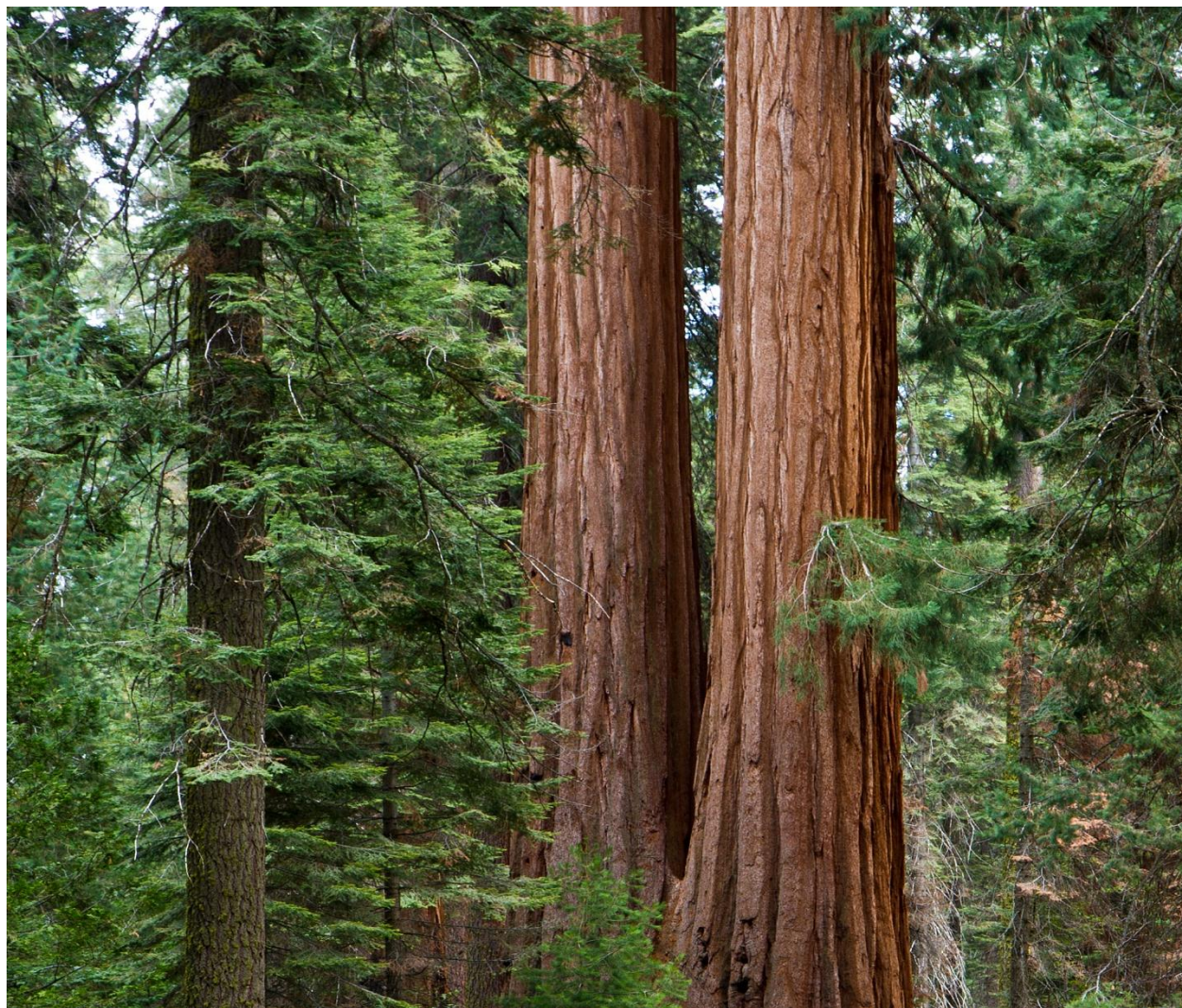




A Natural Resource Condition Assessment for Sequoia and Kings Canyon National Parks

Appendix 2 - Air Quality

Natural Resource Report NPS/SEKI/ NRR—2013/665.2



ON THE COVER

Giant Forest, Sequoia National Park
Photography by: Brent Paull

A Natural Resource Condition Assessment for Sequoia and Kings Canyon National Parks

Appendix 2 - Air Quality

Natural Resource Report NPS/SEKI/ NRR—2013/665.2

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The National Park Service, Natural Resource Stewardship and Science office in Fort Collins, Colorado, publishes a range of reports that address natural resource topics. These reports are of interest and applicability to a broad audience in the National Park Service and others in natural resource management, including scientists, conservation and environmental constituencies, and the public.

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This document contains subject matter expert interpretation of the data. The authors of this document are responsible for the technical accuracy of the information provided. The parks refrained from providing substantive administrative review to encourage the experts to offer their opinions and ideas on management implications based on their assessments of conditions. Some authors accepted the offer to cross the science/management divide while others preferred to stay firmly grounded in the presentation of only science-based results. While the authors' interpretations of the data and ideas/opinions on management implications were desired, the results and opinions provided do not represent the policies or positions of the parks, the NPS, or the U.S. Government.

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Scope of analysis

The condition of air quality in Sequoia and Kings Canyon National Parks (SEKI) is a consequence of its geographic location relative to significant sources of air pollution. The Parks are downwind of numerous major urban areas with associated industrial activity, two heavily-travelled transportation corridors (I-5 and CA 99), and the extensive San Joaquin Valley agricultural landscape which is among the largest producers of agricultural products in the U.S. (CDFA 2011). Long-range transport of pollutants from Asia and elsewhere also contribute to upwind sources (Huang et al 2010). The Parks are within the San Joaquin Valley Air Basin (SJVAB, Figure 1), the southern half of California's broad central valley, which is classified as "extreme non-attainment", the worst category relative to federal standards of ozone (USEPA 2011). Over ten percent of the state's population lives in the eight counties of the SJVAB and air pollution emission sources within the SJV account for about 14 percent of total statewide emissions (CARB 2009). Since 1980, population growth in the SJV has been more rapid than in other parts of California, and is projected to reach over 5 million by 2020 (almost 12% of state population, CARB 2009).

Geographically, the SJVAB is rimmed to the west by the Coast Range, to the south by the Tehachapi Range, and to the east by the Sierra Nevada, and therefore constrains pollutants within its boundaries. General surface air flow patterns carry contaminants southeastward from their sources, through the San Joaquin Valley to the southern end where they are blocked by the Tehachapi Range. From here, pollutants are forced upward and outward, into major drainages, or re-circulate northward along the western flanks of the Sierra Nevada in a vortex called the Fresno Eddy. Sequoia and Kings Canyon National Parks are situated on the eastern edge of the SJVAB and the lower slopes and drainages of the western flanks of the Parks are severely impacted by air pollution, including the Fresno Eddy (Figure 1, also Beaver et al. 2010). Airflow patterns in the SJVAB that bring pollutants, especially ozone, to the Parks have been well-characterized (Beaver et al 2010). Factors influencing direction, strength, and pollutant load include incursions of marine air through major western passes into the SJVAB, upslope and downslope winds in the Sierra Nevada, temperature, and air pressure patterns. The conditions of highest pollutant loading to the west slopes of the Sierra occur during anticyclonic events, high pressure systems that push down and trap polluted air onto the lower slopes of the Sierra Nevada, near the foothills of Sequoia (Beaver et al 2010). Sequoia and Kings Canyon National Parks have the most severe air pollution impacts on vegetation of any other western national park (Sullivan et al. 2001).

The Mediterranean climate of California has implications for timing, type, and distribution of pollutant deposition. Very little precipitation falls between June and October, temperatures are hot, and the air is dry, favoring particulate and gaseous pollutant deposition. Pollutants can build up on leaf and other surfaces, and then wash off when it rains, creating pollutant pulses into soils, streams and lakes. Precipitation between the months of November and May delivers wet chemical species, but generally cleans the air of particulates. This, combined with cooler temperatures, leads to decreased dry and gaseous pollutants in the winter. High elevation snowpack is generally deep, up to 4 m, and can sequester pollutants throughout the winter until released by summer snowmelt.

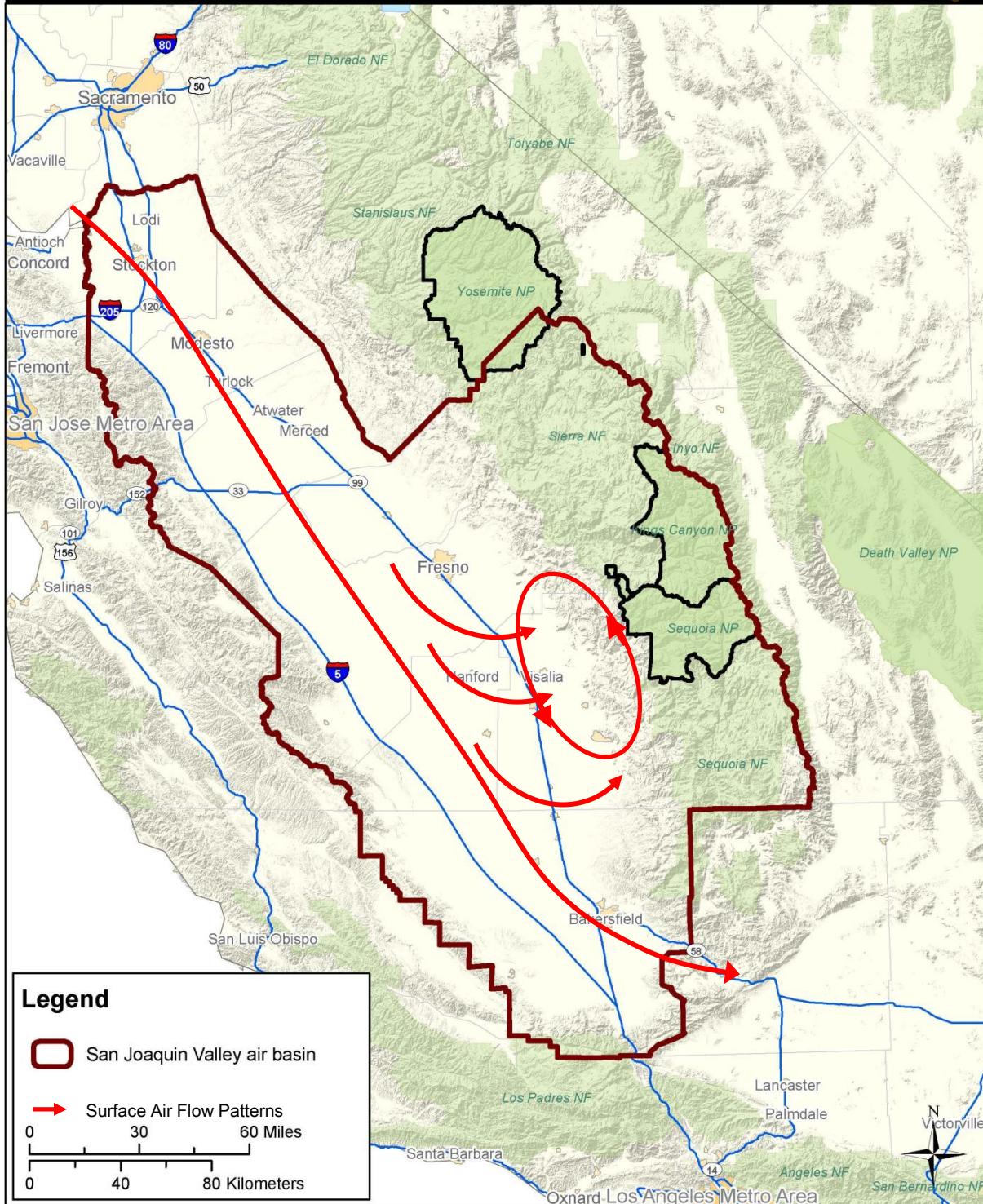


Figure 1. Sequoia and Kings Canyon National Parks (SEKI) are shown in the context of the San Joaquin Valley air basin. Red arrows show generalized surface air flow patterns that bring pollutants from sources towards SEKI. The Fresno Eddy is a CCW vortex that recirculates pollutants northward along the Parks' western extent.

Pollutants of concern to SEKI, and summarized in this report, include ozone and its precursors, wet and dry nitrogen deposition, wet and dry sulfur deposition, fine and coarse particulates, mercury, pesticides, and other contaminants. This chapter will report on spatial patterns and trends of pollutants in the Parks to the degree possible, given the capability of data sources. The assessment will be made relative to national standards, ecosystem effects, and reference conditions, drawing from ongoing monitoring data, from pollutant assessment reports, and from the literature.

Statutory protection

SEKI air quality is protected by elements of the Organic Act, the Wilderness Act, and the Clean Air Act. Sequoia and Kings Canyon National Parks air sheds are afforded the greatest degree of air quality protection under the Clean Air Act as Class I wilderness areas. Furthermore, “In cases of doubt as to the impacts of existing or potential air pollution on park resources, the Service will err on the side of protecting air quality and related values for future generations (NPS 2006)”. Sequoia and Kings Canyon take multiple approaches to fulfill their legislative responsibility to protect air quality, among which is the routine monitoring of criteria pollutants and visibility. Nationwide, the NPS Air Resources Division responds to air quality management needs by routine publication of Air Quality in the National Parks Reports (NPS-ARD 2002, 2005, 2006, 2008, 2009), which summarize status and trends in all legislatively affected NPS units. Because these reports, and others (eg. Sullivan et al. 2001), provide a wealth of information on pollutants, sources, status and trends, and impacts on SEKI’s natural resources, this chapter seeks mainly to summarize existing reports, fill in gaps with new analyses, and provide more detailed information specific to SEKI. To make the discussion meaningful, however, some background information on pollution chemistry, sources, and impacts will be reiterated in a general way. For more in-depth discussions of these, please see NPS-ARD (2002) and Sullivan et al (2001).

Pollutants

Each pollutant covered in this report is described below, including a summary of its sources and effects. Poor air quality can impact other SEKI resources. From a management perspective, these impacts are Air Quality Related Values¹ (AQRVs), and include a range of effects, both direct and indirect. These are described under Pollutant Effects, for each pollutant described below.

Ozone

Pollutant Sources

Sequoia and Kings Canyon are among the most ozone-polluted national parks in the country, due to their proximity to sources in the heavily polluted San Joaquin Valley. Ozone (O₃) is a secondary pollutant, formed from precursor nitrogen oxides (NO_x) and volatile organic compounds (VOCs) in the presence of sunlight. Population growth and the concomitant increase in industry and transportation generate precursors, and thus contribute to ozone pollution in the SJVAB (see page 29 for discussion). Regulations to control ozone pollution have been strengthened recently with Federal (2008) and state (2006) approval of stricter ozone standards.

¹ An AQRV is a resource that may be adversely affected by a change in air quality. The resource may include visibility or a specific scenic, cultural, physical, biological, ecological, or recreational resource (USFS et al. 2010).

Pollutant Effects

Elevated ozone concentrations are harmful to both humans and vegetation, but this report will focus on the vegetation effects of ozone. Park visitors are alerted to unhealthy air quality by the SEKI daily air advisory program. Vegetation ozone injury is a result of the combined effects of: ozone concentration and duration of exposure; plant susceptibility due to species and endogenous factors such as genotype, age, etc.; and environmental factors that favor plant ozone uptake, such as moisture, light, and temperature (Smith et al. 2008). Extensive ozone injury has been documented in SEKI on ponderosa pines (*Pinus ponderosa*) and Jeffrey pines (*Pinus jeffreyi*) (Arbaugh et al. 1998, Duriscoe 1990, Miller 1996, Pronos et al. 1978). While giant sequoia seedlings (*Sequoiadendron giganteum*) have shown ozone injury in growth chambers at ambient concentrations (Grulke et al. 1998, 1996) and California black oaks (*Quercus kelloggii*) have shown ozone injury at high exposure sites in southern California (Miller 1996, Miller and Millecan 1971, Miller et al. 1989), extant ozone injury on giant sequoia and black oaks has not yet been documented in the Parks. Ozone also can have a broad effect on forested landscapes, potentially altering species composition and influencing pest interactions, soil moisture, and fire regimes (Bytnerowicz et al. 2009, McBride and Laven 1999, Miller et al. 1982, Smith 1974). Other ozone-sensitive plants in SEKI include *Apocynum androsaemifolium*, *A. cannabinum*, *Artemisia douglasiana*, *Physocarpus capitatus*, *Populus tremuloides*, *Rhus trilobata*, *Rubus parviflorus*, *Salix scouleriana*, and *Sambucus Mexicana* (NPS 2006b). Sequoia and Kings Canyon have been characterized as high-risk with respect to ozone injury in plants (Kohut 2007). Despite this risk, however, the only systematic surveys of ozone injury in plants in SEKI have been on *Pinus jeffreyi* and *P. ponderosa*.

Nitrogen

Pollutant Sources

Nitrogen (N) is deposited to the Sierra Nevada in both wet and dry forms: as NH_4^+ and NO_3^- in rain, cloudwater and snow, as NO_x , NH_3 and HNO_3 gas, and as particulate NH_4^+ and NO_3^- . The majority of NO_x emissions in California are from mobile sources and most NH_3 emissions are from livestock waste (Fenn et al. 2010a). Ammonia and HNO_3 gases react to form particulate ammonium nitrate (NH_4NO_3).

Pollutant Effects

SEKI's risk from nitrogen deposition is dependent on both nitrogen load and sensitivity of individual ecosystems to increased inputs (Sullivan et al. 2011). Nitrogen nutrient enrichment, rain and soil-water acidification, changes in nutrient cycling, plant and animal development and competitive interactions, and nitrogen saturation can be results of enhanced nitrogen deposition (see Sullivan et al. 2011 for a Parks-relevant review). This assessment will focus on terrestrial effects. Aquatic effects of enhanced nitrogen deposition are well-documented (Clow et al. 2003, Sullivan et al. 2011), but are beyond the scope of this assessment - they are covered in the Water Quality assessment.

Nitrogen saturation occurs when an ecosystem has surpassed its ability to absorb nitrogen, thus nitrogen is no longer retained, or is toxic to organisms. Nitrogen saturation is well-documented in southern California, especially in the high nitrogen deposition mountain ecosystems downwind of Los Angeles (Fenn and Poth 1999, Fenn et al. 2003). In the Sierra Nevada, nitrogen retention is variable: high-elevation watersheds are reported to lose nitrate in snowmelt, mid-elevation coniferous forests and lower-elevation oak woodlands retain nitrogen, but

chaparral ecosystems in the Sierra Nevada exhibit the greatest nitrate leaching indicating nitrogen saturation (Fenn et al. 2003). These systems have high rates of nitrogen deposition due to proximity to nitrogen emission sources and from atmospheric inversions that concentrate pollutants below an elevation of 1000 to 2000 meters.

Critical loads (CLs) are a means of linking potential ecosystem response to pollutant deposition, and are currently being developed throughout Europe and parts of the United States for various pollutants (Pardo et al. 2011). CLs within each of the vegetation types are determined based on a biological or chemical response to levels of a pollutant. Nitrogen CLs have been identified for several major ecosystem types found in SEKI, based on the fertilizing effects of enhanced nitrogen (Fenn et al. 2010a): grassland (6 kg-N/ha/y), chaparral (10-14 kg-N/ha/y), and mixed conifer (17 kg/ha/y), but CLs for sensitive epiphytic lichen species within those ecosystems are the lower 3-8 kg/ha/y. In this chapter, spatial patterns of nitrogen deposition in the Parks will be examined with respect to critical loads.

Sulfur and particulates

Pollutant Sources

Nationwide, the largest source of anthropogenic SO₂ emissions is from coal-fired power plants. However because California lacks these type of energy generation facilities, here dominant sources include industrial processes and the burning of high sulfur containing fuels by locomotives, large ships, trucks, and non-road equipment (USEPA 2011), and from outside the region (Suarez-Murias and Zulawnick 2008). Of great concern is the poor summertime visibility at SEKI, as well as the adverse health effects of fine (2.5 micron) and coarse (10 micron) particulates (PM 2.5, PM 10). The SJVAB is the greatest source of particulates in SEKI, especially dust and nitrogen from agricultural activities and nitrogen from mobile sources. Smoke from wild- and prescribed fires also contributes a large proportion of particulates. However, about 12% of nitrogen particulates and about 70% of sulfate particulates are estimated to come from outside the region (Suarez-Murias and Zulawnick 2008). This report will include updated elements of the Suarez-Murias and Zulawnick (2008) particulate analysis to include data since 2004.

Pollutant Effects

Sulfur impacts on ecosystems at deposition levels generally seen in the western U.S. include acidification of sensitive lakes with low acid-neutralizing capacity. SEKI lake impacts are considered under the Water Quality chapter. Adverse vegetation effects are considered unlikely to occur where total sulfur deposition rates are less than about 5 kg/ha/yr (Peterson et al. 1992a,b, Sullivan et al. 2001). Particulate visibility impairment affects the visitor experience, and is managed as an AQRV. Since spectacular vistas are a SEKI resource, the poor visibility may ultimately deter visitors from the Parks. Particulate effects on vegetation are due primarily to their chemical additions to the ecosystem, although particulate impairment of plant gas exchange has been reported (Bell and Treshow 2002). This, however, has not been studied at SEKI.

Other Contaminants

Pollutant Sources

Sources of mercury include 1) naturally emitted mercury, from the outgassing of volcanoes and geothermal vents, and evaporation from naturally enriched soils, wetlands, and oceans. This form of mercury quickly volatilizes into the atmospheric pool and become globally distributed; and 2) anthropogenic emissions, in forms that have high deposition rates that contribute to local and regional mercury deposition. Elevated ozone has been shown to exacerbate emissions from mercury-rich and natural soils (Gustin et al. 2008), and could potentially enhance mercury deposition through atmospheric oxidative reactions (Lindberg et al. 2007). Models have shown deposition to be elevated specifically over the Great Basin and Sierra Nevada (Gustin et al. 1997, Bullock et al. 2008). It is currently unclear how much of the mercury deposited in the Sierra comes from natural vs. anthropogenic sources.

Pesticides are, and have been, used extensively in California's heavily-farmed San Joaquin Valley. They are of concern for their toxicity, as well as for their ozone-forming potential, as VOCs (Sullivan et al. 2001). Research results show that pesticides used currently, and pesticides that have been banned for over 30 years in California, are present in current deposition in Sequoia National Park (Landers et al. 2008).

Pollutant Effects

Mercury is of concern in SEKI because, as it moves through natural systems, it bio-accumulates as toxic methylated mercury (MeHg), a neurotoxin and teratogen. There is evidence that National Parks throughout the west, including Sequoia, have elevated mercury deposition which is resulting in mercury levels in fish that are so high they are toxic to humans and other piscivorous animals (Landers et al. 2008). This report will analyze trends in precipitation mercury, as well as summarize mercury and pesticide results from the Western Airborne Contaminants Assessment Project (WACAP) for their implications in SEKI.

Critical questions

Spatial patterns

The analysis of spatial distribution of air pollution in the Parks is limited to the two pollutants for which there exist sufficient spatial data: ozone concentration and total nitrogen deposition.

- What is the spatial pattern of ozone concentration in the region and in the park?
- What is the spatial pattern of nitrogen deposition in the region and in the park?

Temporal patterns

Temporal trend analysis of pollutant concentration and deposition is possible, with available data, for ozone, total nitrogen (wet + dry), total sulfur (wet + dry), precipitation chemistry, dry deposition chemistry, and visibility.

- Is ozone concentration changing over time? The ozone concentration trend is updated to include the last 10 years of data.
- Are nitrogen and sulfur deposition changing over time? Total nitrogen (wet + dry) and total sulfur (wet + dry) deposition for SEKI is analyzed and reported here for the first time.
- Are particulates and visibility changing over time? Particulate and visibility trends are updated to include data since 2004.
- Is precipitation chemistry changing over time?

Data sources and types used in analysis

Ozone

Ozone is currently monitored hourly at two sites in Sequoia as a part of the NPS Air Resources Division network (Table 1): Lower Kaweah (also called Giant Forest, 1902 m) and Ash Mt. (457 m). The nearly 30-year record of hourly ozone monitoring in Sequoia provides data to understand temporal patterns in ozone concentration, but spatial patterns of ozone through the Sierra Nevada have only recently been explored (Bytnerowicz et al. 2003a,b, Cisneros et al. 2010). For this report, a new ozone concentration spatial layer was developed from passive ozone monitor data generously provided by Andrzej Bytnerowicz, USDA-FS, Pacific Southwest Research Station, Riverside, CA. Passive samplers were deployed for two-week periods in 2006, 2007 and 2008 during the highest ozone concentrations period of the year – June through October – to estimate average 24-hour ozone concentrations for each 2-week period. They were located over a broad area of the southern Sierra Nevada, including Sequoia and Kings Canyon National Parks. A description of ozone passive samplers can be found in Bytnerowicz et al. (2003). Ozone injury in *Pinus ponderosa* and *P. jeffreyi* in SEKI was reported across both parks in 1990 (Duriscoe 1990) but has not yet been compared with spatial patterns of ozone concentration. This chapter will analyze recent spatial data and long-term temporal trends in ozone concentration data and revisit forest ozone injury data in light of those analyses.

Table 1. Data for time-series air quality analysis came from networks with long-term monitoring sites in the Parks.

Data Type	Network	Sites in SEKI (monitoring dates)	Website data access
Precipitation Chemistry	NADP-NTN	Giant Forest (1981-)	nadpweb.sws.uiuc.edu/sites/sitemap.asp?state=CA
Mercury in Precipitation	NADP-MDN	Giant Forest (2004-)	nadpweb.sws.uiuc.edu/sites/sitemap.asp?state=CA
Ozone	NPS-ARD	Lower Kaweah (1987-) Ash Mountain (1987-) Lookout Point (1992-2004) Grant Grove (1990-95)	ard-request.air-resource.com
Dry Deposition Chemistry SO₂ HNO₃	CASTNET	Ash Mountain (1997-)	www.epa.gov/castnet/data.html
PM₁₀, PM_{2.5} Aerosols, Visibility	IMPROVE	Ash Mountain (2000-)	vista.cira.colostate.edu/improve/Data/IMPROVE

Nitrogen

Precipitation nitrogen deposition is measured at the Lower Kaweah site in Sequoia as a part of the National Atmospheric Deposition National Trends (NADP/NTN) network. Dry particle nitrogen and nitrogen gas deposition is measured at Ash Mt. as a part of the Clean Air Status and Trends network (CASTNET). Time-series data came from a variety of networks with sites in the parks (Table 1). Spatial nitrogen deposition data were generously provided by Mark Fenn, USDA-FS, Pacific Southwest Research Station, Riverside, CA from spatial data published in Fenn et al. (2010a). To determine spatially explicit nitrogen deposition to major ecosystem types across California, Fenn et al. (2010a) employed a broad suite of techniques and data, including passive air monitors (Bytnerowicz et al. 2001, Bytnerowicz et al. 2003a,b), throughfall resin collectors (Fenn et al. 2010a), inferential deposition calculations, and modeling (Byun and Shere 2006). Simulated deposition data from the USEPA CMAQ (Models-3/Community Multiscale Air Quality) model (Byun and Schere 2006, Tonnesen et al. 2007, fide Fenn et al. 2010a) were used for chaparral and oak woodlands, and for broad scale estimates of deposition to grassland. Throughfall deposition data were used to correct nitrogen deposition in forest and chaparral ecosystems (Fenn et al., 2003a, 2008; Meixner and Fenn, 2004). The CMAQ model is designed to represent both wet and dry deposition of aerosol and gas-phase species. Details of methods for deriving nitrogen estimates are in Fenn et al. (2010a).

Sulfur and particulates

Sulfur in precipitation is measured at Lower Kaweah as a part of the NADP/NTN network, and dry and gaseous sulfur deposition at Ash Mt. for CASTNET. Visibility, and its component particulates, has been measured in Sequoia since 2001 at the Ash Mt. site as a part of the

Interagency Monitoring of Protected Visual Environments (IMPROVE) network². Particles in the 2.5 micron range are collected and speciated to sulfate, nitrate and carbon. Masses are recorded of PM_{2.5} and PM₁₀. The NPS examines the clearest days and haziest days to measure visibility conditions.

Other contaminants

Mercury in precipitation is measured at Lower Kaweah as a part of the NADP Mercury Deposition Network (NADP/MDN). Mercury deposition trend data from this network will be analyzed in this report and compared to results from WACAP. WACAP's efforts in SEKI were focused at Pear/ Emerald Lakes, about 10 km NE of the NADP/MDN site.

² Before 2001, the IMPROVE site was located close to road construction which compromised data, so analysis is done only on data after that time.

Reference conditions

Air quality reference conditions are the background levels of chemical species in air influenced only by local natural sources. These conditions predate any monitoring record, thus must be derived from modeling and comparisons with similar pristine areas.

Natural sources of ozone precursors, NO_x and VOCs, include biogenic emissions, stratospheric incursions, lightning, and wildfire. Appropriate levels to be used for background ozone concentrations have been debated at length, resulting in the concept of Policy Relevant Background (Lefohn et al. 2001, Fiore et al. 2003). The current U.S. Environmental Protection Agency (EPA) baseline background O₃ level used in risk assessment is 40 ppb. Because it is federally recognized, we used this value as a threshold value below which ozone concentrations were categorized as not different than reference levels, or “good” (Figure 28). Recent work has shown that this may be high and background ozone is more likely to be 15-30 ppb, although seasonally and spatially variable (Fiore et al. 2003). Natural sources of nitrogen include bacterial fixation, soil and marine emissions, lightning, and wildfire. Reference conditions of nitrogen deposition in the Sierra Nevada are estimated to be no greater than 1-3 kg/ha/y (Fenn et al. 2010a). Natural sources of sulfur include marine, episodic volcano and wildfire emissions, resulting in an estimated 0.25 kg/ha/y background sulfur deposition (NPS-ARD and USFWS 2002). Natural reference conditions for visibility in SEKI have been estimated by the IMPROVE network at around 7 deciView (dv) for SEKI (NPS-ARD 2009, Fox and Riebau 2009).

While background levels of pollutants can serve as reference conditions for anthropogenic impacts, air quality *standards* serve as reference conditions for which there are legislative mandates and penalties for non-compliance. Criteria pollutants are those pollutants for which the U.S. Environmental Protection Agency (EPA) has established National Ambient Air Quality Standards (NAAQS) as directed by the Clean Air Act. Table 2 lists the criteria pollutants that are measured in SEKI: SO₂, PM₁₀, PM_{2.5}, and ozone, as well as visibility for which there is a state standard, but not a federal standard.

Table 2. Federal National Ambient Air Quality Standards (NAAQS) and California State

Pollutant	Averaging Time	<u>Federal</u>		<u>State</u>
		Primary Standard	Secondary Standard	Primary Standard
Sulfur dioxide	24-hour	--	--	40 ppb
	3-hour	--	500 ppb	--
	1-hour	75 ppb	--	--
PM ₁₀	Annual mean	--	--	20 ug m ⁻³
	24-hour	150 ug m ⁻³	same	50 ug m ⁻³
PM _{2.5}	Annual mean	15 ug m ⁻³	same	12 ug m ⁻³
	24-hour	35 ug m ⁻³	same	--
Ozone	8-hour	75 ppb	same	70 ppb
	1-hour	--	--	90 ppb
Visibility	8-hour	--	--	32 dV

Notes:

Sulfur dioxide: In 2010, the U.S. EPA established a new 1-hour SO₂ standard which is based on the 3-year average of the annual 99th percentile of 1-hour daily maximum

Ozone: The 8-hour standard is the 4th highest maximum daily 8-hour average in a year, averaged over 3 years.

Visibility: Extinction coefficient of 0.23 per kilometer = 32 dV

Source: California Air Resources Board (2010, <http://www.arb.ca.gov/research/aaqs/aaqs.htm>)

Spatial and temporal analyses

In this section patterns and trends in air quality in the region and the Parks are described. The impacts of air quality on resources in the Parks, however, are not discussed here, but rather in a later section, “Interactions with other focal resources”.

Spatial trends

Spatial distribution of ozone and nitrogen across the Parks depends on source strength, transport characteristics, topography, and surface and chemical characteristics influencing deposition. However, generally, ozone and nitrogen concentration have been reported to be highest closest to the SJVAB and decrease with distance eastward and northward across the range (Priesler and Schilling 2003, Frącek 2003, Fenn et al. 2010a). The Parks boundaries serve as the limit of the Parks spatial analysis, and the Protected Area Centered Ecosystem (PACE) boundary defines what is called the “region” for the SEKI NRCA. The region provides the context for the Parks. See the main body of report for discussion of why the PACE boundary was chosen for the region.

Ozone Concentrations

To create a continuous extrapolated ozone surface, average 24-hour ozone concentrations over each 2-week sample period were used as the response variable in a geographically weighted regression with mean temperature, elevation, distance from Fresno, distance to the bottom of the nearest drainage, and maximum normalized wind velocity over each two-week sampling period. Temperature and elevation were highly correlated, so only elevation was kept because of its continuous spatial coverage. The best fit included elevation, distance to drainage bottom and maximum normalized wind velocity. The same equation was applied independently to each 2-week time period, with a resulting R^2 for each regression. The median adjusted R^2 was 45% for all 2-week periods, and the highest frequency of R^2 was in the 50-70% range.

Using the resulting regression equation, a continuous ozone concentration layer was extrapolated to the region for each 2-week period, at 3500m resolution, which is the finest resolution at which error was minimized. Because ozone concentration and distribution are source- and weather-dependent, the magnitude of ozone concentration between time-periods were very different, however the distribution of ozone through each period was generally similar. For the purposes of this report, the average of the summer months for the three years is the best representation of the summer high ozone season to which the PACE region and the Parks are, on average, exposed (Figures 2 and 3).

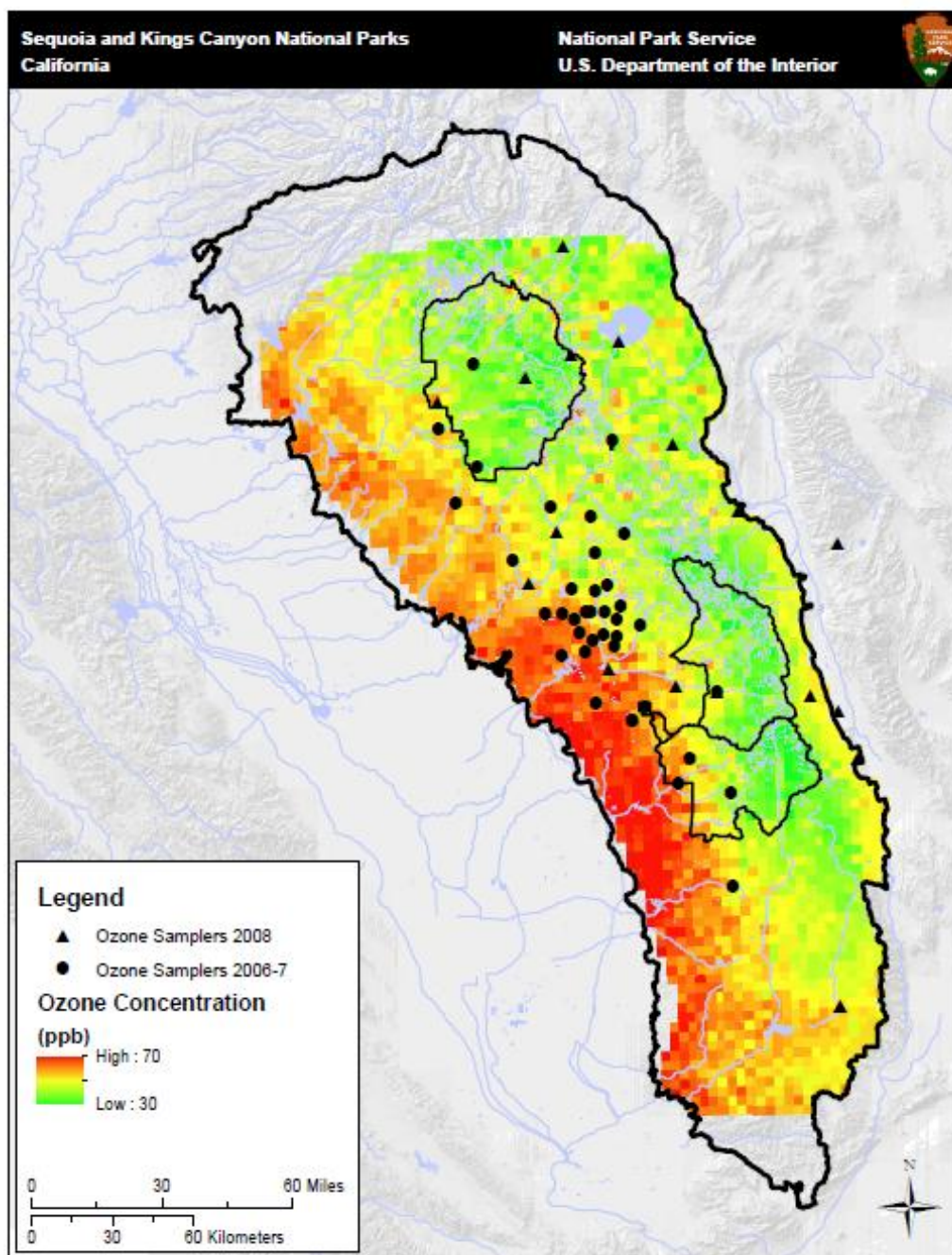


Figure 2. Average ozone concentration for the PACE region, from a geographically weighted regression analysis of passive ozone monitors sampled over the summer and early fall months (June-October) in 2006, 2007 and 2008. Grid size is 3500m. Yosemite, Sequoia and Kings Canyon National Parks boundaries are outlined within the PACE.

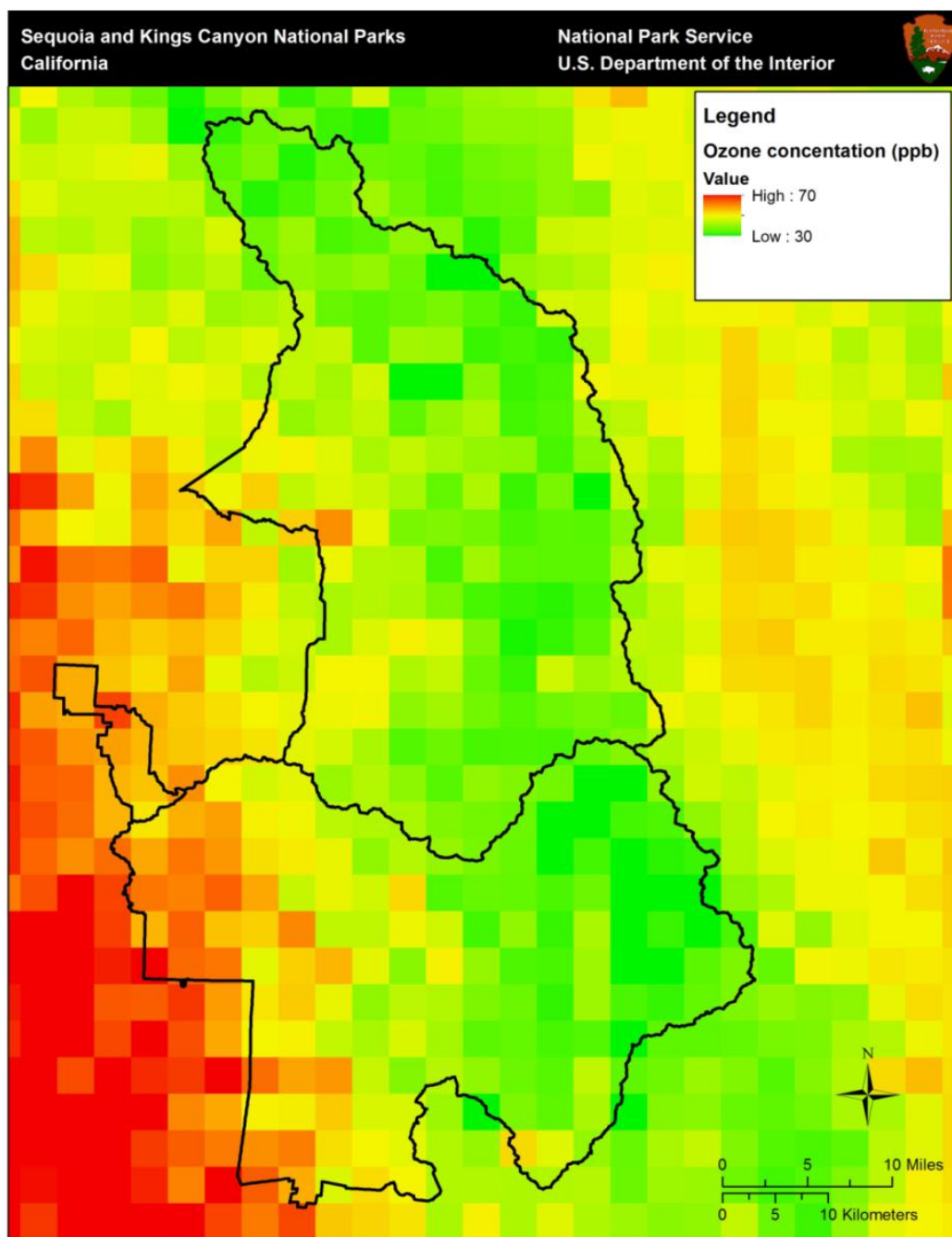


Figure 3. Average ozone concentrations for Sequoia and Kings Canyon, from a geographically weighted regression analysis of passive ozone samplers run continuously for 2-week periods over the summer and early fall months (June-October) in 2006, 2007 and 2008. Grid size is 3500m.

Over the entire PACE region, average June-October ozone concentrations are generally highest on the western side, and specifically to the west of Sequoia and Kings Canyon Parks (Figure 2), showing the influence of the sources in the SJVAB and surface airflow patterns southeastward toward the Parks, as well as the recirculation effect of the Fresno Eddy. The northern portion of the PACE, including Yosemite National Park, is less ozone-impacted. Yosemite is protected

from the SJVAB plume by distance and topography. The higher elevations of the Sierra Nevada contain large areas that are difficult to access, therefore have fewer sampling points. Extrapolation to these areas is less certain, however it is consistent with the observation that higher elevations generally exhibit lower concentrations because ozone is scrubbed out by vegetation as it moves with air flow upward during the day and, at night, downslope flows bring cleaner air from aloft. The eastern slope of the Sierra and the PACE experience incursions of ozone from the Owens Valley, so ozone concentrations are moderately high to the east of the Parks as well. Airflow patterns bring ozone to the Owens Valley through the San Joaquin River drainage to the north (Cisneros et al 2010), and also from Los Angeles, but concentrations are much lower than on the western slopes.

Compared to the PACE region, Sequoia experiences high elevated summer-fall ozone concentrations along the western slopes, especially in the Kaweah watershed (Figure 3). The Kaweah watershed is the region of the Parks in closest proximity to the SJVAB and the Fresno Eddy (Beaver et al. 2010). In the Kaweah watershed, the highest concentrations are generally below 2000-2300 m, with no evident persistence of elevated ozone concentrations into the higher regions of the Parks. This is consistent with Cahill et al. (1996) who showed a sharp reduction in ozone and aerosols between Giant Forest at 6000 feet (1830 m) and Emerald Lake at 10,000 feet (3050 m). This supports the concept of a limited pollutant transport efficiency over the mountains to downwind sites in the central and southern Sierra Nevada. There is also evidence of high ozone concentrations in the mouths of the deeper drainages, particularly the Middle Fork of the Kings River and moderate ozone penetration in to the South Fork.

Nitrogen Deposition

Spatial patterns in total nitrogen deposition for the PACE region (Figure 4) and for Sequoia and Kings Canyon National Parks (Figure 5) were clipped from the data layer described in Fenn et al. (2010).

Overall, total nitrogen deposition in the PACE reflects proximity to sources in the SJVAB on the western side. The foothills receive slightly higher, around 5-10 kg N/ha/y, and the conifer belt above the foothills, with higher leaf area index, receives the highest deposition. Nitrogen deposition declines with distance from the source, so that nitrogen deposition in the central and eastern side of the PACE is low, around 0-5 kg N/ha/y.

All of Kings Canyon and most of Sequoia receives relatively low nitrogen deposition inputs, around 0-5 kg N/ha/yr. The western area of Sequoia, however, is a region of elevated nitrogen deposition, with widespread deposition around 5-10 kg N/ha/y, some areas on the western edge getting up to 15 kg N/ha/y, and two hotspots receiving around 15-20 kg N/ha/y. The higher nitrogen deposition seems to be constrained to an elevational band below 2000-2300 m.

Leaf area is a significant factor in nitrogen deposition, since conifer needles have more surface area than chaparral. However, the conifer zone receives nitrogen deposition as fog as well. CMAQ is poor at modeling fog deposition, so Fenn et al (2010a) used a throughfall correction at sites with fog deposition, including SW Sierra Nevada sites of which Giant Forest is also included.

Broad corroboration of this observed pattern comes from Bytnerowicz et al (2001), who measured pollutants along an elevational gradient in SEKI and showed a linear drop (dilution) of gaseous reactive nitrogen (NH_3 , HNO_3) with increasing elevation up to 2300 m. HNO_3 is a significant portion -- roughly a third -- of total nitrogen (see discussion on page 20).

The results of Fenn et al. (2010a) are, to date, our best estimate of the spatial trends of total nitrogen deposition throughout the Parks. That they match generally with ozone concentration spatial trends provides broad corroboration of a high pollution deposition zone on the western edge of Sequoia Park up to about 2000-2300 m elevation.

Temporal trends

Temporal trends in pollutants are a consequence of a complex mix of the trends in emissions, magnitude of sources, regulatory controls, and changes in weather and airflow patterns that deliver pollutants to the parks.

Ozone

Ozone has been measured hourly at 4 sites in SEKI, but only 2 of those maintain current measurement operations (Table 1): Ash Mt. and Lower Kaweah, both in the Kaweah River drainage on the western slopes of the Sierra. As noted earlier in the chapter, these sites are in the area of the parks that receive the highest ozone concentrations, thus represent among the worst exposure. Other sites with some period of continuous measurement include Grant Grove on the western slopes, and Lookout Point, in the East Fork of the Kaweah watershed.

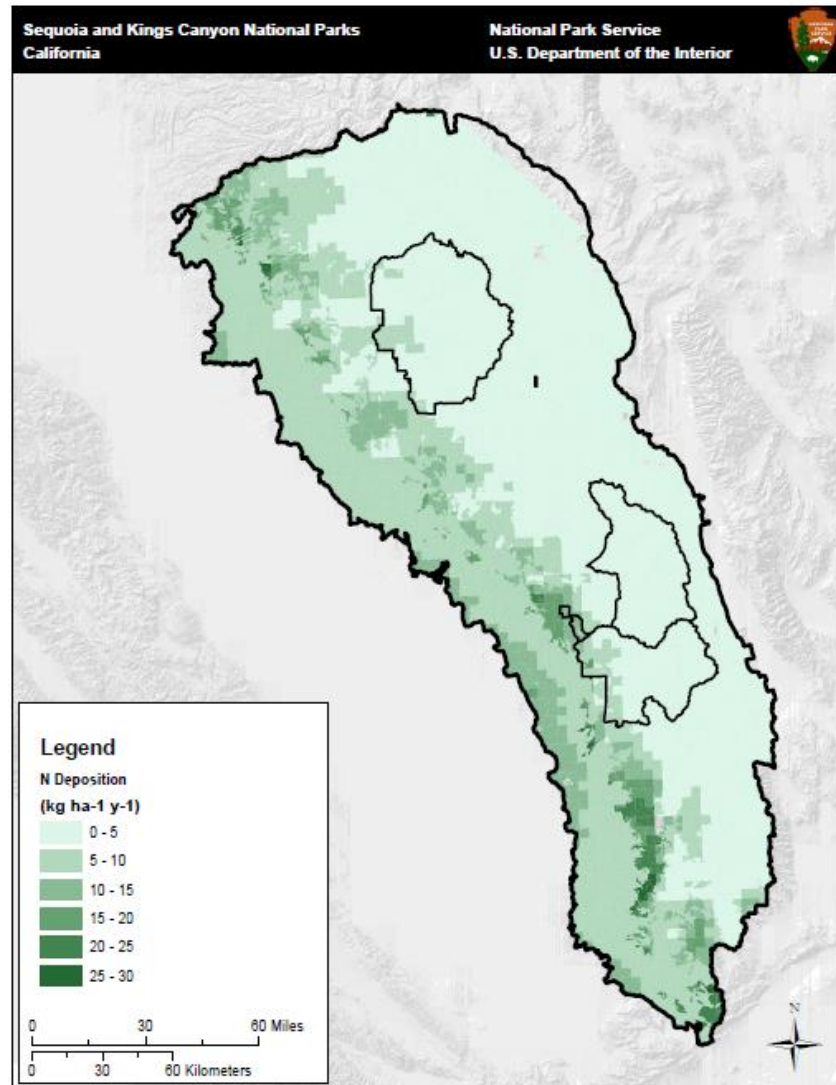


Figure 4. Average total nitrogen deposition (wet+dry, kg N/ha/yr) for the PACE region, data courtesy of Mark Fenn, USDA Forest Service, Riverside, CA. Also in Fenn et al. (2010a).

While annual ozone concentration averages 46.7 and 45.8 ppb, respectively at Ash Mt. and Lower Kaweah from 1987 to 2010 (Figure 6a), average summer ozone (June-September) is much higher (Figure 6b). Neither annually, nor in summer, at either site, is there a linear trend in ozone concentration when mean annual data are plotted against year ($p < 0.05$).

Ozone concentrations change over the day, over the season, and over the landscape, due to the interaction of temperature, light, air flow patterns, and chemistry, leading to times of high human and plant exposure, and times of low exposure. These patterns of ozone concentration, in turn, are critical to understanding impact of ozone exposure on people and vegetation. In people, ozone can exacerbate asthma and cause respiratory stress. In vegetation, ozone enters leaves through stomatal pores that open and close in response to environmental cues like light and moisture, and can disrupt important plant physiological processes. Greatest ozone impact occurs when ozone concentrations are high and physiological activity is high, leading to maximum ozone uptake. Because ozone formation is temperature dependent, a typical seasonal pattern of ozone concentration affecting the general SEKI region shows high concentrations in the summer, peaking in July and August (Figure 7).

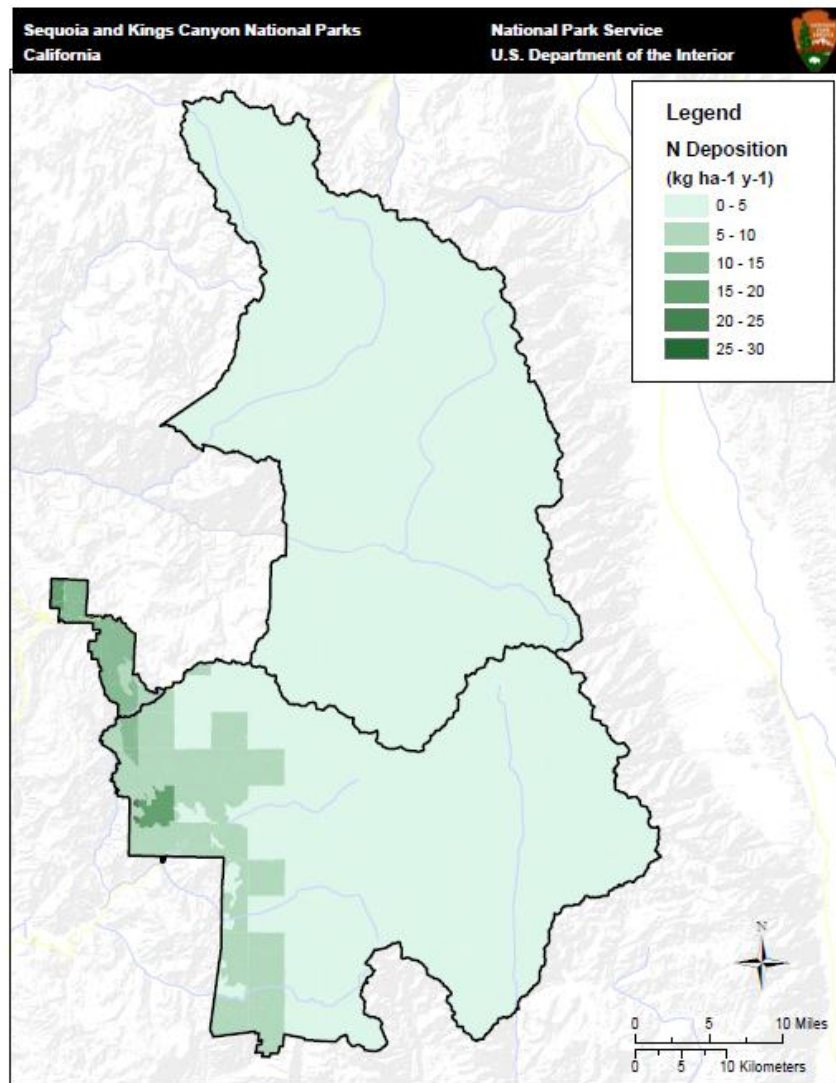


Figure 5. Average total nitrogen deposition (wet+dry, kg N/ha/yr) in Sequoia and Kings Canyon Parks, showing highest deposition along the western edge of Sequoia, closest to sources in the San Joaquin Valley air basin. Data courtesy of Mark Fenn, USDA Forest Service, Riverside, CA. Also in Fenn et al (2010a).

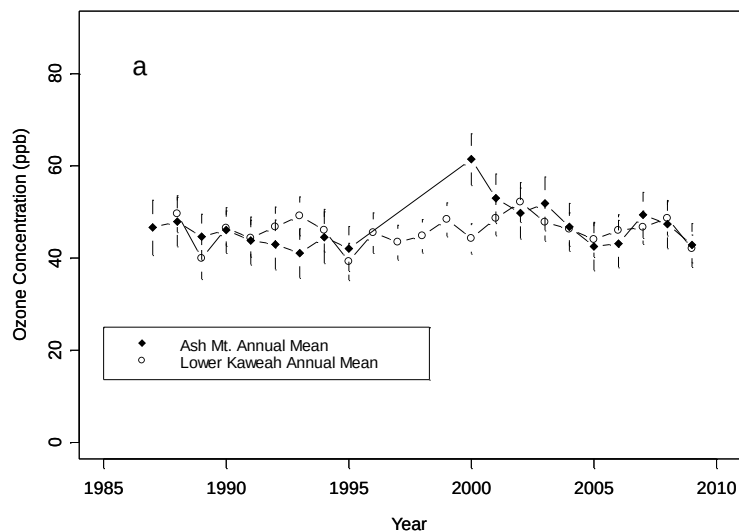


Figure 6a. Annual mean ozone concentration at the 2 long-term continuous monitoring sites in Sequoia and Kings Canyon Parks. Data from NPS-ARD.

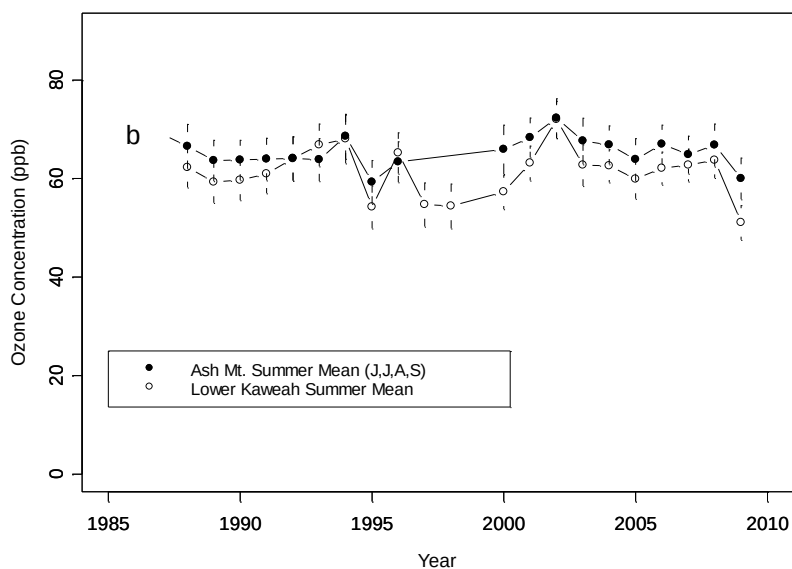


Figure 6b. Mean summer ozone concentration at the 2 long-term continuous monitoring sites in Sequoia and Kings Canyon Parks. Summer months are June, July, August, September. Data from NPS-ARD.

Diurnal patterns of ozone concentration are more variable, and depend on elevation (Van Ooy and Carroll 1995), location relative to the SJVAB plume (Beaver et al 2010, Cisneros et al. 2010), and daily patterns of upslope and downslope air movement. Pronounced diurnal patterns are generally observed closer to drainage bottoms and the SJVAB plume, while flatter diurnal patterns are usually seen at higher elevations (Priesler et al. 2003) and greater distance from sources. Seasonal and diurnal patterns in ozone matter because of their interaction with plant physiological activity, which can have a different seasonal or diurnal cycle than ozone. Diurnal patterns of ozone at SEKI for July-August are shown in Figure 8. Peak ozone concentration occurs at 4-5p. The combined influence of the source plume and elevation can be seen in the similar shape of the diurnals at each site, with a slightly flatter peak at higher elevation between sites of 1445 m. Also note that the diurnal peaks later in the afternoon at Lower Kaweah, showing the lag as the plume moves eastward and upslope.

So, while maximum ozone concentrations occur generally late afternoon in mid- to late summer, this may not correspond to maximum ozone uptake. The prolonged California summer dry season (little rain from June through October) can cause vegetation to shut stomata progressively earlier in the day throughout the season, until by September or October they may be shut most of the day (Panek 2004, Emberson et al. 2000a,b). This means that vegetation may not be taking up ozone in the afternoon or late in the season or both (Figure 9), and so ozone *flux* into vegetation (shown as red triangles) may be low even if ozone *concentrations* (shown as a black line) are high (Figure 9,

from Panek and Ustin 2004; see also Emberson et al. 2000a,b, Panek and Goldstein 2001, Panek 2004). Maximum ozone uptake in *P. ponderosa* occurred in mid-April in Kings Canyon (Figure 9).

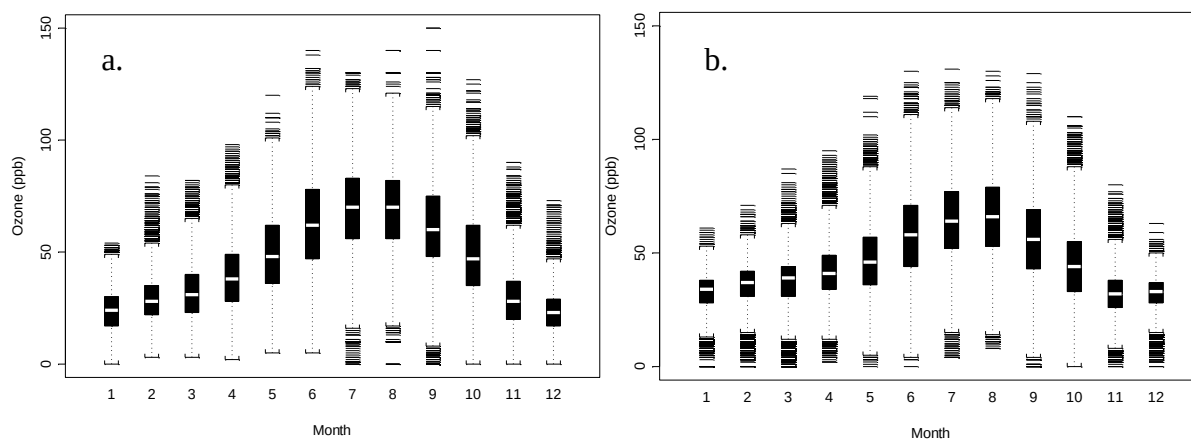


Figure 7. Boxplots of hourly ozone concentrations for the period of record by month, at (a) Ash Mt. and (b) Lower Kaweah from 1987 – 2010, showing a seasonal pattern peaking in July and August. Median, quartiles, and outliers are shown for each month.

In summary, the relationship between measured ambient ozone concentration and plant ozone injury is complex and not yet well-characterized, but is a result of the combined effects of ozone concentration and exposure duration, plant susceptibility, and environmental factors that favor plant ozone uptake, such as moisture and temperature (Panek 2004, Smith et al. 2008).

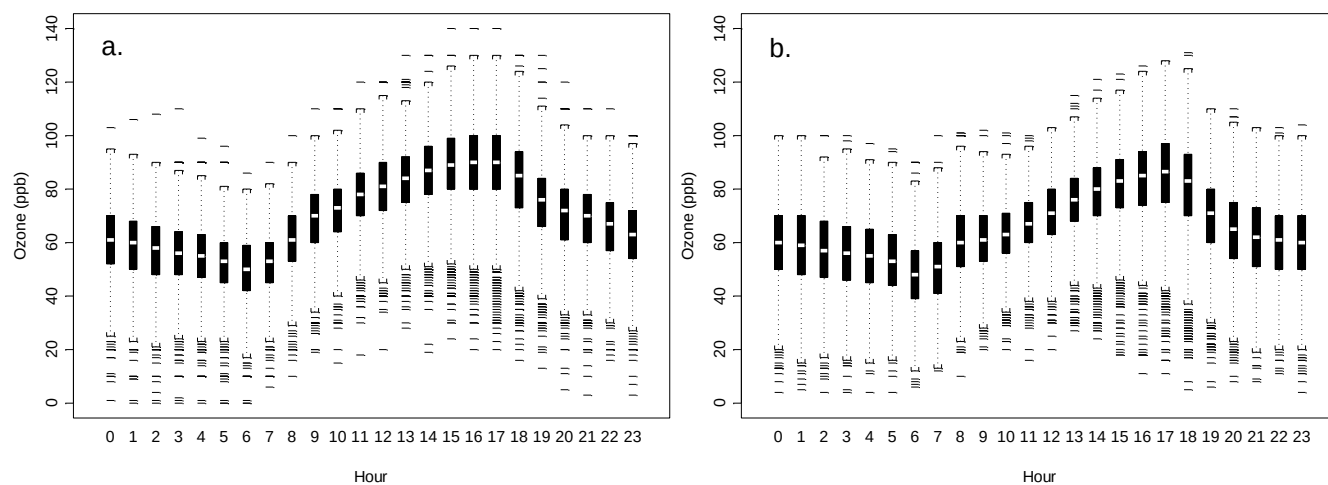


Figure 8. Hourly ozone concentrations in July and August, at Ash Mt.(a) and Lower Kaweah (b) from 1987 – 2010, showing diurnal pattern peaking around 3-5 pm at Ash Mt. and around 5-6 pm at Lower Kaweah. Median, quartiles, and outliers are shown for each hour.

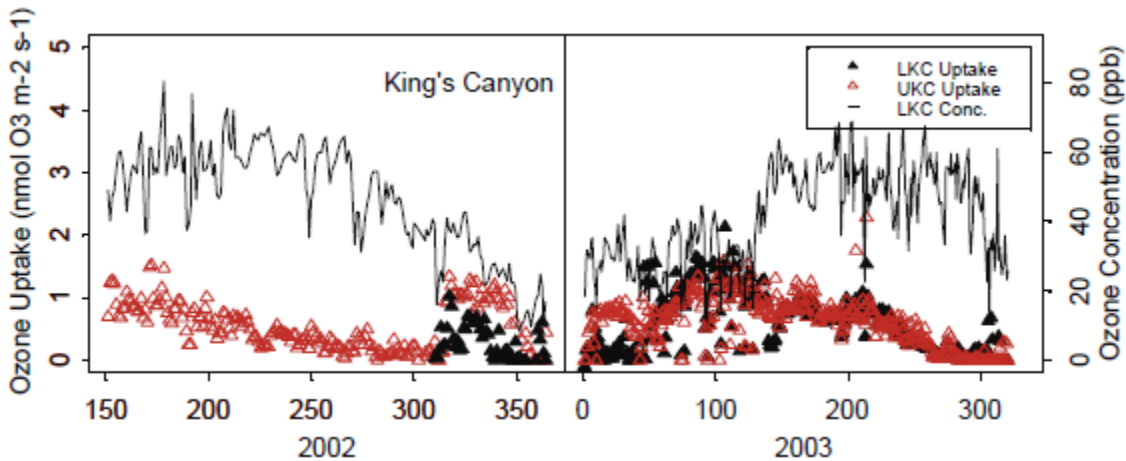


Figure 9. Mean mid-day (10:00 – 14:00 PST) ozone uptake and concentration trends plotted against Day of Year, over a 1.5-year period in 2002-2003 in Kings Canyon. Note the high ozone concentrations over the summer months (Days 152-275), but the decreasing trends in ozone uptake in response to decreasing stomatal conductance. LKC= Lower King's Canyon, UKC = Upper King's Canyon (Panek and Ustin 2004).

Ozone concentrations relative to reference conditions

In 2008, the U.S. EPA significantly strengthened its primary (human health) national ambient air quality standards for ground-level ozone to 75 ppb, as well as establishing a secondary standard (welfare, including vegetation) equal to the primary standard. The California primary eight-hour standard since 2005 is yet more rigorous, at 70 ppb. In both cases, the standard is the annual 4th-highest daily maximum 8-hour ozone concentration averaged over 3 years. In 2009, however, NPS chose to evaluate trends in ozone concentrations using annual 8-hr metrics, instead of the 3-year average, as the annual statistic is available over a longer period (NPS-ARD 2009), thus this is what is reported here.

Despite state and federal regulation, limited improvement in O_3 has been achieved over the last decade, due to continued population growth in the source areas (Figure 26c). Ozone exceeds the federal and state primary and secondary standard at all locations where it is routinely monitored in SEKI. Plotted annually at the 4 sites with continuous ozone concentration data, Federal and State 8-hour ozone standards have been violated at all sites since 1990 (Figure 10).

Using policy relevant background levels as a reference condition, that is an annual average of 40 ppb, and comparing to an average of SEKI's longest record, Ash. Mt. (Figure 6a), SEKI is on average 7 ppb over background levels, and has been as much as 20 ppb over background in the last decade of record.

Other indices designed to relate ambient ozone concentrations to plant exposure and injury include SUM0, SUM60, W126, and AOT 40 (see review by Musselman et al. 2006). All of them are derived from hourly concentration data. In January 2010 the USEPA proposed the W126 as a new secondary ozone standard, which uses a sigmoidal function to assign higher weight to higher concentrations of ozone. The rationale is that higher exposures of ozone are non-linearly more damaging to vegetation than lower. All sites with continuous measurements in SEKI exceed the proposed W126 standard of 7-15ppb-hrs (data not shown). The spatial nature

of this report precludes the use of any of the plant exposure indices for spatial reference conditions, because passive ozone data don't provide the necessary hourly resolution.

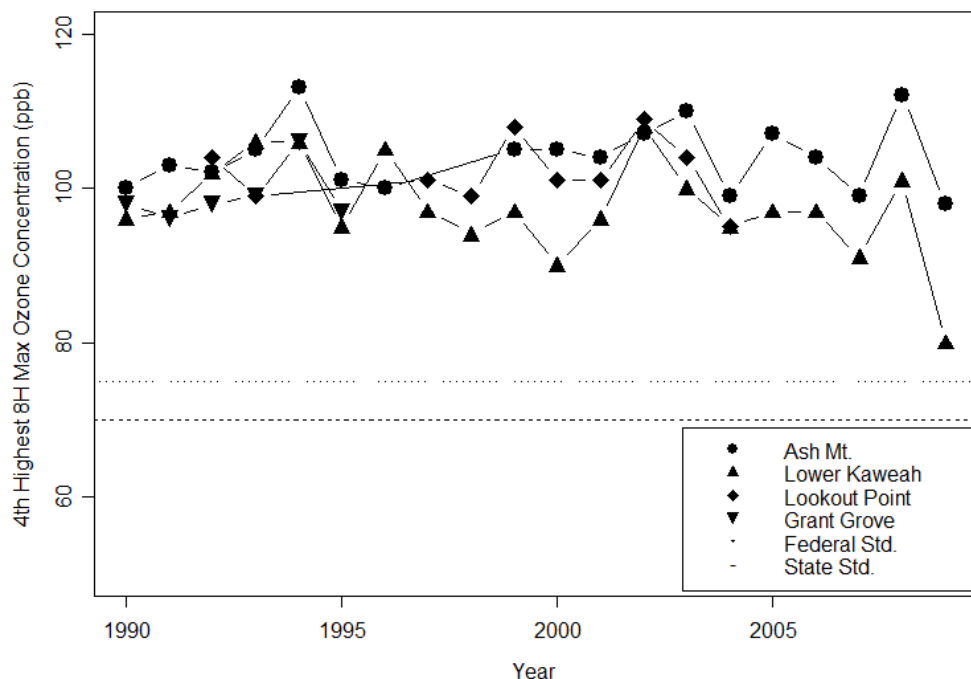


Figure 10. NAAQS 8-hour ozone standard at sites which have had more than 5 years continuous monitoring in Sequoia and Kings Canyon Parks. The Federal primary and secondary (75 ppb) and state primary (70 ppb) 8-hour standards are shown as stippled lines. Data from NPS-ARD.

Wet Deposition Chemistry

Precipitation volume and chemistry are continuously measured in the Kaweah watershed at the same site as ozone, and have been measured since 1981. Most precipitation falls between October and May because of California's Mediterranean climate. The record of precipitation monitoring allows for assessment of long-term trends in wet deposition inputs into the Park. NADP annual deposition data using water years (October – September) was analyzed to look for linear trends (Table 3 and Appendix 1). Deposition data, not concentration data, is analyzed because the focus of this report is pollutant impacts on ecosystems rather than magnitude of -- and trends in -- pollutant sources.

Of the chemical species in wet precipitation, only NH_4^+ shows a significant positive trend over the measurement period, although NO_3^- is not declining to reflect California source emission reductions (CARB 2009). The major sources of atmospheric NH_4^+ , NO_3^- and their precursors (NH_3 and NO_x) in California are livestock waste (80% of NH_3 statewide emissions) and mobile source emissions (86% of NO_x statewide emissions) (CARB 2009).

Chemical species showing decreasing trends are SO_4^{2-} , Total Base Cations, Mg, Na, and Cl.

Table 3. Linear trends in wet deposition chemistry since 1981 (kg/ha/y) using NADP annual deposition data and water year (October – September). If trends are significantly different than 0 ($p < 0.10$), then trendline information is shown.

Species	Annual Deposition (kg/ha) (SD)	p-value of trendline (direction of trend +/-)	Trendline ¹
NO ₃	4.49 (1.36)	0.51	
NH ₄	1.80 (0.73)	0.07 (+)	y= 1.4+0.03(x)
Inorganic N	2.42 (.84)	0.33	
SO ₄	2.62 (1.28)	0.003 (-)§	Log(y)=1.2-0.03(x)
H	0.033 (0.02)	0.25 §	
Base Cations	1.38 (0.50)	0.04 (-)	y=1.7-0.02(x)
Ca	0.44 (0.17)	0.36	
Mg	0.11 (0.06)	0.0009 (-)§	Log(y)=-1.8-0.036 (x)
K	0.18 (0.08)	0.19§	
Na	0.65 (0.29)	0.04 (-)	y=0.83-0.01(x)
Cl	1.17 (0.92)	0.03 (-)§	Log(y)=0.36-0.027(x)
NH ₄ x NO ₃		p=0.00, R ² =0.64	NH ₄ = 1.81+1.48(NO ₃)

1. Trendline format: y is chemical species deposition (kg/ha/y), x is number of years since monitoring began

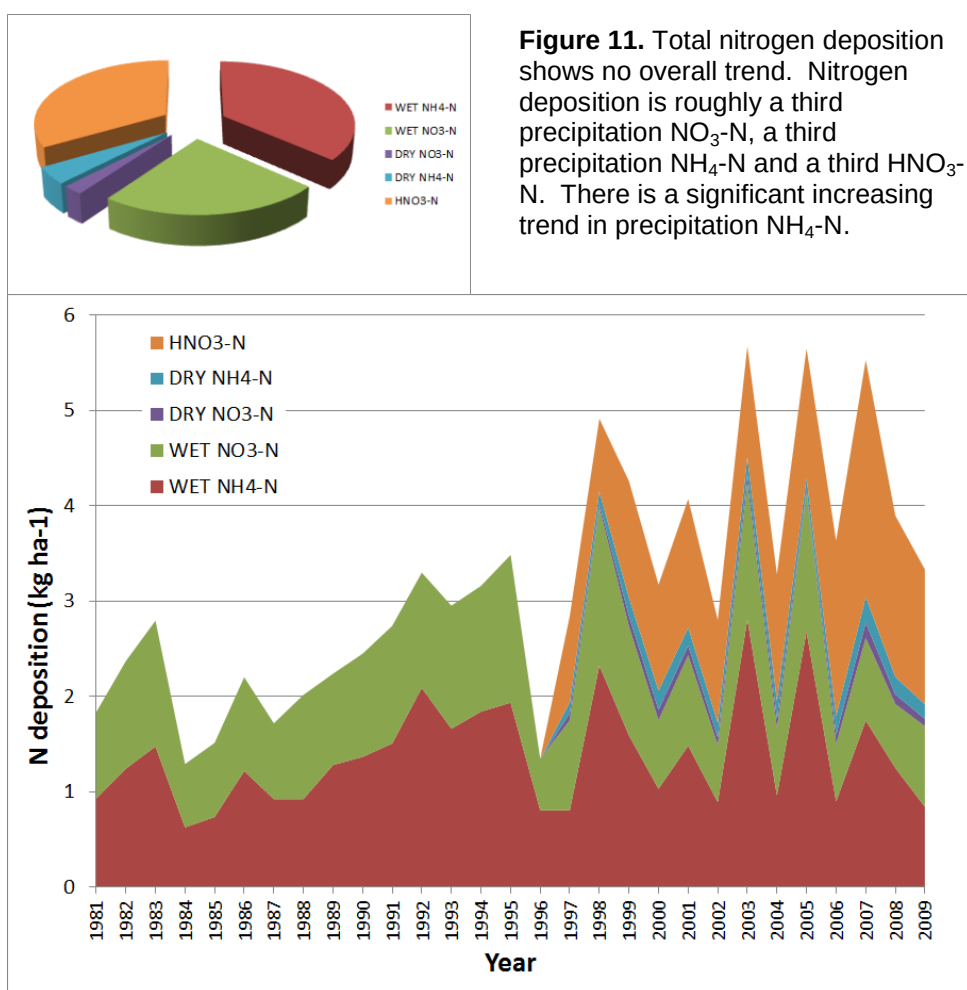
§ Deposition data log-transformed to satisfy Shapiro-Wilk normality test ($p=0.01$)

Deposition was summed from both NADP wet deposition and CASTNET dry deposition networks to estimate total deposition annually over the past 28 years. Dry deposition is derived from air particle concentrations measured at the Ash Mt. site and dry deposition velocities calculated from CASTNET's Multi-Layer Model (MLM) using meteorological, vegetation, and land use data from the site (NPS-ARD 2006).

Resulting estimated total deposition is based on available data at these sites, but probably under-estimates total deposition, because 1) dry deposition was under-sampled from 1996-2002 (data not shown); 2) dry and gaseous species scavenged from the air by trees, then deposited to ground as throughfall is not measured; and 3) dry deposition velocities are derived from Ash Mt. site variables, where vegetation is sparse and leaf area low.

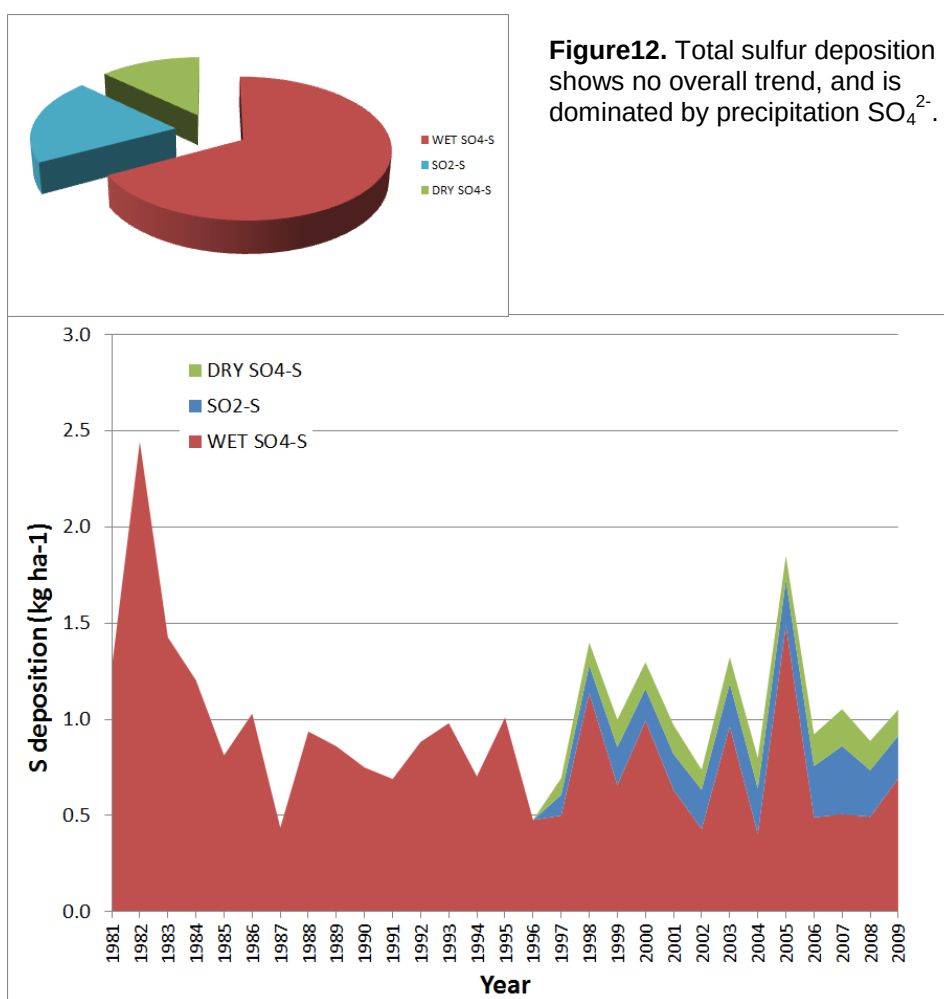
Total Deposition-Nitrogen

Wet nitrogen deposition was divided roughly evenly between $\text{NO}_3\text{-N}$ and $\text{NH}_4\text{-N}$ and generally rose from 1981 until 1995 because of increasing precipitation $\text{NH}_4\text{-N}$, then varied widely from year to year after that (Figure 11). $\text{HNO}_3\text{-N}$ deposition is at least equivalent to both $\text{NO}_3\text{-N}$ and $\text{NH}_4\text{-N}$ deposition, but might be more, so the three species together are responsible for the bulk of N deposition in the Parks. Total nitrogen deposition can exceed 5 kg/ha/y, but not every year, since there is significant variability from year to year. This value is slightly lower than estimated by Fenn et al. (2010a) for this site (5-10 kg/ha/y), which is consistent, as their estimate also includes throughfall. Fenn et al. (2000) estimated that throughfall could account for 66% of total N deposition at a low N deposition site in the San Bernardino Mts., CA. A 5 kg/ha annual deposition rate reaches the critical load thresholds for sensitive lichen communities in chaparral and mixed conifer forests.



Total Deposition-Sulfur

S deposition is dominated by wet $\text{SO}_4\text{-S}$, which comprises roughly 2/3 of all sulfur deposition. SO_2 and particle SO_4^{2-} contribute roughly equal amounts to the remaining third (Figure 12). Total sulfur deposition is variable from year to year, but can exceed 1.8 kg/ha/y, or roughly 1.5 kg/ha/y above background. No significant linear trend exists in total sulfur deposition ($p=0.8$ on data log-transformed to satisfy normality criteria), although the first 4 years of the 1980's saw high sulfur deposition and 1982 was the highest on record. It is possible that California's strict diesel-fuels regulations, adopted in 1988 and resulting in an 82% reduction in SO_2 emissions from diesel, contributed to the decline in sulfur deposition (CARB 2000). As adverse vegetation effects are considered unlikely to occur at rates less than about 5 kg/ha/yr (Peterson 1992c), vegetation is unlikely to be impacted at SEKI. Potential acidification of high elevation lakes is addressed in the Water Quality chapter.



Mercury in Precipitation

Mercury in precipitation has been measured by the NADP-MDN network since 2003. There has been no significant trend in mercury concentration or mercury deposition since then, however mercury values are high compared with other western sites (Figure 13).

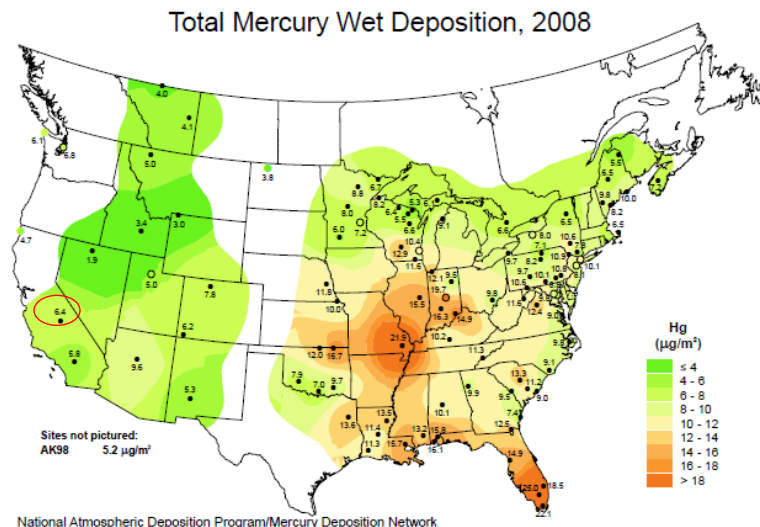
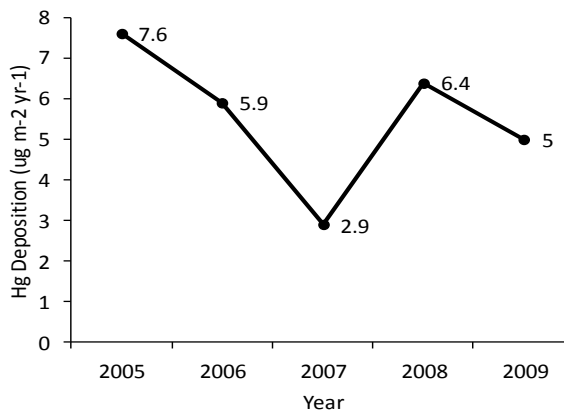


Figure 13. Mercury deposition at Sequoia National Park is generally high compared to other western U.S. sites where it has been measured (red circle). NADP-MDN.

The NADP network has shown mercury deposition to be much higher than measured in WACAP. In comparison to WACAP snow data, in which SEKI reported $\sim 1600 \text{ ng/m}^2/\text{y}$, NADP data show that SEKI has up to $7,600 \text{ ng/m}^2/\text{y}$ Hg annual deposition (Figure 14, see especially year 2005).

Figure 14. Mercury deposition shows no overall trend, and is almost five times higher than reported in the WACAP report (Landers et al. 2008).



Because warm-season deposition was not measured in the WACAP study, total annual deposition of mercury to the parks is higher than the snow fluxes reported there. Clearly there is significant mercury deposition outside of the time period measured by WACAP. Seasonality to mercury deposition is variable from year to year, but much of it can deposit in spring and fall (Figure 15). Mercury dry deposition, including deposition to leaf surfaces and subsequent deposition via throughfall and litterfall is a missed pathway that may bring significant quantities of Hg into the ecosystem. CMAQ modeling efforts suggest that dry deposition of mercury in the SEKI region could be 20-30% of total deposition (Schmeltz 2011).

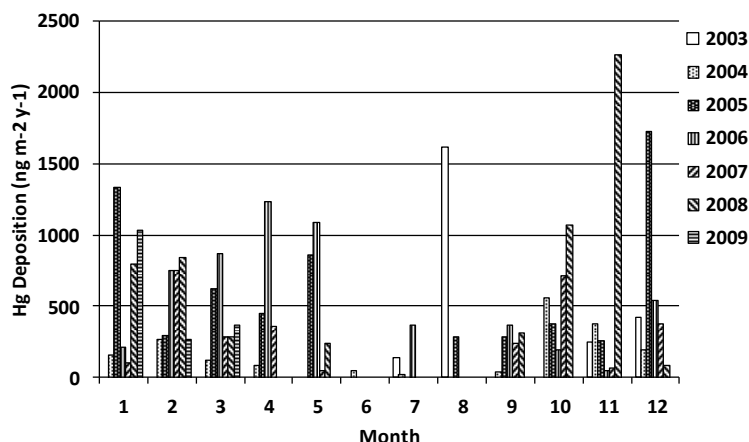


Figure 15. Wet mercury deposition occurs all year, with significant deposition in non-winter months.

Visibility and particulates

Following the method of Suarez-Murias and Zulawnick (2008), we analyzed the IMPROVE (Interagency Monitoring of Protected Visual Environments) visibility and particulate data by determining the worst 20% of visibility days between 2001 and 2009 and identifying the causal particulate species. Values before 2001 were not included because of sensor placement near construction and roads.

The threshold for inclusion in the worst 20% of visibility days was a light extinction of 71.5 mm^{-1} . Like Suarez-Murias and Zulawnick (2008), we found that highest total light extinction (ie, worst visibility) was in wet winter months. The worst months are February and November (Figure 16), and visibility degradation is driven by nitrate (Figure 17). Nitrate accounts for 45% of visibility impairment, on average, on the worst visibility days. Elevated organic carbon drives extinction when nitrates are very low (28% of visibility loss). Sulfates fluctuate, but are generally higher in spring and summer, and account for only a few worst days in April through July, on average 14% of visibility impairment. Contrast this with eastern Parks, where sulfate particles account for 60-85% of the visibility impairment. The worst years measured were the last years of the data record, 2007 and 2008 (Figure 18).

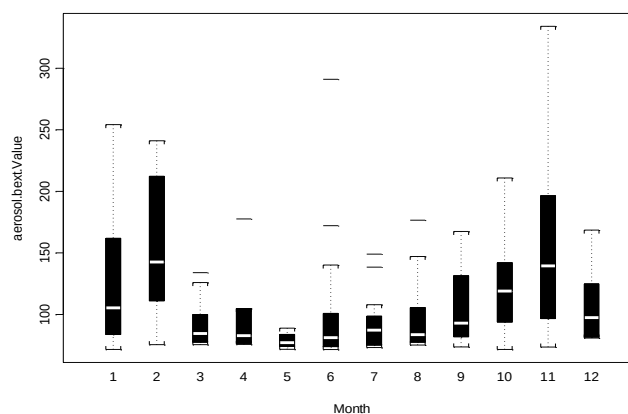


Figure 16. On average, the highest aerosol extinction is in February and November.

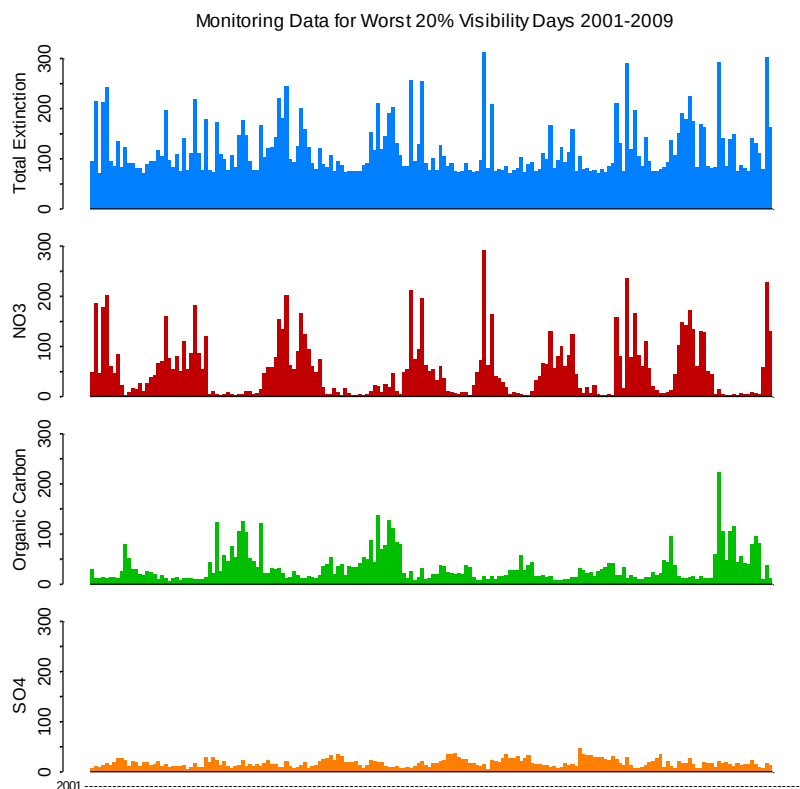


Figure 17. Nitrate accounts for 45% of visibility impairment, on average, on the 20% worst visibility days. Elevated organic carbon drives extinction when nitrates are very low (28% of visibility loss). Sulfates fluctuate, but are generally higher in spring and summer, and account for only a few worst days in April through July, on average 14% of visibility impairment.

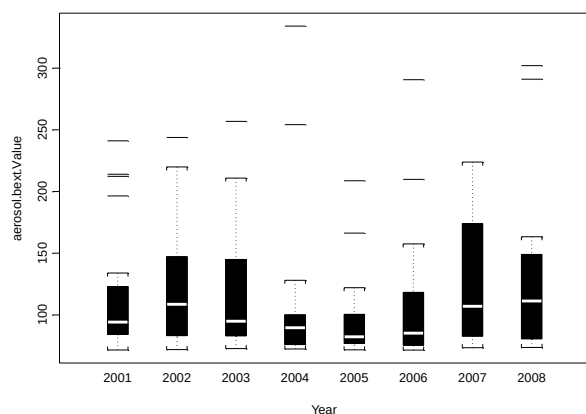


Figure 18. The last 2 years of data, 2007 and 2008, show the highest level of aerosol extinction.

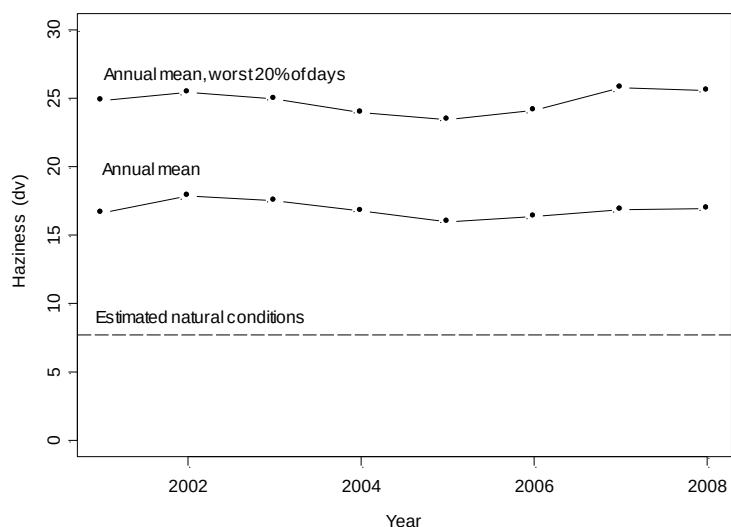


Figure 19. On average, haziness is roughly double natural conditions. On the worst 20% of visibility days, visibility impairment is more than three times higher than natural conditions. Natural conditions are defined as 7 dV (NPS-ARD 2009).

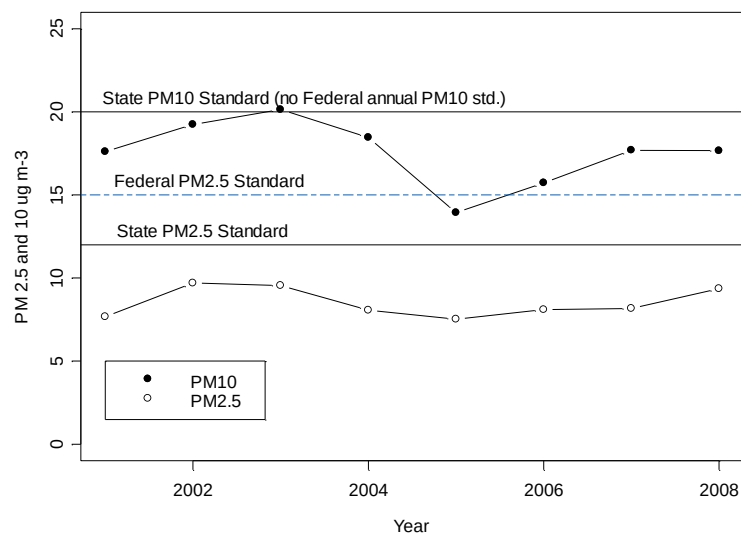


Figure 20. Fine particulate matter (PM2.5) and coarse particulate matter (PM10) show no significant trend, and are below the federal and state standards.

Other Contaminants

Pesticides are generally transported via upslope air movement to more remote regions of Sequoia National Park (Sullivan et al. 2001). Close to the SJVAB, pesticide concentration generally decreases with elevation up to 533 m elevation and then levels off up to 1,920 m (LeNoir et al. 1999). Across the broad high elevations of SEKI, Bradford et al. (2011) found up to 15 chemicals, including current- and historic-use pesticides, in sediment and tadpoles in 28 sampled ponds, as well as endosulfans in the air. Concentrations were very low and chemical compositions were generally similar among sites, and not correlated with distance from SJVAB, suggesting that chemical transport processes did not strongly influence concentration differences among sites.

WACAP (Western Airborne Contaminants Assessment Program)

The WACAP study was a 3-year investigation into contaminant SOCs (semi-volatile organic compounds such as pesticides), mercury, and nutrients in air, vegetation, snow and lake

sediments and fish. While the area studied in SEKI was small, the results were important. SEKI had among the highest concentrations for current-use pesticides, compared with other parks. The sources of these compounds are most likely regional agriculture close to the park. All results reported here are from the WACAP report (2010). A more comprehensive study of air, snow, vegetation, and lakes in SEKI is warranted based on these results.

Current-use pesticides and historic-use pesticides, some of which have been banned since the early 1970's (HCB, a-HCH, dieldrin, DDT, PCBs), and PAHs were detected in the air. Most concentrations in air ranked high relative to those in other parks. Concentrations of many current-use pesticides and historic-use pesticides were high in snow, producing high snow deposition fluxes. In contrast, mercury concentrations in the snow were generally low. SOC and mercury concentrations in SEKI vegetation were in the median to highest ranges among the parks, attributable partly to intensive regional agriculture. Concentrations of many SOCs in lichens increased with elevation. Pesticides scrubbed from the air by vegetation probably contribute significant contaminant loads to the ecosystem via canopy throughfall and litterfall.

Although this chapter focuses on vegetation effects, it is worth noting that current-use SOC concentrations in SEKI fish were among the highest measured in any park. Dieldrin, mercury and DDT levels in fish exceeded thresholds for piscivores, including humans.

In summary, the agricultural region upwind of SEKI is a source of contaminants, including current and historic-use pesticides, for the Parks. These are carried by wind patterns upslope to high elevations to end up in vegetation, snow and lakes. Since Sullivan et al. (2001) and LeNoir et al. (1999) suggest that greater deposition occurs at lower elevations, it is likely that significant but unmeasured deposition of contaminants is occurring at lower elevations in the Kaweah drainage.

Analysis Uncertainty

The spatial ozone concentration data are interpolated from widely spaced passive ozone monitors. Uncertainty is associated with several steps of the analysis. Uncertainty in passive ozone measurement in general in the Sierra Nevada (Lee 2003), described as the mean absolute percent difference between ozone measured with passive samplers and continuous monitors, was

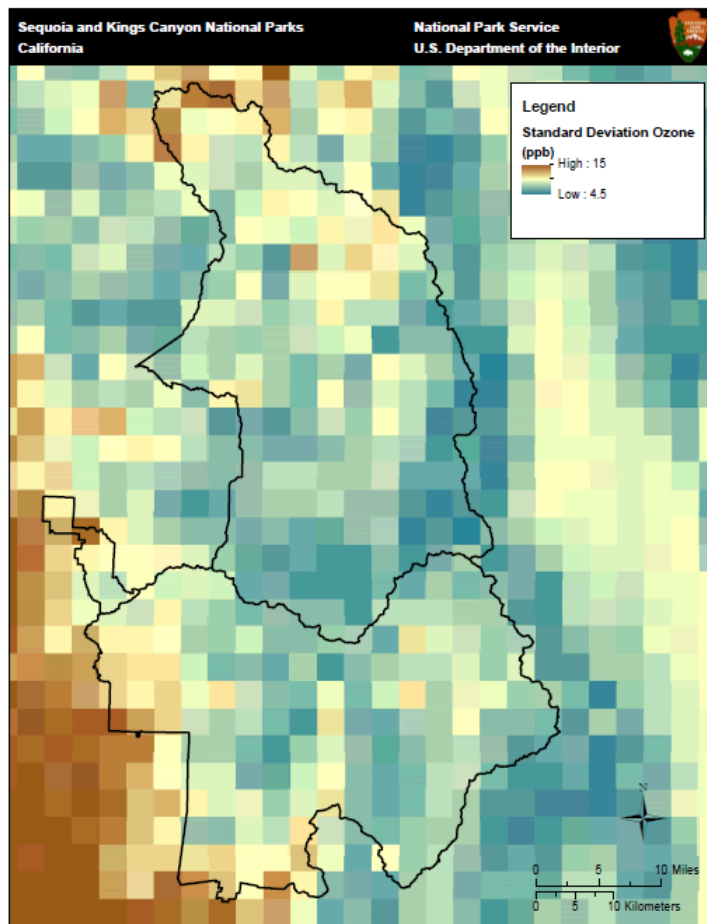


Figure 21. Standard deviation of ozone concentration measured biweekly from June through October, averaged over all sites and all years 2006-2008.

reported at 7.6%. When averaged over the season as we did, it dropped to 5.8%. This describes the uncertainty in measuring ozone concentration with passives. The geographically weighted regression (GWR) brings in another element of uncertainty. The 3500m cell size of the analysis was chosen for two reasons. First, it maximized the fit of the regression, and it was the scale at which previous studies minimized the error in their interpolation (Frączek 2003). The best fit included elevation, distance to drainage bottom and maximum wind velocity. This model had a median adjusted R^2 of 45% for the population of all 2-week passive ozone measurement periods over the three years. The highest frequency of R^2 was in the 50-70% range. The standard deviation of the mean of all 2-week measurement periods is shown in Figure 21. The greater the ozone concentrations, the greater the standard deviation. The area along the western edge of Sequoia shows the greatest variation between measurement periods.

The spatial nitrogen deposition data are modeled using CMAQ. Fenn et al (2010a) compared throughfall N deposition at 26 forested sites in the Sierra Nevada and San Bernardino Mountains with CMAQ estimates and found reasonable agreement when throughfall deposition was <6-7 kg ha/y, but as nitrogen deposition increased CMAQ underestimated nitrogen deposition. There was, however, a significant linear relationship between CMAQ deposition and throughfall nitrogen deposition ($y = -12.20 + 4.34x$; $R^2=0.80$). Ground-truthing of the CMAQ estimates is still required in chaparral, but preliminary work shows the Ash Mt. and Chamise Creek area get around 7-10 kg ha/y (J. Sickman and M. Fenn, preliminary data, fide Fenn, *pers. comm.*)

The corrected CMAQ estimate of nitrogen deposition by Fenn et al (2010a) are our current best estimate of the spatial distribution of nitrogen deposition, but verification through on-the-ground measurements is still needed throughout the Parks. Comparisons of individual study estimates of nitrogen deposition, such as those at Emerald Lake by Sickman et al. (2003), would provide an independent test of model output. Until then, the results of the Fenn et al (2010a) work should be taken as a good first estimate of the spatial trends of nitrogen deposition in the Parks. That they match generally with ozone concentration spatial trends provides broad corroboration of a high pollution deposition zone on the western edge of Sequoia Park up to about 2300 m elevation.

Interactions with other focal resources

Poor air quality can impact other SEKI resources, in some cases described specifically as Air Quality Related Values (AQRVs). For SEKI, AQRVs include ozone impacts on vegetation, such as chlorotic mottle, or decline of ozone-sensitive species; nitrogen AQRVs include fertilization effects, as previously discussed, changes in terrestrial and aquatic community structure, and soil and lake acidification. Particulate AQRVs include visibility impairment. The only AQRVs that can be spatially characterized in SEKI with the available data are 1) chlorotic mottling on pine foliage due to ozone injury, and 2) critical loads of nitrogen deposition.

Ozone injury in pines

Ozone impacts on pine include foliar damage manifest as a characteristic yellowing called chlorotic mottle. Chlorotic mottle has been observed on sensitive pines (*Pinus jeffreyi* and *P. ponderosa*) in the Sierra and Sequoia National Forests and, to a lesser extent, in the Stanislaus, Eldorado, and Tahoe National Forests (Arbaugh et al. 1998, Duriscoe 1990, Miller 1996, Peterson et al. 1995, Pronos et al. 1978, Campbell et al. 2000).

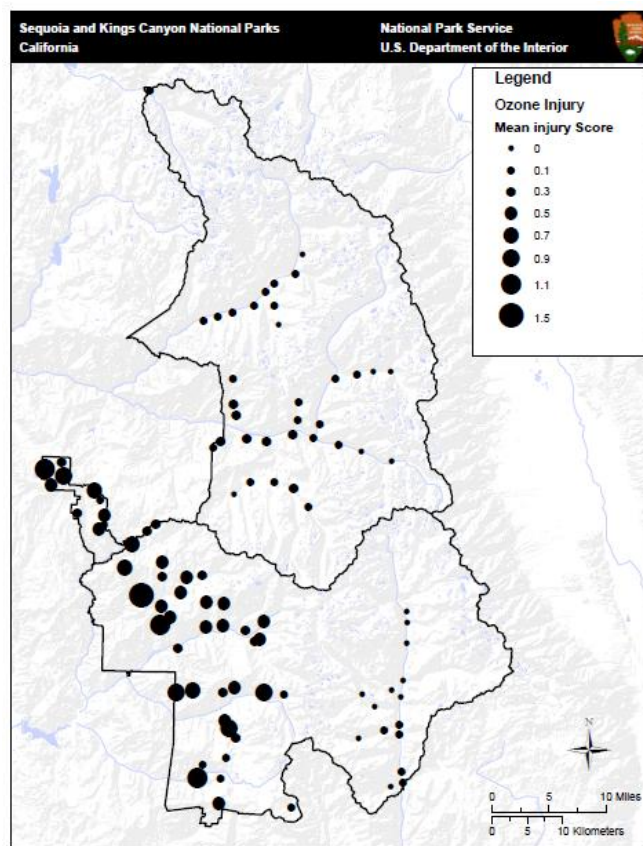


Figure 22. Ozone injury in *Pinus jeffreyi* and *P. ponderosa* (OII, ozone injury index) from Duriscoe ozone injury survey in Sequoia and Kings Canyon Parks in 1985 (Redrawn from Duriscoe 1990).

While patterns of ozone injury have been documented in Sequoia, Kings Canyon, Saguaro and Yosemite National Parks (Duriscoe 1990), these patterns have never been quantitatively compared with spatial patterns of ozone concentration over the same area. The spatial ozone patterns generated for this report allow for this comparison in Sequoia and Kings Canyon National Parks for the first time. When the results of the Duriscoe (1990) ozone injury survey (Figure 22) are overlaid with the spatial pattern of ozone concentration from the geographically weighted regression model (Figure 3), we get a model that explains 48 % of the variation (Figure 23). Given that 1) the ozone injury survey was done in 1980-1985, twenty-five years before the passive ozone monitor work, and 2) the ozone concentration surface is a mean of three years of summertime passive data, and 3) the ozone data represent ambient ozone concentration and not ozone uptake, that it explains as much variation as 48% is remarkable. Clearly more work is needed to explore the details of the broad landscape patterns of ozone concentration dynamics affecting ozone injury in pines.

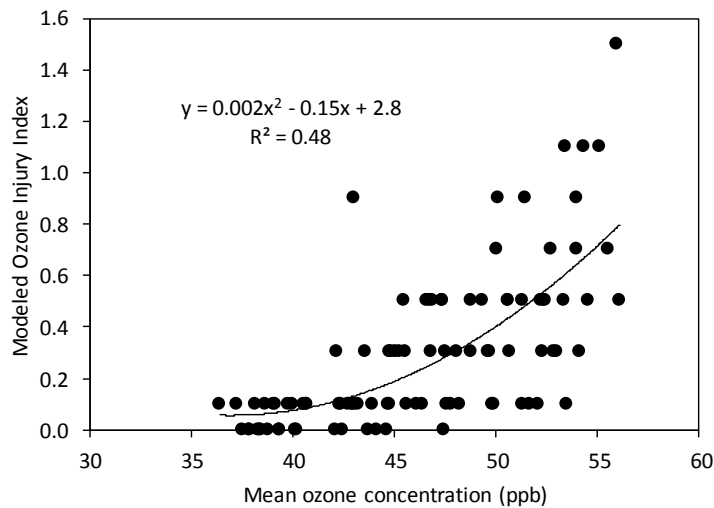


Figure 23. Ozone injury in *Pinus jeffreyi* and *P. ponderosa* from Duriscoe survey were compared with spatial mean ozone concentration map. The resulting model ($R^2=0.48$), was applied to the spatial ozone map to estimate spatial distribution of ozone injury in ozone sensitive pine.

been reached in the extent of their range (3-8 kg N/ha/y), CL for chaparral along the Park's western edge has been reached (10-14 kg N/ha/y), and CL is being approached – and exceeded in 2 hotspots – for mixed conifer in the Kaweah watershed of Sequoia (17 kg/ha/y). Baron (2006) defines a CL for sensitive high elevation lakes as 1.5 kg/ha/y, so much of the high elevation region of SEKI may be impacted by current levels of nitrogen deposition, however pollution impacts on lakes is beyond the scope of this chapter and is covered elsewhere (see Water Quality chapter).

When the model is used to predict ozone injury across SEKI, areas of potential damage to pine are visible. Areas of particular concern include the western edge of Sequoia, notably the Grant Grove area. Also of concern are areas where rivers cut deep drainages into the Sierra, including the Kern, and the middle and south forks of the Kings River (Figure 24).

Nitrogen

Comparing spatial distribution of total nitrogen deposition to critical loads (CLs) of nitrogen defined for ecosystems in the Parks (Fenn et al. 2010a), CL for sensitive epiphytic lichens in conifer and chaparral has

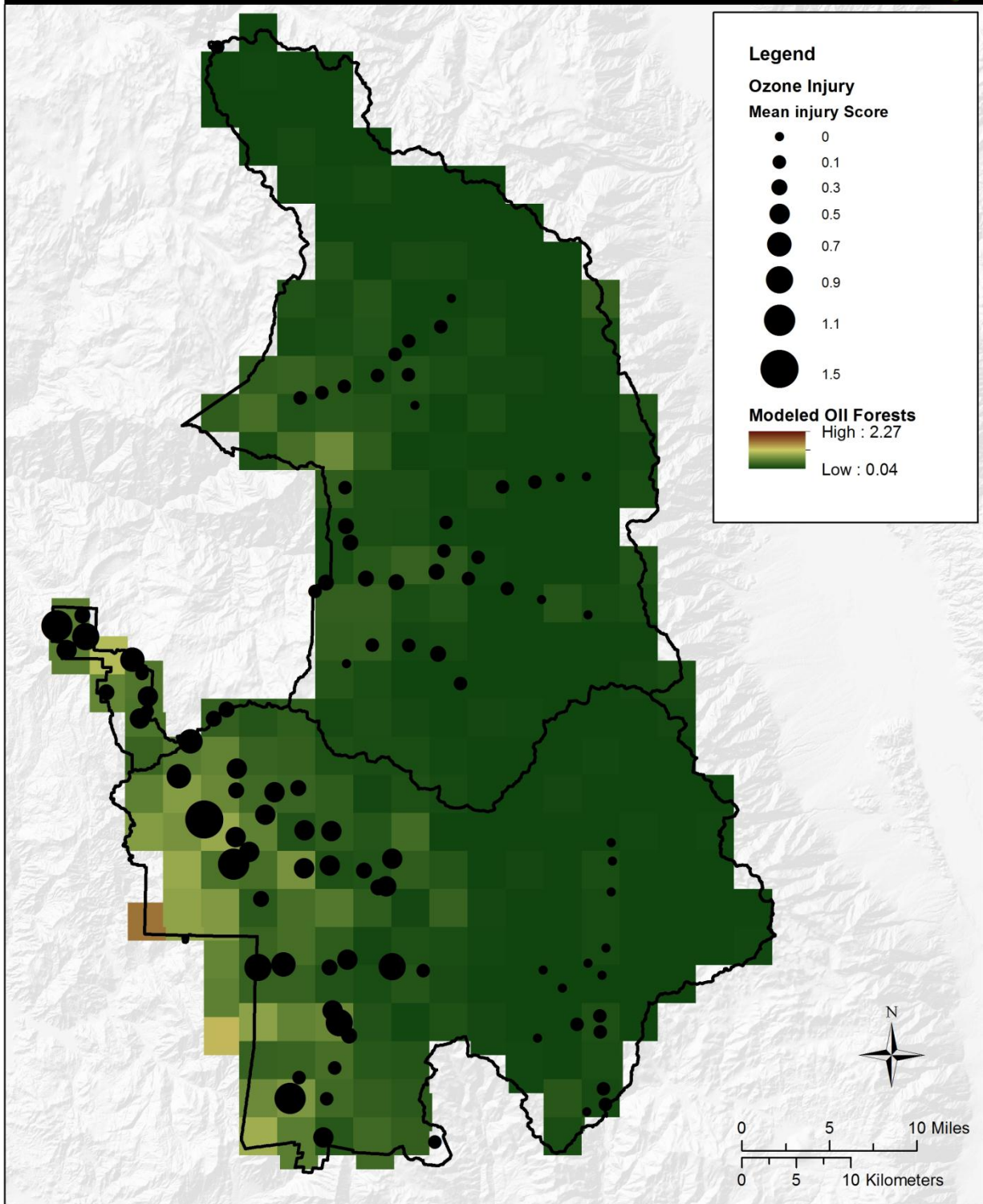


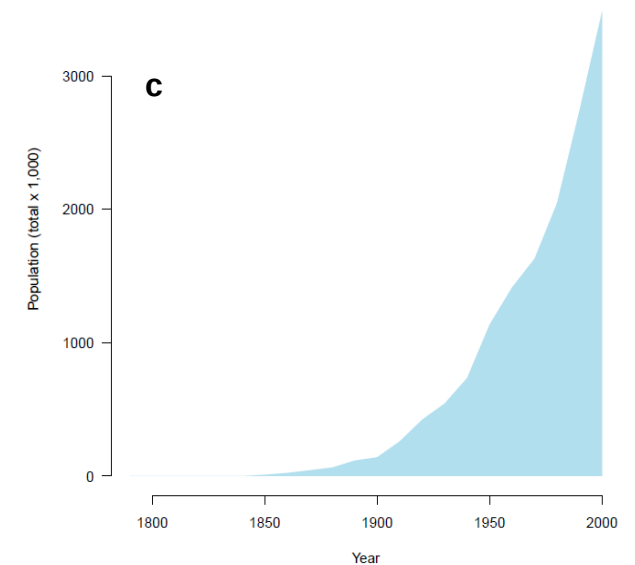
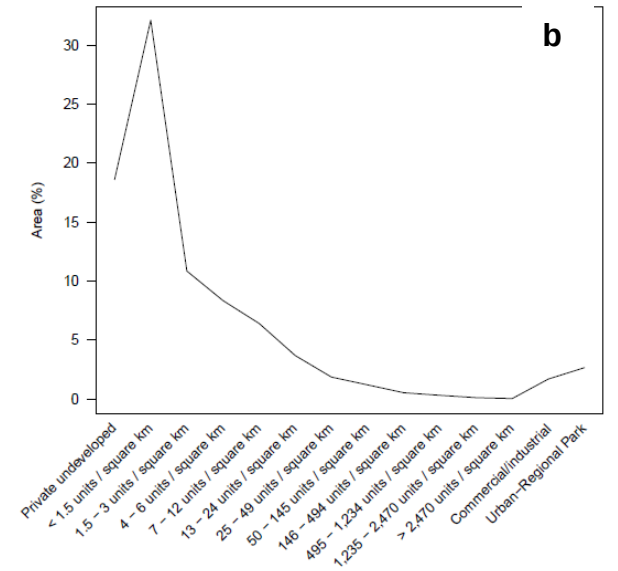
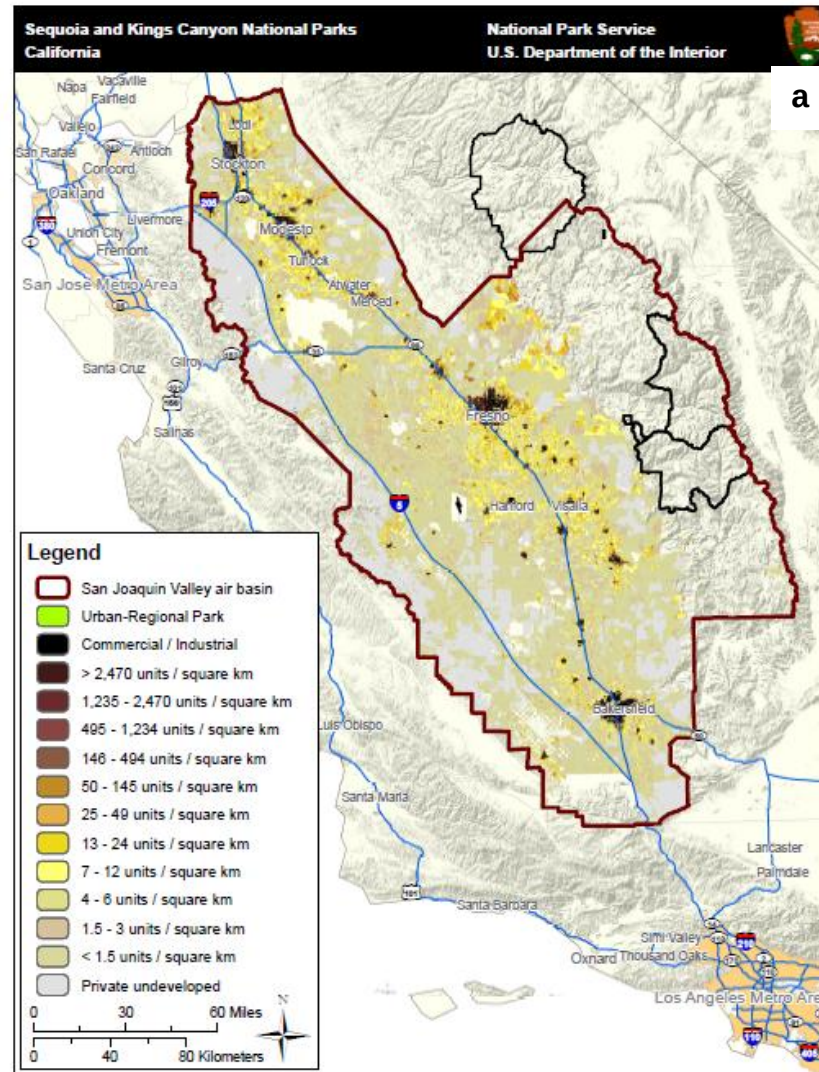
Figure 24. Estimate of the spatial distribution of ozone injury in three-needle pine, based on Duriscoe ozone injury survey, the spatial ozone map, and a simple model of the relationship between the two. OII = Ozone Injury Index (Duriscoe 1990).

Stressors

Land use /fragmentation

Land use upwind of SEKI, particularly development trends in the San Joaquin Valley air basin (SJVAB) has a significant impact on air quality in SEKI, (Figure 25a). As mentioned early in this chapter, over ten percent of the state's population lives in the eight counties of the SJVAB and has more than tripled since 1950 (Figure 25c) and is

Figure 25. Land use linked with increasing population in the SJVAB. Housing density (a) spatially and (b) as a percent of total area. (c) population increases over the last 150 years of the 20th century. Source: B. Monahan, NPScape, NPS. 2011.



projected to reach over 5 million by 2020 - almost 12% of state population (CARB 2009).

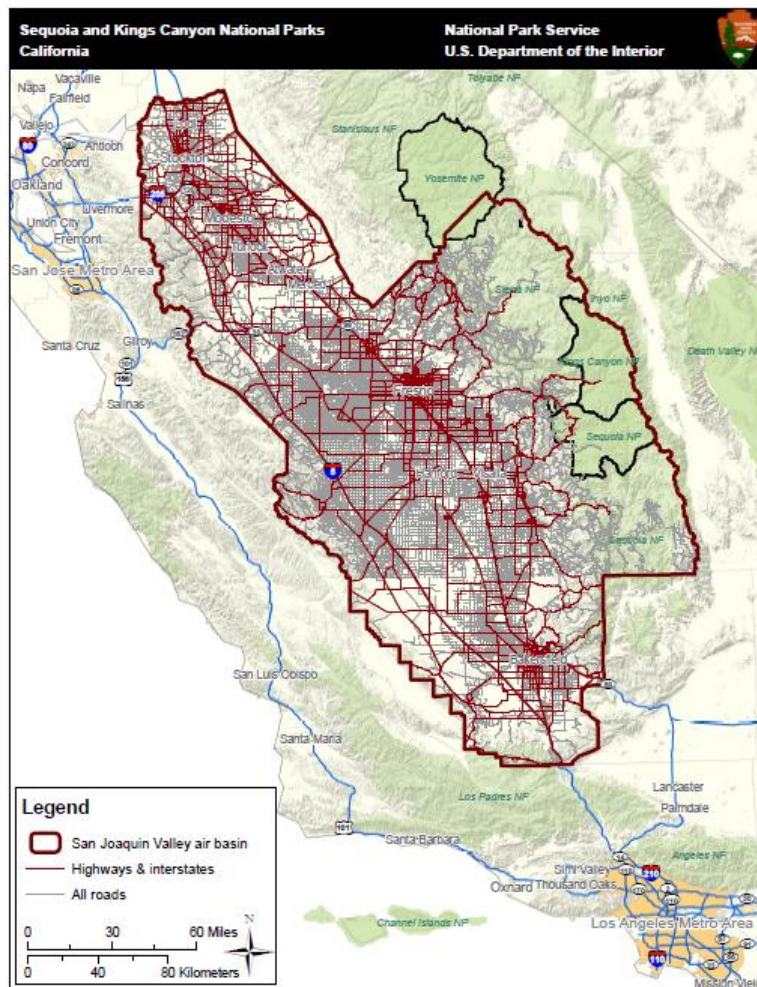


Figure 26. Land use linked with increasing population in the SJVAB. Roads in the SJVAB are shown. Source: B. Monahan, NPScape, NPS. 2011.

Most of the population lives in low-density housing (Figure 25 a,b) spread throughout the valley resulting in generally greater vehicle miles traveled (Figure 26). Long commutes include the 80% of San Joaquin region commuters who make 120,000 daily trips west over the Coast Range to jobs located primarily in the San Francisco Bay Area and Silicon Valley (San Joaquin Partnership 2011, Manteca Economic Development Division 2011). This pattern of development greatly increases mobile source emissions. Commercial and industrial land use generally follows the CA99 highway corridor through the major urban areas of Stockton, Modesto, Fresno and Visalia (Figure 25a). Electricity use and combustion from industrial and mobile sources contribute to NO_x, VOC, and ultimately ozone pollution. Air pollution emission sources within the SJVAB account for about 14 percent of total statewide emissions (CARB 2009).

As mentioned earlier in this chapter, the extensive SJVAB agricultural landscape (Figure 27a) is among the largest producers of agricultural products in the U.S. (CDFA 2011). Cultivated agriculture covers roughly 33% of the airshed's area (Figure 27b). Agriculture is a source of a number of pollutants, particularly nitrogen, but also pesticides, VOCs, dust, and other particulates. Row crops are more prevalent on the western edge of the SJVAB, with generally heavier fertilizer and pesticide use, and orchards on the eastern side (data not shown).

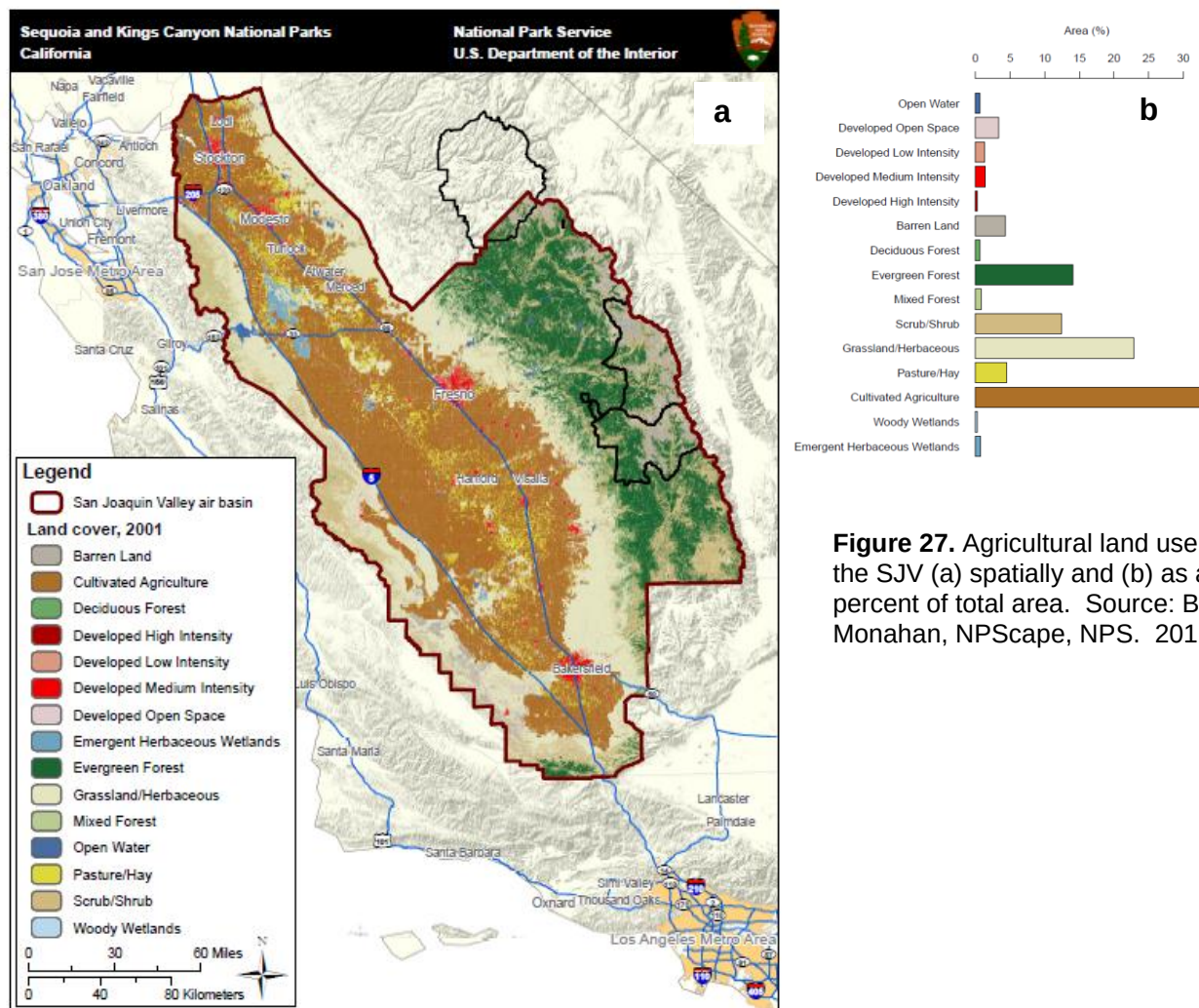


Figure 27. Agricultural land use in the SJV (a) spatially and (b) as a percent of total area. Source: B. Monahan, NPSCape, NPS. 2011.

Climate change

General

Regional air quality is affected by meteorological conditions that change as climate changes. The average maximum or minimum temperature and/or changes in their spatial distribution and duration, lead to changes in atmospheric reactions, to the frequency and pattern of cloud cover, the frequency and intensity of stagnation episodes or a change in the mixing layer that controls the mixing of polluted air with background air. Emissions of hydrocarbons and NO_x from plants and soil are sensitive to temperature and light levels, leading to changes in their concentrations.

Deposition rates to vegetative surfaces are a function of moisture, temperature, light intensity, and other factors. (Weaver et al. 2009)

Ozone

Surface ozone formation from NO_x and VOCs in the presence of sunlight is a temperature dependent reaction, so increased temperatures directly influence ozone formation. Indirect climate effects include climate-induced changes in wildfire abundance and severity. California ecosystems are already drying out, and wildfires are intensifying (Westerling et al. 2006), potentially contributing to the number of ozone NAAQS exceedances (Pfister et al. 2008). Warmer temperatures and increased CO₂ may lead to increased plant growth, which may increase VOC emissions and indirectly ozone. However, since many VOC-emitting plants in California, for example oaks, are water limited, it is possible that if precipitation doesn't also increase, plant growth may not be able to respond to increased CO₂ and temperature.

It is possible, however, that climate change may lead to changes in climate variables that do not favor increases in O₃ concentrations. Climate scenarios are not conclusive on whether California will be wetter or drier, but if it is wetter, increases in cloudiness (and hence decreases in sunlight and O₃ photoproduction) might have net O₃ concentration decreases, despite increased temperatures (Weaver et al. 2009).

Altered fire regimes

Ozone and fire are loosely connected in a positive feedback relationship. Ozone injury to trees rarely kills trees outright, but stresses them so they have fewer reserves to respond to other stresses, including pests, pathogens and fire. Stressed trees succumb more readily to fires. Fire emissions include many compounds, including precursors to ozone formation: NO_x, CO and VOCs (Goldammer et al. 2009). Resulting intense wildfire periods increase ozone concentrations nearby and potentially far downwind from the fire location to levels that exceed current standards (Pfister et al. 2008), which further stress trees.

As well as ozone, fire also affects visibility through emissions of fine particulates (PM_{2.5}) and organic hydrocarbons. As mentioned earlier, visibility in Class I wilderness areas like SEKI is protected under the Clean Air Act, so Parks collaborate with local Air Quality Districts to implement Smoke Management Programs to manage haze. Emissions from prescribed, agricultural and wildland fires are tracked through emission inventories and used in regional air quality modeling (Fox and Riebau 2009).

Other pollution impacts of fire include elevated trace gas, aerosol, and CO₂ levels, nitrogen deposition, acid precipitation and local climatic changes (Bazzaz 1990, Fan et al. 1990, Vitousek et al 1997). Smoke particles can pollute surface water, soil, or wash into streams and lakes, disrupting water quality and aquatic ecosystems (Goldammer et al. 2009).

Condition Assessments

Ozone concentration and nitrogen deposition are the two pollutants for which there is sufficient spatial information to develop a Parks-wide spatial air quality condition assessment. The assessment of air quality was integrated using USGS hydrologic units from a standardized catalog of river basins across the United States, each accompanied by a unique hydrologic unit code (HUC). HUCs are hierarchically-nested, from the smallest (HUC 12 sub-watersheds) to largest (HUC 2 regions). HUC 10 landscape units are of a size roughly equivalent to watersheds, but may differ from local watershed delineations, in accordance with the specific USGS HUC rule-set. We will therefore refer to these hydrologic units as HUC 10 landscape units and not watersheds. Refer to Figures 29, 31 and 32 to view the HUC 10 landscape units used in this analysis.

Ozone concentration

Table 4. Values used to calculate the conversion coefficient between the average ozone spatial layer and the average federal 8H standard layer for the same time periods in 2006-2008.

Site	Avg. ozone	Federal 8H Standard	Coefficient
Ash Mt	58.5	105	1.7949
Lower Kaweah	56.6	96.3	1.7014
Average	57.6	100.7	1.7483

We developed an integrity index for ozone based on the federal NAAQS 8H primary and secondary standards (75 ppb). The spatial map (Figure 3) shows average ozone over the summer and early fall months, over 3 years. We developed a spatial integrity

index representing the federal standard by identifying the points where the ozone standard is

continuously monitored on our spatial ozone map and calculating the conversion coefficient between average ozone concentration and the federal ozone standard during that same period (Table 4). Grant Grove and Lookout Point monitors were not in operation in 2006-2008 (Figure 10). We then multiplied the entire ozone concentration layer by the coefficient to convert concentration to the federal ozone standard (Figure 28). The “poor condition” threshold was defined as anything

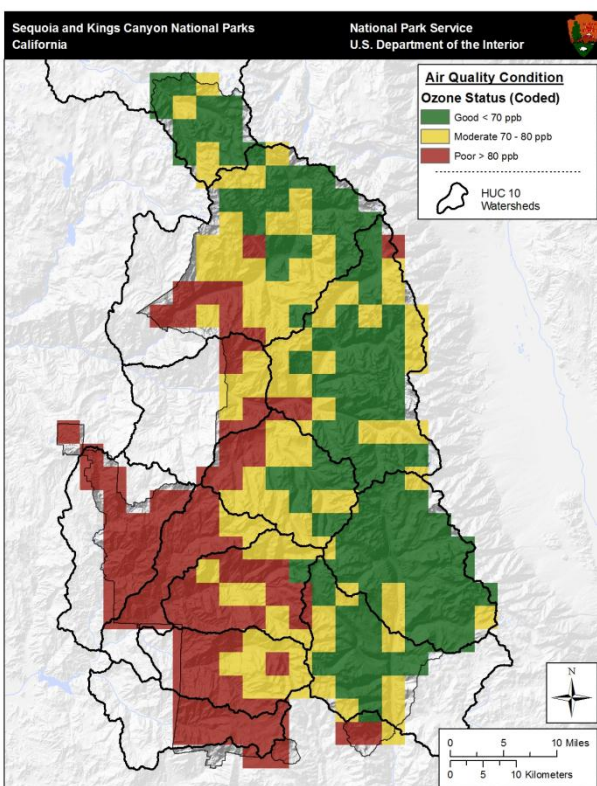


Figure 28. For the ozone condition assessment, we assigned good, moderate and poor thresholds for each of the 3500m units, based on the spatial ozone map converted to federal 8H standard. The overall HUC10 landscape unit condition was the dominant status class in the HUC10 unit derived from the spatial ozone map. In the cases of even distribution between good, moderate and poor, we assigned a moderate status.

equal to or greater than the federal 8H secondary standard threshold of 75 ppb (+ 5 ppb to be conservative in accounting for uncertainty). “Good” integrity is the reference level of ozone, that is an average 40 ppb or lower (when multiplied by the conversion factor is a standard of 70 ppb), defined by the federal EPA as the background ozone reference level, however it is also an internationally accepted threshold below which vegetation injury is uncommon (AOT40). Moderate is everything between poor and good. From the calculated condition layer, we estimated condition percent area cover in each of the HUC 10 landscape units (Figure 29). The lack of trend in the condition of the Parks relative to ozone, is based on there being no trend in ozone concentration (Figure 9) or in the federal 8H ozone standard (Figure 10). The resulting condition map shows the entire western half of Sequoia National Park, the Kaweah drainage, rated as “poor” on the ozone component of the air quality condition assessment. Similarly, the western half of Kings Canyon has “moderate” ozone air quality condition (Figure 29). The eastern side of both parks, protected from the western exposure to the SJVAB by distance and high elevation topographic barriers, rates as “good” ozone air quality condition. Note that the ozone air quality condition map is very similar to the spatial map showing ozone injury in pines, an AQRV related to ozone (Figure 24).

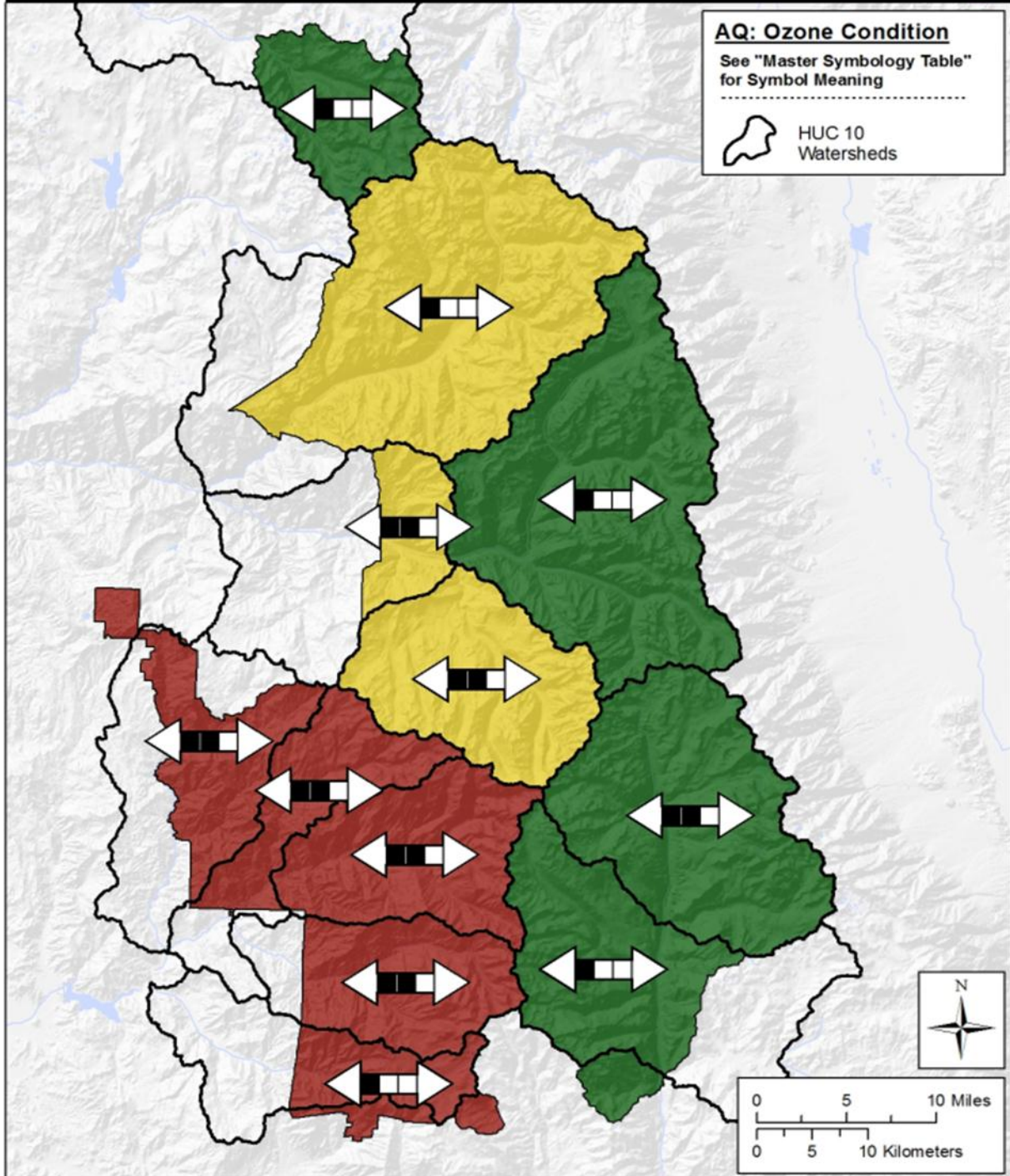


Figure 29 The final ozone condition assessment map showing spatial condition of the ozone component of air quality.

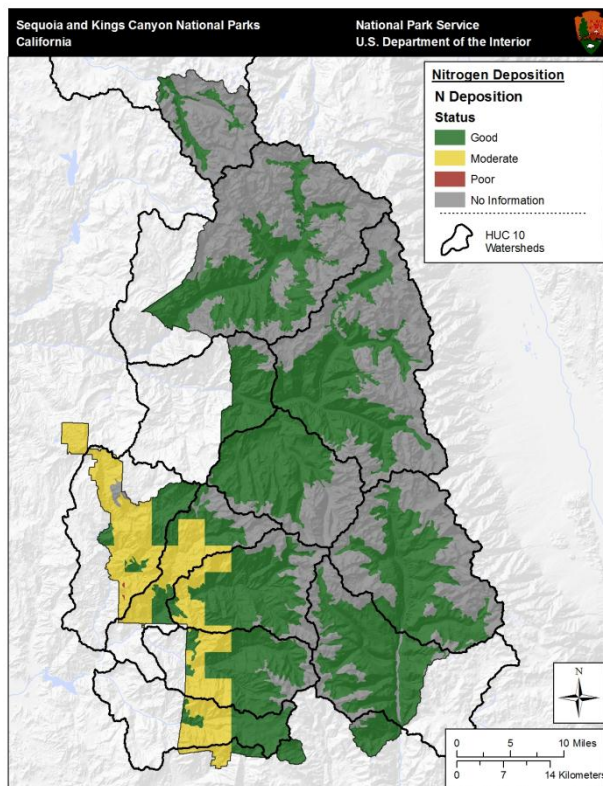
Nitrogen deposition

Nitrogen deposition critical loads for all California ecosystems were defined by Fenn et al (2010a), so we applied those values to define condition thresholds in SEKI for each ecosystem type studied (Table 5). The threshold between good and moderate condition was defined at nitrogen deposition loads where the most sensitive epiphytic

lichen communities show shifts away from acidophyte species. Poor condition was defined at nitrogen deposition levels where nitrogen leakage into streams is measured. For ecosystem types in which a critical load had not been calculated (barren, alpine, riparian), we did not calculate a threshold and designated the area “no data”. For the very small area of grassland and pinyon-juniper in the Parks, the critical load is based on the nitrogen advantage for exotic grass invasion. For these, no moderate status was defined, but all sites were well within the “good” condition status. The resulting assignment of condition by ecosystem type is shown in Figure 30. The

Table 5. Thresholds for Parks condition from nitrogen deposition were based on critical loads defined by Fenn et al (2010a). Units are kg-N/ha/y.

Ecosystem Type	Status		
	Poor	Moderate	Good
Chaparral - Oak Woodland	≥ 10	5.5-10	< 5.5
Mixed Conifer Forest	≥ 17	5.2-17	< 5.2
Alpine		not calculated	
Barren		not calculated	
Grasslands	≥ 6	none	< 6
Riparian		not calculated	
Pinyon-Juniper	≥ 6.3	none	< 6.3



dominant condition in each HUC10 determined the overall condition of the entire. No trend information exists for nitrogen deposition based on data from the Parks.

The resulting condition map (Figure 31) shows nitrogen-related air quality condition to be generally good, except for the northwestern side of Sequoia National Park, where condition is moderate. Note that this condition map is developed with the critical load concept of reference condition, which links nitrogen deposition to ecosystem response, which is an AQRV. Note also that susceptibility of high elevation lakes to nitrogen deposition is not covered in this condition assessment, as it falls

Figure 30. For the nitrogen deposition condition assessment, we assigned good, moderate and poor thresholds based on critical loads and ecosystem types defined in Fenn et al. (2010). The overall HUC10 landscape unit condition was the dominant status class in the HUC10 unit.

under the Water Quality chapter. The structure of the HUC 10 landscape unit assessment unfortunately removes important variability within each HUC 10 unit. It is important to note for the overall condition assessment that the foothills region along the western edge of Sequoia Park is a region of potential nitrogen saturation (Figure 30). The predominance of green in southwestern HUC 10 units is, in part, an artifact of the distribution of foothill vegetation in that area such that it does not dominate those HUC 10 units (Figure 31).

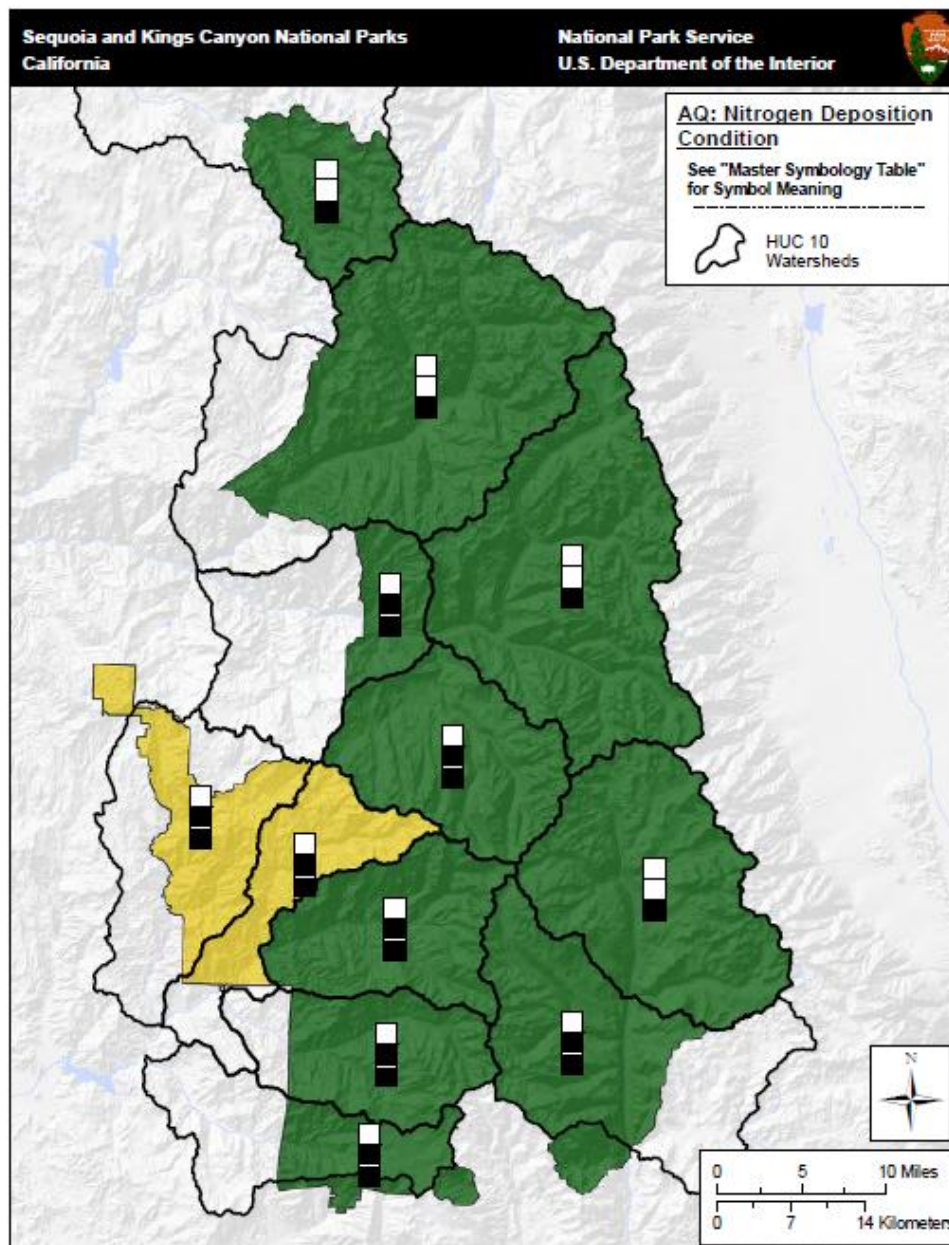


Figure 31. Air quality condition with respect to nitrogen deposition, shows spatial distribution of nitrogen deposition exceedance of critical loads that cause ecosystem change. The condition map combines deposition loads with ecosystem response (risk) to that load. The northwest area of Sequoia is the area of greatest risk. Note that susceptibility of high elevation lakes to nitrogen deposition is not covered in this condition assessment, as it falls under the Water Quality chapter.

Summary of Condition

The summary of air quality in Sequoia and Kings Canyon Parks takes into account the temporal and spatial data both, to create a comprehensive picture of the parks. The temporal record doesn't have good spatial representation, but monitors are well-situated to capture the worst air quality conditions in the parks. Table 6 summarizes conditions by pollutants, with an emphasis on long-term temporal data. Contaminants outside of mercury are not listed because the data is too sparse to provide a solid temporal or spatial assessment. It should be noted that, regardless, pesticide and other contaminant data show conditions that warrant significant concern and further study.

Table 6. Summary of air quality condition in Sequoia and Kings Canyon National Parks.

Metric	Integrity Measure	Condition	Summary Comments
Ozone	Federal 8h Standard for human health and welfare (including vegetation health) 75 ppb	POOR	No year is below standard . Condition assessment for worst western edge of park – interior and higher elevations have lower concentrations, but are less well characterized.
Total Nitrogen Deposition	Critical Load for sensitive elements (lichens) of mixed conifer and chaparral ecosystems (3-8 kg ha ⁻¹ yr ⁻¹ , Fenn et al. 2010a).	MODERATE	Possible nitrogen saturation of foothills region of Parks. Critical thresholds for vegetation only - does not include response of sensitive alpine lakes.
Total Sulfur Deposition	Peterson 1992c: < 5 kg S ha ⁻¹ yr ⁻¹	GOOD	No trend.
Visibility	Reference condition is 7 dV	POOR	Visibility in Parks is half natural conditions. No trend.
PM2.5	State std. 12 ug m-3 Federal std. 15 ug m-3	GOOD	No trend in PM2.5 or PM10
PM10	State std. 20 ug m-3	GOOD	
Mercury	Federal contaminant health threshold in fish for piscivorous mammals and birds, and for human recreational consumption.	POOR	Mercury deposition may be under-represented by current monitoring, but is generally high compared to other western U.S. sites where it has been measured. No trend.

The spatial air quality condition assessment in SEKI aggregates air quality impacts by taking into account the worst of all the air quality threats to SEKI's resources and AQRVs. Air pollution threats should not be averaged or added, because the physical interactions and effects of pollutants on park resources are not averaged or added, but interact in complex ways that are beyond the scope of this chapter. In this case, ozone concentration spatial condition relative to the primary and secondary federal ozone standard shows worse air quality condition than nitrogen deposition, thus is the spatial layer that defines the overall air quality condition of the parks (Figure 32).

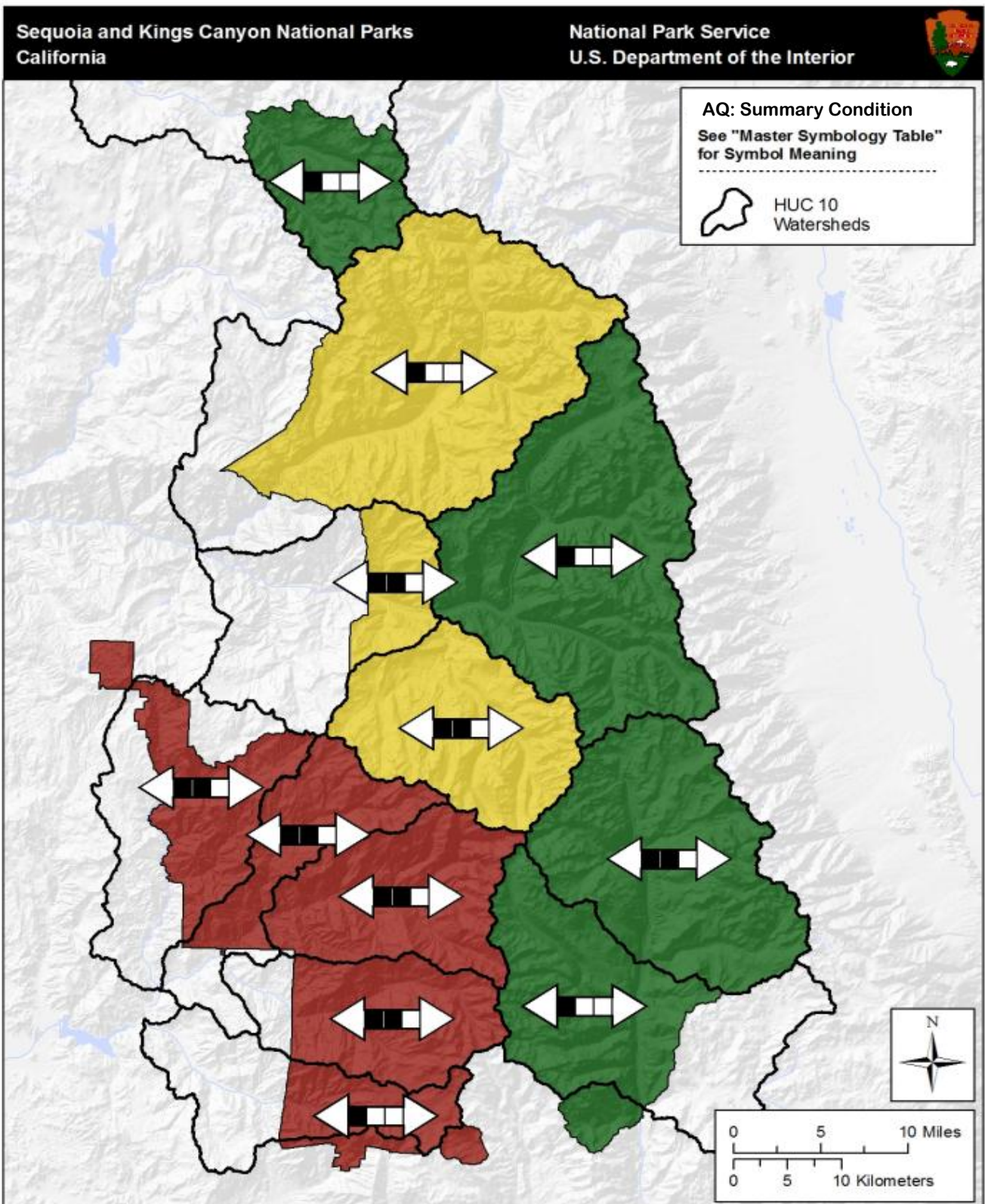


Figure 32. The spatial summary of the air quality condition assessment in the Parks, accounting for the worst of air quality threats to the Parks.

Level of confidence in assessment

Ozone concentration

Uncertainty inherent in passive sampling of ozone was discussed previously. Because of this uncertainty. The geographically weighted regression approach to interpolate ozone concentrations between measured points introduced further uncertainty. Because of these combined uncertainties, we never assigned “High” confidence to a HUC-10 unit. Interpolation uncertainty increases with distance from the passive samplers. We characterized our confidence in the interpolated result as “moderate” in the areas in close proximity (within 3500m, our grid size) to passive samplers or the continuous monitors. For sites more than 3500m from a passive sampler, our confidence was “low”.

Nitrogen deposition

As discussed earlier, the spatial nitrogen deposition data are modeled using CMAQ. When Fenn et al (2010) compared throughfall nitrogen deposition with CMAQ estimates they found reasonable agreement <6-7 kg ha/y, but as nitrogen deposition increased CMAQ underestimated N deposition. They found a significant linear relationship modeled and throughfall nitrogen deposition ($R^2=0.80$). So, our confidence in CMAQ to estimate nitrogen deposition in the Parks is moderate at best. In parts of the Parks where critical loads have yet to be defined, and large areas of “no information” exist, our confidence score is low.

Gaps in understanding

While SEKI are among the most pollution-impacted Parks in the country, routine air quality monitoring is confined to a geographically small portion of the park. Air quality in broad expanses of the Parks remains under-sampled due to remoteness, difficulty of access, and lack of resources. It is clear that at current levels, poor air quality is negatively impacting people, plant and animal populations and processes in the Parks. But the patterns of pollutant dispersal throughout the Parks are currently not well understood, and the impacts of pollutants in all but a few species and ecosystem types remain unclear. Dry deposition and throughfall deposition of major ions, particularly nitrogenous species, are input pathways that aren't well-characterized. Pesticides and mercury are poorly understood and highly toxic inputs that need characterization.

Recommendations for future study/research

The most cost-effective means of monitoring deposition to large expanses of remote terrain is to invest in 1) inexpensive, low-power monitoring equipment such as passive samplers and battery-powered ozone monitors, and to 2) develop and improve models and other remote means of characterizing deposition and its effects, especially options for characterizing deposition to mountainous terrain. Much progress has been made with nitrogen deposition modeling, but modeling approaches designed to characterize ozone deposition, while they exist, have been slowed by the need to include plant physiological response to drought.

Remotely sensed variables that look at change should be pursued. Particularly important would be variables that can be attributed to nitrogen fertilization effects or ozone injury effects. But with modeling and remote sensing comes the challenge of verifying model results. So, we recommend in conjunction with the above, establishing plots to ground-truth models, such as strategic placement of solar-powered ozone monitors, maintenance of broad passive ozone monitoring networks, and routine monitoring of ozone injury in multiple plant species, including but not limited to pines. This is

particularly important since the Parks' statutory mandate is to protect AQRVs, thus AQRVs should be routinely monitored. AQRVs include vegetation and ecosystem effects: ozone injury to plants, changes in plant community composition and chemistry, changes in lichen community composition and chemistry, nutrient cycling studies especially in the foothills region to capture nitrogen saturation. These plots would serve the dual purpose of characterizing AQRVs while also providing ground-truthing to models and remote sensing approaches.

We recommend participation in increased efforts to characterize mercury and pesticide deposition, such as The North American Atmospheric Mercury Speciation Network (AMNet-NADP).

Because nitrogen deposition in the Parks, and throughout the western U.S. are in the form of NH_4^+ (Fenn et al. 2010b), which is very likely from ammonia emissions, we recommend participation in the NADP AMoN ammonia network.

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