



Air Quality Related Values (AQRVs) for Mediterranean Coast Network (MEDN) Parks

*Effects from Ozone; Visibility Reducing Particles; and
Atmospheric Deposition of Acids, Nutrients and Toxics*

Natural Resource Report NPS/MEDN/NRR—2016/1162



ON THE COVER

Photograph of air quality related values within various National Park units. Wildflowers, clear views, aquatic species, and lichens may all be threatened by air pollution.

Photographs courtesy of the National Park Service

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Summary

This report describes the Air Quality Related Values (AQRVs) of the Mediterranean Coast Network (MEDN). Networks have been defined by the National Park Service (NPS) based on geography and shared natural resource characteristics. AQRVs are those resources sensitive to air quality and include streams, lakes, soils, vegetation, fish and wildlife, and visibility. The MEDN parks that are included in the NPS Inventory and Monitoring (I&M) Program, and discussed in this report, are Channel Islands National Park (CHIS), Santa Monica Mountains National Recreation Area (SAMO), and Cabrillo National Monument (CABR).

Sullivan et al. (2011a, 2011b) and Kohut (2007a) conducted risk assessments for acidification, eutrophication, and ozone (O_3) for the MEDN parks; their results are summarized in this report. This report also describes air pollutant emissions and air quality, and their effects on AQRVs. The primary pollutants likely to affect AQRVs include nitrogen (N) and sulfur (S) compounds (nitrate [NO_3^-], ammonium [NH_4^+], and sulfate [SO_4^{2-}]); ground-level O_3 ; haze-causing particles; and airborne toxics. Background for this section can be found in “Air Quality Related Values (AQRVs) in National Parks. Effects from Ozone; Visibility Reducing Particles; and Atmospheric Deposition of Acids, Nutrients and Toxics” (Sullivan 2016).

Regional emissions from vehicles, urban areas, agricultural sources, marine vessels and fires affect air quality in MEDN parks. California is the most populous state in the nation; many emissions are concentrated around major population centers including Los Angeles, with a population of over 17 million people, and San Diego with over 3 million people. SAMO is the largest urban national park in the United States.

Atmospheric S and N pollutants can cause acidification of streams, lakes, and soils. Sulfur emissions and deposition are relatively low in California compared to the eastern United States. In contrast, N emissions and deposition are high, derived mainly from motor vehicles, agriculture, fires, and other combustion sources. Wildland ecosystems in the vicinity of Los Angeles receive the highest N deposition in the United States (Fenn et al. 1998, Fenn et al. 2002, Fenn et al. 2003b). Although N emissions and deposition have decreased in southern California in recent decades, they remain high (Fenn et al. 2003b).

N deposition can also cause undesirable enrichment of natural ecosystems, leading to changes in plant species diversity and soil nutrient cycling. Based on model results, terrestrial vegetation in portions of SAMO is exposed to substantial anthropogenic N deposition of around 15-20 kg N/ha/yr (Sullivan et al. 2011a). Modeled N deposition to CHIS and CABR is relatively low (2-3 kg/ha/yr), due to prevailing westerly winds at their island and coastline locations. The predominant vegetation types in the MEDN that are sensitive (to nutrient N enrichment) are arid and semi-arid vegetation. Much of the data regarding effects of N deposition on arid land vegetation are from the coastal sage scrub (CSS) communities of southern California, downwind of the Los Angeles Basin, where dry N deposition is very high. Native shrub and forb seedlings in the CSS community are unable to compete with dense stands of exotic grasses under high N loading, and thus are gradually replaced by the grasses, especially following disturbance such as fire. CABR was ranked Very High and CHIS

and SAMO were ranked High in Ecosystem Sensitivity to nutrient N enrichment in a coarse screening study by Sullivan et al. (2011a). This is because they have substantial coverage of vegetation types expected to be highly sensitive to nutrient N enrichment effects. Fenn et al. (1996) concluded that symptoms of N saturation were evident in chaparral sites in southern California that received atmospheric N deposition of 20 to 25 kg N/ha/yr or higher, levels higher than those found in the MEDN parks.

In the San Bernardino Mountains near SAMO, up to 50% of lichen species that occurred in the region in the early 1900s have disappeared, with a disproportionate number of locally extinct species being epiphytic cyanolichens (Fenn et al. 2003a). Of the epiphytic lichen species known to occur prior to the early 1900s on coast live oak (*Quercus agrifolia*) in the chaparral and oak vegetation communities of the Los Angeles Basin, about half are no longer present (Fenn et al. 2003a). This loss of lichen diversity has been attributed to both O₃ and N inputs. In the Los Angeles Basin, Fenn et al. (2008) found a distinct shift in lichen community composition from N-sensitive to more tolerant lichen species at N deposition at or above 3.1 kg N/ha/yr. Complete elimination of acidophytes (i.e., species that tolerate high acidity) from the lichen community was predicted at an N deposition level of about 10.2 kg N/ha/yr. These findings suggest that lichen communities in parts of SAMO may also have been significantly affected by current levels of N deposition.

Deposition of N has contributed to the displacement of native herbaceous species by non-native invasive annual grasses. Fenn et al. (2010) reported the results of a field study in southern California along a gradient of N deposition ranging from about 8.7 to 19 kg N/ha/yr. The coverage of non-native grasses increased with atmospheric N deposition and soil N content. Native forb richness decreased across the N deposition gradient from 67 to 16 species per site within the deposition range from 9 to 11 kg N/ha/yr (Fenn et al. 2010). Based on these data, Fenn et al. (2011a) concluded that the critical N load was about 10 kg N/ha/yr to protect plant species richness¹. The deposition level at which NO₃⁻ leaching becomes pronounced in this region (stream NO₃⁻ concentration > 14 µeq/L) is about 17 kg N/ha/yr. Thus, current levels of N deposition may be affecting plant communities in SAMO.

Ozone pollution can harm human health, reduce plant growth, and cause visible injury to foliage. Levels of O₃ are high in many portions of southern California, and all three MEDN parks are located in areas designated nonattainment for the national O₃ standard because O₃ levels are considered unhealthy for both humans and plants at times. Elevated O₃ exposure and N deposition, in combination, have contributed to changes in nutrient cycling in mixed conifer forests of the Los Angeles Basin. Carbon storage has increased in woody above-ground biomass; foliar turnover has increased; and forest litter production and accumulation have increased (Fenn et al. 2011a). The susceptibility of trees to disturbance, including insect infestation, drought, and disease, has increased, resulting in increased tree mortality. One result of exposure to elevated levels of both O₃ and N is increased production and accumulation of litter on the forest floor (Fenn et al. 2003c, 2005, 2011a).

¹ Richness reflects the number of species present; it is a component of diversity, along with evenness of species occurrences.

This latter effect inhibits establishment of many native understory species and increases susceptibility to stand-replacing wildfire.

Growing season O₃ exposure varies considerably throughout the MEDN. At CABR and CHIS, levels of O₃ exposure are generally relatively low, despite the fact that there are short periods when levels exceed the national standard. In addition, there are frequent or extended periods of low soil moisture, which constrains plant uptake of O₃. These features resulted in the assignment by Kohut (2007b) of a low risk of foliar injury in response to O₃ exposure at these two parks. Both of these parks are outside the major zone of influence of N emissions generated in the Los Angeles Basin. At SAMO, however, high O₃ exposure creates additional potential for foliar injury and the risk was judged by Kohut (2007b) to be High. Nevertheless, in a given year, dry soil conditions can reduce the likelihood of foliar injury in this park, and the level of risk varies from year to year.

Particulate pollution can cause haze, reducing visibility. Although it is not monitored in or near the MEDN parks, haze levels are generally high throughout much of southern California. Sea salt contributes to natural haze at CHIS and CABR, and to some extent at SAMO, but human-caused pollution undoubtedly also contributes substantially to haze in these parks.

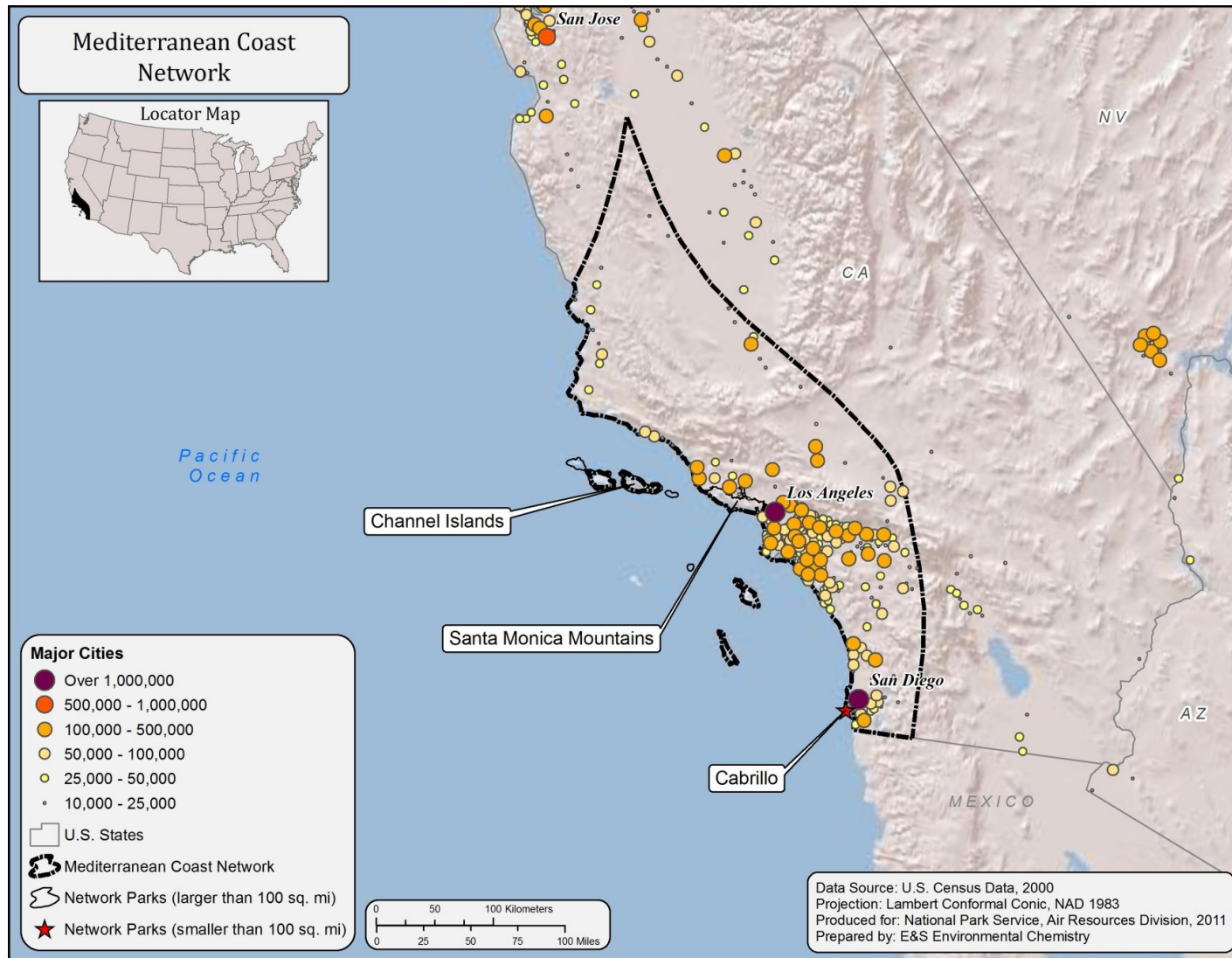
Airborne toxics, including mercury (Hg) and other heavy metals, can accumulate in food webs, reaching toxic levels in top predators. Although data on air toxics are generally not available for the MEDN parks, elevated levels of both banned and current use pesticides and industrial contaminants have been found in Sequoia and Kings Canyon National Parks to the north of the MEDN.

Background

California is the most populous state in the nation, and vehicular emissions and other nonpoint and point sources of air pollution are concentrated around major population centers. The two regions of California that show the most severe air quality problems are the south coast, in and around the MEDN, and the San Joaquin Valley to the northeast (Suarez-Murias et al. 2009).

There are three parks in the Mediterranean Network (MEDN): Channel Islands National Park (CHIS), Santa Monica Mountains National Recreation Area (SAMO), and Cabrillo National Monument (CABR). Two of the parks, CHIS and SAMO, are larger than 100 square miles. Larger parks generally have more available data with which to evaluate air pollution sensitivities and effects. In addition, the larger parks generally contain more extensive resources in need of protection against the adverse impacts of air pollution. Urban development in and around Los Angeles and south to San Diego is substantial. Map 1 shows the network boundary, the location of each park, and population centers with greater than 10,000 people around the network.

To the east of the MEDN parks is the Central Valley, one of the most intensively cultivated agricultural regions of the United States. Much of the coastal sage scrub vegetation that occurred in southern California in pre-settlement times has been replaced by agriculture and urban development.



Map 1. The MEDN boundary, locations of parks, and population centers greater than 10,000 people around the MEDN.

Atmospheric Emissions and Deposition

Sulfur dioxide (SO₂) emissions have been low in California since the early 1980s and do not appear to pose a significant concern for downwind parks (Sullivan et al. 2001). County-level sulfur (S) emissions in 2002 reported by Sullivan et al. (2011b) were mostly less than 5 tons per square mile per year (tons/mi²/yr) of SO₂ within the MEDN. There were no SO₂ point source emissions of any magnitude; all individual sources were less than 5,000 tons/yr. Both SO₂ emissions and atmospheric sulfate (SO₄²⁻) concentrations at Interagency Monitoring of Protected Visual Environments (IMPROVE) Network visibility monitoring sites decreased during the 1990s throughout much of the western United States (Malm et al. 2002). Decreases in emissions were largest (-37%) in California. Nevertheless, emissions of SO₄²⁻ from ocean vessels constitute a potentially important contributor to visibility impairment² at coastal California parks.

The major sources of nitrogen (N) emissions in the western United States are transportation, agriculture, power plants, and industry. The highest emissions source areas of nitrogen oxides and reactive organic gasses (precursors of ozone [O₃]) in California are the South Coast, San Francisco Bay Area, and San Joaquin Valley air basins (Sullivan et al. 2001). In general, annual county N emissions were between 1 and 20 tons/mi²/yr, except in and around the Los Angeles Basin, where they were in the range of 20 to 50 tons/mi²/yr (Sullivan et al. 2011a). There were few N point sources of any magnitude within this network. Nonpoint source emissions, especially from motor vehicles, constituted the vast majority of the N emissions, with other significant sources of N from fossil-fueled power plants. Although N emissions in the MEDN region are very high at some locations, they have decreased during recent decades through implementation of aggressive air pollution emissions reduction programs.

County-level emissions near MEDN, based on data from the EPA's National Emissions Inventory (NEI) during a recent time period (2011), are depicted in Maps 2 through 4 for SO₂, oxidized N (NO_x), and reduced N (NH₃), respectively. Throughout most of the network region, SO₂ emissions were less than 1 ton/mi²/yr, but they were higher (1-5 tons/mi²/yr) near SAMO (Map 2). Spatial patterns in NO_x emissions showed substantially higher values, at some counties more than 25 tons/mi²/yr (Map 3). Emissions of NH₃ near MEDN parks, which is largely of agricultural origin, were in the range of 0.5-8 tons/mi²/yr (Map 4).

Asian sources may also contribute some amount of N deposition to the West Coast. Emissions of N in Asia are higher than in North America (Fenn et al. 2003b) and have increased dramatically in recent decades (Akimoto and Narita 1994). Emissions of N can be transported across the Pacific Ocean from Asia as peroxyacetyl nitrate (PAN), which has a lifetime in the atmosphere of days to weeks. Because of the comparatively short lifetime of NO_x in the atmosphere (generally less than 24 hr), conversion of NO_x to a longer-lived form (such as PAN) is necessary for long-range transport of N. Once in the troposphere, PAN can be transported long distances over a period of many days and

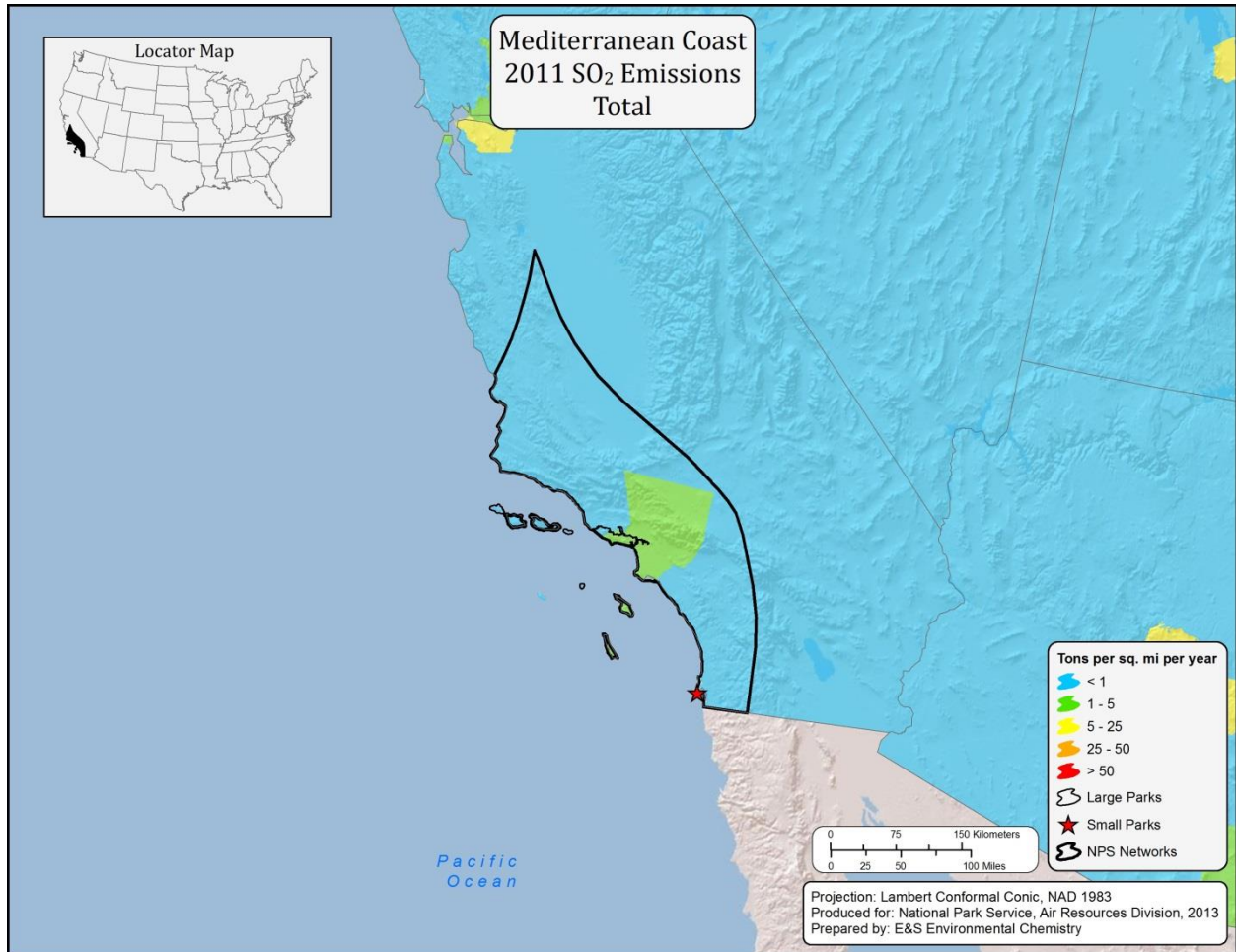
² *Visibility impairment* means any humanly perceptible change in visibility (light extinction, visual range, contrast, coloration) from that which would have existed under natural conditions.

then moved back into the boundary layer, where it regenerates NO_x and nitric acid (Kotchenruther et al. 2001). Long-range transport of PAN from Asia to North America is well documented (Jaffe et al. 1999). Thus, it is likely that NO_x emissions in Asia contribute some, albeit an unknown amount of, N deposition to the western United States, including to the MEDN. If Asian NO_x emissions continue to increase as predicted (cf., Klimont et al. 2001, Streets and Waldhoff 2000), trans-Pacific sources of N may become more important to west coast locations of the United States in the future.

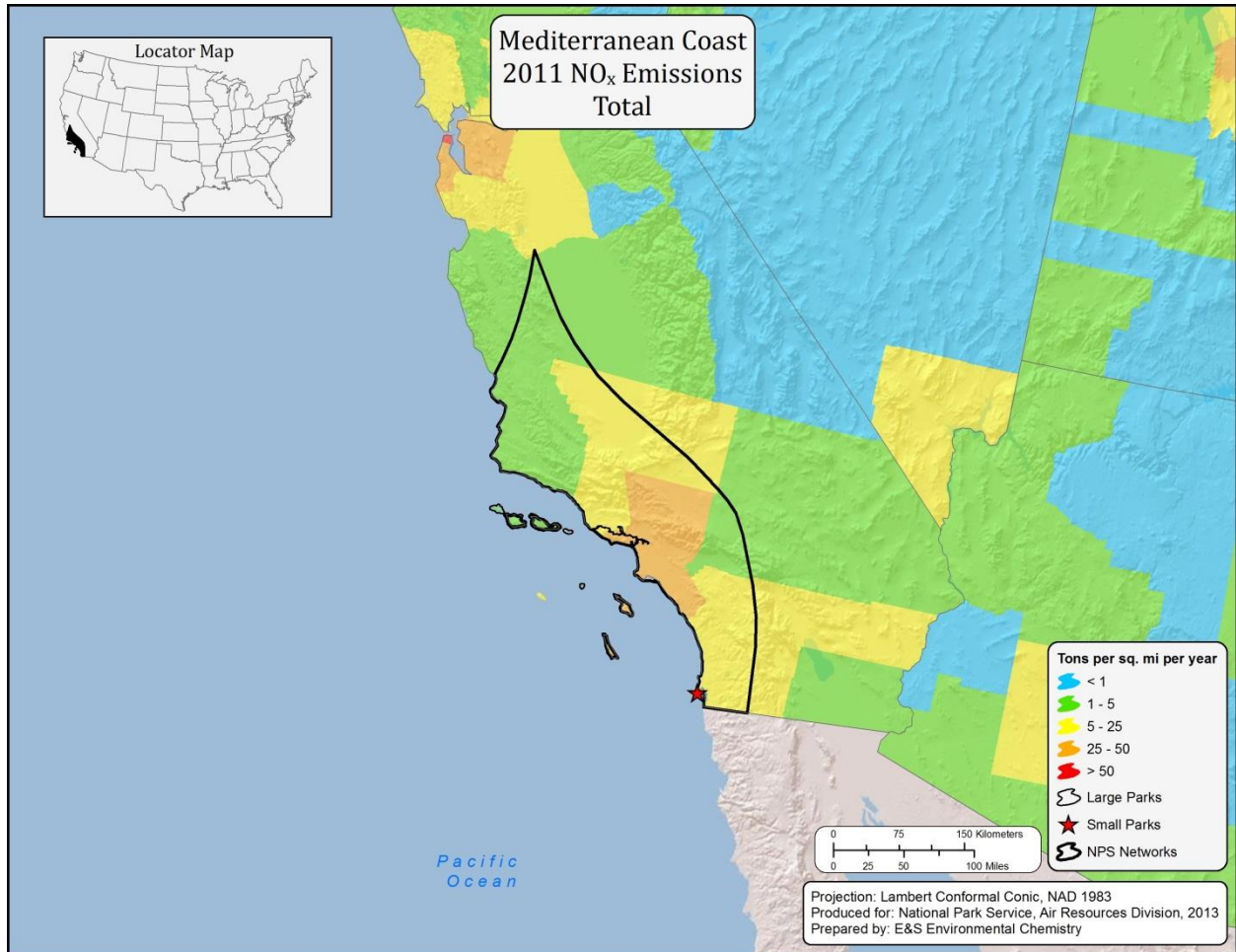
Total modeled S deposition within the network region ranged from less than 2 kg S/ha/yr to more than 15 kg S/ha/yr in 2002 (Sullivan et al. 2011b). Most areas within the network region had S deposition values less than 5 kg S/ha/yr. Total modeled N deposition within the network region ranged from less than 2 kg N/ha/yr at some of the coastal areas in the northern part of the region to as high as 15 to 30 kg N/ha/yr in the Los Angeles Basin in 2002. Because of the dry climate, most deposition is in dry form (particles or gases). Precipitation is low and wet N deposition is low in the region, in the range of 1 to 3 kg N/ha/yr in CABR and CHIS. Dry N deposition is highly variable, and ranges up to nearly 20 kg N/ha/yr in parts of SAMO and up to 30 kg N/ha/yr or more in proximity to major urban NO_x emissions source areas (Bytnerowicz and Fenn 1996, Fenn and Bytnerowicz 1997, Takemoto et al. 2001).

Recently, Schwede and Lear (2014) documented a hybrid approach developed by the National Atmospheric Deposition Program (NADP) Total Deposition (TDEP) Science Committee for estimating total N and S deposition. This approach combined monitoring and modeling data. Modeling was accomplished using the Community Multiscale Air Quality (CMAQ) model (Byun and Schere 2006). Priority was given to measured data near the locations of the monitors and to modeled data where monitoring data were not available. In addition, CMAQ data were used for N species that are not routinely measured in the monitoring programs: PAN, N_2O_5 , NO, NO_2 , HONO, and organic NO_3 . The total deposition estimates are considered to be dynamic, with updates planned as new information becomes available. TDEP data reported here were developed in late 2013 and are designated version 2013.02.

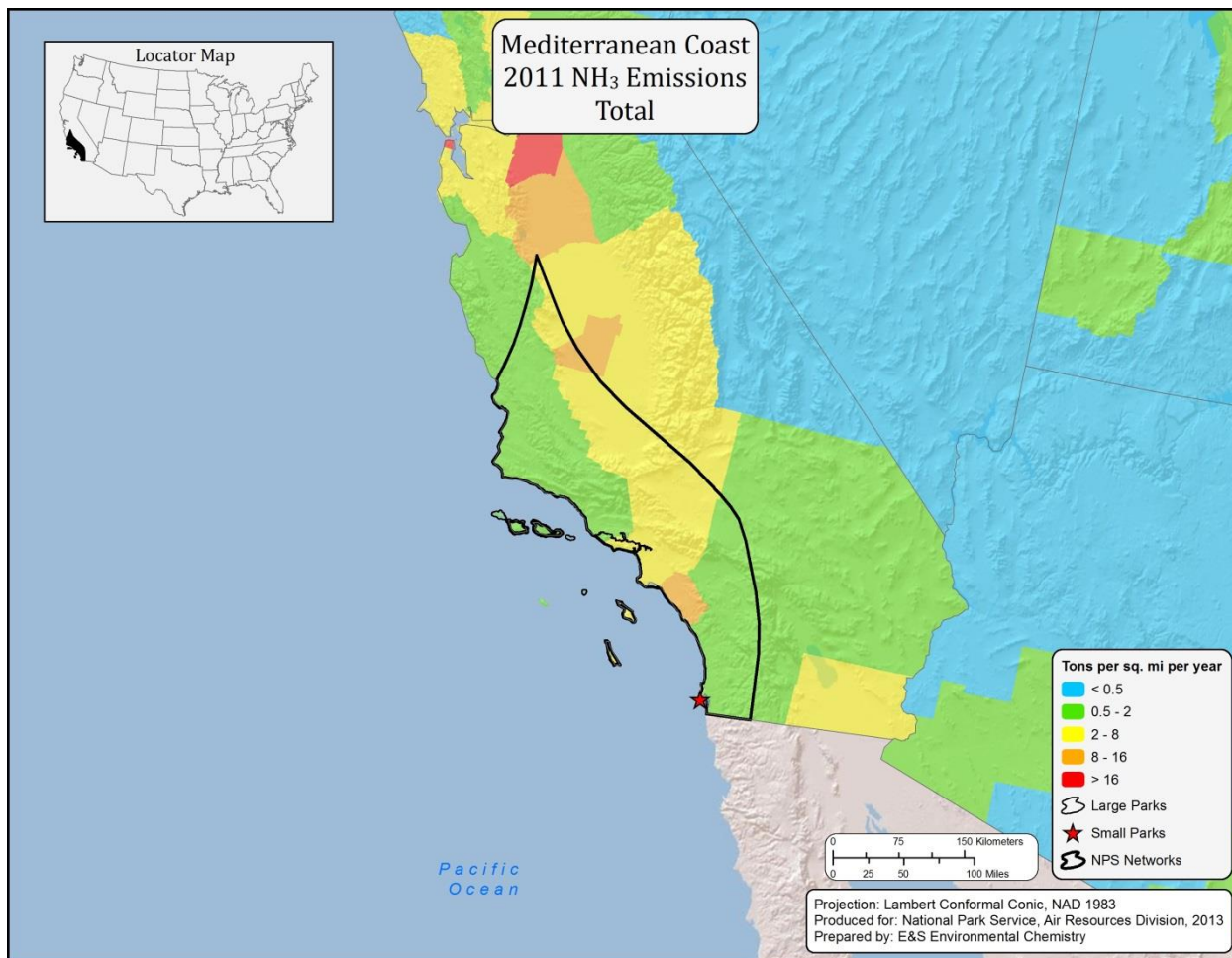
Atmospheric S deposition levels have increased substantially at CABR and CHIS, but decreased at SAMO since 2001, based on TDEP estimates (Table 1). Estimated total N deposition over that same time period remained relatively unchanged at all three parks. Oxidized N deposition decreased at CABR and SAMO. Reduced N deposition (NH_4^+ , NH_3) increased at all of the parks in the network since the monitoring period 2000-2002. In CABO and SAMO, reduced N deposition more than doubled. Total S deposition in and around MEDN for the period 2010-2012 was relatively low (< 5 kg S/ha/yr) at park locations (Map 5). Oxidized inorganic N deposition for the period 2010-2012 was in the range of 5-10 kg N/ha/yr throughout most of the park lands within MEDN (Map 6). Most areas received less than 10 kg N/ha/yr of reduced inorganic N from atmospheric deposition during this same period (Map 7). Total N deposition was > 10 kg N/ha/yr at many park locations (Map 8).



Map 2. Total SO₂ emissions, by county, near the MEDN for the year 2011. Data from EPA's National Emissions Inventory.



Map 3. Total NO_x emissions, by county, near the MEDN for the year 2011. Data from EPA's National Emissions Inventory.



Map 4. Total NH₃ emissions, by county, near the MEDN for the year 2011. Data from EPA's National Emissions Inventory.

Table 1. Average changes in S and N deposition between 2001 and 2011 across park grid cells at MEDN parks. Deposition estimates were determined by the Total Deposition Project, TDEP, based on three-year averages centered on 2001 and 2011 for all ~4 km grid cells in each park. The minimum, maximum, and range of 2011 S and N deposition within each park are also shown.

Park Code	Park Name	Parameter	2001 Average (kg/ha/yr)	2011 Average (kg/ha/yr)	Absolute Change (kg/ha/yr)	Percent Change	2011 Minimum (kg/ha/yr)	2011 Maximum (kg/ha/yr)	2011 Range (kg/ha/yr)
CABR	Cabrillo	Total S	1.20	3.27	2.06	171.3%	3.27	3.27	0.00
		Total N	13.26	12.02	-1.24	-9.4%	12.02	12.02	0.00
		Oxidized N	11.06	6.77	-4.29	-38.9%	6.77	6.77	0.00
		Reduced N	2.20	5.26	3.06	139.4%	5.26	5.26	0.00
CHIS	Channel Islands	Total S	1.72	2.47	0.75	49.3%	1.03	3.38	2.35
		Total N	3.72	3.91	0.20	5.0%	2.29	6.64	4.35
		Oxidized N	2.61	2.69	0.08	2.8%	1.48	4.73	3.25
		Reduced N	1.11	1.23	0.12	10.5%	0.81	1.90	1.10
SAMO	Santa Monica Mountains	Total S	1.58	1.21	-0.37	-24.0%	0.87	2.57	1.70
		Total N	15.51	11.16	-4.34	-4.2%	8.42	21.81	13.39
		Oxidized N	12.64	5.67	-6.96	-36.8%	4.23	10.63	6.39
		Reduced N	2.87	5.49	2.62	111.6%	3.32	12.48	9.16

Acidification

The network rankings for acid Pollutant Exposure, Ecosystem Sensitivity to acidification, and Park Protection developed by Sullivan et al. (2011a) in a coarse screening assessment yielded an overall network acidification Summary Risk ranking for the MEDN that was moderate relative to other Inventory and Monitoring (I&M) networks. The overall level of concern for acidification effects on I&M parks within this network was judged to be Moderate (Sullivan et al. 2011a, Sullivan et al. 2011b).

SAMO ranked in the second highest quintile among I&M parks in acid Pollutant Exposure, whereas CABR and CHIS were ranked Moderate, in the middle quintile, for this theme (Sullivan et al. 2011b; Table 2). The two larger parks were ranked in the second highest quintile (High) in Ecosystem Sensitivity to acidification, having substantial numbers of low-order streams on moderately steep terrain that might be expected based on topography to be at least somewhat sensitive to acidification effects. The smaller park (CABR) was ranked in the second lowest quintile in Ecosystem Sensitivity to acidification (Table 2). While rankings are an indication of risk, park-specific data, particularly data on ecosystem sensitivity, are needed to fully evaluate risk from acidification.

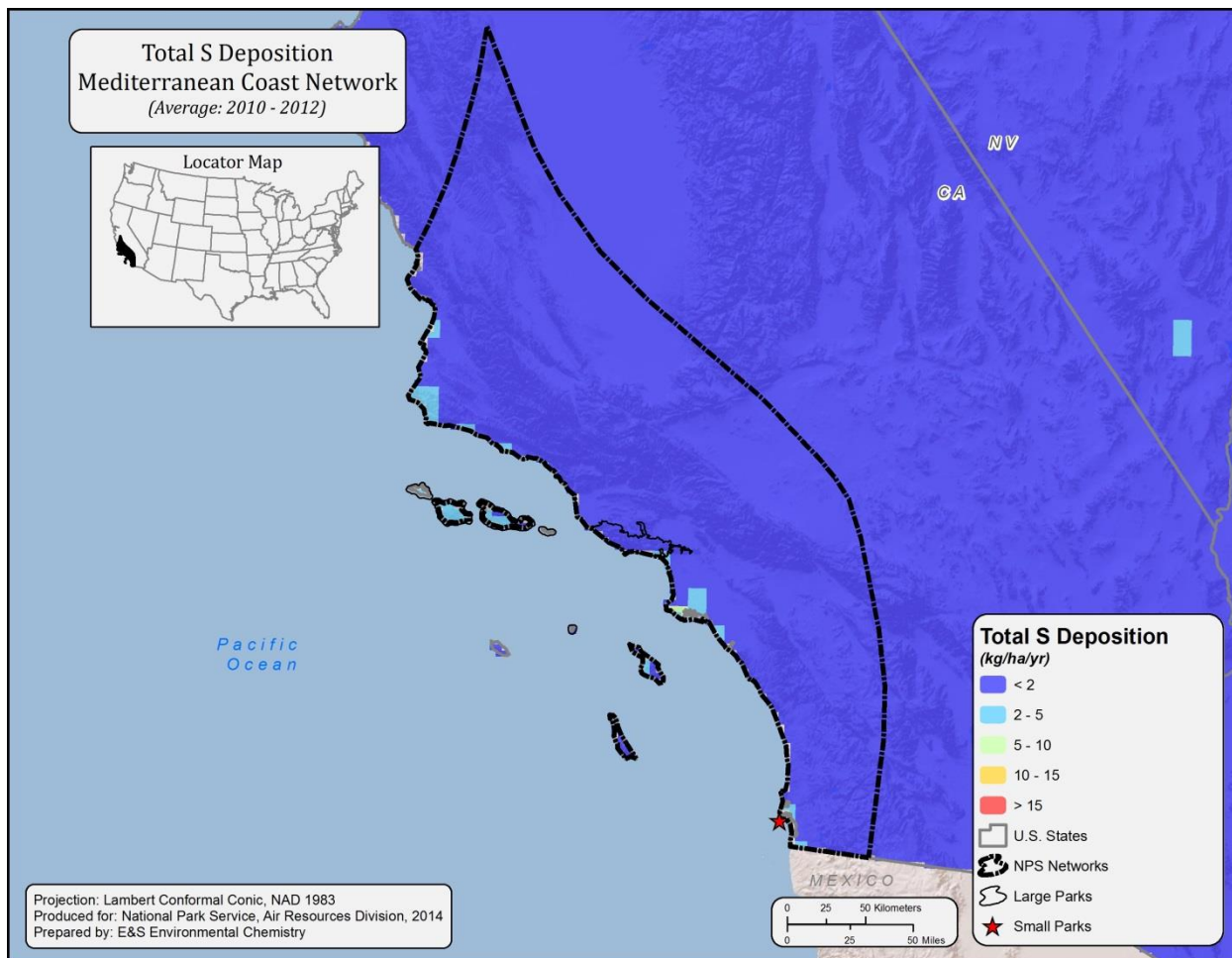
Soil acidification from atmospheric N deposition has been documented in areas of the San Bernardino Mountains that receive very high levels of N deposition (Breiner et al. 2007, Wood et al. 2007). Soils under chaparral in some portions of the San Gabriel Mountains have also undergone substantial acidification (Fenn et al. 2011a, Wood et al. 2007). Nevertheless, there are no data suggesting soil acidification in any of the MEDN parks. Similarly, there are no data suggesting that aquatic acidification a concern in the MEDN. In view of the scarcity of precipitation and high evapotranspiration in this network, such effects are considered unlikely.

Table 2. Estimated I&M park rankings¹ according to risk of acidification impacts on sensitive receptors (Source: Sullivan et al. 2011b)

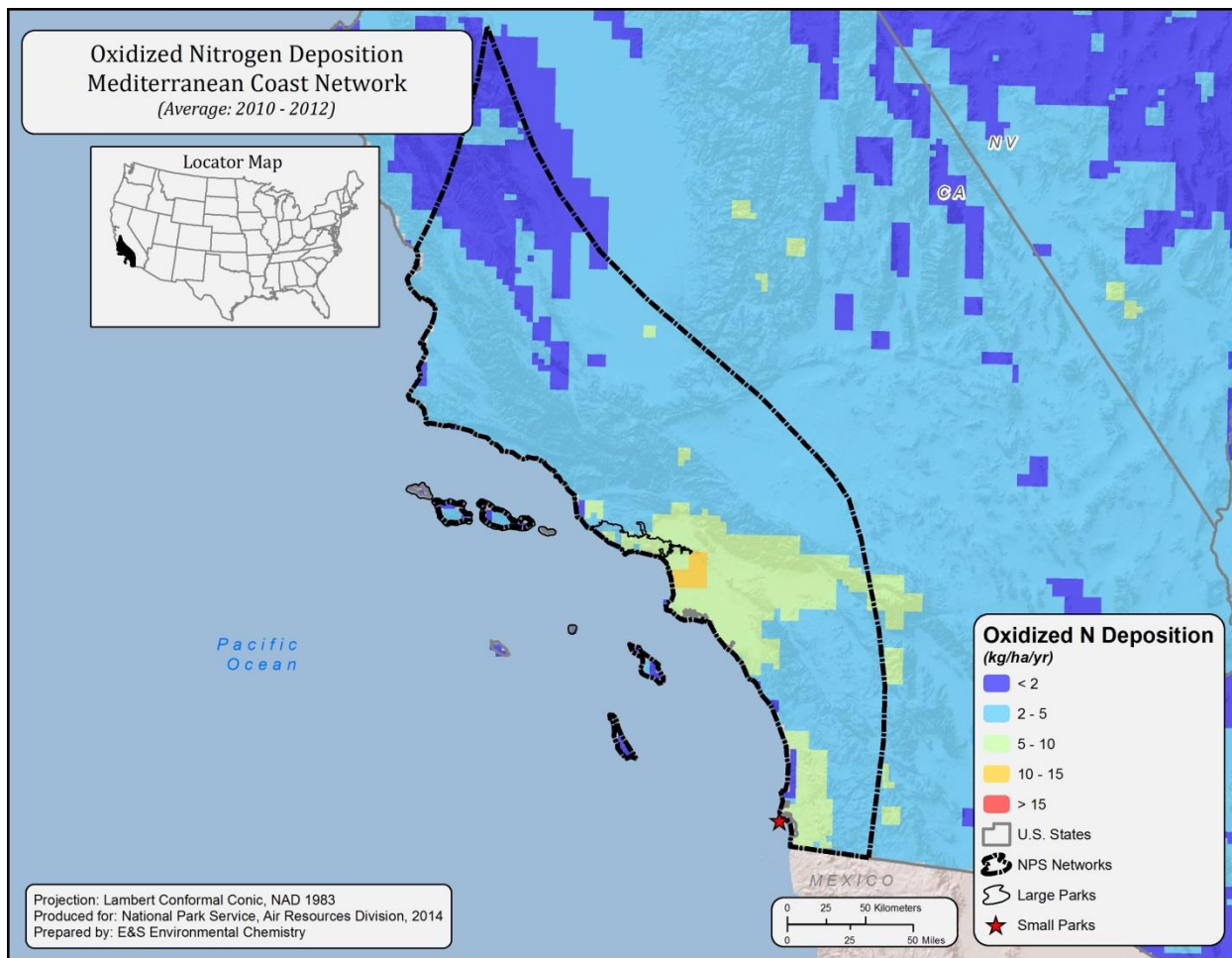
Park Name ²	Park Code	Estimated Acid Pollutant Exposure	Estimated Ecosystem Sensitivity to Acidification
Cabrillo	CABR	Moderate	Low
<i>Channel Islands</i>	CHIS	Moderate	High
<i>Santa Monica Mountains</i>	SAMO	High	High

¹ Relative park rankings are designated according to quintile ranking, among all I&M Parks, from the lowest quintile (Very Low risk) to the highest quintile (Very High risk).

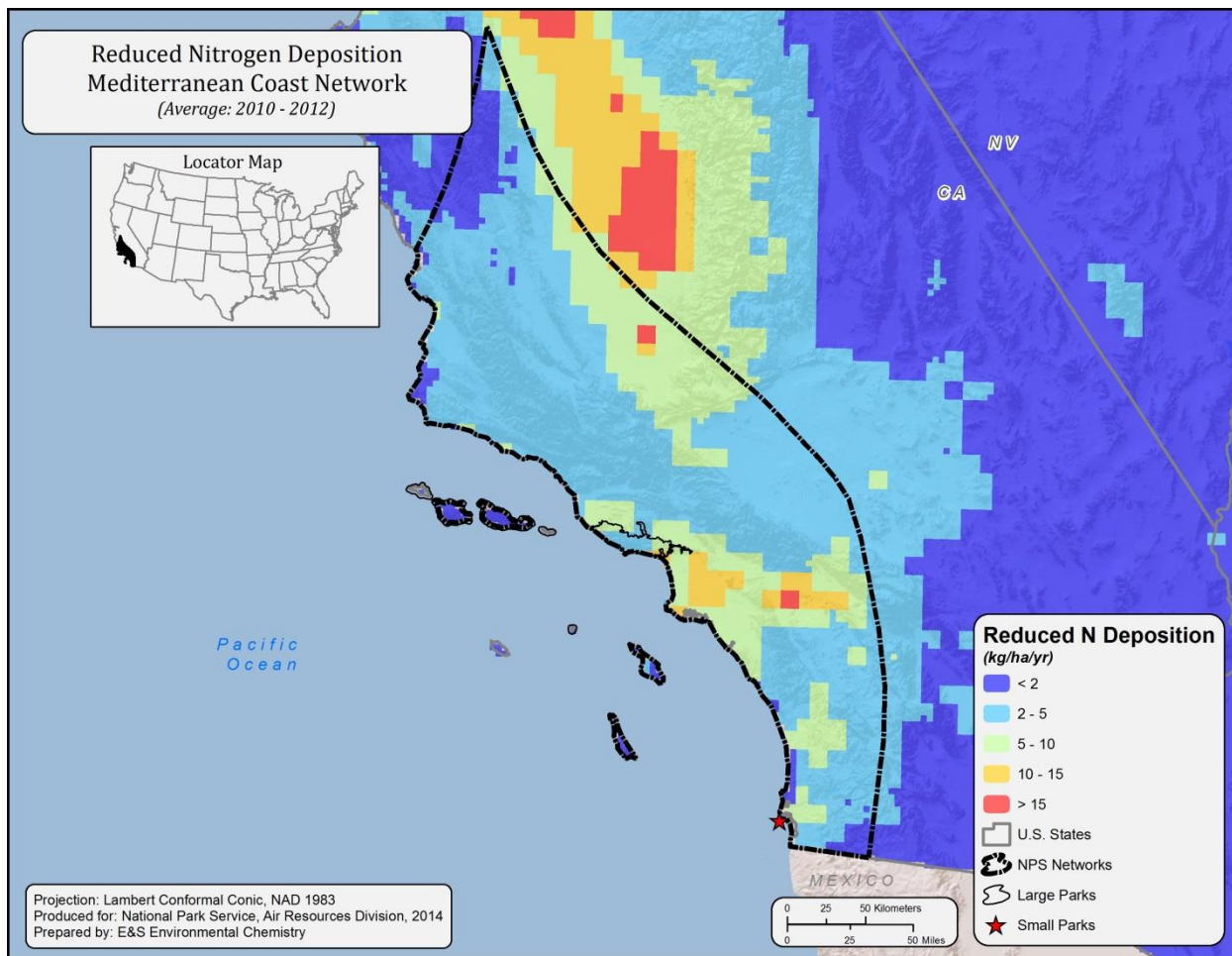
² Park names are printed in bold italic for parks larger than 100 mi².



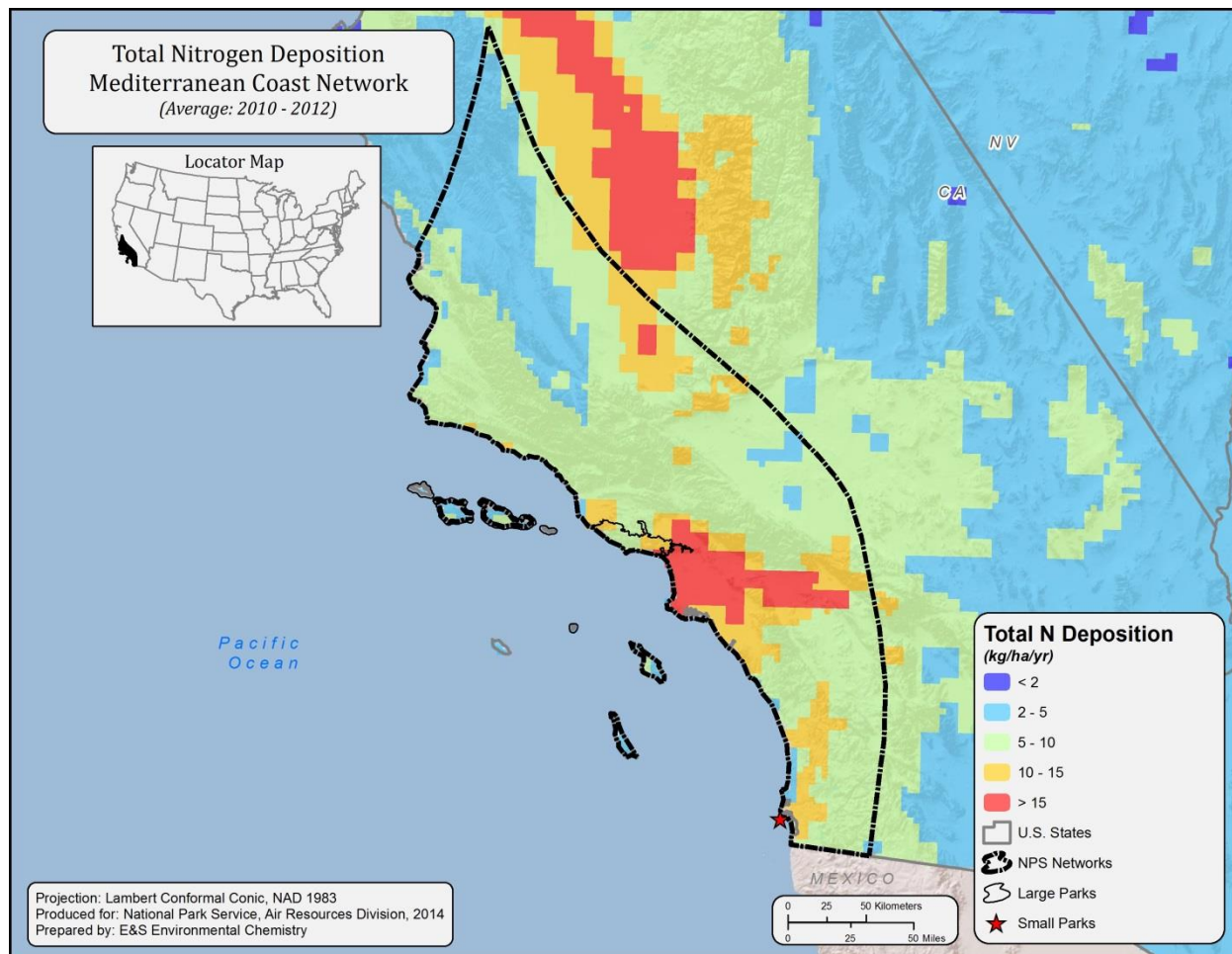
Map 5. Total S deposition for the three-year period centered on 2011 in and around MEDN. (Source: Schwede and Lear 2014)



Map 6. Total oxidized inorganic N deposition for the three-year period centered on 2011 in and around MEDN. (Source: Schwede and Lear 2014)



Map 7. Reduced inorganic N deposition for the three-year period centered on 2011 in and around MEDN. (Source: Schwede and Lear 2014)



Map 8. Total N deposition for the three-year period centered on 2011 in and around MEDN. (Source: Schwede and Lear 2014)

Nutrient Nitrogen Enrichment

The network rankings for nutrient N Pollutant Exposure, Ecosystem Sensitivity to nutrient N enrichment, and Park Protection were judged by Sullivan et al. (2011a) in a coarse screening assessment of eutrophication to yield an overall nutrient enrichment network Summary Risk ranking for MEDN that was at the top of the middle quintile among all networks. The overall level of concern for nutrient N enrichment effects on I&M parks within this network was judged to be Moderate to High (Sullivan et al. 2011a).

SAMO ranked in the highest quintile (Very High) among I&M parks in nutrient N Pollutant Exposure due to its proximity to Los Angeles, whereas CABR ranked High (the second highest quintile) for this theme (Sullivan et al. 2011a; Table 3). CHIS, located off-shore, receives less N pollutant exposure, and was ranked only Moderate. All three parks were ranked High or Very High in Ecosystem Sensitivity to nutrient N enrichment, having substantial coverage of vegetation types expected to be highly sensitive to nutrient N enrichment effects. In addition, CHIS contains a diverse flora, with many rare and endemic plant species that might be sensitive to human-caused N inputs. Although rankings provide an indication of risk, park-specific data, particularly regarding nutrient-enrichment sensitivity, are needed to fully evaluate risk from nutrient N addition.

In general, the predominant sensitive vegetation type in MEDN is arid and semi-arid vegetation. Some arid and semi-arid ecosystems in the network are believed to be sensitive to, and affected by, nutrient N enrichment effects due in part to the high levels of atmospheric N deposition in portions of the network. In addition, N tends to be limiting to plant growth in arid and semi-arid plant communities where adequate moisture is available to support plant growth. Much of the data regarding effects of N deposition on arid land vegetation in southern California are from the coastal sage scrub (CSS) communities, downwind of the Los Angeles Basin, where dry N deposition is very high. Native shrub and forb seedlings in the CSS community are unable to compete with dense stands of exotic grasses under high N loading, and thus are gradually replaced by the grasses, especially following disturbances such as fire (Cione et al. 2002, Eliason and Allen 1997, Yoshida and Allen 2001).

Table 3. Estimated park rankings¹ according to risk of nutrient enrichment impacts on sensitive receptors (Source: Sullivan et al. 2011a)

Park Name ²	Park Code	Estimated Nutrient N Pollutant Exposure	Estimated Ecosystem Sensitivity to Nutrient Enrichment
Cabrillo	CABR	High	Very High
<i>Channel Islands</i>	CHIS	Moderate	High
<i>Santa Monica Mountains</i>	SAMO	Very High	High

¹ Relative park rankings are designated according to quintile ranking, among all I&M Parks, from the lowest quintile (Very Low risk) to the highest quintile (Very High risk).

² Park names are printed in bold italic for parks larger than 100 mi².

Coastal sage scrub occurs in near-coastal areas extending from central to southern California and southward into Mexico. Annual forbs are diverse in CSS plant communities, and include many rare species and species of concern (Fenn et al. 2011a). Plant productivity tends to be low, and highly responsive to inputs of atmospheric N deposition. Higher levels of N input contribute to increased cover of non-native invasive annual grasses, decreased cover of native forbs and shrubs, and reduced native forb diversity. These effects have been shown in both experimental N fertilization studies and observations along N deposition gradients (Fenn et al. 2011a).

Cox et al. (2014) investigated the influence of landscape conditions on vegetative conversion from CSS to exotic annual grasses in Riverside County, California. They included in their analyses measures of climate, topography, vegetation, land use, atmospheric N deposition, and fire. More than one-third of the study sites were converted to exotic grassland between 1930 and 2009. The sites that were converted had higher N deposition than sites that remained as CSS. Cox et al. (2014) concluded that CSS conservation and restoration actions would be most successful when focused on sites receiving < 11 kg N/ha/yr of N deposition and having low invasion of exotic grasses. The CL for protecting CSS resources was estimated to be 11 kg N/ha/yr. Fenn et al. (2010) reported modeled N deposition in the study area ranging from 5.3 (well below the CL) to 23.8 (well above the CL) kg N/ha/yr.

Fire is expected to have an important influence on terrestrial N eutrophication processes in the MEDN. Shortly after fire, N uptake to support plant growth tends to be substantial and may not reach steady state if fires are recurring (Fenn et al. 2008). Forests near the MEDN also show strong seasonality in N mobilization during precipitation events in the relatively wet winter season and N buildup in the terrestrial environment during dry summers. This pattern can be exacerbated during drought cycles (Riggan et al. 1994).

Elevated levels of N deposition have the potential to alter rates and magnitudes of ecosystem processes in the MEDN. Areas of chaparral near the MEDN that receive high levels of air pollution have experienced significant environmental change over the past several decades (Fenn et al. 1996, Fenn et al. 2003a). There are differences in environmental characteristics and pollution dynamics of these forests as compared with forests in the eastern United States that also are exposed to relatively high levels of N deposition (Fenn et al. 2003a). In particular, forests in the MEDN generally receive lower levels of precipitation and it tends to be more acidic. Furthermore, litter decomposition rates were altered in areas with highest N deposition, presumably because of lower carbon:nitrogen (C:N) ratios in the forest floor litter (Fenn and Dunn 1989, Fenn et al. 1998). These differences likely affect overall biogeochemical cycling.

Mediterranean forests often exhibit higher C:N in the forest floor than temperate forests. Net nitrification has been shown to increase when C:N is below 20 to 25 throughout the northeastern United States by Aber et al. (2003), and below 31 in red spruce forests in Vermont by McNulty et al. (1991). In contrast, the study by Fenn et al. (2008) in the MEDN region found a critical C:N level of 34, below which there was elevated NO_3^- leaching.

Some of the chaparral plant communities that are most exposed to air pollution from greater Los Angeles receive sufficiently high levels of atmospheric N deposition as to become N-saturated. Chaparral-dominated watersheds in close proximity to N emissions sources in southern California are characterized by very high concentrations of NO_3^- in stream water during high-flow events (Fenn and Poth 1999, Riggan et al. 1985). In contrast, N leaching is low in most watersheds in the Sierra Nevada to the east, and NO_3^- concentrations in these streams are usually below about 1 $\mu\text{eq/L}$. In the western United States, the Forest Service has set policy thresholds for stream NO_3^- concentration in N-limited ecosystems. A concentration more than 2 $\mu\text{eq/L}$ suggests concern for possible over-enrichment of N (Fenn et al. 2011b).

Chaparral in portions of southern California that receive elevated atmospheric N deposition leach substantial NO_3^- in stream water, due in part to a combination of factors, including 1) temporal asynchrony between the period of greatest biotic demand (spring and early summer) and the period of high runoff (winter; Fenn and Poth 1999, Fenn et al. 2011a), and 2) high evapotranspiration relative to precipitation amount (Fenn et al. 2011a). The deposition level at which NO_3^- leaching becomes pronounced (stream NO_3^- concentration > 14 $\mu\text{eq/L}$) is about 17 kg N/ha/yr (Fenn et al. 2008). Modeling results suggest that this level of N deposition has been exceeded in portions of SAMO.

Loss of N to the atmosphere via de-nitrification is relatively high in N-saturated chaparral in southern California, in the range of about 2 to 3 kg N/ha/yr (Anderson and Poth 1989, Fenn and Poth 2001). This process removes some N from impacted watersheds, but contributes to atmospheric buildup of greenhouse gasses.

Evaluation of N saturation in chaparral must consider the prevalence of fire in this ecosystem type. The stand-replacing fire interval is only about 40 to 60 years (Minnich and Bahre 1995). Much of the deposited N that has accumulated in the litter and above-ground vegetation is released to the atmosphere during fire, and fire contributes to pulses of NO_3^- in streamwater (Anderson and Poth 1989, Riggan et al. 1994). Much of the accumulated N in the mineral soil generally remains on-site after a fire (Fenn et al. 2011a).

In the highly polluted forests of the Los Angeles Basin, the following changes have been observed and have been attributed at least in part to atmospheric N deposition: increased foliar N, increased N to phosphorus (P) ratios, increased foliar NO_3^- , high NO_3^- concentrations in soil and soil solution, and high mineralization and nitrification rates (Fenn et al. 1996, Fenn et al. 1998, Fenn and Poth 1999, Padgett et al. 1999). Fenn et al. (1996) concluded that symptoms of N saturation were evident in chaparral sites in the MEDN region that received atmospheric N deposition of 20 to 25 kg N/ha/yr or higher. This is higher than the N deposition levels commonly encountered in many forest ecosystems in the eastern United States and may be high enough to cause nutrient N enrichment effects on plants (Sullivan et al. 2011a).

Coastal sage scrub vegetation in portions of southern California is in the process of being converted to exotic annual grassland. This change has been occurring over at least the past three decades. This loss of CSS vegetation is attributable to agriculture, grazing, frequent fire, and urbanization (Allen et

al. 1998), but N deposition also appears to play an important role. In areas where shrubs still occur, they now have understories dominated by exotic annual grasses and forbs, which have largely replaced the native herbaceous species. Frequent fire creates a feedback, with increasing dominance of highly flammable grasses which burn much more frequently than the shrubs which had a 25-30 year fire cycle (Minnich and Dezzani 1998a).

Although increased N deposition, fire frequency, and drought have all been hypothesized as causes of vegetation type conversion from CSS to grassland, little direct evidence is available. Kimball et al. (2014) constructed rain shelters in a CSS community in the foothills of the Santa Ana Mountains following a wildfire. They manipulated water and N inputs and collected community composition data for four years. Both N addition and drought slowed recovery of shrub cover. At the completion of the study, average native plant cover ranged from 20% (water reduction, N addition plots) to over 80% (water addition, ambient N input plots). Addition of N decreased the cover of native plants and increased the cover of non-native grasses.

The conversion of CSS to exotic annual grassland has likely been exacerbated by concurrent hydrologic change. Wood et al. (2006) documented a change in subsurface hydrology. The depth and rate of rainwater percolation into wildland hillslope soils was reduced after loss of CSS shrubs. There was a decrease in the distribution of rainwater to soil layers deeper than about 100 cm, and the predominant zone of soil water moved up to the upper 25 cm of soil. This hydrologic shift concentrates N in the upper soil and favors shallow-rooted annual grasses over deeper-rooted shrubs (Wood et al. 2006).

Observed changes in fungus, lichen, and other plant community composition may be a result of increased N loading. It has been hypothesized, but not demonstrated, that the decline in CSS community plant species could be linked to the decline of the arbuscular mycorrhizal community (Egerton-Warburton and Allen 2000). Nitrogen-induced changes in arbuscular mycorrhizal fungi may affect the growth of native seedlings. Larger-spored fungal genera (*Scutellospora* and *Gigaspora*) have decreased in number due to a failure to sporulate, with a concomitant proliferation of small-spored species, including *Glomus aggregatum*, *G. leptotichum*, and *G. geosporum*. This suggests selective pressure favoring the smaller-spored species of fungi (Egerton-Warburton and Allen 2000), and that N enrichment of the soil might alter the arbuscular mycorrhizal species composition and diversity. Additionally, community composition of epiphytic lichens can be altered by relatively small increases in N deposition (Fenn et al. 2003a). Most epiphytic lichens meet their nutritional requirements from atmospheric deposition and can store N in excess of their nutritional needs (van Herk 1999). In the San Bernardino Mountains, up to 50% of lichen species that occurred in the region in the early 1900s have disappeared, with a disproportionate number of locally extinct species being epiphytic cyanolichens (Fenn et al. 2003a, Nash and Sigal 1999).

Fenn et al. (2011a) estimated the critical load (CL) of N deposition based on changes in arbuscular mycorrhizal spore density and root infection along a depositional gradient ranging from about 9 to 20 kg N/ha/yr in western Riverside County, California. The estimated critical load was near 10 kg N/ha/yr (Table 4). Lichen communities in chaparral and oak woodlands in California have been

substantially changed by atmospheric N deposition. However, the response of lichens to N has been confounded by concurrent exposure to O₃ (Fenn et al. 2011a).

Table 4. Spore counts and percent root infection of arbuscular mycorrhizal fungi along an N deposition gradient in western Riverside County, California. Sites are arranged from north to south along an urban to rural gradient (Data from Egerton-Warburton and Allen 2000). Values in bold show a change in spore density and root infection and are interpreted to represent a critical load of N deposition. (Source: Fenn et al. 2011a)

Site	Spores/g Soil	% Root Infection	Soil N (µg/g)	N Deposition (kg N/ha/yr)	
				CMAQ ¹	Inferential ²
Jurupa Hills	23	19	86.2	19.6	.
Waterman Rd.	54	18	15.6	16.9	
Box Springs Mt.	66	17	26.4	14.7	20.2
Mockingbird Cyn	74	24	56.8	12.5	
Lake Mathews	70	23	39.5	11.1	
Motte Reserve	80	27	26.1	11.1	8.9
Hemet	78	22	12.8	9.6	
Lake Skinner	103	39	7.5	8.7	6.6
Santa Margarita	105	45	9.7	8.7	

¹ N deposition using the CMAQ model is reported as modeled wet plus dry deposition (Tonnesen et al. 2007).

² Deposition calculated with the inferential method also includes wet plus dry deposition to vegetation and soil; Bytnerowicz, A.; Jovan, S. Unpublished data on file with Andrzej Bytnerowicz, Research Ecologist, Forest Fire Laboratory, 4955 Canyon Crest Dr., Riverside, CA 92507

Of the epiphytic lichen species known to occur prior to the early 1900s on coast live oak in the chaparral and oak vegetation communities of the Los Angeles Basin (Hasse 1913), about half were no longer present by the early 1980s (Ross 1982). This loss of lichen diversity has been attributed to both O₃ (Nash and Sigal 1999, Ross and Nash 1983) and N inputs (Riddell et al. 2008). Other contributing factors may have included habitat loss, urbanization influence on humidity, and increased fire incidence (Riddell et al. 2008).

Lichen richness and abundance have been shown to decrease with NH₃ emissions and N deposition in the Central Valley (Jovan and McCune 2005, Jovan 2008). Fenn et al. (2011a) reported that, of the 53 lichen survey sites in California that had relatively clean air scores (> 0), only three occurred in 4-km grid cells where the CMAQ model (Byun and Schere 2006) predicted more than 5.5 kg N/ha/yr of total N deposition. Fenn et al. (2008) found a distinct shift in lichen community composition from N-sensitive to more tolerant lichen species at N deposition of 3.1 kg N/ha/yr. Complete elimination of acidophytes (i.e., species that tolerate high acidity) from the lichen community was predicted at an N deposition level of about 10.2 kg N/ha/yr.

Fenn et al. (2008) examined eutrophication responses from N deposition in ponderosa pine (*Pinus ponderosa*) forests in California. Epiphytic lichens were used as bioindicators of N effects (Blett et al. 2003, Bobbink et al. 2003, Fenn et al. 2003a). Fifteen of the 24 study sites were located in the Sierra Nevada Mountains and 9 in the San Bernardino Mountains. Lichen species were classified into

three groups based on their responses to atmospheric deposition of N (Geiser and Neitlich 2007, Jovan and McCune 2006, Jovan 2008). Acidophytes are sensitive to small increases in N deposition (Van Herk et al. 2003), and can be eliminated from the lichen community at moderate levels of deposition. This group included beard lichens (e.g., *Bryoria fremontii*) and other species used by wildlife for forage and nesting material in the Sierra Nevada (McCune et al. 2007). Neutrophytes (species that thrive under conditions of moderate N availability) are more tolerant of N input. Nitrophytes (species that thrive under conditions of high N availability) are fast growing, respond to increased NH_3 deposition, and favor high pH substrates, including the bark of hardwood trees.

Fenn et al. (2011a) summarized data reflecting the relationship between atmospheric N deposition and stream NO_3^- concentration in chaparral vegetation in three areas of southern California: 1) San Dimas Experimental Forest in the San Gabriel Mountains (Meixner et al. 2006, Riggan et al. 1985, Riggan et al. 1994), Devil Canyon watershed in the western San Bernardino Mountains (Fenn and Poth 1999, Meixner and Fenn 2004), and 3) Chamise Creek in SEKI to the east of the MEDN (Fenn et al. 2003a, Fenn et al. 2003c). Furthermore, 56 chaparral streams were sampled and analyzed for NO_3^- concentration across the greater Los Angeles Basin and northern San Diego County by Riggan et al. (1985). The concentrations of NO_3^- in streams located in chaparral watersheds in California are the highest reported in North America, with concentrations in some streams exceeding 200 $\mu\text{eq/L}$ at some times (Fenn and Poth 1999, Fenn et al. 2003a, Fenn et al. 2003c, Fenn et al. 2011a, Riggan et al. 1985, Riggan et al. 1994). Nevertheless, total N flux is typically less than about 10 kg N/ha/yr because evapotranspiration is very high relative to precipitation water input (Fenn and Poth 1999, Riggan et al. 1985).

Near San Jose, N deposition has contributed to displacement of native herbaceous species by non-native invasive annual grasses (Weiss 1999). Portions of inland Riverside and San Bernardino counties receive levels of atmospheric N deposition up to about 20 kg N/ha/yr and have been converted largely to non-native annual grasses over the last several decades (Allen et al. 1998, Fenn et al. 2011a, Minnich and Dezzani 1998b, Talluto and Suding 2008).

Fenn et al. (2010) reported the results of a field study along a gradient of N deposition ranging from about 8.7 to 19 kg N/ha/yr. The coverage of non-native grasses increased with atmospheric N deposition and soil N content. The cover of native shrubs and forbs was inversely correlated ($P < 0.001$) with N deposition. Native forb richness decreased across the N deposition gradient from 67 to 16 species per site within the deposition range from 9 to 11 kg N/ha/yr (Fenn et al. 2010). Based on these data, Fenn et al. (2011a) concluded that the critical N load in this region was about 10 kg N/ha/yr to protect plant species richness.

Biodiversity of understory plants in mixed conifer forests decreased substantially over a 30-year period in areas of the San Bernardino Mountains that received high N deposition (Allen et al. 2007). This effect has been attributed to the introduction and establishment of invasive non-native plant species. In three of six study sites, about 20 to 40% of native understory plant species were lost between 1973 and 2003, as invasive species cover increased (Allen et al. 2007, Fenn et al. 2011a).

Fenn et al. (2011a) used percent cover and richness of plant groups (i.e., native forb, shrub, exotic grass) to estimate the critical load of N deposition to protect plant biodiversity along an urban to rural gradient in Riverside County, California. Observed changes in native forb species richness suggested a critical load near 10 kg N/ha/yr (Table 5).

Table 5. Percent cover and richness (per 3 ha) of plant groups along an N deposition gradient in western Riverside County, California. Sites are arranged from north to south along an urban to rural gradient. Forb species richness in bold shows a rapid drop in richness, suggesting a critical load of 10 kg N/ha/yr by the CMAQ model (between 9.0 and 11.1 kg N/ha/yr), and 7.8 kg N/ha/yr by the inferential method (between 6.6 and 8.9 kg N/ha/yr). (Source: Fenn et al. 2011a)

Site	Exotic Grass (% Cover)	Native Forb (% Cover)	Shrub (% Cover)	Native forbs (No. of species per 3 ha)	Soil N (µg/g)	N Deposition (kg N/ha/yr)	
						CMAQ ¹	Inferential ²
Jurupa Hills	63.5	4.0	2.2	16	37.7	19.6	
Box Springs	69.2	18.5	2.4	31	32.6	14.7	20.2
Botanic Garden	36.0	25.4	0.2	20	28.9	13.4	
Lake Perris	0.5	26.1	2.8	30	20.3	11.1	
Mott Reserve	6.7	14.3	11.2	37	30.6	11.1	8.9
Lopez Cyn	11.1	19.6	19.3	67	9.6	9.0	6.6
Tucalota Hills	1.5	35.7	35.0	50	10.5	8.7	

¹ N deposition using the CMAQ model is reported as modeled wet plus dry deposition (Tonnesen et al. 2007).

² Deposition calculated with the inferential method also includes wet plus dry deposition to vegetation and soil; Bytnerowicz, A.; Jovan, S. Unpublished data on file with A. Bytnerowicz, Research Ecologist, Forest Fire Laboratory, 4955 Canyon Crest Dr., Riverside, CA 92507

Elevated O₃ exposure and N deposition, in combination, have contributed to changes in nutrient cycling in mixed conifer forests of the Los Angeles Basin. Carbon storage has increased in woody above-ground biomass; foliar turnover has increased; and forest litter production and accumulation have increased (Arbaugh et al. 1999, Fenn et al. 2003a, Fenn et al. 2011a). The susceptibility of trees to disturbance, including insect infestation, drought, and disease, has increased, resulting in increased tree mortality (Fenn et al. 2003a). Taken together, these changes have increased ecosystem susceptibility to wildfire (Fenn et al. 2011a, Grulke et al. 2009).

Pardo et al. (2011b) compiled data on empirical CL for protecting sensitive resources in Level I ecoregions across the conterminous United States against nutrient enrichment effects caused by atmospheric N deposition. The lower end of the range of empirical CL reported by Pardo et al. (2011b) for protecting sensitive resources in parks in the MEDN against nutrient-N deposition was consistently less than 8 kg N/ha/yr for protection of mycorrhizal fungi, lichens, and herbaceous plants (Table 6). Ambient N deposition reported by Pardo et al. (2011b) at both CHIS and SAMO was higher than that. Thus, the lower end of the estimates of empirical CL for protecting these resources

was exceeded at both parks. Reported ambient N deposition also exceeds the empirical CL for protecting forest vegetation and for preventing NO_3^- leaching in drainage water, but only at SAMO.

Ellis et al. (2013) estimated the CL for nutrient-N deposition to protect the most sensitive ecosystem receptors in 45 national parks. The lowest terrestrial CL of N is generally estimated for protection of lichens (Geiser et al. 2010). Changes to lichen communities may signal the beginning of other changes to the ecosystem that might affect structure and function (Pardo et al. 2011a). Ellis et al. (2013) estimated the N CL for CHIS in the range of 3.1-5.2 kg N/ha/yr for protection of lichens.

Table 6. Empirical critical loads for nitrogen in the MEDN, by ecoregion and receptor from Pardo et al. (2011b). Ambient N deposition reported by Pardo et al. (2011b) is compared to the lowest critical load for a receptor to identify potential exceedance, indicated by graying. A critical load exceedance suggests that the receptor is at increased risk for harmful effects.

NPS Unit	Ecoregion	N Deposition (kg N/ha/yr)	Critical Load (kg N/ha/yr)				
			Mycorrhizal Fungi	Lichen	Herbaceous Plant	Forest	Nitrate Leaching
Channel Islands NP	Mediterranean California	8.2	7.8 - 9.2	3.1 - 6	6 - 33	17 - 39	10 - 17
Santa Monica Mountains NRA	Mediterranean California	19.8	7.8 - 9.2	3.1 - 6	6 - 33	17 - 39	10 - 17

Ozone Injury to Vegetation

A great deal of research has been conducted in southern California on the effects of O₃ on forest communities, especially ponderosa pine. Although ponderosa pine is highly sensitive to O₃, it does not occur in the MEDN parks. The O₃-sensitive plant species that are known or thought to occur within the I&M parks found in the MEDN are listed in Table 7. Those considered to be bioindicators, because they exhibit distinctive symptoms when injured by O₃ (e.g., dark stipple), are designated by an asterisk. No O₃ sensitive and/or bioindicator species have been identified for CABR, but CHIS and SAMO each contain several (Table 7).

The W126 and SUM06 exposure indices calculated by NPS staff are given in Table 8, along with Kohut's (2007a) O₃ risk ranking. The NPS and Kohut ranking systems differ. The NPS ranking system (NPS 2010) is a quick assessment of O₃ condition that ranks O₃ exposure levels according to injury thresholds from the literature (Heck and Cowling 1997), using a 5-year average of either the W126 or SUM06 index. The former is a measure of cumulative O₃ exposure that preferentially weights higher concentrations; the latter is a measure of cumulative O₃ exposure that includes only hourly concentrations over 60 ppb. Both metrics are calculated over a 3-month period. The W126 was classified as Moderate exposure at values between 7 and 13 ppm-hr, as defined by NPS (2010). Values higher than 13 ppm-hr were classified as High exposure, and values lower than 7 ppm-hr were classified as Low exposure. The SUM06 was classified as Moderate at values between 8 and 15 ppm-hr. Higher and lower values were classified as High and Low, respectively, as defined by NPS (2010). Using these criteria, O₃ at the MEDN parks is rated Low to Moderate at CABR and CHIS and High at SAMO.

Table 7. Ozone-sensitive and bioindicator plant species known or thought to occur in the I&M parks of the MEDN. (Data Source: E. Porter, National Park Service, pers. comm., August 30, 2012); lists are periodically updated at <https://irma.nps.gov/NPSpecies/Report>).

Species	Common Names	Park ¹		
		CABR	CHIS	SAMO
No Species Present		x		
<i>Ailanthus altissima</i> *	Tree-of-heaven			x
<i>Apocynum cannabinum</i>	Dogbane, Indian hemp			x
<i>Artemisia douglasiana</i> *	Mugwort		x	x
<i>Quercus kelloggii</i>	California black oak		x	
<i>Robinia pseudoacacia</i>	Black locust		x	x
<i>Salix gooddingii</i>	Gooding's willow			x
<i>Sambucus mexicana</i> *	Blue elderberry		x	x

¹ Park acronyms are printed in bold italic for parks larger than 100 mi².

* Bioindicator species

Kohut's approach constitutes a more rigorous assessment of potential risk to plants. It considers both O₃ exposure and environmental conditions (soil moisture). Kohut also used injury thresholds from

the literature, but evaluated a different O₃ metric (after Lefohn et al. 1997), the W126 over a 5-month period in conjunction with the N100 (number of hours over 100 ppb O₃). The rationale for the N100 statistic is that higher O₃ concentrations are most likely to cause plant injury. Kohut examined five individual years of O₃ exposure and soil moisture data and considered the effects of low soil moisture on O₃ uptake each year when assigning risk. Soil moisture is important because dry conditions induce stomatal closure in plants, which has the effect of limiting O₃ uptake and injury. In areas where low soil moisture levels correspond with high O₃ exposure, uptake and injury are limited by stomatal closure even when exposures are relatively high.

The results of both ranking systems should be considered when evaluating the potential for O₃ injury to park vegetation. The Kohut approach considered environmental conditions that significantly affect plant response to O₃, but exposures have likely changed since the time of the assessment (1995-1999). The NPS approach considers more recent O₃ conditions (2005-2009), but not environmental conditions.

Kohut (2007b) assessed O₃ risk in the three MEDN parks, one based on in-park monitoring data (CHIS) and the other two based on kriging of data from surrounding monitoring stations. The exposure indices varied across the parks. The SUM06 index exceeded the threshold for foliar injury in CABR and SAMO, but not in CHIS. The criteria for injury under the W126 index were not satisfied in CABR or CHIS, but were satisfied in SAMO. At SAMO, high O₃ exposure creates the potential for foliar injury and the risk was judged by Kohut (2007b) to be High. Nevertheless, in a given year, dry soil conditions can reduce the likelihood of foliar injury even in this park, and the level of risk therefore varies from year to year.

Table 8. Ozone assessment results for I&M parks in the MEDN based on estimated average 3-month W126 and SUM06 ozone exposure indices for the period 2005-2009 and Kohut's (2007a) ozone risk ranking for the period 1995-1999¹.

Park Name ²	Park Code	W126		SUM06		Kohut O ₃ Risk Ranking
		Value (ppm-hr)	Ranking	Value (ppm-hr)	Ranking	
Cabrillo	CABR	10.46	Moderate	12.28	Moderate	Low
<i>Channel Islands</i>	CHIS	7.05	Moderate	7.81	Low	Low
<i>Santa Monica Mountains</i>	SAMO	16.34	High	20.82	High	High

¹ Parks are classified into one of three ranks (Low, Moderate, High), based on comparison with other I&M parks.

² Park names are printed in bold italic for parks larger than 100 mi².

Background levels of O₃ entering southern California from the Pacific Ocean can be determined from measurements taken at CHIS, which are generally similar to comparable background measurements from Trinidad Head in northern California (Oltmans et al. 2008). The O₃ monitoring site at CHIS had a 10-year record of surface O₃ observations, although monitoring was discontinued in 2004. During the winter through early summer period (December-July), the seasonal pattern of O₃ concentrations

at CHIS was similar to that measured at Trinidad Head, with relatively low peak values near 45 ppb. Patterns at the two sites differed during the second half of the annual cycle, with a second peak occurring during the fall season at CHIS, but not at Trinidad Head. There were relatively infrequent cases of O₃ concentration > 70 ppb at CHIS. These higher concentrations were associated with air flow from the mainland, which is influenced by human-caused emissions from the Los Angeles Basin.

Visibility Degradation

Haze is not measured in or near the MEDN parks. Nevertheless, analyses conducted by the Western Regional Air Partnership (WRAP) indicated that both natural and human-caused emissions sources contribute to substantial visibility impairment throughout the western United States. The natural sources include wildfire and biogenic (derived from natural biological sources) emissions from vegetation. In addition, air pollution sources both inside and outside the WRAP domain, including international off-shore shipping and sources from Mexico, Canada, and Asia, can in some cases be substantial contributors (Suarez-Murias et al. 2009).

Visibility modeling results reported by Suarez-Murias et al. (2009) for IMPROVE monitoring site locations in wilderness areas managed by the Forest Service in the MEDN region are summarized in Table 9. In order to achieve Regional Haze Rule goals of no human-caused visibility degradation by 2064, a haze reduction of 60 to 65% is required at these locations.

Source apportionment analyses were conducted for the California Regional Haze Plan (Suarez-Murias et al. 2009), using regional-scale, three-dimensional air quality models that simulate emissions, chemical transformations, and transport of criteria air pollutants and fine particulate matter. Effects on visibility in Class I areas in California were then evaluated for organic C, SO₄²⁻, and NO₃⁻, which are the three major drivers of haze in California.

Table 9. IMPROVE monitors and visibility (deciviews¹) at Class 1 areas in the MEDN region. (Source: Suarez-Murias et al. 2009)

			Current Conditions (2000-2004 Baseline)		Future Natural Conditions (2064 Goals)	
Improve Monitor (name and elevation in meters)		Class I Area(s)	Worst Days	Best Days	Natural Worst Days	Improvement Needed from Current Visibility on Worst Days
RAFA (957 m)	San Rafael	San Rafael Wilderness	18.8	6.4	7.6	60%
SAGA (1791 m)	San Gabriel	San Gabriel Wilderness Cucamonga Wilderness	19.9	4.8	7.0	65%

¹ The deciview visibility metric expresses uniform changes in haziness in terms of common increments across the entire range of visibility conditions, from pristine to extremely hazy conditions. Because each unit change in deciview represents a common change in perception, the deciview scale is like the decibel scale for sound. A one deciview change in haziness is a small but noticeable change in haziness under most circumstances when viewing scenes in Class I areas.

Directly emitted anthropogenic organic C was estimated to contribute half or less of the atmospheric organic C in most areas of California except near San Francisco. Anthropogenic sources of NO_x contributed at least half of the NO₃⁻ in all Class I areas in California except Redwood National Park. The anthropogenic SO₄²⁻ contribution ranged from 1 to 35% at various Class I areas, with highest contributions in southern California. The primary drivers of haze in California derive mostly from

natural sources (organic C), mobile sources (NO_3^-), and off-shore and non-WRAP regional sources (SO_4^{2-} ; Suarez-Murias et al. 2009).

Toxic Airborne Contaminants

Data on the occurrence and potential effects of atmospheric deposition of toxic materials in national park lands in the MEDN are generally not available. However, elevated levels of both banned and current-use pesticides and industrial contaminants were found in vegetation, fish, and sediments in SEKI to the north of the MEDN during the Western Airborne Contaminants Assessment Project (Landers et al. 2008). Analyses and modeling suggested that intensive regional agriculture was at least partly responsible for these contaminants. It is likely that MEDN parks have also received airborne toxics deposition from long range trans-Pacific transport.

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