Potential Impact of Ozone on Coniferous Forests of the Interior Southwestern United States

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Despite the well-documented negative impacts of ozone on the health of coniferous forests in southern California and the significant growth experienced by southwestern cities over the past several decades, the ozone/forest dynamic in the interior portion of the southwestern United States has been largely ignored. Primarily through a review of literature pertaining to most aspects of ozone and its impact on forest health, this article provides insights on the ozone/forest dynamic within coniferous forests of the interior Southwest. It is suggested that ozone absorption in southwestern coniferous forests may equal that in southern California, owing to the long-distance transport of atmospheric pollutants into the interior Southwest and the presence of the North American monsoon. Nevertheless, research gaps identified in this article suggest a need for future research on ozone exposure levels and the ozone sensitivities of conifer species and varieties in southwestern coniferous forests. *Key Words: climate, coniferous forests, ozone, southwestern United States.*

Humans have been altering coniferous forests extensively in the western United States in the last century through timber-harvesting, livestock-grazing, and fire suppression. A more recent agent of change is atmospheric pollution. Cities are major sources of atmospheric pollutants, particularly chemicals that lead to the production of ozone. Ozone is the only regionally dispersed atmospheric pollutant for which rigorous scientific evidence exists of its adverse impact on forests (Woodman 1987; Cowling 1989). Ozone exposure is known to disrupt forest ecosystems, mainly through increased tree mortality and associated changes in forest structure.

A prime example of the severe damage ozone can cause to forests can be found in the Los Angeles area, where the harmful effect of ozone on conifers has been studied since at least the early 1960s (e.g., Miller et al. 1963). Heavy emissions of pollutants from anthropogenic sources (U.S. Environmental Protection Agency 2000) combined with atmospheric conditions conducive to elevated ozone concentrations (Holzworth 1962; Lu and Turco 1996) have resulted in the death of most ozone-sensitive trees in the nearby San Bernardino Mountains since the 1950s (Takemoto, Bytnerowicz, and Fenn 2001).

Substantial ozone-induced forest damage in the San Bernardino Mountains is one of relatively few cases of well-documented declines in forest health resulting from atmospheric pollution (Innes 1992).[1] Lack of documentation does not mean that atmospheric pollution levels are typically not high enough to damage trees, or that most trees are not sensitive to ozone. Rather, it illustrates how the complex interrelationships between atmospheric pollution and forest health hinder the identification of exact causes of forest damage. The inextricable linkage between ozone and forest damage in the San Bernardino Mountains is a legitimate reason alone for considering ozone-induced declines in forest health in nearby regions, such as the interior portion of the southwestern United States—the "interior Southwest."

For the purposes of this article, the "interior Southwest" is defined as a region containing all of Arizona and large portions of Nevada and New Mexico (Figure 1). This region has undergone tremendous population growth over the past 50 years (Goudy 2001). As a result of this growth, several large urban areas are located next to mountains topped with coniferous forests. Hence, it is plausible that the environmental problem in the San Bernardino Mountains also exists in the interior Southwest.

Little is known about ozone-induced damage to coniferous forests in the interior Southwest, despite the region's proximity to the Los Angeles area and the obvious juxtaposition of urban areas and forests in the region (Figure 1). This critical research gap results in part from the absence of a region-specific conceptual framework for the topic. Therefore, the principal aim of this article is to propose the conceptual underpinnings of the ozone and forest-health dynamic in the interior Southwest. Through a review of relevant literature and an explanation of the resulting conceptual model, this article addresses the following questions:

- What is the potential for significant ozone-induced damage in the coniferous forests of the interior Southwest?
- How does the potential vary across the region?

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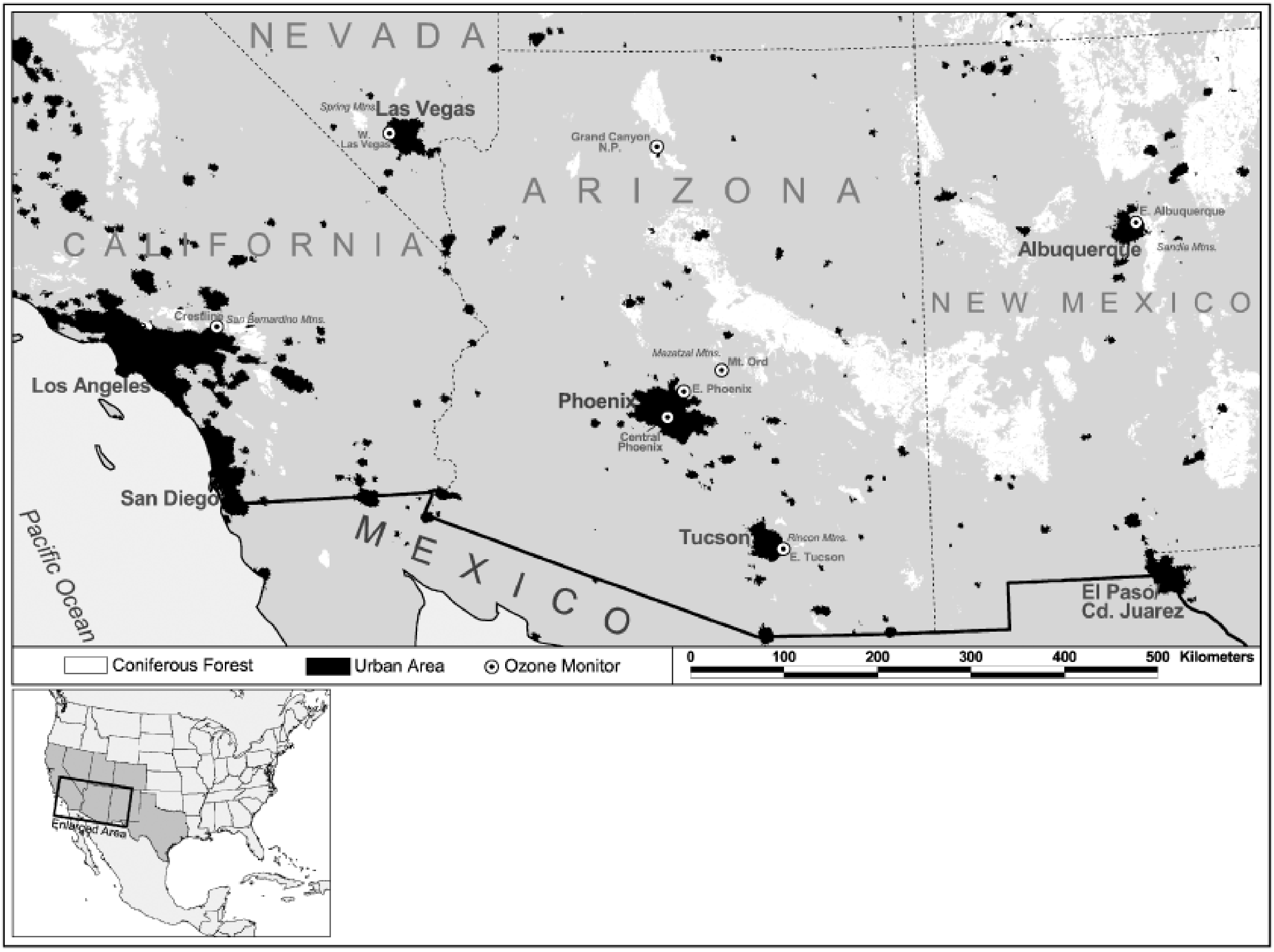

Figure 1. Map of the southwestern United States showing urban areas, mountain-situated coniferous forests, and select ozone monitors. Urban areas were determined using satellite-measured nighttime light-intensity data available from the National Geophysical Data Center, while coniferous forests were located using land-cover data provided by the United States Geological Survey's National Gap Analysis Program.

The work presented in this article is intended to: (1) highlight the major ozone/forest concepts, (2) integrate the ozone/forest concepts with Southwest-specific information, (3) speculate on the magnitudes and controlling processes of phenomena related ultimately to ozone-induced forest damage in the interior Southwest, and (4) distill the conceptual and speculative information into several key future research questions. Table 1 presents descriptions of relevant data (i.e., climate, land cover, population, pollutant emissions, motor-vehicle usage, and ozone) used to fulfill the objectives.

One purpose of this article is to encourage researchers to explore the ozone/forest issue in the interior Southwest. Pollution-induced damage to forests can reduce genetic resources and alter ecological structure and function (Tingey, Hogsett, and Henderson 1990). A thorough understanding of anthropogenic stressors, including ozone, is needed to retard the future degradation of forest health. This synthesis of information could also influence policy decisions related to the monitoring of ozone and the surveying of forests to assess the magnitude of ozone-induced damage. The verification of high ozone levels and ozone-induced damage to southwestern coniferous forests—especially those forests in national wilderness areas and national parks—may lead to decisions that could ultimately diminish pollutant-emitting activities in various locales.[2]

Coniferous Forests and Urban Areas in the Interior Southwest

Most of the interior Southwest is situated in the Basin and Range physiographic province, which is characterized

Table 1. Descriptions and Sources of Relevant Data

Data Type	Year(s)	Source
Monthly precipitation totals and average temperature values for Palisade Ranger Station, Arizona	1965–1981	Western Regional Climate Center
Spatially resolved land-cover data for Arizona, California, Colorado, Nevada, New Mexico, and Utah	Late 1990s	United States Geological Survey's National Gap Analysis Program
Spatially resolved nighttime light-intensity data	1997	National Geophysical Data Center
County-specific population data	1990–2000	United States Census Bureau
MSA-specific pollutant emissions and vehicle-miles-traveled estimates	1995	United States Environmental Protection Agency's National Emission Trends Database
Tucson-specific pollutant emissions and vehicle-miles-traveled estimates	1998	Diem and Comrie (2002)
Hourly ozone data for a monitor at Crestline[1] in the San Bernardino Mountains	1996–1999	California Air Resources Board
Hourly ozone data for monitors at central Phoenix,[2] eastern Phoenix,[3] and Mt. Ord[4]	1997–2000	Arizona Departmental of Environmental Quality
Hourly ozone data for a monitor at Saguaro National Park (East Unit)[5] near Tucson, Arizona	1997–2000	Pima County Department of Environmental Quality
Hourly ozone data for monitors at western Las Vegas,[6] Grand Canyon National Park (AZ),[7] and eastern Albuquerque[8]	1997–2000	United States Environmental Protection Agency

Notes: The monitors' notations refer to U.S. EPA site identification numbers, which are as follows: [1]060710005442011, [2]040133002442011, [3]040132005442011, [4]040139701442011, [5]040190021442011, [6]320030073442011, [7]040058001442011, and [8]350011012442011.

by mountain ranges typically rising more than 2,100 m above alluvial basins. While semiarid and arid climates prevail in the basins, the climate at higher elevations is considerably cooler and wetter. This climate supports coniferous forests dominated by ponderosa pine (*Pinus ponderosa*), often growing in pure stands (Brown 1998). Interior ponderosa pine (*Pinus ponderosa* var. *scopulorum*), which is the most common conifer in the interior Southwest (Zwolinski 1996), and Arizona ponderosa pine (*Pinus ponderosa* var. *arizonica*) are the two recognized varieties in the region. Ponderosa pine hybrids may also exist (Epperson et al. 2001). Other common conifers in the mountains near southwestern cities include bristlecone pine (*Pinus aristata*), Douglas fir (*Pseudotsuga menziesii*), limber pine (*Pinus flexilis*), and white fir (*Abies concolor*). Not all conifer species exist on every mountain range. A typical southwestern coniferous forest has a warm, temperate climate, with daily maximum temperatures reaching 25°C in the summer, daily minimum temperatures dropping to -4°C in winter, and annual precipitation totals exceeding 750 mm.[3] The forests receive moisture in winter as midlatitude storms move through the region (Comrie 1996); in the summer, the North American monsoon brings heavy precipitation primarily to the southern and eastern portions of the region (Adams and Comrie 1997; Comrie and Glenn 1998).

Before sketching the conceptual model of ozone and forest health, it is important to note human activities in the recent past that have shaped the coniferous forests. Those activities established the forest conditions on which atmospheric pollution acts. In the late nineteenth century, Anglo settlers modified the forests through timber-harvesting (deBuys 1985) and livestock-grazing (Savage and Swetnam 1990). Intentional fire exclusion was entrenched by the early twentieth century (Swetnam and Baisan 1996), which enabled the accumulation of fuels. The region's coniferous forests shifted away from low-intensity, surface fire regimes toward high-intensity, stand-replacing fire regimes (Weaver 1951; Cooper 1960; Swetnam and Baisan 1996). Anthropogenic impacts on southwestern coniferous forests have resulted in a large potential for increased tree density (Habeck 1990) and crown fires (Covington and Moore 1994; Swetnam and Baisan 1996).

As forest management practices continued to alter the character of montane forests throughout the second half of the twentieth century, the populations of the interior Southwest's urban areas began growing rapidly. The completion of federally funded water projects (Reisner 1993), the availability of air conditioners, and an arrival of relocated companies (Prytherch 1999) facilitated the influx of people. Additionally, climate and natural amenities (e.g., scenery) became prime factors attracting residents to the interior Southwest (Logan 1995; Prytherch 1999).

Because almost all growth occurred after the automobile became the dominant mode of transportation, the cities grew outward. Presently, the largest metropolitan statistical areas (MSAs) in the interior Southwest are Phoenix, Las Vegas, Tucson, and Albuquerque, with respective populations of ~3,252,000, ~1,563,000, ~844,000, and ~713,000. These MSAs are still experiencing large population growth rates; their median population growth rate from 1990 to 2000 was approximately 35 percent, compared to a 13-percent median growth rate for all MSAs with similar population totals.[4] The population growth rate is associated with a much larger relative increase in geographic area of the MSAs. For example, Las Vegas's urbanized area nearly quadrupled between 1980 and 2000 (Schrank and Lomax 2001). As an example of the influence of population density on motor-vehicle usage, the average daily vehicle miles traveled per capita is approximately 20 percent higher in southwestern MSAs than in northeastern MSAs.[5] Motor vehicles are a major source of atmospheric pollutants; therefore, as with the situation in the Los Angeles area, increased motor vehicle usage in and near southwestern cities has led to decreased air quality. It is crucial to emphasize that urbanization is a relatively recent phenomenon in the interior Southwest, and pollutant emissions resulting principally from the excessive, growth-induced motor-vehicle usage may be the major anthropogenic stressor for local coniferous forests.

Conceptual Model of Ozone and Forest Health in the Interior Southwest

The impact of ozone on coniferous forests involves many processes, some of which are exceedingly complex. This section explores only the major processes. Climate plays a large part in the ozone/forest dynamic, because it directly and indirectly affects the following core components of the ozone/forest-health conceptual model: (1) pollutant emissions and ozone production, (2) pollutant transport, (3) ozone removal prior to absorption, (4) ozone exposure, (5) ozone absorption, (6) ozone-induced damage, and (7) indirect effects of ozone-induced damage (Figure 2). These components are examined below.

Pollutant Emissions and Ozone Production

Surface ozone is produced typically by the oxidation of volatile organic compounds (VOCs) in the presence of nitrogen oxides (NO_x) and sunlight (Chameides et al. 1992). In most southwestern MSAs, vegetation (i.e., biogenic sources) is presumed to be the largest VOC source (Diem and Comrie 2000), and on-road motor vehicles are the largest NO_x source (U.S. Environmental Protection Agency 2000; Diem and Comrie 2002). Ozone production at the surface would be minimal without the anthropogenic contribution of NO_x.

Climate controls pollutant emissions and ozone production mainly through temperature and solar-radiation effects. Ultraviolet radiation, which is mainly solar in origin, is the ultimate limiting factor, because ozone cannot be produced in its absence. Temperature and solar radiation are generally correlated positively with biogenic VOC emissions (Guenther et al. 1993). Concerning anthropogenic emissions, temperature is also correlated positively with VOC emissions from evaporated gasoline and solvents and NO_x emissions from power plants (Diem and Comrie 2001). In most cases, hot and sunny conditions are associated with increased ozone production (Sillman 1999; Diem and Comrie 2001).

Although the hot and sunny climate of the interior Southwest is conducive to ozone production, the highest ozone concentrations in the nation are not found in the interior Southwest (U.S. Environmental Protection Agency 2002) for the following reasons: (1) arid and semiarid areas, which have little to moderate vegetative cover, have relatively low contributions of VOCs from biogenic sources; (2) southwestern MSAs, which have a relative lack of industrial emissions (U.S. Environmental Protection Agency 2000), are not among the largest emission source areas in the nation; and (3) the MSAs are geographically isolated (e.g., the average distance between the MSAs is ~500 km), meaning that, unlike in the eastern United States, urban pollution plumes at the surface may not coalesce and, in turn, magnify pollution levels. Despite the above characteristics that limit a regional ozone problem, the potential does exist for elevated ozone levels in southwestern coniferous forests.

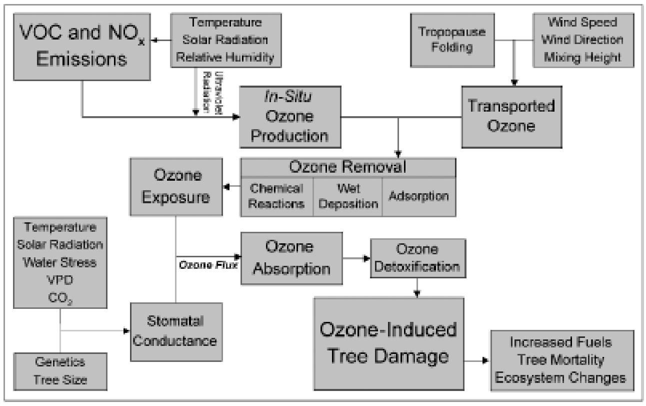

Figure 2. Schematic diagram of the major ozone/forest concepts. Note the complex linkages between pollutant emissions and ozone-induced tree damage.

This potential is buttressed by changes in ozone-production chemistries, the transport of pollutants over large distances, and intrusions of stratospheric ozone.

Pollutant Transport

Horizontal transport of atmospheric pollutants can occur over a range of geographic scales, and it can occur within both the planetary boundary layer (i.e., the lowest portion of the troposphere that is influenced by surface friction) and the free troposphere (i.e., the portion of troposphere above the planetary boundary layer). On a synoptic scale, atmospheric pollutants in the planetary boundary layer not only can accumulate slowly within high-pressure cells and be transported hundreds of kilometers (Vukovich et al. 1977; Comrie 1990, 1994) but also can be vented into the free troposphere, where they can be transported long distances rapidly by synoptic-scale systems (Pryor and Hoffer 1992; McKendry and Lundgren 2000). Concerning synoptic-scale transport in the western United States, the Los Angeles area has been implicated as a likely source of atmospheric pollution in southern Nevada/northwestern Arizona (Pryor and Hoffer 1992; Davis and Gay 1993a; Pryor et al. 1995), Phoenix (Gaffney et al. 2002), and Tucson (Diem and Comrie 2001). An examination of the synoptic-scale circulation patterns and their monthly frequencies, presented in Davis and Walker (1992), indicates that April, May, and June have the highest potential for pollutant transport from southern California and northwestern Mexico into the interior Southwest. Throughout most of the ozone season (i.e., April through September), the region has a high potential for pollutant transport within the free troposphere (Diem and Comrie 2001). Long-distance pollutant transport is a critical issue in the interior Southwest, because it could bring high atmospheric pollution levels to areas distant from large pollutant-source areas.

Complex topography makes mesoscale wind systems extremely important in the interior Southwest with respect to pollutant transport. Pollutants can be transported consistently from basin-located cities to mountain-top coniferous forests via upslope winds (King, Shair, and Reible 1987; Böhm, McCane, and Vendetta 1995; Lu and Turco 1996; Ellis, Hildebrandt, and Fernando 1999; Ellis et al. 2000; Diem and Comrie 2001). Although strong, low-elevation temperature inversion can restrict the penetration of upslope flows and the subsequent transport of pollutants to forests (King, Shair, and Reible 1987), these inversions are rare in the interior Southwest during the ozone season, because maximum mixing heights exceed 4,000 m during the ozone season (Holzworth 1962).

There also can be significant vertical transfers of ozone from the stratosphere to the lower troposphere. These stratospheric intrusions are the predominant source of tropospheric ozone at remote, midlatitude sites (Singh et al. 1980), and they are "natural" processes not directly influenced by anthropogenic activities. The major mechanism behind stratospheric intrusions is tropopause folding (Viezee and Singh 1980), which is a deformation of the boundary between the troposphere and the stratosphere. Midlatitude synoptic-scale systems and horizontal tongues of tropopause air are responsible for much of the folding (Gettelman and Sobel 2000). Stratospheric intrusions peak in May and June in the middle and high latitudes of the northern hemisphere (Appenzeller, Holton, and Rosenlof 1996; Gettelman and Sobel 2000). For example, in southeastern Wyoming, folding events peak in May, and the associated elevated ozone concentrations add approximately 10 percent to the normal background concentrations (Wooldridge, Zeller, and Musselman 1997).

A peak in folding events in May should also occur in the interior Southwest. The Great Basin trough, a synoptic-scale circulation pattern presumably conducive to stratospheric intrusions, occurs on approximately 23 percent of the days in May, the highest proportion of any month (Davis and Walker 1992; Davis and Gay 1993b). Overall, nearly 90 percent of all troughing days throughout the year occur in May, June, September, and October, and nearly one-third of all troughing days occur in May (Davis and Gay 1993b). A persistent upper-level anticyclone prevents folding events from occurring during the monsoon period (i.e., July and August). Stratospheric intrusions may supplement ozone levels in most of the region's coniferous forests during the pre-monsoon months. The degree of enhancement is unknown at the present time.

Substantial exchanges of ozone can also occur from the planetary boundary layer to the free troposphere. In the Southwest, ozone can be transported from the planetary boundary layer to the free troposphere by clouds and mountains, thereby diminishing the ozone reservoir at the surface. Cloud venting involves vertically developed clouds (Ching, Shipley, and Browell 1988); mountain venting involves strong, mountain-induced updrafts (McKendry and Lundgren 2000). Of course, clouds that develop over mountainous terrain can also vent ozone (Kossmann et al. 1999; McKendry and Lundgren 2000). Ozone that remains in the planetary boundary layer is not necessarily subject to absorption by conifers, because ozone can be removed easily by several different mechanisms.

Ozone Removal Prior to Absorption

Ozone can be removed from the atmosphere through chemical reactions, wet deposition, and adsorption (Rasmussen, Taheri, and Kabel 1975), thereby reducing the supply of ozone available for absorption by conifers. Ozone is destroyed following reactions with NO_x and VOCs. Since the forests are tens of kilometers from major NO_x sources, a presumed absence of high NO_x concentrations in southwestern forests indicates that ozone removal by chemical reactions with those species is not especially prevalent. Chemical reactions within forests may still be important, because Mikkelson and colleagues (2000) suggest that the reaction of ozone with biogenic VOCs contributes significantly to ozone removal. Wet deposition involves precipitation-scavenging such as "rainout" (transfer of ozone to cloud droplets) and "washout" (transfer of ozone to falling precipitation) (Nowak 1994). Adsorption occurs when ozone is deposited onto external surfaces of objects (Matyssek et al. 1995). Considering the prevalence of nearly semiarid climates throughout southwestern coniferous forests, adsorption is probably more important than is wet deposition in the region. In fact, precipitation events are extremely infrequent in May and June (Sellers and Hill 1974). Ozone molecules not removed through the above processes constitute the available ozone to which conifers are exposed.

Ozone Exposure

The eventual ozone exposure levels in southwestern coniferous forests are a complex product of synoptic-scale transport, mesoscale transport, *in situ* ozone production, exchanges between the planetary boundary layer and free troposphere, and ozone removal processes. By no means is ozone in those forests simply a function of *in situ* ozone production (Kelly, Wolff, and Ferman 1984). In fact, because of the relative abundance of biogenic VOCs and scarcity of large NO_x sources, *in situ* ozone production in western forests is dependent on ambient NO_x concentrations (Fehsenfeld et al. 1983), which are usually low. Consequently, ozone exposure levels in southwestern coniferous forests should primarily reflect the horizontal and vertical transport of ozone.

Substantial spatial variations in ozone concentrations can exist throughout montane forests (Miller, Taylor, and Poe 1986). Topographic effects and proximity to urban areas are two major controls of ozone exposure levels within a mountain region (Van Ooy and Carroll 1995). For example, trees on the western end of the San Bernardino Mountains are exposed to considerably higher ozone exposure levels than are those on the eastern end of the range (Takemoto, Bytnerowicz, and Fenn 2001). In the interior Southwest, mountain slopes exposed to urban pollution plumes, in the path of pollution-rich, upper-level air, or both have potentially the highest ozone exposure levels in a region.

Based on the prior information presented in this section of the article, it is likely that relatively high ozone exposure levels exist in some forests in the western United States. As mentioned previously in less detail, the San Bernardino Mountains have exceptionally high ozone exposure levels, owing to heavy pollutant emissions in upwind areas (i.e., Los Angeles), ample sunlight, onshore breezes, and a shallow planetary boundary layer (Holzworth 1962; Lu and Turco 1996). The remainder of this article treats ozone exposure at Crestline, which is situated at the western end of the San Bernardino Mountains and thus has higher ozone exposure levels than do many other sites in the mountain range, as a benchmark in comparisons with ozone exposure throughout the interior Southwest.

It is unfortunate that no truly "clean" ozone-monitoring site currently exists at high elevations in the interior Southwest. Data from such a site would provide an indication of minimal ozone exposure levels in the region. Ozone levels at "clean" sites may be used as proxies for natural background ozone levels (Lefohn, Krupa, and Winstanley 1990). Of course, if long-distance pollutant transport is a common occurrence throughout the region, then few "clean" sites should exist. Lefohn and colleagues (1990) consider the Apache National Forest in eastern Arizona to be a "clean" site, but in a direct comparison with other presumably "clean" sites across the globe, it consistently had higher ozone exposure levels than did most of the other sites. Results presented by Lefohn and colleagues (1990) also indicate that May is typically the month with the highest ozone exposure at Apache National Forest. A remote ozone monitor does exist at 2,150 meters above sea level (m a.s.l.) at Grand Canyon National Park, and monthly ozone exposure levels at that site also reach a maximum in May and decline thereafter (Figure 3).[6] In addition to the transport of pollutants from southern California, stratospheric intrusions may cause the peak in ozone exposure levels. Even though the monitor at Grand Canyon National Park is not located at a "clean" site, its data probably represents minimal ozone levels at coniferous forests in the western portion of the interior Southwest fairly well.

Southwestern coniferous forests near major anthropogenic pollution source areas (i.e., large urban areas) have dramatically higher ozone exposure levels than do more remote forests, such as those at Grand Canyon National Park. The ozone monitor in the interior Southwest that lies both within a coniferous forest and near a major urban area is located at Mount Ord, which is 2,180 m a.s.l. and 80 km northeast of Phoenix in the Mazatzal Mountains (Figure 1). In the Phoenix area, ozone and its precursors are transported from the urban areas to nearby upland areas, such as the Mazatzal Mountains, via upslope winds (Ellis, Hildebrandt, and Fernando 1999; Ellis et al. 2000). Even though anthropogenic emissions of ozone precursor chemicals from the Los Angeles area are over three times larger than emissions from the Phoenix area and the planetary boundary layer over southern California is less than half as thick as that over the interior Southwest (Holzworth 1962), daily ozone exposure levels from April through September at Crestline are only 20 percent higher than those at Mt. Ord.[7] Early-season ozone at Mt. Ord is actually equivalent to that at Crestline (Figure 3). Mt. Ord's relatively high early-season exposure levels most likely result from Phoenix-derived pollution, superimposed on pollutant transport from southern California and stratospheric intrusions.

Grand Canyon National Park and Mt. Ord monitors are in considerably different ozone-exposure environments (i.e., remote versus urban influence at the mesoscale), yet

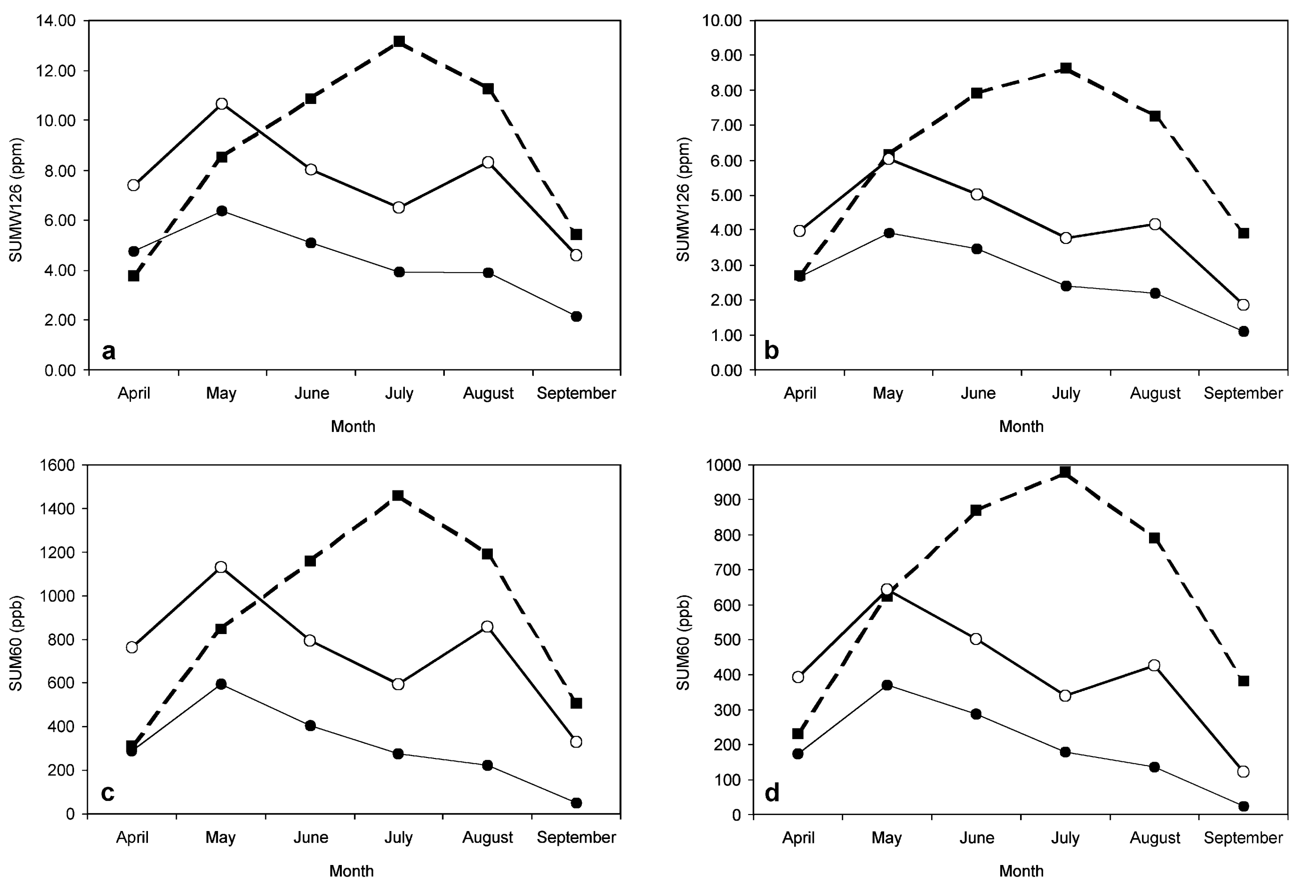

Figure 3. Intraseasonal variations in (a) daily SUMW126, (b) daytime SUMW126, (c) daily SUM60, and (d) daytime SUM60 at Crestline (--■--), Grand Canyon National Park (—●—), and Mt. Ord (—○—). SUMW126 and SUM60 are ozone exposure indices; see endnote 6 for an explanation of them. Scales of the y-axis vary to emphasize differences in ozone exposure levels among the stations.

both have similar intraseasonal variations in ozone exposure. This common intraseasonal behavior should represent the behavior at coniferous forests proximate to the three other cities reasonably well. Urban-fringe monitors downwind of Albuquerque, Las Vegas, Phoenix, and Tucson do not experience as pronounced an early-season exposure maximum compared to the mountain monitors (Figure 4). Thus, the intrusion of stratospheric ozone is probably not as prevalent at low-elevation sites. Increased emissions of ozone precursors (Diem and Comrie 2001, 2002) and a shallower planetary boundary layer lead to a secondary peak in ozone exposure levels in late summer in the four southwestern MSAs. Similar late-season peaks may also occur in forests near the three other large MSAs. A late-season peak in ozone exposure would have major implications concerning ozone's overall impact on forest health.

Hypothetically, coniferous forests near Albuquerque, Las Vegas, and Tucson should have relatively high ozone exposure levels.[8] In fact, exposure in southern Nevada's Spring Mountains may equal that in the western San Bernardino Mountains (Figure 5). The Spring Mountains are essentially downwind (at the synoptic scale) of the Los Angeles area during the first three months of the ozone season, and nearby Las Vegas is probably the dominant late-season pollutant source. In addition, stratospheric intrusions are hypothesized as major suppliers of ozone in the early season, not only in the Spring Mountains, but also at most high-elevation sites in the interior Southwest. Ozone exposure levels in the Rincon and Sandia Mountains are estimated to be almost equal to levels in the Mazatzal Mountains, but much lower than levels in the Spring Mountains. Ozone exposure in the above mountains represents the potential dose of ozone that could impact a conifer if absorption were always maximized. However, processes controlling ozone absorption are highly variable.

Ozone Absorption

Coniferous forests with high ozone-exposure levels do not always have high fluxes of ozone into needles. This is

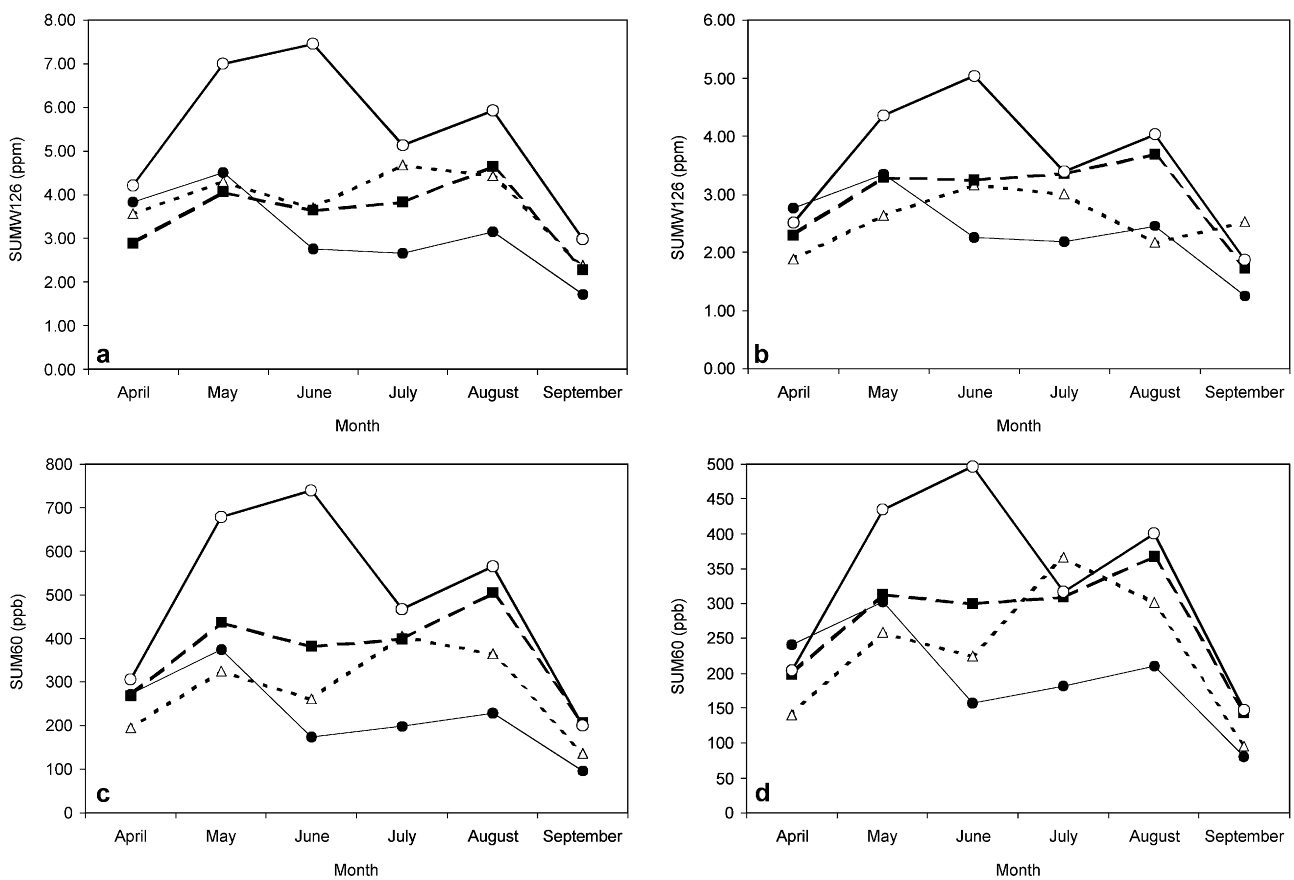

Figure 4. Intraseasonal variations in (a) daily SUMW126, (b) daytime SUMW126, (c) daily SUM60, and (d) daytime SUM60 at western Las Vegas (—○—), eastern Phoenix (--■--), eastern Albuquerque (- -△- -), and eastern Tucson (—●—). Scales of the y-axis vary to emphasize differences in ozone exposure levels among the stations.

important, because only the ozone flux (i.e., absorption of ozone) actually damages the plant (Bauer et al. 2000). The rate of ozone flux is controlled by the concentration of ozone at the needle surface and by stomatal conductance (i.e., conductance of gases into the needle tissue through stomata). Genetics plays a principal role in stomatal conductance, with conductance varying widely among different varieties of the same species (Coyne and Bingham 1982). For example, the stomatal conductance of Pacific ponderosa pine (*Pinus ponderosa* var. *ponderosa*) is approximately two times greater than that of interior ponderosa pine (Monson and Grant 1989). Stomatal conductance is also a function primarily of the following environmental variables: temperature, solar radiation, water stress, needle-to-air vapor pressure deficit (VPD), and exposure to elevated carbon dioxide (CO_2) levels. Studies have shown significant positive correlations between stomatal conductance and temperature (Musselmann and Minnick 2000) and solar radiation (Sharkey and Ogawa 1987). Other studies have shown significant negative correlations between stomatal conductance and needle-to-air VPD (Kolb and Stone 2000), water stress (Tingey and Hogsett 1985; Temple et al. 1993; Panek and Goldstein 2001), and elevated CO_2 concentrations (Pallas 1965; Melillo et al. 1990). Stomatal conductance is a complex process that is affected by many environmental factors, and the overall impact of those factors on stomatal conductance is not fully understood.

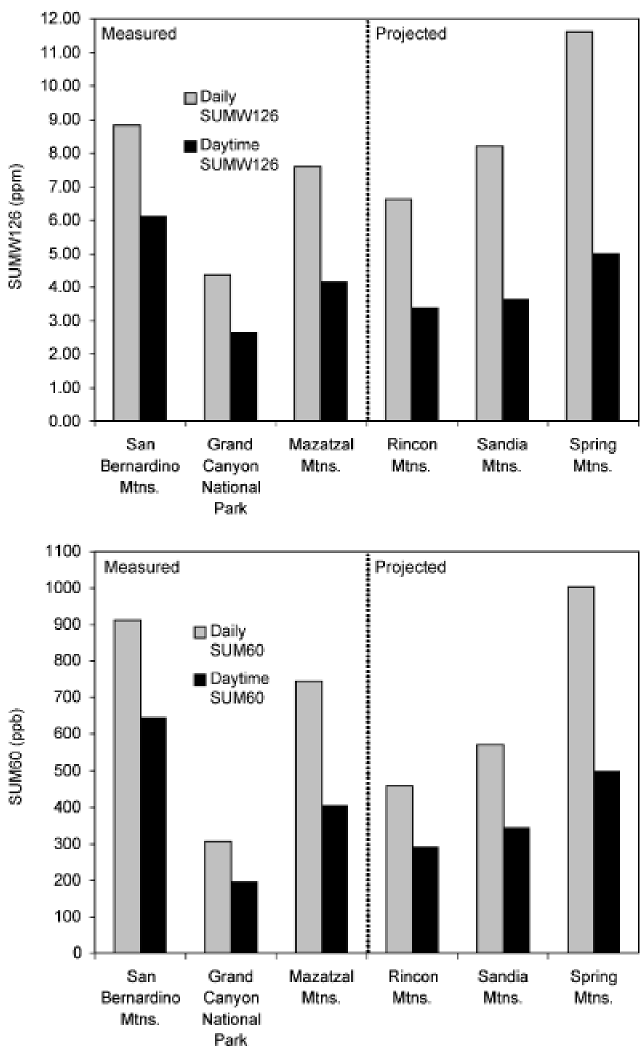

Figure 5. Measured seasonal ozone exposure in the western San Bernardino Mountains, Grand Canyon National Park, and the Mazatzal Mountains, and projected seasonal ozone exposure in the Rincon, Sandia, and Spring Mountains. See endnote 8 for an explanation of the method used to calculate the projected values.

The above environmental controls on stomatal conductance can be highly variable within southwestern coniferous forests, especially those forested areas with much topographic relief. Temperature and light intensity vary greatly across complex terrain. Needle-to-air VPD and water stress decrease with an increase in elevation; thus, increased relative humidity and increased soil moisture at higher elevations should increase stomatal conductance. Although elevated CO_2 concentrations are well documented for the Phoenix area (e.g., Idso, Idso, and Balling 1998, 2001; Wentz et al. 2002), it is not known if "CO_2 domes" exist in the other urban areas or if CO_2 is transported to the coniferous forests. The existence and effects of elevated CO_2 concentrations in coniferous forests cannot be postulated, given the available information.

Moisture-related factors should produce the largest intraregional and intraseasonal variations in stomatal conductance. Based on results for northern Arizona (Kolb and Stone 2000), stomatal conductance throughout most southwestern coniferous forests probably exhibits a bimodal distribution during the ozone season, with peaks in April/May and August/September. Enhanced stomatal conductance in April and May is enabled by increased soil-water availability, provided by winter precipitation. Beginning in July, the monsoon increases relative humidity levels and soil-water availability and decreases needle-to-air VPDs (Kolb and Stone 2000), thereby preventing the mid- to late-summer decrease in stomatal conductance that would occur in the absence of the monsoon. The arrival of the monsoon is potentially the largest control of intraseasonal variations in ozone flux in most of the region's coniferous forests. Monsoonal moisture could increase ozone flux dramatically in the late season.

Nocturnal stomatal conductance by southwestern conifers could increase the daily ozone flux considerably and lead to increased ozone-induced damage to conifers. Although nocturnal stomatal conductance is often lower than daytime stomatal conductance (Musselmann and Minnick 2000), nocturnal ozone flux can still be

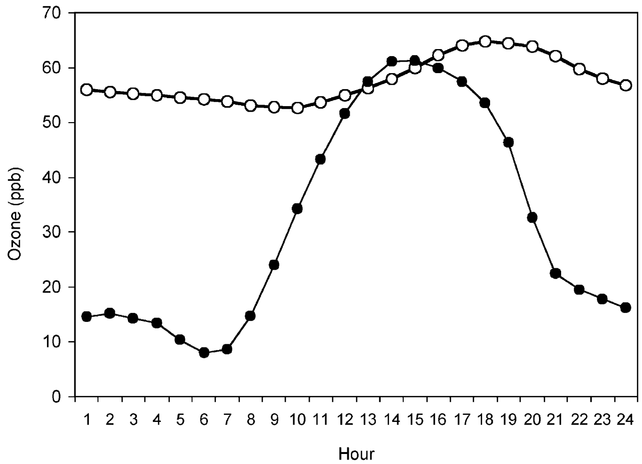

Figure 6. Diurnal variations in ozone concentrations at Mt. Ord (—○—) and central Phoenix (—●—).

significant, resulting from ozone levels remaining high throughout the night. Elevated nocturnal ozone levels result from minimal chemical scavenging by nitric oxide (NO) (Lefohn and Jones 1986). For example, nocturnal ozone levels are much higher at Mt. Ord than in central Phoenix (Figure 6), which is exposed to excessive NO concentrations resulting from motor-vehicle traffic. Although high nocturnal ozone levels probably exist in many southwestern coniferous forests, owing to inadequate knowledge of nocturnal stomatal conductance it is not known if those levels affect those forests. Pacific ponderosa pine may exhibit nocturnal stomatal conductance (Bauer et al. 2000), so the interior and Arizona varieties may have similar behaviors. Notwithstanding the uncertainties associated with ponderosa pine varieties, significant nocturnal stomatal conductance has been measured for Douglas fir (*Pseudotsuga menziesii*) (Blake and Ferrell 1976; Running 1976), which is also a common constituent of southwestern coniferous forests.

The critical role of stomatal conductance may actually yield similar levels of ozone flux between conifers in the Mazatzal, Rincon, Sandia, and Spring Mountains and conifers in the western San Bernardino Mountains, which, as mentioned previously, have had severe ozone-induced forest damage. The lack of summertime moisture in the San Bernardino Mountains may lower stomatal conductance and ozone absorption, especially from July through September (Grulke 1998; Temple and Miller 1998). As discussed earlier, early-season ozone at Mt. Ord is actually comparable to that at Crestline, and the suspected late-season increase in stomatal conductance should increase ozone flux at Mt. Ord and at other forests impacted moderately to severely by the monsoon. Figure 7 shows hypothesized intraseasonal variations in ozone exposure, stomatal conductance, and ozone flux at a typical high-elevation site in the interior Southwest. If these variations are plausible, then late-season ozone flux in coniferous forests near southwestern cities is almost as large as early-season ozone flux. If there is almost an entire season of substantial ozone flux, then the likelihood of ozone-induced damage to southwestern coniferous forests increases. Yet, if many of the conifers exhibit reduced stomatal conductance regardless of environmental conditions, then the magnitude of ozone flux in the interior Southwest may be much lower than that in the San Bernardino Mountains, and the potential for ozone-induced damage may be less than expected.

Ozone-Induced Damage to Conifers

Ozone-induced damage to conifers, which is regulated by ozone flux and the action of passive and active defenses, occurs primarily at the needle level. Passive defenses include those antioxidants that are normally present in plant tissue to detoxify ozone, while active defenses are initiated after ozone enters plant tissue and may include increased antioxidant production (Musselman and Minnick 2000). Temporally, concentrations of antioxidants for detoxification are often lower later in the season (Esterbauer, Grill, and Welt 1980) and at night (Musselmann and Minnick 2000), which further complicates the ozone-induced injury process. Typical effects of chronic exposure to high concentrations of ambient ozone include chlorotic mottling (i.e., yellowing of needles), needle necrosis, reduced needle retention, reduced needle length, increased branch mortality, and decreased radial growth of stems (Stolte et al. 1992; Grulke and Lee 1997). For conifers, chlorotic mottling appears on the older needles and progresses to the current season's needles, premature abscission of needles progresses from older to younger needles, and necrosis extends from the needle tip inwards (Bytnerowicz and Grulke 1992). The functional importance of these changes is that they have been shown to lower net photosynthesis and alter basic processes of water, carbon, and nutrient allocation by trees (Bytnerowicz 1996; McLaughlin and Percy 1999).

Pines tend to be the most ozone-sensitive conifers in western forests, with Pacific ponderosa pine, Jeffrey pine (*Pinus jeffreyi*), and western white pine (*Pinus monticola*) thought to be the most sensitive species/varieties (Bytnerowicz and Grulke 1992). Nevertheless, interior ponderosa pine is possibly three times more ozone-tolerant than is the Pacific variety (Miller, Longbotham, and Longbotham 1983), and it has not exhibited ozone-injury symptoms in the Rocky Mountains (Peterson, Arbaugh, and Robinson 1993). The interior variety's

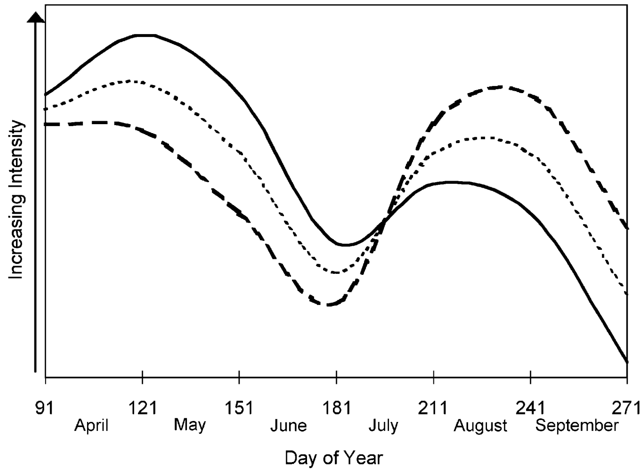

Figure 7. Hypothesized intraseasonal variations in ozone exposure (——), stomatal conductance (– – –), and ozone flux (- - - - - -) at a typical coniferous forest in the southwestern United States. Ozone-exposure estimates are based on levels at Mt. Ord and the urban-fringe monitors, and stomatal-conductance estimates are based on values presented in Kolb and Stone (2000) for northern Arizona. Ozone-flux intensity is estimated by averaging the intensity estimates for ozone exposure and stomatal conductance.

ozone-tolerance is attributed primarily to its low stomatal conductance (Monson and Grant 1989).

It is known that some southwestern conifers are sensitive to ozone. Previous research results provide partial evidence of ozone-induced forest damage in the Rincon Mountains (Duriscoe and Selph 1985; Duriscoe 1990; Diem 2002). Unfortunately, no studies have differentiated between the varieties of ponderosa pine. However, it can be hypothesized that most of the damage occurred to Arizona ponderosa pine, certain hybrids, or both, because interior ponderosa pine is presumably ozone-tolerant.

Both early- and late-season ozone may be a potential threat to the forests containing ozone-sensitive trees. Late-season damage may be caused not only by high ozone fluxes but also by potentially lower concentrations of antioxidants in August and September. A sharp rise in damage should be evident from June to August. Assuming that interior ponderosa is not injured by ozone and that the ozone sensitivity of Arizona ponderosa pine is significant, the most damage in the interior Southwest should occur in the Rincon Mountains in April, May, and August. If this is the case, then ozone-induced ecosystem impacts are the most severe in those mountains.

Indirect Effects of Ozone-Induced Damage

In addition to the direct effects of ozone exposure on individual trees, ozone has more complex impacts on coniferous forest ecosystems. Under the influence of several stressors, tree mortality could result indirectly from high pollution loads. For example, Grulke and colleagues (1998) found that across an ozone-pollution and nitrogen-deposition gradient, root biomass in trees at the least polluted sites were up to fourteen times greater than those at the most polluted sites. These changes can lead to substantial loss of vigor, which can increase a tree's susceptibility to other stresses, such as drought and insect infestations (e.g., bark beetle [*Dendroctonus brevicomis*]), cause abnormally high rates of tree mortality, and initiate changes in forest structure and function (Miller, Taylor, and Wilhour 1982; McLaughlin and Percy 1999; Pronos, Merrill, and Dahlsten 1999). Pollution stress can also reduce the genetic variability within populations by removing sensitive individuals; sensitive trees will ultimately become unable to compete with the less sensitive trees (Binkley, Droessler, and Miller 1992; Bytnerowicz and Grulke 1992). The impacts on individual conifers discussed above can have more significant and long-term consequences when assessed at the ecosystem level.

Ozone exposure could also be responsible for excessive needle senescence, thereby resulting in large amounts of litter on the forest floor. This litter has long-term implications with respect to pine-seedling establishment, carbon sequestration, and fire sensitivity (Grulke and Baldumann 1999). With severe ozone-induced damage, a ponderosa pine may retain needles for only a single year (Stolte 1996) compared to three to five years for a pine in an unpolluted area (Sudworth 1908). Hence, severely damaged conifers could be labeled "deciduous conifers" (Grulke and Baldumann 1999), and their increased needle senescence could contribute to increased ground litter and increased fuel loads. Subsequent tree mortality would increase the quantity of dead fuels. Natural surface fires consume fuels and maintain open stands (Weaver 1951). Because the natural-fire process was essentially eliminated in the late nineteenth century in parts of the interior Southwest, however, fuel loads are at high levels resulting from more than a century of fire suppression (Baisan and Swetnam 1990; Swetnam and Baisan 1996). Increased dead fuels resulting from ozone-induced damage would amplify the potential for exceptionally large fires.

Applying the ozone/forest health concepts to the interior Southwest indicates that there is a strong potential for moderate to severe ozone-induced damage to many of the region's coniferous forests. The same ecological problem that has occurred in the San Bernardino Mountains, where ozone exposure has caused a decline in forest health throughout the range (Innes 1992), may be occurring presently in forests near Albuquerque, Las Vegas, Phoenix, and Tucson.

Conclusions

This review article has synthesized conceptual information from a variety of disciplines to develop a theoretical rationale for ozone impacts on coniferous forests in the interior Southwest. The assessment indicates that it is entirely likely that ozone is damaging southwestern coniferous forests, especially those near the largest cities. Climate is a central component in the ozone and forest-health conceptual model, for it exerts considerable control over ozone exposure and ozone flux in the region. Based on the presence of high ozone-exposure levels in the Mazatzal Mountains near Phoenix, ozone-exposure levels in the mountains downwind of Albuquerque, Las Vegas, and Tucson are hypothesized to be equal to or greater than levels in the Mazatzals. Thus, mountains near the region's four largest cities have estimated ozone-exposure levels high enough to cause damage to sensitive conifers. The suspected high ozone levels in those mountains appear to be a function of anthropogenic pollutant emissions in nearby urban areas, synoptic-scale pollutant transport from southern California, and the intrusion of stratospheric ozone. Early-season ozone exposure in all the above mountains should equal or exceed levels in the severely ozone-damaged San Bernardino Mountains of southern California.

Ozone flux in the interior Southwest should peak in the early-season; however, unlike the case in southern California, there should also be a secondary peak in flux during the late season enabled by the North American monsoon. The monsoonal moisture increases stomatal conductance and subsequent ozone flux. Both early-season ozone and late-season ozone should be important in the interior Southwest, especially in New Mexico and southern Arizona. Even though the environment appears to be conducive to high ozone flux and subsequent high ozone-injury levels, genetic characteristics that reduce stomatal conductance may confer protection on some conifers.

The ozone/forest dynamic, which involves interactions among a diverse array of variables, is complex and contains many uncertainties. Those uncertainties hamper the effort of gaining a full understanding of ozone impacts on southwestern coniferous forests. Ambient ozone levels and the sensitivities of southwestern conifers to ozone represent the most crucial uncertainties. Ambient ozone measurements are unavailable for nearly every forest stand in the interior Southwest. Ponderosa pine dominates most forests, but the ozone sensitivity of Arizona ponderosa pine, which occurs in the southernmost forests, is unknown. Prior results suggest that the sensitivity of interior ponderosa pine is low. The scarcity of data concerning ambient ozone levels and ozone sensitivity, which includes stomatal conductance, causes estimates of ozone flux and subsequent ozone-induced tree damage to be exceptionally speculative.

Determining the relationship between ozone and forest health in the interior Southwest needs to be on the overall scientific research agenda. This article has highlighted substantial research gaps, which yield the following questions:

- How do ozone exposure levels vary both spatially and temporally?
  - How much does the intrusion of stratospheric ozone contribute to ozone levels at high-elevation sites?
  - How much does atmospheric pollution transported over long distances contribute to ozone levels, and where does most of that pollution originate?
  - How complex are "surfaces" of ozone exposure levels within various forested areas?
- How does stomatal conductance vary both spatially and temporally as well as among species and varieties of conifers?
  - How does the North American monsoon regulate stomatal conductance?
  - What is the magnitude of nocturnal stomatal conductance?
- How does ozone-induced forest damage vary both spatially and temporally?
  - What is the relationship between ozone exposure and ozone-induced damage?
  - Which conifers appear to be the least and most sensitive to ozone?
  - What is the role of climate in controlling ozone-induced damage?

Answering the above questions requires several key pieces of information. Ozone concentrations need to be measured in all coniferous forest areas, especially forests near Albuquerque, Las Vegas, Phoenix, and Tucson. Ozone can be measured with continuous ozone monitors and ozone passive samplers (Koutrakis et al. 1993). In addition, extensive forest surveys are needed throughout the region's coniferous forests to assess the magnitude of ozone-induced damage and to determine which tree species/varieties are ozone-sensitive. Surveys should be both intraseasonal and interseasonal in nature to elucidate the impacts of the monsoon and climate variability on ozone-induced damage.

Ozone-induced damage to the southwestern coniferous forests is a potentially serious environmental problem. The many air-quality policy implications concerning both urban and rural locales amplify the problem.

Understanding the linkages between human activities and associated, spatially displaced impacts on nature is a difficult but important endeavor, requiring an approach that emphasizes different geographic scales (e.g., synoptic scale and mesoscale) while holistically examining links between human activities and physical processes. Consequently, the ozone/forest dynamic should be of eminent interest to geographers.

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Notes

1. Forest health is defined in a pathological sense as the incidence of biotic and abiotic factors affecting a population of trees within a forest ecosystem (Ferretti 1997).
2. National wilderness areas and national parks were afforded the highest degree of protection from air pollution-caused deterioration in future years by the Clean Air Act Amendments of 1977. Federal land managers in these Class I air-quality areas are authorized to protect air-quality-related values. Therefore, the land managers advise the state or federal government if emissions from a proposed project may adversely impact a wilderness area (Bunyak 1993).
3. The climate data pertain to the Palisade Ranger Station near Tucson, Arizona, and they were obtained from the Western Regional Climate Center, as noted in Table 1.
4. Population estimates for relevant MSAs were obtained from the U.S. Census Bureau, as noted in Table 1.
5. Estimates of daily vehicle miles traveled for relevant MSAs were obtained from the U.S. Environmental Protection Agency, as noted in Table 1.
6. Ozone exposure indices are used to weight ozone concentrations, thereby providing a more realistic measure of the severity of the exposure of conifers to ambient ozone. Two indices, SUM60 and SUMW126, are employed in this study. SUM60 is the sum of all hourly concentrations greater than or equal to 60 parts per billion (ppb), while SUMW126 is the sum of sigmoidally weighted hourly ozone concentrations; less weight is given to lower ozone concentrations. Hourly ozone concentrations were weighted using the following equation:

$$W_i = [1 + M^* e^{-(A^* C_i)}]^{-1}$$

where M and A were assigned values of 4403 and 126 ppm^{-1}, respectively. W_i is the weighting factor for concentration i, and C_i is the concentration of i (Lefohn and Runeckles 1987).

7. Estimates of anthropogenic emissions of VOCs and NO_x for relevant MSAs were obtained from the U.S. Environmental Protection Agency, as noted in Table 1.
8. Ozone-exposure levels in coniferous forests near Albuquerque, Las Vegas, and Tucson are estimated using the exposure levels at urban-fringe sites situated at or near the bases of forested mountains, which include the Sandia Mountains, Spring Mountains, and Rincon Mountains, respectively. Multiplying the Mt. Ord to eastern Phoenix ratio of average daily and daytime ozone exposures—which are 1.4 (daytime SUMW126), 2.1 (daily SUMW126), 1.5 (daytime SUM60), and 2.0 (daily SUM60)—by the urban-fringe exposure levels provide rough estimates of exposure levels in these three mountain ranges. The accuracies of the estimated ozone exposure levels are unknown; thus, the values should be treated with much less validity than observed values.

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