# 15 TEMPERATE SIERRAS

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## 15.1 Ecoregion Description

Much of the Temperate Sierras ecoregion is located in Mexico, but a northern portion of this ecoregion is found in the southwestern United States (mainly in Arizona and New Mexico) (CEC 1997, Fig. 2.1). The ecoregion description is adapted from CEC (1997). Forested montane regions are typically surrounded at lower elevations by arid and semi-arid deserts, scrublands, or grasslands. Vegetation can be evergreen or deciduous, and is primarily composed of conifers and oaks which grow from 10 to 30 m, sometimes reaching 50 m. This vegetative cover may comprise from one to three tree layers, one or two shrub layers, and an herbaceous stratum. This forest community is characterized by about 3000 vascular plant species, 30 percent of which are endemic to Mexico (CEC 1997).

## 15.2 Ecosystem Responses to N Deposition

Responses to N deposition in this ecoregion (Fenn et al. 1999, 2002a, 2002b) appear to be similar to those in the mixed conifer forest of the Mediterranean California ecoregion (Chapter 13). These effects include elevated foliar N concentrations, increased soil N cycling rates, increased N export in solution and via trace gas emissions, increased susceptibility to pests, changes in native species composition, enhanced establishment of exotic invasive species, and changes in mycorrhizal and epiphytic lichen communities. Although data on N deposition and effects are scarce for the U.S. portion of the Temperate Sierras ecoregion, under current simulated estimates (3 to 6 kg N $ha^{-1}$ $yr^{-1}$) of total N deposition (Fenn et al. 2003) and National Atmospheric Deposition Program (NADP) wet deposition data, we expect that the most likely effects occurring in some areas under current deposition levels are likely to be effects on lichen communities and possibly effects on herbaceous or grass understory communities.

## 15.3 Range of Responses Observed

Little research has been done on N deposition effects in forests of the Temperate Sierras ecoregion in Arizona and New Mexico. However, a combination of modeling and monitoring were used to estimate N deposition fluxes for forests northeast of Phoenix, Arizona, a rapidly expanding metropolis (Fenn et al. 2003). Estimated deposition fluxes to the forested area ranged from 13 to 23 kg $ha^{-1}$ $yr^{-1}$. Further measurements are needed to confirm these data and to determine the extent of the affected areas, but these estimates suggest that current levels of N deposition are likely to have significant impacts on these forests. The spatial extent of affected areas may be relatively limited.

Because we found no studies relating N inputs to ecosystem responses in the United States, we summarize research conducted in the Mexico City air basin (Fig. 15.1) within this ecoregion, as it may shed some light on the levels at which responses may be observed. Mexico City (Fig. 15.1) is located within the larger Basin of Mexico, sometimes called the valley of Mexico. The 9600 $km^2$ basin is defined by mountain ranges surrounding the valley floor. The Mexico City air basin refers to the ground-level atmosphere of the basin. In Mexican mountain pine (*Pinus hartwegii*) forests in the Mexico City air basin, the primary documented response to N deposition was nitrate ($NO_3^-$) leaching (Fenn et al. 1999, Fenn et al. 2002b). Foliar N was sometimes higher in the more polluted sites, but this result was inconsistent. Also, soil N levels did not always correspond with N deposition (Fenn et al. 1999, 2002b, 2006). The inconsistent response of foliar and soil N may be due to the inherent high soil N fertility of the volcanic soils prevalent in the Mexico City air basin (Fenn et al. 2002b, 2006). The role of N deposition in the historical and current deterioration of native lichen communities is unclear, as lichen community impacts in the Mexico City Air Basin began decades ago

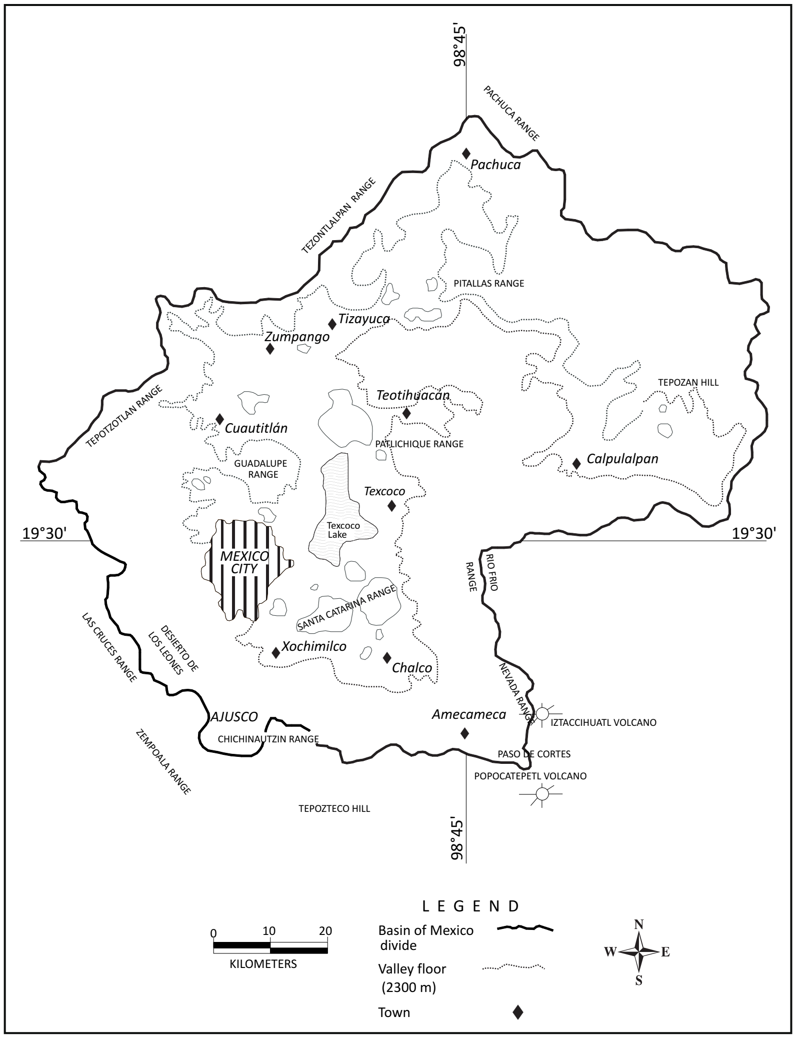


Figure 15.1—The Mexico City air basin, showing major mountain ranges, towns, and Mexico City. The high N deposition forested sites studied, Desierto de los Leones and Ajusco, are in the southwestern portion of the basin. The low deposition forest sites are in the southeastern portion of the basin within the Rio Frio and Nevada mountain ranges.

when sulfur (S) emissions and deposition were much greater than current levels and greater than nitrogen oxides ($NO_x$) emissions. However, it is likely that as S deposition levels have decreased in the Mexico City air basin in the past decade (Fenn et al. 2002a), the relative importance of N deposition on the remaining lichen communities has increased.

In the Mexico City air basin, $NO_3^-$ concentrations in stream and spring water were highly elevated in forested catchments south and southwest of Mexico City. Nitrogen deposition in throughfall at the highly N-saturated Desierto de los Leones National Park was 18.5 kg $ha^{-1}$ $yr^{-1}$ in 1996/1997 (Fenn et al. 1999). Nitrate concentrations in springs and streamwater at this site were frequently 20 to 90 µeq $L^{-1}$ (Fenn et al. 2002b), indicating that this site is strongly N saturated. Elevated $NO_3^-$ concentrations were also found at the Ajusco site, where N deposition is believed to be lower than at the Desierto site (Fenn et al. 2002a), but annual deposition fluxes have not been quantified. Because the Andisols in these study sites are typically high in N, even without elevated N deposition, no clear patterns of increased net N mineralization, nitrification, soil N or foliar N were observed (Fenn et al. 2002b). Nitrogen fertilization sometimes results in little plant growth response (Fenn et al. 2002b, 2006); preliminary evidence suggests a depressive growth effect from added N in the Mexico City air basin in some cases, possibly because added N induces greater phosphorus (P) deficiency (Fenn et al. 2006, 2002b). Widespread P deficiency, presumably because of the high P-fixation capacity of these Andisols (Shoji et al. 1993), and the high inherent N fertility in these study sites, may enhance the tendency of these catchments to leach $NO_3^-$ with increasing atmospheric N deposition.

At the Ajusco Mexican mountain pine site located south and downwind of Mexico City, an unknown soil factor was highly toxic to growth of river redgum (*Eucalyptus camaldulensis)* seedlings and the percentage of short roots colonized by putative symbiotic fungi in association with external mycelium was severely reduced (Fenn et al. 2006, Perea-Estrada et al. 2005). In the absence of fertilization, the percentage of eucalyptus roots that were ecto- and endomycorrhizal was consistently lower in high deposition sites. When eucalpytus seedlings were fertilized with P alone or a combination of N and P, a strong positive plant growth response occurred and the levels of root-associated fungi no longer differed between the polluted and clean sites (Perea-Estrada et al. 2005). Furthermore, at two of the three study sites in the Mexico City air basin, added N caused a significant reduction in the number of short roots of eucalyptus seedlings colonized by symbiotic fungi with external mycelium (Perea-Estrada et al. 2005). Further mechanistic studies are needed before firm conclusions can be reached regarding the role of N deposition in the strongly growth-depressive effects of soil from the Ajusco site. However, the negative effects of added N on symbiotic root fungi and the positive effects of added P on plant growth and root-associated fungi suggest that excess N may exacerbate P deficiency, particularly if symbiotic fungi important for P uptake are inhibited by high N levels in soil.

### 15.3.1 Lichens

Lichen responses to air pollutants in the Temperate Sierras ecoregion have not been studied, except in the Mexico City air basin. A series of studies contrasted fir and oak forests in Desierto de los Leones and Sierra de las Cruces (both south of Mexico City) to those of El Chico National Park, a relatively unpolluted site about 100 km to the northeast. Epiphytic lichen diversity at Desierto de los Leones is severely impoverished compared to El Chico National Park. Desierto de los Leones may have lost nearly 50 percent of its lichen species and lichen abundance is reduced by 60 percent, presumably as a result of the severe air pollution 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). Short-term lichen transplant experiments in Sierra de las Cruces, a montane region just south-southwest and downwind of the urbanized zone, showed 30 percent lower carbon fixation and 15 to 25 percent chlorophyll degradation compared to samples in El Chico National Park (Zambrano and Nash 2000, Zambrano et al. 1999). These results suggested that chronic air pollution (ozone ($O_3$), N- and S-containing compounds) is a major cause of lichen decline in forests surrounding the city, along with a variety of other anthropogenic disturbance factors (Zambrano et al. 2002). In the most recent study, Valencia-Islas et al. (2007) showed that the composition and location of phenolics in thalli of the sensitive parmotrema lichen (*Parmotrema stuppeum*) and tolerant Asahina's cartilage lichen (*Ramalina asahinae*) was consistent with their contrasting survival in forests stressed by $O_3$ (oxidative air pollution). Such responses illustrate potential confounding effects to lichen communities by codominant acidifying and oxidative pollutants in contrast to environments dominated solely by nutrient N.

## 15.4 Critical Loads Estimates

We estimate a N critical load for $NO_3^-$ leaching from pine stands in the Mexico City air basin of 15 kg ha$^{-1}$ yr$^{-1}$ (Table 15.1). This critical load is based on streamwater data from two regions of low N deposition east of Mexico City and two regions south and southwest of Mexico City (Fenn et al. 1999, 2002b). At the Desierto de los Leones site, where throughfall N deposition was 18.5 kg ha$^{-1}$ yr$^{-1}$ in 1996/1997, $NO_3^-$ concentrations in springs and streamwater were frequently 20 to 90 µeq L$^{-1}$ (Fenn et al. 2002b), indicating that this site is strongly N saturated and well above the critical load. Elevated $NO_3^-$ concentrations were also found at the Ajusco site, where N deposition is believed to be lower than at the Desierto site (Fenn et al. 2002a), but annual deposition fluxes have not been quantified. Uncertainty in this estimated critical load is largely due to an insufficient number of study sites spanning a range of deposition fluxes and because deposition data are from only 1 year for throughfall from one high and one low deposition site (Fenn et al. 1999). Also note that high levels of $NO_3^-$ leaching in the Mexico City air basin occur both during

**Table 15.1—Empirical critical loads of nutrient N for Temperate Sierras ecoregion forests in the Mexico City air basin[a]. Reliability rating: ## reliable; # fairly reliable; (#) expert judgment**

| Site | Critical Load for N deposition[b] *kg N ha$^{-1}$ yr$^{-1}$* | Reliability | Response | Comments | Study |
|---|---|---|---|---|---|
| New Mexico and Arizona forests | 4-7 | (#) | Epiphytic lichens | Assumes 60% eutroph threshold and 30-180 cm precipitation | Based on application of Geiser et al. 2010 model |
| Las Cruces and Chichinautzin Ranges S/SW of Mexico City | 15 | # | Elevated $NO_3^-$ in stream and spring waters | Data are from Mexican mountain pine sites in the Desierto de los Leones National Park and Ajusco | Fenn et al. 1999, 2002b |

[a] A variety of studies in the Mexico City air basin have demonstrated that air pollution is an important factor contributing to major changes in lichen communities, to the decline of sacred fir, and possible reductions in plant growth, root production and mycorrhizal symbionts. However, because of insufficient data and the co-occurrence of multiple pollutants, the role of N deposition is unclear and critical loads for N deposition cannot be determined without further study.

[b] Nitrogen deposition was measured as throughfall (Fenn et al. 1999).

the rainy summer growing season and during the winter dry season (Fenn et al. 1999, 2002b).

Although lichen species have disappeared and lichen communities have been severely altered in the Mexico City air basin, it is difficult to estimate a N critical load for lichen effects because of the lack of historical pollution exposure and deposition data and the potential effects of multiple pollutants. Emissions of S, N and heavy metals have been historically high; $O_3$ exposure until recently has been among the highest in the world (Fenn et al. 2002a). Geiser et al. (2010) developed a method for relating N deposition to lichen community response. They select a threshold for the shift in the community composition from oligotroph- to eutroph-dominated. This approach is described in detail in Chapter 4. For U.S. portions of the ecoregion, our best estimate of N critical loads for epiphytic lichens is 4 to 7 kg ha$^{-1}$ yr$^{-1}$. This estimate assumes a response threshold of about 60 percent eutrophs, mean annual precipitation of 30 to 180 cm, and general applicability of the Geiser et al. (2010) model to temperate forests.

## 15.5 Comparison to Critical Loads for Other Regions

There are no relevant lichen data from Europe for comparison with this ecoregion; critical loads developed for North American deserts or Northwestern Forested Mountains may have some applications. The lichen data from Mexico, summarized above, demonstrate a response in lichen community structure, but no conclusions can be made regarding a N critical load because trace metals, $O_3$, and S and N deposition were all at high levels during the decades of rapid industrial and population growth in the Mexico City air basin (Fenn et al. 2002a).

The estimated critical load for $NO_3^-$ leaching in the Mexico City air basin (15 kg ha$^{-1}$ yr$^{-1}$) is lower than that of mixed conifer forests in Mediterranean California. This difference may be due to the high N fertility and low P availability of the Andisols common in the Mexico City air basin. Because N is not strongly limiting in the Mexico City air basin, presumably, less added N is needed to induce N losses from the ecosystem. The critical load for $NO_3^-$ leaching in the Mexico City air basin is uncertain because it is based on only 1 year of deposition data, but assuming the deposition data approximate longer term inputs, we conclude that the Mexican forests are more prone to $NO_3^-$ leaching than the forests of California (Fenn et al. 2008). In the Mexico City air basin with N deposition of 18.5 kg N ha$^{-1}$ yr$^{-1}$, $NO_3^-$ concentrations in stream and springwater were commonly at high levels (20 to 90 µeq L$^{-1}$), even during the growing season (Fenn et al. 2002b). By comparison, a throughfall deposition of 17 kg N ha$^{-1}$ yr$^{-1}$ was determined as the critical load for incipient increases in $NO_3^-$ leaching (peak values above 14 µeq L$^{-1}$) in California mixed conifer forests during the winter dormant wet season (Fenn et al. 2008).

The critical load for incipient $NO_3^-$ leaching in the Mexico City air basin cannot be determined without streamwater sampling and N deposition measurements from sites at intermediate levels of N deposition. Nitrogen deposition levels leading to $NO_3^-$ leaching in the Mexico City air basin appear to be within the range described from surveys of temperate forest catchments in Europe and North America (Dise and Wright 1995, Gundersen et al. 2006, MacDonald et al. 2002, Stoddard et al. 2001).

## 15.6 Future Research Directions and Gaps in Knowledge

Future research could be directed at mountain forests outside of Phoenix, Arizona, to determine actual deposition fluxes and ecosystem impacts at different deposition levels. Follow up studies are needed in forests surrounding Mexico City to better understand nutrient cycling responses to N deposition of forests growing on N-rich but P-limited Andisols. Preliminary results indicate the value of developing critical load for N effects on mycorrhizal fungi, root production, and lichen communities, and of further refinement of the estimated critical load for $NO_3^-$ leaching. Emissions of nitrogenous trace gases from soil in response to N deposition should also be investigated, particularly considering the high N concentrations in soil and the role of these gases in smog formation and climate change. The possible role of N deposition in combination with S deposition and elevated $O_3$ exposure merits further investigation as contributing or causal factors in the severe decline of sacred fir (*Abies religiosa* ) in the more polluted regions of the Desierto de los Leones National Park (Alvarado-Rosales and Hernández-Tejeda 2002, Fenn et al. 2002a).

Forest Inventory and Analysis lichen community surveys have been completed for the parts of this ecoregion in Arizona and New Mexico. Analysis of these lichen data along depositional N gradients estimated by NADP and the Environmental Protection Agency's Models-3/Community Multiscale Air Quality (CMAQ) simulation model (Tonnesen et al. 2007) could refine the lichen-based critical load estimate for this ecoregion. More research is also needed to better understand the separate effects of acidifying and oxidizing versus fertilizing N-containing/N-derived pollutants on vegetation and lichens.

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