15. Summary of Air Pollution Impacts on Forests in the Mexico City Air Basin

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Oxidant air pollution symptoms were first reported in bioindicator plants in the Mexico City Air Basin (MCAB) in 1971 (de Bauer 1972). Classic injury symptoms on well-known bioindicator plants strongly supported the presumption that symptoms were caused by photochemical oxidants, of which ozone (O_3) is the primary pollutant. Symptoms in indicator plants characteristic of injury caused by peroxyacetyl nitrate (PAN), ethylene, and sulfur dioxide (SO_2) were also reported (de Bauer 1972; de Bauer and Hernández-Tejeda 1986). These discoveries were followed in 1976 by the observation of O_3 injury symptoms in native pine species in forests in Ajusco (AJ), just south of Mexico City (de Bauer and Hernández-Tejeda 1986; Krupa and de Bauer 1976). Then in the 1980s, a dramatic and severe decline occurred in sacred fir (*Abies religiosa*) stands in the highly polluted area of the Desierto de los Leones (DL) National Park southwest of Mexico City (Alvarado-Rosales and Hernández-Tejeda 2002). Thousands of trees died, leaving dead zones called cemeteries. Air pollution was widely believed to be an important causal factor in the widespread mortality of sacred fir within the park (Ciesla and Macias-Samano 1987). Topographic conditions, including volcanic mountain ranges circumscribing much of the Basin, thermal atmospheric inversions, and prevailing winds which carry pollutants from the urban zone to forested areas to the south-southwest (SSW), create conditions favoring high pollution exposure for these forests (Bravo and Torres 2002; Jáuregui 2002).

Continuing increases in population and in the number of motor vehicle miles driven in the Basin and increasing traffic congestion (Enríquez 2000) indicate that chronic and severe air pollution exposures will be a problem in these forests for years to come, or at least until more stringent emission-control regulations are in place and newer less-polluting vehicles become more prevalent. The ultimate solution to air pollution in the MCAB would be greater use of less-polluting mass-transit systems and a technological breakthrough in developing an economically viable and readily available clean energy source for motor vehicles. Fuel efficient and low polluting hybrid (electric and gasoline) vehicles are now coming on to the market in a few countries. However, in order for this or other new technologies to have a major impact on fossil fuel emissions in the MCAB, it would be necessary for millions of people to purchase new vehicles to replace older high-polluting vehicles, a cost-prohibitive alternative for many inhabitants of the MCAB. In addition, a large fraction of the emissions are from heavy-duty vehicles such as semitrailers, trucks, buses, and vans from which emissions are high.

Although current pollution control measures in the MCAB are insufficient to reduce the concentrations of all pollutants to levels below the governmental standards, progress has been made. For example, levels of SO_2 have decreased significantly since 1993 and rarely exceed the 24-hour standard. However, annual average levels of SO_2 are still slightly above the annual standard (Bravo-Alvarez and Torres-Jardón 2002). With the introduction of low-lead gasoline in 1986 and unleaded gasoline in 1990, similar progress has been made in reducing emissions of lead. The reduced incidence of of emergency levels of O_3 (from 177 days in 1992 to 7 days in 1999) suggests that limited progress has been made in reducing O_3 levels. (SMA 1999; http://sma.df.gob.mx/publicaciones/aire/inf_cal_aire99/ indice.htm). However, O_3 levels continue to be a major air pollution problem. The O_3 air quality standard is exceeded by more than 4 to 5 hours per day for at least 300 days per year (Bravo-Alvarez and Torres-Jardón 2002). Nitrogen (N) and sulfur (S) deposition levels also remain highly elevated in forests downwind of Mexico City (Fenn et al. 1999), notwithstanding the greatly reduced S content of fuel.

Major Pollutants with Potential to Impact Forests in the MCAB

The major air pollution types in the MCAB that have the potential to impact sensitive plant species or some aspect of forest ecosystem function and the major characteristics of these types of air pollution are listed in Table 15.1. The available evidence indicates that under current air pollution exposures, the pollutants with the greatest ecological and environmental effects are O_3 and N pollutants. Sulfur dioxide probably was an important pollutant with observable biotic effects in sensitive organisms such as lichens and some plant species, particularly

Table 15.1. Characteristics and potential forest effects of the major pollutant types in the Mexico City Air Basin (MCAB)

Pollutant	Characteristics	Major effects on forests in the MCAB
Ozone	A secondary gaseous pollutant, a product of photochemical reactions of NO_x and hydrocarbons; highly phytotoxic	Severe visible foliar and crown injury in sensitive pine species; may be an important factor in sacred fir decline; symptoms in some understory species; crop loss also documented
Nitrogen oxides	Primary pollutants, mainly from fossil fuel emissions, ozone precursor	Main effects may be as an ozone precursor; source of N compounds deposited to forested watersheds which "fertilizes" the forest and increases nitrate levels in runoff
Nitrate and nitric acid vapor	A secondary pollutant from NOx	Increases N fertility of the forests and increases nitrate in runoff; direct injury from nitric acid hasn't been studied in the MCAB
Peroxyacetyl nitrate	A secondary gaseous photochemical oxidant, occurring at lower concentrations than ozone	Concentrations not usually high enough to cause visible injury to vegetation in rural areas; the MCAB may be an exception, but data is lacking in rural areas
Sulfur dioxide	Mainly from combustion of fossil fuels, petroleum and natural gas industries, smelting and refining processes; a gaseous pollutant	Likely effects on sensitive organisms such as lichens; more severe in the north of the Basin; concentrations are lower in forests in the south of the Basin, but there is little data from forested areas; levels have decreased as a result of low sulfur fuels
Sulfate	A secondary pollutant from SO_2 emissions	Causes S enrichment of the forests; increased S content of plants; possible role in soil acidification and cation leaching in more exposed forests
Heavy metals	Mainly from motor vehicles and industry. No data available on deposition to forests	Lead was the major heavy metal. Emissions have greatly decreased since the mid-1990s. Phytotoxic symptoms not reported, unknown effects on more sensitive organisms and soil processes; Potential water quality problem
Ammonium-ammonia	Mainly from intensive livestock operations; some industrial sources	Also increases N fertility of forests; may contribute to soil acidification due to increased nitrification rates
Volatile organic compounds (VOC)	From fossil fuels, industry and emissions from plants	VOC are precursors of ozone and other oxidants. Direct effects of VOC on vegetation in the MCAB has not been studied

in the northwestern industrial sector (Zambrano et al. 2002), but concentrations have decreased since the early 1990s. Sulfur dioxide concentrations are higher in the northern part of the Basin, while forests are found predominantly in the southern portion of the Basin. Much of the SO_2 emitted from the industrial areas in the north is presumably transformed to sulfate aerosols and transported to the southwest by the predominant winds (Bravo-Alvarez and Torres-Jardón 2002). Phytotoxic symptoms attributable to SO_2 have not been reported for forest species in the MCAB. However, pollutants can have significant physiological effects on plants without inducing visible symptoms (Bussotti and Ferretti 1998; Evans et al. 1996; Krupa et al. 1982). Furthermore, controlled air pollution exposures have not been carried out for the major overstory species in the MCAB with any of the major pollutants. For example, it has not yet been confirmed which, if any, pollutants may cause the foliar symptoms observed in declining sacred fir trees (Alvarado-Rosales and Hernández-Tejeda 2002). Thus, the lack of reported SO_2 injury symptoms in forests of the MCAB could possibly reflect the lack of study or knowledge of symptom development in the native species. Considering the historically high S emissions from the MCAB, and the, as yet, high atmospheric sulfate deposition inputs in throughfall, it seems clear that S deposition in high-pollution forests of the MCAB has increased the S concentration in foliage (Fenn et al. 1999, López-López et al. 1998) and is probably affecting nutrient cycling processes to some degree (see Fenn et al. 2002a).

Although we are not aware of direct measurements of atmospheric deposition rates for lead (Pb) and other heavy metals in forested areas of the MCAB, data on metal concentrations in forest soils and vegetation (Castro-Servín et al. 1997; Fenn et al. 2002a) and recent dendrochemical analyses (Watmough and Hutchinson 1999) demonstrate that heavy-metal deposition, particularly in the case of Pb, was high in the past. Lichen studies also demonstrate high deposition rates of heavy metals in some parts of the Basin (Zambrano et al. 2002). With the elimination of leaded gasoline, the ecological effects of heavy-metal deposition may now be less of an issue than it was a few years ago. However, because of the accumulation of metals in soil and soil organic matter, heavy metals can be biologically important even after deposition levels decrease (Bargagli 1998). Lead that has accumulated in soil and organic matter throughout the watersheds can continue to be exported in drainage waters for many years, and there is preliminary evidence that this is occurring in the Basin of Mexico (Fenn et al. 2002a).

As mentioned, O_3 and N compounds appear to be the pollutants with the greatest potential for ecological and environmental impacts in the MCAB. Nitrogen pollutants can include substances in several physical (gaseous, particulate, dissolved) and chemical (reduced, oxidized, ionic) forms. The main biotic effects of O_3 and N pollutants are usually quite different, inasmuch as O_3 is phytotoxic and decreases the growth and productivity of sensitive plant species. Nitrogen, on the other hand, is a growth stimulant and is also the plant nutrient most frequently limiting plant growth in temperate forests (Vitousek and Howarth 1991). On the other hand, phosphorus (P) is often limiting in sites with volcanic soils such as in the Basin of Mexico (Fenn et al. 2002b).

By the time nitrogen oxides from Mexico City have been transported to forested areas outside of the city, the N pollutants have been largely converted to particulate nitrate or nitric acid vapor (HNO_3). Nitrogen and S deposition occurs mainly as gaseous or particulate deposition in dry form or as wet deposition during precipitation events. Washoff from forest canopies of accumulated dry deposition occurring as throughfall, is a major mechanism for deposition of air pollutants to forest ecosystems. In high-pollution areas, a significant fraction of dry-deposited pollutants, especially nitrogenous compounds, are not washed off by precipitation but are retained within the canopy. Thus, throughfall measurements in areas of elevated N deposition commonly underestimate total N deposition by approximately 30% to 40% (Fenn et al. 2000; Lovett and Lindberg 1993).

Based on recent atmospheric measurements in Mexico City (Gaffney et al. 1999) and symptoms in bioindicator plants (de Bauer and Hernández-Tejeda 1986), phytotoxic levels of PAN occur in Mexico City, However, possible effects of PAN in forests in the MCAB have not been studied. Levels of PAN measured in north central Mexico City are the highest reported in the world since similar levels were reported near Los Angeles, California, in the late 1970s. Concentrations in the southwest of the Basin, in or near the major forested areas, are expected to be even higher (Bravo-Alvarez and Torres-Jardón 2002). PAN levels in Mexico City exhibit a strong diurnal pattern, indicating that this pollutant is transported out of the city almost completely during each diurnal period (Gaffney et al. 1999). This suggests that forests downwind of the city, such as the DL Park, may be exposed to high concentrations of PAN on a regular basis. Further research is needed to evaluate the impacts of PAN exposure on vegetation in forests of the MCAB.

Plants in the MCAB are exposed to many other organic pollutants besides PAN (Bravo-Alvarez and Torres-Jardón 2002). The effects of volatile organic compounds (VOC) on vegetation have not been investigated in the MCAB. Based on a review of the literature, it was concluded that the indirect effects of VOC, as precursors to O_3 formation, are more important than the direct effects on vegetation (Smidt 1994). However, this same author concluded that the direct effects of VOC on forest decline have not been well defined. The concentrations of formaldehyde (one example of an important atmospheric VOC) in the MCAB are among the highest reported in the literature. The peak concentration of formaldehyde (110 ppb) was reported in the spring of 1993 at a site to the southwest of the urban zone (Baez et al. 1995). However, common bean plants exposed to 400 ppb formaldehyde for 7 hours per day, 3 days per week for 4 weeks grew larger than the control plants (Mutters et al. 1993). Mutters et al. concluded that formaldehyde concentrations much greater those that occur in urban areas are required to injure bean foliage.

Air Pollution Impacts on Forest Health and Sustainability

Geographic Location of Air Pollution Studies in the Mexico City Air Basin

Much of the work on air pollution effects in the MCAB has focused on the DL National Park. However, the first reported observation of air pollution injury symptoms to forest species was that of O_3-induced injury to *Pinus leiophylla* and *P. hartwegii* in the AJ area (Fig. 15.1) (de Bauer and Hernández-Tejeda 1986; Krupa and de Bauer 1976). Air pollution gradient design studies have been useful in evaluating the effects of air pollutants on forests and forest species in the San Bernardino Mountains in southern California (Fenn and Bytnerowicz 1993; Fenn and Dunn 1989; Miller et al. 1986) and in the forests of the MCAB as well (de Bauer and Hernández-Tejeda 1986; Fenn et al. 1999; Fenn et al. 2002b; Hernández-Tejeda 1984). Hernández-Tejeda studied O_3 injury in pine species along a transect or air pollution gradient following a route parallel to the highway from Mexico City to Cuernavaca (de Bauer and Hernández-Tejeda 1986; Hernández-Tejeda 1984). The most exposed forest site in the MCAB is the DL National Park, which is adjacent to the urban zone and directly downwind (to the

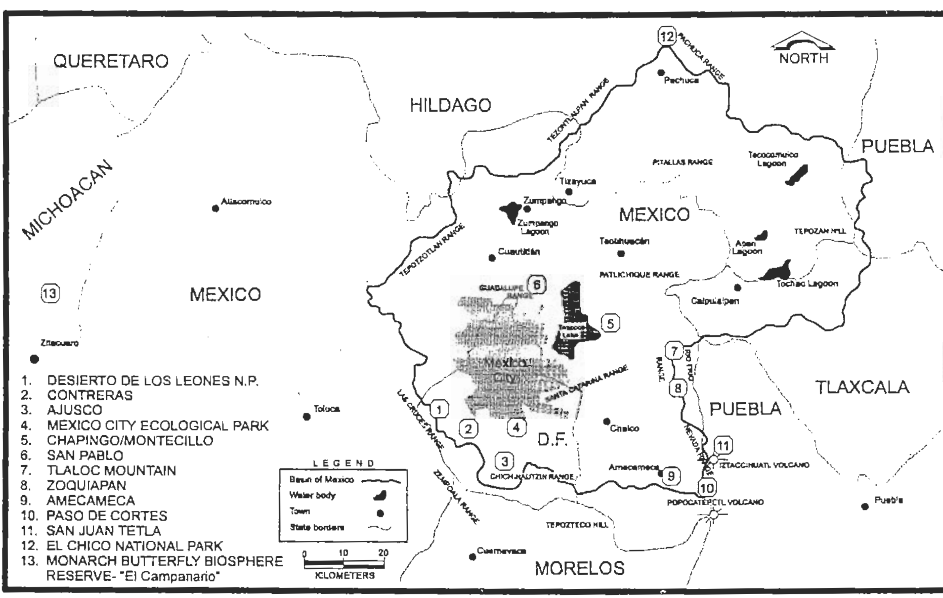

Figure 15.1. Location of the sites used by various researchers in studies of air pollution effects on forests in the Basin of Mexico. High-pollution sites are those to the SSW of Mexico City. Other sites indicated were mainly used as relatively low-pollution exposure sites. (Modified from Figure 3.1 in this volume.)

southwest) of Mexico City. The AJ area is SSW of the city (Fig. 15.1), and is also considered to be a high-pollution area, although it is apparently not as directly exposed to air pollution masses as DL. Lichen studies were carried out at high-pollution study sites including DL and the Mexico City Ecological Park located at the southwestern fringe of Mexico City (Fig. 15.1) (Zambrano et al. 2002). Forest sites which have been used as low-pollution sites (although their pollution levels are only low relative to the high-pollution sites) are found to the east or southeast of the city. Zoquiapan (ZOQ) National Park (Fenn et al. 1999), Paso de Cortes within the Iztaccíhuatl-Popocatépetl National Park (Fenn et al. 2002b), the municipality of Amecameca on the slopes of Popocatépetl at 13 km from Paso de Cortés (Alvarez et al. 1998), and San Juan Tetla in the state of Puebla (Fig. 15.1) Castro et al. 1997; Fenn et al. 2002a) are examples of such (relatively) low-pollution sites. In dry deposition and throughfall studies at DL and ZOQ, it was concluded that S deposition at ZOQ was higher than would have been expected based on the location of the site in relation to Mexico City because of volcanic outgassing from the Popocatépetl volcano during the sampling period (Fenn et al. 1999). Researchers using any forest sites within the zone of influence of Popocatépetl as low-pollution study sites should keep in mind that emissions from the volcano are likely to affect the atmospheric chemistry and deposition occurring in the area (Delmelle et al. 2001), thus possibly compromising the value of these sites as controls. Other control sites that have been used include the El Chico National Park just north of the boundary of the Basin of Mexico (Zambrano et al. 2002), the El Campanario study site within the Monarch Butterfly Biosphere Reserve in the state of Michoacán (Terrazas and Bernál-Salazar 2002), and San Pablo, located 45 km northeast of Mexico City (Watmough and Hutchinson 1999). Additional data is needed to better characterize air pollution exposures at these study sites. Recent developments in the use of passive samplers in conjunction with strategically located electronic monitors (Cox and Malcolm 1999; Koutrakis et al. 1993; Krupa and Legge 2000; Runeckles and Bowen 2000; Yamada et al. 1999) will facilitate collection of the data needed to better define pollution gradients in the MCAB.

Sacred Fir Decline

The most dramatic effect attributed to air pollution in the MCAB is the severe forest dieback of *Abies religiosa* in the DL National Park (Alvarado-Rosales et al. 1993; Cibrian-Tovar 1989; Ciesla and Macias-Samano 1987). Mortality of sacred fir as the end stage of this syndrome was first observed in 1981 (Cibrian-Tovar 1989) and increased throughout the 1980s. In recent years, mortality rates have declined, but symptoms of decline are still present in the stands. Sacred fir decline was so severe in the 1980s that entire stands were decimated, cleared, and replanted with alternative species. Replanting with sacred fir was not successful, possibly because of unfavorable conditions for seedling survival. The etiology of the decline appears to be complex, and the role of air pollution in the decline is not definitively known. Foliar symptoms begin as a whitish stippling. These lesions later coalesce and develop into a reddish-brown discoloration of the foliage. These symptoms are putatively caused by O_3, although controlled exposures to O_3 have not been performed with this species. However, the decline is particularly severe at the ends of ravines where air pollutants accumulate. Furthermore, foliar symptoms were prevented in branches of sacred fir enclosed in charcoal-filtered branch chambers or treated with an antitranspirant known to greatly reduce O_3 injury (Alvarado-Rosales et al. 1991, 1993). Based on the available evidence, oxidant air pollution is widely believed to play an important role in the decline of sacred fir in the park (Alvarado-Rosales and Hernández-Tejeda 2002; Ciesla and Macias-Samano 1987). Controlled exposures to O_3 will be carried out in the near future to determine whether O_3 causes the observed foliar symptoms. However, other phytotoxic pollutants, such as PAN, could also play a role in the observed foliar injury (Gaffney et al. 1999; Temple and Taylor 1983), although this has not been studied.

Stand densification and overmaturation of the trees are also believed to be contributing, and possibly critical, factors in the development of fir decline. The prohibition of silvicultural management of the forest in the park results in overcrowded and stressed trees. In neighboring forests, where thinning and harvesting for timber have occurred throughout the years, no decline has occurred. Drought, possibly exacerbated by the harvesting of water from springs throughout the park, may also contribute to the decline. Nutritional deficiencies, insect attacks, and pathogens have also been considered as contributing factors, but it is difficult to determine if these are casual factors in the decline or secondary factors occurring in declining trees (Alvarado-Rosales and Hernández-Tejeda 2002).

Ozone Injury of Pine

The primary forest species known to be severely impacted by air pollution is *Pinus hartwegii*, which is highly sensitive to O_3 injury (de Bauer and Hernández-Tejeda 1986; Miller et al. 2002). The development of injury symptoms in *P. hartwegii* is virtually identical to the classic symptoms described for *P. ponderosa* in the Los Angeles Air Basin in southern California (Miller et al. 2002), where high O_3 levels also occur in forested areas. Ozone-induced symptoms in both pine species include thinning of the crown, chlorotic mottling on the foliage, and premature senescence and abscission of the older foliage. In the San Bernardino Mountains in southern California, mature *P. ponderosa* trees exposed to elevated O_3 concentrations and N deposition have dramatically reduced fine root biomass, and greater foliar litterfall and litter accumulation (Grulke et al. 1998; Takemoto et al. 2001). These factors have not been studied in forests of the MCAB, but reduced root production as a result of O_3 exposure is a likely effect in the Mexican forests as well, considering that reduced root: shoot ratios are a general response of plants stressed by O_3 (Andersen et al. 1997;

Landolt et al. 1997; Scagel and Andersen 1997). Ozone-induced injury symptoms have also been reported on native understory species in the MCAB, including *Sambucus mexicana* (*Sauco*), *Sicyos* sp. (*Chayotillo*), *Eupatorium* sp., *Piqueria trinervia*, and *Solanum* (de Bauer and Hernández-Tejeda 1986).

In the MCAB, O_3 levels are high in both summer and winter, but in California, O_3 levels are very low in winter (Miller et al. 1994). Even more significant is the fact that for much of the summer, when O_3 levels are highest in California, soil moisture is low, thus severely limiting stomatal conductance and foliar uptake of O_3 (Temple and Miller 1998). The temporal asynchrony between the greatest available soil moisture and high O_3 levels limits the amount of O_3 injury incurred by ponderosa pine. In contrast, soil moisture levels are high in summer in the MCAB, and at the sametime, O_3 levels rise well above the threshold concentrations for inducing O_3 injury. Thus, because of concurrent high levels of O_3 and soil moisture, there is greater potential for O_3 injury in sensitive species in the MCAB, although O_3 injury scores were similar between high-pollution sites in the MCAB and the San Bernardino Mountains in the Los Angeles Air Basin (Miller et al. 1994; Miller et al. 2002).

Dendrochronology Studies

Chlorotic mottle on pine needles and other well-defined injury symptoms on indicator plant species provide strong evidence of areas where air pollution is exerting significant biological effects on sensitive forest species in the MCAB. Dendrochronological studies indicated that radial growth rates for *P. hartwegii* decreased significantly at DL and, to a lesser degree, at AJ from the 1970s onward (Alarcón et al. 1993). Similar growth trends have been demonstrated in several tree ring studies of *A. religiosa* at DL (Alvarado et al. 1993; Bernal-Salazar et al. 2002; Terrazas and Bernal-Salazar 2002). Xylem tracheid length and wall thickness also decreased after the 1970s, the period corresponding to the occurrence of severe air pollution problems in the MCAB. These and other tracheid modifications were similar to those previously reported for declining trees exposed to air pollutants (Terrazas and Bernal-Salazar 2002). These alterations in wood tissue structure in air pollution impacted trees of *A. religiosa* appear to be good indicators of physiological air pollution stress in trees.

Effects of Heavy Metals in Forests of the MCAB

Atmospheric deposition rates of Pb and other heavy metals have not been measured in forests in the MCAB, but data on metal concentrations in lichens, foliage, and soil from several studies demonstrate that inputs of heavy metals are greater in the forested areas to the SSW of Mexico City where exposure to O_3 and other pollutants is known to be higher. As a result of the introduction of unleaded fuels in 1990 and the phase out of leaded gasoline in 1997, Pb deposition to forested areas in the MCAB has undoubtedly decreased greatly in recent years. The available evidence suggests that levels of heavy metals in the plant species studied to date are not sufficient to cause noticeable phytotoxic effects. However, this is not to say that ecological effects on more sensitive species and processes have not occurred. Litter decomposition is known to be inhibited by elevated levels of heavy metals (Laskowski et al. 1994) and N transformation processes in soil can be affected at levels of heavy metals as low as three times background concentrations (Bääth 1989; Tyler et al. 1989).

Watmough and Hutchinson (1999) studied the trace metal chemistry of soil and tree rings of *A. religiosa* at DL and at San Pablo, a control site 45 km upwind (northeast) of Mexico City. Surface soils at DL contained elevated levels of trace metals, especially Pb, Cd and Zn. Lead and Cd levels were higher in tree rings formed since the 1960s. However, peaks in Cd and Pb were found in rings formed between 1920 and 1940, corresponding to the heartwood–sapwood boundary, especially in trees growing in contaminated soils with low pH. The authors suggested that Pb and Cd taken up by roots of sacred fir accumulate in the heartwood, whereas Pb and Cd entering through the bark are transported radially to a much lesser extent and more accurately reflect changes in trace-metal deposition. The accumulation of Cd and Pb in the heartwood–sapwood boundary limits the usefulness of this technique, at least with *A. religiosa*. However, it is possible that this technique may be more appropriate in other species as a methodology for determining temporal patterns in pollutant uptake in forests of the MCAB. Bargagli (1998) suggested that whenever possible, ring-porous trees such as oaks, elms, and ashes should be used for these types of studies because in these species, most of the water moves through only the current year's wood vessels.

There is a report of Pb concentrations in excess of the Mexican drinking water standard in spring water from the Ajusco area (Morales 1998). In subsequent streamwater studies at forest sites across the MCAB, including streams and springs in the high-pollution sites AJ and DL National Park, Pb concentrations tended to be higher at the two high-pollution sites (Fenn et al. 2002a), but concentrations were generally much lower than reported by Morales (1998). Further studies are needed to more fully assess the severity and extent of Pb export in drainage waters.

Air Pollution Effects on Lichen Communities

Epiphytic lichen diversity at DL is severely impoverished compared to El Chico National Park, a low-pollution site 100 km northeast of Mexico City. DL may have lost nearly 50% of its lichen species and lichen abundance is reduced by 60%, presumably as a result of the severe air pollution levels occurring at this site. Based on historical herbarium collections, the decline in lichen diversity appears to coincide with the period of accelerated industrial and population growth of Mexico City since the 1930s and 1940s (Zambrano et al. 2002). Concentrations of airborne metals such as Cu, Pb, and Zn in lichens from forests near the city indicate that metal deposition was more than twice as high in forests near the city as in forests far from the city. Short-term lichen transplant experiments in Sierra de las Cruces, a montane region just SSW and downwind of the urbanized zone, showed a 30% lower carbon fixation and 15% to 25% chlorophyll degradation compared to samples in El Chico National Park (Zambrano and Nash 2000; Zambrano et al. 1999). These results suggest that chronic air pollution is a major cause of lichen decline in forests surrounding the city, along with a variety of other anthropogenic disturbance factors.

Air Pollution Effects on Urban Forests

Because of the diversity of plant species grown in Mexico City and the surrounding urban zones, and considering the exposure of vegetation to elevated concentrations of several phytotoxic pollutants in the MCAB (Table 15.1), there is little doubt that sensitive species have been severely impacted. However, few studies have directly addressed pollution impacts on urban trees. One documented example is the discovery of O_3 injury in eucalyptus seedlings growing in Mexico City. That the observed injury symptoms in eucalyptus were in fact caused by O_3 was subsequently confirmed in controlled O_3 exposure studies (Hernández-Tejeda et al. 1981). Unfortunately, most reports of air pollution injury in urban vegetation are anecdotal or consist of observations without experimental confirmation of the cause of plant injury. Martínez and Chacalo (1994) discuss the major tree species grown in Mexico City and report on the air pollution sensitivity of the described species. However, the basis on which a particular species is reported to be sensitive to air pollution is not given, nor do they mention to which air pollutants the plants are sensitive or tolerant. This type of unsubstantiated general information is limited in its reliability, specificity, and utility.

More studies on the effects of air pollution on urban trees and vegetation are needed before clear guidelines can be established concerning the preferred species for use in urban plantings. In the absence of more information on air pollution sensitivity of the species used in the MCAB, possibly the best initial guide would be information from the literature on studies of the responses of similar species to air pollution. To this end, Hernández-Tejeda and de Bauer (1989) published a synopsis of air pollution effects on vegetation in the MCAB, along with lists of plant species, mainly from North America, Europe, and Mexico, and their relative sensitivity or tolerance to the major pollutants, O_3, PAN, nitrogen oxides, and SO_2.

Nitrogen Saturation: Effects of N Deposition on Nutrient Cycling and Water Quality

Nitrogen saturation can be defined as the long-term removal of N limitations on biotic activity in an ecosystem, accompanied by a decrease in N-retention capacity. Recent studies in forests of the northern hemisphere have shown that with chronic atmospheric N inputs, an excess of N accumulates in the system, and the system becomes N saturated. The classic symptom of an N-saturated watershed is the export of excess N as nitrate, which is leached through the soil profile and into the groundwater and surfacewater system. Thus, forests normally exhibiting a closed N cycle become open systems, and excess N is lost, mainly as leached nitrate, and to a lesser degree as nitrogenous trace gasses emitted from soil as the result of microbial processing of soil N. The latter process typically results in greater greenhouse gas emissions (e.g., N_2O, NO) from forested lands (Fenn et al. 1996; Fenn et al. 1998; Gasche and Papen 1999). The phenomenon of N-saturated watersheds demonstrates that air quality (atmospheric N pollutants) and water quality (nitrate levels) are intricately linked in forested watersheds receiving high atmospheric N deposition. Nitrogen deposition is also believed to stimulate forest growth, up to a point, although this is an open and complicated issue, especially because increased N availability can also impair mycorrhizal and root production and function and causes other disruptions of normal ecosystem processes (Fenn et al. 1998; Skeffington and Wilson 1988).

Nitrogen levels in vegetation and nitrate concentrations in runoff appear to be much more affected by N deposition in the San Bernardino Mountains east of Los Angeles, California, than in the DL National Park (Fenn 1991; Fenn and Poth 1999; Fenn et al. 1996; Fenn et al. 1999), based on comparisons with sites on the low end of N deposition gradients in both regions. Furthermore, there was greater foliar growth in pine trees fertilized with N in southern California, especially in N-limited sites with annual throughfall depositions of less than 12 kg/ha (Kiefer and Fenn 1997), but foliar growth of pine trees at sites with low or high N deposition in the MCAB did not respond to N fertilization. These results, and the inherently low C : N ratios and high total N content of the Mexican forest soils, suggest that the pine trees growing on these volcanic soils are not severely N limited. Some studies suggest that phosphorus may be limiting tree growth in forests in the MCAB (Fenn et al. 2002b), which would not be surprising considering the well-known high P fixation capacity of these andisols (Brady and Weil 1999; Fenn et al. 2002b). This may also explain why N concentrations in foliage often did not differ between sites across the pollution gradient in the MCAB. In a few instances, foliar N levels were significantly higher at the polluted sites, but differences were small compared to responses to N deposition in the San Bernardino Mountains in California (Fenn et al. 1996; Fenn et al. 1999).

In summary, it appears that the most significant effect of chronic N deposition to forests in the MCAB is to increase nitrate concentrations in springs and streams which provide drinking water for the adjacent urban population. However, preliminary evidence from several studies also suggests a relationship between greater soil acidity, Al mobilization, and loss of soil Ca and Mg at the more exposed high elevation pine sites in the DL (reviewed in Fenn et al. 2002a in this volume). More research is needed to evaluate these findings and to determine the possible role of atmospheric deposition of N and S in this phenomenon. Further study of the effects of N and S deposition on ecosystem processes is also warranted in vegetation types other than pine forests. For example, deposition of N, S, and other pollutants is expected to be greater under *A. religiosa* (e.g., Fenn et al. 2002a; Watmough and Hutchinson 1999) because of its greater surface area for collection of atmospheric deposition (Van Ek and Draaijers 1994), thus possibly increasing the magnitude of perturbations on nutrient cycling and plant function in fir stands.

Air Pollution Effects on Agronomic and Herbaceous Species

Oxidant air pollution injury symptoms have been observed in the MCAB on a number of agronomic species, on native understory species at forested sites, and on many well-known bioindicator species (de Bauer 1972; de Bauer and Hernández-Tejeda 1986; Laguette-Rey et al. 1986; Manning 1998; Ortíz-García et al. 2002). Oxidant air pollution injury has been reported for pinto beans, tobacco, lettuce, oats, soybeans, radish, petunia, eucalyptus seedlings, spinach, annual bluegrass (*Poa annua*), *Sicyos* sp., *Eupatorium* sp., *Piqueria trinervia*, and *Solanum verrucosum*, among others in the MCAB. In one study with pinto bean crops growing near Montecillo in the State of Mexico (a site usually upwind of Mexico City), bean production of an O_3-tolerant variety (Canario 107) was reduced by 4.5% when plants were exposed to ambient levels of oxidants. In contrast, bean production of an O_3-sensitive variety (Pinto 111) was reduced by 40.7% when plants were exposed to ambient levels of photochemical oxidants (Laguette-Rey et al. 1986; Ortíz-García et al. 2002). The observation of visible O_3 injury in so many different plant species in the MCAB demonstrates the widespread effects of oxidant air pollution on vegetation in the Basin. Furthermore, it is likely that physiological effects are occurring in other species before visible injury is apparent, or in some cases, when no visible foliar injury symptoms ever develop (Evans et al. 1996; Krupa et al. 1982).

Summary

The first evidence that air pollution was impacting vegetation in the MCAB was the observations of foliar injury symptoms in bioindicator plants (de Bauer 1972). These symptoms were attributed to O_3, PAN, SO_2, and possibly other pollutants. Subsequently, O_3-induced injuriy to foliage and crowns of pine trees were reported in forests to the south and southwest of Mexico City (de Bauer and Hernández-Tejeda 1986; Krupa and de Bauer 1976). Ozone is considered to be the pollutant that impacts most severely on vegetation within the urban zone and in forests downwind of the city. *P. hartwegii*, the most O_3-sensitive pine species, is severely impacted by the high O_3 exposures occurring SSW of the metropolitan area (Miller et al. 2002). The potential for O_3 injury is particularly high in the MCAB because O_3 levels are high during the summer rainy season, when soil moisture availability and stomatal conductance are greatest. These factors enhance O_3 uptake and injury. The decline of *A. religiosa* (*oyamel*) in the DL National Park is now a world-famous example of dramatic dieback and mortality of entire forest stands, presumably, at least in large part, as a result of air pollution stress (Alvarado-Rosales and Hernández-Tejeda 2002). However, other factors, such as the lack of stand thinning, are also believed to contribute, predisposing the trees to further decline.

Concentrations of PAN, and possibly concentrations of other oxidants, may also be damaging vegetation in some areas of the MCAB, but this has not been investigated. Aside from bioindicator evidence of PAN injury, the effects of organic pollutants in the MCAB or plants have not been studied, and little information is available to suggest whether organic pollutants are a significant threat to vegetation. Atmospheric deposition of Pb in the MCAB has undoubtedly decreased dramatically since the introduction of unleaded gasoline in September, 1990, and the phase out of leaded gasoline in August, 1997. The limited available evidence suggests that present foliar concentrations of heavy metals in forest species are not phytotoxic (Fenn et al. 2002a). Sulfur dioxide concentrations decreased in the early 1990s as a result of regulatory mandates limiting emissions. However, sensitive organisms in the northeast and northwest sectors of the urban zone, where concentrations are highest, may still be impacted by exposure to SO_2. Deposition of ionic forms of N and S in throughfall are equally high in forested areas southwest of the city. The effects of these chronic nutrient inputs to the forest are only beginning to be investigated and understood.

The most significant effect of elevated N deposition SW of Mexico City seems to be the increased levels of nitrate in runoff from forested watersheds downwind of Mexico City. This phenomenon has possible implications for the quality of water extracted from these forested watersheds that are expected to provide a relatively pristine water source for the adjacent urban populations. However, notwithstanding the increased streamwater nitrate concentrations, even the highest value reported to date from springs in DL (132 μeq/L) are still well below the Mexican and U.S. drinking water standard (714 μeq/L or 10 ppm as nitrate nitrogen). Further studies are needed to evaluate the extent of Pb export in drainage waters from forested watersheds in high-pollution sites to the SSW of Mexico City.

The ecological perturbations caused by severe exposures to air pollution in forests located downwind of Mexico City are expected to continue for the near future (the next 5–10 years), largely as a result of stubbornly high levels of O_3 and emissions of N oxides. The scenario over the longer term is more uncertain and depends largely on the effectiveness of regulatory emissions control strategies. Based on recent trends, it is hoped that pollutant levels will continue to fall. Forest responses to these downward trends will depend on how long it takes to reduce levels sufficiently to allow sensitive species to recover. Of course, some of the ecological damage, such as the loss of lichen diversity and of sensitive species, is probably irreversible (Zambrano et al. 2002). Preservation, and in some cases, prevention of further deterioration of the forests and green areas within the Basin of Mexico will certainly depend on the cooperation of government officials, regulatory agencies, and the general public. It is hoped that the increasing interest in ecological preservation and greater awareness of the importance of the ecological services provided by forests will make this goal achievable. Air pollution is but one of the anthropogenic factors threatening the preservation of the remaining forest ecosystems in the MCAB.

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