years (Asher 1956), with average 24-hour concentrations up to 120 parts per billion (ppb; U.S. EPA 1996). Pollutant transport from nearby Los Angeles is the likely mechanism responsible for the death of most ozone-sensitive

pine species on these Mountains (Miller et al. 1963; Miller et al. 1995; Arbaugh et al. 1998). Decline in sensitive, dominant forest species results in forest stand alteration and eventually complete ecosystem transformation (U.S. EPA 1996). Forests not only harbor most of the world's biodiversity, they also protect water resources, manage climate, provide oxygen and help support local, regional and global economies.

Effects on Plant Physiology

Ozone can damage plant leaf tissue when a sufficient amount of ozone molecules are able to pass through a series of permeable layers in the leaf to reach the mesophyll, a spongy tissue critical to photosynthesis (Chameides 1989; Hewitt et al. 1990; U.S. EPA 1996; Kuehler and Flagler 1999). If met by antioxidants such as ascorbic acid at any point a long this pathway, ozone molecules may be scavenged prior to reacting with vulnerable plant cell tissues (Kuehler and Flagler 1999). An ozone molecule first diffuses into a plant leaf through one of its many stomata (leaf pores), which regulate gas exchange by allowing for sufficient carbon dioxide (CO_2) uptake while limiting water loss through evapotranspiration (Hewitt et al. 1990). The stomata open to a cavity where the ozone molecule can then dissolve in an aqueous layer lining this inner air space of the leaf then proceed to penetrate the cell well (U.S. EPA 1996). Once inside the plant's cellular membrane it can react with polyunsaturated fatty acids and begin its destructive oxidation processes (Chameides 1989). Through the oxidation of plant tissue, ozone can interfere with any of the various processes of photosynthesis (e.g. Friend and Tomlinson 1992; Krupa et al. 2000; Vollenweider et al. 2003).

Photosynthesis is the process by which plants use CO_2 and energy from sunlight to make food. Ozone disrupts photosynthetic processes in sensitive and tolerant plants and can have any

number of consequences on the plant, many of which have yet to be documented (Chappelka and Samuelson 1998; Krupa et al. 2000). In addition to the effects on growth, biomass and foliar injury, reduced photosynthetic rates can also result in water stress, nutrient deficiency and variations in carbon allocation (Friend and Tomlinson 1992; Vollenweider et al. 2003). Tree age, differential sensitivity among species and site-specific environmental and climatic variables should also be considered as these factors often modify individual study results (Wiselogel et al. 1991; Manning and Krupa 1992; Chappelka and Samuelson 1998; Chappelka et al. 1999b.; Krupa et al. 2000).

The most common visible sign of ozone-induced damage to plants is foliar injury. This is generally an indication that antioxidants have attempted to shield vulnerable leaf tissues from the oxidation processes of ozone (Krupa et al. 2000; Friend et al. 1992). Proven indicators of ozone-induced foliar injury include bleaching, bronzing, flecking, stippling, mottling, chlorosis, necrosis, tip burn and premature senescence (Krupa et al. 2000; Vollenweider et al. 2003). These visual symptoms of ozone injury are seen in acute (Friend and Tomlinson 1992) to chronic (Arbaugh et al. 1998) exposure situations and can affect sensitive (Chappelka et al. 1999b.) as well as tolerant species (Neufeld et al. 2000). As such, foliar injury is a good indicator of disruption of photosynthetic processes. However, ozone-induced damage to plant species can occur with or without foliar injury (Horton et al. 1990; Taylor 1994; Barbo et al. 2002).

Ozone has been shown to alter both tree growth and biomass production by interfering with stomatal conductance, respiration and photosynthesis, as well as reducing CO_2 uptake, assimilation and fixation (Chappelka and Samuelson 1998; Krupa et al. 2000). Plant growth, biomass production and other physiological processes are generally interrelated due to their dependence on optimum photosynthesis; yet damage to one component may not necessarily

disturb another (Chappelka and Samuelson 1998; Somers et al. 1998; Massman et al. 1999; Krupa et al. 2000). For example, ozone may disrupt a plant's photosynthetic processes by decreasing stomatal conductance and reducing CO_2 uptake. The plant may exhibit signs of foliar injury, but not symptoms of reduced growth (Horton et al. 1990; Taylor 1994), or vice versa (Edwards et al. 1995). However, when this same plant is grown in a much less predictable competitive environment, height and diameter growth and biomass production may decrease while exhibiting no signs of visible injury (Barbo et al. 2002; Laurence and Andersen 2003). It is therefore, important to note that while ozone-induced damage may take a variety of forms, interactions within and between plant species are highly complex and their countless responses to ozone exposure are not fully understood.

Impacts to Forest Ecosystems

Estimating ozone risk to forests based on the responses of tree species contained therein is a difficult task. There have been many attempts to do so using techniques such as statistical (McLaughlin and Downing 1995; Kuehler and Flagler 1999; Massman et al. 1999) and simulation modeling (Hogsett et al. 1997; Barbo et al. 2002), interpolation of surfaces (Lefohn et al. 1997; Phillips et al. 1997) and extrapolation from tree to stand conditions (Taylor 1994; Kolb and Matyssek 2001). However difficult to estimate, species-specific responses to ozone can influence forest ecosystem stability by modifying overall forest productivity (Hogsett et al. 1997; Coulston et al. 2003). Coulston et al. (2003) describe that by directly impacting tree growth, ozone can transform "forest succession, forest composition… and forest dependent wildlife, insects and pathogens." If species are dominant in areas where they are predicted to be at risk, their ecological and economical importance to these areas could be threatened (Arbaugh et al.

1998; Coulston et al. 2003). For example, Coulston et al. (2003) explain that loblolly pine and sweetgum (*Liquidambar styraciflua*) are fundamental species that co-exist throughout the Southeast. Loblolly pine is an important commercial product as well as an indicator species in pine and pine-hardwoods stands in the area, which provide habitat for many game and nongame species. Sweetgum contributes to the local foodweb by providing its seeds as a food source to birds, squirrels and chipmunks. Damage to these species by ozone could therefore, affect the biodiversity of southern pine and pine-hardwood habitats as well as world-wide timber industries.

Ecological impacts to forest ecosystems can indirectly influence the economic benefits of that system. Ozone changes to single tree physiology have been noted; height and diameter growth can be reduced, biomass can be lost and photosynthetic processes altered resulting in foliar injury, water stress and nutrient deficiency (Vollenweider et al. 2003). Changes to key sensitive species within a community can slowly but eventually alter genetic traits (Coulston et al. 2003). As this species or even certain genotypes of this species is weakened, more tolerant competitive species may succeed within the forest community possibly eradicating the weaker sensitive species and altering the composition of the community (Vollenweider et al. 2003). Should this happen within an economically important forest ecosystem such as loblolly pine, the timber industry would be greatly affected and much capital would stand to be lost (Vollenweider et al. 2003). Similarly, if key forests and their unique habitats decline, revenue generated from recreational activities such as hunting, fishing, hiking and camping would be reduced (Coulston et al. 2003).

Ozone in the Southeastern United States

For the purposes of this study, the southeastern United States will be defined according to the U.S. EPA Region 4. By this definition, the Southeast includes the states of Alabama, Florida, Georgia, Kentucky, Mississippi, North Carolina, South Carolina and Tennessee (U.S. EPA 2004c.). This definition was chosen so geographic areas mentioned in this study would correspond to the federal ozone standards and data cited herein.

The southeastern United States is a region that has recently had elevated levels of ozone (U.S. EPA 1996). For example, Atlanta has exceeded federal air quality standards for at least the past ten years, as have other large cities in the region including Birmingham, Nashville and Charlotte (U.S. EPA 2004a.). This could possibly be attributed to its temperate climate, its abundance of vegetation (potential BVOC emissions), its history of industry (potential NO_X and VOC emissions), a profusion of automobile traffic and long commutes and long-range transport from surrounding industrialized cities from Louisiana to the Carolinas (Neufeld et al. 2000). The highest ozone concentrations in the Southeast occur from March to October in metropolitan areas such as those listed above and at high elevations in the Appalachian Mountains (U.S. EPA 1996). However, these data could be misleading, as ozone monitoring in rural areas of the Southeast is uncommon.

The Southeast represents a good location for ozone-related investigations because conditions are favorable to ozone production. The lush vegetation of the mixed pine-hardwood forests together with the abundance of cultivated lawns, results in the likelihood of a vast quantity of BVOC emissions. BVOC concentrations vary by region due to the type of vegetation present and are an important component to consider in ozone-related studies, as they are natural precursors to ozone production (Geron et al. 1995; Atkinson 2000; Pompe and Veber 2001).

Additionally, the magnitude of industrial plants and the amount of vehicles present in sizeable cities in the Southeast such as Atlanta, Birmingham and New Orleans can lead to potentially high levels of NO_X and VOC emissions (Geron et al. 1995). Precise combinations of these variables can lead to potentially high ozone levels (Krupa et al. 2000).

Forests of the Southeastern United States

Forests of the southeastern United States are among the most diverse in North America and primarily consist of hardwood and mixed pine-hardwood systems (Powell et al. 1992; USDA Forest Service 2004a.). Dominant forest types include: oak-hickory and maple-beech-birch in the Appalachian mountain region; oak-pine and loblolly-shortleaf pine in the Piedmont region; and longleaf-slash pine and oak-gum-cypress in the coastal plains region (Powell et al. 1992; USDA Forest Service 2004a.). Approximately half (48%) of the wetlands of the coterminous U.S. are located in the South; 74% of these wetlands are classified as “forested” (U.S. Fish and Wildlife Service 2004). Additionally, southeastern forests contain the highest concentration of freshwater aquatic diversity in the world (U.S. Fish and Wildlife Service 2004). Conversely, the southeastern U.S. contains approximately half of the world’s industrial tree plantations where commercial logging rates are high (Powell et al. 1992; USDA Forest Service 2004a.). As this study will examine the impacts of ozone to forest species in the Piedmont province of Georgia, specifically metropolitan Atlanta, only the forests types appropriate to this area will be discussed in further detail.

Oak-Pine communities are located in relatively dry, exposed areas including the Piedmont, Ridge and Valley and Blue Ridge physiographic provinces of Georgia (Kuchler 1964). Dominant species within these forests include various oak species, such as northern red (*Quercus*

rubra), white (*Quercus alba*), black (*Quercus velutina*), scarlet (*Quercus coccinea*) and chestnut (*Quercus prinus*); pine species, including eastern white (*Pinus strobus*), pitch (*Pinus rigida*), Virginia (*Pinus virginiana*), table mountain (*Pinus pungens*) and shortleaf (*Pinus echinata*); as well as other native species such as hickory (*Carya*), white ash (*Fraxinus americana*), hemlock (*Tsuga canadensis*), sugar maple (*Acer saccharum*), red maple (*Acer rubrum*), tulip poplar (*Liriodendron tulipifera*), American basswood (*Tilia americana*), black gum (*Nyssa sylvatica*), black cherry and black birch (*Betula lenta*) (Redington 1978).

Loblolly-shortleaf pine forests are the second most widespread forest type in the eastern U.S. accounting for approximately 64% of coniferous forest in the East (Chappelka and Samuelson 1998). These forests are primarily comprised of loblolly and shortleaf (*Pinus echinata*) pines and cover most of the Piedmont region of the Southeast (Kuchler 1964). Other pine species within this system may include Virginia (*Pinus virginiana*), Pond (*Pinus serotina*) and Pitch (*Pinus rigida*). Hardwood species might include sweetgum, magnolia (*Magnolia grandiflora*), tulip poplar (*Liriodendron tulipifera*) and various oaks (Kuchler 1964).

Potential Impacts of Ozone Exposure to Southeastern Forests

Ozone effects on vegetation have been well documented in studies in the southeastern U.S. since the 1950s when it was first identified as injurious to plants such as tobacco (*Nicotiana tabacum*), a chief agricultural product at that time (Middleton et al. 1950). Research on several types of tree species indigenous to the Southeast has revealed varying degrees of sensitivity to ozone (Wiselogel et al. 1991; Lefohn et al. 1992; Qui et al. 1992; Richardson et al. 1992; Taylor 1994; Neufeld et al. 1995; Somers et al. 1998; Chappelka et al. 1999a. & b.; Neufeld et al. 2000). Black cherry (Hogsett et al. 1997; Somers et al. 1998; Chappelka et al. 1999a.) and tulip poplar

(Hogsett et al. 1997; Somers et al. 1998; Chappelka et al. 1999b.) consistently display a high sensitivity to ozone exposure. Common understory species that display a high sensitivity to ozone include blackberry and sumac species (Barbo et al. 1998). Moderately sensitive species in the region include loblolly pine (Wiselogel et al. 1991; Qui et al. 1992; Richardson et al. 1992; Taylor 1994; Hogsett et al. 1997; Lefohn et al. 1997), eastern white pine (Bennett et al. 1994; Hogsett et al. 1997) and sugar maple (Hogsett et al. 1997). Insensitive species are Virginia pine (Hogsett et al. 1997; Neufeld et al. 2000) and Red Maple (Hogsett et al. 1997). Sensitivity has been determined through controlled chamber studies by measuring height and/or diameter growth, percent biomass and the presence of foliar injury.

Based on species distribution maps (USDA Forest Service 2004a.), most forests in the Southeast have the potential to be adversely affected by ozone based on sensitivity rankings determined through previous research studies. Several species within each of the forest types listed above have been examined for ozone injury. A majority of this research has been conducted in Great Smoky Mountains National Park, which lies near the southern end of the Appalachian Mountains along the North Carolina and Tennessee borders. The results of these studies indicate that black cherry and tulip poplar appear to be among the most highly sensitive to ozone (Neufeld et al. 1995; Hogsett et al. 1997; Somers et al. 1998; Chappelka et al. 1999a. & b.). Other researchers have estimated risk to southeastern forests using statistical and simulation modeling techniques and have determined that forests of the Piedmont and coastal plains regions of the Southeast are also potentially at risk (Hogsett et al. 1997; Barbo et al. 2002; Coulston et al. 2003).

Although the findings from previous studies are significant, many of the studied species are found mainly in rural and upland areas and therefore, do not adequately represent the overall

range of ozone damage to vegetation for the entire southeastern region. Of the wide-ranging forest communities, the loblolly-shortleaf pine forest system has been the focus of most studies in the Southeast (Wiselogel et al. 1991; Qui et al. 1992; Richardson et al. 1992; Hogsett et al. 1997; Lefohn et al. 1997). This is probably due to the moderate sensitivity of loblolly pine to ozone (Taylor 1994) and its great economic importance as a major timber product in the Southeast (Chappelka and Samuelson 1998; Coulston et al. 2003).

Previous Studies of Ozone Effects on Loblolly Pine (Pinus taeda)

Loblolly pine is the dominant tree species of the granite outcrop ecosystem that is Stone Mountain and its response to ozone has been studied in depth (e.g., Pye 1988; Taylor 1994; Lefohn et al. 1997; Coulston et al. 2003). Results from previous investigations of loblolly pine response to ozone exposure have indicated that this species is moderately sensitive to ozone (Wiselogel et al. 1991; Qui et al. 1992; Richardson et al. 1992; Hogsett et al. 1997; Lefohn et al. 1997; Coulston et al. 2003). Noted signs include inhibited growth (Pye 1988; Taylor 1994), biomass loss (Shafer and Heagle 1989; Hogsett et al. 1997), reduced photosynthesis rates (Wiselogel et al. 1991; Richardson et al. 1992) and foliar injury (Horton et al. 1990; Kuehler and Flagler 1999). A majority of studies have been chamber fumigation experiments on loblolly seedlings using varying degrees of ozone concentrations in competitive (Barbo et al. 2002; Laurence and Andersen 2003) and non-competitive environments (Shafer and Heagle 1989; Horton et al. 1990; Richardson et al. 1992; Flagler and Chappelka 1995; Reinert et al. 1996). Other studies have tested ozone exposure-response functions on loblolly seedlings and saplings in greenhouse experiments (Horton et al. 1990; Wiselogel et al. 1991; Shafer and Heagle 1989)

and field analyses of mature loblolly growth in its natural state (McLaughlin and Downing 1995).

A review of the literature indicates that most of the research on loblolly pine pertains to the types of injury caused by exposure to ozone. The researchers of these studies examined any of four major physiological processes typically influenced by ozone exposure: 1) photosynthesis; 2) foliar health; 3) growth (height and radial); and 4) biomass production. The following four sections will summarize the leading publications in each of these areas. In many cases, the study examines more than one parameter of loblolly injury and therefore, will be mentioned in the section that most closely relates to the authors' main objectives.

Photosynthesis

Ozone has been shown to reduce photosynthetic rates in several plant species (Lefohn et al. 1997; Coulston et al. 2003). Biochemical composition can be altered resulting in reduced growth, biomass loss and premature needle senescence (Chappelka and Samuelson 1998). The physiology of loblolly needles is highly complex and much is still unknown about species-specific photosynthetic responses resulting from ozone exposure.

Ozone and acidic precipitation have been shown to affect photosynthesis in loblolly pine species. Richardson et al. (1992) examined these effects in three families of loblolly pine in North Carolina over the course of one year (March to December). There were no statistically significant outcomes of acid rain treatment on photosynthesis in any of the three families. The response of photosynthesis to ozone was different in each of the three families. Photosynthesis rates were reduced by as much as 80+% in the 3X ozone treatment chambers by October. By the end of the growing season, needles of the saplings in the 3X and 2.25X treatment chambers had

begun to prematurely senesce. The most important finding from the Richardson study indicated that after only one growing season, photosynthesis rates were reduced by an average of 10% in the sub-ambient chambers and 25% in the 1.5X ambient chambers. These exposure concentrations are typical of those found during the growing season in the Southeast. Therefore, there is potential for reduced photosynthetic rates and thus growth in loblolly pines growing in this region under ambient conditions.

Ozone affects on photosynthesis have also been shown to alter the biochemical composition of loblolly pines. Friend et al. (1992) studied the biochemical composition of one family of loblolly pine exposed to varying levels of ozone, acid rain and magnesium in Tennessee over three growing seasons. Needle and stem tissue was extracted and the amount of carbohydrate reserves present was analyzed. The total dry weight of carbohydrate concentrations were 24-56% less in the ambient ozone treatment chambers compared to the sub-ambient ozone treatment chambers. This is possibly due to its influence on foliar starch. A definite variation in maximum (April) and minimum (September) starch concentration in loblolly foliage was observed. The authors state that this information could be used to enhance future stand-level experiments and modeling efforts at a relatively low cost.

The role of antioxidants in protecting plants from ozone exposure has also been examined. Kuehler and Flagler (1999) evaluated the effects of antioxidant treatments (sodium erythorbate, or Ozoban and ethylenediurea, or EDU) on photosynthetic function of loblolly pine seedlings exposed to ozone in Texas. Foliar injury was observed early in the experiment, but subsided over time indicating the tree’s ability to recover from ozone damage. This phenomenon was not attributed to protection from the antioxidants. Additionally, neither of the two antioxidants enhanced photosynthesis rates in the seedlings in all cases, nor at all ozone concentration levels.

However, this could be attributed to experimental design flaws and further research is needed to better understand the ozone mitigation potential of these two antioxidants.

Foliar Injury

Ozone-induced foliar injury is usually caused by an interference with one or more functions of photosynthetic processes (Friend et al. 1992; Krupa et al. 2000). Visible symptoms on loblolly pine needles include chlorotic mottling, necrosis and tip burn; needles may also prematurely senesce (Krupa et al. 2000). Though foliar injury generally indicates a disruption in photosynthesis (Richardson et al. 1992), it does not necessarily correlate with growth (Horton et al. 1990; Taylor 1994).

Interference with photosynthesis often induces foliar injury in loblolly pines. Friend and Tomlinson (1992) explored the physiological function of twelve loblolly pine seedlings in Tennessee exposed to ozone through analyzing foliar carbon dynamics. The authors found that ozone altered the amount of total carbon as well as the partitioning of carbon. Total carbon loss resulted in short term retention of carbon followed by a rapid export of carbon into sugars and carbon loss to respiration. Ozone also caused a decrease in partitioning of carbon into protein and starch and an increase in partitioning of carbon into organic acids, residue and lipids. These results imply a shirt in carbon partitioning from those responsible for carbon storage to those used in injury repair. This shift can eventually lead to reduced growth rates if sufficient carbon is not available and/or if ozone exposure levels in the immediate environment are high.

Ozone exposure can also interfere with other nutrients and their cycling processes within vegetation. Edwards et al. (1995) evaluated the nutritional responses of 1296 loblolly pine seedlings and 90 red spruce saplings to ozone exposure and acidic precipitation over three

growing seasons. No increased nutrient leaching or acidification occurred in the soil of the loblolly pine seedlings in response to the acid rain treatments. In contrast, the soil of the red spruce saplings negatively to the acid rain treatments resulting in a lower pH and the leaching of calcium and magnesium. With regard to ozone exposure, the foliage of the red spruce saplings was not greatly disturbed. The loblolly pine seedlings exhibited greater mineral concentrations in foliage and a reduction in biomass. However, the authors found no significant affects of ozone exposure on the nutrient content of loblolly foliage.

Foliar injury surveys can be used to assess areas at risk for ozone injury. Coulston et al. (2003) conducted a regional assessment of ozone sensitive trees in the eastern U.S. using data from previous studies to identify areas of potential risk. Through the use of kriging, each of the 512 field plots was assigned a biosite index of severity based on foliar injury. Based on these parameters, loblolly pine (mean biosite index = 20.4) was determined a sensitive species and at moderate to high risk for ozone injury within their range. Additionally, the authors ranked loblolly pine as one of the four species determined to be at risk on a regional scale from exposure to current levels of ambient ozone. They suggested that more in-depth studies are needed to assess the actual impacts of ozone on the growth of loblolly pine, especially in the Southeast.

Growth

Ozone effects on height growth in loblolly pine have been examined over the past four decades. Recent research, conducted since the late 1980s, is most relevant to this study. The majority of results from these studies indicate that loblolly pines exposed to ozone exhibit reduced growth, namely height, diameter, stem circumference and root growth. However, the

literature shows that as knowledge accumulates and scientific methods improve we begin to see varying, sometimes contradictory, results.

A majority of research on loblolly pine has resulted in similar conclusions. Pye (1988) reviewed 43 tree species and hybrids from the data of 25 experiments to determine the consequences of ozone exposure on each. His article conferred that loblolly pine exposure to ozone resulted in reduced height and diameter growth, biomass production and photosynthetic rates, but stated that assessment methods need to be improved in order to accurately quantify damage. He is also one of the first researchers to pose the issue of differential sensitivity among genotypes. Similarly, Shafer and Heagle (1989) conducted a 3-year study on loblolly pine in North Carolina and found increasing damage (reduced height growth and biomass production) with increasing ozone concentrations. These researchers also found variations in response due to differential sensitivity among seedlings. Additionally, they reported that reductions in growth were not correlated with visible injury, an important visible indicator of ozone damage at that time.

During the 1990s, ozone-related research substantially increased in the Southeast, largely as a result of the Forest Health Monitoring Program of the USDA Forest Service, begun in 1990. With research efforts focused in this area, studies of loblolly pine became more prevalent. Horton et al. (1990) examined seedlings of three loblolly pine families grown in two different soil substrates. Results from this study showed that exposure to increasing concentrations of ozone resulted in reduced diameter growth (15-36% less) and biomass production (4-23% less), as well as foliar injury (up to 40% damage), but not height growth. Their findings indicate that soil nutrient composition plays an important role in individual species response to ozone exposure. However, their results contradicted earlier findings on different genotype response by

stating that the response to ozone exposure did not vary among the three families. Wiselogel et al. (1991) exposed 35 families of loblolly pine to varying concentrations of ozone. Growth was significantly reduced by exposure to ozone, especially root growth (20% less). Height growth was reduced in most families, except one family that showed increased growth with increasing ozone concentration. Diameter growth was reduced in all families. Foliar injury was evident early in the experiment and resulted in approximately 50% injury to loblolly needles. Photosynthesis was significantly reduced by as much as 30%. Neither net photosynthesis nor foliar injury varied among families, nor did they correlate with growth rates.

Current research in plant growth analyses involves the examination of exposure-response trends. Taylor (1994) was instrumental in exploring the role of genotype in exposure-response trends in loblolly pines. In his report, Taylor examines all of the results from previous studies conducted on loblolly pine (with an emphasis on the Southeast). Differential response, genetic variation, physiological mechanisms and relationships between parameters of ozone response (i.e. visible injury and growth) in loblolly pine are explored in depth. Taylor concluded that foliar injury is not correlated with growth and more importantly, that the threshold for effects on an average loblolly seedling's growth is below the average exposure rates of ozone in some areas of the Southeast. Taylor indicates that more efforts to examine biospheric rather than atmospheric processes are needed to further understand exposure-response relationships in loblolly pine to ozone.

Few studies have been conducted on mature loblolly pine trees in a field setting. McLaughlin and Downing (1995) studied loblolly pine responses to ozone in a non-chamber, natural environment. The authors measured the stem circumference of 28 mature loblolly pines over five years (1988-1992). Ozone effects varied between trees and years, with growth rates

varying by 75% across years. The authors compiled the data into a regression model to quantify the contributions of various environmental variables to growth (i.e. ozone, soil moisture, temperature and rainfall). Interactions between ozone and soil moisture accounted for the majority of variance in the model at 58%, implying a strong relationship between ozone-induced damage and soil moisture.

Because only a small number of studies have been conducted in natural settings, the role of competition has been largely overlooked. Barbo et al. (2002) assessed the impacts of ozone on loblolly pine seedlings in a competitive environment in Alabama over two years. The authors found that loblolly pines in a competitive environment respond to ozone in a much different way than previously reported. In this compelling study, loblolly pine height growth in sub-ambient conditions was less than that of higher concentration levels. Similarly, seedlings grown in sub-ambient conditions exhibited the least diameter growth and the lowest biomass production. This contradiction in results is attributed to competition for resources with other species, but the need for a longer-term study to validate these results is noted.

Biomass Production

Biomass production is also a function of tree growth. Within this field of research, the term total biomass typically refers to the dry weight (in grams) of all living parts of a tree, below (root) and above (shoot) ground. Although many of the authors noted above have examined the affects of ozone on biomass production as a part of tree growth, these authors presented here have focused their research on biomass production and may or may not necessarily include height and/or diameter growth in their analyses.

Because loblolly pine is an important commercial timber crop, especially in the Southeast, there has been ample research on ozone affects on biomass production. Flagler and Chappelka (1995) reviewed and summarized data collected through the Southern Commercial Forest Research Cooperative (SCFRC) and compared it to other studies on various southern pine species. In the studies reviewed, ozone was found to cause adverse modifications on loblolly (and other) pine species, even at current ambient levels. Observed signs throughout the studies included decreased height and diameter growth, decreased biomass, especially in foliage and decreased photosynthesis and other biochemical functions. The authors noted that the most significant finding was that decreased rates of carbon gain in foliage combined with premature senescence of needles was responsible for much of the decreased growth observed in these studies. Lost productivity resulting from ozone damage can greatly reduce tree longevity and thus, represents a substantial concern to the commercial timber industry in the Southeast.

Examinations of exposure-response trends were strengthened in the late 1990s by geographic information systems (GIS). Hogsett et al. (1997) utilized a GIS in the examination of ozone risk to eastern forests. These authors used growth or change in biomass to measure forest productivity. The GIS was created to model exposure-response functions and to extrapolate data from tree to landscape. Results indicate that loblolly pine is estimated to have a 3-6% biomass loss annually over 80% of its range. Predicted relative biomass loss is greatest within the Piedmont region of the Southeast. Based on area-weighted biomass loss of seedlings to estimated exposures across its range (for 1988), loblolly pine was grouped within the moderately sensitive category. The authors do indicate their uncertainties in their model and of risk characterization in general.

Research efforts to understand the role of antioxidants in protecting plants from ozone exposure continue to contribute new information to the field. Manning et al. (2003) conducted a three-year study on the growth 834 loblolly pine seedlings in Texas exposed to ozone and treated with antioxidants (Ozoban and EDU). Their results indicate that treatment with antioxidants did not prevent foliar injury and incidence and severity were not related to any specific treatment level. Treatment with Ozoban and EDU did increase growth of the seedlings in the second and third growing seasons, although the results were not statistically significant when analysis of variance (ANOVA) was employed to scrutinize the data. The authors note that much longer studies are needed to accurately assess the long-term consequences of treatment on tree growth under ambient ozone conditions.

Potential Areas of Concern within Past Research

Ozone studies of loblolly pine are highly complex and have thus, resulted in several research deficiencies. Since it is known that growth reduction does not often correlate to the presence of foliar injury (i.e. Horton et al. 1990; Taylor 1994; Chappelka and Samuelson 1998) and that chlorosis can also be caused by other environmental variables such as lack of light or mineral deficiencies (Allaby 1992), it is necessary to consider other environmental variables that influence loblolly pine growth and physiology and/or to examine other ozone-sensitive species coexisting in the same areas. Further, knowledge gained from experiments on seedlings and saplings (i.e. chamber studies) cannot necessarily be applied to mature trees growing in natural, competitive environments. Accordingly, it is difficult to estimate the impact of ozone exposure to mature forests under ambient conditions. Therefore, it is best to conduct studies based on a small number of sample sites within a given area to determine site-specific species response to

local ambient ozone concentrations. Lastly, urban areas contribute to high ozone concentrations in downwind rural and forested sites. Although the southeastern United States is a region that has recently had elevated levels of ozone, studies on the effects of ozone exposure to vegetation in urban environments of the Southeast is lacking.

Research Questions and Objectives

These major gaps in current literature guided the questions and objectives of this research endeavor. The proposed thesis project aims to supplement current research on ozone-induced damage to vegetation by examining mature plant and tree species in their natural setting for foliar injury. Specifically,

- ***How do ambient ozone concentrations in metropolitan Atlanta affect native plant species on Stone Mountain, particularly those of loblolly pine ecosystems?***
- ***If ozone-induced foliar injury is present, to what extent have these species been damaged?***

We predict that loblolly pine species on Stone Mountain will exhibit symptoms of ozone-induced foliar injury and premature senescence, due to the close proximity to pollutant emission sources in Atlanta. Additional ozone-sensitive species known to exist on Stone Mountain, such as blackberry, sumac and black cherry will also exhibit ozone-induced foliar injury, thus supporting the theory that ozone concentrations at Stone Mountain are sufficient to damage indigenous ozone-sensitive vegetation such as loblolly pine.

The objectives of this project will help determine if ozone levels in metropolitan Atlanta are sufficient to injure sensitive plant species by 1) establishing that transport from urban emission sources in Atlanta to Stone Mountain occurs; 2) determining the magnitude of ozone

concentrations near Stone Mountain; and 3) assessing the aforementioned plant species on Stone Mountain for ozone-induced foliar injury. Since this type of field research has not been previously conducted on loblolly pine in the Atlanta area, it represents a major addition to our current literature.

Significance of the Research

The vegetation of Stone Mountain represents a unique ecosystem. Loblolly pine is the last stage of the successional forest in granite outcrop communities. Due to harsh climatic and environmental conditions typical of outcrop associations, these species tend to have a much lower growth rate and often exhibit a dwarfed appearance compared to loblolly pines found in the Piedmont region (Houle and Delwaide 1991). This same environment is probably responsible for its high sensitivity to stress and climatic fluctuations. As such, the loblolly pine species of Stone Mountain are potentially more susceptible to ozone damage than those of the lowland species members. For this reason, the vegetation of Stone Mountain should be assessed for ozone damage to ensure the optimal health of this ecosystem. It is possible through adaptations to this stark environment that these loblolly pine species have become tolerant to air pollutants such as ozone. It is also possible that any atmospheric chemistry change could disrupt its physiological function to the point of its eventual demise in this community. Further research on the topic of ozone-induced damage to vegetation will significantly supplement the existing bank of knowledge scientists have accumulated on this subject.

It is important for research to be conducted in major metropolitan areas such as Atlanta to determine if the vegetation has been adversely altered by ozone. In large urban areas, air quality is closely tied to transportation as vehicle emissions represent a large portion of pollution. In

1998, Atlanta was deemed in violation of the Clean Air Act and the 13-county metropolitan area was designated as nonattainment. As a result, the use of federal transportation funds were restricted by the federal government and national publicity about Atlanta's air pollution and traffic problems increased. In response, the Georgia General Assembly established the Georgia Regional Transportation Authority (GRTA) in 1999 to combat air pollution through addressing issues such as traffic congestion and poor planning developments throughout the Atlanta region.

With the amount of developed land increasing three times as fast as population, Atlanta is losing approximately 50 acres of trees per day (GRTA 2004). Atlanta's pollution problem is not getting any better: the 13-county metropolitan area is still in nonattainment. Therefore, it is greatly important to understand how ozone disturbs the health of Atlanta's urban forest ecosystems. Knowing where ozone impacts vegetation in the region can lend clues about pollutant emission sources and long-range transport. Such information is vital to policymakers working to minimize Atlanta's pollution problems to bring the city back into attainment with the Clean Air Act and to preserve the health of it citizens and urban forest ecosystems.

Study Area

The Atlanta area can be used as a "natural" laboratory for examining the impacts of ozone on trees. Atlanta is one of the largest metropolitan areas (Figure 1) in the Southeast and is the only metropolitan area in the region currently listed as a "serious" 1-hour ozone nonattainment area (U.S. EPA 2004b.). Five to twenty-eight percent of days during the 2000-2003 ozone seasons exceeded federal air quality standards (GDNR, EPD 2003). Despite its history of nonattainment, Atlanta's urban forests have not been the focus of any major studies on ozone exposure impacts to vegetation. Based on the Forest Inventory Analysis of the USDA Forest

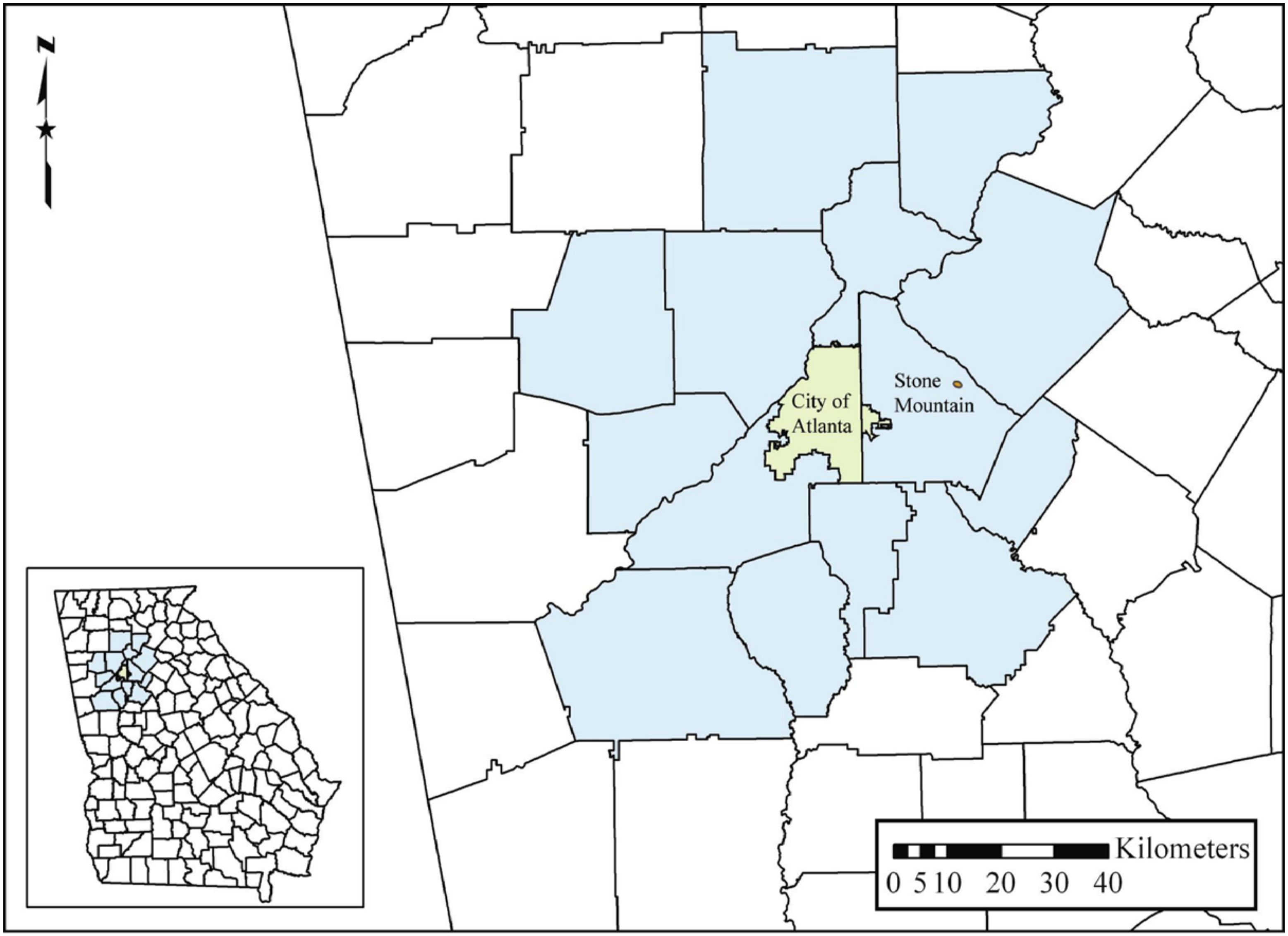

Figure 1. Map of metropolitan Atlanta ozone nonattainment area.
The 13-county metropolitan region is represented by counties shaded in blue. The City of Atlanta is represented by the green area, and Stone Mountain by the orange-colored mound. Source: Diane M. Styers, 2004.

Service (2004a.), loblolly pine accounts for a majority of urban tree cover types in and around Atlanta and has also been shown to be at least moderately sensitive to ozone exposure (Taylor 1994; Hogsett et al. 1997; Barbo et al. 2002). It is thus imperative to conduct a field examination of ozone damage on this major land cover type and its associated understory species in order to evaluate the health of Atlanta's urban forest ecosystems.

The study area is located on Stone Mountain (Figure 2) approximately 26 kilometers east/northeast of downtown Atlanta, Georgia. Stone Mountain is a granite outcrop ecosystem situated on of about 236 hectares of exposed granite (Stone Mountain Park 2004). The collection sites are located at approximately UTM 16N 764223E 3744333N (WGS84/NAD83) and range approximately 252 meters in elevation from 261 to 513 meters above mean sea level (amsl) (Stone Mountain Park 2004).

This study area was chosen for several reasons: 1) Stone Mountain is a unique ecosystem with species that have been shown to be sensitive to ozone; 2) this ecosystem is near a major urban area that has violated the federal ozone standard in recent years; and 3) there is a potentially high amount of ozone exposure to the vegetation of Stone Mountain due to its climate and physical setting. Atlanta's latitude and altitude result in a long growing season, thus giving the vegetation ample time for ozone uptake (April to October). Furthermore, the vegetation on Stone Mountain is likely to have higher ozone exposure potential than that of lower Piedmont locations due to its mountain-top setting (approximately 513 meters amsl). The results of this study should complement previous research on ozone-induced needle damage to loblolly pine in a natural, urban setting. Additionally, it will provide relevant data on ozone damage to vegetation in metropolitan Atlanta, an area of the Southeast that has been greatly overlooked in this field of research.

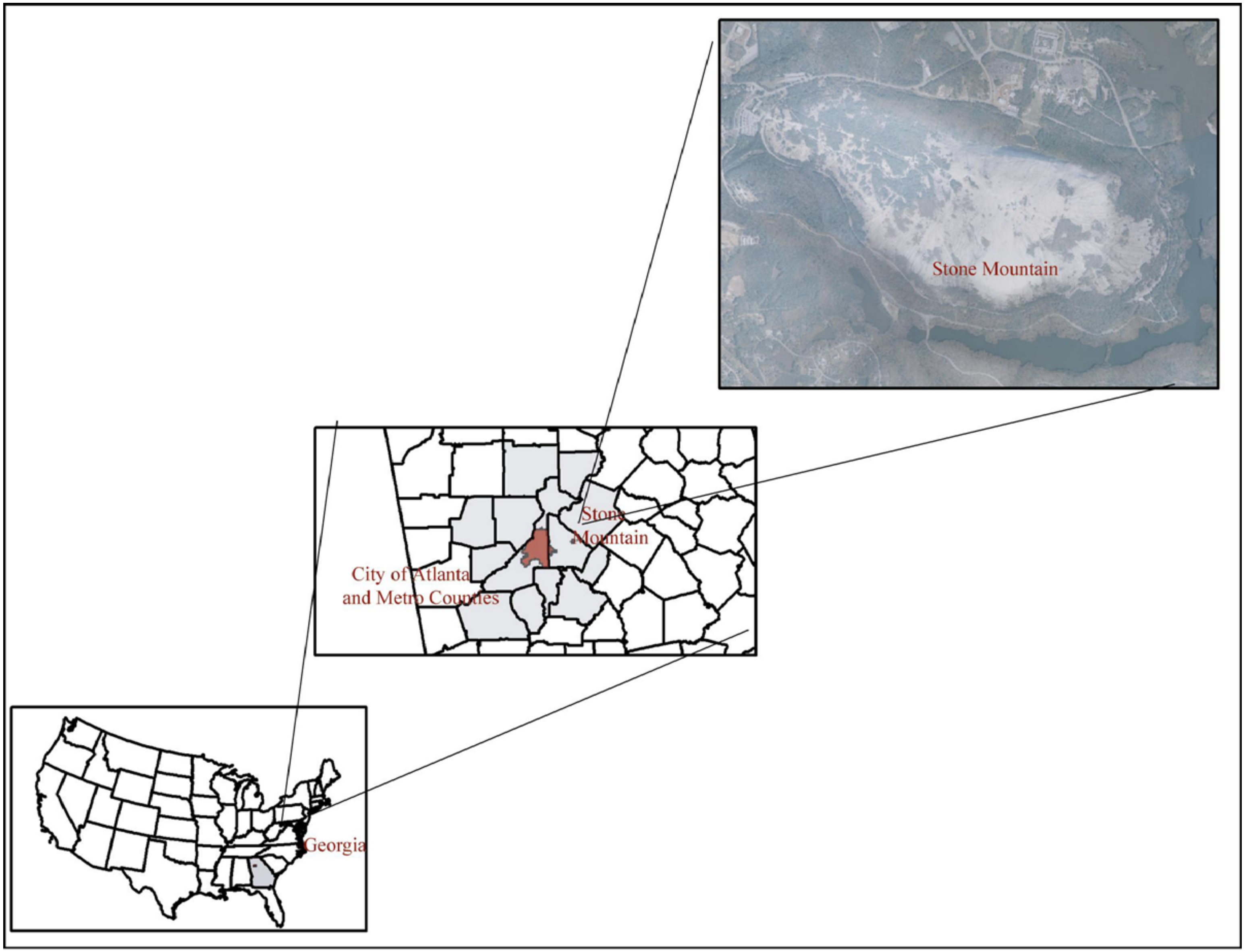

Figure 2. Map of study area: Stone Mountain in Georgia.
Source: Diane M. Styers, 2004.

Chapter 2. Data and Methods

Data and Methods

To fully assess the presence and extent of ozone-induced foliar injury to vegetation on Stone Mountain, upper-level wind data, ozone-concentration data and plant-leaf data were examined. Wind data were collected to establish monthly and seasonal wind direction and speed. These data will establish pollution transport trends from urban emission sources in Atlanta. Ozone levels in metropolitan Atlanta, specifically near the study area, were also evaluated by analyzing ozone-concentration data for four ozone seasons. Standard monthly and seasonal ozone indices were calculated to determine if ambient ozone concentrations are sufficient to damage vegetation in metropolitan Atlanta. Plant data were collected in early August 2004 to determine the presence and degree of foliar injury at each sampling site and damage extent across the study area.

Data

Upper-level atmospheric data were obtained from National Climatic Data Center for the years 1996-2002. Data were collected for the Peachtree City Station (#72215; UTM 16N 727058E 3692748N, Elevation 246 meters amsl) at the 850 millibar (mb) level for 12Z. These data were utilized because they comprise the longest record without excessive missing data. The wind data were used to establish lower-tropospheric wind directions and velocities for the metropolitan Atlanta area. Seasonal wind patterns were determined through the compilation of hourly data for the months May to September for the previous eight years.

Hourly ozone data were acquired from the Georgia Department of Natural Resources, Environmental Protection Division, Air Quality Branch for the 2000 to 2003 ozone seasons (April to September). These data were used because only the past two to four years are relevant when examining pine foliage, as most species tend to drop their needles after a few growing seasons. Data was collected for the following six stations (Figure 3) surrounding Stone Mountain: Atlanta (131210055), Conyers (132470001), Henry (131510002), Lawrenceville (131350002), South Dekalb (130890002) and Tucker (130893001). These hourly data allowed the calculation of standard monthly and seasonal ozone indices including daily 8-hour maximum, SUM0, SUM60, W126 and AOT40.

Plant-leaf data were collected from field studies on Stone Mountain. Loblolly pine, blackberry, sumac and black cherry species were examined for ozone-induced foliar injury, as these are the dominant plant species on Stone Mountain with well-documented sensitivity to ozone (refer to *Chapter 1*). These analyses (discussed in more detail in the *Methods* section) produced data describing the percent of species within the study area exhibiting potential ozone injury and the degree of severity of damage to each injured species.

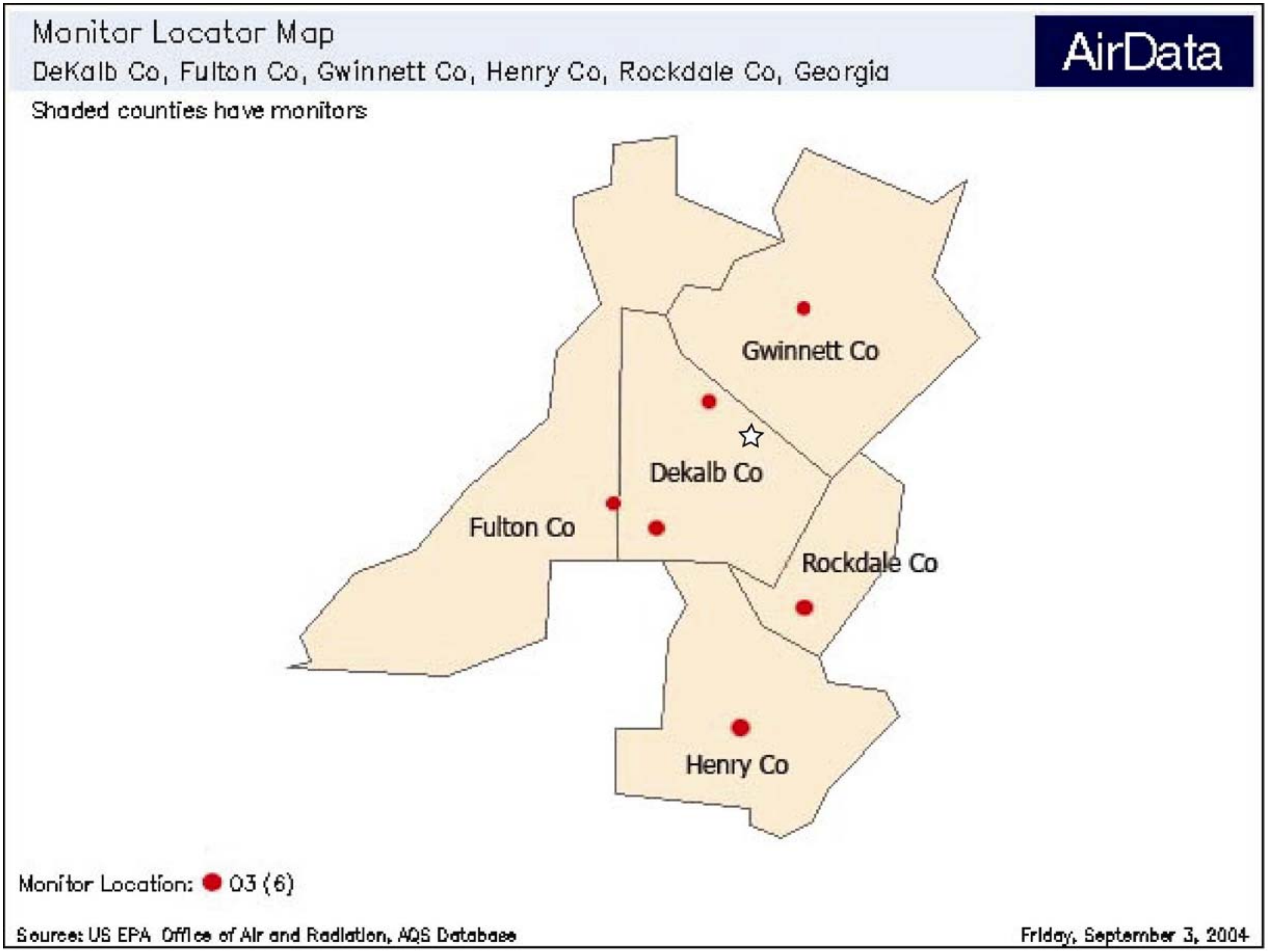

Figure 3. Locations of metropolitan Atlanta ozone monitors used in this study.
(Star indicates the location of Stone Mountain.) Source: United States Environmental Protection Agency, 2004. Alterations by Diane M. Styers, 2004.

Methods

Hourly wind data at the 850 mb level were aggregated to monthly and seasonal (May to September) averages to determine lower-tropospheric and near-surface wind directions and velocities for the Atlanta area. These averages were used to determine the location of Stone Mountain in relation to Atlanta where wind patterns could transport ozone from emission sources located in and around the city. Monthly wind speeds were assessed for relevance in ozone production, as stagnant air with low winds represent the ideal condition for ozone formation. Monthly wind directions were also evaluated to determine which months have the highest potential for pollutant transport from Atlanta. Pollutants transported could include ozone, as well as ozone precursors such as NO_X and VOCs. Monthly wind directions were compared to corresponding monthly ozone concentrations at the six monitors located between Atlanta and Stone Mountain. This assessment should help determine the role of pollutant transport from metropolitan Atlanta to the vegetation located on Stone Mountain.

An examination of historical hourly ozone data was conducted to determine standard ozone indices. Daily 8-hour maximum is defined as the maximum 8-hour average concentration recorded at a given monitor (U.S. EPA 1990). The daily 8-hour maximum standard is currently set by the U.S. EPA (1990) at 0.08 parts per million (ppm), or 80 ppb, with the third highest daily maximum concentration averaged over the previous three years. SUM0 is defined as the daily sum of all valid[1] ozone hours for April through September (USDA Forest Service 2004b.). "The SUM0 statistic is computed for all days with valid data during the year or selected/growing season. Units are PPB-HRS. N is the number of days during the year and selected/growing season with calculated daily SUM0 exposure" (USDA Forest Service 2004b.). SUM60 represents the daily sum of all valid ozone hours greater than or equal to 60 parts per billion for

[1] Valid ozone hours are those hours with recorded data.

April through September (USDA Forest Service 2004b.). "The SUM60 statistic is computed for all days with valid hourly ozone concentrations equaling or exceeding 60 ppb during the year and selected/growing season. Units are PPB-HRS. N is the number of days during the year or season with one or more hourly ozone concentrations equaling or exceeding 60 ppb" (USDA Forest Service 2004b.). W126 is an ozone index that multiplies each specific concentration by a sigmoidally weighted function, giving greater emphasis to higher hourly concentrations but still including the lower ones, then sums all values (USDA Forest Service 2004b.). The W126 weighting function provides a weighting value that is unique for each hourly ozone concentration. The weighting function, as described by [Lefohn and Runeckles 1987] is: W126 = SUM n, I=1 (wi ci) where W126 = monthly W126 exposure index, wi = weighted value for hourly concentration I and ci = hourly ozone concentration" (units are in ppm; USDA Forest Service 2004b.). AOT40 ("accumulated exposure over a threshold of 40 ppb" for daylight hours during a growing season) is a European standard that indicates the sum of the differences between hourly ozone concentration and 40 ppb for each hour when the concentration exceeds 40 ppb (European Environment Agency 2004c.). AOT40 is calculated by taking the "sum of the difference between hourly concentrations greater than 40 ppb and 40 ppb over a given period using only the one hour values measured between 8:00 and 20:00 each day" (British and Irish Legal Information Institute 2004).

Thirty sampling sites with an approximate ten-meter diameter were randomly selected based on satellite imagery obtained from the United States Geologic Survey (USGS 2002) for Stone Mountain (resolution approximately 0.3 meters) prior to entering the field (Figure 4). A global positioning system receiver was used to locate each of the sample sites once in the field. A portable database was constructed using a Trimble® GeoExplorer® Geo XT™ handheld

Figure 4. Locations of Stone Mountain field sampling sites (each site ~10m diameter).
Note: Elevation increases from northwest (base) to southeast (peak). Source: Diane M. Styers, 2004.

device to record relevant sample site information including point location (in UTM coordinates), vegetation type and the presence and extent of ozone-induced foliar injury at the time of data collection. The total number of species affected per site in relation to the total number of species present per site was noted and expressed as a percentage. Foliar injury on each affected plant species was classified into four categories: 0=none, 1=minimal, 2=moderate and 3=severe. The general overall vegetation condition of each site was noted as well.

A survey of visible foliar injury was conducted on Stone Mountain in August 2004. Ozone-injury data were collected from a minimum of 30 sample sites on the dome of Stone Mountain. Loblolly pine species at the base of Stone Mountain could not be visually assessed with a hand lens since the heights of the lowest branches were greatly out of sight and reach. The leaves of each mature loblolly pine species ($\geq$10 cm diameter at breast height, or dbh) and each understory species ($\geq$ 1 cm dbh) within each sampling site were examined for ozone-induced foliar injury. Foliar injury was defined as chlorotic mottling and premature senescence for loblolly pines and red stippling or flecking for blackberry, sumac and black cherry species. Although the examination of ozone-induced injury to loblolly pine was the main objective of this study, it is important to assess the health of other co-existing sensitive species to provide supporting evidence that the injury observed was likely due to ozone exposure rather than another variable.

The following variables related to ozone-induced foliar injury were used to estimate damage to the four sensitive species selected for examination on Stone Mountain: 1) ratio of number of species injured per site to number of species per site; 2) mean degree of foliar injury per plant, per site; 3) ratio of number of species injured per study area to number of species per study area; and 4) percent total injury per study area. Overall vegetation condition was noted for each site based on a broad visual assessment of the health of the whole plant.

Chapter 3. Results and Discussion

Pollutant Transport Potential

Hourly wind data was obtained from for the years 1996-2002 from the Peachtree City Station (#72215) at the 850 mb level. The results from this wind data analysis established lower-tropospheric and near-surface wind direction and velocity for the metropolitan Atlanta area, as determined through the aggregation of hourly data for the months May to September for each year. These results indicate that Stone Mountain is located downwind from pollutant sources in metropolitan Atlanta. As such the potential exists for ozone, NO_X and VOC transport from the Atlanta area to Stone Mountain. Ozone concentrations downwind from Atlanta are also likely greater than those found at upwind locations.

Lower-tropospheric winds at the 850 mb level over Atlanta most frequently prevail from the west (Figure 5) at approximately 11.25 kilometers per hour (kph; Table 1). These data represent the total wind direction frequency for each month of the season for all years combined. The data show that wind direction fluctuates little from May to August typically blowing from the southwest (208º-230º), then changes in September to blow in more from the south (170º). However, seasonal averages from 1999 and 2001 show a more southerly wind flow (188º and 203º, respectively) than do the other years (1996-228º, 1997-227º, 1998-219º, 2000-222º and 2002-227º). Monthly wind speeds vary throughout the season from 8.87 to 14.34 kph. Winds are highest in May (14.34 kph) and gradually decrease each month through August (8.87 kph), before slightly increasing again in September (11.11 kph). July and August have the most calm winds (10.22 and 8.87 kph, respectively), while May and June are the highest (14.34 and 11.69

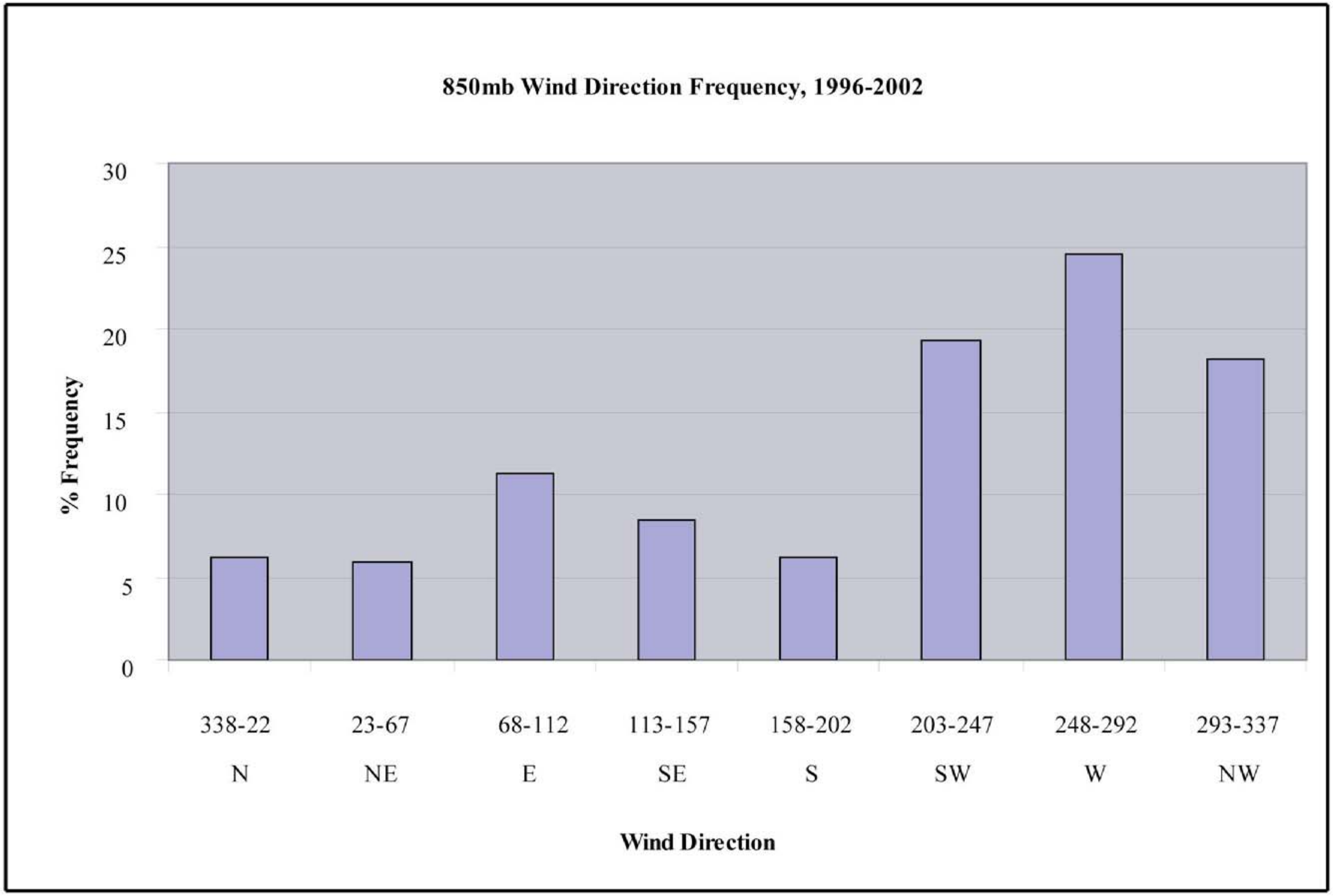

Figure 5. Histogram of 850mb wind direction frequency in metropolitan Atlanta from 1996 to 2002.
Source: Diane M. Styers, 2004.

Table 1. Mean wind speeds at 850mb in metropolitan Atlanta from 1996 to 2002.
Source: Diane M. Styers, 2004.

Speed (in kph)								
850 mb	**1996**	**1997**	**1998**	**1999**	**2000**	**2001**	**2002**	**Monthly Mean**
May	12.94	15.80	13.33	12.68	14.82	14.48	16.32	14.34
June	9.40	9.83	15.14	12.34	10.22	12.65	12.25	11.69
July	12.60	11.09	13.94	7.27	10.69	9.03	6.95	10.22
August	8.40	9.91	7.03	8.82	9.70	8.72	9.53	8.87
September	12.71	10.44	10.65	9.45	10.41	10.56	13.57	11.11
Yearly Mean	11.21	11.42	12.02	10.11	11.17	11.09	11.72	**---> 11.25**

kph, respectively). The lowest winds during the period of study were in 1999, with an average seasonal wind velocity of 10.11 kph. July and August wind speeds during 1999 were 7.27 and 8.82 kph respectively. The greatest wind speeds occurred in 1998 (12.02); however, these data still represent fairly calm wind conditions.

Wind velocity data indicates that conditions are favorable for ozone production, since the average wind speeds in Atlanta are relatively calm. This is probably due to the presence of high pressure systems associated with typical summer weather in the Southeast. Additionally, results of the wind direction data analyses indicate that there is potential for pollutant transport to Stone Mountain due to its location downwind from the Atlanta area. Ozone produced in Atlanta could possibly end up at Stone Mountain in just 2 to 3 hours after formation. Additionally, ozone precursors such as NO_X and VOCs could be transported from emission sources in Atlanta and combine to produce ozone at or near Stone Mountain in a similar frame of time.

Ambient Ozone Concentrations in Metropolitan Atlanta

Hourly ozone data was acquired for the 2000 to 2003 ozone seasons (April to September) from six monitors located near Stone Mountain. Data from these years were examined because only the past two to four years are relevant when examining pine foliage, as loblolly pines tend to drop their needles after 18 months, or two growing seasons. The stations used in this study were: Atlanta (131210055), Conyers (132470001), Henry (131510002), Lawrenceville (131350002), South Dekalb (130890002) and Tucker (130893001). These hourly data were used to calculate standard monthly and seasonal ozone indices including daily 8-hour maximum, SUM0, SUM60, W126 and AOT40.

Daily 8-hour maximum is defined as the maximum 8-hour average concentration recorded at a given monitor (U.S. EPA 1990). This average cannot exceed the daily 8-hour maximum standard, currently set at 0.08 ppm (80 ppb), more than three times in three years (U.S. EPA 2004a.). This standard was enacted by the U.S. EPA in order to monitor potential human health consequences of high ozone exposure periods at the county level. Counties that violate the Clean Air Act by exceeding the daily standard more than the three times allotted are placed on a nonattainment list maintained by the U.S. EPA. Many large metropolitan areas such as Atlanta have been on this list since this new standard took effect in June 2004 (the previous air quality standard used was based on the daily maximum 1-hour average concentration). Daily 8-hour maximum violations calculated for this study were for the summer months of June, July and August (JJA), as this is the suitable time for elevated ozone concentrations based on increased solar radiation, high temperatures and low wind speeds (Sillman 1999; Krupa et al. 2000).

Daily 8-hour maximums were exceeded for each of the six monitors used in this study. Exceedances were greatest at the Atlanta monitor (131210055) for JJA from 2000 to 2003, with a total of 66 exceedances for the four-summer season period. This result is to be expected as this monitor is located in downtown Atlanta amongst many sources of NO_X and VOC emissions. Of greater note is that the Tucker monitor (130893001), located downwind from Atlanta and approximately 8 km WNW of Stone Mountain, was ranked second for overall exceedance values (60 exceedances for JJA, 2000-2003), displaying ozone concentrations near that of the Atlanta monitor. The remaining four monitors in proximity to the study area also displayed a high number of exceedances during this period: Henry (131510002) had 54, South Dekalb (130890002) had 43, Lawrenceville (131350002) had 39 and Conyers (132470001) had 35. These numbers reveal that Atlanta area ozone concentrations exceeded the daily 8-hour

maximum standard each year examined during the summer months alone. Had the entire ozone season been considered in this analysis (March to October), the number of exceedances per year could have been even greater. Out of a total 72 monthly exceedances (3 months of 4 years at 6 monitors), only 6 months had zero exceedances or 8% of the overall time period. This is worth noting as it reveals periods of low ozone levels; yet, since the daily 8-hour maximum standard cannot be exceeded more than three times in three years these zero monthly values are insignificant on this smaller temporal scale. A summary of these data appears in Table 2.

Regarding ozone concentrations on a seasonal basis throughout the study period, there is a bimodal distribution of exceedance values. The summer of 2000 had the highest number of exceedances by far, at 151 for all monitors combined, which is 49% greater than the next highest exceedance season (2002 – 77) and 77% greater than the other two seasons (2001 – 35 and 2003 – 34). The fact that the summer of 2003 had the least number of exceedances is important to note, as ozone impacts to loblolly pine during the 2003 season could still be evident on the needles of the previous season's whorls.

For comparison purposes, the daily 8-hour maximum was exceeded at the Crestline ozone monitor in San Bernardino County in California 71 days in 2003 (Figure 6; California Air Resources Board 2005). The Atlanta monitor had 7 exceedances for the peak months (JJA) of the ozone season in 2003, a mere 10% of the number of exceedances at Crestline. Although San Bernardino County represents an area of severe ozone, Atlanta area monitors considerably surpass the allowed number of ozone exceedances each year. As such, the values found for Atlanta are still noteworthy, as plant exposure to these levels of ozone are sufficient to cause injury.

Table 2. Summary data of daily 8-hour maximum ozone exceedances for Atlanta. Data presented are for six Atlanta-area monitors from 2000 to 2003 (JJA). Source: Diane M. Styers, 2004.

S. Dekalb					
1308900021	**2000**	**2001**	**2002**	**2003**	**Monthly Total**
June	4	0	5	4	13
July	7	2	4	0	13
August	10	2	4	1	17
Season Total	21	4	13	5	**43**

Tucker					
1308930011	**2000**	**2001**	**2002**	**2003**	**Monthly Total**
June	7	3	5	4	19
July	12	4	3	2	21
August	11	2	5	2	20
Season Total	30	9	13	8	**60**

Atlanta					
1312100551	**2000**	**2001**	**2002**	**2003**	**Monthly Total**
June	7	0	5	4	16
July	14	3	5	2	24
August	14	3	8	1	26
Season Total	35	6	18	7	**66**

Lawrenceville					
1313500021	**2000**	**2001**	**2002**	**2003**	**Monthly Total**
June	6	1	3	4	14
July	4	2	2	1	9
August	9	2	3	2	16
Season Total	19	5	8	7	**39**

Henry					
1315100021	**2000**	**2001**	**2002**	**2003**	**Monthly Total**
June	4	0	5	3	12
July	15	3	3	1	22
August	12	2	4	2	20
Season Total	31	5	12	6	**54**

Conyers					
1324700011	**2000**	**2001**	**2002**	**2003**	**Monthly Total**
June	1	1	5	1	8
July	3	3	4	0	10
August	11	2	4	0	17
Season Total	15	6	13	1	**35**

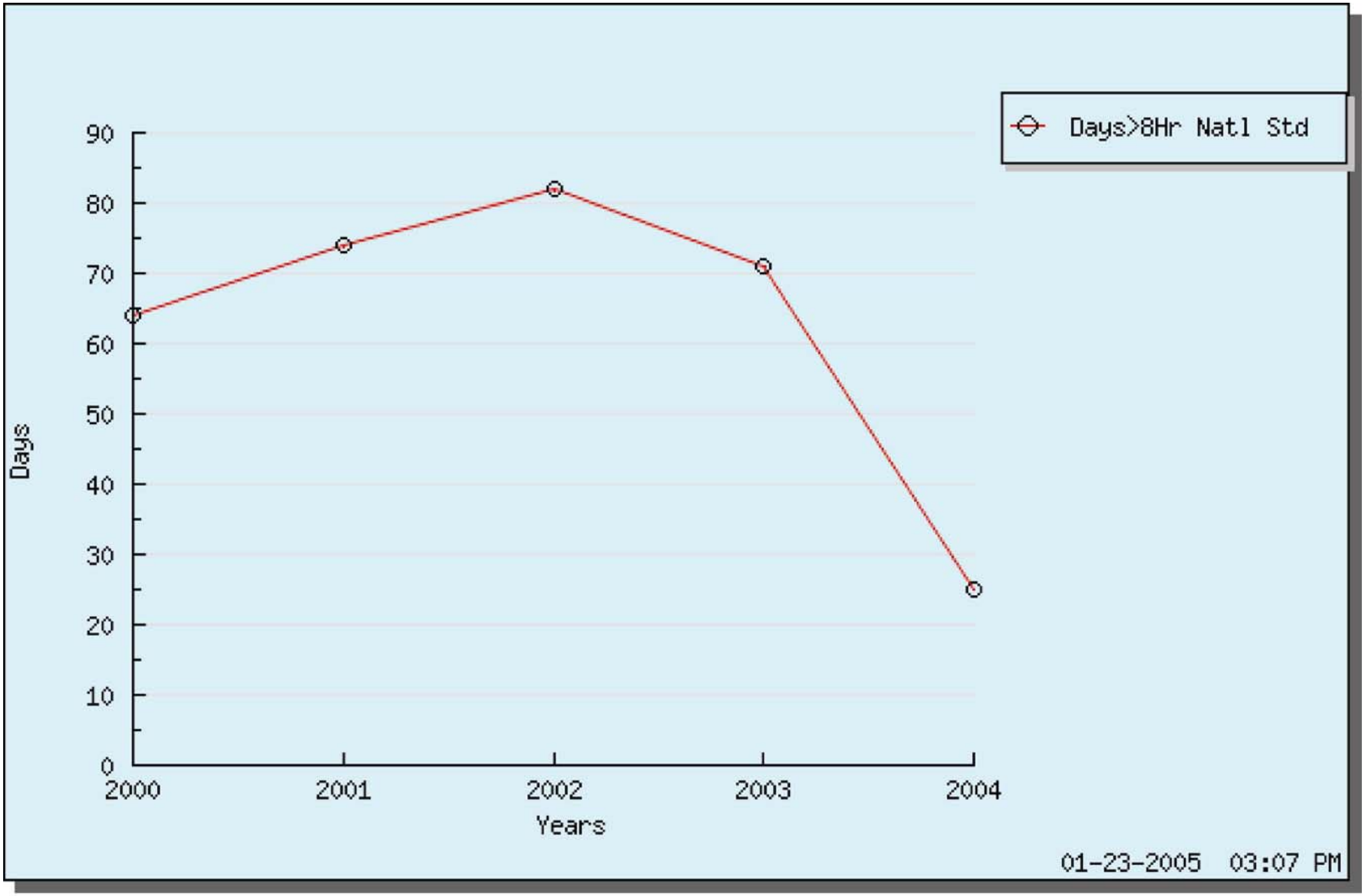

Figure 6. Daily 8-hour maximum exceedances at the Crestline ozone monitor in San Bernardino County, California from 2000 to 2004.
Source: California Air Resources Board, 2005.

SUM0 is defined as the daily sum of all valid ozone hours for April through September (USDA Forest Service 2004b.). SUM0 is one of several indices used by the UDSA Forest Inventory and Analysis Ozone Biomonitoring Program to quantify ambient ozone concentrations into seasonal ozone exposures that characterize typical forest response (USDA Forest Service 2004b.). Unlike the daily 8-hour maximum, which was established by the U.S. EPA to monitor human health, SUM0 and other OBP indices are used to evaluate ozone exposure risks to vegetation health. The SUM0 index places a greater emphasis on lower ozone concentrations than higher ones (Lefohn et al. 1989). The SUM0 statistic represents the total hourly ozone concentration (>0 ppb) recorded for a given monitor. These concentrations can then be aggregated to total daily, monthly and seasonal ozone exposure values.

For the six Atlanta area monitors, SUM0 values were calculated from hourly concentrations to daily totals which were then aggregated to monthly and seasonal total values (Figure 7). Ozone values for each monitor location increase steadily beginning in April, reaching peak ozone concentrations in July and then decreasing thereafter with a sharp drop occurring in September. The highest SUM0 value occurs at the Tucker monitor (130893001) at 26,551 ppb for July over the four-year period; the lowest occurs at the South Dekalb monitor (130890002) at14,490 ppb for September over the four-year period.

SUM60 is defined as the daily sum of all valid ozone hours greater than or equal to 60 parts per billion for April through September (USDA Forest Service 2004b.). SUM60 is an index used by the OBP to monitor plant exposure potential, as ozone concentrations over 60 ppb are generally sufficient to damage plant tissue of sensitive species. Long-range exposure to high levels of ozone could possibly result in reduced photosynthesis, diminished growth, biomass

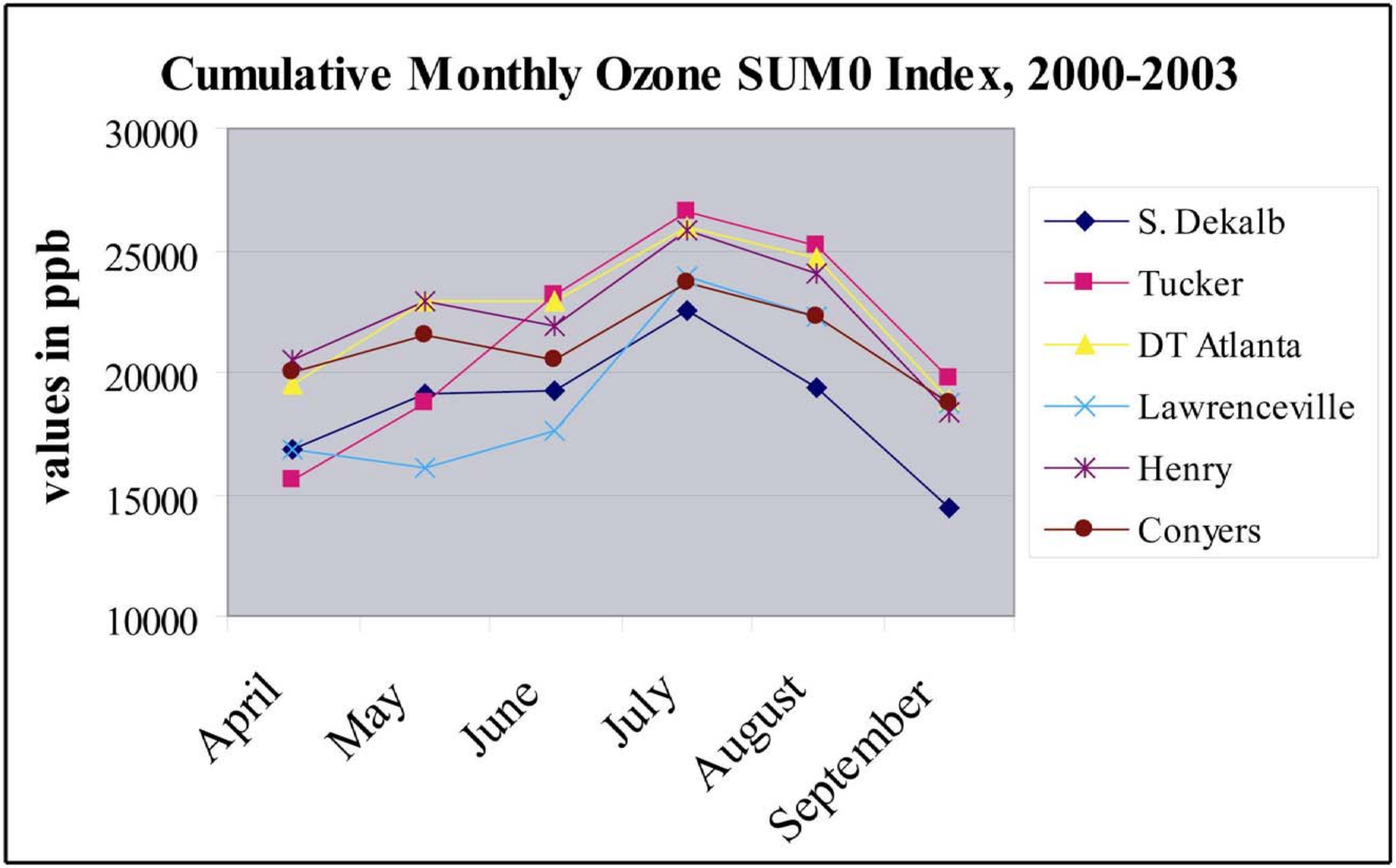

Figure 7. Cumulative monthly ozone SUM0 index for six Atlanta monitors from 2000 to 2003.
Source: Diane M. Styers, 2004.

loss and foliar injury[2], even in less sensitive species. As such, the SUM60 index is more useful for evaluating plant exposure potential than SUM0, which emphasizes lower ozone concentrations (Lefohn et al. 1989). The SUM60 statistic represents the total hourly ozone value (>60 ppb) recorded for a given monitor.

SUM60 values were calculated for the six area monitors from hourly concentrations into daily totals which were then aggregated to monthly and seasonal total values (Figure 8). Ozone concentration trends are similar to those of the SUM0 results. Again, the highest SUM60 value occurs at the Tucker monitor (130893001) at 12,335 ppb for July over the four-year period; interestingly, the lowest also occurs at the Tucker monitor at 2,323 ppb for April over the four-year period. SUM60 values for each of the six monitors display a similar increase in ozone concentrations from low April values to July and August highs with a steep drop occurring in September where values return to near April levels.

W126 is an ozone index that multiplies each specific hourly concentration (in ppm) by a sigmoidally weighted function, giving greater emphasis to higher hourly concentrations but still including the lower ones, then sums all values (USDA Forest Service 2004b.; Lefohn and Runeckles 1987). The W126 index is heralded by some to be a more accurate index than SUM0 or SUM60 as it is more limiting than SUM0, weighting higher concentrations more heavily than lower ones, yet less restrictive than SUM60 as it does not completely exclude values less than 60 ppb (6 ppm; Lefohn et al. 1989). As such, the W126 statistic represents the total sigmoidally weighted hourly ozone value recorded for a given monitor.

---

[2] Ozone exposure concentrations do not always indicate the amount of ozone absorbed by a plant because processes responsible for ozone absorption are highly variable and differ among species and environments.

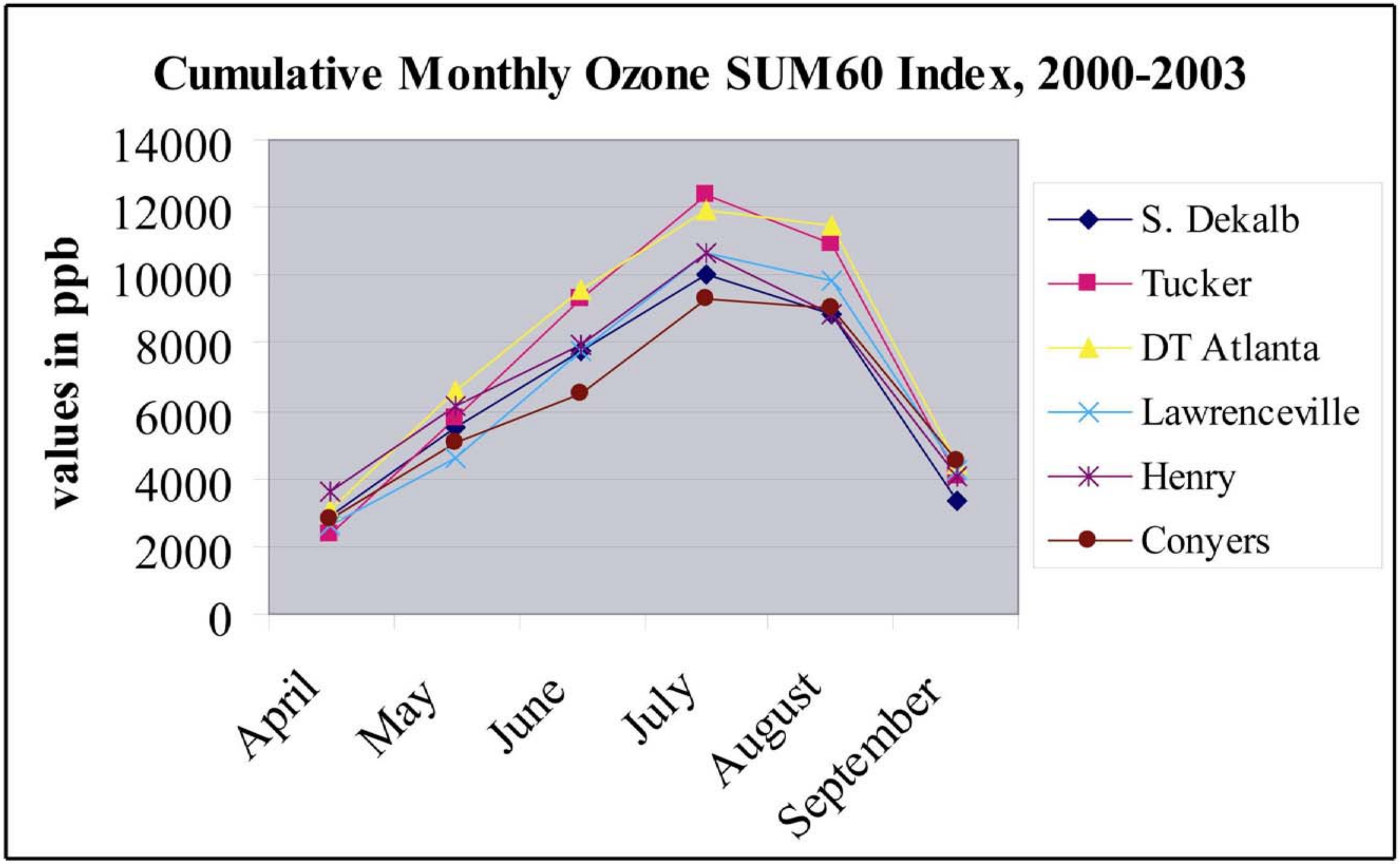

Figure 8. Cumulative monthly ozone SUM60 index for six Atlanta monitors from 2000 to 2003.
Source: Diane M. Styers, 2004.

W126 values were calculated for the Atlanta monitors from hourly concentrations into daily totals then aggregated to monthly and seasonal total values (Figure 9). W126 values are similar to those of the SUM60 values. Results from statistical analyses of previous studies have found that SUM60 and W126 values are highly correlated (Diem 2002). The results obtained from this project concur with those findings. Again, the highest W126 value occurs at the Tucker monitor (130893001) at 127.01 ppm (127,010 ppb) for July over the four-year period; the lowest also occurs at the Tucker monitor at 32.28 ppm (32,280 ppb) for April over the four-year period. W126 values for each of the six monitors display a similar trend in seasonal ozone concentration as the SUM0 and SUM60 results.

AOT40, or "accumulated exposure over a threshold of 40 ppb", is a European standard that indicates the sum of the differences between hourly ozone concentration and 40 ppb for each daylight hour (8:00 – 20:00) when the concentration exceeds 40 ppb (European Environment Agency 2004c.). This index was developed to assess ozone exposure potential to vegetation across Europe (European Environment Agency 2004d.). In fact, ozone exceedance limitations protecting vegetation are nearly half that of those enacted to protect human health (European Environment Agency 2004a. & b.).

AOT40 values were calculated for the six monitors from hourly concentrations (only the hours from 8:00 to 20:00 were included) to daily totals then aggregated to monthly and seasonal total values (Figure 10). AOT40 concentration trends were similar to those of the SUM60 and W126 values. The highest W126 value occurs at the Tucker monitor (130893001) at 12,675 ppb for July over the four-year period; the lowest also occurs at the Tucker monitor at 3,297 ppb for April over the four-year period. Thus, the European AOT40 index supports the results from those used to calculate forest risk in the U.S.

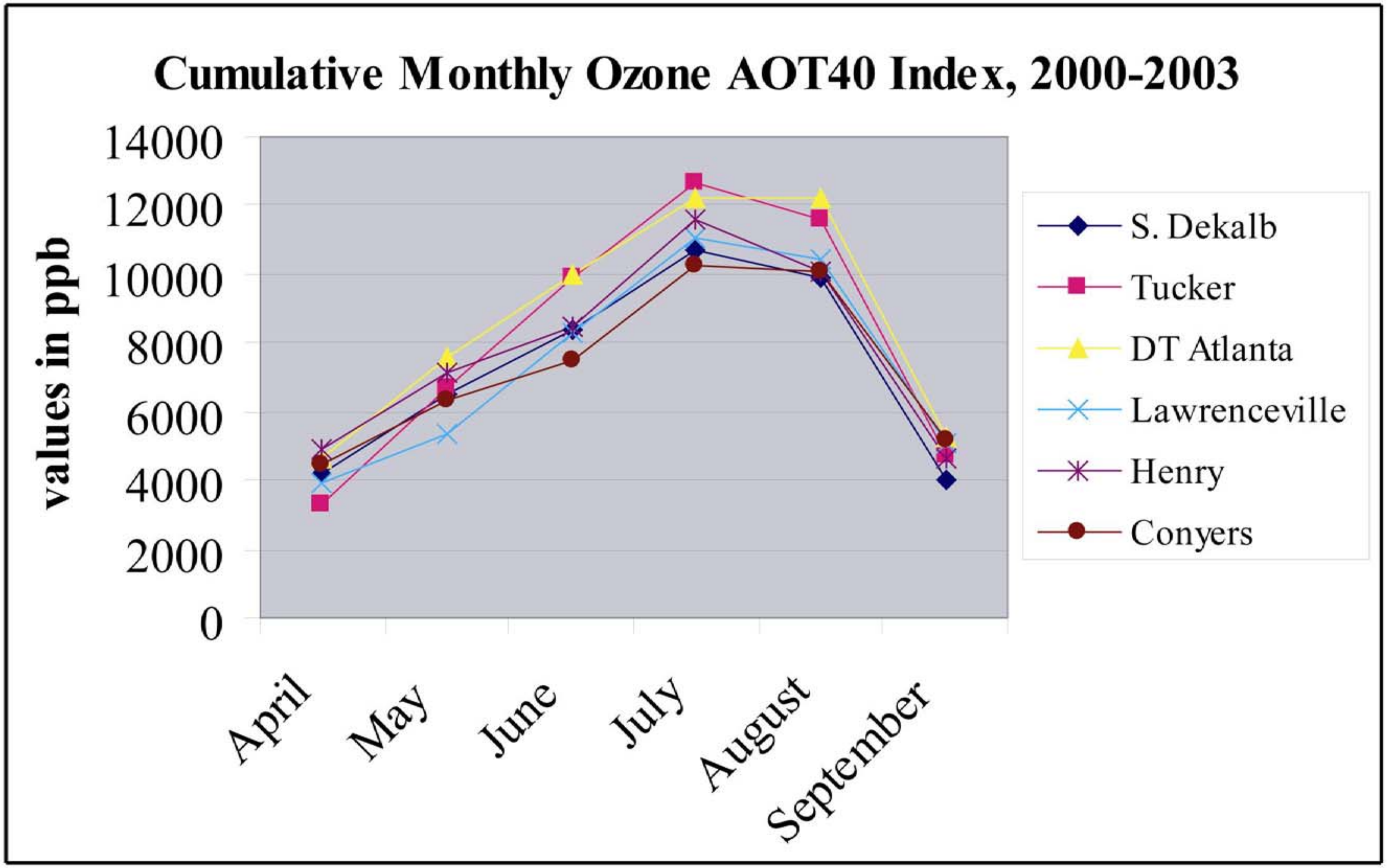

Figure 9. Cumulative monthly ozone AOT40 index for six Atlanta monitors from 2000 to 2003.
Source: Diane M. Styers, 2004.

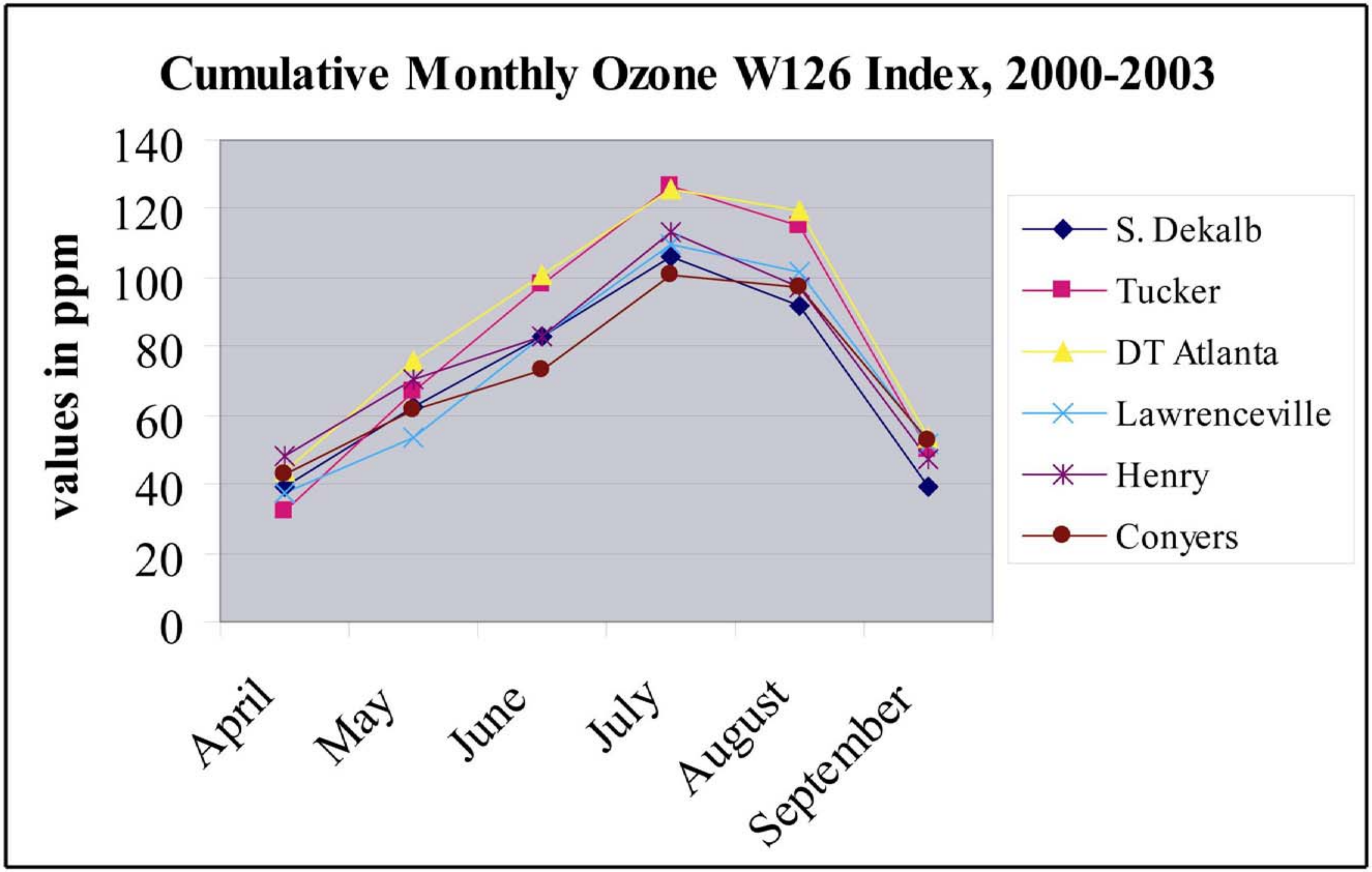

Figure 10. Cumulative monthly ozone W126 index for six Atlanta monitors from 2000 to 2003.
Source: Diane M. Styers, 2004.

Analysis of these various ozone indices revealed that ambient ozone concentrations at metropolitan Atlanta monitors are high enough to damage vegetation exposed to those levels. The highest concentration values were found at the Tucker monitor, located downwind from Atlanta and in close proximity to Stone Mountain (8 km WNW). These concentrations are frequently well above the 60 ppb necessary to injure vegetation. These results provide sufficient evidence to support the possibility of finding ozone-damaged vegetation on Stone Mountain.

Visible Foliar Injury to Vegetation on Stone Mountain

Plant-leaf data was collected in early August 2004 from sample sites located on Stone Mountain for loblolly pine, blackberry, sumac and black cherry species. All plant species located within the approximate 10-meter diameter of each of the 30 sample sites were examined for ozone-induced foliar injury (Figure 11 a. & b.). Loblolly pine was the major plant species present on Stone Mountain. It was located in abundance within each of the 30 sample sites. Since several sites consisted of only loblolly pine (those with none of the other three species of concern present), these were the sites used to examine the health of loblolly pines. Needles of only the lower branches within reach were examined, as injury usually begins on the lower branches and works up the tree. Needles of trees located on sites with an adequate amount of any of the other species were also; however, if no damage was detected, these species were not counted in the total number of species for that site. This decision was made during the course of the field work due to the large amount of loblolly pines present. Since the total number of loblolly pines analyzed would not alter the results of this study, it was assumed that this decision was reasonable. This resulted in a better overall analysis of Stone Mountain as a whole ecosystem. A summary of dominant plant distribution per site appears in Table 3.

a. 

b. 

Figure 11 a. Typical loblolly pine patch site and b. typical understory site on Stone Mountain. Source: Diane M. Styers, 2004.

Table 3. Dominant plant distribution per site.
Source: Diane M. Styers, 2004.

Dominant Species	# of sites	% of total sites	total # of species	total % of species
Blackberry	2	7	17	4
Sumac	10	33	142	15
Blackberry/Sumac	2	7	21 b / 21 s	N/A (incl. above)
Black Cherry	3	10	15	1
Loblolly only	13	43	881	80
ALL	30	100	1,097	100
ALL except loblolly	17	57	216	20

Of the 881 loblolly pine species within thirteen sites (80% of total species) assessed on Stone Mountain, none of the needles examined exhibited chlorotic mottling or any other signs of foliar injury (Figure 12a.). There did appear to be senescence of the previous season's needles on several of the species, especially at higher elevations (Figure 12b.), but this cannot be absolutely attributed to ozone damage as other variables can not be ruled out. Other factors that can cause premature senescence include drought and invasion by pests and disease (Skelley et al. 1985). Additionally, loblolly pine species occurred on all slope aspects and at all elevations.

Thirty-eight blackberry species within four sites (two of these sites were evenly mixed with blackberry and sumac species) were examined for foliar injury. Blackberry species represented 4% of the total number of species and 7% of the total sites examined as a part of this study (blackberry-sumac sites represented an additional 7% of the total sites). Five species within one of the blackberry-only site showed no signs of foliar injury (Figure 13a.). One of the blackberry-only sites had severe damage on all twelve species located within the site (Figure 13b.). This site was located at the base of Stone Mountain near the park's sight-seeing train tracks. It is possible that exposure to daily high levels of NO_X emissions from the train accounts for the injury present in these species, as no other blackberry species on Stone Mountain exhibited such severe damage. This is evidence that ozone production at Stone Mountain is NO_X-limited. Overall, 71% (12 out of 17) of the blackberry species examined within the blackberry-only sites exhibited severe foliar injury. This phenomenon occurred in 50% (1 out of 2) of the blackberry sites.

a.

b.

Figure 12 a. Healthy loblolly pine needles with no chlorotic mottling and b. dead loblolly pine needles exhibiting senescence of last season's needles on Stone Mountain.
Source: Diane M. Styers, 2004.


a.


b.

Figure 13 a. Healthy blackberry plant and b. blackberry leaves showing severe foliar injury on Stone Mountain. Source: Diane M. Styers, 2004.

Within the two mixed blackberry-sumac sites, species distribution was evenly spread. Twenty-one blackberry and twenty-one sumac species were examined for foliar injury. In one site, 17 out of 28 species (61%) exhibited moderate foliar injury (11 blackberry and 6 sumac). In the other site, 3 out of 14 species (21%) exhibited minimal foliar injury (2 blackberry and 1 sumac). Overall, 48% of total blackberry and sumac species (62%, or 13 out of 21 and 33%, or 7 out of 21, respectively) examined within the mixed sites exhibited minimal to moderate foliar injury. Foliar injury was present in 100% (2 out of 2) of the mixed blackberry-sumac sites.

Sumac species (163 plants) within twelve sites (two of these sites were evenly mixed with blackberry and sumac species) were examined. Sumac species represented 15% of the total number of species and 33% of the total sites examined as a part of this study (blackberry-sumac sites represented an additional 7% of the total sites). Thirteen sumac species within three sites and the remaining sixty-three species within seven of the other sites exhibited no foliar injury (54% of species within sumac-only sites; Figure 14a.). Twenty-two out of fifty-nine species (37%) within four sites showed minimal damage. Forty-one out of fifty-four species (76%) within two sites exhibited moderate damage. Three out of sixteen species (19%) within one site showed severe injury (Figure 14b.). Overall, 46% (66 out of 142) of the sumac species examined within the sumac-only sites exhibited minimal to severe foliar injury. This damage occurred in 70% (7 out of 10) of the sumac sites. However, sumac leaves can display red stippling in response to many types of invasion other than by ozone (Skelley et al. 1985). On sumac plants where red stippling occurred with other types of leaf damage (black spots, bite areas, etc.), ozone may not be the only possible cause of foliar injury (Figure 15). Without chemical or physiological analyses, foliar injury on sumac cannot be entirely attributed to ozone damage.

a. 

b. 

Figure 14 a. Healthy sumac plant and b. sumac leaves showing severe foliar injury on Stone Mountain. Source: Diane M. Styers, 2004.

Figure 15. Sumac plant with pest and/or disease damage (red coloring occurs in areas surrounding brown spots and bite areas). Source: Diane M. Styers, 2004.

Fifteen black cherry species within three sites were examined for foliar injury. One of the black cherry sites had no signs of damage on any of the seven species located within the site (Figure 16a.). Within the other two sites, four out of eight (50% each site) black cherry plants exhibited moderate foliar injury (Figure 16b.). Overall, 27% (4 out of 15) of the black cherry species examined these sites exhibited moderate foliar injury. This damage was present in 67% (2 out of 3) of the black cherry sites. Black cherry species represented 1% of the total number of species and 10% of the total sites examined as a part of this study.

Of the total 1097 species examined, only 102 (9%) plants within the 30 study sites located on Stone Mountain exhibited some degree of foliar injury (Table 4). None of the 881 loblolly pines appeared to be damaged as assessed by the presence of chlorotic mottling. Twelve out of thirty sites (40%) contained plants with some degree of foliar injury, where five sites and 17% of the damage found was minimal; five sites and 17% of the damage found was moderate; and two sites and 7% of the damage found was severe. If loblolly pine species are excluded from this analysis (as they accounted for 43% of the sites but none of the 881 plants exhibited any damage) the results are slightly different. The damaged plants would still exhibit an overall average minimal degree of injury, but twelve out of the seventeen (or 72%) sites contained plants with some degree of damage. Five sites and 23% of the damage found would be minimal; five sites and 34% of the damage found would be moderate; and two sites and 15% of the damage found would be severe. A summary of species damage severity appears in Table 5.

a.

b.

Figure 16 a. Healthy black cherry plant and b. black cherry leaf showing moderate foliar injury on Stone Mountain. Source: Diane M. Styers, 2004.

Table 4. Summary statistics for site and species injury incidence on Stone Mountain.
Source: Diane M. Styers, 2004.

Dominant Species	# sites with damage	% of total sites	total # of species	% of total species
Blackberry	1	50	12	71
Sumac	7	70	66	46
Blackberry/Sumac	2	100	13 b / 7 s	48
Black Cherry	2	67	4	27
Loblolly only	0	0	0	0
ALL	12	40	102	9
ALL except loblolly	12	71	102	47

Table 5. Summary statistics for species damage severity on Stone Mountain.
Source: Diane M. Styers

Dominant Species	# of sites NONE	% of sites NONE	# of sites MINIMAL	% of sites MINIMAL	# of sites MODERATE	% of sites MODERATE	# of sites SEVERE	% of sites SEVERE	Mean damage magnitude
Blackberry	1	50	0	0	0	0	1	50	MINIMAL - MODERATE
Sumac	3	30	4	40	2	20	1	10	MINIMAL
Blackberry/Sumac	0	0	1	50	1	50	0	0	MINIMAL - MODERATE
Black Cherry	1	33	0	0	2	67	0	0	MINIMAL
Loblolly only	13	100	0	0	0	0	0	0	NONE
ALL	18	60	5	17	5	17	2	7	MINIMAL
ALL except loblolly	5	28	5	23	5	34	2	15	MINIMAL

Chapter 4. Conclusions

This study explored ozone pollution damage to vegetation in metropolitan Atlanta. The impacts of ozone exposure to loblolly pine, black cherry, sumac and blackberry on Stone Mountain were examined because they are located in a high exposure area downwind from urban pollution emission sources and the granite outcrop communities located there are part of a unique ecosystem in the Southeast. Wind data were analyzed to determine that pollution transport from Atlanta is possible. Ambient ozone-concentration data were examined to establish that ozone levels are sufficient to damage vegetation in the metropolitan area. Plant-leaf data were assessed to verify ozone injury to area vegetation and to determine the degree of damage suffered during a lower-than-average ozone season in Atlanta. Based on these results, it is likely that ozone damage to vegetation is widespread across the Atlanta area, especially at downwind locations.

Results from the wind and ozone data analyses clearly indicate that ozone levels in metropolitan Atlanta are sufficient to injure sensitive plant species in near-source areas and at downwind locations. Area monitors in proximity to Stone Mountain have recorded ozone concentrations at significant levels during the 2000 through 2003 ozone seasons. Ozone peaks occur consistently in July and lows in September. Calculations of several standard ozone exposure indices have revealed area ozone values above that needed to damage vegetation. Although ozone exposure does not necessarily correspond to uptake amounts by plants, it is still possible to conclude that at these concentrations it is likely that vegetation in these areas of high ozone concentrations is being affected by this oxidant.

Plant-leaf data analyses revealed some degree of injury to sensitive species on Stone Mountain. A visual survey of vegetation on Stone Mountain included the examination of needles of loblolly pines as well as leaves of blackberry, sumac and black cherry species. These additional species were examined to provide supporting evidence that the injury observed is likely due to ozone exposure rather than another variable, as loblolly pines do not always exhibit foliar injury as a response to ozone oxidation. As noted in the literature, each of these four species is sensitive to ozone to some degree. However, due to differential sensitivity in and between species, each plant may respond to ozone in different ways and at varying degrees of severity. This could be a possibility in this case, as the loblolly pines examined on Stone Mountain as part of this study did not exhibit foliar injury while the three broadleaf understory species did. Although chlorotic mottling was absent, some degree of senescence was observed on several of the loblolly pine species but other causes (i.e., drought, mineral leaching, wind damage, etc.) cannot be ruled out without further analyses. Thus, the visible evidence from the understory species indicates that it is possible that loblolly pines on Stone Mountain could be suffering damage from ozone without exhibiting foliar injury.

It is a fortunate finding that loblolly pine species on Stone Mountain appeared to be free from foliar injury. Although it is possible that these species could be suffering other, non-visible forms of oxidant injury, it is likely that the damage is not severe. This could possibly be attributed to adaptation features these Stone Mountain species have acquired through time. These species live and grow in a coarse environment, experiencing greater climatic fluctuations, harsh sun exposure, less nutrient-rich soil and less water uptake ability. Adaptations to such an environment could possibly render these species more tolerant of other environmental stressors,

such as exposure to harmful pollutants. Such a tolerance would explain why foliar injury was absent in Stone Mountain loblolly pines, but present in the more sensitive understory species.

The fact that blackberry, sumac and black cherry species examined displayed varying degrees of ozone-induced foliar injury confirms that ozone concentrations in metropolitan Atlanta are sufficient to damage sensitive vegetation at Stone Mountain and other metropolitan locations. As such, the health of Atlanta's urban forests could be at risk. These data show that ozone damage can occur in areas downwind from Atlanta. Such information is important to Atlanta's citizens and policymakers as damage is occurring outside the area covered by continuous ozone monitors. Vegetation surveys such as the one conducted as part of this study, in combination with passive ozone samplers and other inexpensive data collection methods in metropolitan areas not currently monitored for ozone are imperative to understanding the accurate extent of ozone exposure in Atlanta. It is likely that other metropolitan areas are experiencing similar or worse air quality conditions.

It has been shown that ozone is capable of damaging vegetation to a severe degree. However, not all species are created equal. Loblolly pine on Stone Mountain has proved to be a more tempered species than its coexisting understory neighbors and its lowland cousins. This could possibly be attributed to an adaptation that has occurred allowing these specific trees the ability to tolerate more stressful climatic and environmental conditions. It could also be that injury has occurred to these plants, but that it is not visibly noticeable without magnification. Without further chemical and/or other plant physiological analyses to investigate any diminished plant functions typical of ozone oxidation, its resiliency is unknown. This is a difficult but imperative endeavor, but one that could be approached from many different disciplines.

The results from this field experiment has provided basis for several major conclusions. First, not all plant types respond similarly to ozone exposure, nor do members of the same species. Further, plant physiological data and sensitivity rankings obtained from chamber experiments on seedlings do not always correlate to mature plant responses to ambient ozone concentrations in the field. Long-range transport of ozone and its precursors from urban areas is likely the cause of injury at downwind rural and forested locations. If damage is evident on Stone Mountain, it is likely that ozone is affecting the health of vegetation in many other locations around metropolitan Atlanta. Quick, inexpensive surveys such as this could provide the data necessary to make cognizant land management and development decisions. Such information is vital to policymakers working to minimize Atlanta's pollution problems to preserve the health of it citizens and urban forest ecosystems.

This project aimed to supplement previous research by examining the impact of ozone on vegetation downwind of an urban area in the southeastern United States. The results from this experiment contribute to the knowledge base of ozone effects on mature plant health in its natural environment. These data could be built upon by more longitudinal studies to develop exposure-response relationships for average, below-average and above-average ozone seasons. Further, additional locations around metropolitan Atlanta could be assessed to examine the spatial extent of ozone-induced injury across the region. In order to standardize these analyses, passive ozone monitors should be placed in close proximity to the injury survey areas for a more accurate assessment of "true" ambient ozone concentrations exposed to vegetation in these areas at the time of assessment. This work could have been improved by such additions, but still provides relevant data about a "snapshot" in time for Stone Mountain.

The protection of our natural environment is vital to the health of ourselves and many generations of humans to come. Vegetation responses indicative of pollution injury should be a warning sign that the air we breathe is just not healthy. Loblolly pines on Stone Mountain may be tolerant of ozone, but loblolly pines elsewhere and many other species may not. And just because high ozone levels may not be produced in many rural and forested locations outside the city, does not mean that it cannot be transported long distances to get there. Atlanta's ozone is partially responsible for making the Great Smoky Mountains the most polluted National Park in America. If Atlanta's ozone can impact forests 250 kilometers away, the ones existing within its bounds will not likely fair any better.

References

Allaby, M. 1992. *The Concise Oxford Dictionary of Botany*. Oxford: Oxford University Press.

Aneja, V.P., R. Mathur, S.P. Arya, Y. Li, Murray, G. C. Jr. and T.L. Manuszak. 2000. Coupling the Vertical Distribution of Ozone in the Atmospheric Boundary Layer. *Environmental Science and Technology* 34:2324-2329.

Arbaugh, M.J., P.R. Miller, J.J. Carroll, B. Takemoto and T. Procter. 1998. Relationships of ozone exposure to pine injury in the Sierra Nevada and San Bernardino Mountains of California, USA. *Environmental Pollution* 101:291-301.

Asher, J.E. 1956. Observation and Theory on X-Disease or Needle Dieback. Arrowhead Ranger District, San Bernardino National Forest, U.S. Forest Service, U.S. Department of Agriculture Forest Service Report.

Atkinson, R. 2000. Atmospheric chemistry of VOCs and NOX. *Atmospheric Environment* 34:2063-2101.

Barbo, D.N., A.H. Chappelka, G.L. Somers, M.S. Miller-Goodman and K. Stolte. 1998. Diversity of an early successional plant community as influenced by ozone. *New Phytologist* 137: 653-662.

Barbo, D.N., A.H. Chappelka, G.L. Somers, M.S. Miller-Goodman and K. Stolte. 2002. Ozone impacts on loblolly pine (*Pinus taeda* L.) grown in a competitive environment. *Environmental Pollution* 116:27-36.

Bennett, J.P., R.L. Anderson and M.L. Mielke. 1994. Foliar injury air pollution surveys of eastern white pine (*Pinus strobus* L.): a review. *Environmental Monitoring & Assessment* 30:247-274.

Bennett, J.P. and M. Banerjee. 1995. Air pollution vulnerability of 22 Midwestern parks. *Journal of Environmental Management* 44:339-360.

Boehm, M., B. McCune and T. Vandetta. 1994. Ozone regimes in or near forests of the western United States: Part 2. Factors influencing regional patterns. *Journal of the Air & Waste Management Association* 45: 477-489.

British and Irish Legal Information Institute. 2004. The Air Quality Ozone Wales Regulations 2003 No. 1848 (W.198). http://www.wales-legislation.hmso.gov.uk/legislation/wales/wsi2003/20031848e.htm

California Air Resources Board. 2005. Air Quality Trend Summaries. http://www.arb.ca.gov/adam/cgi-bin/db2www/polltrendsb.d2w/Branch

Chameides, W.L. 1989. The Chemistry of Ozone Deposition to Plant Leaves: Role of Ascorbic Acid. *Environmental Science & Technology* 23:595-600.

Chameides, W.L. 1992. "Tropospheric Ozone: Formation and Fate." In: *Surface Level Ozone Exposures and Their Effects on Vegetation*. A.S. Lefohn (ed). : Chelsea, MI: Lewis Publishers, Inc.

Chappelka, A.H. and L.J. Samuelson. 1998. Ambient ozone effects on forest trees of the eastern United States: A review. *New Phytologist* 139:91-108.

Chappelka, A., E. Hildebrand and J. Skelly. 1999a. Mature black cherry used as a bioindicator of ozone injury. *Water, Air and Soil Pollution* 116:261-266.

Chappelka, A., G. Somers and J. Renfro. 1999b. Visible ozone injury on forest trees in Great Smoky Mountains National Park, USA. *Water, Air and Soil Pollution* 116:255-260.

City-Data.com, http://www.city-data.com/city/Stone-Mountain-Georgia.html.

Comrie, A.C. 1990. The climatology of surface ozone in rural areas: A conceptual model. *Progress in Physical Geography* 14:295-316.

Comrie, A.C. 1994. Tracking ozone: Air-mass trajectories and pollutant source regions influencing ozone in Pennsylvania forests. *Annals of the Association of American Geographers* 84:635-651.

Coulston, J.W., G.C. Smith and W.D. Smith. 2003. Regional assessment of ozone sensitive tree species using bioindicator plants. *Environmental Monitoring and Assessment* 83:113-127.

Davis, J.M. and P. Speckman. 1999. A model for predicting maximum and 8-hr average ozone in Houston. *Atmospheric Environment* 33: 2487-2500.

Diem, J.E. 2002. Remote assessment of forest health in southern Arizona, USA: evidence for ozone-induced foliar injury. *Environ Management* 29(3):373-84.

Diem, J.E. 2004. Explanations for the Spring Peak in Ground-level Ozone in the Southwestern United States. *Physical Geography* 25(2):105-129.

Diem, J.E. and A.C. Comrie. 2001. Air quality, climate and policy: A case study of ozone pollution in Tucson, Arizona. *The Professional Geographer* 53:469-491.

Edwards, G.S., R.E. Sherman and J.M. Kelly. 1995. Red Spruce and Loblolly Pine Nutritional Responses to Acidic Precipitation and Ozone. *Environmental Pollution* 89:9-15.

Elbern, H, J. Kowol, R. Sladkovic, A. Ebel. 1997. Deep stratospheric intrusions: A statistical assessment with model guided analyses. *Atmospheric Environment* 31: 3207-3226.

European Environment Agency. 2004a. Ambient air quality standard for ozone for health protection. http://star.eea.eu.int/showResult.asp?selection=3191&themeID=4&class=inst&instr=Regulation

European Environment Agency. 2004b. Ambient air quality standard for ozone for the protection of vegetative growth. http://star.eea.eu.int/showResult.asp?selection=3193&themeID=4&class=inst&instr=Regulation

European Environment Agency. 2004c. AOT40 definition. http://glossary.eea.eu.int/EEAGlossary/A/AOT40.

European Environment Agency. 2004d. Tropospheric Ozone in EU - The consolidated report. http://reports.eea.eu.int/TOP08-98/en/page008.html

Flagler, R.B. and A.H. Chappelka. 1995. Growth response of southern pines to acidic precipitation and ozone. In: *Impact of Air Pollutants on Southern Pine Forests*. Fox, S. and R.A. Mickler (eds.). NewYork: Springer-Verlag.

Friend, A.L., P.T. Tomlinson, R.E. Dickson, E.G. O'Neill, N.T. Edwards and G.E. Taylor, Jr. 1992. Biochemical composition of loblolly pine reflects pollutant exposure. *Tree Physiology* 11:35-47.

Friend, A.L. and P.T. Tomlinson. 1992. Mild ozone exposure alters ^{14}C dynamics in foliage of *Pinus taeda* L. *Tree Physiology* 11:215-227.

Georgia Department of Natural Resources. 2003. Environmental Protection Division, Air Quality Branch, Annual Air Data, 1996-2004. http://www.air.dnr.state.ga.us/amp.

Georgia Regional Transportation Authority. 2004. http://www.grta.org/default.asp

Geron, C.D., T.E. Pierce and A.B. Guenther. 1995. Reassessment of biogenic volatile organic compound emissions in the Atlanta area. *Atmospheric Environment* 29:1569-1578.

Gregg, J.W., C.G. Jones and T.E. Dawson. 2003. Urbanization effects on tree growth in the vicinity of New York City. *Nature* 424:252-254.

Hewitt, C.N., Lucas, P., Wellburn, A.R. and Fall, R. 1990. Chemistry of ozone damage to plants. *Chemistry & Industry (London)*15:478-481.

Hogsett, W.E., E.H. Lee and J.E. Weber. 1997. An approach for characterizing tropospheric ozone risk to forests. *Environmental Management* 21:105-120.

Horton, S.J., R.A. Reinert and W.W. Heck. 1990. Effects of ozone on three open-pollinated families of *Pinus taeda* L. grown in two substrates. *Environmental Pollution* 65:279-292.

Houle, G. and A. Delwaide. 1991. Population structure and growth-stress relationship of Pinus taeda in rock outcrop habitats. *Journal of Vegetation Science* 2.1:47-58.

Kelly, N.A. and R.F. Gunst. 1990. Response of ozone to changes in hydrocarbon and nitrogen oxide concentrations in outdoor smog chambers filled with Los Angeles air. *Atmospheric Environment Part A* 24:2991-3005.

Kolb, T.E. and R. Matyssek. 2001. Limitations and perspectives about scaling ozone impacts in trees. *Environmental Pollution* 115:373-393.

Krupa, S., M.T. McGrath, C. P. Andersen, F. L. Booker, K. O. Burkey, A. H. Chappelka, B. I. Chevone, E. J. Pell and B. A. Zilinskas. 2000. Ambient Ozone and Plant Health. *Plant Disease* 85: 4-12.

Kuchler, A.W. 1964. *Potential Natural Vegetation of the Conterminous United States.* American Geographical Society, Special Publication No. 36.

Kuehler, E.A. and R.B. Flagler. 1999. The effects of sodium erythorbate and ethylenediurea on photosynthetic function of ozone-exposed loblolly pine seedlings. *Environmental Pollution* 105:25-35.

Laurence, J.A. and C.P. Andersen. 2003. Ozone and natural systems: understanding exposure, response and risk. *Environment International* 29:155-160.

Lefohn, A.S. and V.C. Runeckles. 1987. Establishing standards to protect vegetation - Ozone exposure/dose considerations. *Atmospheric Environment* 21:561-568.

Lefohn, A.S., V.C. Runeckles, S.V. Krupa and D.S. Shadwick. 1989. Important considerations for establishing a secondary ozone standard to protect vegetation. *Journal of the Air Pollution Control Association* 39:1039-1045.

Lefohn, A.S., B.G. Lockaby and R.S. Meldahl. 1992. The characterization and comparison of ozone exposure indices used in assessing the response of loblolly pine to ozone. *Atmospheric Environment, Part A* 26:287-298.

Lefohn, A.S., H.P. Knudsen and W. Jackson. 1997. Effect of surface ozone exposures on vegetation growth in the Southern Appalachian Mountains: identification of possible areas of concern. *Atmospheric Environment* 31:1695-1708.

Manning, W.J., R.B. Flagler and M.A. Frenkel. 2003. Assessing plant response to ambient ozone: growth of ozone-sensitive loblolly pine seedlings treated with ethylenediurea or sodium erythorbate. *Environmental Pollution* 126:73-81.

Massman, W.J., R.C. Musselman and A.S. Lefohn. 1999. A conceptual ozone dose-response model to develop a standard to protect vegetation. *Atmospheric Environment* 34:745-759.

McKendry, I.G. and Lundgren, J. 2000. Tropospheric layering of ozone in regions of urbanized complex and/or coastal terrain: a review. *Progress in Physical Geography* 24(3):329-354.

McLaughlin, S.B. and D.J Downing. 1995. Interactive effect of ambient ozone and climate measured on growth of mature forest trees. *Nature* 374:252-254.

Middleton, J.T., J.B. Kendrick and H.W. Schwalm. 1950. Injury to Herbaceous Plants by Smog or Air Pollution. *Plant Disease Reporter* 34:245-252.

Miller, P.R., J.R. Parmeter, O.C. Taylor and E.A. Cardiff. 1963. Ozone injury to the foliage of *Pinus ponderosa*. *Phytopathology* 53:1072-1076.

Miller, P.R., K.W. Stolte, D.M. Duriscoe and J. Pronos, (technical coordinators). 1995. Evaluating ozone air pollution effects on pines in the western United States. Pacific Southwest Research Station, U.S. Forest Service, U.S. Department of Agriculture; Gen. Tech. Rep. PSW-GTR-155. 78 p.

Mueller, S.F., A. Song, W.B. Norris, S. Gupta and R.T. McNider. 1996. Modeling Pollution Transport of High Ozone in the Southern Appalachian Mountains. *Journal of Applied Meteorology* 35:2105-2120.

Neufeld, H.S., Ben-Hui Yu and E.H. Lee. 1995. Sensitivity of seedlings of black cherry (Prunus serotina Ehrh.) to ozone in Great Smoky Mountains National Park: Exposure-response curves for biomass. *New Phytologist* 130:447-459.

Neufeld, H.S., W.D. Hacker and E.H. Lee. 2000. Seedling insensitivity to ozone for three conifer species native to Great Smoky Mountains National Park. *Environmental Pollution* 108:141-151.

Phillips, D.L., D.T. Tingey and E.H Lee. 1997. Use of auxiliary data for spatial interpolation of ozone exposure in southeastern forests. *Environmetrics* 8:43-61.

Pompe, M. and M. Veber. 2001. Prediction of rate constants for the reaction of O_3 with different organic compounds. *Atmospheric Environment* 35:3781-3788.

Powell, D.S., J.L. Faulkner, D.R. Darr, Z. Zhu and D.W. MacCleery. 1992. Forest Resources of the United States, 1992. USDA Forest Service, Rocky Mountain Forest and Range Experiment Station, General Technical Report RM-234.

Pye, J.M. 1988. Impact of Ozone on the Growth and Yield of Trees: A Review. *Journal of Environmental Quality* 17:347-360.

Qui, Z., Chappelka, A.H., Somers, G.L., Lockaby, B.G. and Meldahl, R.S. 1992. Effects of ozone and simulated acidic precipitation on above- and below-ground growth of loblolly pine (*Pinus taeda*). *Canadian Journal of Forest Research* 22:582-587.

Redington, R.J. 1978. Survey of the Appalachians. South Egremont:Taconic Publishers.

Reinert, R.A., S.J. Horton and S.R. Shafer. 1996. Responses of loblolly pine to ozone and simulated acidic rain. *Canadian Journal of Forest Research* 26:1715-1723.

Richardson, C.J., T.W. Sasek and E.A. Fendick. 1992. Ozone exposure-response relationships for photosynthesis in genetic strains of loblolly pine seedlings. *Forest Ecology & Management* 51:163-178.

Roelofs, G.J., A.S. Kentarchos, T. Trickl, A. Stohl, W.J. Collins, R.A. Crowther, D. Hauglustaine, A. Klonecki, K.S. Law, M.G. Lawrence, R. Von Kuhlmann, M. Van Weele. 2003. Intercomparison of tropospheric ozone models: Ozone transport in a complex tropopause folding event. *Journal of Geophysical Research, D. Atmospheres* 108:[np].

Seinfeld, H. 1989. Urban air pollution: State of the science. *Science* 243:745-52.

Shafer, S.R. and A.S. Heagle. 1989. Growth responses of field-grown loblolly pine to chronic doses of ozone during multiple growing seasons. *Canadian Journal of Forest Research.* 19:821-831.

Sillman, S. 1999. The relation between ozone, NOX and hydrocarbons in urban and polluted rural environments. *Atmospheric Environment* 33:1821-1845.

Skelley, J.M., D.D. Davis, W. Merrill, E.A. Cameron, H.D. Brown, D.B. Drummond and L.S. Dochinger (eds.). 1985. Diagnosing Injury to Eastern Forest Trees. Agriculture Information Service, College of Agriculture, Penn State University.

Somers, G.L., J.R. Renfro and A.H. Chappelka. 1998. Empirical evidence of growth decline related to visible ozone injury. *Forest Ecology and Management* 104:129-137.

Stone Mountain Park, http://www.stonemountainpark.com/newsite/parkinfo_faq.asp

Tanaka, P.L., D.T. Allen, S. Oldfield, J.D. Neece and C.B. Mullins. 2000. Anthropogenic sources of chlorine and ozone formation in urban atmospheres. *Environmental Science and Technology* 34: 4470-4473.

Taylor, G.E. Jr. 1994. Role of genotype in the response of loblolly pine to tropospheric ozone: Effects at the whole-tree, stand and regional level. *Journal of Environmental Quality* 23:63-82.

United States Bureau of the Census. 2001. Population Change and Distribution: 1990 to 2000.

United States Department of Agriculture, Forest Service. 2004a. Forest Inventory and Analysis Program. http://srsfia2.fs.fed.us/.

United States Department of Agriculture, Forest Service. 2004b. Ozone Biomonitoring Program. Exposure definitions. http://www.fiaozone.net/exposuredescrip.htm

United States Environmental Protection Agency. 1990. Ozone and Carbon Monoxide Design Value Calculations. EPA, Office of Air Quality Planning and Standards, Research Triangle Park, North Carolina; EPA Memorandum. http://www.epa.gov/ttn/naaqs/ozone/ozonetech/laxton.htm

United States Environmental Protection Agency. 1996. Air quality criteria for ozone and related photochemical oxidants. EPA, Office of Air Quality Planning and Standards, Research Triangle Park, North Carolina; EPA Report No. EPA/600/P-93/004bF.

United States Environmental Protection Agency. 1999. Smog—Who Does It Hurt? What You Need to Know About Ozone and Your Health. (AirNow).

United States Environmental Protection Agency. 2004. AirData. http://www.epa.gov/air/data/

United States Environmental Protection Agency. 2004. Photochemical Assessment Monitoring Stations (PAMS). http://www.epa.gov/oar/oaqps/pams/arealist.html

United States Environmental Protection Agency. 2004. Regions. http://www.epa.gov/epahome/locate2.htm

United States Fish and Wildlife Service. 2004. http://www.fws.gov/

United States National Oceanic and Atmospheric Administration, National Climatic Data Center. 2004. Comprehensive Aerological Reference Data Set, Global Radiosonde Data. http://www.ncdc.noaa.gov/oa/climate/cards/cards_homepage.html

United States National Park Service, Great Smoky Mountains National Park. 2004. http://www.nps.gov/grsm/pphtml/nature.html.

United States National Research Council. 1992. Rethinking the Ozone Problem in Urban and Regional Air Pollution. Committee on Tropospheric Ozone Formation and Measurement. Washington DC: National Academy Press.

Viezee, W., W.B. Johnson and H.B. Singh. 1983. Stratospheric ozone in the lower troposphere II – assessment of downward flux and ground-level impact. *Atmospheric Environment* 17:1979-1993.

Vollenweider, P., M. Ottiger and M.S. Günthardt-Goerg. 2003. Validation of leaf ozone symptoms in natural vegetation using microscopical methods. *Environmental Pollution* 124:101-118.

Wiselogel, A.E., J.K. Bailey J.K. and R.J. Newton. 1991. Growth response of loblolly pine (*Pinus taeda* L.) seedlings to ozone fumigation. *Environmental Pollution* 71:43-56.

Zaveri, R.A., R.D. Saylor, L.K. Peters, R. McNider and A. Song. 1995. A model investigation of summertime diurnal ozone behavior in rural mountainous locations. *Atmospheric Environment* 29: 1043-1065.

Zhang, J. and S.T. Rao. 1999. The Role of Vertical Mixing in the Temporal Evolution of Ground-Level Ozone Concentrations. *Journal of Applied Meteorology* 38: 1674-1691.