ASSESSMENT OF AIR QUALITY AND AIR POLLUTANT IMPACTS IN CLASS I NATIONAL PARKS OF CALIFORNIA

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LIST OF ACRONYMS AND UNITS

AERP	Aquatic Effects Research Program
AIRS	Atmospheric Information Retrieval Systems
Al	Aluminum
ANC	Acid neutralizing capacity
ARD	Air Resources Division (of the National Park Service)
AQRV	Air quality related value
b_{abs}	Light absorption coefficient
b_{ext}	Light extinction coefficient
BLM	Bureau of Land Management
b_{Ray}	Rayleigh extinction coefficient
Ca	Calcium
CARB	California Air Resources Board
CASTNet	Clean Air Status and Trends Network
C_B	Base cations
Cl	Chloride
CO	Carbon monoxide
DOC	Dissolved organic carbon
EMAP	Environmental Monitoring and Assessment Program
EPA	Environmental Protection Agency
FOREST	Forest Ozone Response Study
GIS	Geographic information system
H	Hydrogen
HCO_3	Bicarbonate
HF	Hydrogen fluoride
H_2S	Hydrogen sulfide
IAP	Integrated Assessment Report
JOTR	Joshua Tree National Park
IMPROVE	Interagency Monitoring of Protected Visual Environments
K	Potassium
LABE	Lava Beds National Monument
LAVO	Lassen Volcanic National Park
LAC	Limit of acceptable change
MAGIC	Model of Acidification of Groundwater in Catchments
MCAB	Mountain Counties air basin
Mg	Magnesium
MOHAVE	Measurement Of Haze And Visual Effects
N	Nitrogen
Na	Sodium
NAAQS	National Ambient Air Quality Standards
NADP	National Atmospheric Deposition Program
NADP/NTN	National Atmospheric Deposition Program/National Trends Network
NAPAP	National Acid Precipitation Assessment Program
NAWQA	National Water Quality Assessment
NH_3	Ammonia
NH_4-N	Ammonium-nitrogen

NO	Nitric oxide
NO_2	Nitrogen dioxide
NO_3-N	Nitrate-nitrogen
NO_X	Nitrogen oxides
NPS	National Park Service
OII	Ozone Injury Index
O_3	Ozone
P	Phosphorus
PFT	Perfluorocarbon tracer
PINN	Pinnacles National Monument
PM-10	Particulate matter less than 10 :m diameter
PM-2.5	Particulate matter less than 2.5 :m diameter
PORE	Point Reyes National Seashore
PSD	Prevention of significant deterioration
REDW	Redwood National Park
RH	Relative humidity
S	Sulfur
SCOIAS	Sierra Cooperative Ozone Impact Study
SDAB	San Diego Air Basin
SEKI	Sequoia and Kings Canyon National Parks
SFBAAB	San Francisco Bay Area air basin
SFU	Stacked filter unit
SIP	State implementation plan
SJV	San Joaquin Valley
SJVAB	San Joaquin Valley air basin
SO_2	Sulfur dioxide
SO_4	Sulfate
SoCAB	South Coast air basin
STORET	Storage and Retrieval Database (EPA)
$SUMO_x$	Cumulative index of air pollution exposure based on total hours exceeding a threshold level x, with units typically expressed as ppb-hr or ppm-hr
SVAB	Sacramento Valley air basin
SVR	Standard visual range
USFS	U.S. Forest Service
USFWS	U.S. Fish and Wildlife Service
YOSE	Yosemite National Park
VMT	Vehicular miles traveled
VOC	Volatile organic compound
VR	Visual range
VWM	Volume-weighted mean
YBP	Years before present

<u>Units</u>

b_{ext}	light extinction coefficient
cfs	cubic feet per second
cm	centimeter
dv	deciview

ha	hectare
kg	kilogram
km	kilometer
L	liter
m	meter
meq	milliequivalent
mg	milligram
Mm^{-1}	inverse megameter
ppbv	parts per billion by volume
ppmv	parts per million by volume
yr	year
:eq	microequivalent
:g	microgram
:m	micrometer
:S	microsiemen

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EXECUTIVE SUMMARY

1. Background

This report summarizes current and potential future conditions of air quality and air pollution impacts in Class I national parks[1] located in California, and recommends monitoring and research activities that could be implemented to acquire critical new data. The subject parks include Joshua Tree National Park (JOTR), Lassen Volcanic National Park (LAVO), Lava Beds National Monument (LABE), Pinnacles National Monument (PINN), Point Reyes National Seashore (PORE), Redwood National Park (REDW), Sequoia and Kings Canyon National Parks (SEKI), and Yosemite National Park (YOSE). The focus of this report is on the effects of pollutants (ozone, sulfur dioxide, nitrogen oxides, particulates, sulfate, and ammonia) on air quality and on selected air quality related values (terrestrial vegetation, aquatic systems, visibility). Other pollutants (e.g., pesticides) are also discussed for specific locations where these pollutants may have potential impacts on natural resources.

In general, air quality adjacent to some Class I areas in California is considerably better than in areas of the eastern United States. Air quality in many areas is among the best in the United States, in particular at near-coastal northern locations and in remote sections of the Sierra Nevada and Cascade Mountains. However, southern airsheds on the western slopes of the Sierra Nevada (Sequoia, Kings Canyon (referred to jointly throughout the report as SEKI), and Yosemite National Parks) and in the southeastern deserts (Joshua Tree National Park) experience elevated ozone concentrations and nitrogen deposition, especially during summer months. Other Class I areas throughout the state experience seasonal to chronic periods of poor air quality, especially high ozone. These areas include Pinnacles National Monument and Lassen Volcanic National Park.

California is the most populous state in the nation, and vehicular emissions and point sources are concentrated around major population centers. The highest emission source areas of nitrogen oxides and reactive organic gasses (precursors of ozone) are the South Coast, San Francisco Bay Area, and San Joaquin Valley air basins. These emissions are mainly from motor vehicles, with other significant sources of nitrogen from fossil-fueled power plants and of reactive organic gasses from cleaning and surface coatings and solvent evaporation. Sulfur dioxide emissions have been low in California since the early 1980s and do not currently pose a significant concern for Class I areas.

2. Visibility Impairment

Visibility impairment occurs at all of the National Park Service monitoring sites in California. Monitoring data show coarse mass concentrations, particles between the size of 10 microns and 2.5 microns, are highest along the Pacific Coastal Mountains and lowest in the Sierra-Humbolt region. Fine aerosol concentrations are highest in the Southern California region. On an annual basis, organics contribute the most to light extinction at parks in the Sierra-Humbolt area (Lassen Volcanic National Park) and Sierra Nevada area (Yosemite, Sequoia, and Kings Canyon National Parks), whereas sulfates contribute the most to extinction in the Pacific Coastal Mountain parks (Pinnacles National Monument, Point Reyes National Seashore, Redwood National Park), and nitrates are the largest contributors to extinction in southern California.

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[1] Class I areas include national parks over 2,430 ha (6,000 ac) and national wilderness areas over 2,025 ha (5,000 ac) that were in existence before August, 1977.

3. Air Quality and Atmospheric Deposition

Urban-area emissions are known to affect air quality in California's national parks, particularly those parks in the Sierra Nevada and southeastern deserts. Prevailing winds carry precursors and secondary pollutants along known transport routes into the mountains and deserts to the south and east of urbanized areas. National Parks along the coast or in northern California are not as affected by urban air pollution, due to their remoteness and location upwind of the major urban centers. Ozone and particulate matter are the most problematic air pollutants, and are of concern for their effects on public health, visibility and vegetation. In addition, nitrogen deposition may affect aquatic and terrestrial systems, though it is not thought to be a public health concern.

Ozone measurements in Joshua Tree, Sequoia, and Kings Canyon National Parks and Pinnacles National Monument exceed the level of the federal one-hour ozone standard (120 parts per billion, ppb) on one or more days per year; they frequently exceed the level of the California state standard (90 ppb) and are well above typical background levels on many other days. Some ozone measurements in Yosemite National Park have been moderately high, remaining below the federal one-hour level, but above the California standard. The annual maximum ozone levels in Lassen Volcanic National Park have been well below the federal exceedance limit, though close to the California limit (some above, some below). Redwood National Park and Point Reyes National Seashore showed no exceedance for either the California or federal standard. In Lava Beds National Monument, where only weekly-average data are available, measurements are generally in the low range compared to other parks in the state.

From the early 1980s through 1997, ozone concentrations have been declining in the South Coast and San Francisco Bay Area air basins. During this same period, ozone concentrations have remained roughly steady in the San Joaquin Valley Air Basin. Particulate concentrations have been declining since 1988 in all of these air basins.

4. Environmental Effects

Class I parks throughout California exhibit enormous diversity of natural resources, levels and kinds of air pollution, and associated sensitivity to air pollution degradation and effects on sensitive receptors. Topographic variations and proximity to the Pacific Ocean, coupled with geological diversity associated with plate tectonics, uplift, glaciation, and volcanic and seismic activity, contribute to large temperature and moisture gradients. The result is an enormous diversity of natural resources, many of which are sensitive to air pollution degradation. Airflow patterns and varying proximity to major emissions source areas cause Class I parks to experience very different air pollution regimes. In addition, resource sensitivities differ greatly across the parks. Air pollution damage, especially damage associated with exposure of sensitive plants to ozone, varies from virtually none in Lava Beds National Monument, Point Reyes National Seashore, and Redwood National Park, to significant in Sequoia and Kings Canyon National Parks. Table A-1 provides an overview of pollutant exposures, resource sensitivity, and documented or probable effects to date.

The potential impacts of nitrogen and sulfur deposition on high-elevation surface waters have been well-studied in Sequoia National Park and to a lesser extent elsewhere in the Sierra Nevada. There have been no clearly documented effects of air pollution on the aquatic resources of any of the Class I parks in California. It is probable, however, that the levels of nitrogen deposition received in high-elevation portions of Sequoia, Kings Canyon, and Yosemite National Parks have contributed to slight episodic acidification of surface waters during snowmelt and

Table A-1. Relative rating of pollutant exposures, sensitivity of surface waters to acidification from sulfur or nitrogen deposition, and known or suspected effects on terrestrial and aquatic receptors in the Class I national parks in California.

Class I Area	Pollutant Exposure[1]			Sensitivity to Surface Water Acidification[2]	Known or Suspected Effects[3]				
					Terrestrial Resources			Aquatic Resources	
	Ozone	Sulfur	Nitrogen		Ozone	Sulfur	Nitrogen	Sulfur	Nitrogen
JOTR	H	L	M	L	+	-	○	-	-
LAVO	M	L	L	H	++	-	-	-	-
LABE	L	L	L	L	○	-	-	-	-
PINN	H	L	L	L	+	-	-	-	-
PORE	L	L	L	L	-	-	-	-	-
REDW	L	L	L	M	-	-	-	-	-
SEKI	H	L	M	H	+++	-	○	-	○
YOSE	M	L	L	H	+++	-	-	-	○

[1] Pollutant exposure: L=low, M=medium, H=high
[2] Sensitivity to acidification: L=low, M=medium, H=high
[3] Known or suspected effects:
- - = adverse effects unlikely
- ○ = adverse effects may occur, but are not documented and are presumed to be small in magnitude or non-existent
- + = adverse effects are suspected
- ++ = low to moderate damage has been confirmed
- +++ = significant damage has been confirmed

spring rain storms. Slight chronic acidification may also have occurred in the most acid-sensitive aquatic systems. Such effects on water chemistry, if they have occurred, have been small in magnitude and it is unlikely that adverse biological impacts have occurred.

However, Lassen Volcanic, Sequoia, Kings Canyon, and Yosemite National Parks contain some of the most acid-sensitive lakes in the world. A combination of factors (including steep slopes, scarcity of soil, deep snowpacks, and resistance of the bedrock to weathering) result in lake and stream waters that are almost completely devoid of any buffering capacity. Even relatively moderate amounts of nitrogen or sulfur deposition could lead to chronic, and especially episodic, acidification, with significant adverse effects on in-lake and in-stream biota. Because of this high acid-sensitivity, a long-term aquatic monitoring program is needed in these four parks to document if and when any future adverse impacts occur.

Amphibian decline is an important environmental concern throughout the western United States, including the Class I areas of California. A number of potential causes have been suggested, including acidic deposition. There is no clear evidence suggesting that air pollution has contributed to the observed dramatic declines in amphibian populations, but it cannot be completely ruled out. Multiple causes may be involved, including past fish stocking, ultraviolet

radiation, pesticides, disease, climatic changes, and perhaps acidic deposition. Additional research and monitoring are needed.

Effects of air pollution on vegetation have been documented in Sequoia, Kings Canyon, Yosemite, and Lassen Volcanic National Parks, all of which are located in mountainous areas of the Sierra Nevada and Cascade Mountains. High ozone exposure along the western edge of the mountains throughout late spring and summer provides sufficient stress in the bioindicators ponderosa pine and Jeffrey pine to induce a gradient in foliar injury from south to north; injury is very severe at Sequoia and Kings Canyon National Parks, severe at Yosemite National Park, and moderate at Lassen Volcanic National Park. Reduced foliar biomass has been documented at Sequoia, Kings Canyon, and Yosemite National Parks, and reduced photosynthetic rates and tree growth have been documented at Sequoia and Kings Canyon National Parks.

These air pollutant effects, which have been documented through research and monitoring over the past 20 years, provide clear evidence of a cause-and-effect relationship between elevated levels of ambient ozone and impacts to vegetation resources in the national parks of California. These effects, particularly at Sequoia and Kings Canyon National Parks, are among the most severe in North America for non-point source pollution. Altered plant physiological response and productivity represent significant changes in ecosystem functional characteristics at broad spatial scales. It is not clear if this will ultimately lead to altered distribution and abundance of forest species, as has apparently occurred in the more polluted San Bernardino Mountains.

No air pollutant effects on vegetation have been observed in Joshua Tree National Park, Lava Beds National Monument, Pinnacles National Monument, Point Reyes National Seashore, or Redwood National Park. Some effects in Joshua Tree National Park seem likely, given high ozone exposure in the park and the results of *in-situ* exposure studies under controlled conditions, although field surveys have not indicated any injury. It may be that low physiological activity during high ozone exposure in summer protects desert plants from injury, but the condition of vegetation in the park bears continued monitoring. It is also recommended that a survey of the bioindicator ponderosa pine at Lava Beds National Monument be considered to ascertain if there is any ozone injury, and more importantly, to provide a baseline against which future condition could be compared in the event of increased ozone exposure. Such baseline monitoring of sensitive bioindicator plant species for ozone might also be of future value at Point Reyes National Seashore and Redwood National Park in the event that ozone exposures increase substantially in the future.

No injury to vegetation due to pollutants other than ozone has been documented at any of the Class I national parks in California. Exposures to sulfur and nitrogen at all of the Class I parks are low relative to known vegetation damage thresholds.

A number of park-specific studies are needed to more fully characterize current resource conditions. Continued monitoring of air pollution exposure and effects on sensitive receptors is of critical importance. Regional studies are also needed, in particular modeling activities to further refine the spatial distribution of ozone effects on vegetation and to quantify the dose/response functions for the effects of nitrogen deposition on sensitive aquatic resources.

5. Summary for Each Class I Park

Joshua Tree National Park

The resources in Joshua Tree National Park of greatest concern with respect to air pollution degradation are visibility and vegetation. Ozone concentrations remain high in the park, with exceedance of the federal maximum daily 1-hr standard occurring every year from 1992-1997.

The primary source of ozone and ozone precursors in the park is emissions from the South Coast Air Basin, where ozone concentrations have been declining in recent years. However, no such trend in declining ozone has been detected in Joshua Tree National Park, and no definitive explanation is available for the difference between trends in the South Coast and the park.

Based on data collected 80 km away, wet sulfur and nitrogen deposition at Joshua Tree National Park are well below levels expected to cause damage. Dry deposition measurements for the period 1995 through 1998 suggest that dry sulfur deposition is low, but dry nitrogen deposition is modest, although still below the range thought to cause damage to vegetation. A new NADP monitoring station was installed in 2000.

Surface water resources are generally lacking at Joshua Tree National Park. Water quality is not an important air quality related value in this park.

There are no documented effects of air pollution on vegetation growing naturally in the park, although controlled fumigation studies have shown that a number of native plant species are quite sensitive to this pollutant, and ozone effects were noted on one plant species grown in a biomonitoring garden in the park. There appears to be high potential for vegetation injury caused by ozone, although injury symptoms are known for relatively few desert plant species and can be difficult to diagnose.

Aerosol monitoring was conducted for a short period during a special study in 1992 at Joshua Tree National Park, but there is not an extensive visibility data set available for this park. However, park staff observations note that visibility impairment frequently occurs at the park, particularly during summer. Winter visibility conditions seem to be better when the prevailing airflows are not from the Los Angeles basin. As part of the IMPROVE Network expansion, a new aerosol sampler was located in the park during the winter 2000 season.

Exposures to ozone and, to a lesser extent, nitrogen deposition are high to moderate and continual monitoring of ozone concentrations and dry N deposition will be important. Additional monitoring of ozone effects on vegetation is also recommended.

Lassen Volcanic National Park

Lassen Volcanic National Park is located in northern California, relatively remote from urban areas. It is believed that emissions from the adjacent Sacramento Valley Air Basin may affect air quality in the park, despite the fact that Lassen Volcanic National Park is at the northern end of the Sacramento Valley Air Basin whereas the urbanized portion of the air basin is to the south.

Pollution levels in the park are low to moderate. Ozone maximum daily 1-hour concentrations exceeded the state standard (90 ppb) during three out of the six years from 1992 through 1997, but did not exceed the federal standard (120 ppb). Wet deposition measurements for sulfur and nitrogen are not currently available within the park, but can be roughly estimated using wet-deposition measurements at a nearby site (approximately 65 km southeast of the park). These estimates indicate that both sulfur and nitrogen wet deposition are well below the minimum levels thought to be associated with damage to sensitive resources. A wet deposition monitoring site was recently installed within the park. Dry deposition data are available for the period 1996 through 1998, and indicate low values for both sulfur and nitrogen.

Relatively little research has been conducted in Lassen Volcanic National Park on the sensitivity of aquatic ecosystems to acidification from sulfur and/or nitrogen deposition. Current deposition levels are very low and are not likely to have caused any adverse impacts to aquatic ecosystems. However, available lake survey data suggest that high-elevation lakes in Lassen Volcanic National Park may be, as a group, more sensitive to acidification than the aquatic

resources in any other western park. More complete characterization of these resources and initiation of a long-term lake monitoring program are needed to better establish baseline conditions and determine if and when future impacts occur to these ultra-sensitive resources.

Some ponderosa pine and Jeffrey pine in the park currently have foliar symptoms of ozone injury, with unknown impacts on growth and productivity. Although injury is not as high as at national parks in the southern Sierra Nevada, this injury represents a significant impact of air pollution at a location quite remote from urban areas of California. Annual biological monitoring is needed to track the condition of pines in the park.

Visibility is frequently impaired at Lassen Volcanic National Park, with visibility conditions varying by season. Reconstructed light extinction generated from aerosol data showed that the worst visibility occurs during summer when the seasonal average visual range is 118 km. The best visibility conditions occur during winter when the seasonal average visual range is 223 km. The annual average visibility at the park is 165 km. Visibility impairment is largely due to organics and sulfates.

The principal resources and potential damages of concern in this park include acidification of aquatic ecosystems, visibility degradation, and ozone injury to plants. The aquatic, vegetative, and scenic resources in this park are outstanding and merit a high level of vigilance regarding resource protection. For that reason, additional resource characterization and monitoring efforts are needed. A number of additional research and monitoring activities are recommended, including chemical and biological characterization studies of lakes, and continued monitoring of lake chemistry and biology, ozone damage to bioindicator plant species, wet and dry deposition, ozone concentrations, and visibility.

Lava Beds National Monument

Lava Beds National Monument is located in a remote region of northern California. Transport from adjacent California air basins is not expected.

Ozone is measured only at a single passive ozone monitoring site within the monument. During the period 1995-1998, ozone measurements from this sampler were amongst the lowest of all parks in California.

Wet and dry deposition of sulfur and nitrogen are not measured within the monument. However, wet deposition measurements from a nearby site (85 km west of the park) are well below the minimum levels thought to be associated with vegetation damage. It is believed that dry deposition rates are not high, and total deposition rates are well below probable damage thresholds.

Surface water resources are lacking at Lava Beds National Monument and water quality is not an important air quality related value in this monument. There are no documented effects of air pollution on vegetation. Given that air quality is good at the monument, there is probably little concern about potential effects on plants in the near future. However, a survey for injury in the ozone bioindicator ponderosa pine is recommended to verify that there is currently no injury, to provide the baseline data in the event of elevated ozone levels in the future, and to provide background information to be integrated with data collected on pines as part of the Sierra-wide Forest Ozone Response (FOREST) study.

As part of the IMPROVE network expansion, a new aerosol sampler was installed in Spring 2000 to monitor visibility at the monument.

There are no significant air pollution effects concerns in this monument at the present time. Pollutant exposures are uniformly low and there are no known or suspected adverse effects on sensitive receptors.

Pinnacles National Monument

Pinnacles National Monument is located in a mostly non-urban air basin; however, pollutant transport from the San Francisco Bay Area or North Central Coast (Monterey area) Air Basins occurs. Data from the hourly ozone monitor within the monument show that ozone concentrations exceeded the federal maximum daily 1-hr hourly ozone standard (120 ppb) for two of the six years between 1992 and 1997 and consistently violated the state maximum hourly standard (90 ppb). Between 1990 and 1997, 8-hour ozone maxima at Pinnacles National Monument were 88 to 109 ppb, suggesting that compliance with the federal 8-hour ozone standard (80 ppb, 3-year average of fourth-highest annual values) may be problematic.

An NADP wet deposition monitor was recently installed within Pinnacles National Monument, but data are not yet available. Estimates of wet deposition, based on data from Salinas, show sulfur and nitrogen deposition rates well below levels thought to cause damage. Dry deposition data are available for 1996 through 1999, and show dry deposition of both sulfur and nitrogen well below levels thought to cause damage. Sulfur dioxide data collected from 1988 to 1995 indicated concentrations were well below the levels at which plant injury has been documented.

Surface water resources in Pinnacles National Monument are not sensitive to acidification from nitrogen or sulfur deposition. There are no known effects of air pollution on vegetation in the monument. It should be noted, however, that the visual characteristics of symptoms of air pollutant injury are known for relatively few species in the shrubland vegetation that dominates Pinnacles National Monument.

Visibility impairment at Pinnacles National Monument can be evenly attributed to sulfates, nitrates, organics, light absorbing carbon, and coarse mass. Smoke from frequent fires is suspected to reduce visibility during summer. Reconstructed extinction generated from aerosol data showed that the average annual visual range is 90 km. During summer the seasonal average visual range is 87 km. The worst visibility occurs during autumn, when the seasonal average visual range is 84 km, and the best seasonal average visual range is 113 km, which occurs during winter.

The primary impact of ecological concern with respect to air pollution degradation in Pinnacles National Monument is the potential effect of ozone on vegetation. Exposure to ozone is high, although adverse impacts have not been documented. Additional monitoring efforts are recommended, including passive ozone measurements at multiple locations, ozone effects monitoring of the bioindicator blue elder, and continued aerosol monitoring.

Point Reyes National Seashore

Point Reyes National Seashore is located in western Marin County, within the San Francisco Bay Area Air Basin, a predominantly urban air basin with substantial emissions of air pollutants. However, its coastal location near the northwestern edge of the air basin, combined with prevailing northwesterly winds off the Pacific Ocean, place Point Reyes National Seashore in a generally upwind position relative to the urbanized portions of the San Francisco Bay Area Air Basin. When prevailing winds change in fall, air quality and visibility are reduced.

Ozone measurements within the seashore are available from 1989 to 1992 only, when ozone concentrations were low relative to other parks within the state. Low ozone concentrations are typical of coastal locations in the San Francisco Bay Area.

Wet and dry deposition are not measured within Point Reyes National Seashore. Comparisons with two nearby sites show rates well below the level of wet deposition for sulfur

and nitrogen that are thought to cause damage. Dry deposition rates are not expected to be substantially higher than wet deposition.

There are a number of nonpoint source water quality problems at Point Reyes National Seashore, but acidification from acidic deposition is not one of them. Nitrogen and sulfur deposition are low and the streams are apparently not highly sensitive to acidification. However, the seashore has high amphibian and salmonid abundance and diversity in coastal streams, and these would be sensitive to adverse effects of acidification, if it was to occur. There are no known effects of air pollution on vegetation in the seashore. Exposure to ozone and other air pollutants at this location generally is low enough that one would not expect effects on most plant species in the foreseeable future. It should be noted that the visual characteristics of symptoms of air pollutant injury are known for relatively few species that commonly occur in the coastal plant communities in the seashore.

Visibility monitoring data show visibility impairment at Point Reyes National Seashore is largely due to sulfates, nitrates and coarse mass. Unlike many areas in the western United States, a small fraction of light extinction can be attributed to light absorbing carbon at Point Reyes National Seashore. Seasonal variations in visibility occur. Reconstructed extinction generated from aerosol data show the worst seasonal average visual range is 60 km, which occurs during summer. The best seasonal average visual range (94 km) occurs during winter. The annual average visual range is 77 km.

There are no significant air pollution effects concerns in this park at the present time. Pollutant exposures are uniformly low and there are no known or suspected adverse effects on sensitive ecological receptors.

Redwood National Park

Redwood National Park's coastal location in remote northern California, combined with prevailing northwesterly winds off the Pacific Ocean, place it in a generally upwind position relative to most emission sources.

Data from the hourly ozone monitor located within the park show that ozone concentrations and exposures for the period 1988 to 1995 were low relative to other areas of the state; data for later years are unavailable.

Wet and dry deposition rates for sulfur and nitrogen are not available within the park. However, comparisons can be made with measurements from a nearby site (10 km northeast of the park). These data show that both wet and dry non-marine sulfur and nitrogen deposition are well below levels thought to cause damage.

Streams in Redwood National Park are moderately dilute and some may be somewhat sensitive to acidification from acidic deposition. Atmospheric inputs of nitrogen and sulfur are very low, however, and there appears to be no real risk of future adverse aquatic effects. It should be noted, however, that Redwood National Park contains an abundant and diverse amphibian fauna, which could be sensitive to acidification, and also an important anadromous fisheries resource, which is highly sensitive to acidification damage.

There are no known effects of air pollution on vegetation in Redwood National Park. Exposure to ozone and other pollutants at this location generally is low enough that one would not expect effects on most plant species in the foreseeable future. However, a survey for injury in Jeffrey pine is recommended to verify the absence of current injury, to provide baseline data in the event that pollutant exposure increases dramatically in the future, and to provide background data on pines to be integrated with the Sierra Nevada FOREST study.

Visibility impairment frequently occurs at Redwood National Park and is largely due to sulfates, nitrates, and organics. Visibility conditions vary by season and are strongly influenced by weather, winds, and coastal fog. Reconstructed extinction generated from aerosol data showed the annual average visual range is 79 km. The seasonal average visual range in the summer is 79 km, and the worst seasonal average visual range is 73 km, which occurs in the autumn. The best seasonal average visual range is 139 km, which occurs during winter.

There are no significant air pollution effects concerns in this park at the present time. Pollutant exposures are uniformly low and there are no known or suspected adverse effects on sensitive ecological receptors.

Sequoia and Kings Canyon National Parks

Sequoia and Kings Canyon National Parks are located within the San Joaquin Valley Air Basin, and are exposed to pollutants transported from the San Joaquin Valley and from the San Francisco Bay Area Air Basin. Several air quality monitoring sites are located within the parks.

Ozone levels are high at all of the monitoring sites. The annual daily 1-hour maximum exceeded the state standard (90 ppb) in every year that data were available, and regularly exceeded the federal standard (120 ppb). SUM60 indexes were well over 100,000 ppb-hours for all sites for most years. Passive ozone samplers show that the highest ozone levels are evident at the western end of Sequoia National Park, adjacent to the San Joaquin Valley.

Wet deposition of sulfur in the parks has been below levels thought to cause damage to vegetation. However, wet deposition of nitrogen, though generally below levels thought to cause damage, was above the minimum suspected damage thresholds during some years at some sites. Dry deposition data are available for 1999 from a CASTNet site installed at Lookout Point, during which time dry nitrogen deposition was moderate but dry sulfur deposition was low. At higher elevations (above 2700 m), a substantial portion of the total annual deposition is released rapidly during spring when the accumulated snowpack melts, and therefore nitrogen loading may be high during short time periods.

There has been a great deal of research on the chemistry and biology of lakes within Sequoia and Kings Canyon National Parks. Significant effort has been expended to determine the extent to which current levels of acidic deposition may have contributed to chronic and/or episodic acidification and the extent of acidification that would be required to elicit adverse biological impacts. The results of these studies paint a rather complete picture of the current acid-base status and biological conditions of freshwater resources in Sequoia and Kings Canyon National Parks. These results are also very relevant to Yosemite National Park, and likely to acid-sensitive high-elevation aquatic resources in Lassen Volcanic National Park.

Alpine and subalpine lakes and streams in Sequoia and Kings Canyon National Parks are extremely acid-sensitive. A high percentage have acid neutralizing capacity < 50 µeq/L and many have acid neutralizing capacity below 30 µeq/L. It is possible (but not demonstrated) that some of the most acid-sensitive lakes and streams have been chronically acidified by current levels of acidic deposition, but such acidification, if it has occurred, has been quite small in magnitude. It is likely that many of the lakes and streams that exhibit lowest acid neutralizing capacity experience episodic acidification in response to current levels of sulfur, and especially nitrogen, deposition. Such an effect is superimposed on natural episodic acidification processes, especially base cation dilution.

There is no evidence to suggest that current levels of acidic deposition have caused either chronic or episodic biological effects. Available data suggest that such effects are unlikely. It is very likely, however, that adverse biological effects would be manifested under moderately

increased deposition of sulfur or nitrogen. Baseline conditions have been rather well described. Continued monitoring of aquatic chemistry and biology would be able to document if and when future changes occur.

Sequoia and Kings Canyon National Parks have the most severe effects of air pollution on vegetation in California outside the Los Angeles Basin, more impacts than any other western national park. Ozone-induced foliar injury in ponderosa pine and Jeffrey pine is widespread and severe in the western regions of the parks. Reduced foliar biomass and growth at some locations indicate that ecosystem productivity has clearly been affected by chronic, long-term exposure to elevated levels of ozone in the mixed conifer zone. Foliar injury has been documented in giant sequoia seedlings but not in mature trees. Long-term monitoring of forest condition and productivity is needed at these parks, with potential expansion to other known bioindicators of ozone injury.

Impaired visibility frequently occurs, largely due to organics, sulfates, and nitrates. Smoke occasionally causes reduced visibility during the summer months. Visibility conditions vary by season. Reconstructed extinction generated from aerosol data showed that the worst visibility occurs in the summer, when the seasonal average visual range is 60 km. The best visibility conditions occur in the winter when the seasonal average visual range is 93 km. The annual average visibility at Sequoia and Kings Canyon National Parks is 63 km.

These parks have several significant resource concerns related to the current high (ozone) to moderate (nitrogen deposition) pollutant exposures that the parks receive. Ozone damage to sensitive vegetation is high, as is visibility degradation, both primarily in the western portions of the parks. Aquatic effects from sulfur and nitrogen deposition are currently not significant, but aquatic resource sensitivity is very high and therefore continued monitoring is important. A number of recommendations are provided, primarily focused on continuation of monitoring activities, including wet and dry deposition, ozone concentration, water quality, ozone injury to vegetation, and aerosol concentrations.

Yosemite National Park

Yosemite National Park, located in the central Sierra Nevada, is adjacent to the San Joaquin Valley. It is affected by emissions transported from the San Joaquin Valley and may also be affected by pollutant transport from the San Francisco Bay Area Air Basin. Several monitoring sites are located within the park.

At three sites where ozone is monitored, the second-highest annual daily 1-hour maxima were below the federal one-hour ozone standard (120 ppb) during all years from 1992 through 1997, but the California hourly-ozone standard of 90 ppb was exceeded at all sites. Passive ozone sampling was conducted at several sites within the park; no discernable spatial pattern was evident.

Wet deposition rates for sulfur and nitrogen were measured at three sites within the park. All sites showed deposition rates below the levels thought to cause vegetation damage. Dry deposition of sulfur and nitrogen was measured at one site, similarly showing levels below those thought to cause damage.

High-elevation lakes and streams in Yosemite National Park are extremely sensitive to acidification from even moderate levels of sulfur or nitrogen deposition. Some slight episodic, and perhaps chronic, acidification may occur under current levels of deposition. It is unlikely that such acidification, if it has occurred, has adversely impacted biota to date. Little research or monitoring has been conducted on sensitive, high-elevation aquatic resources in the park, however. A long-term monitoring program is recommended.

Yosemite National Park has severe ozone-induced foliar injury in ponderosa pine, the second worst impacts in a western national park. Ozone injury is widespread in the western portion of the park and foliar biomass is lower in symptomatic trees, although there is no indication that ozone has reduced tree growth. Long-term monitoring of forest condition and productivity is needed at Yosemite National Park for the foreseeable future.

Visibility impairment at the park is largely due to organics and sulfates. Smoke from frequent fires is suspected to reduce visibility during the summer months. Reconstructed extinction generated from the aerosol data at Yosemite National Park showed seasonal variations in visibility conditions. The best seasonal average visual range (211 km) occurs during the winter months, and the worst seasonal average visual range (107 km) occurs during the summer months. The annual visual range is 124 km.

Yosemite National Park shares the same resource concerns as Sequoia and Kings Canyon National Parks. Ozone damage to sensitive vegetation is substantial. Visibility degradation is an important issue. Surface water resources, although not currently impacted to any significant extent, are highly sensitive to acidification if sulfur or nitrogen deposition increase substantially in the future. Continued monitoring of wet and dry deposition, ozone concentration, water quality, ozone injury to vegetation, and aerosol concentrations are recommended.

6. Recommendations

Additional work is recommended in many of the Class I parks to better characterize the extent of current damage to air quality related values. In particular, more information is needed regarding the extent of the damage that is occurring to vegetation from exposure to high concentrations of ozone.

We recommend that the National Park Service consider at least some additional monitoring of air quality in most of the parks discussed in this report. For example, short-term quantification of the spatial patterns of ozone exposure using passive samplers is an inexpensive way to establish a reference point in time. While continuous analyzer data for ozone are more expensive to obtain, collection of such data is the best way to obtain a reference with which future conditions can be compared.

Monitoring in combination with modeling is an ideal mechanism for estimating the impacts of local sources, especially for aquatic effects. Several of the Class I parks in California contain some of the most acid-sensitive lakes in the world; relatively small increases in nitrogen or sulfur deposition would cause significant adverse effects to aquatic biota in highly sensitive lakes. Although the current chemistry of acid-sensitive lakes in Sequoia, Kings Canyon, and Yosemite National Parks is reasonably well established, additional measurements of acid neutralizing capacity and other characteristics of high mountain lakes in Lassen Volcanic National Park will establish a reference and identify potentially sensitive resources, should air quality deteriorate in the future. Types and locations of monitoring are suggested in this report, in the event that there is sufficient interest and funding for such activities in the future. Regional modeling studies are also suggested to ascertain the deposition levels at which highly-sensitive aquatic receptors would be damaged.

Ongoing and future visibility monitoring is necessary in the California NPS Class I areas to identify sources and extent of visibility impairment. Data and in-depth modeling and analysis are required to further evaluate historical trends and future projections of impacts from existing and future sources. Future research is also recommended in the Class I areas to minimize the uncertainty in estimates of how aerosol species affect visibility.

I. INTRODUCTION

National parks and other Class I areas in California include lands of exceptional ecological and cultural significance. The National Park Service (NPS) maintains the world's most admired and imitated system of parks. Millions of visitors each year are attracted by the outstanding scenery and the unspoiled nature of park ecosystems. However, the ecological integrity of these ecosystems is threatened by increasing demands on park usage and pollutant emissions outside park boundaries. Air quality is fundamentally important to the preservation of healthy ecosystems. Most of the Class I national parks, national monuments, and the national seashore in California receive generally low levels of atmospheric pollutants (Sisterson et al. 1990, Smith 1990). Nevertheless, sensitive aquatic and terrestrial ecosystems, especially those at high elevation and downwind of major urban, agricultural or industrial areas, can potentially be degraded by existing or future pollution or in some cases have been degraded by existing pollution (D. Peterson et al. 1992a). Elevated emission levels of nitrogen oxides and elevated concentrations of ozone (O_3) have been measured near and in some Class I areas within the state (Sisterson et al. 1990). Some areas also have impaired visibility.

Recognizing the valuable role that scientific research can play in proper park management, the NPS Director commissioned the National Research Council to review the NPS's research program. The review committee (National Research Council 1992) concluded that there was an urgent need to accelerate research in the parks to:

- inventory the natural resources in order to protect and manage the resources and detect changes;
- better understand the natural ecosystems in the parks; and
- assess specific threats to the parks.

In accordance with these recommendations, the Air Resources Division (ARD) of the NPS initiated a series of projects to assess air quality in selected parks. The subject of this report is one such project, which is designed to take a proactive position in assessing potential threats from air pollution to the Class I national parks, monuments, and seashore in California.

To maintain healthy ecosystems and protect visibility, it is imperative that land managers monitor and assess levels of atmospheric pollutants and ecological effects in the lands managed by the NPS. Knowledge of emissions inventories, coupled with scientific understanding of transport mechanisms and the effects of pollutants on natural resources, will provide land managers with a framework within which to protect sensitive resources from degradation due to atmospheric pollutants.

The Clean Air Act and the NPS Organic Act provide mandates for protecting air resources in NPS areas. In Section 160 of the Clean Air Act, Congress stated that one of the purposes of the Act is to "preserve, protect and enhance the air quality in national parks, national wilderness areas, national monuments, national seashores and other areas of special national or regional natural, recreation, scenic, or historic value." According to the Clean Air Act and subsequent amendments, federal land managers have "...an affirmative responsibility to protect the air quality related values (AQRVs)...within a Class I area." AQRVs include visibility, flora, fauna, bodies of water and other resources that may be potentially damaged by air pollution. A Class I designation allows only small increments of pollution above already existing levels within the area. National Parks over 2,430 ha (6,000 ac) and national wilderness areas over 2,025 ha (5,000 ac), that were in existence before August of 1977, are designated as Class I areas. All other areas managed by the NPS are designated Class II, and a greater amount of air quality degradation is allowed by the Clean Air Act.

The NPS Class I areas within California include (alphabetically) Joshua Tree National Park (JOTR), Lassen Volcanic National Park (LAVO), Lava Beds National Monument (LABE), Pinnacles National Monument (PINN), Point Reyes National Seashore (PORE), Redwood National and State Parks (REDW), Sequoia/Kings Canyon National Parks (SEKI), and Yosemite National Park (YOSE; Figure I-1).

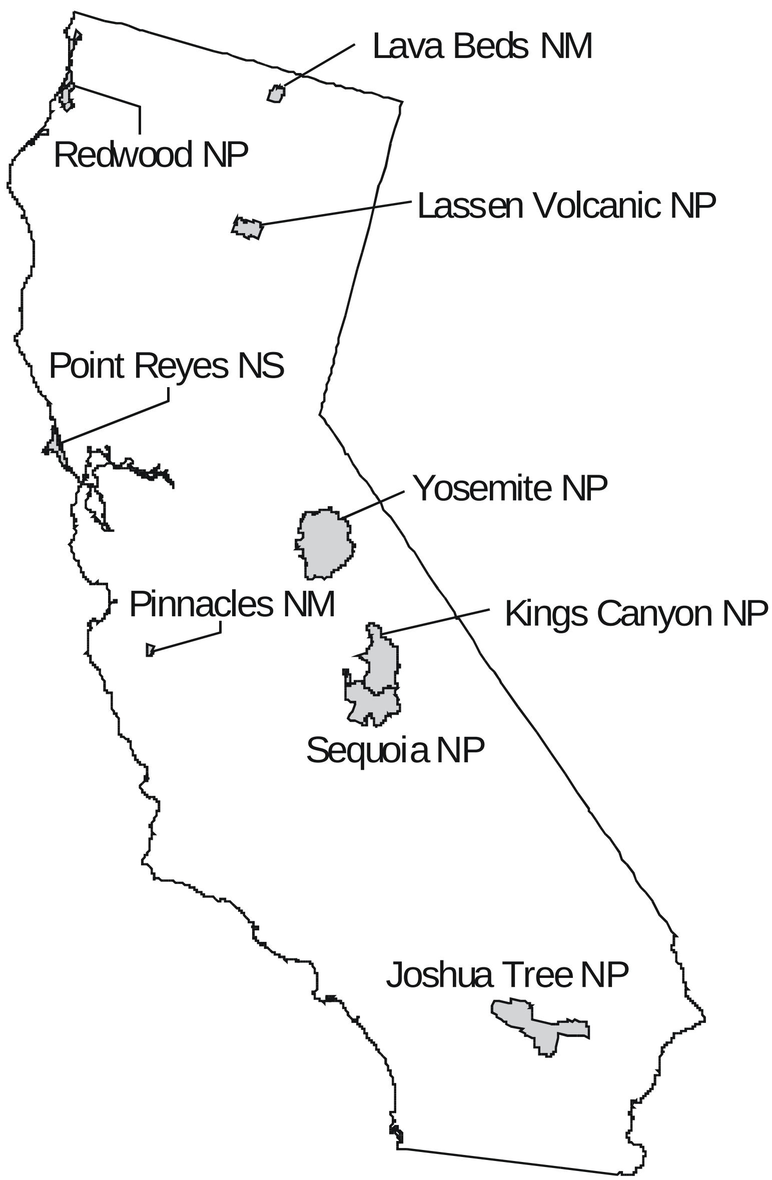


Figure I-1. Map of California showing location of the Class I national parks, monuments, and seashore.

A. OBJECTIVES

The principal goal of this report is to evaluate the status of air quality and air pollution effects and to identify information needs for air quality related issues in NPS Class I areas in California. To support the mandate to protect AQRVs in Class I areas, the following specific objectives have been identified for the report:

- provide updated summaries of monitoring data on visibility and on pollutant concentrations and deposition, both temporally (hourly, seasonally, annually) and spatially (regional, statewide, park);
- conduct comprehensive analyses of documented and potential ecological effects of various atmospheric pollutants and exposures (chronic, episodic) on terrestrial and aquatic systems;
- compile inventories of pollution-sensitive components or receptors of ecosystems, and the critical or target loading of pollutants that would be likely to cause changes in the sensitive receptors;
- assess key knowledge deficits and additional information required to adequately protect resources sensitive to potential degradation by poor air quality.

The report addresses these objectives by providing a summary of current and historical monitoring data for pollutants, a description of the resource base in each area, a synthesis of knowledge on the ecological effects of atmospheric pollutants, and a park-specific assessment of pollution vulnerability.

B. SCOPE AND ORGANIZATION

The scope of this report is limited to addressing potential threats in Class I areas to: (1) terrestrial resources (primarily from nitrogen (N) and sulfur (S) deposition [including gaseous forms], and ozone exposure), (2) aquatic resources (primarily from N and S deposition), and (3) visibility (primarily from particulates and aerosols). Exposure to trace metals, pesticides, radionuclides, and organic toxins are not addressed or are mentioned only in a cursory fashion.

Although the report attempts to address many of the critical issues facing each park, partial coverage of some topics should not be interpreted as a judgment that these topics are not important or relevant to the issue of air pollution effects. These omissions often reflect the lack of information on these topics rather than their ecological significance.

It is hoped that this report will serve audiences including staff with the NPS Air Resources Division, regional air quality coordinators, individual park staff, and organizations dealing with air quality issues in California. The report is structured to present relevant information on regional issues and then to discuss individual NPS Class I areas.

It should be noted that some aspects of measuring air pollution and air pollution effects are evolving and scientists remain divided with respect to appropriate assessment techniques. We have not attempted to resolve these issues in this report but have simply identified acceptable monitoring strategies.

C. BACKGROUND

1. Air Quality Regulations

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 I-1). Standards were established for the pollutants that are emitted in significant quantities throughout the country and that may be a danger to public health and welfare. The primary NAAQS is designed to protect human health while the secondary NAAQS is designed to protect the public welfare from the adverse effects of the pollutant. The Clean Air

Act defines public welfare effects to include, but not be limited to, "effects on soils, water, crops, vegetation, manmade materials, animals, wildlife, weather, visibility and climate, damage to and deterioration of property, and hazards to transportation, as well as effects on economic values and on personal comfort and well-being". The standards are defined in terms of deposition-averaging times, such as annual or hourly, depending on the type of exposure associated with health and welfare effects. For some pollutants, there are both short-term and long-term standards. Criteria pollutants include ozone (O_3), carbon monoxide (CO), nitrogen dioxide (NO_2), sulfur dioxide (SO_2), particulate matter less than 10 µm (PM-10), particulate matter less than 2.5 µm (PM-2.5) and lead (Pb). Baseline data on criteria pollutants collected by a national monitoring system are used to determine if the NAAQS are met and to track pollutant trends.

Air quality within Class I lands of the NPS system is subject to the "prevention of significant deterioration (PSD)" provisions of the Clean Air Act. The primary objective of the PSD provisions is to prevent substantial degradation of air quality in areas that comply with NAAQS, and yet maintain a margin for industrial growth. A PSD permit from the appropriate

Table I-1. National Ambient Air Quality Standards.

Constituents	Averaging Time	Primary Standard	Secondary Standard
Sulfur Dioxide	Annual Arith. Mean 24-hour[a] 3-hour[a]	30 ppbv 140 ppbv none	none none 500 ppbv
PM_{10}	Annual Arith. Mean[b] 24-hour[b]	50 µg/m^3 150 µg/m^3	same same
$PM_{2.5}$[d]	Annual[b] 24-hour[b]	15 µg/m^3 65 µg/m^3	same same
Carbon Monoxide	8-hour[a] 1-hour[a]	9,000 ppbv 35,000 ppbv	same same
Ozone	8-hour[c,d] 1-hour	80 ppbv 120 ppbv	same same
Nitrogen Dioxide	Annual Arith. Mean	50 ppbv	same
Lead	Calendar Quarter	1.5 µg/m^3	same

[a] Concentration is not to be exceeded more than once per calendar year.
[b] Compliance is based on concentrations averaged over a 3-year period.
[c] Compliance is based on a 3-year average of the annual fourth-highest daily maximum 8-hour concentration.
[d] Standards under litigation; 1-hr zone standard in effect until issue resolved.

air regulatory agency is required to construct a new pollution source or modify an existing source (Bunyak 1993). A permit application must demonstrate that the proposed polluting facility will (1) not violate national or state ambient air quality standards, (2) use the best available control technology to limit emissions, (3) not violate either Class I or Class II PSD increments for SO_2, NO_2, or particulate matter (Table I-2), and (4) not cause or contribute to adverse impacts to AQRVs in any Class I area (D. Peterson et al. 1992a).

Table I-2. Prevention of significant deterioration increments (in µg/m³); PSD increments are not defined for ozone or PM-2.5.

Constituent	Averaging Time	Class I	Class II	Class III
Sulfur Dioxide	Annual Arith. Mean	2	20	40
	24-hour	5	91	182
	3-hour	25	512	700
PM-10	Annual Arith. Mean	4	17	34
	24-hour	8	30	60
Nitrogen Dioxide	Annual Arith. Mean	2.5	25	50

The PSD increments are allowable pollutant concentrations that can be added to baseline concentrations. The values chosen as PSD increments by Congress were not selected on the basis of concentration limits causing impacts to specific resources. Therefore, it is possible that pollution increases exceeding the legal Class I increments may not cause damage to Class I areas. It is also possible that resources in a Class I area could be affected by pollutant concentrations that do not exceed the increments. The role of the federal land manager is to determine if there is potential for additional air pollution to cause damage to sensitive receptors whether or not the PSD increments have been exceeded. Even if a proposed facility is not expected to violate Class I increments, the federal land manager can still recommend denial of a permit by demonstrating that there will be adverse impacts in the Class I area. Provisions for mitigation can be recommended by the federal land manager to the agency that issues permits.

The following questions must be answered for PSD permit applications:

- What are the identified sensitive AQRVs in each Class I area that could be affected by the new source?
- What are the air pollutant doses that may affect the identified sensitive AQRVs?
- Will the proposed facility result in pollutant concentrations or atmospheric deposition that will cause or contribute to the identified critical dose being exceeded?
- If the critical dose is exceeded, what amount of additional pollution is considered "insignificant"?

The first two questions are land management issues that should be answered on the basis of management goals and objectives for the Class I area. The last two are technical questions that must be answered on the basis of analyses of emissions from the proposed facility and predictions of environmental response to a given pollutant concentration (Peterson et al. 1992a,b).

The 1999 Regional Haze Regulations require states (and tribes who choose to participate) to review how pollution emissions within the state affect visibility at Class I areas across a broad region (not just parks within the state). These rules also require states to make "reasonable progress" in reducing any effect this pollution has on visibility conditions in Class I areas and to prevent future impairment of visibility. The states are required by the rule to analyze a pathway that takes the Class I areas from current conditions to "natural conditions" within 60 years. "Natural conditions" is a term used in the Clean Air Act, which means that no human-caused pollution can impair visibility. This program, while aimed at Class I areas, will improve regional visibility conditions throughout the country.

In May, 1998, the U.S. EPA issued the Interim Air Quality Policy on Wildland and Prescribed Burning, for federal land managers and air regulators. Objectives of the new policy

include avoidance of public health effects from increased prescribed burning and improvement of visibility in Class I areas. States are required to develop Smoke Management Programs. California issued revised Title 17, Smoke Management Guidelines for Agricultural and Prescribed Burning. The objectives of Title 17 are to:

- provide increased opportunity for burning,
- develop a statewide smoke management network,
- minimize smoke and health impacts on the public, and
- comply with EPA's interim policy (S. Ahuja, USDA Forest Service, Mendocino National Forest, pers. Comm.)

2. Deposition

Given the structural complexity of landforms within California and the broad distribution of major pollution source areas, a network of air-quality and atmospheric deposition monitors in or near Class I areas is needed to quantify local deposition rates accurately. However, general patterns of N and S deposition can be inferred from state and national databases. It is known that air pollution in some areas has increased considerably during the last 30 to 40 years, while decreasing in others. Dispersion and transport of pollutants vary locally. The challenge is to quantify the spatial distribution of this exposure relative to the location of sensitive receptors. Establishment of a current reference point for air quality, in combination with additional monitoring, is needed to detect long-term trends.

The estimation of deposition of atmospheric pollutants in high-elevation areas in the western United States is especially difficult because all components of the deposition (rain, snow, cloudwater, dryfall and gases) have seldom been measured concurrently. Even measurement of wet deposition remains a problem because of the logistical difficulties in operating a site at high elevation. Portions of the wetfall have been measured by using snow cores (or snow pits), bulk deposition, and automated sampling devices such as those used at the National Atmospheric Deposition Program/ National Trends Network (NADP/NTN) sites. All of these approaches suffer from limitations that cause problems with respect to developing annual deposition estimates. The snow sampling includes results for only a portion of the year and may seriously underestimate the load for that period if there is a major rain-on-snow event. Bulk deposition samplers are subject to contamination problems from birds and litterfall. Weekly ambient concentration data are available at some sites within the state, collected within the Clean Air Status and Trends Network (CASTNet), from which dry deposition rates will eventually be calculated when the methods of calculation have been satisfactorily refined.

The need to measure or estimate cloudwater, dryfall and gaseous deposition complicates the difficult task of measuring and monitoring total deposition. Cloudwater can be an important portion of the hydrologic budget in coastal areas and some high elevation forests (Harr 1982), and failure to capture this portion of the deposition input could lead to substantial underestimation of annual deposition. Furthermore, cloudwater chemistry has the potential to be more acidic than rainfall. Dryfall from wind-borne soil can constitute major input to the annual deposition load, particularly in arid environments. Aeolian inputs from dryfall can provide a major source of acid neutralization not generally measured in other forms of deposition. Gaseous deposition is calculated from the product of ambient air concentrations and estimated deposition velocities. The derivation of deposition velocities is subject to considerable debate. In brief, there is great uncertainty regarding the amount of current deposition of atmospheric pollutants in the Class I areas and throughout many of the mountainous regions of the western United States.

3. Gaseous Pollutants of Concern

a. Ozone

Ozone is a secondary pollutant formed by the photooxidation of nitrogen oxides (NO_x) and volatile organic compounds (VOCs). Ozone is a colorless gas and is a component of photochemical haze which can develop during the clear warm weather associated with high pressure systems. Ozone is an important regional pollutant because it forms during transport of its precursors (VOC and NO_x), and can occur at high concentrations in areas remote from precursor sources. Typical tropospheric average background concentrations are generally thought to be in the range of 25 to 40 ppbv; the exact values have been the subject of some debate, and background levels could exceed 40 ppbv at some times and places. However, the weekly average levels of ozone in "pristine" areas can be as low as 10 to 25 ppbv (Altshuller and Lefohn 1996, Cooper and Peterson 2000), with maximum hourly ozone concentrations generally less than about 50 to 80 ppbv. Some areas in southern California experience ozone concentrations exceeding 150 to 200 ppbv (maximum hourly concentrations). Ozone is a potential threat to high-elevation plant species because concentrations and total exposure can be higher at higher elevations under appropriate atmospheric conditions (Loibl et al. 1994; Sandroni et al. 1994; Brace and Peterson 1996,1998). At some locations in the Sierra Nevada and southern California, ozone concentrations and total exposure increase with elevation to 1200-1500 m, then decrease (Van Ooy and Carroll 1995, Bytnerowicz et al. 1999a).

Ozone causes injury to highly sensitive species of plants at concentrations as low as 40 to 60 ppbv (Treshow and Anderson 1989). Ozone enters plant leaves as a gas and dissolves in the presence of water. The resulting free radicals oxidize proteins of cell membranes, including those of the thylakoid membranes where photosynthesis takes place. Injury includes leaf discoloration, reduced photosynthetic rates, lowered sugar production, reduced growth and possibly death.

There is a broad range of plant species sensitivities to ozone. Available plant injury data are therefore not consistent in suggesting a damage threshold. In general, we can conclude that there may be adverse effects with prolonged exposure to ozone concentrations greater than 60 ppb, and there are likely effects with prolonged exposure to concentrations greater than 80 ppb.

It is helpful for park managers to know approximate thresholds of cumulative air quality degradation at which potential injury to plants might be expected. This topic has been discussed at length by air pollution experts in an effort to establish a secondary standard as part of the National Ambient Air Quality Standards. Although there is considerable professional disagreement about this issue, Heck and Cowling (1997) note that a range of 3-month, 12-hour SUM06 ppm-hour values can be used as a first-order guide for foliar injury to vascular plants:

8-12 ppm-hr	Foliar injury in natural ecosystems
10-15 ppm-hr	Growth effects to tree seedlings in natural forest
12-16 ppm-hr	Growth effects to tree seedlings and saplings in plantations
15-20 ppm-hr	Yield reductions of 10% in agricultural crops

Because of the wide range of variability in the sensitivity of plant species to ozone exposure, the actual range of SUM06 values at which injury is observed may vary considerably.

b. Sulfur Dioxide

Sulfur dioxide (SO_2) is a product of fossil fuel combustion. Some of the largest emitters are coal-fired electric power plants and smelters. However, no such large emitters (> 1000 tons/year) are located in California. Elsewhere, forest dieback near power plants has been

documented since the mid 1800s. Although more stringent regulations have reduced emissions over the last 50 years, SO_2 continues to be a major pollutant of concern in many areas of the United States. SO_2 is a precursor of pollutants which cause acidic deposition and visibility impairment.

Like ozone, SO_2 is a gas and enters the leaf through the stomata. Inside the leaf it disrupts mesophyll cell functioning, causing reduced productivity of the leaf. SO_2 injury in plants is characterized by leaf bleaching and chlorosis, necrotic lesions, and early senescence. Prolonged exposure can weaken a plant making it susceptible to pathogens and other organisms. Some species are sensitive to chronic exposures of as low as 25 ppbv (Treshow and Anderson 1989).

There are few data on the effects of S compounds on mature trees or other native plants, and there is a wide range of sensitivities to ambient S compounds (Davis and Wilhour 1976, Smith 1990). Limited data on tree seedlings (Hogsett et al. 1985, P.R. Miller unpublished data) indicate that SO_2 concentrations below 20 ppbv (24-hour mean) do not produce visible injury symptoms. For ponderosa pine (*Pinus ponderosa*) and lodgepole pine (*P. contorta*), slight injury is found with chronic exposures above 40 ppbv and moderate injury above 65 ppbv. Slight injury is found for Douglas-fir (*Pseudotsuga menziesii*) above 65 ppbv. In order to maximize protection of all plant species, acute SO_2 concentrations should not exceed 40 to 50 ppbv, and annual average 24-hour SO_2 concentrations should not exceed 8 to 12 ppbv.

c. VOCs and NO_x

Volatile organic compounds (VOCs) and nitrogen oxides (NO_x) are not criteria pollutants, but they are important precursors of ozone. NO_x is also a precursor for pollutants that cause acidic deposition and visibility impairment, and VOCs can contribute to visibility impairment. Automobiles and stationary fossil fuel burning systems are the major anthropogenic sources of NO_x in the United States. Naturally occurring NO_x compounds originate from soils, wildfire, lightning and decomposition. Biogenic sources of NO_x are comparable to or less than anthropogenic sources in most areas (Chameides et al. 2000).

Anthropogenic sources of VOCs include motor vehicle exhaust, gasoline vapors, stationary fuel combustion, commercial and industrial processes, and emissions from solid wastes (Smith 1990). Natural systems, particularly soils and vegetation, produce VOCs and emit them to the atmosphere; trees in particular emit the highly reactive hydrocarbons isoprene and terpene. Globally, biogenic sources of VOC exceed anthropogenic sources, while in localized, urban areas anthropogenic sources typically dominate. VOCs include a large number of hydrocarbons which vary greatly in reactivity (Chameides et al. 2000).

4. Vegetation and Bioindicators

Gaseous air pollutants can reduce photosynthesis, growth, and productivity of sensitive plant species (Treshow and Anderson 1989, Smith 1990, Runeckles and Chevone 1992, Chappelka and Chevone 1992), even at relatively low exposure levels (c.f., Reich and Amundson 1985). Some species are sensitive to episodes of high ambient exposure, while others are sensitive to lower chronic exposure over an extended period of time. Plants that are physiologically stressed from air pollutants can have reduced vigor, which in turn can lead to greater susceptibility to additional stresses (Innes 1993), such as insect attack (Pronos et al. 1999) and fungal pathogens (Fenn and Dunn 1989).

Oxidant air pollutants are a particular concern for national park lands in California and elsewhere in the western United States. National parks can experience greater cumulative ozone exposure, higher minimum values, and higher concentrations than the upwind urban areas where most of the pollutant sources are located (Böhm 1992). In many places, ozone concentrations

tend to increase with elevation (Brace and Peterson 1998, Cooper and Peterson 2000), although there are some exceptions (Bytnerowicz et al. 1999a). This can lead to elevated values in mountainous locations. Outside California, it has generally been difficult to estimate ozone exposure in wildlands because of a sparse dataset collected primarily in urban areas. Fortunately, a spatially dispersed set of monitoring data has been collected in California, including samples from several national parks collected over a significant period of time. Such data make it possible to quantify the relationship between air quality and observed effects for a limited set of sensitive species for much of the state (Arbaugh et al. 1998, Takemoto 2000, Takemoto et al. 2000).

Bioindicators are those species for which pollutant sensitivity has been documented and for which data exist on their dose-response to pollutants and on symptomatology. In some cases, bioindicators detect exposure of a pollutant at a site where air quality monitoring data are not available. Ozone and SO_2 are the most extensively studied pollutants regarding impacts on vegetation. Some of this work has fortunately been conducted on native Californian species and at various locations in the state. However, it should be noted that identifying symptoms of air pollutant injury is difficult, because visual symptoms are generally poorly documented for vegetation in western North America, and only a small fraction of the thousands of native plant species in California have been screened for sensitivity to air pollutants. Lichens are often cited as a taxonomic group that is sensitive to air pollution (e.g., Sigal and Nash 1983, Nash and Gries 1991). Unfortunately, despite the work conducted to date, diagnostic symptoms of injury are difficult to verify and quantify in the field (Stolte et al. 1993), and like vascular plants, symptoms have been documented for relatively few species in California.

a. Symptomatology

Pollutants can cause injury to various plant tissues including leaves, stems and roots. Foliar injury is the most visible form of damage, although it often can be confused with other biophysical injuries, including abrasion, desiccation, insect herbivory, and fungal pathogens. Diagnosis of pollutant injury in the field is often difficult, and should be verified in conjunction with a plant pathologist or other plant scientist familiar with air pollutant symptomatology.

Ozone symptoms in conifer, hardwood, and herbaceous foliage that could be considered "typical" include chlorosis, stipple (uniform black spots), and accelerated needle and leaf loss (Miller et al. 1983; Thompson et al. 1984a,b; Hogsett et al. 1989; Treshow and Anderson 1989; Stolte 1996; Brace et al. 1999). Common symptoms of SO_2-induced foliar injury are summarized in Table I-3 (Krupa and Manning 1988, Krupa et al. 1998).

Injury induced by SO_2 includes interveinal necrosis, dieback of leaf and needle tips, and necrotic spots (Thompson et al. 1980, 1984a,b; Treshow and Anderson 1989; Legge et al. 1998). Common symptoms of SO_2-induced foliar injury are summarized in Table I-4 (Legge et al. 1998).

There are few data or descriptions of the effects of nitrogen oxides on plant species, with the best description of symptomatology in Bytnerowicz et al. (1998). Concentrations of nitric oxide (NO) and nitrogen dioxide (NO_2) at locations remote from point sources are rarely high enough to cause visible injury symptoms. The adverse effects from long-term exposures can be reduced photosynthesis and growth in the absence of visible injury (Taylor 1968, Saxe 1994). At moderate levels, NO_x generally stimulates plant growth. If short-term, acute exposures of NO and NO_2 occur from sources adjacent to sensitive vegetation, the most common symptoms include necrotic lesions, chlorosis and other discoloration, and accelerated leaf senescence (Bytnerowicz et al. 1998).

Table I-3. Common symptoms of ozone-induced foliar injury (from Krupa and Manning 1988, Krupa et al. 1998).

Acute Injury	Chronic Injury
<u>Conifers</u>	
• Banding; clear bands of chlorotic tissue develop on semi-mature needle tissue following ozone episodes. • Tipburn; characterized by dying tips of young elongating needles. At first reddish-brown in color later turning brown, injury spreading	• Flecking and mottling; flecking is the earliest symptom on the older needles of conifers. Mottling is generally associated with diffuse chlorotic areas interspersed with green tissue on first-year needles. • Premature senescence; early loss of needles.
<u>Broad-leaved plants (hardwood trees, herbaceous plants)</u>	
• Bleaching (unifacial or bifacial); small unpigmented necrotic spots or more general upper surface bleaching. Palisade cells and, where injury is more severe, upper epidermal cells collapse and become bleached. • Flecking; small necrotic areas due to death of palisade cells, metallic or brown, fading to tan, gray or white. • Stippling; small punctate spots where a few palisade cells are dead or injured, may be white, black, red, or red-purple. • Bifacial necrosis; when the entire tissue through the leaf is killed, bifacial dead areas develop, ranging in color from white to dark orange-red. While small veins are usually killed along with the other tissue, larger veins frequently survive	• Pigmentation (bronzing); leaves turn red-brown to brown as phenolic pigments accumulate. • Chlorosis; may result from pigmentation or may occur alone as chlorophyll breakdown. • Premature senescence; early loss of leaves, flowers, or fruit.

Table I-4. Common symptoms of SO_2-induced foliar injury (from Legge et al. 1998).

Acute Injury	Chronic Injury
Conifers	
• Reddish-brown tip necrosis extending downward. With successive acute exposures distinct banding of necrotic tissue may develop in older needles. Premature shedding of older needles common.	• Chlorosis at the tip spreading downward. May lead to premature leaf senescence.
Broad-leaved plants (hardwood trees, herbaceous plants)	
• Interveinal necrosis sometimes starting at the margins with an initial grayish-green water-soaked appearance. Upon drying, these areas exhibit a bleached ivory color in most species, and in others a brown, red or black color. In contrast, on compound leaves injury may consist of irregular necrotic or dead areas between the side veins, often closer to the midrib. Similarly in lobate or palmately-veined leaves, the necrotic areas generally appear closer to the point of branching of the main veins rather than towards the margin.	• Interveinal chlorosis and sometimes silvery appearance. Accumulation of non-green pigmentation, leading to premature leaf senescence.
Monocotyledonous plants (narrow-leaved)	
• Yellowish-white or ivory-colored necrotic streaks that develop near the leaf tips and extend downward between the veins toward the base of the leaf. Leaf margins are also commonly necrotic.	• Interveinal chlorosis starting at the tip and extending downward towards the base of the leaf.

Reference photos and descriptions of pollutant injury in western plant species can be found in Thompson et al. (1984a), Flagler (1998), and Brace et al. (1999). Flagler (1998) also includes descriptions and photos of symptomatology for air pollutants other than those discussed in this report.

b. Documentation of Air-Pollutant Effects in California

California has benefitted from a long history of scientific investigations of the effects of air pollution on vegetation, going back to the initial identification of ozone injury to ponderosa pine in the 1950s. Four decades of research and monitoring, particularly the long-term studies of USDA Forest Service scientist Paul Miller, have clearly established quantitative linkages

between elevated levels of ambient ozone and a range of effects in common forest species of California. Much of what we know about the spatial distribution and magnitude of air pollution effects on vegetation (focused primarily on ozone) in the state is summarized in Peterson et al. (1991), Miller (1992), Peterson and Arbaugh (1992), Arbaugh et al. (1998, 1999), Miller and McBride (1999), Takemoto (2000), and Takemoto et al. (2000). The major findings and implications of these studies for national parks in California are described below.

Physiological Effects

Considerable variation in the response of trees to air pollutants is caused by differences in the pollutant dose, phenological stage, age of leaves exposed, seed source, nutritional status of plants, and the integrated effects of multiple stresses (Bytnerowicz and Grulke 1992). There are several mechanisms for phytotoxicity, depending on the type of air pollutant. Physiological effects at the cellular and subcellular level can significantly affect plant function in the absence of visible symptoms of injury.

Gaseous air pollutants typically enter the plant through stomata, so the duration of stomatal opening greatly affects the pollutant dose assimilated. Once a pollutant enters a plant cell, there are many biochemical processes that are potentially affected (Wellburn 1988, Heath and Taylor 1997). Ozone, SO_2, and other pollutants (e.g., peroxyacetyl nitrate, PAN) increase the potential for the formation of harmful free radicals. Many of these free radicals are highly reactive and disrupt various metabolic processes through oxidation, substitution for other compounds, and toxicity. Disruption of photosynthetic processes (e.g., damage to mesophyll cells and degradation of chlorophyll and chloroplasts in ozone toxicity; Grulke et al. 1996) is one of the most deleterious effects of air pollutants, because it results in lower photosynthetic rates, (Patterson and Rundel 1989, Grulke 1999), rapidly reduces plant vigor and productivity, and can affect internal resource allocation in the absence of visible symptoms (Matyssek and Innes 1999).

The physiological effects of NO_2 and other NO_x compounds are not well studied. Very elevated levels of NO_2 appear to induce plasmolysis through lipid breakdown in membranes (Wellburn 1990). During the photochemical smog formation process, NO is rapidly oxidized to NO_2, which subsequently reacts with hydroxyl radicals, producing HNO_3 (nitric acid) vapor. The effects of gaseous HNO_3 are poorly known, although this compound appears to be unique in that a significant quantity of deposited N enters the plant body both via stomatal uptake and a transcuticular pathway (Bytnerowicz et al. 1999b). Exposure to HNO_3 under experimental conditions has been shown to modify surficial foliar characteristics in some species (Bytnerowicz et al. 1998). It has also been shown to produce elevated concentrations of NO_3^- and changes in nitrate reductase activity (Norby et al. 1989), leading to the inference that long-term N nutrition could be affected (Bytnerowicz et al. 1999b).

The degree of pollutant injury depends on the effective dose, which is a function of concentration, length of exposure, and stomatal aperture (Kozlowski and Constantinidou 1986). Ozone injury of cells and tissues is essentially the same in woody and herbaceous plants (Bytnerowicz and Grulke 1992). Injury generally occurs first in the most photosynthetically active tissues, with disruption of chloroplasts in the palisade and mesophyll tissue. Overlying epidermal cells are usually uninjured. The loss of photosynthetic tissue results in visible chlorosis and necrosis. In conifers, these symptoms typically occur in older needles and progress to younger needles.

In plants injured by SO_2, tissue injury is more prominent between the veins of broadleaf species and is distinct from uninjured tissue along the veins. The injured tissue develops chlorosis and eventually necrosis. In conifers, the mesophyll cells of needles are the most

susceptible to SO_2, but other cells are also subject to damage (Kozlowski and Constantinidou 1986).

Injury development in plants subjected to elevated concentrations of N compounds is not well understood, although it has been shown that acidic precipitation, particularly nitric acid, can injure the cuticular layer and substomatal cavities (Taylor et al. 1988). The contact time of acidic droplets or films on the leaf surface determines the degree of damage (Wellburn 1988). Acid-induced injury under experimental conditions typically requires exposure to very acidic solutions (pH < 3; Temple 1988, Turner et al. 1989), and injury in the field is rarely observed. Elevated levels of nitric acid vapor have been shown to damage ponderosa pine needles (Bytnerowicz et al. 1998), although again injury in the field is rarely observed.

The relationship of pollutant exposure to seasonal variation in physiological activity can have significant physiological effects and result in visible injury in plants (Grulke 1999). Gaseous uptake generally is higher during periods when soil water potential is high. Therefore, the potential for pollutant uptake is higher during periods when soil moisture is high and plants are most metabolically active (after bud break in trees), typically from late winter to early summer in most parts of California. Recent data collected in the Sierra Nevada have documented that stomatal conductance in ponderosa pine generally is higher during the early growing season (spring, early summer) when soil moisture is high (unpublished data from Yosemite National Park, Sequoia-Kings Canyon National Parks, and other locations by J. Panek, http://www.cbe2.ced.berkeley.edu/panek/papersom.htm). Therefore, the physiological inference is that injury is more likely to be manifested by ozone exposure during the spring when gaseous uptake is higher. This finding has implications for how ozone exposure indices are quantitatively compared to ozone injury measurements.

Interactions with insects, pathogens, and other pollutants (Bytnerowicz and Grulke 1992) can accentuate the stress complex for plants exposed to a particular pollutant. This has been well documented for ozone and ponderosa pine. The most common stress complex includes ozone exposure, drought stress, and bark beetles (especially mountain pine beetle [*Dendroctonus ponderosae*] and western pine beetle [*D. brevicomis*]; Stolte 1996, Pronos et al. 1999). This interaction is particularly prominent in the mixed conifer forests of southern California and the southern Sierra Nevada. For example, during the late 1980s and early 1990s, ozone-stressed trees were subjected to several years of low precipitation. This reduced xylem pressure, and many trees were sufficiently weakened that they were susceptible to bark beetles, which proliferated through local and regional outbreaks, resulting in high levels of mortality in many areas. Drought stress seems to be a significant factor in reducing the vigor of bigcone Douglas-fir (*Pseudotsuga macrocarpa*) as well (Peterson et al. 1995). The prevalence of annosus root rot (*Heterobasidion annosum*) and black stain (*Leptographium wageneri*), fungal pathogens of ponderosa pine, appears to be accentuated by elevated levels of ambient ozone (Fenn et al. 1990, Pronos et al. 1999), although the extent of this interaction has not been quantified in the field.

Physiological effects on lichens are not as well understood as for vascular plants, although some consistent patterns have been observed (Eversman and Sigal 1987). In general, elevated ozone exposure under experimental conditions has been shown to reduce photosynthesis and produce structural changes in some species. Elevated SO_2 reduces respiration and increases membrane permeability. Elevated NO_x reduces chlorophyll content. The physical (visible) manifestation of pollutant injury is highly variable and often difficult to detect, especially under field conditions.

Spatial Distribution of Injury

The Forest-Ozone Response Study (FOREST), conducted during 1991-1994, quantified the distribution of ozone injury to forest species throughout the state of California (Rocchio et al. 1993, Cahill et al. 1996, Arbaugh et al. 1998). This study was linked to the Sierra Cooperative Ozone Impact Study (SCOIAS), another regional study that focused primarily on monitoring of ambient ozone and associated weather variables in the Sierra Nevada (Van Ooy and Carroll 1995). The FOREST study added to the spatial resolution of long-term research and monitoring of ozone effects by Paul Miller and colleagues in southern California (summarized in Miller 1992, Miller and McBride 1999) and to extensive surveys of injury and growth patterns in the Sierra Nevada by Peterson and colleagues (Peterson et al. 1987, 1991, Peterson and Arbaugh 1988, 1992). Other assessments of ozone injury have been done for individual national forests and national parks (e.g., Pronos and Vogler 1981, Vogler 1982, Warner et al. 1983), but the FOREST results are the most spatially comprehensive and the most recent.

The FOREST data are comprised of ozone-injury surveys at 11 sites (all with associated ozone-monitoring data), ranging from LAVO in northern California to San Bernardino National Forest in southern California, with sites distributed throughout the westside Sierra Nevada within the elevational range of mixed conifer forest. The study included two sites in YOSE, two sites in SEKI and one site in LAVO. The final analysis (Arbaugh et al. 1998) also included four additional sites from the San Bernardino Air Pollution Gradient Study (SBGS, Miller 1992), thereby extending the range of exposure severity.

Visible ozone injury to ponderosa pine and Jeffrey pine (*Pinus jeffreyi*) was detected at all sites, ranging from 26.8% of trees at the northernmost site to 100% at the southernmost (Table I-5). Mean whorl retention ranged from 6.5 to 2.7 for these same sites (although Jeffrey pine, which is more dominant in the north, has more needles than ponderosa pine). Ozone Injury Index (OII; Schilling and Duriscoe 1996) ranged from 6.3 in the north to 65.1 in the south. It was determined that OII was strongly correlated with the Forest Pest Management (FPM) injury index used by the Forest Service (Pronos et al. 1978), which indicates that data collected by personnel from different organizations can likely be combined for robust analyses. Most importantly, there were significant correlations between OII and various ozone indices over the geographic range of the study (Figure I-2), which suggests that (1) ozone-monitoring data can be used to predict ozone injury in sensitive pine species and (2) conversely, injury data for pines (much less expensive to collect than ozone data) are a reasonable indicator of the level of ozone exposure at a particular geographic location. This means that ponderosa pine and Jeffrey pine, which are present in several Class I national parks in California, can be used as true bioindicator species for ozone sensitivity and exposure. The FOREST study is one of the first field studies to demonstrate a quantitative relationship between tree injury and cumulative ozone exposure (SUMO). It is unclear how this relationship would be derived for species less sensitive than ponderosa pine and Jeffrey pine.

The FOREST study corroborates a previous analysis of ozone injury (presence of chlorotic mottling and needle retention) to ponderosa pine in the Sierra Nevada (Peterson et al. 1991, Peterson and Arbaugh 1992), which found a gradient of injury, ranging from low injury at northern sites (Tahoe National Forest) to high injury at southern sites (Sequoia National Park and Sequoia National Forest); this study also included sites at YOSE. Again, strong correlations were found between injury level and ozone exposure. With essentially the same results from two studies that used different methods at different points in time, one can have high confidence in the inference that ozone injury increases from north to south in the Sierra Nevada and that the injury is directly proportional to ambient ozone exposure.

Table I-5. Crown injury characteristics of trees for individual sites in the FOREST study (from Arbaugh et al. 1998). Means are for 1991-1994 for all sites except Manzanita Lake (1992-1994) and Barton Flats (1992-1995).[1]

	National Forest or National Park	Mean Ozone Injury Index (± 1 SD)	Mean Whorl Retention (Yr)	Mean Trees with Visible Injury (%)
FOREST Sites (north to south)				
Manzanita Lake	Lassen NP	6.3 (10.5)	6.5	26.8
White Cloud	Tahoe NF	27.3 (21.0)	4.9	68.4
Sly Park	El Dorado NF	27.3 (23.1)	4.4	62.0
Five Mile	Stanislaus NF	27.3 (21.0)	5.0	66.4
Camp Mather	Yosemite NP	14.7 (18.9)	5.4	41.4
Wawona	Yosemite NP	14.7 (21.0)	5.0	38.6
Shaver Lake	Sierra NF	16.8 (18.9)	5.3	46.2
Grant Grove	Sequoia NP	37.8 (18.9)	5.0	93.8
Giant Forest	Sequoia NP	41.3 (18.2)	4.8	92.3
Mountain Home	Sequoia NF	29.4 (23.1)	4.9	67.5
Barton Flats	San Bernardino NF	46.2 (14.7)	4.5	99.2
SBGS Sites (east to west)				
Heart Bar	San Bernardino NF	29.4 (16.8)	5.5	95.9
Camp Osceola	San Bernardino NF	42.0 (18.9)	4.8	95.3
Camp Angeles	San Bernardino NF	54.6 (10.5)	3.9	100
Camp Pavika	San Bernardino NF	65.1 (6.3)	2.7	100

[1] Reprinted from Arbaugh, M.J., P.R. Miller, J.J. Carroll, B. Takemoto, and T. Procter, 1998, Relationships of ozone exposure to pine injury in the Sierra Nevada and San Bernardino Mountains of California, USA, Environmental Pollution 101:291-301, with permission from Elsevier Science.

It should be noted that the highest injury to ponderosa pine and Jeffrey pine in California is consistently found in the San Bernardino Mountains of southern California (Miller 1992, Miller et al. 1997, Miller and Rechel 1999), where injury has also been observed in several understory perennials (blue elder [*Sambucus mexicana*], Douglas sagewort [*Artemisia douglasiana*], wormwood [*A. dracunculus*], evening primrose [*Oenothera elata* var. *hookeri*]) and annuals (hairy peppergrass [*Lepidium virginicum* var. *pubescens*], spreading groundsmoke [*Gayophytum diffusum*], stiffbranch bird's beak [*Cordylanthus rigidus*]) (Temple 1999). Ozone injury in this area is now lower than it was prior to the 1980s, which is likely the result of improved air quality and perhaps selection for ozone-tolerant genotypes (Miller et al. 1989, Arbaugh et al. 1999, Miller and Rechel 1999), but this injury is considerably higher than other locations in California. SEKI, and to a lesser extent YOSE, clearly have the worst ozone injury of the national parks that have been monitored in California.

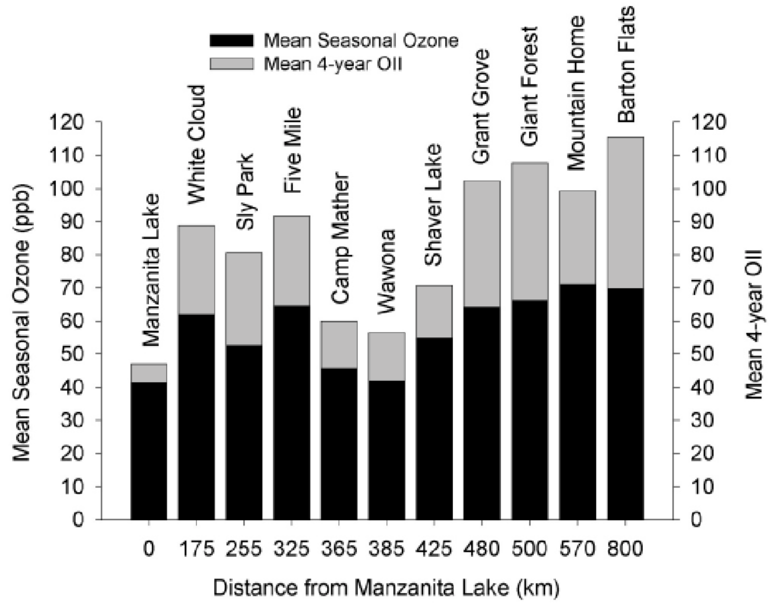


Figure I-2. Average Ozone Injury Index (OII) scores and average cumulative ambient ozone (SUM0) over the four-year study period for FOREST sites in the Sierra Nevada and San Bernardino Mountains (1992 to 1995), ranging from Manzanita Lake in Lassen Volcanic National Park in the north to Barton Flats in the San Bernardino Mountains in the south. The OII is a unitless index that ranges for this dataset from 6.3 at Manzanita Lake to 46.2 at Barton Flats. (Reprinted from Arbaugh et al. 1998)

Shrubby species of chaparral and coastal sage scrub vegetative assemblages are generally not regarded as very sensitive to air pollutants. Net assimilation and growth are highest during spring when soil moisture is high and temperature is optimal. Most species have physiological adaptations to summer drought, and gas exchange during summer is minimal when air pollution (especially ozone) is highest. Therefore, many of these species likely do not take up high concentrations of pollutants in the field (Temple 1999), except perhaps as seedlings (Stolte 1982). Westman (1979, 1985) reported that several species of coastal sage scrub (in the Santa Monica Mountains) showed ozone injury in the field, and that species richness in this vegetation type was inversely correlated with ozone exposure. However, controlled-exposure studies of coastal sage scrub (Preston 1986) and chaparral (Stolte 1982) found that all species were resistant to ozone exposure, and reports of high sensitivity in chaparral species are not generally regarded by air pollution pathologists as scientifically credible (P. Miller, P. Temple, pers. comm.).

Based on an assessment of potential effects of air pollutants on vegetation in JOTR, injury symptoms in Mojave Desert and Colorado Desert plants have not been documented in the field (Temple 1989). However, experimental studies using controlled exposures have identified desert species that are potentially sensitive to ozone, SO_2, and NO_2 (Thompson et al. 1980, 1984a,b; Temple 1989). In a controlled-exposure study, Thompson et al. (1980) tested the sensitivity of five perennial species and five annual species native to the Mojave Desert and

Colorado Desert to SO_2 and NO_2, and found a wide range of responses in terms of visible injury symptoms and growth. In a major air-pollutant screening effort, Thompson et al. (1984a,b) used controlled exposures to determine the sensitivity of 49 Mojave Desert species to ozone and SO_2. They found considerable variation in sensitivity, with evening primrose (*Oenothera* spp.) and catseye (*Cryptantha* spp.) having some of the highest levels of injury. Results for individual species studied by Thompson et al. (1980;1984a,b) are not summarized here, but were used to compile the table of species sensitivities for JOTR (Table III-6).

Temple (1989) tested the sensitivity of the woody perennials catclaw acacia (*Acacia greggii*), desert willow (*Chilopsis linearis*), skunkbush sumac (*Rhus trilobata*), and Goodding's willow (*Salix gooddingii*) to controlled exposures of ozone, ranging up to 200 ppbv over 4-hour periods. This study found that skunkbush sumac exhibited foliar injury at concentrations as low as 100 ppbv, while Goodding's willow exhibited foliar injury at 200 ppbv; other species had no injury symptoms. Surprisingly, skunkbush sumac exposed to elevated levels of ozone also grew faster, but the fact that it had such clear symptoms at relatively low exposures indicates high sensitivity and good potential as a bioindicator.

Effects on Growth and Productivity

Numerous studies have demonstrated that prolonged exposure to ozone under experimental conditions can reduce growth of ponderosa pine, Jeffrey pine, and other species (e.g., Temple 1988; Temple et al. 1992, 1993; Temple and Miller 1994; Takemoto et al. 1997). It has been more difficult to infer clear relationships between ozone exposure and tree growth in the field in California, primarily due to high variability in growth patterns and site characteristics.

Long-term studies of tree growth in the San Bernardino Mountains indicate that the growth of mature ponderosa pine and Jeffrey pine was lowest between 1950 and 1975, during which ozone concentrations in the Los Angeles Basin were higher than in earlier and later time periods (Arbaugh et al. 1999). Growth reductions generally were found in trees with the most severe levels of ozone injury (Miller et al. 1997). In addition, growth of mature bigcone Douglas-fir in the same area was significantly lower after 1950, and growth was generally lower at sites with higher ozone exposure (Peterson et al. 1995, Arbaugh et al. 1999). However, both studies found that there is a strong interaction with drought, and that trees, especially the older ones, tend to have larger growth reductions and slower recovery of growth following periods of low precipitation. So although ozone has a major impact on tree growth in the San Bernardino Mountains, drought—alone or in combination with ozone stress—can also reduce growth.

Ozone has also been shown to reduce tree growth in the southern Sierra Nevada. In SEKI, Jeffrey pine with ozone injury were shown to have 11% lower growth than trees without injury (Peterson et al. 1987). An extensive study of ponderosa pine at 56 sites in the Sierra Nevada quantified growth of trees with and without injury symptoms, ranging from the Tahoe National Forest to Sequoia National Forest (Peterson et al. 1991, Peterson and Arbaugh 1992). Significant growth reductions since 1950 were found in Sequoia National Park and Sequoia National Forest, where ozone injury and ozone exposure were greatest. Growth reductions, often quite severe, were also documented in YOSE, but were likely caused by the prevalence of annosus root rot (Peterson et al. 1991).

One of the most striking examples of the ecological effects of ozone is the documentation of high levels of ponderosa pine and Jeffrey pine mortality since the 1950s in areas of severe ozone injury in the San Bernardino Mountains (Miller 1992, McBride and Laven 1999). This has resulted in accelerated plant succession within the mixed conifer forest in this region, with shade tolerant—and mostly ozone tolerant—understory species such as white fir (*Abies concolor*) and incense cedar (*Calocedrus decurrens*) becoming more dominant and pines

becoming less dominant. Lower cone production in injured ponderosa pine (Luck 1977) and higher levels of litter accumulation under trees with severe ozone injury (Miller and Rechel 1999) may also inhibit regeneration of pines, which require mineral soil for germination. A recent increase in pine seedlings, which may be associated with reduced overstory mortality, reduced ozone injury, and possibly more ozone-tolerant genotypes suggests that ponderosa pine may be able to increase its dominance in the future if ozone exposure does not increase (McBride and Laven 1999).

The kind of ozone-related ecosystem change documented in the San Bernardino Mountains is rare elsewhere in California. However, severe ozone injury, reduced tree growth, and high rate of mortality in ponderosa pine have been observed in the Deer Ridge area of Sequoia National Park (Peterson et al. 1991, D.L. Peterson, personal observation). This site overlooks the Kaweah River Valley and receives some of the higher ozone exposures in the park (c.f., Table IX-7). The rapid loss of the overstory at this site may represent ozone-induced ecosystem change at the local level.

Additional subtle effects may also be affecting the mixed conifer forests of southern California and southern Sierra Nevada. For example, prolonged exposure to elevated ambient ozone may reduce root biomass in mature ponderosa pine (Grulke et al. 1998, Grulke 1999), which may predispose this species to increased stress during periods of low soil moisture. Given that summer drought is a normal part of the Mediterranean climate of California, this could be a significant long-term stress.

Elevated levels of N deposition have the potential to alter rates and magnitudes of ecosystem processes. High N deposition, in addition to ozone, may be a factor in reduced root production (Grulke et al. 1998, Grulke and Balduman 1999). In the highly polluted forests of the Los Angeles Basin, the following changes have also been observed: increased foliar N, increased N: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). These changes in the biogeochemical characteristics of the mixed conifer forest may eventually affect other ecosystem processes such as growth, interspecific competition, and decomposition (Fenn and Dunn 1989). There is no evidence to suggest that N deposition is sufficiently high in any of the Class I parks in California so as to cause N-saturation of forests and consequent adverse effects on forest health. Total N deposition is less than about 3 kg/ha/yr at all of the parks, which is well below the levels at which N-saturation of forest ecosystems have been demonstrated. In contrast, mixed conifer forests in some portions of the San Bernardino Mountains near Los Angeles have been judged to be N-saturated, based on high NO_3^- concentrations in soil solution and foliage, enhanced litter decomposition and N-mineralization rates, low C:N ratios, and other indicators of excess N availability (Fenn et al. 1996). Total N deposition in this area ranges from about 35 to 45 kg/ha/yr. At two relatively low-pollution sites at the eastern end of the San Bernardino Mountains (furthest from Los Angeles) and at three mixed conifer forest sites in the San Gabriel Mountains, vector analysis of the foliar response to experimental ammonium nitrate (NH_4NO_3) fertilization indicated that N was still growth-limiting despite N deposition levels of 6 to 11 kg/ha/yr (Kiefer and Fenn 1997). Current levels of N deposition to the Class I parks in California are well below the levels at which significant ecosystem changes have been documented. Although extremely high rates of N deposition and quantifiable effects have not been observed outside of southern California, long-term accumulation of N at other locations in California where N deposition is above background levels could conceivably have some effects in the future.

c. Species Sensitivity

In this report, each section for the individual Class I areas of California includes a list of plant species and their sensitivities to ozone, SO_2, and NO_x. These lists are based primarily on visible injury (rarely growth) observed in controlled exposures to air pollutants, field research and monitoring, and to some extent expert opinion. Information is drawn from the primary literature and various reports in which sensitivity to air pollutants has been assessed, including previous reports developed for the NPS Air Resources Division.

Sensitivity is ranked in the general classes low, medium and high. These classes are roughly keyed to sensitivity classes developed for estimating air pollution effects in vegetation of California wilderness areas (D. Peterson et al. 1992a), and can be used to relate potential foliar injury to pollutant exposures. These sensitivity classes can be used to identify severity of injury as well as to indicate potential injury thresholds. Species with high sensitivity generally will be the first to exhibit injury symptoms and therefore deserve greater emphasis in monitoring.

Ozone-sensitivity classes for conifers (Table I-6) are based primarily on symptomatology of ponderosa pine (var. *ponderosa*) and Jeffrey pine in California in the field and under experimental conditions (Miller and Millecan 1971; Pronos and Vogler 1981; Peterson and Arbaugh 1988; Peterson et al. 1991; Peterson and Arbaugh 1992; Temple et al. 1992, 1993). Because these data are from quantitative studies at a variety of locations, Table I-6 should be highly relevant for assessing ozone injury in ponderosa pine and Jeffrey pine in national parks throughout California. Applicability to other coniferous species may vary considerably with respect to specific symptoms. However, because quantifiable symptoms for other species are rare, it is recommended that these sensitivity classes be applied to all coniferous species. Similarly, because quantifiable symptoms for SO_2 (Thompson et al. 1984a,b) and NO_x (Flagler 1998) are rare, these sensitivity classes can be cautiously applied to both SO_2 and NO_x for the time being.

Table I-6. Sensitivity classes for pollutant injury in coniferous trees with respect to foliar characteristics (after D. Peterson et al. 1992a).

Sensitivity Class	Needle Age Class with Chlorotic Mottle (years)	Needle Retention as Percent of Uninjured Tree (%)
Low	≥ 5	≥ 71
Medium	3 - 4	41 -70
High	1 - 2	≤ 40

It has been suggested that the high, medium, and low sensitivity classes are associated with the following respective ozone concentrations (7-hour growing season mean, based on the highest continuous 7-hour period per day): 61-70, 71-90 and > 90 ppbv (D. Peterson et al. 1992a). The association of these pollutant concentrations with specific symptoms and sensitivity classes should be considered a general guideline only, and may not be relevant for all species.

Pollutant-sensitivity classes for hardwoods (Table I-7) are based primarily on data for quaking aspen (*Populus tremuloides*) and a few other hardwood species. Numerous studies have documented the sensitivity of this species to ozone and SO_2 under field and experimental

Table I-7. Sensitivity classes for pollutant injury in hardwood trees and herbaceous plants with respect to foliar characteristics (after D. Peterson et al. 1992a).

Sensitivity Class	Percent Leaf Area with Stippling
Low	≤ 40
Medium	41-60
High	≥ 61

conditions (Wang et al. 1986, Karnosky et al. 1992, Coleman et al. 1996), although there is considerable variability in sensitivity among different genotypes (Berrang et al. 1986).

Diagnostic ozone symptomatology for quaking aspen includes chlorosis, stippling, necrotic spotting, and leaf margin burn. Symptoms can vary seasonally, with stippling being most prominent in the spring and black, bifacial (both leaf surfaces) spotting appearing in late summer (J.P. Bennett, pers. comm.). Great care must be taken in distinguishing pollutant symptoms from the effects of various pathogens and insect herbivores commonly found on this species.

Table I-7 should be relevant for assessing ozone injury in quaking aspen. Applicability to other hardwood species may vary considerably with respect to specific symptoms. However, because quantifiable symptoms for other species are rare, it is recommended that the sensitivity classes for quaking aspen be applied to all hardwood species. It is also recommended that these sensitivity classes be applied to herbaceous species, because their leaf morphology is similar to that of hardwood trees. Similarly, because quantifiable symptoms for SO_2 (Thompson et al. 1984a) and NO_x are rare, these sensitivity classes can be cautiously applied to both NO_x and SO_2 for the time being.

Species sensitivities in this report are derived from Davis and Wilhour (1976), Miller et al. (1983), Thompson et al. (1984a), Esserlieu and Olson (1986), Hogsett et al. (1989), Temple 1989, Bunin (1990), D. Peterson et al. (1992a), J. Peterson et al. (1992a), National Park Service (1994), Electric Power Research Institute (1995), Flagler 1998, Peterson and Sullivan (1998), Temple 1999, and Brace et al. (1999). Estimated sensitivity (low, medium, high) from these sources was cross-referenced with summaries of vascular flora in the NPFlora database and lichens in the NPLichen database for each Class I park in California. Summary tables of sensitivities to air pollutants for each species are listed in sections for individual parks. These tables should be used only as general guides to species sensitivities because of the uncertainty and variability of data on which they are based. Visible injury normally would be expected to appear only in species with high (and possibly medium) sensitivity.

d. Potential Effects of Nitrogen Deposition on Terrestrial Ecosystems

During the past decade, much has been written about the potential effects of elevated levels of N deposition on various terrestrial ecosystems. A recent article (Fenn et al. 1998) suggests that several forest ecosystems of North America have "excess" N, often termed "N saturation." This terminology should be used cautiously, because N in soluble form commonly moves through and out of terrestrial systems when conditions are not optimal for uptake. Examples include when (1) saturated hydrologic flow occurs in soils, (2) N is removed during periods of low vegetation uptake and growth, and (3) N accumulates over a period of time and moves out of the terrestrial system via hydrologic flow (e.g., snowmelt).

Of more substantive concern are the effects of long-term accumulation of N on various ecosystem processes. Increased levels of N have the potential to increase foliar N content, alter N metabolism in plants (including winter hardiness), alter soil chemistry and N cycling, affect mycorrhizae, modify soil microbial dynamics, and alter plant growth (Takemoto 2000, Takemoto et al. 2000).

It is likely that at least some forest ecosystems in southern California are experiencing some effects of long-term accumulation of N in soils. Mixed conifer forests in the San Bernardino Mountains currently have higher foliar N content in conifers in areas with highest N deposition (Fenn et al. 1996, 1998, Fenn and Poth 1999). Furthermore, litter decomposition rates are higher in areas with highest N deposition, presumably because of lower C:N ratios in the forest-floor litter (Fenn and Dunn 1989, Fenn et al. 1998). This is a significant change in an ecosystem process that has implications for biogeochemical cycling in general. It is also believed that increased N deposition is a contributing factor (along with ozone exposure and acidic deposition) to reduced root biomass in mixed conifer forest of the San Bernardino Mountains (Grulke et al. 1998, Grulke 1999, Grulke and Balduman 1999). This is a significant change in C allocation with the potential to predispose trees to drought stress and to alter long-term growth and productivity.

At the present time, wet (and presumably total) N deposition rates in national parks of California are much lower than total N deposition measured in mountain ecosystems adjacent to the Los Angeles Basin. Therefore, the kinds of effects being detected in the forests adjacent to Los Angeles are unlikely to be manifested in other areas of California in the near future. However, it should be noted that even slightly elevated N deposition could potentially affect ecosystem processes through long-term N accumulation, especially at locations with relatively low soil fertility (c.f., Baron 1992).

5. Aquatic Resources and Sensitive Indicators

The potential effects of S deposition on surface water quality have been well studied throughout the United States, particularly within the Environmental Protection Agency's (EPA's) Aquatic Effects Research Program (AERP), a component of the National Acid Precipitation Assessment Program (NAPAP). Major findings were summarized in a series of State of Science and Technology Reports (e.g., Baker et al. 1990, Sullivan 1990) and the final NAPAP policy report, the 1990 Integrated Assessment (NAPAP 1991). Although aquatic effects from N deposition have not been studied as thoroughly as those from S deposition, concern has been expressed regarding the role of NO_3^- in acidification of surface waters (particularly during hydrologic episodes), the role of NO_3^- in the long-term acidification process, the contribution of ammonium (NH_4^+) from agricultural sources to surface water acidification, and the potential for anthropogenic N deposition to stimulate eutrophication of freshwaters and estuaries (e.g., Sullivan 1993, Wigington et al. 1993, Sullivan et al. 1997, NAPAP 1998)

Atmospheric deposition of S and N (as NO_3^- and as NH_4^+, which can be quickly nitrified to NO_3^-) often cause increased concentrations of SO_4^{2-} in drainage waters and can, in some cases, cause increased concentrations of NO_3^-. An increase in the concentration of either of these mineral acid anions will generally result in a number of additional changes in water chemistry. These can include:

- increased concentration of base cations (Ca^{2+}, Mg^{2+}, K^+, Na^+)
- decreased concentration of acid neutralizing capacity (ANC)
- increased concentration of hydrogen ion (H^+) (decreased pH)
- increased concentration of dissolved Al

Increased concentrations of H^+ and/or Al occur only in response to higher concentrations of SO_4^{2-} or NO_3^- when ANC has decreased to near or below zero. At higher ANC values, contributions of SO_4^{2-} or NO_3^- are mainly balanced by increasing base cation concentrations and some decrease in ANC. High concentrations of H^+ or Al can be toxic to fish and other aquatic biota.

If NO_3^- leaches into stream or lakewater as a result of increased N deposition, the result can be eutrophication or acidification. If N is limiting for aquatic primary production, the added NO_3^- will generally result in increased algal productivity, which can cause disruption of aquatic community dynamics. If N is not limiting (P or some other nutrient can be limiting, for example), then the added NO_3^- will remain in solution, possibly leading to acidification.

Surface waters that are sensitive to acidification from acidic deposition of S or N typically exhibit a number of characteristics. Such characteristics either predispose the waters to acidification and/or correlate with other parameters that predispose the waters to acidification. Although precise guidelines are not widely accepted, general ranges of parameter values that reflect sensitivity are as follows (c.f., Sullivan 2000).

Dilute - Waters have low concentrations of all major ions, and therefore specific conductance is low (< 25 µS/cm). In areas of the West which have not experienced substantial acidic deposition, highly sensitive lakes and streams are often ultradilute, with specific conductance less than 10 µS/cm.

Acid neutralizing capacity - ANC is low. Acidification sensitivity has long been defined as ANC<200 µeq/L, although more recent research has shown this criterion to be too inclusive. Waters sensitive to chronic acidification generally have ANC<50 µeq/L, and waters sensitive to episodic acidification generally have ANC<100 µeq/L. In the Sierra Nevada, where acidic deposition is generally low and not expected to increase dramatically, ANC values of 25 µeq/L and 50 µeq/L probably protect waters from any foreseeable chronic and episodic acidification, respectively.

Base cations - Concentrations are low in non-acidified waters, but increase substantially in response to acidic deposition. In relatively pristine areas, the concentration of (Ca^{2+} + Mg^{2+} + Na^+ + K^+) in sensitive waters will generally be less than about 50 to 100 µeq/L.

Organic acids - Concentrations are low in waters sensitive to the effects of acidic deposition. Dissolved organic carbon (DOC) and associated organic acids cause water to be naturally low in pH and ANC, or even to be acidic (ANC<0), but also impart substantial pH buffering at these low pH values. Waters sensitive to acidification from acidic deposition in the West generally have DOC less than about 3 to 5 mg/L.

pH - pH is low, generally less than 6.0 to 6.5 in acid-sensitive waters. In areas that have received substantial acidic deposition, acidified lakes are generally those that had pre-industrial pH between 5 and 6.

Acid anions - Sensitive waters generally do not have large contributions of mineral acid anions (e.g., SO_4^{2-}, F^-, Cl^-) from geological or geothermal sources. In particular, the concentration of SO_4^{2-} in drainage waters would not be substantially higher than could reasonably be attributed to atmospheric inputs, after accounting for probable dry deposition and evapotranspiration.

Physical characteristics - Sensitive waters are usually found at moderate to high elevation, in areas of high relief, with flashy hydrology and minimal contact between drainage waters and soils or geologic material that may contribute weathering products to solution. Sensitive streams are generally low order. Sensitive lakes are generally either small drainage systems or small seepage systems that derive much of their hydrologic input as direct precipitation to the lake surface.

The Sierra Nevada and Cascade Mountains constitute the mountain ranges with the greatest number of sensitive aquatic resources in the West. Sensitive lakes are dilute bicarbonate systems and, unlike many low-ANC lakes in the East, have very low concentrations of DOC. Acid anion concentrations in most western lakes are extremely low in fall samples, but limited analyses of lake chemistry in spring generally show higher concentrations of NO_3^- and SO_4^{2-} (Williams and Melack 1991). The extremely dilute nature of many western lakes raises concerns regarding potential increases in acid anions, derived from acidic deposition, during spring snowmelt. The available data from intensive study sites in the West (e.g., Loch Vale, CO, Emerald Lake Basin, CA, and the Glacier Lakes Watershed, WY) suggest that episodic depression of stream pH is more pronounced than for lake systems, yet no systematic regional stream chemistry data are available in the West with which to assess the regional sensitivity of streams to acidic deposition or the importance of episodic processes to stream chemistry (Sullivan 2000).

The Sierra Nevada and portions of the Cascade Mountains (including LAVO, SEKI, and YOSE) are particularly sensitive to potential acidic deposition aquatic effects because of the predominance of granitic bedrock, thin acidic soils, large amounts of precipitation, coniferous vegetation, and extremely dilute waters (McColl 1981, Melack et al. 1985, Melack and Stoddard 1991, Sullivan 2000). It appears that chronic acidification has not occurred to any significant degree. It is possible, however, that episodic effects may have occurred under current deposition regimes.

Concentrations of SO_4^{2-} in western lakes are generally low, but in some cases, for example in part of Kings Canyon National Park, geologic sources contribute substantial amounts of SO_4^{2-} to lakewaters. Nitrate concentrations were virtually undetectable in most western lakes sampled by EPA's Western Lakes Survey in the fall (Landers et al. 1987). However, in some cases, fall NO_3^- concentrations were surprisingly high. For example, in the Sierra Nevada about 10% of the lakes had NO_3^- concentrations above 5 µeq/L (Sullivan and Eilers 1994). Thus, some high-elevation lakes in the Sierra Nevada may be experiencing N deposition sufficiently high to cause chronic NO_3^- leaching, and likely associated chronic and episodic acidification, albeit small in magnitude.

In contrast to S, which is generally conservative in the West (Stauffer 1990), N is assimilated rapidly in most watersheds. Whereas a conceptual model of S distribution in lakes suggests that most lake populations will exhibit a normal distribution around some positive value (Turk and Spahr 1991), a companion conceptual model for N presented by Sullivan and Eilers (1994) suggests that NO_3^- distributions for undisturbed watersheds should be highly skewed towards zero. As N loading exhausts the capability of the watershed to assimilate NO_3^- and NH_4^+, "leakage" will be exhibited as an extended regional distribution. Where watershed disturbance is severe (e.g., logging, cattle grazing, cropland, urbanization), NO_3^- concentrations in drainage waters can be substantial.

Most lakes receive the majority of their hydrologic input from water that has previously passed through the terrestrial catchment. As long as N retention in the terrestrial system remains high, as is generally the case for forested ecosystems, N concentrations in lakes will remain low in the absence of contributions from land use (e.g., agriculture) or other pollution sources. However, if N retention in the catchment is low and the lake has not yet acidified, N deposition can in some cases increase primary production. Lakes that are most likely to be low in base cations (therefore potentially sensitive to acid deposition) and also N-limited are often systems overlaying volcanic bedrock (these rocks are often high in P).

Spring snowmelt can flush into lakes and streams N that was deposited in the snowpack from atmospheric deposition or N mineralized within the soil during winter. In general, NO_3^-

concentrations in the snowpack of the California mountains are slightly higher in the Sierra Nevada than in the Cascade Mountains in the northern portion of the state (Laird et al. 1986). In some alpine and subalpine western lakes, the concentration of NO_3^- remains somewhat elevated throughout the growing season. This may be related to the extent of snow cover and effects of the cold temperatures on biological uptake processes, hydrological flowpaths across exposed bedrock and talus, and/or saturation of the uptake capacity of terrestrial and aquatic biota (Sullivan 2000).

A substantial component of the NO_3^- in western lakewaters may have been derived from mineralization of organic N and not directly from atmospheric deposition (Williams et al. 1996). It is likely that microbial activity under the snowpack plays an important role in both the production of inorganic N before the snowmelt begins and also in the immobilization of N during the initial phases of snowmelt before vegetation becomes active (Brooks et al. 1996). The recognized importance of mineralization, the production of inorganic N from the breakdown of organic material, and subsequent conversion to NO_3^- (nitrification) as a source of streamwater NO_3^- does not imply, however, that atmospheric N deposition is not driving this flux. It is likely that mineralization and nitrification processes release N to surface waters that was derived largely from deposition and was cycled through the primary production of the previous growing season.

In the absence of adequate site-specific research data, computer models can be used to predict pollution effects on aquatic ecosystems and to perform simulations of future ecosystem response (Cosby et al. 1985, Agren and Bosatta 1988). The Model of Acidification of Groundwater in Catchments (MAGIC), a lumped-parameter mechanistic model, has been used throughout North America and Europe and extensively tested against the results of diatom reconstructions and ecosystem manipulation experiments (e.g., Wright et al. 1986; Sullivan et al. 1992; Sullivan and Cosby 1995; Cosby et al. 1995,1996; Sullivan 2000). Watershed models that include N dynamics should prove valuable to management agencies which require quantitative predictions of pollution impacts and control programs. Nitrogen dynamics have recently been added to the MAGIC model (Jenkins et al. 1997).

Topographic relief is also a contributing factor to acidic deposition sensitivity in the West because the mountainous terrain contributes to major snowmelt events that may cause episodic pH and ANC depressions. These snowmelt events can result in multiple exchanges of the water volume in sensitive lakes. The short residence time of many high-elevation lakes not only contributes to elevated sensitivity to episodic acidification during snowmelt events, but also reduces the relative importance of in-lake alkalinity generation processes.

Episodic acidification is an important issue for surface waters in the Sierra Nevada. A number of factors pre-dispose such systems to potential episodic effects (Peterson and Sullivan 1998), including:

1. the abundance of dilute to ultradilute lakes (i.e., those having extremely low concentrations of dissolved solutes), exhibiting very low concentrations of base cations, and therefore ANC, throughout the year;
2. large snowpack accumulations at the high elevation sites, thus causing substantial episodic acidification via the natural process of base cation dilution; and
3. short retention times for many of the high-elevation drainage lakes, thus enabling snowmelt to rapidly flush lake basins with highly dilute meltwater.

In most cases, episodic pH and ANC depressions during snowmelt are driven by natural processes (mainly base cation dilution) and nitrate enrichment (cf. Wigington et al. 1990, 1993; Stoddard 1995). Where pulses of increased SO_4^{2-} are found during hydrological episodes, they are usually attributable to S storage and release in streamside wetlands or S retention in

watershed soils. This is probably attributable to the observation, based on ratios of naturally-occurring isotopes, that most stream flow during episodes is derived from pre-event water. Water stored in watershed soils is forced into streams and lakes by infiltration of meltwater via the "piston effect." This is not necessarily the case for high-elevation watersheds in the Sierra Nevada and Cascade Mountains, however. Such watersheds often have large snowpack accumulations and relatively little soil cover. Selective elution of ions in snowpack can therefore result in relatively large pulses of both NO_3^- and SO_4^{2-} in drainage water early in the snowmelt (Sullivan 2000).

The N loading to alpine and subalpine systems may be functionally much higher than is reflected by the total annual deposition measured or estimated for the watersheds. There are several reasons for this. First, the actual N loading to both soils and drainage waters at high-elevation sites during summer is comprised of both the ambient summertime atmospheric loading and also the loading of the previous winter which was stored in the snowpack and released to the terrestrial and aquatic systems during the melt period, often largely occurring during May through July. For this reason, the N loading from atmospheric deposition during summer can actually be substantially higher than the annual average atmospheric loading. Second, soil waters are often completely flushed during the early phases of snowmelt in alpine areas. Such flushing can transport to surface waters a significant fraction of the N produced in soils during winter by subnivian mineralization of the primary production of the previous summer. This N load from internal ecosystem cycling will generally be larger in areas that receive significant N deposition because the gross primary production of alpine ecosystems often tends to be N-limited (Bowman et al. 1993). Thus, the functional N loading to terrestrial and aquatic runoff receptors in alpine and subalpine areas during the summer growing season is much higher than the annual average N loading for the site. This is especially true during the early phases of snowmelt, when soil waters are flushed from shallow soils and talus areas and when a large percentage of the ionic load of the snowpack is released in meltwater (Sullivan 2000).

6. Visibility

The NPS monitors visibility conditions and supports studies to determine the causes of visibility impairment (haze) at many parks and wilderness areas nationwide. The purpose of this monitoring is to characterize current visibility conditions, identify the specific chemical species and their emission sources that contribute to visibility impairment, and to document long-term trends to assess the effects of changes in emissions. The NPS cooperates and shares resources with other federal land management agencies, states, and the U.S. EPA in the Interagency Monitoring of Protected Visual Environments Program (IMPROVE). IMPROVE monitoring is conducted at eight California NPS Class I areas in the Pacific Coast, Sierra-Humboldt, and Sierra Nevada regions. View-only (non-IMPROVE) monitoring has also been conducted at LABE in the Sierra-Humboldt region. Table I-8 summarizes visibility monitoring efforts conducted in these California regions since 1982. The following section describes the types of visibility monitoring listed in the table.

IMPROVE Program descriptions and spatial distribution summaries presented in the following sections were compiled by the NPS Air Resources Division (Denver) for the period March 1988 through February 1999. Some of the provided data and graphical summaries were extracted from the "Spatial and Seasonal Patterns and Long Term Variability of the Composition of Haze in the United States: An Analysis of Data from the IMPROVE Network" for 1992 - 1995 (Sisler et al., 1996), and the IMPROVE report for the period March 1996 through February 1999 (Malm et al. 2000).

Table I-8. Visibility Monitoring in Class I National Parks of California 1982 to Present.

Region	Site	IMPROVE Visibility Monitoring		
		Particle (Aerosol)	**Optical**	**View**
Pacific Coast	Pinnacles National Monument	3/1988 - Present	3/1988 - 7/1993	8/1986 - 1/1995
	Point Reyes National Seashore	3/1988 - Present	None	6/1987 - 4/1995
	Redwood National Park	3/1988 - Present	None	6/1987 - 3/1995
Sierra-Humboldt	Lassen Volcanic National Park	3/1988 - Present	None	6/1987 - 4/1995
	Lava Beds National Monument	None	None	8/1986 - 9/1991
Sierra Nevada	Sequoia/Kings Canyon National Parks	3/1994 - Present (Excluded from Regional Summaries)	None	None
	Yosemite National Park	3/1988 - Present	8/1988 - Present	9/1982 - 4/1995
Southern California	Joshua Tree National Park	9/1992 - 9/1993 (Special Study - MOHAVE data excluded from Regional Summaries)	None	6/1982 - 9/1992

It should be noted that data summaries for the Sierra Nevada region only include YOSE. The period of record for SEKI (3/1994 - 2/1999) differed too much from the YOSE period of record (3/1988 - 2/1999) to represent an accurate combined regional summary.

a. Visibility Characterization

Visibility is usually characterized by visual range (the greatest distance that a large, black object can be seen against a viewing background) expressed in kilometers (km) or light extinction (the sum of light scattering and absorption per unit distance) expressed in inverse megameters (Mm^{-1}). These two characterizations are inversely related; a visual range (VR) of 391 km signifies the best possible visibility and corresponds to a light extinction (b_{ext}) of 10 Mm^{-1}; as visual range decreases, light extinction increases. Neither visual range nor extinction is linear with respect to increases or decreases in perceived visual air quality. For example, a 15 km change in visual range or 2 Mm^{-1} change in extinction can result in a scene change either unnoticeably small or very apparent depending on the baseline visibility conditions. Therefore, a third visibility characterization, the deciview (dv), was derived by Pitchford and Malm (1994) to index a constant fractional change in extinction or visual range. The advantage of this characterization is that equal changes in deciview are equally perceptible across different baseline conditions. Higher deciview values signify poorer visibility. A zero deciview corresponds to Rayleigh scattering (clean air), 10 Mm^{-1}, or a visual range of 391 km.

Of the three visibility indices, the light extinction coefficient (b_{ext}, commonly called extinction) is the characterization most used by scientists concerned with causes of reduced visibility. Extinction can be directly calculated from light transmittance measurements (measured extinction) or derived from measured particle concentrations (reconstructed extinction). Direct relationships exist between the concentrations of particles and gases and their contribution to the extinction coefficient. Understanding these relationships provides a method of estimating how visibility would change with changes in the concentrations of these atmospheric constituents. This methodology, known as “extinction budget analysis", is important for assessing the visibility consequences of proposed pollutant emission sources, or

for determining the extent of pollution control required to meet a desired visibility condition. Extinction, both measured and reconstructed, is the primary visibility characterization provided in this report.

b. Mechanisms and Sources of Visibility Impacts

Visibility impairment results from both scattering and absorption of light by gases and particles suspended in the air. Light scattering results from the natural Rayleigh scatter from air molecules, (which causes the blue sky) and the scattering caused by suspended particles in the atmosphere (aerosols). Rayleigh scatter is approximately 10 Mm^{-1}, but varies with altitude and the associated density of the atmosphere. Particle scatter can be caused by natural aerosols (e.g., wind-blown dust and fog) or by man-made aerosols (e.g., sulfates, nitrates, organics, and other fine and coarse particles). The effect of particle scatter depends on the particle size, chemical properties, hydroscopic properties, and mass concentration of the particles in the atmosphere. Fine particles have the largest effect on visibility. Fine particles with sizes near the light wavelength 0.4 - 0.7 mm are the most efficient light scatterers. In addition, when water is associated with sulfates, nitrates, and some organics, the total light scattering can increase substantially over corresponding dry conditions. In most parts of the country, sulfates and organics make up the largest mass fractions of the fine aerosol. Although not as abundant, nitrates are also a major contributor to visibility impairment. Coarse particles from natural and man-made sources also affect scattering but have less influence than fine particles.

Light is also absorbed by gases and particles. Nitrogen dioxide (NO_2) is the only major visible light-absorbing gas in the lower atmosphere. Elemental carbon (soot) is the dominant light-absorbing particle in the atmosphere.

With few exceptions, average concentrations of sulfate, organics, and elemental carbon are highest in summer. Nitrate concentrations are generally highest in winter or spring; however, nitrate trends observed in the Southern California region often show higher summer concentrations. Soil concentrations are highest in spring or summer.

c. IMPROVE Station Description and Rationale

A fully complemented IMPROVE station could include fine and coarse particle (aerosol) monitoring, optical monitoring, and view monitoring with photography. A brief summary of each follows:

<u>Particle Monitoring</u>

Particle monitoring provides concentration measurements of atmospheric constituents that contribute to visibility impairment. Until recently, four independent IMPROVE sampling modules were used to automatically collect two 24-hour samples of suspended particles each week by drawing air through filters. (Note: In 2000, sampling frequency changed from twice a week to every third day.) Three of the four samplers (modules A, B, and C) collect fine particles with diameters < 2.5 mm, which are especially efficient at scattering light. The fourth sampler (module D) collects coarse particles with diameters up to 10 mm. Coarse particles do not scatter light well, and therefore, do not contribute much to visibility impairment unless there is a large quantity of coarse particles in the air. The module A, B, and C filters are analyzed to determine the gravimetric mass and chemical composition of the collected particles. These filters are analyzed specifically to estimate the elemental composition and mass of sulfate, nitrate, organic, and elemental carbon species. In addition, the module A filter is used to

estimate the light absorption. All data obtained from the IMPROVE sampling modules are used to determine chemical concentrations and to reconstruct extinction from the known extinction efficiencies of certain species.

Detailed descriptions of the aerosol sampler, laboratory analysis, and data reduction procedures can be found in the Standard Operating Procedures and Technical Instructions for the IMPROVE Aerosol Sampling Network (U.C. Davis 1996).

Aerosol sampler data are used to reconstruct the atmospheric extinction coefficient in Mm^{-1} (inverse megameters) from experimentally determined extinction efficiencies of important aerosol species. Higher extinction coefficients signify lower visibility. Reconstructed extinction data are used as background conditions to run plume and regional haze models. These data are also used in the analysis of visibility trends and conditions.

Standard Visual Range (SVR) can be expressed as:

$$SVR = 3912/(b_{ext} - b_{Ray} + 10)$$

where b_{ext} is the extinction coefficient expressed in Mm^{-1}; b_{Ray} is the site specific Rayleigh values (elevation dependent); 10 is the Rayleigh coefficient used to normalize visual range; and 3912 is the constant derived from assuming 2% contrast detection threshold. The theoretical maximum SVR is 391 km. Note that b_{ext} and SVR are inversely related: for example, as the air becomes cleaner, b_{ext} values decrease and SVR values increase.

Haziness as expressed in deciview is defined as:

$$\text{(haziness) } dv = 10 \ln(b_{ext} / 10Mm^{-1})$$

where b_{ext} is the extinction coefficient expressed in Mm^{-1}. A one dv change is approximately a 10% change in b_{ext}, which is a small but perceptible scenic change under many circumstances. The deciview scale is near zero (0) for a pristine atmosphere and increases as visibility is degraded.

Optical Monitoring

Optical monitoring provides a direct quantitative measure of light extinction to represent visibility conditions. Water vapor in combination with suspended particles can affect visibility, so optical stations also record temperature and relative humidity. Optical monitoring uses ambient, long-path transmissometers and ambient nephelometers to collect hourly-averaged data. Transmissometers measure the amount of light transmitted through the atmosphere over a known distance (between 0.5 and 10.0 km) between a light source of known intensity (transmitter) and a light measurement device (receiver). The transmission measurements are electronically converted to hourly averaged light extinction (b_{ext}, scattering plus absorption). Ambient nephelometers draw air into a chamber and measure the scattering component of light extinction. Data from both of these instruments are recovered at a central location for storage and analyses. Optical measurements of extinction and scattering include meteorological events such as cloud cover and rain. The data are "filtered" by flagging invalid data points with high relative humidities (RH>90%). This filtering process is assumed to remove the effects of weather from the data set. Although nephelometer monitoring has been conducted for special studies performed in the California area, no nephelometer monitoring has been conducted for

any individual NPS Class I area in the Pacific Coast, Sierra-Humboldt, or Sierra Nevada regions.

The measured extinction data are used to verify the calculated reconstructed extinction and can also be used to run plume and regional haze models and to analyze trends and conditions. Because of the larger spatial and temporal range of the aerosol data, reconstructed extinction data are preferred.

View Monitoring

View monitoring is accomplished with automated 35mm camera systems. Cameras typically take three photographs a day (9:00, 12:00, and 15:00) of selected scenes. The resulting slides are used to facilitate data interpretation, and form a photographic record of characteristic visibility conditions. Based on April 1995 recommendations of the IMPROVE Steering Committee, view monitoring has been discontinued at all NPS Class I areas that have a 5 year (or greater) photographic monitoring record. Over a five-year period, varying degrees of visibility impairment at Class I areas can be adequately photographed and recorded. No view monitoring has been conducted at any California NPS visibility monitoring site since April 1995.

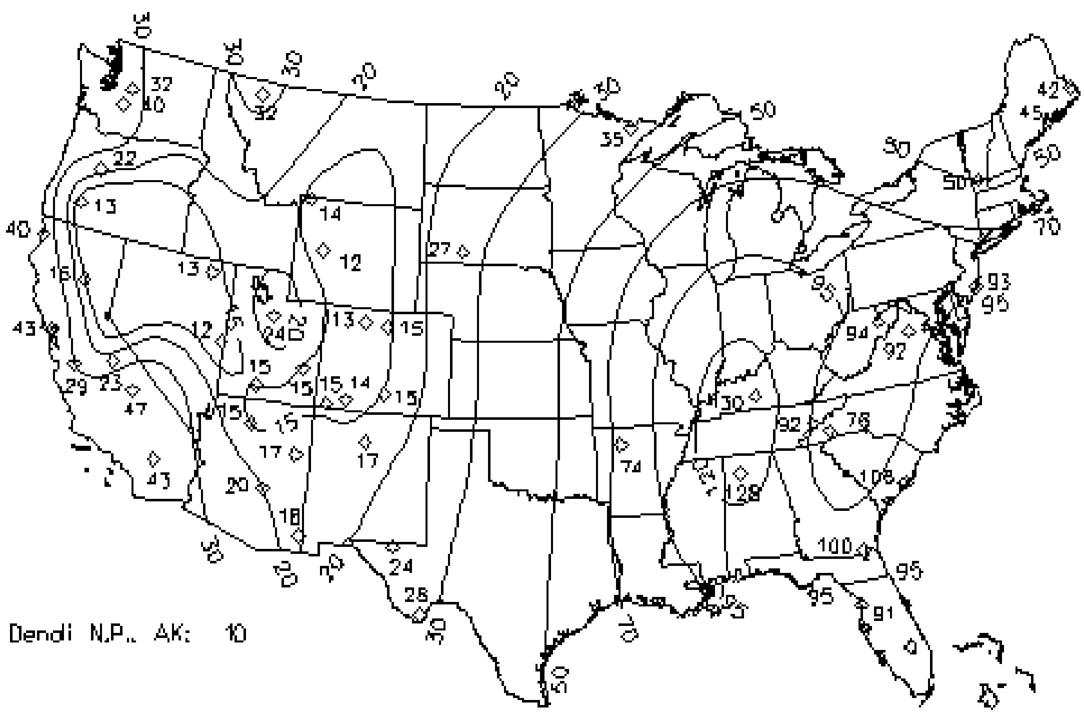
The IMPROVE visibility network consists of 110 particle monitoring sites selected to provide regionally representative coverage and data for all 156 mandatory federal Class I areas. Optical monitoring is conducted at 20 of the 110 IMPROVE sites. Cameras will be placed at new IMPROVE sites where view monitoring has never been conducted. View monitoring is currently conducted at five IMPROVE sites outside of California. A detailed description of the current and expanding IMPROVE visibility monitoring network may be found in Malm et al. (2000).

d. Overview of Visibility Conditions for the Pacific Coast, Sierra-Humboldt, and Sierra Nevada Regions

Particle, optical, and view monitoring data for the Pacific Coast, Sierra-Humboldt, and Sierra Nevada regions were extracted from IMPROVE network data archives and summarized for the March 1988 through February 1999 period.

Figure I-3 summarizes the spatial distribution of total reconstructed light extinction (including Rayleigh) averaged by site over three years (March 1996 through February 1999), as presented in the IMPROVE report (Malm et al. 2000). Figure I-4 provides a graphic summary of measured light extinction by geographic region and by season for the period March 1988 through February 1999. Seasonal arithmetic means of regional IMPROVE transmissometer data were summarized by the following visibility metric categories:

- Worst - conditions represent a mean visual range less than 42 km (b_{ext} >93 Mm^{-1})
- Below Average - conditions represent a mean visual range from 42 km to 78 km (b_{ext} from 93 Mm^{-1} to 50 Mm^{-1})
- Above Average - conditions represent a mean visual range from 78 km to 145 km (b_{ext} from 50 Mm^{-1} to 27 Mm^{-1})
- Best - conditions represent a mean visual range greater than 145 km (b_{ext} <27 Mm^{-1}) For both time periods, the highest light extinction (>100 Mm^{-1}) occurred in the eastern United States and the lowest extinction (<20 Mm^{-1}) occurred in the intermountain West.



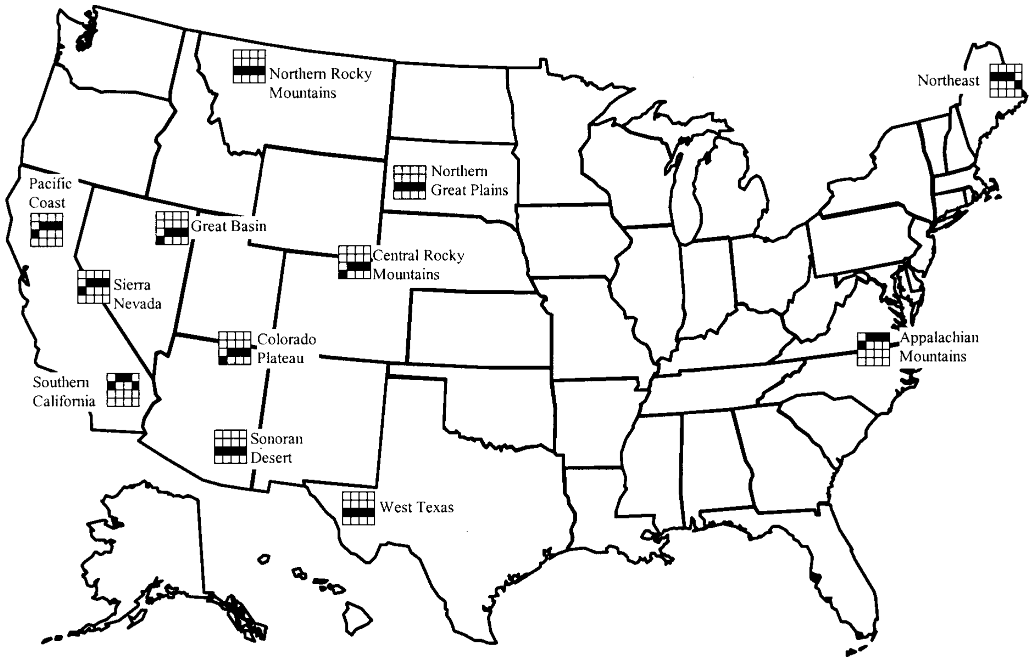
IMPROVE network particle and optical (transmissometer) monitoring sites are represented

Figure I-3. Three-year averages of total reconstructed light extinction coefficient (Mm^{-1}) calculated from the aerosol concentrations at each operational site in the IMPROVE network for the period March 1996 through February 1999.

in Figures I-3 and I-4 respectively. Site-specific seasonal and annual averages of reconstructed and measured extinction for the period March 1988 through February 1999 are provided in Sections III through X. Caution should be used when comparing reconstructed and measured extinction. Given differences in measurement periods and averaging methods, as well as relative humidity filtering methods and effects on light extinction efficiencies, the ratio of reconstructed extinction to measured extinction is seldom greater than 80%.

Figure I-3 shows a three-year average (March 1996 – February 1999) of total reconstructed light extinction, calculated from aerosol concentrations at each IMPROVE monitoring site throughout the U.S. As shown in Figure I-3, the worst visibility conditions in California largely occur in the southern areas of the State. Figure I-4 shows seasonal visibility conditions across the U.S., calculated using data from transmissometer sites in the IMPROVE Network. As shown in Figure I-4, the best visibility at the monitoring sites in California occurs during the winter season.

Tables I-9 and I-10 summarize the seasonal and annual averages of reconstructed light extinction coefficients for the Pacific Coast, Sierra-Humboldt, and Sierra Nevada regions (March 1988 through February 1999). Table I-9 also shows the breakdown of non-Rayleigh extinction among fine and coarse particle scattering, and absorption. Table I-10 further



Visibility Metric:

Average conditions...	Arithmetic mean of all visibility data for all regions. (Visual Range = 78 km, b_{ext} = 50 Mm^{-1})
Best conditions...	Arithmetic mean of the 20% least impaired visibility data for all regions. (Visual Range = 145 km, b_{ext} = 27 Mm^{-1})
Worst conditions...	Arithmetic mean of the 20% most impaired visibility data for all regions. (Visual Range = 42 km, b_{ext} = 93 Mm^{-1})

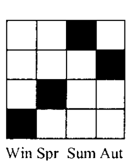
Box Plot Key:

Columns - Represent Meteorological Seasons
- Column 1 - **Winter** (December, January, and February)
- Column 2 - **Spring** (March, April, and May)
- Column 3 - **Summer** (June, July, and August)
- Column 4 - **Autumn** (September, October, and November)

Rows - Represent Defined Visibility Ranges
- Row 1 - Average seasonal visibility is in the "**Worst**" visibility metric category.
- Row 2 - Average seasonal visibility is in the "**Below Average**" visibility metric category.
- Row 3 - Average seasonal visibility is in the "**Above Average**" visibility metric category.
- Row 4 - Average seasonal visibility is in the "**Best**" visibility metric category.

Example Box Plot:

"**Worst**"
"**Below Average**"
"**Above Average**"
"**Best**"



Visual Range less than 42 km ($b_{ext} \geq 93$ Mm^{-1})
Visual Range from 42 km to 78 km (b_{ext} from 93 Mm^{-1} to 50 Mm^{-1})
Visual Range from 78 km to 145 km (b_{ext} from 50 Mm^{-1} to 27 Mm^{-1})
Visual Range greater than 145 km ($b_{ext} < 27$ Mm^{-1})

Figure I-4. Seasonal visibility across the continental United States calculated using b_{ext} data from each operational transmissometer site in the IMPROVE network for the period March 1988 through February 1999.

identifies the contributions of sulfate, nitrate, organics, elemental carbon, and coarse particles (including fine soil) to the non-Rayleigh aerosol light extinction. Figure I-5 graphically depicts the percentage of total light extinction (including Rayleigh) contributed by sulfate, nitrate, organics, elemental carbon, and coarse particles.

Pacific Coast

The Pacific Coast region includes three NPS Class I areas along and near the coast of northern California: PINN, PORE, and REDW. Aerosol monitoring was conducted at all three sites for eleven years (1988-1999). As shown in Table I-9, the average annual extinction during the 1988 through 1999 period for this area is 53.1 Mm^{-1} with 81% due to non-Rayleigh aerosol extinction. The annual variance is moderate and ranges between 53.7 Mm^{-1} during the summer and 46.9 Mm^{-1} in the winter. As shown in Table I-10, sulfate extinction obtains its maximum in the summer at 24.6 Mm^{-1}, when nitrate extinction is near its minimum at 7.9 Mm^{-1}. When nitrate extinction obtains its maximum of 14.2 Mm^{-1} during the winter, sulfate extinction is at its minimum of 9.3 Mm^{-1}. Organic extinction and elemental carbon absorption obtain their maxima in the autumn at 8.4 Mm^{-1} and 3.3 Mm^{-1} respectively. On an annual basis, the largest contributor to non-Rayleigh aerosol extinction is sulfate (44.1%), followed by nitrate (25.1%), organics (13.9%), soil and coarse material (11.7%), and elemental carbon absorption (5.3%). The contribution from sulfate shows considerable variation, ranging from a high in the summer of 56.3% to 25.2% in the winter when its contribution is eclipsed by nitrate, which contributes 38.5%.

Table I-9. Seasonal and annual average reconstructed light extinction (Mm^{-1}) apportioned by general category for the Pacific Coast, Sierra-Humboldt, and Sierra Nevada regions (March 1988 through February 1999).

Season	Total Extinction	Natural Rayleigh Extinction	Non-Rayleigh Extinction	Fine Scattering	Coarse Scattering	Absorption
Pacific Coast						
Spring	49.5	10.0	39.5	32.8	5.0	1.7
Summer	53.7	10.0	43.7	37.1	5.2	1.3
Autumn	52.5	10.0	42.5	34.4	4.8	3.3
Winter	46.9	10.0	36.9	30.0	4.0	2.9
Annual	53.1	10.0	43.1	36.0	4.8	2.3
Sierra-Humboldt						
Spring	24.3	10.0	14.3	10.9	1.9	1.5
Summer	31.9	10.0	21.9	16.8	2.8	2.4
Autumn	23.4	10.0	13.4	9.6	1.8	2.0
Winter	18.6	10.0	8.6	5.5	1.5	1.6
Annual	24.5	10.0	14.5	10.5	2.0	1.9
Sierra Nevada						
Spring	32.4	10.0	22.4	17.8	2.9	1.7
Summer	39.1	10.0	29.1	21.8	3.6	3.7
Autumn	35.2	10.0	25.2	18.8	3.3	3.1
Winter	22.5	10.0	12.5	9.8	1.7	1.1
Annual	33.3	10.0	23.3	18.0	2.9	2.4

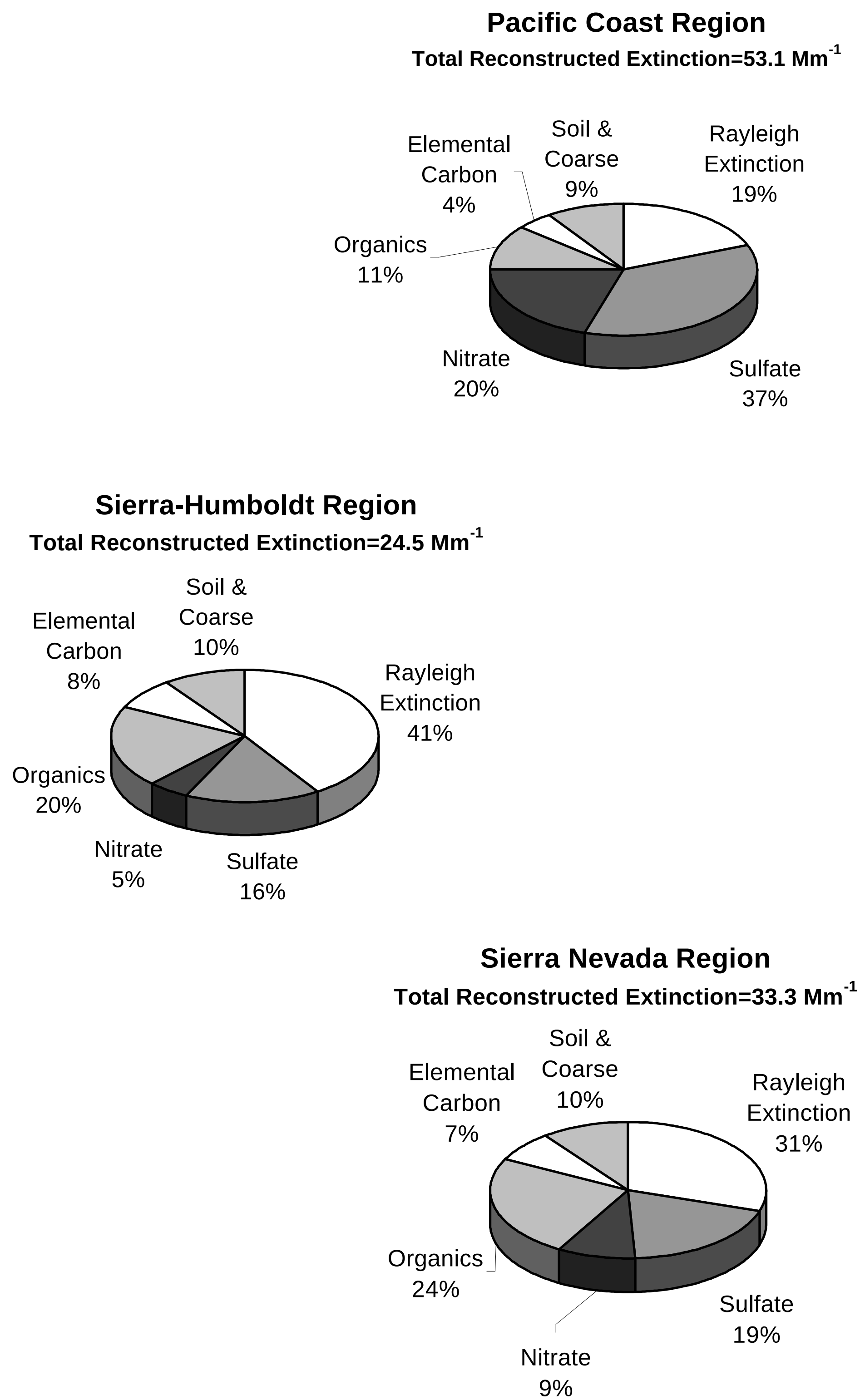


Figure I-5. Annual average percentage of total light extinction (including Raleigh) contributed by sulfate, nitrate, organics, elemental carbon, and coarse particles for the Pacific Coast, Sierra-Humboldt, and Sierra Nevada regions (March 1988 through February 1999).

Table I-10. Contributions of various types of fine particles (Mm^{-1}) to the total seasonal and annual average non-Rayleigh aerosol light extinctions for the Pacific Coast, Sierra Humboldt, and Sierra Nevada regions (March 1988-February 1999).

Season	Non-Rayleigh Aerosol Extinction	Sulfate	Nitrate	Organics	Elemental Carbon (Absorption)	Soil & Coarse Material
Pacific Coast						
Spring	39.5	19.0	8.9	4.6	1.7	5.4
Summer	43.7	24.6	7.9	4.4	1.3	5.5
Autumn	42.5	16.7	8.9	8.4	3.3	5.1
Winter	36.9	9.3	14.2	6.3	2.9	4.1
Annual	43.1	19.0	10.8	6.0	2.3	5.0
Sierra-Humboldt						
Spring	14.3	4.7	1.6	4.0	1.5	2.5
Summer	21.9	7.3	1.7	7.2	2.4	3.3
Autumn	13.4	2.8	0.8	5.6	2.0	2.2
Winter	8.6	2.0	1.0	2.4	1.6	1.6
Annual	14.5	4.0	1.2	4.9	1.9	2.5
Sierra Nevada						
Spring	22.4	7.4	4.1	5.6	1.7	3.6
Summer	29.1	6.5	1.9	12.6	3.7	4.3
Autumn	25.2	5.2	2.6	10.4	3.1	3.9
Winter	12.5	3.0	3.4	3.3	1.1	1.8
Annual	23.3	6.4	3.1	8.0	2.4	3.4

Sierra-Humboldt

The Sierra-Humboldt region resides further north in the Sierra Nevada and Humboldt mountain ranges. Aerosol monitoring was conducted eleven years (1988 - 1999) at both Crater Lake National Park in southern Oregon and LAVO in northern California. As shown in Table I-9, for this region, the total reconstructed light extinction averaged 24.5 Mm^{-1} with maximum extinction in summer (31.9 Mm^{-1}) and minimum extinction in winter (18.6 Mm^{-1}). Seasonality is primarily driven by variations in sulfate and organic extinctions. Organics contribute the most to extinction (33.9%), followed by sulfate (27.7%), soil and coarse particles (17.0%), elemental carbon (13.1%), and nitrate (8.3%; Table I-10).

Sierra Nevada

Two NPS sites, YOSE and SEKI represent the Sierra Nevada region. Aerosol and optical monitoring were conducted at YOSE for eleven years (1988 - 1999). SEKI had module A and D samplers since March 1992 but was not instrumented with a full IMPROVE sampler until March 1994. Aerosol monitoring was conducted at the SEKI site for five years (1994 - 1999). Because the period of record between these two sites differs, only the YOSE aerosol data are included in the Sierra Nevada aerosol regional summaries. The average reconstructed light extinction for this region is 33.3 Mm^{-1} with a seasonal component that has a winter minimum of 22.5 Mm^{-1} and a summer maximum of 39.1 Mm^{-1} (Table I-9). The seasonality is driven primarily by organics and elemental carbon (absorption) with both species peaking during the summer at 12.6 Mm^{-1} and 3.7 Mm^{-1}, then dropping to 3.3 Mm^{-1} and 1.1 Mm^{-1}, their minima,

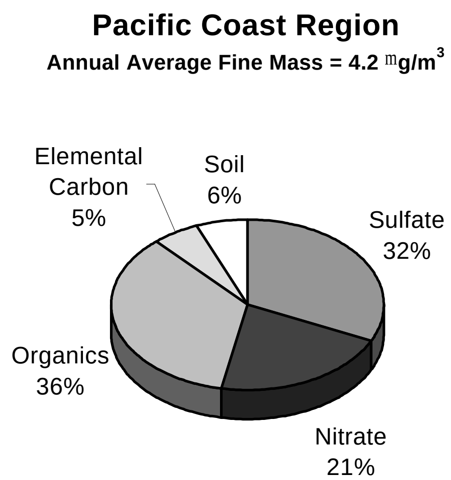
during the winter (Table I-10). Sulfate contributes to a lesser extent to the seasonality; its maximum occurs in the spring at 7.4 Mm^{-1} and its seasonal low occurs in the winter at 3.0 Mm^{-1}. On an annual average, organics contribute the most to aerosol extinction (34.3%), followed by sulfate (27.5%), soil and coarse material (14.7%), then nitrate (13.1%), and finally absorption (10.3%).

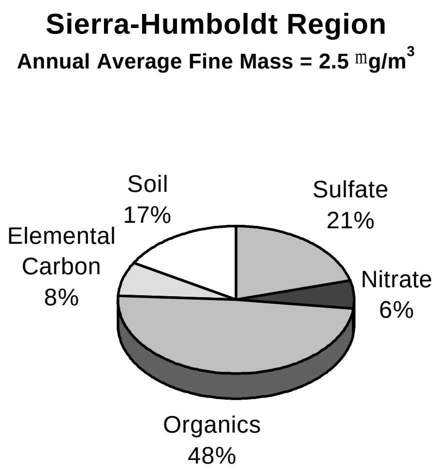
Figure I-6 graphically depicts the measured fine aerosol mass budgets (in percent) for each region. These summaries represent the seasonal and annual mean concentrations for the March 1988 through February 1999 period.

Table I-11 shows the mass concentrations (: g/m^3) of fine and coarse aerosol, and the chemical composition (mass budgets) of fine aerosol for the Pacific Coast, Sierra-Humboldt, and Sierra Nevada regions.

Table I-11. Measured fine and coarse aerosol mass concentrations (in mg/m^3) for the Pacific Coast, Sierra-Humboldt, and Sierra Nevada regions (March 1988 through February 1999).

Season	Coarse Mass	Fine Mass	Sulfate	Nitrate	Organics	Elemental Carbon	Fine Soil
Pacific Coast							
Spring	8.4	3.7	1.3	0.7	1.2	0.2	0.3
Summer	8.7	4.0	1.9	0.7	1.1	0.1	0.2
Autumn	8.0	4.9	1.3	0.8	2.1	0.3	0.3
Winter	6.6	4.2	0.8	1.4	1.6	0.3	0.1
Annual	7.9	4.2	1.3	0.9	1.5	0.2	0.3
Sierra-Humboldt							
Spring	3.2	2.5	0.6	0.2	1.0	0.1	0.6
Summer	4.6	3.5	0.7	0.2	1.8	0.2	0.5
Autumn	3.0	2.6	0.5	0.1	1.4	0.2	0.4
Winter	2.4	1.3	0.2	0.1	0.6	0.2	0.2
Annual	3.4	2.5	0.5	0.2	1.2	0.2	0.4
Sierra Nevada							
Spring	4.8	3.8	1.0	0.5	1.4	0.2	0.7
Summer	6.0	6.3	1.6	0.5	3.2	0.4	0.7
Autumn	5.5	4.9	1.0	0.5	2.6	0.3	0.6
Winter	2.8	1.8	0.3	0.4	0.8	0.1	0.1
Annual	4.8	4.2	1.0	0.5	2.0	0.2	0.5





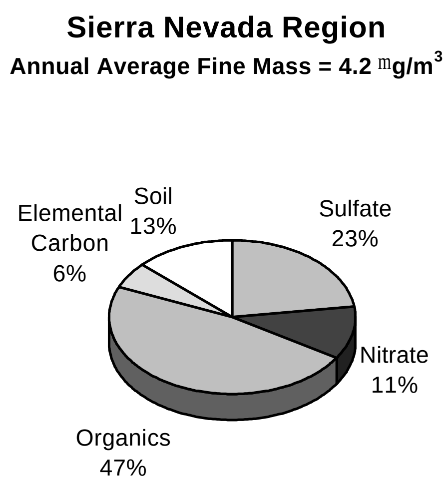


Figure I-6. Annual average measured fine aerosol mass budgets (in percent) for the Pacific Coast, Sierra-Humboldt, and Sierra Nevada regions (March 1988 through February 1999).

e. Visibility Special Studies for California

In addition to the IMPROVE network, the California region has been the focus of many studies that examined visibility, haze, and the sources of pollutants responsible for visibility impairment. The most noted to date has been the seven-year congressionally funded study known as Project MOHAVE (Measurement Of Haze And Visual Effects).

Project MOHAVE was an extensive monitoring, modeling, and data assessment project designed to estimate the contributions of the Mohave Power Plant to haze at Grand Canyon National Park (GRCA). The power plant is located approximately 75 miles southwest of GRCA. It emits up to 40,000 tons of SO_2 each year, and is one of the largest sources of SO_2 in the western United States.

The field study component of the project was conducted in 1992 and contained two intensive monitoring periods (~30 days in the winter and ~50 days in the summer). Unique, non-depositing, non-reactive perfluorocarbon tracer (PFT) materials were continuously released from the power plant stack during the two intensive periods to enable the tracking of emissions specifically from the Mohave power plant. Tracer, ambient particulate composition, and SO_2 concentrations were measured at about 30 locations in a four-state region, including southern California, southern Nevada, southern Utah, and Arizona.

Project MOHAVE operated under the joint technical and program management of the EPA and Southern California Edison in close partnership with the NPS. Numerous other organizations contributed to the operations and assessment work of the project. Since the end of the field study component of the project, data assessment and modeling efforts were undertaken by the many participants and have led to numerous papers and reports.

Conclusions of the study stated: "Detailed analysis of field measurements was unable to link elevated sulfate concentrations with Mohave Power Plant emissions. In general, the concentrations of visibility-impairing species seemed to be affected by regional sources and regional meteorology. Several analyses of concentration patterns and of distributions of the PFT and of other natural tracers all concluded that the dominant sources of GRCA visibility impairment were area sources (principally urban) in Southern California, Arizona, and northern Mexico. The Las Vegas urban area was also implicated in some analyses." (Project MOHAVE Final Report, M. Pitchford, March 1999).

f. Interpretation and Visibility Projections

Data from other IMPROVE visibility sites around the country have been presented graphically (Figures I-2 through I-5) so that visual air quality in the Pacific Coast, Sierra-Humboldt, and Sierra Nevada regions can be understood in comparison with other U.S. regions. The characterizations of trends can be a highly subjective exercise. The quantity of available data and the techniques employed for determining slopes and their significance can vary depending on the statistical technique employed. There is also the underlying year-to-year variability due to meteorology. Only recently has the IMPROVE aerosol network, initiated in March 1988, matured to a point where long-term trends of average ambient aerosol concentrations and reconstructed extinction can be assessed.

Long-term trends fall into three categories: increases, decreases, and insignificant changes. Given the NPS visibility sites and regions summarized for this report, the majority of California data show either no change or a decreasing trend in extinction, indicating improving visibility. Detailed trends analysis is available in the IMPROVE report (Malm et al. 2000).

Ongoing and future monitoring is necessary to identify local source impacts. Additional data and in-depth modeling and analysis are required to further evaluate regional trends and future projections of impact from existing and future sources.

One aspect of visibility impairment, the incidence of prescribed fires and wildfires, is likely to increase in the coming decades as a result of past fire suppression. "Emissions from fire may represent the single most important change in air quality in the next 50 years" (Grand Canyon Visibility Transport Commission, 1996). The effects of fire on visibility will likely vary from year to year, depending on the frequency and intensity of natural and/or prescribed fires in each region.

7. Fire

An important goal of the NPS is the maintenance of natural ecological processes and those processes associated with indigenous people (e.g., burning by Native Americans) prior to European settlement. In national parks throughout California, including in particular SEKI and YOSE, concern has been expressed in recent years about the potential effects of nearly a century of fire exclusion on ecological processes within those parks. The NPS is currently using a prescribed burning procedure to increase the frequency of fire as a disturbance process (Christensen et al. 1987; Parsons 1990, 1994; Piirto et al. 1998). Prior to initiation of the current fire program, exclusion of fire through suppression and creation of fuel discontinuity (through logging, agriculture, and roads outside the parks) resulted in accumulation of large quantities of dead and living fuels in some forest ecosystems. Concern has particularly been expressed for the giant sequoia/mixed conifer stands in SEKI and YOSE. Today, prescribed burning is being implemented widely as a fire management tool on NPS lands.

Fire is one of the most important environmental factors that affects ecosystems in national parks throughout California. Fires were historically a common occurrence, especially in the mixed conifer region of California. Fire has played a variety of roles, including reduction of understory density; seed bed preparation for some tree species; recycling and volatilization of nutrients; regulating plant species distributions, the mosaic of age classes, vegetation types, and wildlife habitat; reduction of numbers of trees susceptible to attack by insects and diseases; and the reduction of intense fire hazard associated with unusual accumulations of fuels.

Natural fire regimes help stabilize park vegetative communities and ecosystems. In some situations, certain plant species are dominant because they are adapted to the successional stages that are maintained by fire. If fire has been restricted, the community structure may change and new species become dominant. For example, in the mixed conifer community, ponderosa pine is favored by open sunny conditions which occur subsequent to fire. In contrast, incense cedar is favored by the deep shade of the dense forest. The structure of the mixed conifer community therefore changes under changing fire regimes. The chaparral community is stabilized by infrequent, but high intensity fires which recycle nutrients and prepare the seed bed for resprouting. Thus, there are dramatic effects of fire on the vegetation, and wildlife species are similarly affected.

Policies related to the management of fire within the national parks have evolved during the past century, ranging from efforts to eliminate all fire to the more recent widespread recognition of the importance of restoring and maintaining fire as a natural ecological process (Parsons and Botti 1996). It is now well known that interactions of vegetation and fire are largely responsible for shaping the ecosystems that most parks have been established to protect. This thinking was incorporated into NPS policy in 1968 when it was formally recognized that the presence or absence of fire is important in determining the characteristics of an area. This

new fire policy permitted use of prescribed burning and allowed lightning-caused fires to be left to burn in order to help to accomplish approved management objectives (van Wagtendonk 1991). It was recognized that fire exclusion had not eliminated fire but rather it had most often simply changed local fire regimes so that fires were less frequent, but in many cases more intense, with results that could be potentially detrimental to management goals of the NPS. By 1988, 26 parks had operational natural fire programs and many others had active programs for igniting prescribed fires (Parsons and Botti 1996). The goal of restoring fire as a natural process was intended to allow burning at similar frequencies and intensities and presumably with similar effects as would have occurred in the absence of fire exclusion.

Interagency recognition of risks and expenses associated with wildland fire management culminated in a 1995 Final Report of the Federal Wildland Fire Management Policy and Program Review. The Secretary of the Interior accepted and endorsed the principles, policies, and recommendations contained in the report, and directed the NPS to implement them. Adoption of a set of umbrella federal fire policies will enhance operations across administrative boundaries and improve capabilities to meet the challenges posed by current wildland fire conditions.

There are a number of obstacles and challenges that must be confronted if the NPS is to meet its goal of restoring fire to something approaching its natural role in park ecosystems. One significant constraint associated with reaching that goal is air quality regulations and their effects on smoke emissions (Parsons and Botti 1996). The importance of the burning of woody fuels as an air quality issue is partly related to visibility impairment. Burning also produces fine particulate, which is regulated by the EPA and is a significant health concern. Those effects can often be minimized by proper timing and preparation for burning (Hall 1972).

Prescribed natural fire programs on federal lands adjacent to heavily populated areas are threatened by conflicting laws. federal land managers' ability to prescribe fire is encumbered by the potential impacts of smoke. To some extent the Wilderness Act and Clean Air Act are at odds on this issue (Chambers and Duncan 1995). The intent of the Clean Air Act was to limit pollutants from human-made sources, but it has been interpreted and been made applicable to lightening ignitions as well. In California, the state board determines and designates from meteorological data days when agricultural burning will be prohibited within each air basin. The burn day requirement is applicable to elevations below 1,829 m. Fire in national park ecosystems can have significant short-term impacts on fine particulate concentrations in the air and also on visibility. Park managers must address the conflict between the ecological need for using fire as a management tool and the public's desire for clean air and good visibility. At the same time that air quality regulations are becoming more stringent, the ecological need for the use of prescribed fire is increasing in many parks and wilderness areas. Fuel loadings have increased to relatively high levels in many cases, a situation that may conflict with fire and wilderness management (Haddow 1995).

II. REGIONAL CHARACTERISTICS

A. ENVIRONMENTAL SETTING

1. The Environment

The environment of California is highly complex, and that complexity is well reflected in the public lands within the state, including Class I areas managed by the NPS. There is more climatic and topographic variation in California than in any other region of comparable size in the United States (Schoenherr 1992). Elevation ranges from 87 m below sea level in Death Valley to 4,418 m on Mount Whitney in Sequoia National Park, representing the lowest and highest elevations in the lower 48 states. Precipitation is similarly variable, from some of the driest areas of the United States to northwestern forests that receive in excess of 300 cm of precipitation per year. Temperature ranges can be extreme, from well below zero in the mountains to over 50 °C in some desert locations. Geological complexity is high, having been influenced heavily by plate tectonics, faulting, volcanism, uplift, and glaciation.

Because California contains a rich diversity of climate and topography, a broad range of habitats for plants and animals has developed. There are more than 5,000 native plant species and nearly 1,000 vertebrate animal species. About one-third of the plant species and one-half of the animal species (including invertebrates) are considered endemic to the state (Schoenherr 1992).

Class I parks in California are distributed across several landforms, including the Sierra Nevada, Cascade Mountains, the coastal and Coast Range Mountains, the Great Basin Desert, and the interface between the Mojave and Colorado Deserts.

The Sierra Nevada is the dominant landform in the state. It is comprised of a large granitic batholith that extends 640 km in length and 80 km in width. The High Sierra was produced by three stages of folding, faulting, and uplift. The first began more than 200 million years ago and was followed by erosion and subsidence. The second stage of uplift occurred about 135 million years ago and was accompanied by batholithic intrusions of granite. This uplift was also followed by a period of erosion. Beginning with the Tertiary Period, a final stage occurred, consisting of four main uplifts and several lesser ones, which raised the range as a series of fault blocks many thousand feet higher than the depression of the eastern block now occupied by the Owens River Valley. Three distinct glacial intervals occurred during the Pleistocene Epoch glaciation, forming U-shaped valleys, cirques, and moraines, and leaving behind glacial eratics.

Most of the uplift occurred along the Sierra Nevada fault zone, which stretches along the eastern edge of the range. This fault zone defines the vertical transition between peaks of the eastern Sierra Nevada, many of which are over 4,200 m elevation, and the Owens Valley, 3,200 m below on the western edge of the Great Basin. Heavy precipitation falls on the western slope of the Sierra Nevada, largely as snow at the higher elevations. The eastern slope lies in the rain shadow of the high mountains and is much drier. Wilderness sections of Yosemite (YOSE) and Sequoia and Kings Canyon (SEKI) National Parks, together with national forest wilderness areas in the Sierra Nevada, comprise the largest roadless area in the contiguous United States (Schoenherr 1992).

The current Mediterranean climate of much of California is dominated by cool, wet winters and dry summers. Not all of California has a typical Mediterranean climate, however. High mountain areas are much colder, northwestern coastal areas wetter, and eastern desert areas drier than the normal Mediterranean climate (Barbour et al. 1993). There are strong climatic gradients from west to east in response to moist air and storm tracks from the Pacific Ocean and orographic effects of the mountains. Annual precipitation increases about 17 cm for every 300 m of elevation up to about 2,400 m elevation. Above that level, most of the moisture has been extracted from the air masses, and precipitation lessens (Barbour et al. 1993). Thus, the high

peaks areas are quite dry. The transition of rain to snow is a major factor controlling vegetation distribution and land use. The general climatic pattern of the past 150 years has been relatively warm and wet, containing one of the wettest half-century intervals of the past 10,000 years (Stine 1996). There is growing concern that climate change in California may have broad ecological impacts on natural resources throughout the state, including Class I areas (Field et al. 1999).

The Sierra Nevada Ecosystem Project (SNEP) recently assembled and assessed regional data necessary to assist in making important policy decisions for future management of the Sierra Nevada. Alternative management strategies were evaluated to meet the goal of maintaining the health and sustainability of Sierra Nevada ecosystems while providing resources to meet human needs. Statistics compiled by the SNEP are relevant to the Class I regional assessment contained herein, and are summarized below.

The lands adjacent to the Class I parks in California are likely to undergo significant land conversion through continuing population growth over the next half century (c.f., Duane 1996). This will result in a range of direct and indirect effects on ecosystem structure and function. Fire regimes in settled areas and adjacent wildland ecosystems will be further altered. Habitat area will be reduced, leading to increased habitat fragmentation. Competition will intensify for available water.

The Sierra Nevada contains more than 3,500 species of native vascular plants. About 400 plant species occur only in the Sierra Nevada, of which 218 are considered rare or threatened by the California Native Plant Society or by federal or state agencies. The foothills woodland and chaparral plant communities, in particular, have been altered by human settlement and contain many rare or threatened plant species.

The SNEP mapped 945 areas of special significance throughout the Sierra Nevada. These areas contain features of particular ecological, cultural or geologic diversity. A feature was considered significant if it was unusually rare, diverse, or representative of natural diversity. Extensive areas within YOSE and SEKI were included in this designation (Millar et al. 1996).

Plant community distributions were mapped by SNEP throughout the Sierra Nevada and southernmost Cascade Mountains. In addition, five key coniferous forest types were assessed for successional status: eastside mixed conifer, white fir, montane mixed conifer, upper montane red fir, and eastside pine. These are the forest types in which structural complexity continues to increase with stand age for at least several centuries, and for which the ecological differences between late successional and earlier successional stages are distinctive. Conifer forests in the middle elevation zones of the Sierra Nevada that are not disturbed by logging or severe fire tend to develop complex structures and high species richness. There are large standing dead and down logs within a mosaic of large mature stands interspersed with openings and younger stands. The mature forest is characterized by high horizontal and vertical complexity, thereby providing habitat for many species that are not found to occur in earlier successional forests. High-quality late-successional middle-elevation conifer forests were found throughout the region, but were especially well-represented in YOSE, SEKI, and Lassen Volcanic National Park (LAVO). Large portions of these parks were designated as areas of late successional emphasis, because they contain much of the remaining large contiguous areas of late-successional forests in middle-elevation conifer types (Davis and Stoms 1996, Franklin and Fites-Kaufmann 1996).

Water is the primary limiting factor that determines the distribution and abundance of many plants and animals throughout California by shaping the landscape and providing a range of habitat conditions. Lakes, streams, and wetlands support a rich diversity of plants and animals in both the aquatic and surrounding riparian habitat. However, these aquatic and riparian systems are the most altered and impaired habitats in the Sierra Nevada (Kondolf et al. 1996) and likely

in the state. Dams and water diversions to meet downstream demands of a growing human population have severely impacted water flows, temperature, and biodiversity. In addition, riparian and in-stream habitats have been severely altered by mining, grazing, and timber harvest. Excessive sediment yields due to up-watershed disturbance, introduction of non-native fish species, habitat degradation, and water quality degradation have all contributed, both regionally and locally, to impaired aquatic and riparian condition and function.

Aquatic and riparian habitats have been extensively altered by human development throughout the state. Of 67 types of aquatic habitat categorized by Moyle (1996), almost two-thirds (64%) were reported to be declining in quality and abundance and many at risk of disappearing completely. A variety of factors contribute to this deterioration. Sedimentation is an extremely important contributing factor, especially in association with roads. Overgrazing of riparian zones, direct modification of streams by dams, diversions and channelization, and mining are all important contributors.

Lassen Volcanic National Park is situated at the southern end of the Cascade Mountains, adjacent to the northern extent of the Sierra Nevada. The Cascade Mountains are volcanic in origin and extend from northern Washington to Lassen Peak. Like the Sierra Nevada, the Cascade Mountain Range exhibits a wide range of climatic conditions, governed by elevation and orographic effects. Precipitation averages about 200 cm per year on the western slope, decreasing substantially on the Modoc Plateau and other high desert regions to the east. Within California, there are two Cascade volcanic peaks: Mount Shasta and Lassen Peak. The latter, located in LAVO, is the most recently active volcano in California.

The Coast Range of California extends from near the Oregon border to the Santa Ynez Mountains of the Transverse Ranges near Santa Barbara. Formed by the movement of tectonic plates, the Coast Range rises abruptly from sea level to nearly 2,000 m elevation in a series of northwest to southeast trending ridges and valleys associated with faulting and folding (Schoenherr 1992). Geological diversity is high, although most rocks are sedimentary, of marine origin. The climate along the coast is heavily influenced by fog and the moderating influence of the cold water offshore. Precipitation exceeds 300 cm at some Coast Range locations, most of which occurs as winter rain.

Coast redwoods occur in the coastal zone of the northern Coast Range, and have been preserved in Redwood National and State Parks (REDW), located just south of the Oregon border. Further south, Point Reyes National Seashore (PORE) is comprised of sedimentary rocks derived from the ocean floor and granite brought up from southern California along the San Andreas fault. Temperatures are relatively constant in near-coastal areas from day to night and season to season, due to the heat transfer properties of the ocean waters. The coastline has been extensively altered by geologic activity and erosional processes. Rocky headlands, sandy beaches, seaside cliffs, estuaries, marine terraces, and sand spits contribute to a wide diversity of plant and animal habitats in these coastal Class I areas.

Chaparral vegetation dominates the southern Coast Range, especially on south-facing slopes. Here, long dry summers associated with the Mediterranean climate favor the drought-tolerant chaparral species. Pinnacles National Monument (PINN) and lower elevation sections of Sequoia National Park preserve the most extensive tracts of chaparral habitat in the NPS system.

Desert areas lie to the east, in the rain shadow of the Cascade Mountains, Sierra Nevada, and Transverse and Peninsula Ranges. Desert areas are characterized by low precipitation (< 25 cm/yr), and include a substantial part of California. The Modoc Plateau is a high desert east of the Cascade Mountains at an elevation of about 1,400 m. It was formed by lava flows, many of which occurred while the Cascade Mountains were being formed. Lava Beds National

Monument (LABE) includes extensive lava flow areas and a rich diversity of cave resources. To the south of the Great Basin Desert is the Mojave Desert, followed by the lower elevation Colorado Desert. The latter is a subdivision of the Sonoran Desert, which stretches to northwestern Mexico and southwestern Arizona. Joshua Tree National Park (JOTR) straddles the border between the Mojave and Colorado Deserts, and includes spectacular high-desert scenery and a high diversity of desert biota.

2. Vegetation of California: the National Park Settings

The vegetation of California is the most diverse in the United States in terms of number of species. The latitudinal extent of the state, climatic variability, topographic variability, geomorphic variability, and broad elevational range contribute to a large number of habitats that support this diverse flora. The national parks included in this report encompass a significant proportion of the primary vegetative groupings in California (Küchler 1964; Figure II-1). It should be noted that the vegetation map displayed here is just one of several vegetation classifications for the state.

Of the 20 vegetative groupings in *Terrestrial Vegetation of California* (Barbour and Major 1988), the Class I parks in the state include 12 groupings, covering the Californian, Sierran, Pacific Northwest, and Hot Desert floristic provinces. These vegetative groupings are described briefly below, with summaries derived from chapters in Barbour and Major (1988). The authors for specific chapters in Barbour and Major (1988) are listed in parentheses, as well as the national parks for which the vegetative grouping is relevant.

<u>Beach and Dune</u> (from Barbour and Johnson)
(PORE)

Beaches are characterized by a maritime climate, high exposure to salt spray and sand blast, and a shifting, sandy substrate with low water-holding capacity. Dunes include open habitat that typically extends from the foredune to inland vegetation types on stabilized substrate. The latter generally have less salt spray and lower soil salinity than beaches and foredunes.

Some of the more common beach species include the rhizomatous American dune grass (*Elymus mollis*) (native) and European beach grass (*Ammophila arenaria*) (exotic). Both species help to stabilize sandy soils, although beach grass grows so densely that it can exclude other species. Nearly all of the dominant beach taxa are perennials, and many share the common traits of being herbaceous, evergreen, succulent, and often prostrate. Shifting sands and human disturbance often remove significant amounts of vegetation, although the non-native *Ammophila arenaria* forms ridges that limit the natural shifting of sands.

Dunes often have considerable topographic variation, including gently rolling areas, ridges, and highly eroded concavities. Dunes are characterized by successional sequences ranging from pioneer beach grasses to late-successional tree species; this sequence typically ranges from grass-dominated foredunes to diverse herbaceous communities to shrub- and tree-dominated stabilized ridges with increasing distance from the beach. Shifting sands, blowouts, and depressions result in a mosaic of topographic diversity and associated diversity in successional state and species composition.

<u>Coastal Salt Marsh</u> (from MacDonald)
(PORE)

Coastal salt marshes are restricted to the upper intertidal zone of protected shallow bays, estuaries, and coastal lagoons, with environmental gradients established in response to elevational changes in the frequency and duration of tidal flooding. This results in vertical

have physiological and regenerative adaptations to drought and periodic fire disturbance. Many species are sclerophyllous, which reduces moisture loss in the dry Mediterranean climate. Dominant woody species in chaparral include chamise (*Adenostoma fasciculatum*), manzanita species (*Arctostaphylos* spp.), ceanothus species (*Ceanothus* spp.), toyon (*Heteromeles arbutifolia*), and poison oak (*Rhus diversiloba*). Several oak species are also found in some chaparral types.

Community dynamics and succession of chaparral are closely linked to fire. In typical high-intensity fires, most or all of the aboveground vegetation is killed. Sprouting species initiate vegetative growth soon after the fire, and significant cover is achieved after a few years. Other species have seeds that sprout prolifically after fire, and they too can attain dominance in the early-successional stand. High concentrations of various secondary chemical compounds in chaparral species affect competitive interactions, contribute to very high temperatures during fires, and encourage post-fire hydrophobicity of soils.

<u>Montane and Subalpine Vegetation of the Sierra Nevada and Cascade Range</u> (from Rundel et al.) (LAVO, SEKI, YOSE)

These forests encompass a diversity of species and vegetative assemblages that vary by latitude, elevation, aspect, and local geomorphology and soils. Lower montane forests are dominated by ponderosa pine on xeric sites. At higher elevations, mixed conifer forest is prevalent, including ponderosa pine, white fir (*Abies concolor*), incense cedar (*Calocedrus decurrens*), sugar pine (*Pinus lambertiana*), and California black oak. Jeffrey pine replaces ponderosa pine within this forest type at higher locations and on shallow soils. In the south-central Sierra Nevada, giant sequoia (*Sequoiadendron giganteum*) is a distinctive component of the mixed conifer forest, and dominates selected groves in terms of size and biomass; this is the world's largest tree species in terms of mass. Periodic fire disturbance plays an important role in stand dynamics and succession in these forests by reducing stem density and encouraging regeneration of pines and giant sequoia. Fire exclusion and extensive logging during the 20th century have significantly altered landscape patterns, fuels, fire regimes, and successional patterns in much of the state.

Upper montane forests have very different species composition than lower elevation forests. Red fir (*Abies magnifica*) is typically the dominant species, often found mixed with white fir at lower elevations. On cooler and/or wetter (higher snowpack) sites, lodgepole pine (*Pinus contorta* var. *murrayana*) is common and sometimes dominant. Because the canopy of upper montane forests is so dense, the understory is often minimal. Regeneration of trees in the understory or gaps is limited by snow accumulation and by the availability of seeds through periodic large cone crops. Fire in these stands typically causes high mortality. Subalpine forest occupies the highest and snowiest sites of all montane forests. Dominated by mountain hemlock (*Tsuga mertensiana*), western white pine (*P. monticola*), whitebark pine (*P. albicaulis*), foxtail pine (*P. balfouriana*), and limber pine (*P. flexilis*), these forests are typically located on shallow, granitic soils. The growing season can be less than 10 weeks, so growth rates of these species are very low, although several species can be very long-lived. While there has been some logging in upper montane forests, subalpine forests have remained relatively free from human activities, with the exception of fire exclusion.

Dominant species composition of the above forest types does not change much between the southern Sierra Nevada and southern Cascade Range. However, the location of individual forest types is typically shifted downward in elevation at northern locations.

Alpine (from Major and Taylor)
(LAVO, SEKI, YOSE)

The alpine zone is above treeline and typically has moderate to high snowfall and cold temperatures, which greatly limit growth and regeneration. Snow avalanches, frost heaving, and cold winds are all stresses found more frequently in the alpine zone. Species diversity can be relatively high in a particular area, with species distribution and abundance governed by differences in radiation, soil moisture, soil fertility, and snowpack distribution. The primary physiognomic contrast observed in the field relates to differences between low shrubs, graminoids (grasses, sedges), and herbaceous species. Species dominance varies significantly along a latitudinal gradient in California.

Redwood Forest and Associated North Coast Forests (from Zinke)
(REDW)

Redwood forest is a distinctive vegetative assemblage found along the western coast of California. Coast redwood (*Sequoia sempervirens*) is the world's tallest species (up to 112 m tall) and dominates this forest in terms of stature and growth rates. Redwood forests contain a diversity of other species, including Douglas-fir, western hemlock, grand fir (*Abies grandis*), Sitka spruce (*Picea sitchensis*), western red cedar (*Thuja plicata*), tanoak, and Pacific madrone. The proportion of individual species within a particular stand varies significantly by latitude and distance from the coast.

Although these forests were historically subject to infrequent fire, the primary disturbance during the 20th century has been logging, with greater than 90% of the original redwood stands having been cut. Older, uncut forests are typified by multi-story structure, gaps, and patchy distribution of understory vegetation. The best representatives of old-growth redwood forest now exist in REDW and other isolated reserves.

Sagebrush Steppe (from Young et al.)
(LABE)

Sagebrush steppe is a series of generally treeless, shrub-dominated communities located along the eastern and northeastern boundary of California. *Artemisia* species are the dominant shrubs, with perennial bunch grasses, other shrubs (e.g., rabbitbrush [*Chrysothamnus* spp.]), and herbaceous species subdominant. Sagebrush (*Artemisia tridentata*) is the dominant species in most areas, with the most common grass species being bluebunch wheatgrass (*Agropyron spicatum*) and Idaho fescue (*Festuca idahoensis*). Shrub steppe grades into western juniper (*Juniperus occidentalis*) and ponderosa pine at higher elevations, and adjacent to forested communities, diversity tends to be higher including the subdominant antelope bitterbrush (*Purshia tridentata*), plum (*Prunus* spp.), currant (*Ribes* spp.), and snowberry (*Symphoricarpos* spp.).

Sagebrush steppe is characterized by cold winters and warm, dry summers. Optimal temperature and soil moisture are concurrent only during late spring, so most herbaceous growth occurs during this brief period. Moisture availability can vary considerably from year to year. Fire is an important form of disturbance in sagebrush steppe, with fire generally killing sagebrush, which then regenerates by seed; many other species, including other shrubs and bunchgrasses, are prolific sprouters and can dominate a site for up to 20 years. While fire has been excluded over much of the landscape during the 20th century, it appears that the increased dominance of exotic cheatgrass (*Bromus tectorum*) has increased fire frequencies at some locations and altered successional patterns. Grazing has had a major impact on overall species

composition and fire ecology through selective removal of forage species and introduction of exotic species.

Transmontane Coniferous Vegetation (from Vasek and Thorne)
(JOTR, LABE)

Northern juniper woodlands occur on open, rolling topography and on mountain ridges and slopes. Dominated by western juniper, these woodlands have a number of different shrubs in the understory, including sagebrush species, curl-leaf mountain mahogany (*Cercocarpus ledifolius*), Utah serviceberry (*Amelanchier utahensis*), desert gooseberry (*Ribes velutinum*), and antelope bitterbrush. A wide variety of herbaceous flora is also found in juniper woodlands. At higher elevations, juniper woodland grades into Jeffrey pine. Fire tends to kill the juniper overstory, resulting in an increased dominance of other shrub species for up to several decades after fire. It has been suggested that due to fire exclusion, juniper now occupies a greater proportion of the landscape than it had previously.

Mojave Desert Scrub (from Vasek and Barbour)
(JOTR)

This vegetation type includes a broad range of desert shrubland and woodland vegetation in the Mojave Desert at elevations below coniferous woodland and forest. Compared to the Sonoran Desert, the Mojave Desert has relatively low plant species diversity. Frequently, creosote bush (*Larrea tridentata*) alone or with one or a few associates, dominates desert communities, with creosote bush and white burroweed (*Ambrosia dumosa*) forming the most characteristic association. Species diversity tends to increase with topographic diversity. Creosote bush scrub occurs on well-drained, sandy flats and upland slopes throughout much of the Mojave Desert.

Xerophytic saltbush scrub is dominated by three species of saltbush (*Atriplex* spp.) that occur in basins and valleys throughout the Mojave Desert region. Halophytic saltbush scrub occurs on playas, in sinks, and near seeps with surface or ground water high in mineral content. Several succulent species in the Chenopodiaceae dominate halophytic communities. Gradients between the saltbush scrub vegetation types can be found near dry lakes, with the halophytes dominant near the old lake shore, xerophytes increasingly dominant remote from the lake shore, and often creosote bush replacing the xerophytes with sufficient increase in elevation and reduction in salinity.

Joshua tree woodland is a distinct type of desert scrub vegetation dominated by Joshua tree (*Yucca brevifolia*) in the overstory and various shrubs and herbs in the understory. It occurs between the range of creosote bush scrub and pinyon and California juniper (*Juniperus californica*) woodlands in the high desert. This species of *Yucca* attains considerable stature and has a wide range of growth forms.

Sonoran Desert (from Burk)
(JOTR)

The vegetation of the Sonoran desert encompasses several specific associations including creosote bush scrub, cactus scrub, wash woodland, palm oasis, saltbush scrub, and alkali sink. These associations are typically differentiated along clines of soil salinity and water availability.

Creosote bush scrub is dominated by creosote bush and white burroweed, which were discussed above. Cactus scrub is of limited distribution and is usually surrounded by creosote bush scrub. Dominant species include prickly pear species (*Opuntia* spp.), barrel cactus (*Echinocactus acanthoides*), and hedgehog cactus (*Echinocereus engelmanii*). Wash woodland

occurs in the margins of arroyos and can support a relatively dense growth of trees, including palo verde (*Cercidium floridum*), desert ironwood (*Olneya tesota*), honey mesquite (*Prosopis glandulosa*), and desert willow (*Chilopsis linearis*). Palm oases, which are limited to riparian areas with a year-round water supply, are dominated by California fan palm (*Washingtonia filifera*). Saltbush scrub is dominated by *Atriplex* species and found in areas with relatively high soil salinity, as described above. Alkali sink is found in high-salinity soils along the margins of the Salton Sea and the Colorado River, and is dominated by iodine bush (*Allenrolfea occidentalis*) and Mojave seablite (*Suaeda torreyana*).

3. Fish and Wildlife

Animal species diversity is high in many portions of California. For example, the Sierra Nevada contains about 400 species of terrestrial vertebrates, including 232 birds, 112 mammals, 32 reptiles, and 25 amphibians. Three modern species that were once well-distributed throughout the region have now disappeared: grizzly bear (*Ursus arctos*), California condor (*Gymnogyps californianus*), and least Bell's vireo (*Virea belli pusillus*). About 17% of the Sierran terrestrial fauna are listed by California as being of special concern or by federal land managers as endangered, threatened, or sensitive (Graber 1996).

Stohlgren and Quinn (1992) evaluated existing natural resources data for 40 national parks and monuments in the NPS Western Region, which includes California. The majority of the park species lists were less than 80% complete in species, geographic, and ecologic (community type) coverage. None of the parks had had parkwide systematic surveys. Data were particularly deficient, or lacking entirely, for invertebrates and nonvascular plants. Stohlgren and Quinn (1992) concluded that there was an urgent need to develop a hierarchical framework of methods to collect and synthesize biological resource data over large spatial and long temporal scales.

The Point Reyes Peninsula has one of the highest diversities of bird species in the United States. Over 480 bird species have been recorded at PORE, about half of the bird species in the country.

At least 30 non-native fish species have been introduced to Sierran waters, ten of which have become widespread. Fish introductions have been especially pronounced at the higher elevations (> 1,800 m) which lacked any fish fauna prior to the trout introductions that began during the nineteenth century. Predatory trout have dramatically altered lake and stream ecosystems and have been implicated in the decline of a number of amphibian species, in particular the pronounced decline of the mountain yellow-legged frog (*Rana muscosa*). The historic distribution of the mountain yellow-legged frog corresponded very closely with location of the zone of historically fishless lakes and streams (Jennings 1996). Fish were introduced to many lakes in this high-elevation zone and are believed to have played a role in the recent decline of this species. Of the 40 species of fish native to the Sierra Nevada, eight are formally listed as threatened or endangered and twelve are candidates for listing. Four other species are in decline in the Sierra Nevada, although less threatened elsewhere (Jennings 1996). Rare fish are also found in coastal streams, including the last remaining wild runs of coho salmon in California. Principal causes of the declines have included dams and water diversions; overharvest; introductions of exotic fish species; alterations of stream channels; and watershed disturbance from grazing, mining, logging, and road building.

There has been an increasing interest during the 1990s within California regarding the protection and management of amphibian and reptile resources on public lands. This interest is due, at least in part, to the widely recognized problem of declining amphibian populations throughout the world (Blaustein and Olson 1991). In addition, 28 taxa in California are in need

of some degree of protection, and 22 species are listed as threatened or endangered under state and federal regulations (Jennings and Hayes 1994, Jennings 1995).

Serious declines and extirpation of frogs and toads have been documented in many areas of the United States. National park areas in California have been affected, including YOSE, SEKI, and LAVO. Apparently well-protected populations have disappeared for no obvious reasons. Possible causes for these declines include acid deposition, disease, habitat loss, increased ultra-violet light, or a combination of such factors. The decline and extinction of amphibians has also apparently been aggravated in many parks by the introduction of non-native fish. Historically it was possible for amphibians to use streams as dispersal corridors. However, the presence of introduced fish in many of the Sierran lakes and streams has greatly reduced or eliminated this possibility (Bradford et al. 1993).

Bradford and Gordon (1992; see also Bradford et al. 1993, 1994a) studied several species of amphibian as potential indicators of adverse ecological effects of acidic deposition in the Sierra Nevada. They conducted laboratory dose-response studies to determine the sensitivity of four amphibian species to low pH and elevated aluminum concentration in early life stages, and they also conducted field surveys to characterize the abundance and associated water chemistry of amphibian populations at high elevation. The results suggested that amphibians were at little risk from low pH in waters that might be acidified to an estimated extreme of pH 5.0 due to acidic deposition. The results also suggested that amphibians were at little risk from the Al levels that were tested, which ranged from 39 to 80 µg/L. Nevertheless, the authors concluded that the possibility exists that observed sublethal effects in response to pH as high as 5.25, or elevated aluminum at the above indicated levels, might represent significant threats to amphibian populations due to reduced growth rate and early hatching. Potential breeding sites were surveyed for two declining and one nondeclining species at high-elevation within 30 randomly selected survey areas. The authors compared chemical parameters between sites that contained the species versus sites that lacked the species. No significant differences were found, however, suggesting that the water chemistry was not different among the sites inhabited by the three species. These findings implied that acidic deposition was unlikely to have been a cause of recent amphibian population decline in the Sierra Nevada (Bradford and Gordon 1992; Bradford et al. 1993, 1994a). Laboratory dose-response studies were conducted for the mountain yellow-legged frog, Yosemite toad, Pacific tree frog, and long-toed salamander (*Ambystoma macrodactylum*). Survival was high for all four species at higher pH levels, but declined dramatically as a function of pH in the mid to low 4s. Al caused a significant reduction in total length of Yosemite toad tadpoles at pH 5.3 and 5.8, but not at pH 4.9, and for long-toed salamander larvae at pH 5.3, but not 4.9 or 5.8. Al has often been found to be less toxic to fish at lower pH values, particularly those below 5, than at higher pH values.

The sensitivity of aquatic amphibians to air pollution impacts is especially pertinent to examine for a number of reasons (c.f., Bradford and Gordon 1992). Many of the physiological and ecological characteristics of amphibians render them particularly sensitive to environmental change or degradation (Blaustein and Wake 1990). Aquatic amphibians are also especially pertinent to examine in the Sierra Nevada. They were historically the only native aquatic vertebrates throughout most of the High Sierra. Subsequently many amphibian populations were eliminated by introduced fishes. Aquatic amphibians at high elevation in the Sierra Nevada breed during or shortly after snowmelt, thereby exposing to the acidifying influence of the Sierran snowmelt the early life stages that are the most sensitive to acidification (Freda 1990).

Perhaps the greatest justification for focusing attention on amphibians in the High Sierra is that dramatic population declines appear to have already occurred or are in progress for at least two of the five species that still occur in the High Sierra (Pierce 1985, Bradford and Gordon

1992). These are mountain yellow-legged frog and Yosemite toad, both of which are restricted to high elevation. Many of the population declines of these species have occurred in seemingly pristine environments, including areas within SEKI, YOSE, and numerous wilderness areas. Prior to the introduction of game fish, mountain yellow-legged frog was probably the most prevalent aquatic vertebrate in mid and high-elevation lakes in the Sierra Nevada (Bradford 1989). Recent surveys have indicated, however, that the species has disappeared from much of its former range in the past two to three decades (Phillips 1990, Bradford and Gordon 1992). Surveys for Yosemite toad indicated that it had disappeared from approximately 50 percent of its former range (c.f., Bradford and Gordon 1992). For example, populations that were observed regularly since the 1950s in the Tioga Pass area near YOSE have had little or no reproductive success since 1982.

Increased levels of ultraviolet light due to thinning of the upper atmosphere ozone layer has also been suggested as a cause of world-wide amphibian decline (c.f. Wake 1991, Wyman 1990). There is no direct evidence, however, that increased ultraviolet radiation has caused the decline of amphibian species. High elevation populations of amphibians have adaptations to protect against intense ultraviolet radiation, such as dense concentrations of melanin in their eggs and reproductive organs. It is possible, however, that they are near their physiological limits of exposure at current ultraviolet radiation levels. Drost and Fellers (1994) suggested that the observed tendency towards declines of high elevation populations of Pacific tree frog is indirect, but intriguing evidence that some broad-scale phenomenon such as increased ultraviolet light might be involved in the amphibian declines.

The question of potential effects of acidic deposition on aquatic amphibian populations in the Sierra Nevada has been well studied, but definitive answers have not yet been provided. The effects of increased acidity may warrant further study. Surface water acidification is a possible contributor to recent amphibian declines, along with drought, pesticides or other toxic chemicals, increased ultraviolet radiation, and predation by introduced fish (c.f., Bradford and Gordon 1992; Bradford et al. 1992, 1994b, 1998; Drost and Fellers 1994). If acidification of high-elevation surface water does indeed play a role in the observed amphibian decline, such a role is likely highly episodic in nature.

4. Land Use and Human Impacts

Historic and current land uses such as logging, grazing, overharvest of fish, and water diversion in and around the Class I areas have had major impacts on natural resources and their potential sensitivity to environmental stresses, including air quality degradation. Changing NPS policies have further complicated the already complex interactions between air pollution and environmental health, especially with respect to fire suppression and fish stocking. Still other issues, that are regional or global in scope, may affect environmental response functions across multiple Class I areas. The latter include climate change and recent widespread amphibian declines in the western United States.

The primary impact of logging during the past 150 years on the conifer forests in and around many of the Class I areas in California has been to simplify the structure, and presumably also the function, of these forests. Large trees, snags, large woody debris, canopies of multiple heights, and complex spatial mosaics of vegetation have been replaced by relatively continuous, even-aged forests of trees that are similar in height, diameter, and canopy characteristics. Grazing has also had significant effects on vegetation communities in most, or perhaps all, of the parks. Meadows and riparian communities have been impacted more heavily than other vegetation types. Extensive grazing of such communities since the 19^{th} century has altered plant species composition and has been accompanied by extensive invasion of nonnative plants. More

recent cessation of grazing, in combination with fire suppression, may have contributed to the recent invasion of subalpine and montane meadows with conifers.

Human population shifts and patterns of movement throughout California are, and will continue to be, important determinants of air pollution levels and air quality degradation in Class I areas. Changes in mobile source contributions of air pollutants and pollutant precursors to Class I areas, as well as future stationary source development, are closely tied to changes in human population patterns upwind of sensitive resources. In addition, upwind agricultural development and park visitation are heavily influenced by human population dynamics and demographics, and can affect in-park air quality and resource damage.

B. REGIONAL AIR QUALITY

Air quality in portions of California is among the best in the United States, in particular in remote sections of the Cascade Range and the Sierra Nevada and at near-coastal northern locations. However, southern airsheds on the west side of the Sierra Nevada (including those in SEKI and YOSE) experience elevated ozone concentrations and nitrogen deposition, especially during summer. Other Class I areas throughout the state experience seasonal to chronic periods of poor air quality, especially of high ozone concentrations. These areas include PINN, JOTR, and LAVO.

California is the most populous state in the nation, with major population centers located along the Pacific coast and in the Sacramento and San Joaquin valleys. Because the prevailing winds are from the west and northwest, many of California's national parks and Class I areas, including those located in the Sierra Nevada and the southeastern deserts, are usually downwind of the most populated portions of the state. Pollutant transport from urban to more remote regions of California has been studied extensively, though the present state of knowledge does not permit precise numerical estimates of quantitative relations between urban emissions and regional air quality. Nonetheless, urban-area emissions and air quality are known to affect air quality in California's national parks. Consequently, the more readily observed urban air quality problems are a good indicator of the types of air quality impacts to be expected at downwind locations. Moreover, the potential driving influence of upwind emissions on air quality in parks and Class I areas makes statewide trends in emission levels and air quality of broad interest. Indeed, urban trends are generally more readily detectable than trends in the Class I areas because ambient air pollutants are present at higher concentrations in urban than in remote areas, and because emission sources are concentrated in urbanized portions of the state.

Statewide, the air pollutants that have been most problematic and required the most regulatory attention are ozone and particulate matter (PM). National and state regulatory efforts have focused heavily on ozone and PM because they have known adverse effects on human health. Ozone also adversely affects some species of vegetation, and PM contributes to visibility degradation; thus, these same two pollutants are of concern for their effects on the public welfare as well as on public health. In addition, deposition of N species (NO_3^- or NH_4^+) via precipitation, fog, or as dry deposition, may affect aquatic and terrestrial systems; unlike ozone and PM, N deposition has not been a health concern.

A common set of emissions affects ambient air concentrations of ozone and PM, as well as N deposition. Ozone is a secondary pollutant, forming in the atmosphere from chemical reactions involving sunlight, oxides of N (NO_x), and volatile organic compounds (VOCs). PM consists of both primary, or direct, emissions of particulate matter and of secondary PM formed from reactions of gas-phase compounds. Typically, primary PM in the size range of 2.5 to 10 micrometer (μm) is dominated by geologically-derived materials; such coarse PM may be affected by nonanthropogenic sources, but human activities contribute road, construction, or

agricultural dust. In California, PM less than 2.5 μm typically consists largely of organic and elemental C and, especially during the winter months, of NH_4NO_3. In urban areas, organic and elemental C are emitted by automobiles and other mobile sources, as well as sources of wood smoke; like emissions of CO and some VOCs, organic and elemental C are derived from incomplete combustion. Particulate organic C also forms from reactions of gas-phase VOCs. In the Sierra Nevada and other forested regions, wildfires are another source of particulate C. Particulate NO_3^- forms from two gas-phase precursor species: NH_3 and HNO_3. The reaction is an equilibrium reaction, with the particulate phase favored under cool or moist conditions. Ammonia is emitted from many sources, including anthropogenic ones such as feedlots. Nitric acid is produced in the atmosphere from reactions involving NO_x; these same reactions also generate ozone. Besides PM, therefore, NO_x and VOC emissions are of interest for their roles in generating ozone and secondary particulates; VOC and CO emissions further provide some indication of relative emission levels of organic and elemental C. SO_2 emissions in California have been low since the early 1980s; thus, SO_2 and particulate SO_4^{2-} levels tend to be of lesser concern than other pollutants.

1. Emissions

a. Current Emission Levels Within All Air Basins

Tables II-1 through II-4 contain 1995 emissions for SO_x, NO_x, reactive organic gases (ROG - essentially synonymous with VOCs), and particulate matter of 10 μm diameter or less (PM_{10}) for all air basins in California (depicted in Figure II-2), as reported by the California Air Resources Board (CARB). The areas with highest emissions are the South Coast, San Francisco Bay Area, and San Joaquin Valley air basins; depending upon the pollutant, the Sacramento Valley, San Diego, and South Central Coast air basins also exhibit high emission levels. SO_2 emissions are highest in the San Francisco Bay Area (SFBAAB) and the South Coast (SoCAB) air basins, though emissions remain below 30,000 tons/year in each of these basins (Table II-1). The SoCAB has the highest NO_x emissions, primarily from vehicles (Table II-2). The SFBAAB and San Joaquin Valley Air Basin (SJVAB) also have high NO_x emissions, mainly from vehicles, but also with significant contributions from fossil-fueled power plants (Table II-2). ROG emissions are highest in the SoCAB, mainly from vehicles, and followed by emissions from cleaning and surface coatings and solvent evaporation (Table II-3). The SFBAAB and the SJVAB have the next highest emissions of ROG, mainly from vehicles. Other significant sources in the SFBAAB include cleaning and surface coatings and solvent evaporation. In the SJVAB, other significant sources of ROG include petroleum production and marketing and solvent evaporation. The highest particulate emissions occur in the SoCAB (Table II-4), followed by the SJVAB, Salton Sea and Sacramento Valley air basins. Most of these particulate emissions are from dust, construction, farming, residential fuel combustion and waste burning.

As listed, the emissions estimates understate the contributions from motor vehicles. During the past ten years, several studies have shown that ambient concentrations of CO and ROG were definitively greater than would have been expected based upon emission inventory estimates (Lawson et al.1990, Stephens and Cadle 1991). The CARB has re-examined the methods used for estimating motor vehicle emissions and developed a new estimation procedure (EMFAC2000); CARB staff have recommended that the CARB adopt EMFAC2000 to estimate motor vehicle emissions beginning in the year 2000 (CARB 1999c). Compared with past estimates, EMFAC2000 will increase the estimates of statewide motor vehicle emissions of CO by 93 percent, ROG by 78 percent, and NO_x by 68 percent (CARB 1999c).

Table II-1. Annual emissions of SO_x for air basins in California in 1995. (Source: CARB Emissions Website 1999a). Units are 1000 tons/year.

	Stationary Sources					Area-Wide Sources		Mobile Sources		Natural Sources	
Air Basin	Fuel Combustion	Waste Disposal	Cleaning and Surface Coatings	Petroleum Production and Marketing	Industrial Processes	Solvent Evaporation	Misc. Processes	On-Road Motor Vehicles	Other Mobile Sources	Natural (Non-anthropogenic) Sources*	**Total****
Great Basin	0.4	na	na	na	0.0	na	0.0	0.0	0.0	0.0	0.4
Lake County	0.0	na	0.0	na	0.0	na	0.0	0.0	0.0	0.4	0.4
Lake Tahoe	0.0	0.0	na	na	na	na	0.0	0.0	0.0	na	0.0
Mojave Desert	1.5	0.0	na	na	2.6	na	0.0	0.7	1.1	na	5.8
Mountain Counties	0.7	0.0	na	na	0.0	na	0.4	0.4	0.4	na	1.8
North Central Coast	0.4	na	na	0.0	0.4	na	0.0	0.4	0.7	na	1.8
Northeast Plateau	0.0	na	na	na	na	na	0.0	0.4	0.4	na	0.7
Sacramento Valley	0.4	0.0	na	0.0	0.4	na	0.4	1.8	1.8	na	4.4
Salton Sea	0.4	na	na	na	0.0	na	0.0	0.4	0.4	na	1.1
San Diego	1.1	0.0	na	na	0.0	na	0.0	1.8	1.5	na	4.4
San Francisco Bay	3.7	0.0	na	13.1	2.6	na	0.4	2.9	6.9	na	29.9
San Joaquin Valley	5.8	na	na	0.0	2.2	na	0.0	2.6	0.7	na	11.7
South Central Coast	1.1	0.0	na	4.0	0.4	na	0.0	0.7	0.7	na	7.3
South Coast	2.6	0.4	0.0	4.4	0.7	na	0.4	8.4	11.3	na	28.1

* Geologic
** Totals reflect emissions in each category before rounding
na not available

Table II-2. Annual emissions of NO_x for air basins in California in 1995. (Source: CARB Emissions Website 1999a). Units are 1000 tons/year.

Air Basin	Stationary Sources					Area-Wide Sources		Mobile Sources		Natural Sources	
	Fuel Combustion	Waste Disposal	Cleaning and Surface Coatings	Petroleum Production and Marketing	Industrial Processes	Solvent Evaporation	Misc. Processes	On-Road Motor Vehicles	Other Mobile Sources	Natural (Non-anthropogenic) Sources*	**Total****
Great Basin	0.4	0.0	na	na	0.0	na	0.0	2.6	0.7	0.0	3.7
Lake County	0.0	na	0.0	na	0.0	na	0.0	1.5	0.4	0.0	2.2
Lake Tahoe	0.0	0.0	na	na	na	na	0.0	1.1	0.0	0.0	1.1
Mojave Desert	21.5	0.0	0.0	0.0	14.2	na	0.7	24.5	18.3	0.0	80.3
Mountain Counties	2.2	0.0	na	na	0.0	na	0.7	13.1	5.5	0.4	21.9
North Central Coast	7.7	0.0	na	0.0	1.1	na	0.7	14.6	5.5	0.0	29.6
Northeast Plateau	0.4	0.0	na	na	na	na	0.4	6.2	6.2	0.4	13.5
Sacramento Valley	8.0	0.0	0.0	0.7	1.1	na	2.6	65.7	20.4	0.4	98.6
Salton Sea	4.4	na	0.0	na	0.0	na	0.4	16.4	6.6	0.0	27.7
San Diego	5.5	0.0	na	na	0.0	na	1.8	65.7	12.4	0.4	87.6
San Francisco Bay	35.4	0.0	0.0	3.7	1.1	na	8.0	124.1	40.2	0.0	211.7
San Joaquin Valley	62.1	0.0	na	0.0	8.4	na	4.4	91.3	28.5	0.4	197.1
South Central Coast	5.8	0.0	na	0.0	0.0	na	1.5	29.9	11.0	0.4	47.5
South Coast	40.2	0.4	0.0	3.3	2.6	na	12.4	288.4	91.3	0.7	438.0

* Wildfires
** Totals reflect emissions in each category before rounding
na not available

Table II-3. Annual emissions of ROG for air basins in California in 1995. (Source: CARB Emissions Website 1999a). Units are 1000 tons/year.

	Stationary Sources					Area-Wide Sources		Mobile Sources		Natural Sources	
Air Basin	Fuel Combustion	Waste Disposal	Cleaning and Surface Coatings	Petroleum Production and Marketing	Industrial Processes	Solvent Evaporation	Misc. Processes	On-Road Motor Vehicles	Other Mobile Sources	Natural (Non-anthropogenic) Sources*	**Total****
Great Basin	0.0	na	0.0	0.0	na	0.7	0.4	1.8	1.8	0.4	5.1
Lake County	0.4	na	0.4	0.0	0.0	0.7	0.4	1.5	0.7	0.4	4.4
Lake Tahoe	0.0	0.0	0.0	0.0	na	0.4	0.7	1.8	0.7	0.0	3.7
Mojave Desert	1.1	0.0	1.8	0.7	0.7	3.7	2.6	15.7	4.7	0.0	31.8
Mountain Counties	0.4	0.0	1.5	0.4	0.4	3.7	7.3	11.0	11.0	0.7	36.1
North Central Coast	0.4	0.7	3.3	0.7	0.0	7.7	1.8	8.0	1.8	0.4	28.5
Northeast Plateau	0.0	0.0	0.4	0.4	0.0	1.8	4.7	3.7	4.0	0.7	15.7
Sacramento Valley	0.4	0.4	9.5	5.8	2.2	15.3	12.4	54.8	8.8	1.1	109.5
Salton Sea	0.4	0.0	1.5	0.4	0.0	4.0	2.2	10.2	1.8	0.4	20.4
San Diego	0.7	0.7	13.1	1.8	1.8	12.8	4.4	58.4	6.2	1.5	98.6
San Francisco Bay	0.7	2.2	21.5	12.8	4.4	27.0	8.8	109.5	17.2	0.0	204.4
San Joaquin Valley	3.3	2.6	11.0	25.6	5.5	30.3	36.1	69.4	12.8	1.5	197.1
South Central Coast	0.7	0.4	4.7	5.5	0.4	9.1	2.9	23.7	3.7	7.7	58.4
South Coast	3.7	1.1	73.0	23.0	7.7	65.7	18.3	226.3	36.5	2.6	474.5

* Geogenic and wildfires
** Totals reflect emissions in each category before rounding
na not available

Table II-4. Annual emissions of PM_{10} for air basins in California in 1995. (Source: CARB Emissions Website 1999a). Units are 1000 tons/year.

	Stationary Sources					Area-Wide Sources		Mobile Sources		Natural Sources	
Air Basin	Fuel Combustion	Waste Disposal	Cleaning and Surface Coatings	Petroleum Production and Marketing	Industrial Processes	Solvent Evaporation	Misc. Processes*	On-Road Motor Vehicles	Other Mobile Sources	Natural (Non-anthropogenic) Sources**	**Total****
Great Basin	0.0	0.0	na	na	0.4	na	18.6	0.0	0.0	0.7	19.7
Lake County	0.0	na	0.0	na	0.0	na	3.7	0.0	0.0	0.7	4.7
Lake Tahoe	0.0	0.0	na	na	0.0	na	1.5	0.0	0.0	0.0	1.8
Mojave Desert	0.7	0.0	0.0	0.0	33.2	0.0	40.2	1.1	1.5	0.4	76.7
Mountain Counties	0.7	0.0	0.0	na	1.5	na	35.0	0.4	0.4	2.6	40.2
North Central Coast	0.4	na	na	0.0	0.7	na	20.4	0.4	0.4	1.5	24.1
Northeast Plateau	0.4	0.0	na	na	0.4	na	25.9	0.4	0.4	2.9	29.6
Sacramento Valley	0.7	0.0	0.0	0.0	4.0	0.0	80.3	2.2	1.1	2.9	91.3
Salton Sea	0.4	0.0	0.0	na	1.1	0.0	94.9	0.7	0.4	0.7	98.6
San Diego	0.4	1.1	na	na	1.1	0.0	32.1	1.8	1.1	3.7	40.2
San Francisco Bay	1.5	0.0	0.0	0.4	5.1	na	43.8	3.3	2.9	0.0	58.4
San Joaquin Valley	3.7	0.0	0.0	0.0	6.6	na	146.0	2.9	1.5	3.7	160.6
South Central Coast	0.4	0.0	0.0	0.0	0.7	0.0	22.6	0.7	0.7	2.9	28.1
South Coast	2.6	1.1	0.4	0.7	4.4	0.0	135.1	10.2	6.2	7.3	167.9

* Major categories include road dust, fugitive windblown dust, construction, farming, residential fuel combustion and waste burning.
** Wildfires
*** Totals reflect emissions in each category before rounding
na not available

Figure II-2. California air basins. Arrows indicate where pollutant transport is known to occur, but do not represent precise pathways or magnitudes. (Source: CARB 1996a).

Emissions of NO_x from motor vehicles and stationary sources have played major roles in a number of significant air pollution problems in California. For example, NO_x emissions contribute to increased concentration of 3 of the 4 pollutants for which national air quality standards have been exceeded in California: NO_2, ozone, and PM_{10} (CARB 1985).

b. Emission Levels Close to Class I Areas

Emissions from sources located within the counties that surround Class I areas are of particular relevance to air quality within Class I areas. County-level emissions of NO_x, ROG, PM_{10}, CO, and SO_x are shown in Table II-5 for any county within 140 km of each Class I area, as long as that county is in the same air basin as the Class I area. For Class I areas that straddle two air basins (such as YOSE), the table includes all counties within 140 km in both air basins. In areas where transport has been documented across air basins, emissions from counties in the air basin where emissions originate have also been included (for example, emissions from the SoCAB have been listed under JOTR because transport is known to occur between the South Coast and Mojave Desert air basins) (see also Section II-4). In some cases, only a portion of a county is in the air basin of a particular Class I area. In such cases, only the portion of the county in the same air basin as the Class I area is listed, unless transport occurs between the two air basins (for example, in YOSE, which is in the San Joaquin and Mountain Counties air basins, only the portions of El Dorado and Placer counties that are within Mountain Counties air basin are listed; the portions of El Dorado and Placer counties within the Lake Tahoe air basin are not included).

Class I areas in or near the SoCAB are potentially exposed to the highest levels of emissions. These areas include JOTR, and USDA Forest Service-managed Agua Tibia, San Gorgonio, San Jacinto, Cucamonga, and San Gabriel Wilderness areas. Class I areas adjacent to the central and southern San Joaquin Valley are also potentially exposed to high levels of emissions. These areas include YOSE, SEKI, and USDA Forest Service-managed Emigrant, Mokelumne, Ansel Adams, Domeland, John Muir and Kaiser Wilderness areas. Areas in the desert that can be exposed to high emissions from surrounding counties JOTR. Other Class I areas that are located within 140 km of areas of high emissions include PINN and PORE (Point Reyes is typically upwind of pollution sources).

Table II-5. 1995 Emissions from counties within 140 km of Class I Areas. Source: CARB Almanac 1999b; SO_x from CARB Emissions Website 1999a. Units are 1000 tons/year.

Class I Area and County	NO_x	ROG*	PM_{10}	CO	SO_x
Agua Tibia W[3]					
San Diego	86.9	101.1	36.5	624.9	4.0
San Bernardino [1] (So. Coast)	52.9	45.3	24.8	249.3	1.5
Riverside[1] (So. Coast)	53.7	45.3	55.5	280.0	1.5
Los Angeles [1] (So. Coast)	264.6	268.6	77.0	1565.5	22.6
Orange	71.9	90.2	24.1	537.3	1.8
Ansel Adams W[3]					
Alpine	0.4	0.7	0.7	2.9	0.0

Table II-5. Continued.

Class I Area and County	NO_x	ROG*	PM_{10}	CO	SO_x
Mono[2]	1.1	2.2	8.4	14.2	0.0
Inyo[2]	2.2	1.8	9.5	15.7	0.4
Madera	11.3	7.7	8.8	41.2	0.4
Tulare	16.8	18.3	19.3	113.9	0.4
Caribou W[3]					
Plumas	2.9	5.8	6.6	48.2	0.4
Sierra	0.7	1.8	4.0	13.5	0.0
Nevada	4.0	5.5	6.2	40.5	0.4
Lassen	3.7	4.7	8.4	30.7	0.4
Modoc	1.8	1.1	6.6	9.1	0.0
Siskiyou	7.7	9.1	11.7	109.1	0.4
Cucamonga W					
San Bernardino [1] (So. Coast)	52.9	45.3	24.8	249.3	1.5
Riverside [1] (So. Coast)	53.7	45.3	55.5	280.0	1.5
Los Angeles [1] (So. Coast)	264.6	268.6	77.0	1565.5	22.6
Orange	71.9	90.2	24.1	537.3	1.8
San Bernardino [1,4] (Desert)	59.5	19.3	85.4	118.3	4.4
Desolation W[3]					
El Dorado [1] (Lake Tahoe)	0.7	2.6	1.1	28.8	0.0
Placer [1] (Lake Tahoe)	0.4	1.1	0.4	9.9	0.0
El Dorado [1] (Mountain)	5.5	6.9	5.5	54.8	0.4
Placer [1] (Mountain)	1.5	1.5	2.6	9.9	0.0
Sierra	0.7	1.8	4.0	13.5	0.0
Nevada	4.0	5.5	6.2	40.5	0.4
Amador	3.3	3.7	3.3	21.2	0.4
Calaveras	1.5	3.7	3.7	23.4	0.0
Tuolumne	2.9	4.7	3.7	34.3	0.4
Mariposa	0.7	1.8	2.6	10.6	0.0
Dome Land W					
Kern[1] (San Joaquin)	60.2	50.0	24.8	186.5	2.9
Kings	9.5	8.0	13.5	36.5	0.4
Tulare	16.8	18.3	19.3	113.9	0.4
Fresno	39.8	39.4	45.3	205.1	3.7
Emigrant W					
Tuolumne	2.9	4.7	3.7	34.3	0.4
Mariposa	0.7	1.8	2.6	10.6	0.0
Amador	3.3	3.7	3.3	21.2	0.4
Calaveras	1.5	3.7	3.7	23.4	0.0

Table II-5. Continued.					
Class I Area and County	NO_x	ROG*	PM_{10}	CO	SO_x
El Dorado [1] (Mountain)	5.5	6.9	5.5	54.8	0.4
Placer [1] (Mountain)	1.5	1.5	2.6	9.9	0.0
Merced [4]	15.3	10.2	17.5	75.9	0.7
San Joaquin [4]	26.6	28.1	16.4	143.1	1.8
Stanislaus [4]	17.2	16.8	14.2	102.9	1.1
Madera [4]	11.3	7.7	8.8	41.2	0.4
Hoover W					
Alpine	0.4	0.7	0.7	2.9	0.0
Mono [2]	1.1	2.2	8.4	14.2	0.0
Inyo [2]	2.2	1.8	9.5	15.7	0.4
John Muir W[3]					
Mono [2]	1.1	2.2	8.4	14.2	0.0
Inyo [2]	2.2	1.8	9.5	15.7	0.4
Merced	15.3	10.2	17.5	75.9	0.7
Madera	11.3	7.7	8.8	41.2	0.4
Fresno	39.8	39.4	45.3	205.1	3.7
Tulare	16.8	18.3	19.3	113.9	0.4
Kings	9.5	8.0	13.5	36.5	0.4
Joshua Tree NP					
Riverside [1] (Desert)	1.1	0.7	1.1	5.5	0.0
Riverside [1,4] (Salton Sea)	13.9	10.6	14.2	71.5	0.4
San Bernardino [1] (Desert)	59.5	19.3	85.4	118.3	4.4
San Bernardino [1,4] (S. Coast)	52.9	45.3	24.8	249.3	1.5
Riverside[1,4] (So. Coast)	53.7	45.3	55.5	280.0	1.5
Los Angeles [1,4] (So. Coast)	264.6	268.6	77.0	1565.5	22.6
Orange [4]	71.9	90.2	24.1	537.3	1.8
Kaiser W					
Madera	11.3	7.7	8.8	41.2	0.4
Fresno	39.8	39.4	45.3	205.1	3.7
Kings	9.5	8.0	13.5	36.5	0.4
Merced	15.3	10.2	17.5	75.9	0.7
Tulare	16.8	18.3	19.3	113.9	0.4
Kings Canyon NP					
Merced	15.3	10.2	17.5	75.9	0.7
Madera	11.3	7.7	8.8	41.2	0.4
Fresno	39.8	39.4	45.3	205.1	3.7
Tulare	16.8	18.3	19.3	113.9	0.4
Kern [1] (San Joaquin)	60.2	50.0	24.8	186.5	2.9

Table II-5. Continued.					
Class I Area and County	NO_x	ROG*	PM_{10}	CO	SO_x
Kings	9.5	8.0	13.5	36.5	0.4
Lassen Volcanic NP[3]					
Shasta	12.0	11.3	10.6	91.3	0.7
Tehama	6.2	4.0	5.1	27.7	0.4
Butte	7.7	10.6	10.2	60.2	0.4
Sutter	4.0	5.5	6.9	35.8	0.0
Yuba	3.3	4.0	3.3	23.4	0.4
Glenn	3.7	4.0	6.6	25.9	0.4
Lassen	3.7	4.7	8.4	30.7	0.4
Modoc	1.8	1.1	6.6	9.1	0.0
Siskiyou	7.7	9.1	11.7	109.1	0.4
Lava Beds NM					
Lassen	3.7	4.7	8.4	30.7	0.4
Modoc	1.8	1.1	6.6	9.1	0.0
Siskiyou	7.7	9.1	11.7	109.1	0.4
Marble Mountain W					
Modoc	1.8	1.1	6.6	9.1	0.0
Siskiyou	7.7	9.1	11.7	109.1	0.4
Mokelumne W[3]					
Amador	3.3	3.7	3.3	21.2	0.4
Calaveras	1.5	3.7	3.7	23.4	0.0
El Dorado [1] (Mountain)	5.5	6.9	5.5	54.8	0.4
Mariposa	0.7	1.8	2.6	10.6	0.0
Placer [1] (Mountain)	1.5	1.5	2.6	9.9	0.0
Nevada	4.0	5.5	6.2	40.5	0.4
Sierra	0.7	1.8	4.0	13.5	0.0
Tuolumne	2.9	4.7	3.7	34.3	0.4
Alpine	0.4	0.7	0.7	2.9	0.0
Mono [2]	1.1	2.2	8.4	14.2	0.0
San Joaquin [4]	26.6	28.1	16.4	143.1	1.8
Stanislaus [4]	17.2	16.8	14.2	102.9	1.1
Pinnacles NM					
Monterey	20.8	19.7	14.2	102.9	1.1
San Benito	2.6	1.8	4.4	11.3	0.0
Santa Cruz	6.9	10.6	4.7	51.8	0.4
San Mateo [4]	22.6	23.0	8.4	165.3	0.7
Santa Clara [4]	46.0	48.5	17.9	333.2	1.5
Alameda [4]	43.1	43.1	13.1	283.2	3.3

Table II-5. Continued.

Class I Area and County	NO_x	ROG*	PM_{10}	CO	SO_x
Point Reyes NS					
Alameda	43.1	43.1	13.1	283.2	3.3
Contra Costa	50.4	36.1	12.0	219.0	14.6
Marin	7.3	8.8	3.7	64.6	0.4
Napa	3.3	4.4	1.5	27.4	0.0
San Francisco	15.7	18.3	6.2	110.2	2.9
San Mateo	22.6	23.0	8.4	165.3	0.7
Santa Clara	46.0	48.5	17.9	333.2	1.5
Solano [1] (SF Bay Area)	12.4	10.6	3.7	66.4	6.6
Sonoma [1] (SF Bay Area)	10.2	11.3	4.4	84.3	0.4
Redwood NP					
Del Norte	1.5	2.2	3.3	36.5	0.0
Humboldt	8.4	8.8	7.7	72.3	1.1
Trinity	1.1	2.2	6.2	23.0	0.0
San Gabriel W					
Los Angeles [1,4] (Desert)	6.9	6.9	9.5	34.7	0.4
San Bernardino [1] (So. Coast)	52.9	45.3	24.8	249.3	1.5
Riverside [1] (So. Coast)	53.7	45.3	55.5	280.0	1.5
Los Angeles [1] (So. Coast)	264.6	268.6	77.0	1565.5	22.6
Orange	71.9	90.2	24.1	537.3	1.8
San Gorgonio W					
San Bernardino [1] (So. Coast)	52.9	45.3	24.8	249.3	1.5
Riverside [1] (So. Coast)	53.7	45.3	55.5	280.0	1.5
Los Angeles [1] (So. Coast)	264.6	268.6	77.0	1565.5	22.6
Orange	71.9	90.2	24.1	537.3	1.8
San Jacinto W					
Riverside [1,4] (Salton Sea)	13.9	10.6	14.2	71.5	0.4
Imperial [4]	14.2	10.6	55.5	57.7	0.7
San Bernardino [1] (So. Coast)	52.9	45.3	24.8	249.3	1.5
Riverside [1] (So. Coast)	53.7	45.3	55.5	280.0	1.5
Los Angeles [1] (So. Coast)	264.6	268.6	77.0	1565.5	22.6
Orange	71.9	90.2	24.1	537.3	1.8
San Rafael W					
San Luis Obispo	13.1	11.7	10.2	71.2	5.5
Santa Barbara	15.7	19.0	6.9	84.0	0.7
Ventura	21.5	30.7	8.8	149.7	1.1
Kern [1,4] (San Joaquin)	60.2	50.0	24.8	186.5	2.9
Sequoia NP					

Table II-5. Continued.					
Class I Area and County	NO_x	ROG*	PM_{10}	CO	SO_x
Madera	11.3	7.7	8.8	41.2	0.4
Fresno	39.8	39.4	45.3	205.1	3.7
Tulare	16.8	18.3	19.3	113.9	0.4
Kern [1] (San Joaquin)	60.2	50.0	24.8	186.5	2.9
Kings	9.5	8.0	13.5	36.5	0.4
South Warner W					
Lassen	3.7	4.7	8.4	30.7	0.4
Modoc	1.8	1.1	6.6	9.1	0.0
Siskiyou	7.7	9.1	11.7	109.1	0.4
Thousand Lakes W					
Shasta	12.0	11.3	10.6	91.3	0.7
Tehama	6.2	4.0	5.1	27.7	0.4
Butte	7.7	10.6	10.2	60.2	0.4
Glenn	3.7	4.0	6.6	25.9	0.4
Ventana W					
Monterey	20.8	19.7	14.2	102.9	1.1
San Benito	2.6	1.8	4.4	11.3	0.0
Santa Cruz	6.9	10.6	4.7	51.8	0.4
San Mateo [4]	22.6	23.0	8.4	165.3	0.7
Santa Clara [4]	46.0	48.5	17.9	333.2	1.5
Yolla Bolly W[3]					
Humboldt	8.4	8.8	7.7	72.3	1.1
Mendocino	5.8	6.6	6.9	48.2	0.0
Trinity	1.1	2.2	6.2	23.0	0.0
Butte	7.7	10.6	10.2	60.2	4.0
Colusa	4.4	4.0	8.8	32.5	0.0
Glenn	3.7	4.0	6.6	25.9	0.4
Shasta	12.0	11.3	10.6	91.3	0.7
Tehama	6.2	4.0	5.1	27.7	0.4
Yosemite NP[3]					
Amador	3.3	3.7	3.3	21.2	0.4
Calaveras	1.5	3.7	3.7	23.4	0.0
El Dorado [1] (Mountain)	5.5	6.9	5.5	54.8	0.4
Placer [1] (Mountain)	1.5	1.5	2.6	9.9	0.0
Mariposa	0.7	1.8	2.6	10.6	0.0
Madera	11.3	7.7	8.8	41.2	0.4
Tuolumne	2.9	4.7	3.7	34.3	0.4
Fresno	39.8	39.4	45.3	205.1	3.7

Table II-5. Continued.

Class I Area and County	NO_x	ROG*	PM_{10}	CO	SO_x
Merced	15.3	10.2	17.5	75.9	0.7
San Joaquin	26.6	28.1	16.4	143.1	1.8
Stanislaus	17.2	16.8	14.2	102.9	1.1
Tulare	16.8	18.3	19.3	113.9	0.4
Kings	9.5	8.0	13.5	36.5	0.4

Notes for Table II-5
* Reactive Organic Gases
[1] Portion of the county in the air basin
[2] Does not include emissions from Owens and Mono Lake beds
[3] Area includes two air basins; counties in both basins that are within 140 km are included
[4] County (or portion) is in adjacent air basin in an area where transport between air basins occurs

c. Emissions From Major Point Sources

Large point sources of emissions potentially affect local air quality by virtue of their high emissions release rates. In some cases, tall stacks are employed to disperse emissions, potentially affecting transport directions and ranges. Actual situations must be evaluated individually, as emission rates, stack heights, local winds, and California's complex terrain all affect dispersion and potential impact on downwind areas. Large point sources (over 100 tons/year) of SO_x, NO_x, PM_{10}, and ROG emissions are located throughout California, with the largest number found in the Bay Area, SoCAB, Kern County (San Joaquin Valley), and Sacramento area (Figures II-3 to II-6). Statewide, mobile sources are the single largest source category of ROG and NO_x emissions, whereas area sources emit more PM than any other source category (Tables II-2 through II-4).

d. Pesticides

Besides criteria pollutants (ozone, PM, NO_2, SO_2, and CO), a large number of chemical compounds can be found in ambient air monitoring samples at concentrations that are of concern from either a health-effects or an environmental perspective. Among these compounds are a variety of types of pesticides, which are used extensively in California's heavily-farmed San Joaquin and Sacramento valleys. Pesticides vary as to their levels of volatility. Potential impacts are also affected by the degree to which a particular pesticide breaks down in the environment, and the levels of toxicity of the breakdown products. The release and transport of pesticides from their point of application to downwind areas involves numerous other important physical processes. An overview of pesticide usage in non-attainment air basins that contain Class I areas is shown in Table II-6. The amount of pesticide use, particularly in the San Joaquin Valley (SJV), suggests that further study is warranted. As discussed in Section II-4, a known pollution transport route into the Sierra (where several Class I areas are located) is from the eastern side of the SJV. Some research indicates that pesticides used in the SJV may be transported into the Sierra (Zabik and Seiber 1993).

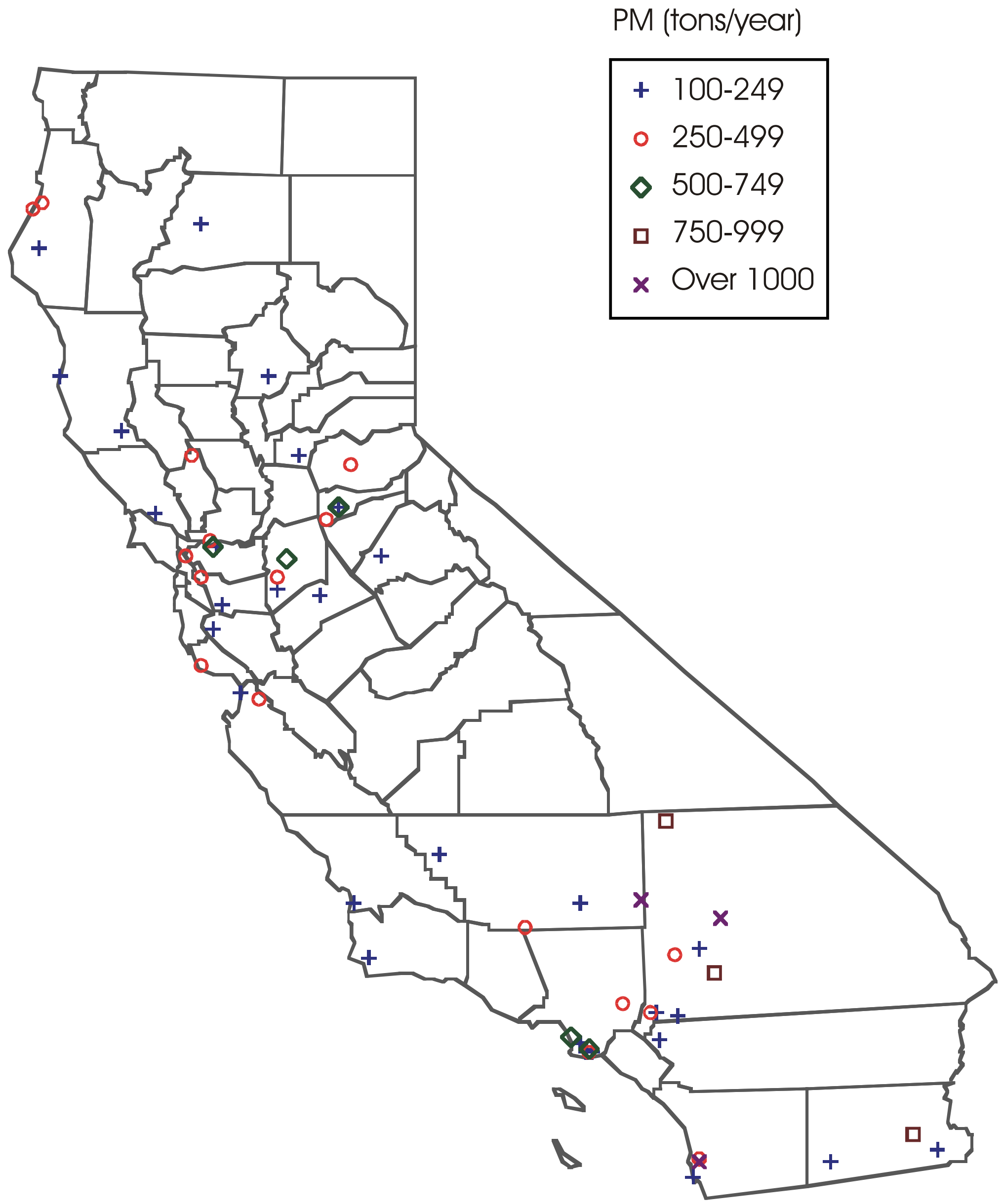


Figure II-3. 1996 PM_{10} point sources for California (source: Andrew Alexis, CARB, personal communication).

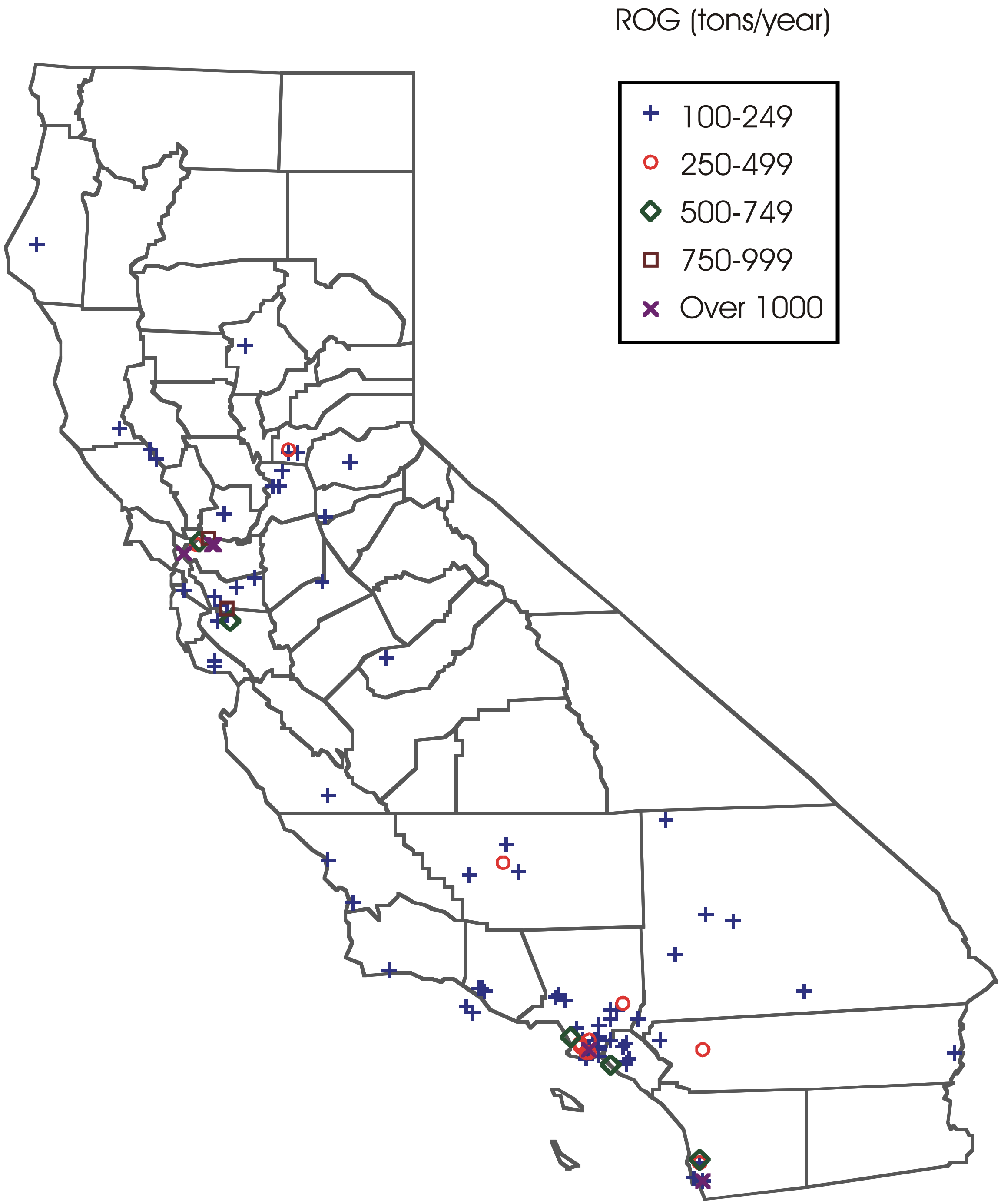


Figure II-4. 1996 Reactive Organic Gas (ROG) point sources for California (source: Andrew Alexis, CARB, personal communication).

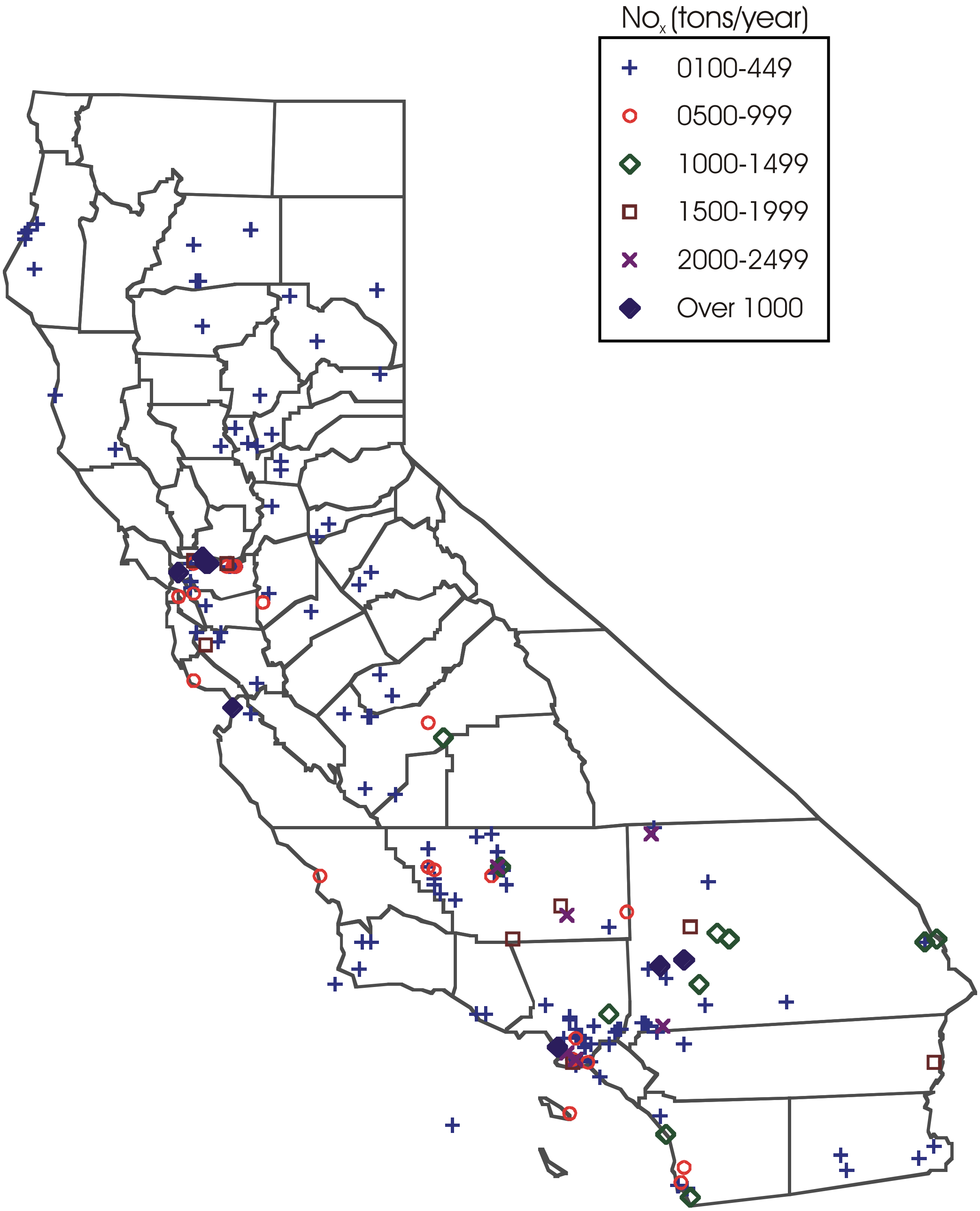


Figure II-5. 1996 NO$_x$ point sources for California (source: Andrew Alexis, CARB, personal communication).

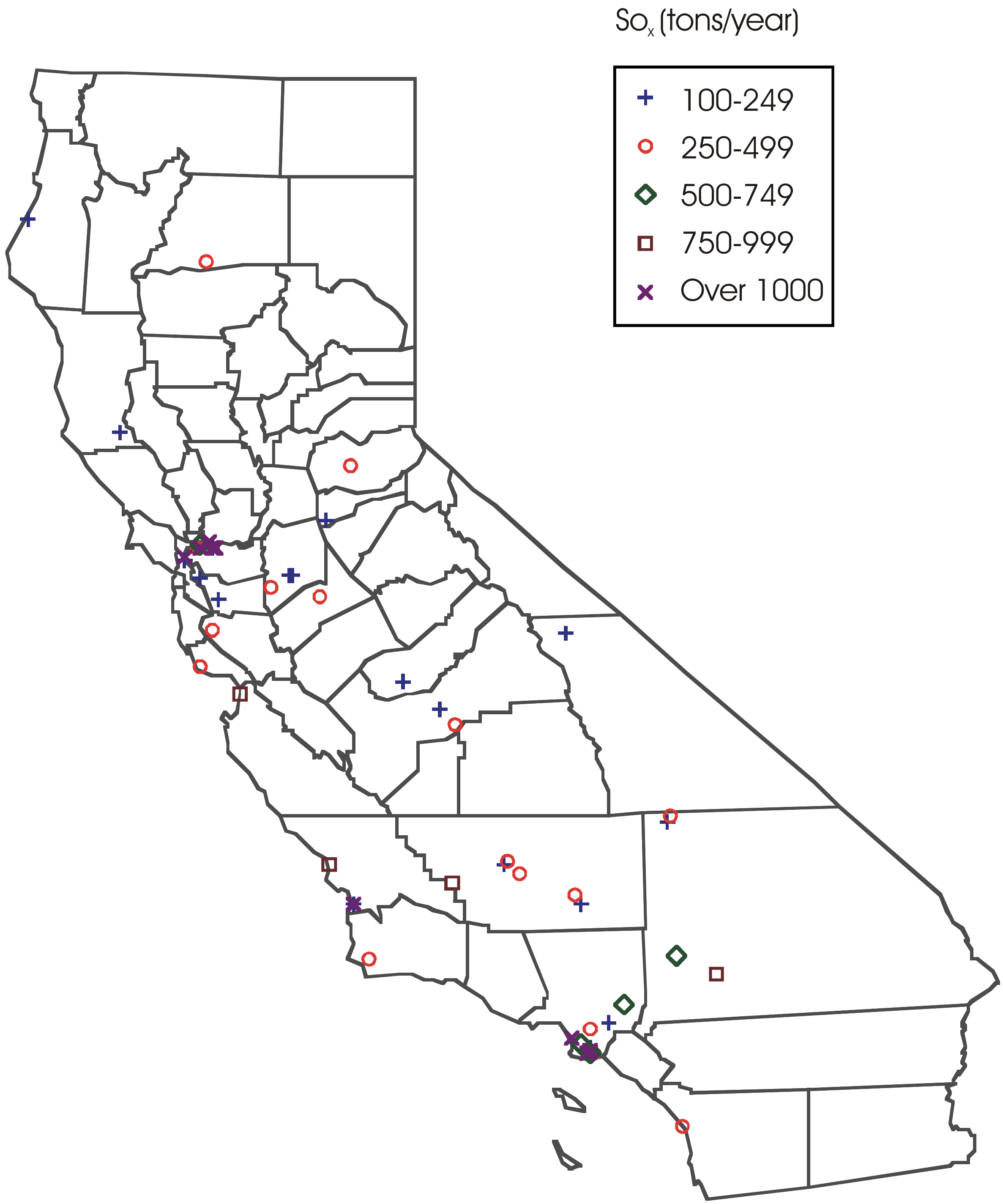


Figure II-6. 1996 SO_x point sources for California (source: Andrew Alexis, CARB, personal communication).

Table II-6. Pesticide applications in three air basins, showing Class I/II areas in each air basin,1994. Data are from non-attainment areas only; table does not include transport from adjacent air basins. (Source: CDPR website, 2000).

Air Basin	Counties	Pesticide VOC (1 million lbs. applied)	Emissions Potential (1 million lbs.)	Class I/II Areas in the Air Basin
San Joaquin Valley	Fresno	55.3	7.0	Sequoia/Kings Canyon NP, John Muir W, Dome Land W, Ansel Adams W, Yosemite NP, Kaiser W, Devils Postpile NM*, Manzanar NHS*
	Kern	33.5	6.1	
	Kings	11.4	2.5	
	Madera	12.6	1.0	
	Merced	13.3	3.2	
	San Joaquin	16.0	3.1	
	Stanislaus	9.7	2.4	
	Tulare	23.4	4.4	
South Coast	Los Angeles	2.9	1.0	San Gabriel W, Cucamonga W, San Gorgonio W, San Jacinto W, Agua Tibia W, Santa Monica NRA*
	Orange	2.9	1.0	
	Riverside	2.0	0.3	
	San Bernardino	0.8	0.2	
Mojave Desert	San Bernardino	.3	0.1	Death Valley NP*, Joshua Tree NP, Mojave NPr*
	Riverside	4.3	0.7	
	Los Angeles	0.2	0.1	

* NPS Class II

2. Ambient Monitoring Data

Gas- and particulate-phase ambient air concentrations of pollutants, as well as deposition and visibility, are monitored at several sites in or near Class I areas. Organizations or networks that are monitoring, or have monitored, air quality on a long-term basis include the California Air Resources Board networks (CARB 1995a, 1997a, and 1998a), the Clean Air Status and Trends Network (CASTNet) operated jointly by USEPA and the NPS, local Air Pollution Control Districts (APCD) (EPA Airs website 1999), the Integrated Monitoring Study of 1995 (IMS95) (P.A. Solomon et al. 1997), Interagency Monitoring of Protected Visual Environments (IMPROVE) (NPS Visibility Website 1999), the National Atmospheric Deposition Program (NADP) (NADP/NTN Website 1999), the California Acid Deposition Monitoring Program (CADMP) (Watson et al. 1991, Blanchard and Tonnessen 1993), and the University of California, Santa Barbara (UCSB) (Melack et al. 1995). Tables II-7 through II-9 list the information available from monitoring networks with sites that are within 50 km of Class I and NPS Class II areas. Generally, more extensive monitoring occurs in national parks than in

Table II-7. Air quality monitoring in California Class I and NPS Class II areas -- gas phase species. Offsite monitors are stations located within 50 km of area boundaries. Data are available from the organizations listed*.

Class I/II Area	Species and Networks				
	Ozone			SO_2	
	Hourly Onsite	Hourly Offsite	Passive Onsite	Onsite	Offsite
Agua Tibia W		ARB			
Ansel Adams W		ARB			
Cabrillo NM #		ARB			
Caribou W		ARB			
Channel Islands NP #	ARB		NPS		ARB
Cucamonga W		ARB			
Death Valley NP #	CASTNet‖			NPS	
Desolation W		ARB			
Devils Postpile NM #		ARB			
Dome Land W		ARB			
Emigrant W		ARB			
Eugene O'Neill NHS #		ARB			
Fort Point NHS #		ARB			
Golden Gate NRA #		ARB			
Hoover W		ARB			
John Muir NHS #		ARB			
John Muir W		ARB			
Joshua Tree NP	CASTNet‖ ARB		NPS**		
Kaiser W		ARB			
Kings Canyon NP		CASTNet	NPS		
Lassen Volcanic NP	CASTNet‖ ARB			NPS	
Lava Beds NM			NPS		
Manzanar NHS #					
Marble Mt. W		ARB			
Mojave NPr #					
Mokelumne W		ARB			
Muir Woods NM #		ARB			

Table II-7. Continued.

Class I/II Area	Species and Networks				
	Ozone			SO_2	
	Hourly Onsite	Hourly Offsite	Passive Onsite	Onsite	Offsite
Pinnacles NM	CASTNet‖ ARB			NPS	
Point Reyes NS	NPS^	ARB	NPS**	NPS	
Redwood NP	NPS^^	ARB		NPS	
San Francisco Maritime NHP #		ARB			
San Gabriel W		ARB			
San Gorgonio W		ARB			
San Jacinto W		ARB			
San Rafael W		ARB			
Santa Monica Mountains NRA #		ARB			
Sequoia NP	CASTNet‖		NPS	NPS	
South Warner W					
Thousand Lakes W		ARB			
Ventana W		ARB			
Whiskeytown-Shasta-Trinity NRA #		ARB			
Yolla Bolly W		ARB			
Yosemite NP	CASTNet‖		NPS	NPS	

* Contact agencies for data:
ARB (CD-ROM available from Planning and Technical Support Division, Air Quality Data Branch)
CASTNet (http://www.epa.gov/castnet/)
NPS (NPS Air Resources Division or http://www2.nature.nps.gov/ard/gas/index.htm;
SO_2 data a at http://capita.wustl.edu/CAPITA/DataSets/IMPROVE/)
NPS Class II Area
** New or soon to be installed site
‖ CASTNet ozone data commence in 1995 (1997 at Sequoia). For some sites, the ARB has earlier data.
^ closed 1993
^^ closed 1995

Table II-8. Air quality monitoring in California Class I and NPS Class II areas -- particulate species. Offsite monitors are stations located within 50 km of area boundaries^^. Data are available from the organizations listed**. Both IMPROVE and IMPROVE-protocol sites are listed as IMPROVE.

	Species and Networks					
	PM_{10}		$PM_{2.5}$		Visibility ^	
Class I/II Area	Onsite	Offsite	Onsite	Offsite	Onsite	Offsite
Agua Tibia W	IMPROVE*		IMPROVE*			
Ansel Adams W		IMPROVE ARB		IMPROVE ARB		IMPROVE
Cabrillo NM #		ARB				
Caribou W		IMPROVE*		IMPROVE*		
Channel Island NP#		ARB		ARB		
Cucamonga W		IMPROVE*		IMPROVE*		
Death Valley NP #	IMPROVE*		IMPROVE*	ARB		
Desolation W	IMPROVE*	ARB	IMPROVE*	ARB		IMPROVE ARB
Devils Postpile NM #		IMPROVE ARB		IMPROVE ARB		IMPROVE
Dome Land W	IMPROVE*	ARB	IMPROVE* ARB			
Emigrant W		IMPROVE		IMPROVE		IMPROVE
Eugene O'Neill NHS #		ARB				
Fort Point NHS #		ARB		ARB		
Golden Gate NRA #		ARB		ARB		
Hoover W	IMPROVE*	ARB	IMPROVE*	IMPROVE ARB		IMPROVE
John Muir NHS #		ARB				
John Muir W		IMPROVE ARB		IMPROVE ARB		IMPROVE
Joshua Tree NP	IMPROVE*	ARB	IMPROVE*			
Kaiser W	IMPROVE*		IMPROVE*			
Kings Canyon NP		IMPROVE ARB		IMPROVE ARB		
Lassen Volcanic NP	IMPROVE		IMPROVE	ARB		
Lava Beds NM	IMPROVE* ARB		IMPROVE*			
Manzanar NHS #		ARB				
Marble Mt. W	IMPROVE*	ARB	IMPROVE*			
Mojave NPr #						
Mokelumne W		IMPROVE* ARB		IMPROVE* ARB		

Table II-8. Continued.

Class I/II Area	Species and Networks					
	PM_{10}		$PM_{2.5}$		Visibility ^	
	Onsite	Offsite	Onsite	Offsite	Onsite	Offsite
Muir Woods NM #		ARB		ARB		
Pinnacles NM	IMPROVE ARB		IMPROVE ARB		IMPROVE	
Point Reyes NS	IMPROVE ARB	ARB	IMPROVE ARB			
Redwood NP	IMPROVE	ARB	IMPROVE	ARB		
San Gabriel W	IMPROVE*	ARB	IMPROVE*	ARB		ARB
San Gorgonio W	IMPROVE	ARB	IMPROVE		IMPROVE	
San Jacinto W		IMPROVE ARB		IMPROVE		IMPROVE
San Rafael W	IMPROVE*		IMPROVE*	IMPROVE		
San Francisco Maritime NHP #		ARB				
Santa Monica Mountains NRA#						
Sequoia NP	IMPROVE	ARB	IMPROVE			
South Warner W		IMPROVE* ARB		IMPROVE*		
Thousand Lakes W		IMPROVE* ARB		IMPROVE*		
Ventana W		ARB				IMPROVE
Whiskeytown-Shasta-Trinity NRA #		IMPROVE ARB		IMPROVE		
Yolla Bolly W		IMPROVE*		IMPROVE*		
Yosemite NP	IMPROVE ARB		IMPROVE ARB		IMPROVE	

** Contact agencies for data:
ARB (CD-ROM available from Planning and Technical Support Division, Air Quality Data Branch)
IMPROVE (http://vista.cira.colostate.edu/improve/)

^^ All visibility-protected Class I areas are included in clusters represented by individual monitoring sites. The monitoring sites representing South Warner and Yolla Bolly-Middle Eel wilderness areas are more than 50 km from them but are considered representative.

\# Class II Area

^ Visual range measured by transmissometer or light scattering measured by nephelometer

* New or soon to be installed site

Table II-9. Air quality monitoring in California Class I and NPS Class II areas -- wet and dry deposition. Offsite monitors are stations located within 50 km of area boundaries. Data are available from the organizations listed##. The ARB data base includes data from the ARB's CADMP network and ARB-sponsored studies conducted by the University of California at Santa Barbara.

	Measurements and Networks			
	Wet Deposition		Dry Deposition	
Class I/II Area	Onsite	Offsite	Onsite	Offsite
Agua Tibia W		ARB^^		
Ansel Adams W		NADP, ARB^^		CASTNet, ARB
Cabrillo NM #				
Caribou W				
Channel Islands NP #	NADP**	ARB^^		ARB
Cucamonga W		NADP, ARB		ARB
Death Valley NP #	NADP*		CASTNet*	
Desolation W	ARB^			
Devils Postpile NM #		NADP, ARB		CASTNet, ARB
Dome Land W		ARB		
Emigrant W		NADP, ARB		CASTNet, ARB
Eugene O'Neill NHS #				
Fort Point NHS #				
Golden Gate NRA #				
Hoover W		NADP, ARB		CASTNet, ARB
John Muir NHS #				
John Muir W		NADP, ARB^		CASTNet, ARB
Joshua Tree NP	NADP*		CASTNet	
Kaiser W		ARB^		
Kings Canyon NP	ARB^	NADP		CASTNet, ARB
Lassen Volcanic NP	NADP*		CASTNet	
Lava Beds NM				
Manzanar NHS #				
Marble Mt. W		NADP, ARB^^		
Mojave NPr #				
Mokelumne W		ARB		
Muir Woods NM #				

Table II-9. Continued.

Class I/II Area	Measurements and Networks			
	Wet Deposition		Dry Deposition	
	Onsite	Offsite	Onsite	Offsite
Pinnacles NM	NADP*	ARB^^	CASTNet	
Point Reyes NS		ARB^^		
Redwood NP		ARB		ARB^^
San Francisco Maritime NHS #				
San Gabriel W		NADP, ARB		ARB
San Gorgonio W		ARB^^		
San Jacinto W				
San Rafael W		ARB^^		ARB^^
Santa Monica Mountains NRA#				
Sequoia NP	NADP, ARB		CASTNet, ARB^^	
South Warner W				
Thousand Lakes W				
Ventana W		ARB		
Whiskeytown-Shasta-Trinity NRA #				
Yolla Bolly W				
Yosemite NP	NADP, ARB		CASTNet, ARB^^	

Contact agencies for data:
ARB (CD-ROM available from Planning and Technical Support Division, Air Quality Data Branch)
CASTNet (http://www.epa.gov/castnet/)
NADP (http://nadp.sws.uiuc.edu/)
Class II Area
* New or soon to be opened site
** Operated 1980-82
^ UCSB sites in operation 1990-1993
^^ Closed in 1994 or earlier

wilderness areas, though there are notable gaps in monitoring even within the national parks. Some wilderness areas are further than 50 km from any existing monitoring location. Additional air-quality data have been obtained during special research studies, as discussed later.

a. Wet and Dry Deposition

Wetfall chemistry and wet deposition in or near REDW, YOSE, and SEKI are shown in Tables II-10 and II-11 (Blanchard et al. 1996; Dwight Oda and Brent Takemoto, CARB,1999, personal communication). Concentrations of species and deposition of S and N are reported for all available years for sites in or close to (within 10 km) these parks.

Annual-average hydrogen ion concentration in precipitation ranges from 2.1 to 5.8 µeq/L (pH 5.68 to 5.24) (Table II-10). These values are not substantially more acidic than the expected value of water in equilibrium with atmospheric CO_2 (pH ~ 5.7). However, individual storm events or meltwater may exhibit higher acidity, posing questions about potential transient impacts (see later discussions).

Except for coastal locations, which are affected by sea-salt SO_4^{2-}, S deposition averages less than 1 kg/ha/yr (equivalently, 3 kg/ha/yr as SO_4^{2-}) in all parks where monitoring has taken place (Table II-11). Multi-year NO_3^- and NH_4^+ wet deposition rates were each in the range of 0.5 to 1.8 kg/ha/yr as N, yielding total wet N deposition rates of 0.8 to 3.0 kg/ha/yr (Table II-11). For individual years, total wet N deposition rates were as high as 4.4 and 5.7 kg/ha/yr in Sequoia National Park (Table II-11).

Blanchard et al. (1996) estimated annual and 10-year wet deposition rates throughout California by interpolating the observations from all NADP/NTN, CADMP, and special-studies monitoring locations for the period 1985 through 1994; they also estimated interpolation uncertainties. Upper-bound deposition values for unmonitored areas were estimated as the sum of the interpolated values plus twice the uncertainty (i.e., mean plus two standard deviations). The results indicated that the 10-year mean wet total N deposition rates were less than 3 (+/- 3, at 2 sigma) kg/ha/yr (as N) throughout the state. Wet S deposition was less than 1.3 (+/- 1, at 2 sigma) kg/ha/yr (as S) throughout the state.

The CADMP co-located wet and dry deposition samplers at ten locations in California (Blanchard et al. 1996). The most striking difference between urban and nonurban locations was the rate of dry deposition of oxidized N species: in the seven urban locations, dry deposition of oxidized N (HNO_3, NO_2, and aerosol NO_3^-) exceeded wet deposition of NO_3^- by factors of ten to thirty (Blanchard et al. 1996). In contrast, at the three nonurban locations (YOSE, SEKI, and Gasquet, located near REDW), mean dry deposition rates of oxidized N were in the range of 0.7 to 0.8 kg/ha/yr (as N), roughly comparable to flux rates of 0.5 to 1.2 kg/ha/yr (as N) wet NO_3^- deposition. Similarly, at these three nonurban sites, dry NH_3 plus aerosol NH_4^+ deposition rates were approximately 0.4 to 1.7 times the rate of wet NH_4^+ deposition and dry S (SO_2 plus aerosol SO_4^{2-}) deposition rates were in the range of 0.7 to 1.3 times the rate of wet SO_4^{2-} deposition (Blanchard et al. 1996). Thus, proximity to higher levels of NO_x emissions is expected to be a key determinant of local N fluxes. Research studies have shown that N deposition rates in the San Bernardino and San Gabriel Mountains could approach 30 to 35 kg/ha/yr (as N) (Bytnerowicz and Fenn, 1996; Fenn and Poth, 1998, 1999; Fenn et al., 1998; Grulke et al, 1998). The results of Chorover et al. (1994) would suggest that nitrogen deposition on the western slope of the southern Sierra Nevada could approach 10 kg/ha/yr (as N) if throughfall N flux was indicative of N loading, but this is unlikely to be the case. The multi-year CADMP mean wet and dry deposition estimates for sites at Yosemite and Sequoia National Parks were closer to ~5

Table II-10. Wetfall chemistry at CADMP, NADP and UCSB sites in or near national parks in California. Units are μeq/L, except precipitation (cm). Source: Blanchard et al. 1996; Dwight Oda and Brent Takemoto, CARB, 1999; Melack et al., 1995.

CLASS I Area	NAME	Water Year*	Prec	H^+	SO_4^{-2}	NH_4^+	NO_3^-	Ca^{+2}	Mg^{+2}	Na^+	K^+	Cl^-
		1985	181.1	4.1	9.3	1.8	1.6	7.7	11.5	46.2	1.7	55.8
Redwood NP	GASQUET**	1986	236.8	4.3	6.4	2.5	1.6	2.1	7.7	30.7	1.2	37.6
	CADMP	1987	185.4	5.1	7.2	2.9	1.7	5.3	8.3	36.1	0.9	43.2
	(117 m.)	1988	173.3	4.2	5.7	3.1	2.4	4.5	7	29.1	0.8	35.1
		1989	213.5	5.2	7.5	1.7	2.4	4.3	7.1	35.7	1.3	38.6
		1990	169.9	5.8	7.4	1.3	1.5	6.8	5.1	25.0	0.8	29.1
		1992	138.0	3.9	6.3	0.8	2.2	5.6	10.6	26.4	1.0	32.4
		1993	268.0	2.7	8.0	0.6	1.9	8.9	13.7	37.6	1.0	43.1
		1994	133.7	3.4	6.0	0.5	1.5	11.7	15.1	29.9	0.9	33.7
		1995	281.1	4.2	6.5	0.7	1.7	14.1	19.8	35.2	1.5	39.6
		1996	296.7	4.9	8.1	0.7	1.8	5.6	14.6	49.7	1.3	42.3
		1997	294.2	4.4	6.4	1.5	1.6	6.3	11.4	34.1	1.2	38.2
		1998	224.4	4.8	7.3	0.6	1.6	10.2	13.9	43.2	1.2	51.7
		Ave.	225.7	4.3	7.0	0.8	1.7	8.7	13.0	35.1	1.1	38.8
Sequoia/Kings	EMRLD LAKE	1991	112.2	3.5	2.5	3.9	3.5	2.1	0.7	1.6	1.6	1.5
NP	UCSB	1992	78.4	3.7	4.3	10.3	6.5	3.9	1.0	1.3	1.5	2.0
	(2824 m.)	1993	256.3	5.1	3.0	3.0	2.8	1.1	0.5	1.7	1.1	2.8
		Ave.	149.0	4.1	3.3	5.7	4.3	2.4	0.7	1.5	1.4	2.1
Sequoia/Kings	MINERAL	1991	64.3	5.1	6.1	14.6	10.5	3.4	1.4	2.3	1.4	2.2
NP	KING	1992	56.0	3.9	5.2	9.8	8.5	4.3	0.9	1.7	1.2	2.3
	UCSB	1993	108.4	3.9	3.4	4.4	3.9	2.6	0.5	1.2	1.1	1.6
	(2694 m.)	Ave.	76.2	4.3	4.9	9.6	7.6	3.4	0.9	1.7	1.2	2.0
Sequoia/Kings	ONION	1991	70.0	6.1	4.2	7.2	6.2	3.3	0.8	1.3	1.0	1.7
NP	VALLEY	1992	52.8	4.0	5.8	6.0	6.9	4.4	0.9	1.8	1.1	2.0
	UCSB	1993	86.2	5.6	3.1	4.1	4.0	2.8	0.7	1.0	1.1	2.9
	(2800 m.)	Ave.	69.7	5.2	4.3	5.8	5.7	3.5	0.8	1.4	1.1	2.2

Table II-10. Continued

CLASS I Area	NAME	Water Year*	Prec	H^+	SO_4^{-2}	NH_4^+	NO_3^-	Ca^{+2}	Mg^{+2}	Na^+	K^+	Cl^-
Sequoia/Kings NP	SEQ-GF/NADP (1902 m.)	1981	50.8	6.8	16.9	13.7	14.1	4.6	1.8	4.4	0.9	5.6
		1982	141	5	8.3	4.8	4.1	2.1	1.4	2.8	0.4	8.5
		1983	182.9	4	4.5	4.4	4.2	1.7	1.3	3.8	0.4	4.5
		1984	84.3	3.3	10.8	9	8.1	3.3	1.7	3.4	1	4.1
		1986	103.6	3.7	6.1	8.2	6.8	1.7	1.8	3.4	0.4	3.6
		1987	53	3.8	7.1	15.3	13.5	2.2	1.3	3.8	0.9	3.5
		1988	77	3.3	7.3	8.2	9.4	4	1.3	3.4	1.2	2.6
		1990	60.9	2.4	8.5	18.6	13.6	2.8	1.1	4.5	0.4	4.5
		1991	64.0	2.1	7.6	18.1	15.3	5.8	1.9	5.3	0.9	5.1
		1992	55.1	2.1	5.6	13.7	10.6	4.7	1.2	3.8	0.5	3.3
		1993	130.1	2.9	6.6	15.0	9.3	1.7	0.9	3.2	0.4	3.1
		1994	70.0	2.8	5.9	18.7	12.2	1.9	1.1	4.0	0.5	3.9
		Ave.	76.0	2.5	6.8	16.8	12.2	3.4	1.2	4.1	0.5	4.0
Sequoia/Kings NP	SEQUOIA ASH MT CADMP (548 m.)	1985	55.9	7.4	8.6	15.9	19.5	4.8	2	15.4	1.2	11.5
		1986	89.9	6	6.3	13.6	9.9	3.3	1.8	6.3	0.9	7.1
		1987	35.9	6.1	11.7	32.5	22.4	6.9	2.7	5.8	1.1	6.8
		1988	46.9	5	9.5	28.1	19.8	10	1.6	3.7	0.6	2.9
		1989	41.8	3.9	8.8	19	12.7	6	2.2	5.5	0.9	8.2
		1990	39.7	3.1	13.6	37.2	28.2	9.1	3.6	6.0	0.8	4.8
		1991	39.4	1.9	8.6	30.1	18.8	6.7	2.3	5.7	0.9	5.8
		1992	36.2	1.8	6.1	18.2	11.3	12.0	4.6	5.2	1.6	4.7
		1993	79.4	2.5	5.9	17.7	9.8	8.9	4.0	4.2	0.9	4.7
		1994	41.0	1.8	7.4	29.0	16.6	18.8	7.4	4.8	1.0	5.5

Table II-10. Continued.

CLASS I Area	NAME	Water Year*	Prec	H^+	SO_4^{-2}	NH_4^+	NO_3^-	Ca^{+2}	Mg^{+2}	Na^+	K^+	Cl^-
Sequoia/Kings NP	SEQUOIA ASH MT CADMP (548 m.)	1995	102.0	3.2	7.3	25.2	14.4	12.1	6.2	4.5	1.4	4.6
		1996	52.2	3.4	4.4	10.2	7.6	6.8	3.6	5.0	0.9	5.0
		1997	82.2	3.8	4.0	15.5	9.9	7.0	2.8	2.7	1.2	2.8
		1998	71.5	4.5	3.3	5.4	5.5	13.3	2.9	6.5	0.9	7.9
		Ave.	60.4	2.9	6.7	20.9	13.6	10.5	4.1	5.0	1.1	5.1
Sequoia/Kings NP	SEQUOIA GF CADMP (1890 m.)	1986	166.7	5.2	4.1	6.8	5.3	1.8	1	4.5	0.5	4.6
		1987	61.3	6.7	7.4	22	14.5	4.7	1.3	2.5	0.5	3.9
		1988	76.3	4.6	6.5	15.4	10.8	9.4	1.1	2.5	0.5	1.8
		1989	70.8	4.2	5.4	7.9	7.2	2.8	1.1	2.1	0.5	3.8
		1990	61.0	4.5	9.2	17.9	13.9	6.0	2.1	4.1	1.0	3.0
		1991	57.2	3.0	8.0	24.9	17.8	7.0	2.3	4.0	1.0	4.0
		1992	55.8	2.6	4.3	10.8	8.7	7.9	3.1	2.8	0.7	2.5
		1993	128.1	2.5	5.6	13.5	8.5	8.2	3.1	2.3	0.7	2.8
		1994	67.6	2.3	6.3	20.4	11.8	13.2	5.9	3.8	0.7	4.3
		1995	162.7	3.5	3.9	9.7	6.6	7.6	5.6	2.9	1.2	2.5
		1996	95.2	3.2	3.3	5.7	4.0	5.2	2.7	3.0	0.8	3.2
		1997	114.1	2.6	2.1	4.1	3.8	6.0	2.5	1.6	0.8	1.4
		1998	75.2	4.0	2.4	4.0	3.8	7.9	1.6	2.3	0.9	2.9
		Ave.	90.8	3.1	5.0	12.3	8.8	7.7	3.2	3.0	0.9	3.0
Yosemite NP	TIOGA PASS UCSB (2993 m.)	1991	97.0	5.5	3.6	5.1	5.2	3.3	0.7	2.4	1.4	1.8
		1992	81.1	5.4	3.8	5.3	5.3	2.9	0.8	1.2	1.1	1.4
		1993	224.0	4.8	3.0	3.2	3.0	2.4	0.6	1.1	1.0	1.3
		Ave.	134.0	5.2	3.5	4.5	4.5	2.9	0.7	1.5	1.2	1.5

Table II-10. Continued.

CLASS I Area	NAME	Water Year*	Prec	H^+	SO_4^{-2}	NH_4^+	NO_3^-	Ca^{+2}	Mg^{+2}	Na^+	K^+	Cl^-
Yosemite NP	YOSEMITE CADMP (1395 m.)	1985	79.7	4.3	7.1	10.7	10.2	8.4	2.9	6	1.6	6.4
		1987	68	8	9	13.8	13.6	3.4	1.5	4.3	1.3	5
		1988	65.5	5.3	4.9	9.4	9.8	4.9	2	2.7	1.1	3
		1989	51.1	5.2	5.1	7.8	7.9	3.2	1.2	2.5	0.5	4.5
		1991	45.5	3.6	5.6	9.7	8.4	3.9	1.9	3.9	0.7	4.6
		1992	57.0	3.4	5.9	9.9	9.0	5.8	2.8	2.6	0.7	2.5
		1993	85.0	3.2	5.0	6.5	6.6	9.4	3.2	2.7	1.0	3.0
		1994	45.1	4.0	5.3	12.4	11.1	17.5	3.6	2.2	0.7	2.4
		1995	143.0	4.4	4.1	6.6	6.0	18.3	3.9	3.4	1.3	3.1
		Ave.	75.1	3.7	5.2	9.0	8.2	11.0	3.1	3.0	0.9	3.1
Yosemite NP	YOSEMITE/ NADP (1408 m.)	1983	231.1	3.8	4	4	4	2	1.2	3.8	0.3	4.4
		1984	129.5	3.1	8.4	5.9	6.8	2.5	1.5	3.9	0.6	4.4
		1985	81.3	4.1	7.3	5.5	6.8	3.2	1.5	3.8	0.4	4
		1990	90.3	3.8	6.3	9.9	9.1	1.6	0.7	2.3	0.2	2.7
		1992	74.1	4.1	5.1	5.7	7.6	2.4	0.9	3.6	0.2	2.9
		Ave.	82.2	4.0	5.7	7.8	8.4	2.0	0.8	3.0	0.2	2.8

* The CARB water year is July 1 through June 30. For instance, water year 1995 is for July 1, 1994 through June 30, 1995.
** Located ~10 km. from Redwood NP

Table II-11. Wet deposition of S and N at CADMP, NADP and UCSB sites in or near National Parks in California (Source: Blanchard et al. 1996; Dwight Oda and Brent Takemoto, CARB 1999, personal communication). Units are kg/ha/yr.

Class I Area	Site	Water Year*	Sulfur	NO_3-N	NH_4-N	Total Inorganic N
Redwood NP	GASQUET** CADMP (117 m.)	1990	2.02	0.37	0.30	0.67
		1992	1.39	0.43	0.16	0.58
		1993	3.41	0.72	0.22	0.94
		1994	1.27	0.28	0.10	0.38
		1995	2.91	0.68	0.28	0.96
		1996	3.86	0.74	0.28	1.02
		1997	3.00	0.66	0.63	1.29
		1998	2.62	0.50	0.19	0.69
		Average	2.56	0.55	0.27	0.82
Sequoia/Kings NP	EMRLD LAKE UCSB (2824 m.)	1991	0.45	0.55	0.61	1.16
		1992	0.54	0.72	1.13	1.84
		1993	1.25	0.99	1.06	2.05
		Average	0.75	0.75	0.93	1.69
Sequoia/Kings NP	MINERAL KING UCSB (2694 m.)	1991	0.63	0.94	1.31	2.26
		1992	0.46	0.67	0.77	1.44
		1993	0.59	0.60	0.67	1.27
		Average	0.56	0.74	0.92	1.65
Sequoia/Kings NP	ONION VALLEY UCSB (2800 m.)	1991	0.47	0.61	0.71	1.32
		1992	0.49	0.51	0.45	0.95
		1993	0.42	0.48	0.50	0.98
		Average	0.46	0.53	0.55	1.08
Sequoia/Kings NP	SEQ-GF/NADP (1902 m.)	1990	0.82	1.16	1.59	2.75
		1991	0.78	1.37	1.62	2.99
		1992	0.49	0.82	1.06	1.88
		1993	1.38	1.70	2.72	4.42
		1994	0.66	1.20	1.83	3.02
		Average	0.83	1.25	1.76	3.01
Sequoia/Kings NP	SEQUOIA ASH MT CADMP (548 m.)	1990	0.86	1.56	2.06	3.63
		1991	0.54	1.04	1.66	2.69
		1992	0.35	0.58	0.92	1.50
		1993	0.76	1.09	1.97	3.06
		1994	0.49	0.95	1.66	2.62
		1995	1.19	2.05	3.60	5.65
		1996	0.36	0.56	0.75	1.31
		1997	0.53	1.14	1.78	2.92
		1998	0.38	0.55	0.54	1.10
		Average	0.61	1.06	1.66	2.72

Table II-11. Continued.

Class I Area	Site	Water Year*	Sulfur	NO_3-N	NH_4-N	Total Inorganic N
Sequoia/Kings NP	SEQUOIA GF CADMP (1890 m.)	1990	0.90	1.19	1.53	2.71
		1991	0.73	1.43	2.00	3.42
		1992	0.39	0.68	0.85	1.52
		1993	1.15	1.52	2.42	3.94
		1994	0.69	1.12	1.93	3.05
		1995	1.01	1.51	2.22	3.73
		1996	0.51	0.54	0.76	1.29
		1997	0.38	0.60	0.65	1.25
		1998	0.28	0.40	0.42	0.83
		Average	0.67	1.00	1.42	2.42
Yosemite NP	TIOGA PASS UCSB (2993 m.)	1991	0.55	0.70	0.69	1.39
		1992	0.50	0.60	0.60	1.20
		1993	1.08	0.93	1.02	1.95
		Average	0.71	0.74	0.77	1.51
Yosemite NP	YOSEMITE CADMP (1395 m.)	1991	0.41	0.54	0.62	1.15
		1992	0.54	0.72	0.79	1.51
		1993	0.68	0.79	0.78	1.57
		1994	0.38	0.70	0.78	1.49
		1995	0.93	1.19	1.31	2.51
		Average	0.59	0.79	0.86	1.64
Yosemite NP	YOSEMITE/NADP (1408 m.)	1990	0.91	1.16	1.25	2.41
		1992	0.60	0.79	0.60	1.39
		Average	0.76	0.97	0.93	1.90

* The CARB water year is July 1 through June 30. For instance, water year 1995 is for July 1, 1994 through June 30, 1995.
** Located ~10 km. from Redwood NP

kg/ha/yr (as N), with about half occurring in precipitation (Table II-11; Blanchard et al., 1996). Nitrogen deposition on the eastern slope of the Sierra Nevada may be less than 1 kg/ha/yr (as N; Bytnerowicz and Fenn 1996; Bytnerowicz et al. 1991, 1992, 2000).

Some controversy exists regarding the levels of S or N deposition that are potentially harmful to ecosystems. The National Acid Precipitation Program (NAPAP) has asserted that a single threshold deposition rate for significant adverse effects does not exist (NAPAP 1998). However, adverse vegetation effects are considered unlikely to occur where total S deposition rates are less than about 5 kg/ha/yr (Peterson et al. 1992). Also, total N deposition rates below 3 kg/ha/yr are not expected to cause injury to vegetation, while those between 3 and 20 kg/ha/yr are potentially injurious depending upon species, or may have long-term effects on ecosystem structure and diversity (Peterson et al. 1992). Though threshold deposition rates for aquatic effects are not well defined, surveys have established that chronic acidification of surface waters in the Sierra Nevada has not occurred or has been small in magnitude (Cahill et al., 1996; NAPAP, 1991; Landers et al., 1987).

b. Ozone

The National Park Service operates sites to monitor hourly ozone concentrations at most of the California national parks. Several ozone air-quality statistics derived from the data are shown in Table II-12, with incomplete years noted. Blanks indicate that data were unavailable for the year. The various statistics tend to be closely related, and locations and periods having high maximum ozone also have high ozone exposure (see Figure II-7). The highest ozone levels were seen at JOTR and SEKI. JOTR tends to have higher ozone exposure levels relative to its maximum 9 am to 4 pm ozone concentrations, because its peak hourly ozone levels often occur between 4 and 6 pm. The lowest ozone levels were seen at YOSE-Wawona Valley, REDW, PINN, and PORE.

Three ozone standards are applicable to sites in California. The Federal National Ambient Air Quality standard (NAAQS) for the maximum hourly ozone concentration is 0.12 parts per million (ppm) or, equivalently, 120 parts per billion (ppb). (Technically, only values of 125 ppb or greater are violations of the NAAQS because measurements are rounded to the nearest 10 ppb before determining compliance.) The federal one-hour ozone standard is violated when a site has more than three days in three years with peak hourly ozone concentrations exceeding 0.12 ppm. More than three exceedances within three years occured during most years at JOTR, and in 1984 at Sequoia - Ash Mountain (SEAM) (Table II-12). The California state 1-hr ozone standard is 90 ppb. It is not to be exceeded except for rare events (typically having a frequency of occurrence of less than once in three years), and a statistical procedure is used for determining rare events. The California one-hour standard has been violated during recent years at all parks except REDW and PORE (Table II-12). Finally, a new federal standard requires that the three-year average of the 4th-highest maximum daily 8-hour ozone concentration not exceed 80 ppb (equivalently, less than 85 ppb since data may be rounded to the nearest 10 ppb). This standard has been challenged and will not be enforced until the courts have resolved the legal questions. For many locations, eight-hour maxima tend to occur during mid-day hours and are therefore comparable in magnitude to the maximum 9 a.m. to 4 p.m. averages shown in Table II-12. The fourth-highest eight-hour maxima will be less than the maximum 9 a.m. to 4 p.m. values that are tabled, but the data do indicate that violations of the eight-hour ozone standard will likely occur at both JOTR and SEKI.

The one-hour maximum ozone NAAQS was primarily intended as a criterion for protecting public health. Other measures of ozone, which incorporate the duration and frequency of high concentrations, are thought to provide better indicators of potential vegetation damage. Such indicators include the seven-hour growing season mean, SUM60, and other measures of ozone exposure (SUM60 is the sum of all hourly ozone concentrations equaling or exceeding 60 ppb). In comparison with SUM60, SUM00 sums all hourly ozone concentrations. The SUM00 correlates with an ozone injury index (see Figure I-2), though some studies suggest that a better correlation is obtained between ozone injury and SUM90 (Cahill et al., 1996). Since various ozone exposure measures, including the W126, all correlate with SUM60, the SUM60 values are discussed here. The maximum 9am to 4pm concentrations ranged from 41 ppb at REDW to 120 ppb at JOTR and Sequoia National Park. JOTR and SEKI had SUM60 indexes over 100,000 ppb-hours during this reporting period. SUM60 indexes were lower at the three YOSE sites, generally ranging from 30,000-60,000 ppb-hour, with only three years at YOSE-Turtleback Dome exceeding 100,000 ppb-hour. The sites at LAVO and PINN had lower SUM60 indexes, ranging from 8,000-98,000 ppb-hour. REDW and PORE had very low SUM60 indexes of 61 and 372 ppb-hour, respectively.

Table II-12. Summary of ozone concentrations and exposure from NPS monitoring sites (Source: Joseph and Flores, 1993; National Park Service, Air Resources Division 2000). Bold-face values of the 3-year average number of exceedances indicate violations of the federal 1-hour ozone standard.

Site	Year	Maximum Daily 1-hour Value (ppbv)	2nd Highest Daily 1-hour Value (ppbv)	Number of Daily Maximum 1-hour Values ≥ 125 ppb	3-Year Average Number of Exceedances	Maximum 9am-4pm Average (ppbv)	Sum60 (ppbv-hour)	Sum06 (ppbv-hour)[a]	Number of Valid Hours of Ozone Measurements
JOTR	1987	148	147	10	na	104	108,426	52,000	4,585
	1988	134	131	3	na	84	123,216	48,000	6,220
	1989	139	126	2	**5.0**	93	76,104	15,000	4,606
	1990	132	120	1	2.0	92	80,151	21,000	5,690
	1991	135	133	6	3.0	107	130,262	65,000	6,845
	1992	138	131	7	**4.7**	95	114,021	57,000	6,726
	1993	138	121	1	**4.7**	85	72,325	16,000	6,052
	1994	165	147	20	**9.3**	102	204,998	72,000	8,183
	1995	151	148	4	**8.3**	90	133,201	45,000	8,250
	1996	146	139	5	**9.7**	109	179,079	69,000	8,290
	1997	149	142	9	**6.0**	101	178,426	65,000	8,190
	1998	142	138	9	**7.7**	92	119,318	43,000	7,362
	1999	137	127	2	**6.7**	95	173,371	64,000	7,929
LAVO	1988	89	82	0	na	75	20,363	13,000	6,493
	1989	92	85	0	na	76	21,462	18,000	8,023
	1990	98	92	0	0	84	na	14,000	7,792
	1991	80	77	0	0	71	12,093	9,000	8,106
	1992	80	80	0	0	64	13,850	11,000	8,125

Table II-12. Continued.

Site	Year	Maximum Daily 1-hour Value (ppbv)	2nd Highest Daily 1-hour Value (ppbv)	Number of Daily Maximum 1-hour Values $ 125 ppb	3-Year Average Number of Exceedances	Maximum 9am-4pm Average (ppbv)	Sum60 (ppbv-hour)	Sum06 (ppbv-hour)[a]	Number of Valid Hours of Ozone Measurements
LAVO	1993	84	72	0	0	67	8,478	6,000	7,741
	1994	92	90	0	0	81	48,090	31,000	8,192
	1995	90	89	0	0	73	21,405	12,000	7,337
	1996	93	83	0	0	77	31,993	19,000	7,352
	1997	81	79	0	0	74	12,231	9,000	7,948
	1998	92	89	0	0	75	33,289	20,000	8,217
	1999	109	108	0	0	82	47,614	25,000	7,449
PINN	1987	146	140	5	na	103	98,034	45,000	6,099
	1988	127	121	1	na	84	63,070	25,000	8,005
	1989	138	107	1	2.3	80	47,603	25,000	7,599
	1990	121	112	0	.7	94	44,440	21,000	7,719
	1991	139	103	1	.7	96	57,814	33,000	7,451
	1992	108	108	0	.3	82	41,308	14,000	6,904
	1993	110	104	0	.3	83	47,754	25,000	7,954
	1994	97	95	0	0	77	44,901	23,000	8,150
	1995	138	110	1	.3	82	49,332	30,000	8,050
	1996	120	118	0	.3	95	64,788	41,000	8,102
	1997	112	92	0	.3	86	36,598	21,000	8,125
	1998	124	113	0	0	85	43,209	32,000	7,918
	1999	107	105	0	0	80	52,155	29,000	7,855

Table II-12. Continued.

Site	Year	Maximum Daily 1-hour Value (ppbv)	2nd Highest Daily 1-hour Value (ppbv)	Number of Daily Maximum 1-hour Values $ 125 ppb	3-Year Average Number of Exceedances	Maximum 9am-4pm Average (ppbv)	Sum60 (ppbv-hour)	Sum06 (ppbv-hour)[a]	Number of Valid Hours of Ozone Measurements
PORE	1989	80	73	0	na	63	2,532	2,000	7,870
	1990	78	67	0	na	55	1,223	1,000	8,072
	1991	72	63	0	0	65	729	1,000	7,983
	1992	66	66	0	0	52	372	0	6,924
REDW	1988	68	60	0	na	44	445	0	8,007
	1989	47	47	0	na	43	0	0	7,521
	1990	61	53	0	0	45	61	0	7,488
	1991	54	52	0	0	44	0	0	7,677
	1992	64	55	0	0	49	249	0	7,925
	1993	54	50	0	0	41	0	0	7,827
	1994	51	50	0	0	47	0	0	8,079
	1995	52	50	0	0	45	0	0	2,771
SEKI Ash Mountain	1982	150	140	10	na	120	152,980	85,000	2,630
	1983	120	120	0	na	110	83,590	61,000	2,625
	1984	130	120	1	**3.7**	93	58,220	17,000	1,057
	1985	140	130	3	1.3	107	168,430	90,000	3,237
	1986	140	120	1	1.7	109	126,560	73,000	3,626
	1987	138	127	4	2.7	116	206,607	77,000	8,266
	1988	124	121	0	1.7	104	186,801	63,000	7,849
	1989	116	114	0	1.3	91	159,744	71,000	8,015

Table II-12. Continued.

Site	Year	Maximum Daily 1-hour Value (ppbv)	2nd Highest Daily 1-hour Value (ppbv)	Number of Daily Maximum 1-hour Values $ 125 ppb	3-Year Average Number of Exceedances	Maximum 9am-4pm Average (ppbv)	Sum60 (ppbv-hour)	Sum06 (ppbv-hour)[a]	Number of Valid Hours of Ozone Measurements
SEKI Ash Mountain	1990	120	119	0	0	93	172,409	68,000	8,111
	1991	122	120	0	0	97	155,846	73,000	8,216
	1992	120	117	0	0	100	159,130	72,000	8,081
	1993	128	127	2	.7	109	146,730	75,000	8,195
	1994	132	126	4	2.0	108	176,692	87,000	8,192
	1995	128	119	1	2.3	107	126,327	71,000	7,933
	1996	124	117	0	1.7	100	131,221	75,000	3,438
SEKI Grant Grove	1990	121	119	0	na	96	134,272	57,000	8,253
	1991	110	108	0	na	97	122,938	62,000	8,164
	1992	124	116	0	0	100	158,909	61,000	8,169
	1993	128	125	2	.7	99	124,904	64,000	8,018
	1994	125	123	1	1.0	100	167,318	79,000	8,210
	1995	115	112	0	1.0	91	101,174	54,000	5,513
SEKI Lower Kaweah	1984	110	110	0	na	93	133,350	63,000	3,085
	1985	130	120	1	na	100	83,500	32,000	1,722
	1986	130	120	1	.7	94	62,840	31,000	2,760
	1987	118	109	0	.7	89	103,626	46,000	7,938
	1988	117	112	0	.3	95	171,415	67,000	8,126
	1989	112	110	0	0	87	119,949	61,000	8,051
	1990	121	112	0	0	92	141,511	58,000	8,221

Table II-12. Continued.

Site	Year	Maximum Daily 1-hour Value (ppbv)	2nd Highest Daily 1-hour Value (ppbv)	Number of Daily Maximum 1-hour Values $125 ppb	3-Year Average Number of Exceedances	Maximum 9am-4pm Average (ppbv)	Sum60 (ppbv-hour)	Sum06 (ppbv-hour)[a]	Number of Valid Hours of Ozone Measurements
SEKI Lower Kaweah	1991	116	112	0	0	97	128986	68,000	8,186
	1992	121	119	0	0	102	163933	70,000	8,182
	1993	129	125	2	.7	99	167315	78,000	8,212
	1994	125	123	1	1.0	104	143409	81,000	7,721
	1995	115	110	0	1.0	93	90029	55,000	8,033
	1996	123	122	0	.3	96	152521	76,000	8,213
	1997	112	111	0	0	90	105677	60,000	8,111
	1998	131	116	1	.3	94	84666	58,000	7,016
	1999	115	108	0	.3	91	132466	62,000	7,377
SEKI Lookout Point	1992	117	115	0	na	105	155121	74,000	6,221
	1993	119	113	0	na	94	93798	30,000	5,865
	1997	120	115	0	0	98	140618	69,000	6,166
	1998	119	117	0	0	99	92214	66,000	6,245
	1999	124	122	0	0	101	171734	77,000	7,771
YOSE Camp Mather	1988	96	95	0	na	84	52699	26,000	5,865
	1989	87	82	0	na	74	24307	21,000	7,284
	1990	86	86	0	0	81	22333	19,000	4,835
	1991	88	86	0	0	76	44521	31,000	4,958
	1992	86	81	0	0	na	na	21,000	7,970
	1993	91	91	0	0	na	na	38,000	8,085

Table II-12. Continued.

Site	Year	Maximum Daily 1-hour Value (ppbv)	2nd Highest Daily 1-hour Value (ppbv)	Number of Daily Maximum 1-hour Values $ 125 ppb	3-Year Average Number of Exceedances	Maximum 9am-4pm Average (ppbv)	Sum60 (ppbv-hour)	Sum06 (ppbv-hour)[a]	Number of Valid Hours of Ozone Measurements
YOSE Camp Mather	1994	97	95	0	0	na	na	49,000	6843
	1995	97	92	0	0	na	na	29,000	4243
	1996	94	91	0	0	na	na	32,000	1956
YOSE Turtleback Dome	1992	100	na	na	na	84	83,770	na	na
	1994	113	111	0	na	92	136,595	62,000	8227
	1995	114	104	0	0	86	82,306	42,000	7903
	1996	107	106	0	0	84	113,904	57,000	8099
	1997	111	107	0	0	88	52,168	27,000	8000
	1998	106	104	0	0	87	92,922	51,000	7287
	1999	96	95	0	0	81	118,407	50,000	7390
YOSE Wawona Valley	1987	133	116	1	na	105	91,746	55,000	6128
	1988	119	113	0	na	90	87,885	52,000	7958
	1989	111	99	0	.3	87	53,428	37,000	8142
	1990	120	116	0	0	94	81,957	53,000	7334
	1991	105	98	0	0	88	40,223	30,000	7528
	1992	111	108	0	0	na	na	38,000	8129
	1993	98	95	0	0	na	na	26,000	6733
	1994	95	92	0	0	na	na	21,000	7890
	1995	110	103	0	0	na	na	32,000	8051
	1996	103	101	0	0	na	na	36,000	6694

[a] Maximum 8 am - 8 pm 90-day rolling average

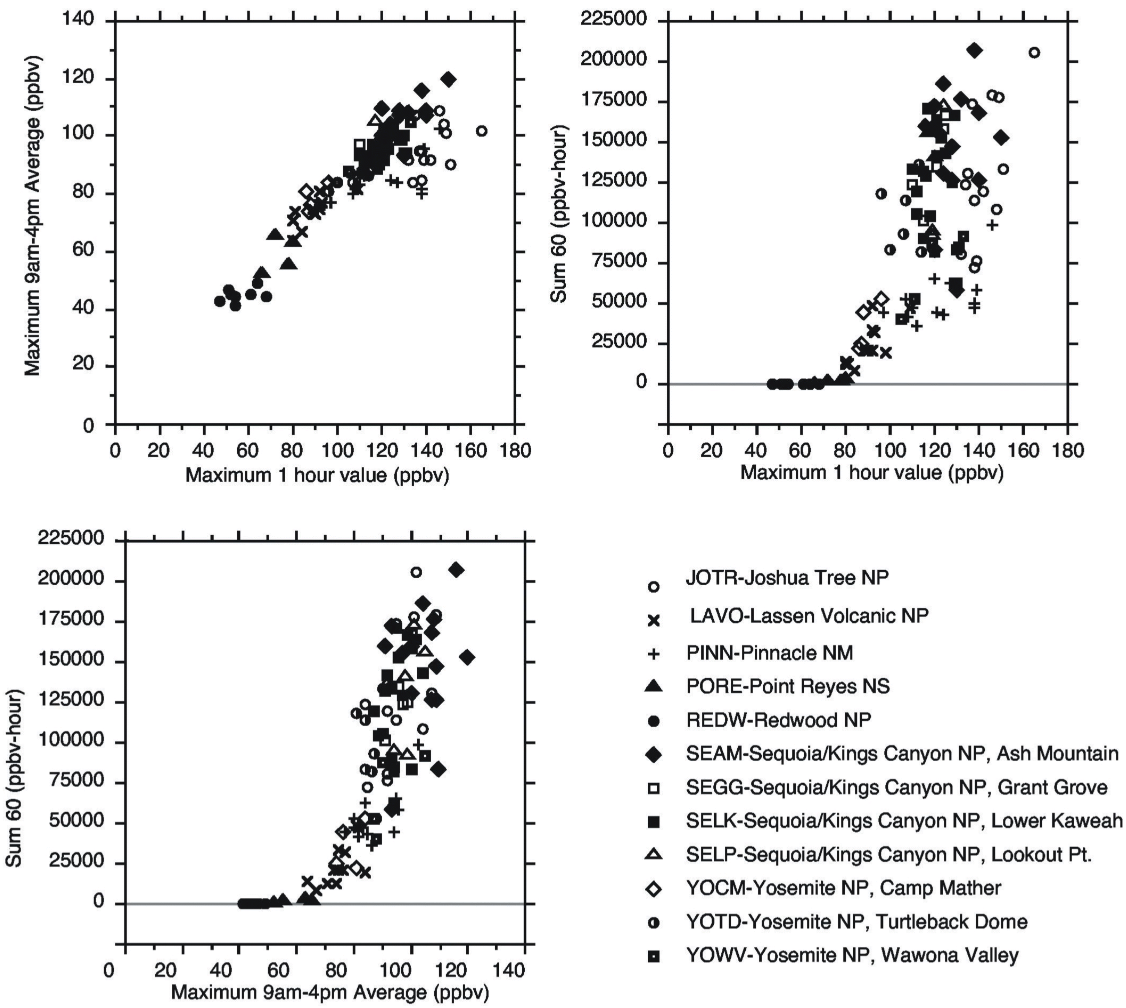


Figure II-7. Comparison of three ozone air-quality statistics. The statistics are the maximum hourly concentration, the maximum 9 a.m. to 4 p.m. ozone concentrations, and ozone exposure (SUM60, or the sum of all hourly ozone concentrations exceeding 60 ppb). Each symbol represents the data for one site during one year.

Passive ozone samplers are located in several Class I areas (NPS Passive Ozone website 1999), providing more extended spatial coverage within some of the parks. These monitors collect samples on a weekly basis, generally for the period of May through September. Average hourly ozone concentrations are calculated based on these weekly samples, which in turn, are reported as an hourly average for the season each year (Table II-13). The data presented in Table II-13 do not show any simple trends in concentrations either over time or with elevation. However, data for 1996 suggest that higher ozone concentrations can be seen along the western side of SEKI (sites at Lower Kaweah, Bear Paw Meadow and Grant Grove), potentially

Table II-13. Summer average hourly ozone concentrations at passive sampling sites within National Parks in California (source: National Park Service, Air Resources Division, NPS Passive Ozone website 1999). Units are ppb.

Sample Locations	Elevation (m)	1995	1996	1997	1998
Lava Beds NM	1451	41.1	41.6	39.3	42.8
Seq/Kings NP-Crabtree	3261		48.4	57.4	53.8
Seq/Kings NP-Le Conte Canyon	2652		49	53.6	52.1
Seq/Kings NP-Bear Paw Meadow	2177		62.8		60.4
Seq/Kings NP-Mineral King	2314		47.4		
Seq/Kings NP-Cedar Grove	1432		39.4		
Seq/Kings NP-Lower Kaweah	1865		64.6		
Seq/Kings NP-Grant Grove	1981		59		
Channel Is. NP-E. Santa Cruz	46			41.9	
Channel Is. NP-Santa Rosa Is., Stiff Leg	17			39.9	
Channel Is. NP-San Miguel Is.	163			35.8	37.8
Channel Is. NP-Anacapa Is.	49			33.4	43.2
Channel Is. NP-Santa Rosa Is., Black Mt.	282			41.3	
Channel Is. NP-site 6	na			36.9	
Joshua Tree NP-Sunrise Well	152			58.3	57.6
Joshua Tree NP-Black Rock	1231			60.9	65.5
Joshua Tree NP-Lost Horse	1256			56.3	59.2
Joshua Tree NP-N. Entrance	866				63.7
Joshua Tree NP-Keys View	1585				71.4
Joshua Tree NP-Cottonwood	305				56.2
Yosemite NP-Camp Mather	1432			42.4	40.2
Yosemite NP-Wawona Valley	1280			35.4	31.7
Yosemite NP-Tioga Road	2393				51.2
Yosemite NP-Tuolumne Meadows	3002				33.4
Yosemite NP-Tioga Pass	3030				51

influenced by pollution outflow from the San Joaquin Valley. In JOTR, 1998 data suggest that higher ozone concentrations can be found toward the northwestern end of the park (Black Rock, Lost Horse, North Entrance and Keys View), areas potentially more exposed to pollutants transported from the SoCAB. For comparison, mean tropospheric background ozone is generally in the range of 25 to 40 ppb, with higher one-hour values (up to 50 to 80 ppbv) occasionally occurring in remote locations (Altshuller and Lefohn 1996).

Lee and Hogsett (in review) recently employed a spatial interpolation method to estimate trophospheric ozone exposure across the western United States, based on data from 179

monitoring stations. The approach used auxiliary digital elevation model (DEM) data at 1-km resolution to predict temperature and ozone exposure (monthly SUM06 - SUM06 is expressed in units of ppm-hours and is equal to SUM60 divided by 1000). Monthly mean daily maximum air temperatures were spatially interpolated using LOWESS nonparametric regression and kriging of the LOWESS residuals and interpolated to 2-km grid points of a DEM and to available ambient air quality monitoring points. Monthly ozone exposures were spatially interpolated using LOWESS fits to relate ozone levels to elevation, predicted temperature, and the geographic coordinates, and then interpolated to 2-km grid points. The elevation-based spatial interpolation procedure produced accurate and precise temperature and ozone exposure surfaces which were logically consistent with local topographical features and atmospheric conditions known to influence ozone formation and transport. The leave-one-out cross-validation mean absolute error was 0.93^{o} C for the monthly mean daily maximum temperature and 1.95 ppm-h for the monthly SUM06 index for June 1994 (Lee and Hogsett in review). Results of the ozone interpolation for California were provided by Henry Lee (U.S. Environmental Protection Agency, Corvallis, OR) and are depicted in Figure II-8. Standard deviations of the predictions are shown in Figure II-9. At locations where the standard deviations are relatively large, collection of additional ozone monitoring data would be particularly useful in order to improve our ability to interpolate ozone exposures.

The spatial pattern for the June, 1994 SUM06 index was complex in central and southern California, where SUM06 values changed abruptly over short distances between locations of differing elevation. The highest SUM06 levels in the western United States (> 25 ppm-h) were predicted for the inland portions of Kern, San Bernardino, Riverside, and San Diego Counties in California. The lowest SUM06 values in the California and NW Coast regions were predicted along the coast, where temperatures were lower and atmospheric conditions were not conducive to ozone formation. Lee and Hogsett (in review) attributed the increasing west to east gradient in SUM06 along the coast to the prevailing westerly winds. Urban ozone concentrations are trapped within the coast subsidence inversion and transported to the heated inland mountain slopes (c.f., Edinger 1973).

c. SO_2

Table II-14 shows the maximum and mean SO_2 integrated daily sample for several Class I areas in California. Yearly means and maxima for each year with data available were computed using daily 24-hour samples (IMPROVE website 1999). Some sites had incomplete data. It was standard to collect 24-hour SO_2 samples every 3-4 days; years when fewer than seventy-five samples were reported are noted. With the exception of the 1990 maximum at LAVO, maxima tended to be well under 2 ppb at all sites in all years. Means ranged from 0.01-0.2 ppb at all sites for all years, again with the exception of LAVO in 1990, when the mean was 0.45 ppb. Mean SO_2 declined over the period of measurement at all locations. In all cases, the ambient SO_2 concentrations were well below the levels at which plant injury has been documented, ~40 to 50 ppb 24-hour average and 8-12 ppb annual average (Peterson et al. 1992).

d. Other Air Quality-Related Monitoring

In addition to the air quality monitoring documented above, several other types of monitoring are ongoing or have been completed over several years. Additional air quality-related monitoring on federal lands includes studies of lake and stream chemistry, lichens, and impacts on pine trees and aquatic amphibians. Table II-15 describes additional air quality-related monitoring on Class I and non-Class I federal lands.

Table II-14. Maximum and mean SO_2, from 24-hour-resolution samples. Samples are collected every 3-4 days, unless noted. (Source: IMPROVE website 1999). Units are ppb. Units were converted from ng/m^3 using standard temperature and pressure.

LOCATION	SO2	1988	1989	1990	1991	1992	1993	1994	1995	1996
Death Valley NP	Maximum	na	na	na	na	na	0.45*	0.43	0.33	0.68
	Mean	na	na	na	na	na	0.16*	0.1	0.06	0.09
Joshua Tree NP	Maximum	na	na	na	0.43*	0.16*	na	na	na	na
	Mean	na	na	na	0.16*	0.08*	na	na	na	na
Lassen Volcanic NP	Maximum	0.63*	0.83*	18.23**	0.11*	0.05**	0.02*	na	na	na
	Mean	0.05*	0.03*	0.45**	0.04*	0.01**	0.01*	na	na	na
Pinnacles NM	Maximum	0.26*	0.34	0.59	0.37	0.33	0.34	0.47	0.32*	na
	Mean	0.09*	0.1	0.14	0.1	0.1	0.08	0.09	0.05*	na
Point Reyes NS	Maximum	0.48*	1.04**	1.25	0.37**	0.78**	0.43*	na	na	na
	Mean	0.06*	0.12**	0.15	0.05**	0.07**	0.03*	na	na	na
Redwood NP	Maximum	0.37*	0.15*	0.15**	0.30*	0.25**	0.04	0.28	0.03*	na
	Mean	0.07*	0.05*	0.03**	0.05*	0.02**	0.01	0.02	0.01*	na
San Gorgonio W	Maximum	na	na	na	0.46	0.31*	na	na	na	na
	Mean	na	na	na	0.14	0.08*	na	na	na	na
Sequoia/Kings NP	Maximum	na	na	na	na	na	0.91*	0.82	0.48	0.89**
	Mean	na	na	na	na	na	0.20*	0.17	0.13	0.19**
Yosemite NP	Maximum	na	na	na	0.63	0.04*	0.36*	0.45	0.32	0.34**
	Mean	na	na	na	0.15	0.02*	0.13*	0.09	0.07	0.08**

na Not available
* Less than 50 samples collected for the year
** 50-75 samples collected for the year

Table II-15. Additional air quality monitoring on Class I and non-Class I federal lands in California (source: USDA 1997 and SFCAP 1997).

Location	Type of monitoring
Class I Lands	
Desolation W	Lichens, lake chemistry
Emigrant W	Lichens, lake chemistry
Marble Mountain W	Lichens
San Gabriel W	Lichens
Agua Tibia W	Lichens
Lassen Volcanic NP	ozone injury (pines), lichens
Yosemite NP	ozone injury (pines), acid deposition
Kings Canyon NP	ozone injury (pines), acid deposition
Sequoia NP	ozone injury (pines), acid deposition
John Muir W	acid deposition (aquatic amphibians)
Ansel Adams W	acid deposition (aquatic amphibians)
Hoover W	acid deposition (aquatic amphibians)
Yolla Bolly-Middle Eel W	acid deposition in lakes
Sierra Nevada	acid depositions in lakes
Non-Class I Federal Lands	
Inyo NF	water, acid deposition (aquatic amphibians), lichens
Tahoe NF	ozone injury (pines)
Eldorado NF	ozone injury (pines)
Sierra NF	ozone injury (pines), acid deposition (aquatic amphibians)
Stanislaus NF	ozone injury (pines)
Sequoia NF	ozone injury (pines), stream sampling
San Bernardino NF	ozone injury (pines)
Mono Basin NF	acid deposition (aquatic amphibians)
Dinkey Lakes W	acid deposition (aquatic amphibians)
Toiyabe NF	acid deposition (aquatic amphibians)
Los Padres NF	NADP
Sierra Nevada	acid deposition in lakes

3. Trends in Ambient Pollutant Concentrations

Emission control efforts have steadily reduced the ambient concentrations of many air pollutants throughout substantial portions of California. The trends are most apparent in the principal metropolitan areas. Declining pollutant levels in the urbanized regions potentially lead to reduced levels of pollutants downwind. Emissions of NO_x in California generally rose throughout the 1970s, and subsequently decreased after 1980 in most air basins due to air pollution control programs. The exception was the SJVAB, where emissions continued to

increase (CARB 1985). Although per vehicle NO_x emissions were reduced substantially, these improvements were largely negated by population and industrial growth.

a. Urban and Regional Ozone Trends

Ozone concentrations in the three most populated areas of the state, SoCAB, SFBAAB, and the San Diego Air Basin (SDAB), have been declining since 1981. The 30 highest daily peak hourly ozone concentrations in a year (annual Top30 mean), when averaged over successive three-year periods, show that ozone concentrations decreased 25 percent in the SoCAB, 15 percent in the SFBAAB, and 12 percent in the SDAB over the period 1981-1990 (CARB, June 1992). In contrast, the 3-year average of the annual Top30 mean declined by 5 percent in the SJVAB and 7 percent in the SVAB. In the Lake County Air Basin, the Top30 mean ozone concentrations decreased 10 percent; in Lake Tahoe Air Basin, Top30 mean ozone concentrations increased 10 percent. It should be noted that both Lake County and Lake Tahoe air basins attain the existing one-hour and proposed eight-hour national ambient air quality standards for ozone.

Another useful air-quality indicator is the highest annual concentration of a pollutant, referred to as the Expected Peak Day Concentration (EPDC). During the period 1982-1992, the basin-wide ozone EPDC decreased 22 percent in the SoCAB, 15 percent in the SFBAAB, and 22 percent in the SDAB (CARB 1995b). During this same period, the EPDC decreased 11 percent in the SJVAB, but remained roughly constant in the SVAB with a decrease of 3 percent in the broader Sacramento area and an increase of 3 percent in the Upper Sacramento Valley.

More recent data show that these trends have continued into the 1990s. Figure II-10 shows maximum 1-hour ozone concentrations for the period 1980 through 1997 (CARB Almanac website 1999b). Peak ozone concentrations in the South Coast, San Francisco Bay, San Diego and Mojave Desert air basins showed a steady decrease over this time period. In contrast, peak ozone concentrations showed little trend in the Sacramento Valley, San Joaquin Valley, Lake Tahoe, and Lake County air basins. Mountain Counties air basin (MCAB) was the only air basin to show an increase in ozone concentrations, with most of the increase occurring between 1984-1988 (55 percent) and little trend occurring after 1989. However, the MCAB trend is least certain, due to the generally lower ozone levels and to the changing locations of the relatively few monitoring sites upon which the statistics were based.

Trends in maximum 8-hour concentrations for ozone showed somewhat similar patterns to those occurring for the 1-hour maxima (Figure II-11). During the period 1980-1997, maximum 8-hour concentrations showed little trend in the SJVAB, Lake County, Lake Tahoe, or Sacramento air basins. Concentrations declined in the SFBAAB, SDAB, and SoCAB, with the highest rate of decline in the South Coast. Maximum 8-hour concentrations increased from 1986 to 1988, and then showed little trend thereafter in the MCAB (data were unavailable for 1980 and 1982-85). The Mojave Desert ozone concentrations were somewhat similar to MCAB, with annually variable increases from 1980-1986, followed by a slow decline through 1997.

The 8-hour ozone design value is defined as the 3-year mean of the 4th-highest maximum daily 8-hour average ozone concentration per year. Trends in the 8-hour design value for ozone were broadly similar to those occurring in the 8-hour maximum (Figure II-12). Lake County, Lake Tahoe and San Joaquin Valley air basins showed little trend over this time period. The 8-hour design values for ozone in the San Diego, Sacramento and San Francisco Bay Area air basins declined modestly, and concentrations in the SoCAB declined steadily during this period. Again, MCAB concentrations reached a peak in 1988, declining slightly afterward.

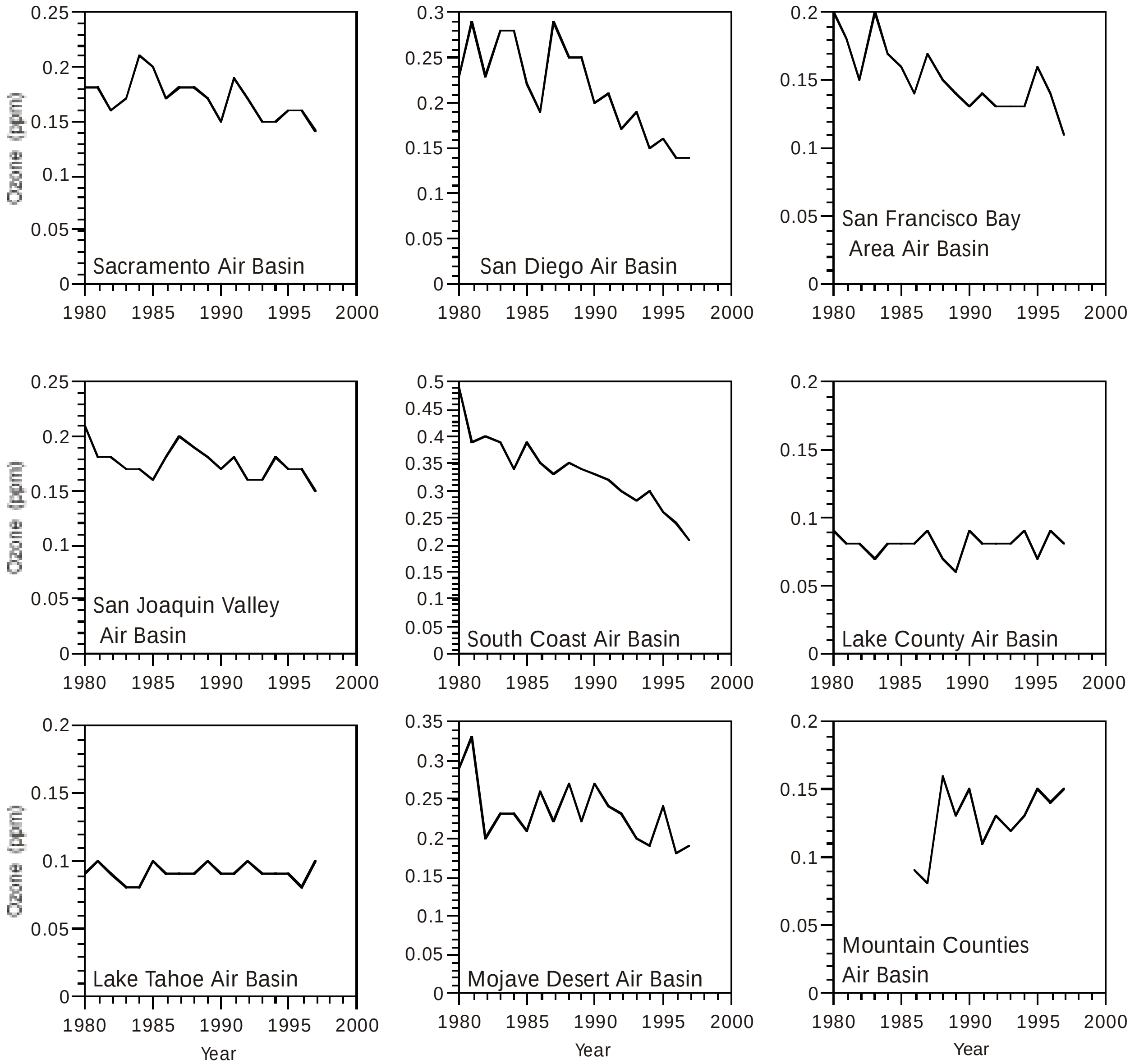


Figure II-10. Maximum 1-hour ozone concentration for selected air basins, 1980-1997. Units are ppm. (Source: CARB Almanac Website 1999b).

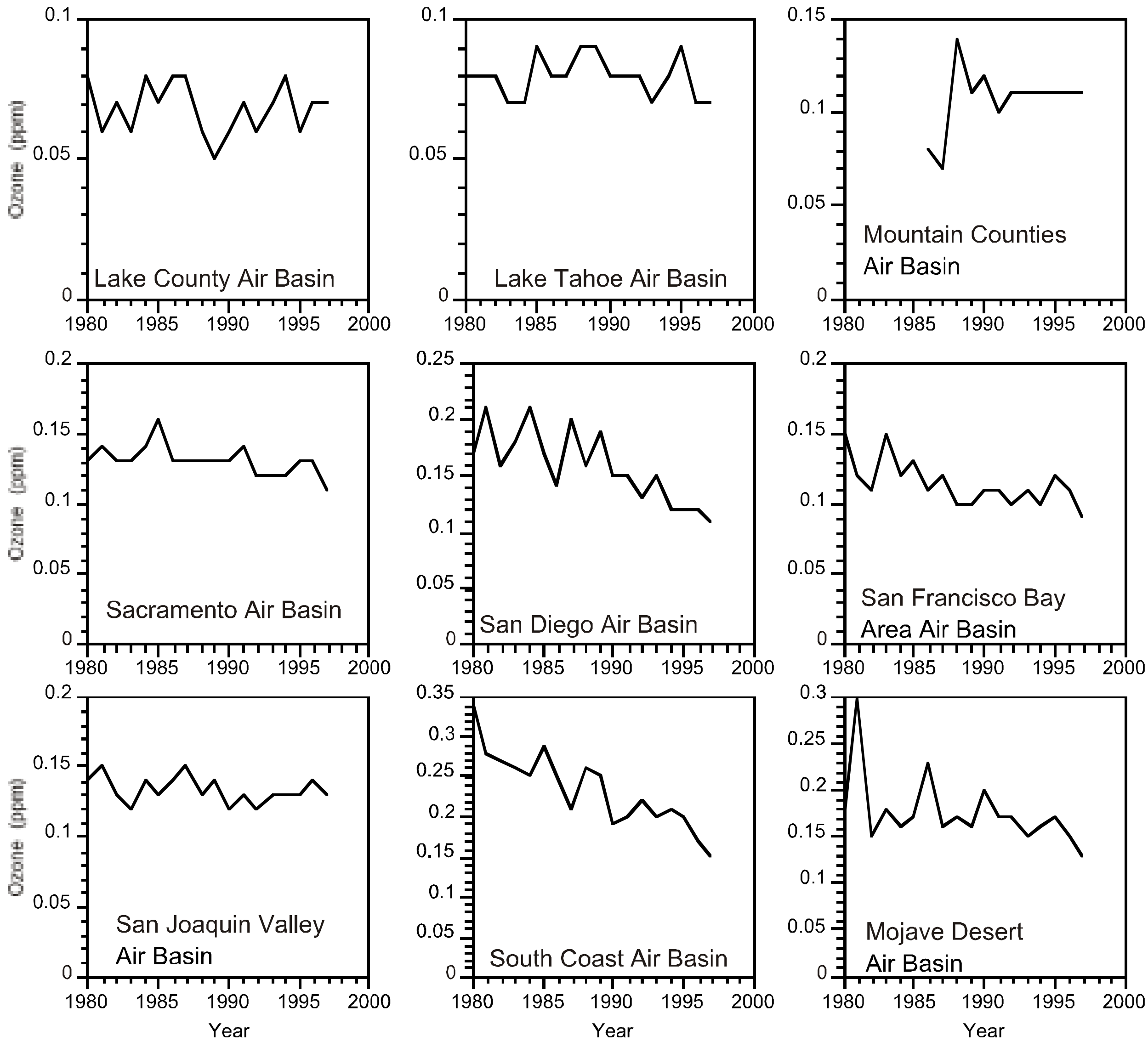


Figure II-11. Maximum 8-hour ozone concentration for selected air basins, 1980-1997 (source: CARB Almanac Website 1999b). Units are ppm.

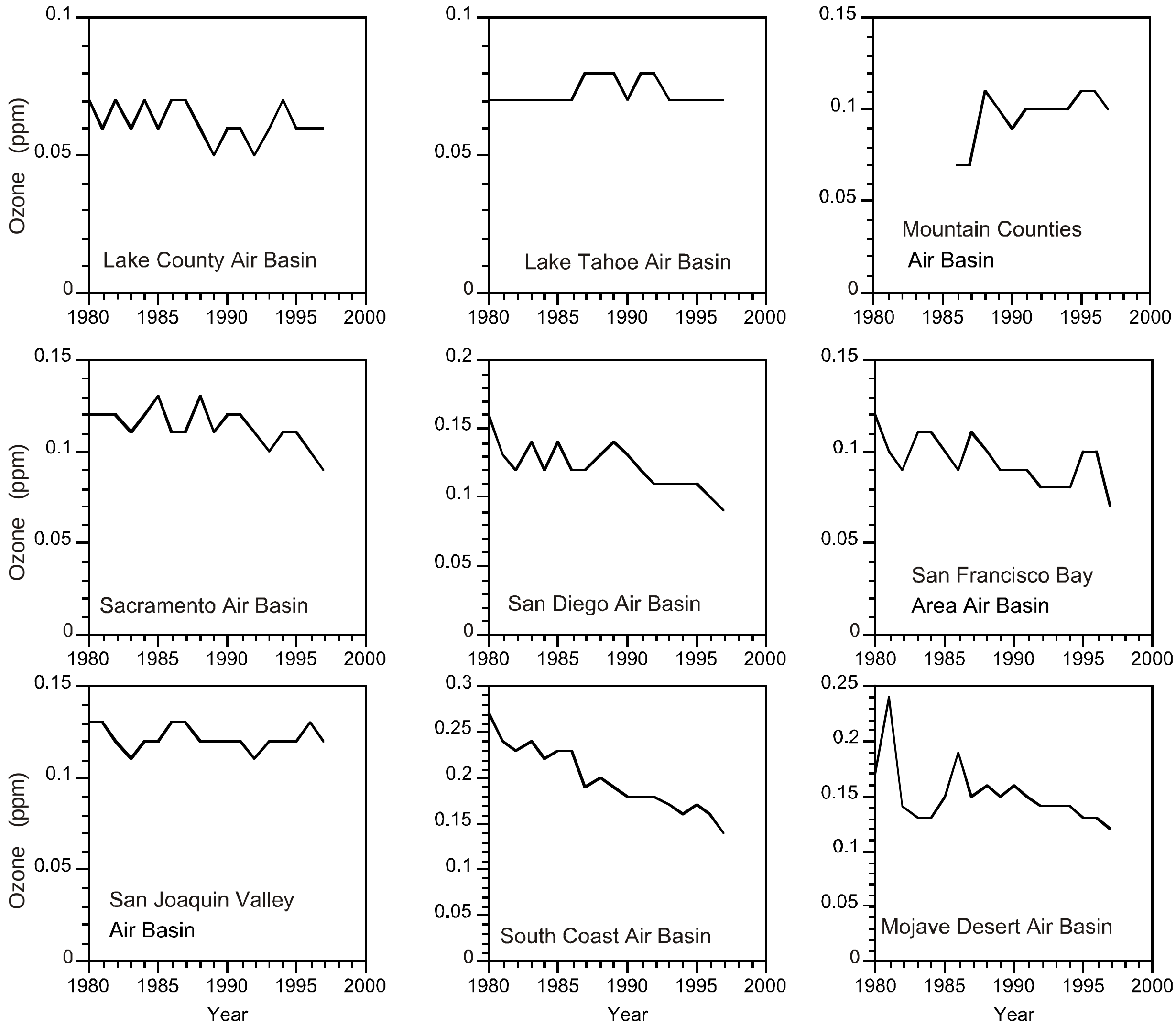


Figure II-12. National 8-hour design value for ozone in selected air basins, 1980-1997. Units are ppm. The 8-hour ozone design value is defined as the 3-year mean of the 4th-highest maximum daily 8-hour average ozone concentration per year. (Source: CARB Almanac Website 1999b)

b. Particulate Trends

PM_{10} annual geometric means have been steadily declining since 1988 in the SoCAB, SFBAAB, MCAB (data unavailable for 1988 and 1989), and SJVAB, with the sharpest declines in the South Coast and the SJV (Figure II-13). The maximum 24-hour concentrations show more year-to-year variability, though they too have been gradually declining in all four of the basins (Figure II-14).

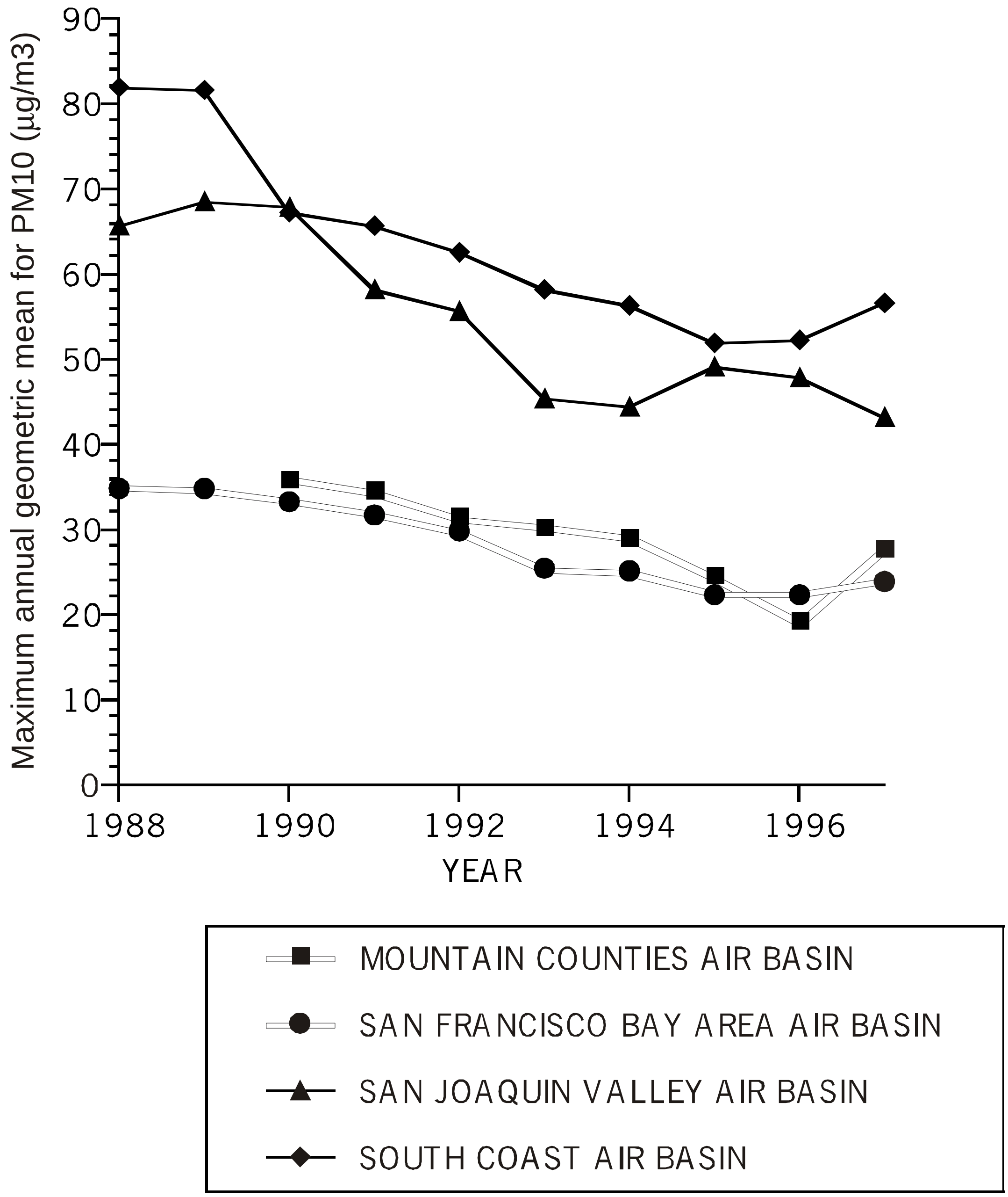


Figure II-13. Maximum annual geometric mean for PM_{10} concentration in selected air basins, 1988-1997. Units are $\mu g/m^3$. (Source: CARB Almanac Website 1999b).

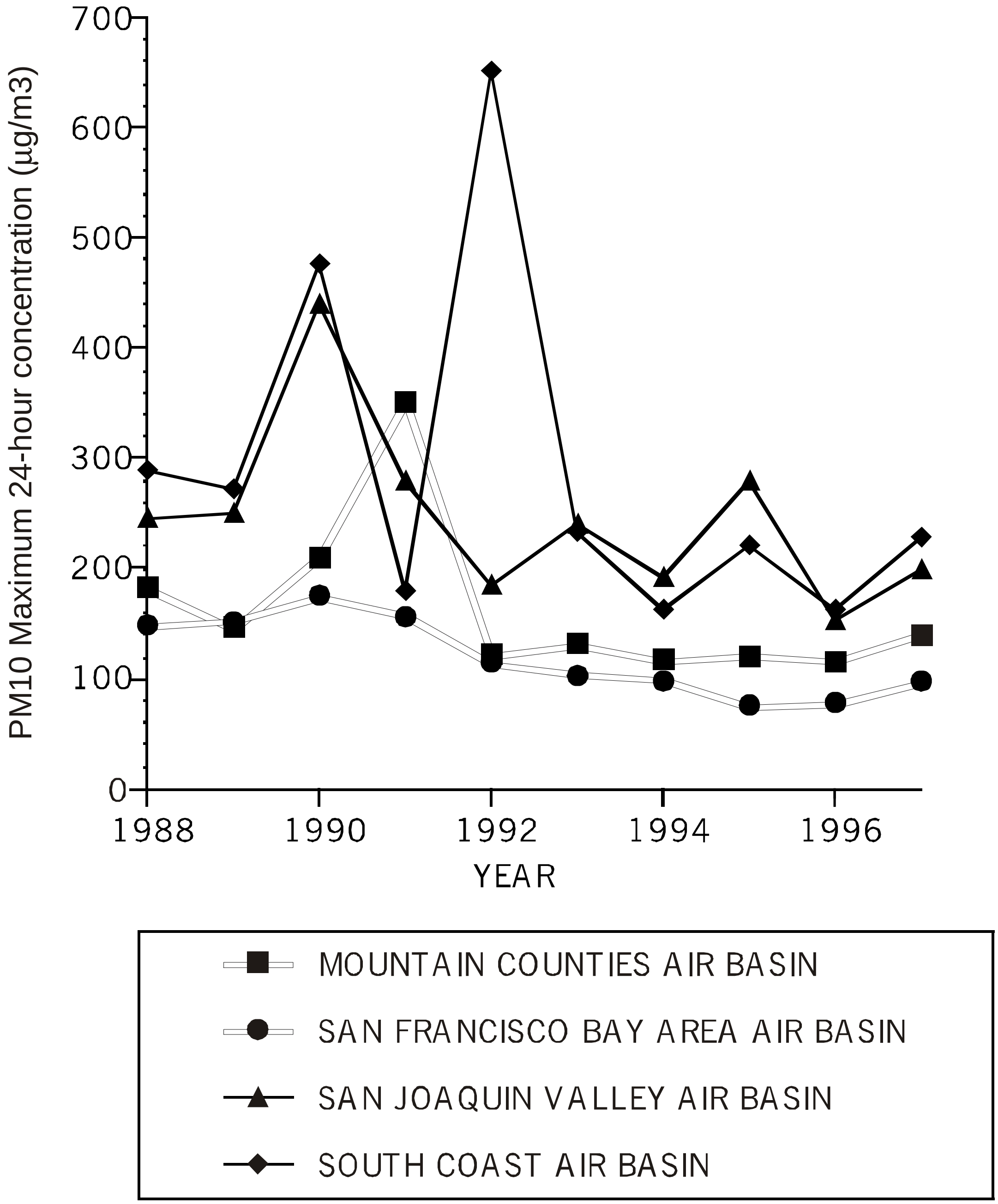


Figure II-14. Maximum 24-hour PM_{10} concentration in selected air basins, 1988-1997. Units are µg/m^3. (Source: CARB Almanac Website 1999b)

c. Carbon Monoxide Trends

Trends since 1980 showed a variable decline in CO levels in the SoCAB, SFBAAB, and SJVAB at both the yearly 1-hour and 24-hour maximum concentrations (Figures II-15 and 16).

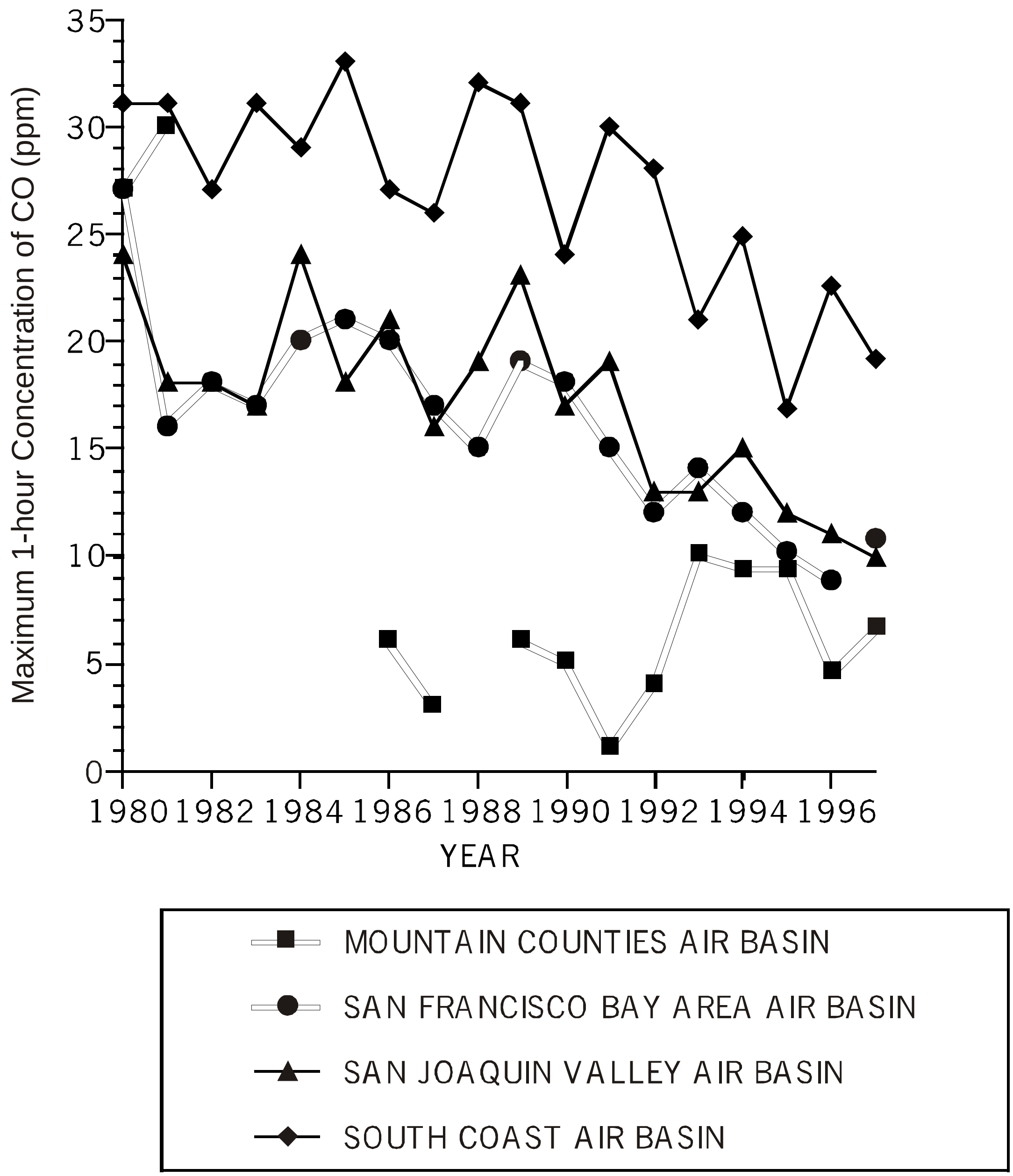


Figure II-15. Maximum 1-hour CO concentration for selected air basins, 1980-1997. Units are ppm. (Source: CARB Almanac Website 1999b)

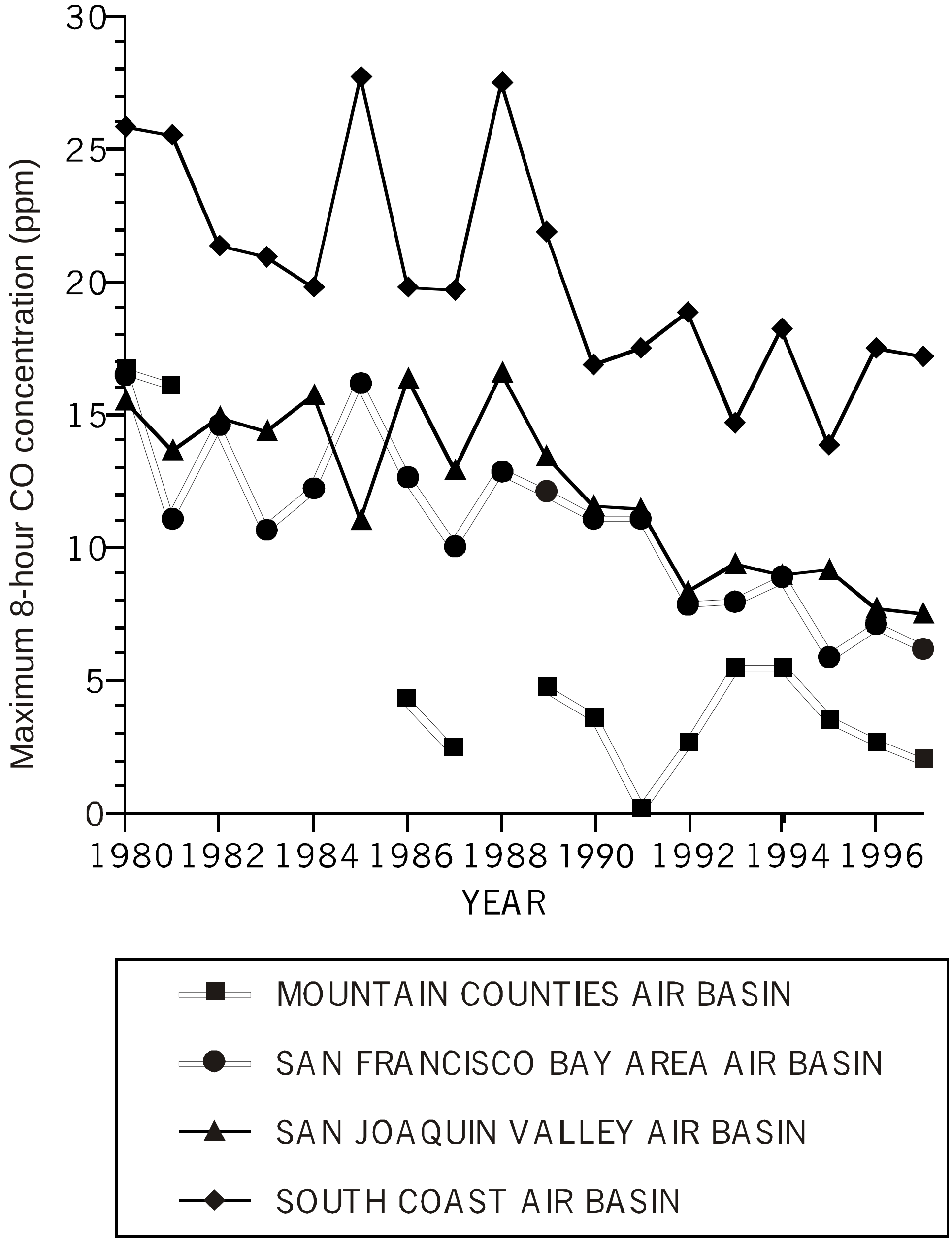


Figure II-16. Maximum 8-hour CO concentration for selected air basins, 1980-1997. Units are ppm. (Source: CARB Almanac Website 1999b).

Expected Peak Day Concentrations for CO showed little trend during the period 1982-1991, and then decreased sharply from 1991-1992 in most areas (CARB 1995b). The decreases coincided with the introduction of oxygenated gasoline during the winter months. EPDC decreased at the basin-maximum site by 17 percent in Los Angeles County (SoCAB), 19 percent in Santa Clara County (SFBAAB), 18 percent in San Diego County (SDAB), and 46 percent in the Lake Tahoe Air Basin.

d. Nitrogen Dioxide Trends

Since 1980, maximum 1-hour concentrations of NO_2 have declined steadily in the SoCAB (Figure II-17). Declines have also occurred in the SFBAAB and the SJVAB, though at a lower rate than in the SoCAB. Ambient NO_2 concentrations in the MCAB were less than 0.1 ppm for the three years of available data.

EPDC trends have been somewhat similar to the annual maxima trends during the period 1981-1993, with some increases and some decreases during the first half of the period and most areas decreasing during the second half of the period. The SoCAB is the only air basin in California that was not in attainment for the National Ambient Air Quality Standard (NAAQS) for NO_2. The peak site in the SoCAB experienced an 11 percent decrease in EPDC for NO_2 during this period.

e. Sulfur Dioxide Trends

Maximum 1-hour concentrations for SO_2 have declined since 1980 in the SoCAB, SFBAAB, and SJVAB (Figure II-18). In the MCAB, SO_2 concentrations were about 0.02 ppm in 1980, the last available measurement.

4. Transport of Air Pollutants

One of the first steps in the effective reduction of ambient pollutant concentrations is to develop quantitative estimates of the contributions of specific emission sources to observed air pollutant concentrations (source attribution). A variety of techniques are available for quantitatively estimating source contributions to each receptor location of concern (source-receptor relationships). However, uncertainties associated with both methods and the available data largely limit current understanding of the relations between emissions in the more urbanized regions of California and ambient concentrations of pollutants in national parks and other Class I areas to a descriptive, or qualitative level. Nonetheless, this descriptive understanding of source-receptor relationships provides a useful characterization of pollutant transport from one portion of California to another.

a. Transport Pathways

California's fifteen air basins (Figure II-2) were originally designated by CARB with the expectation that pollution sources would primarily impact areas within their air basins of origin. However, as more has been learned about the transport of pollutants, it is now known that emissions within some air basins can have a significant impact on neighboring air basins. The CARB has identified twenty-two pathways, known as transport couples, where ozone or ozone precursors from one air basin are transported to an adjacent air basin (Figure II-2). The criteria used to evaluate evidence of transport included: time of day of exceedances, temporal progression of exceedances, air flow patterns (surface and aloft where available), spatial extent of violations, emission levels in both upwind and downwind areas, back trajectory analyses, special field studies, and modeling (not all of these criteria were used for each couple). CARB staff reported that in all of these pathways, transported ozone or ozone precursors contributed

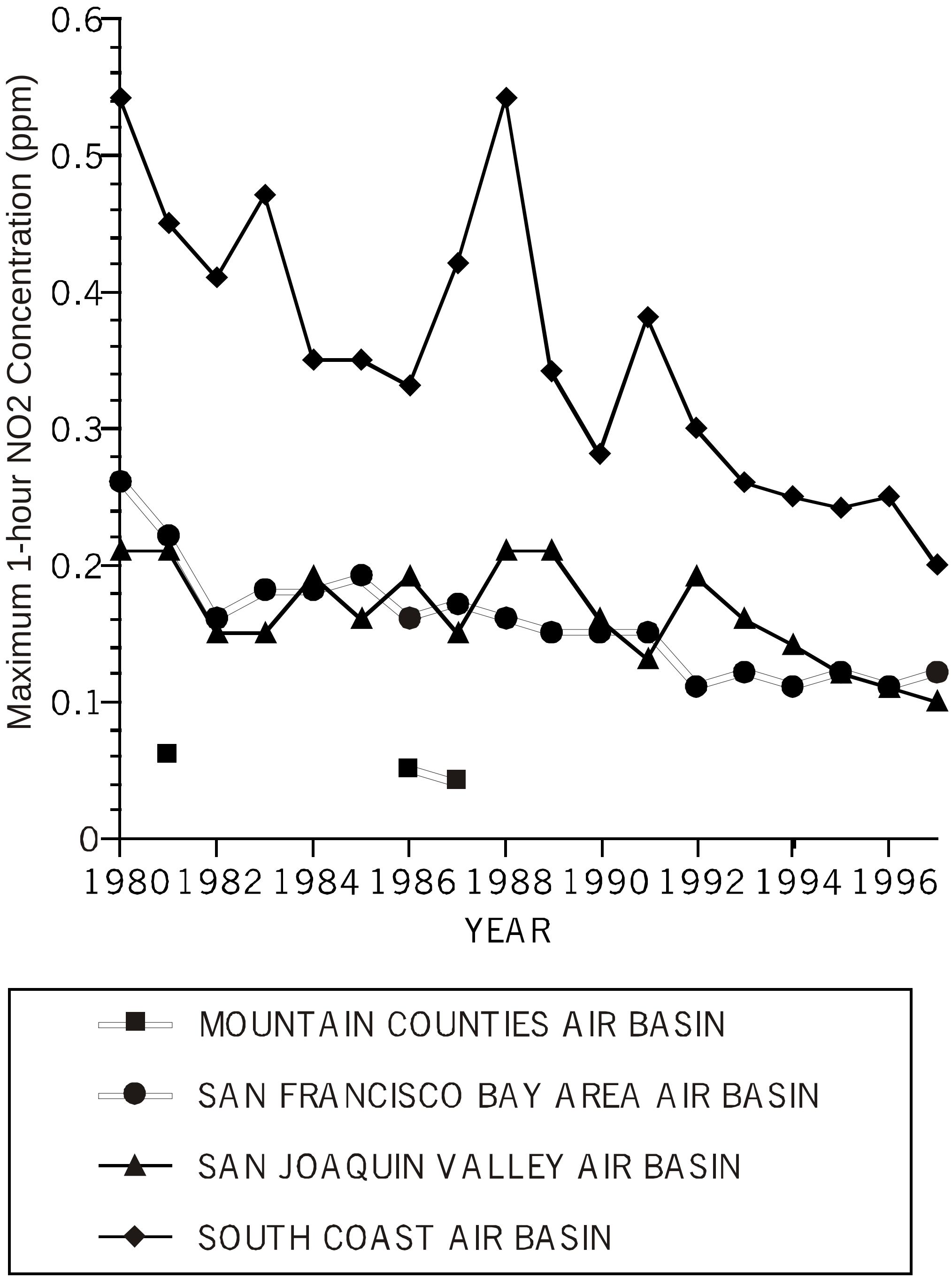


Figure II-17. Maximum 1-hour NO_2 concentration for selected air basins, 1980-1997. Units are ppm. (Source: CARB Almanac Website 1999b).

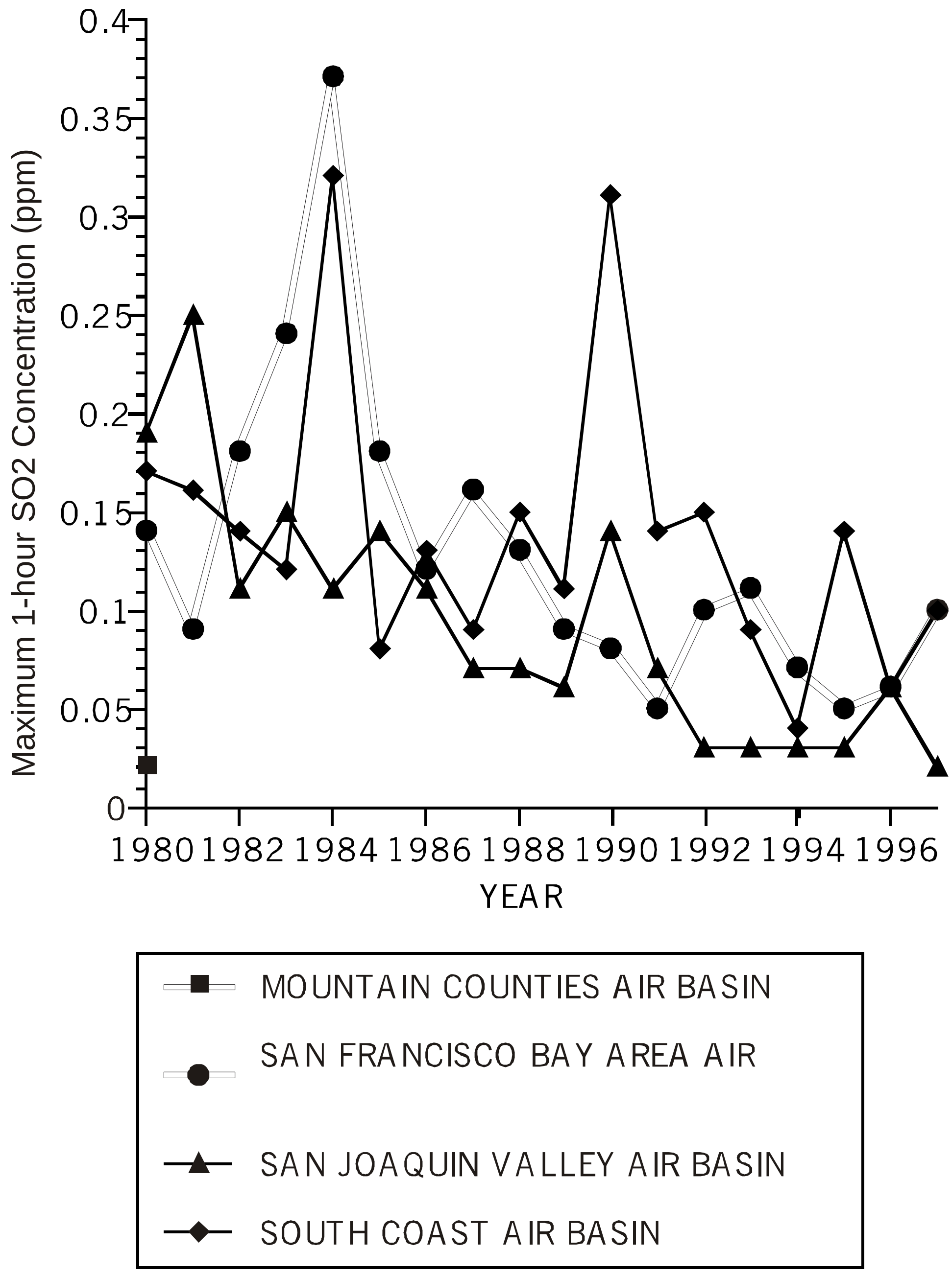


Figure II-18. Maximum 1-hour SO_2 concentration for selected air basins, 1980-1997. Units are ppm. (Source: CARB Almanac Website 1999b).

significantly to producing violations of the State ozone standard in downwind basins (the transported ozone or ozone precursors in combination with emissions from the downwind basin produced the violation). CARB staff also reported that in more than half of these pathways (13 out of 22), the transported ozone or ozone precursors was enough by itself to produce violations. The CARB has characterized the degree of transport as overwhelming (O), significant (S), or inconsequential (I) (Table II-16). Overwhelming transport describes a situation where pollution from the upwind basin is solely responsible for at least one violation of the State ozone standard. Significant transport describes a situation where a combination of pollution from the upwind basin and the basin where the violation occurs are both necessary to produce an exceedance, and inconsequential transport describes a situation where transport of pollution did not significantly contribute to any violations.

Class I areas that are located in air basins that receive transported pollutants are listed in Table II-16. Violations of the ozone standard did not necessarily occur within the listed Class I areas, but only within the air basin in which a given Class I area was located.

b. Transport to the Sierra Nevada

Significance of Transport. Transport from the SFBAAB, the broader Sacramento area and the SJVAB is of particular concern to PORE and also to several Class I areas in the Sierra Nevada, including Desolation Wilderness, Mokelumne Wilderness, and Emigrant Wilderness in the MCAB, and Ansel Adams Wilderness, John Muir Wilderness, SEKI, YOSE, and Domeland Wilderness in the SJVAB (YOSE lies within both air basins).

CARB staff analysis (CARB 1996a) of transport from the broader Sacramento area and the SJV to the central part of the MCAB showed that the transported pollutants overwhelmed local source emissions and were the cause of violations in the western part of those counties, where monitoring sites are located. Transport from the SF Bay Area to the central MCAB also resulted in significant contributions to pollution levels. Both Desolation Wilderness and Mokelumne Wilderness are located on the eastern side of the affected area; however, monitoring data from within their boundaries are unavailable. The CARB did not analyze possible transport from the San Francisco Bay area and the broader Sacramento area into the southern portion of the MCAB, which would include Emigrant Wilderness, Ansel Adams Wilderness, and YOSE, or into the southern SJVAB, which would include SEKI.

Transport from the SJV into the southern portion of the MCAB potentially impacts YOSE, Ansel Adams Wilderness and Emigrant Wilderness. The CARB analysis indicated that violations in the southern portion of the MCAB were primarily caused by transport from upwind areas, and particularly from the SJV. Several of these violations were found at sites within YOSE and within 20-50 km of Ansel Adams and Emigrant Wilderness areas (there are no monitoring sites within those wilderness areas). Other studies document transport from the SJV into the southern Sierra Nevada (see later discussion), but such transport is considered to be intrabasin by CARB.

Summer Wind Flow Patterns. Current understanding of transport patterns indicates that winds during summer typically flow from the northwest to the southeast (up-valley) through the SJV (Roberts et al. 1990, 1995). In broad, qualitative terms, pollutants travel from the San Francisco Bay area into the SJV, predominantly with wind flow through the Carquinez Strait in the northern part of the Bay area, the Altamont Pass in the East Bay, and the Santa Clara Valley and Pacheco Pass in the South Bay (Roberts et al. 1990, 1995). Typical northwest airflow also brings pollutants from the broader Sacramento area into the SJVAB and MCAB. Pollutants that have been transported into the SJV, along with fresh emissions from sources within the SJV, are transported further up the valley.

Table II-16. Transport routes and importance of the transport (source: CARB 1996a).

Class I Areas Within the Downwind Air Basin*	Transport Route	Transport Characterization**
Kings Canyon NP Sequoia NP Yosemite NP John Muir W Ansel Adams W Dome Land W	San Francisco Bay Area to San Joaquin	O,S,I
Yosemite NP Desolation W Mokelumne W Emigrant W	San Francisco Bay Area to Mountain	O
Yosemite NP Desolation W Mokelumne W Emigrant W	Broader Sacramento Area to Mountain	O
Yosemite NP Desolation W Mokelumne W Emigrant W	San Joaquin to Mountain	O
Agua Tibia W	Mexico to San Diego	O,S,I
San Rafael W	San Joaquin to South Central Coast	S,I
Joshua Tree NP	South Coast to Mojave Desert	O,S
Lassen Volcanic NP Thousand Lakes W Caribou W Yolla Bolly W	Broader Sacramento to Upper Sacramento Valley***	O,S
Kings Canyon NP Sequoia NP Yosemite NP John Muir W Ansel Adams W Dome Land W	Broader Sacramento to San Joaquin	S,I
Point Reyes NS	Broader Sacramento to San Francisco Bay Area	S,I
Joshua Tree NP	San Joaquin to Mojave Desert	O
San Rafael W	South Coast to South Central Coast	S,I

Table II-16. Continued.

Class I Areas Within the Downwind Air Basin*	Transport Route	Transport Characterization**
San Gabriel W Cucamonga W San Gorgonio W	South Central Coast to South Coast	S,I
Pinnacles NM Ventana W	San Francisco Bay to North Central Coast	O,S
Pinnacles NM Ventana W	San Joaquin to North Central Coast	S
Hoover W	San Joaquin to Great Basin Valleys	O
San Rafael W	California Coastal Waters to South Central Coast***	S

* Most cases of intra-basin transport are not included in this table. Intra-basin transport from urban areas to Class I areas within the SoCAB includes the Agua Tibia, Cucamonga, San Gorgonio, San Gabriel, and San Jacinto wilderness areas.
** O-overwhelming, S-significant, I-inconsequential; see text for further description
*** Transport within the same air basin

Two important meteorological features significantly affect pollutant transport within the SJV. First, a low-level (100 to 1000 m) nocturnal jet forms in the SJV nearly every summer night (~2100 to 0500 hours), with mean velocities of ~ 7 to 10 m/s from Stockton to Fresno and ~ 5 m/s in Bakersfield (Smith et al. 1981). The jet facilitates rapid overnight movement of daytime pollutants in aloft layers. The jet's velocity is lower at Bakersfield than in the northern and central SJV because blockage by the Tehachapi Mountains and the southern Sierra and Coast Range decelerates the winds.

The second key feature is known as the Fresno Eddy, which is an elongated cyclonic (counterclockwise) circulation between Fresno and Delano centered approximately along or west of Highway 99 (Roberts et al. 1995). The eddy develops near sunrise (~0300 to 0700 hours) on most summer mornings, peaks about 9:00 a.m., and persists as late as about noon (Roberts et al. 1995). The significance of the eddy is that it recirculates pollutants around the SJV, distributing them throughout the southeastern portion of the valley. The eddy transports fresh emissions and carryover-ozone from the Fresno area to the eastern side of the SJV, north and east of Fresno. Daytime winds carry polluted air masses into the foothills and higher elevations.

One of the principal mechanisms for transport into the central and southern Sierra Nevada is an afternoon upslope wind flow, which occurs as radiation heats the westward-facing mountain slopes and the overlying air rises. By early evening, cooling of the slopes reverses the flow (though usually not to the same extent as the afternoon upslope flow), allowing recirculation of pollutants. Upslope flows affecting YOSE and SEKI were studied during the 1990 San Joaquin Valley Air Quality Study (SJVAQS) ozone episodes of July 27-30 and August 3-6 (Lehrman et al., 1994). Radar profilers, or rawinsondes, provided vertical profiles of winds at three valley locations: Mariposa Reservoir, ~60 km southwest of Yosemite Valley, Reedly, ~ 50 km west of

the SEKI boundary, and Visalia, ~ 50 km west of the southern portion of SEKI. The durations of the upslope flows, and the fraction of the total air flow between ground level and 2000 m above ground level that was diverted upslope, are listed in Table II-17. Substantial portions of the air masses at each of the three valley locations were diverted upslope. The durations of the upslope flow, and the wind speeds, were sufficient to transport air from the valley sites into YOSE and SEKI.

Ewell et al. (1989a) examined meteorology and aerosol concentrations during a ten-day intensive study at three sites in SEKI in August 1985. Winds moved up-valley (to the south) and upslope (to the east) during the day and down-valley (north) and downslope (west) during the night. Ash Mountain, at the lowest elevation of the sites studied, showed the highest levels of upslope and downslope wind flow from the SJV. Afternoon northwesterly winds were linked to strong upslope flows at Ash Mountain. However, by late evening (2300 PDT), northwesterly

Table II-17. Characteristics of upslope air flows at Mariposa Reservoir, Reedly, and Visalia during the 1990 SJVAQS (source: Lehrman et al. 1994).

Location	Date	Onset of upslope flow (PDT)	Duration (hours)	Maximum upslope component (m/s)	Diverted flow (percent)
Mariposa	July 27	10	10	9	60
	July 28	12	4	2	40
	July 29	10	6	4	60
	August 3	10	7	3	40
	August 4	10	10	3	40
	August 5	10	5	4	60
	August 6	10	8	3	60
Reedly	July 27	10	11	5	70
	July 28	12	11	4	50
	July 29	11	8	2	40
	August 3	10	8	3	40
	August 4	12	8	2	50
	August 5	12	6	4	40
	August 6	11	10	2	40
Visalia	July 27	10	6	5	60
	July 28	10	8	3	40
	July 29	09	8	2	40
	August 3	15	2	2	-
	August 4	10	4	3	50
	August 5	not present	-	1	-
	August 6	10	10	2	30

flows were still strong in Fresno, but Ash Mountain experienced downslope flows. As a conceptual model, Ewell et al. (1989) proposed that, at night, pollution-laden air from the valley is met by downsloping clean air from the mountains. By morning, this zone of convergence is compressed into a narrow zone or a "smog front." Subsequently, the morning and afternoon upslope wind flow then carries the smog front into the Sierra foothills.

Winter Wind Flow Patterns. Ventilation of the SJV is poorer during winter than summer (Smith and Lehrman 1996). Peak air flows up the valley are insufficient to transport air from the northern to southern ends of the valley in less than two days (Smith and Lehrman 1996). Maximum mixing heights during periods of impaired air quality are often in the range of 200 to 500 m (Smith and Lehrman 1996, Lehrman et al. 1998). Valley-wide episodes of high PM occur when ventilation is poor and PM typically builds up over several days (Smith and Lehrman 1996). Such PM episodes are associated with high pressure over the southwestern U.S. and restricted vertical mixing (Lehrman et al. 1998), warm temperatures aloft, weak or offshore pressure gradients, net transport speeds of less than 1 m/s, and maximum transport distances of about 25 to 30 km before flow reversals occur (Smith and Lehrman 1996). During winter stable conditions, widespread and nonuniform dispersion of tracer material occurs over areas exceeding 50 by 70 km after 24 hours, with little evidence of interaction between the northern and southern SJV (Smith et al. 1981). Periods of poor air quality are terminated by the passage of frontal systems (Blanchard et al. 1999).

PM data suggest that monitoring locations situated at elevations of 500 to 1000 m in the Sierra Nevada are frequently above the mixed layer, and are thus effectively decoupled from the higher winter PM concentrations occurring in the SJV (Blanchard et al. 1999). PM concentrations at the CADMP monitoring locations at Bakersfield (BA), Sacramento (SA), Sequoia (SE), and Yosemite (YO) show seasonal maxima at Sequoia and Yosemite during summer months, whereas maxima at Sacramento and Bakersfield occur during winter (Figure II-19).

Tracer Studies. Tracer studies are useful for quantifying dispersion characteristics of plumes, providing empirical data for evaluating trajectory models, characterizing atmospheric transport, and conducting material balances for use in quantitative source apportionment. Planned tracer releases, using inert, nondepositing gases, such as sulfur hexafluoride (SF_6) or perfluorocarbons, and tracers of opportunity can be used. The accuracy of the results is dependent upon the mass of tracer release, the density and configuration of the sampling network, and the quality of sampling and chemical analysis (Rappolt and Teuscher 1995). Tracer experiments are typically of a few days duration at most, so conclusions are temporally limited to the periods studied. Generally, tracers do not provide much information on plume height and vertical diffusion, since nearly all tracer experiments are conducted using surface-level sampling.

Atmospheric transport studies are logistically less demanding than other types of tracer applications, since documentation of tracer locations and travel times typically suffices to meet the objectives of such studies (Rappolt and Teuscher 1995). Caution must be exercised in inferring transport distances, because the deposition rates of the pollutants of interest exceed those of the tracers by many orders of magnitude. Planned tracer releases have been used to qualitatively track the transport of pollutants from the Los Angeles metropolitan area into the Mojave Desert (Reible et al. 1982), from a power plant south of Las Vegas into Arizona (Green 1998), within the Los Angeles area (Horrell et al. 1989), and from sources near ground level compared with tall stacks (England et al. 1989). During the 1990 SJVAQS, perfluorocarbon tracer releases revealed transport from the San Francisco Bay area approximately 250 km down the length of the SJV (Rappolt and Quon 1995). The SJVAQS operated 72 tracer monitoring

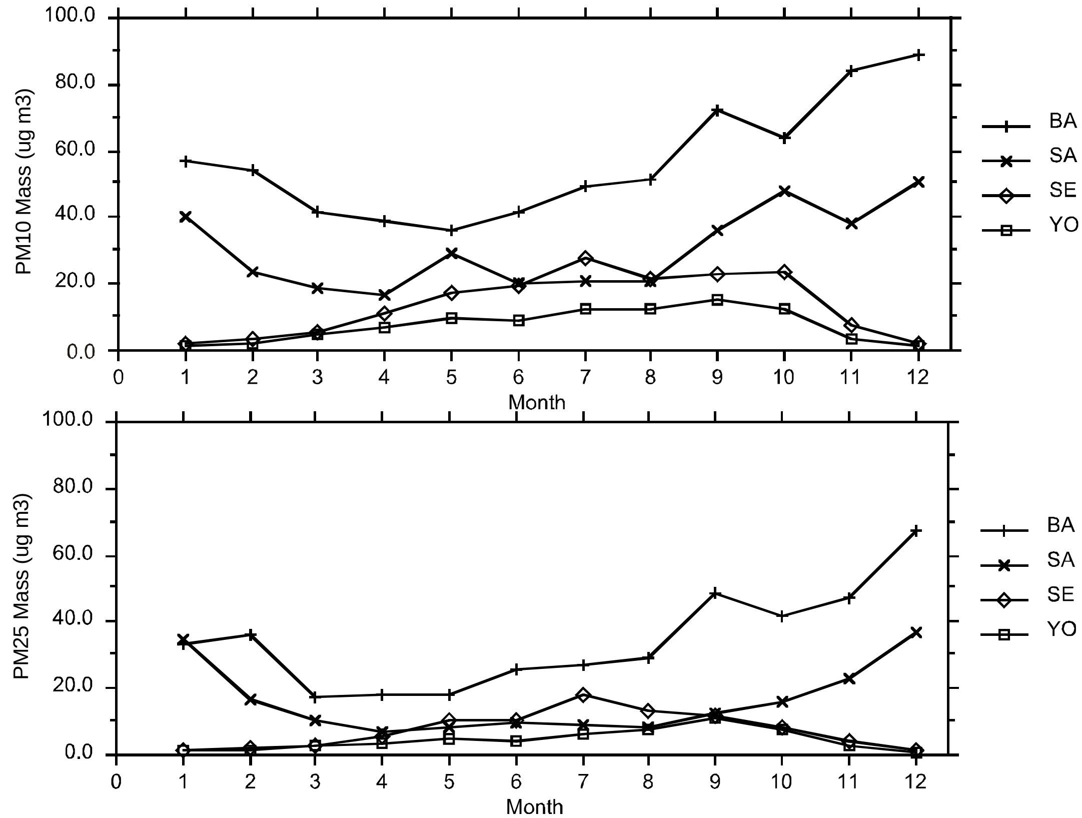


Figure II-19. Monthly-average PM_{10} and $PM_{2.5}$ concentrations at the CADMP monitoring locations at Bakersfield (BA), Sacramento (SA), Sequoia (SE), and Yosemite (YO). Data were collected as two 12-hour samples once every six days from 1989 through 1994. Units are $\mu g/m^3$.

sites within a domain of approximately 150 by 250 km (Rappolt and Teuscher 1995), which provided a site density sufficient for revealing atmospheric transport distances, transport times, and the relative magnitudes of transported versus fresh emissions (Smith and Lehrman 1995).

Tracer studies were conducted in central California during the summer and fall of 1990 to determine the fate of pollutants released in Pittsburgh, San Jose and Hayward in the San Francisco Bay Area (Tracer Technologies Inc. 1991) and from Stockton, Fresno, and Bakersfield in the SJV (Tracer Technologies Inc. 1992). The ambient samples showed the tracers traveling from the Bay area through the eastern SJV into Bakersfield (90% of released tracer was recovered). Tracer plumes from Fresno and Bakersfield were transported east, with the tracer from Fresno found in Cedar Grove and Onyx (southern Sierras) and the tracer from Bakersfield transported straight east. Tracer concentrations identified several transport pathways from the SJV into the Sierra: Merced River Valley into YOSE; San Joaquin River Valley into Wawona (YOSE), Mamouth Pool Reservoir, and Dinkey Creek (Central Sierra Nevada); Kings Canyon into Cedar Grove and Giant Forest (SEKI); Kaweah River Valley into Ash Mountain and Giant Forest (SEKI); and Kern River Canyon into Democrat Station and Lake Isabella (southern Sierras).

Tracer dilution rates were calculated to evaluate the relative impact of pollution transported from urban areas into the Sierran sites on a summer (August) and a fall (October) day (Tracer Technologies 1992). The results showed a different pattern of distribution for each of the two days. The August test results showed that Stockton releases were predominantly responsible for tracer detected in the Sierra north of Fresno; Fresno releases dominated tracer found in the Sierra at one site 48 km north of Fresno, two sites 48 km south of Fresno and one site at the far southern end of the SJV; and Bakersfield releases dominated tracer found at one Sierra site 80 km north of, and one site due east of, Bakersfield. The October test results showed that Stockton releases were responsible solely for tracer found due east of Stockton and approximately 160 km to the southeast of Stockton on the eastern side of the Sierra; Fresno releases were predominantly responsible for tracer found at Sierran sites ranging from 80 km north of Fresno (at YOSE) through all sites south of YOSE on the western slopes, and at the southern end of the SJV. The only exception was one site to the southeast of Bakersfield where Bakersfield tracer dominated (Fresno tracer dominated at one site just 32-48 km due east of Bakersfield). The August and October tracer results, together with the July tracer releases conducted as part of the SJVAQS, indicated that northern releases (Stockton or Pittsburgh) can influence Sierran sites from YOSE southward. Fresno releases influence Sierran sites to the east and south, and under stagnant conditions, to the north. The available data indicated that Bakersfield releases impacted the Sierra to the east and southeast, but did not travel east of Ash Mountain.

Mass balance calculations were done for the October, 1990, tracer releases to determine how much of a particular tracer release remained in the study area (Tracer Technologies 1992). The TAAPS network had only 32 samplers for an area of 116,000 km^2, so that mass balance calculations were limited by the density of the network. Despite the limitation, during the October test, 85% of the tracer mass was accounted for 56 hours after release.

To determine how typical the meteorological conditions were (Tracer Technologies 1992), weather and ozone levels for the entire season (July-October 1990) were compared to the release days. Including the test period, there were 9 days that were similar to the August test days and 37 days similar to the October test days.

Back Trajectory Analyses. Based on data from August 5 and 6, 1990, back trajectories were calculated from several receptors in the SJV and the Sierra for times of peak surface ozone (Blumenthal et al. 1997). The calculations were made using a two-dimensional trajectory model with gridded wind fields; trajectories were calculated for surface flows and two flow regimes

aloft (at 400 m agl and 1000 m agl). Table II-18 describes the results of these back trajectories. Trajectories showed transport into the SJV, particularly into Fresno and Crows Landing, from the SF Bay Area and the Sacramento Area. Transport was also evident into the Sierra Nevada.

Table II-18. Back trajectories in the SJV and Sierra Nevada, showing wind flow source for three different elevations at 24 hours, unless noted (source: Blumenthal et al. 1997).

Receptor	Surface Flow Source	400 m agl Flow Source	1000 m agl Flow Source	Date
Fresno (SJV)	SF Bay	Sacramento area	west of Sacramento	August 5
Kern River Canyon (southern Sierra)	San Luis Obispo	Modesto, SF Bay Area	not calculated	August 5
Kings River (central Sierra)	Kings Canyon*	SF Bay Area	west of Sacramento	August 5
Academy (east of Fresno)	Kings Canyon	Modesto	west of Sacramento	August 6
Mariposa (central Sierra)	Santa Cruz	west of Sacramento	above mixing ht.	August 6
Kern River Canyon (southern Sierra)	Bakersfield	Bakersfield (between Fresno and Merced 48 hr earlier)	Bakersfield (between Fresno and Merced 48 hr earlier)	August 6
Crows Landing (SJV)	SF Bay Area	west of Sacramento (8 hr. earlier)	east of Sacramento (8 hr. earlier)	August 6

c. Transport to the Southeast Desert

Wind Flow Patterns. During summer, the southern part of the Mojave Desert (south of the city of Mojave) is affected by air flow from the SoCAB (Lehrman et al. 1999). Flows from the SJV influence the area from Mojave northward (Lehrman et al. 1999). The relative magnitudes of the flows passing from the SJV into the Mojave Desert through Walker Pass, Tehachapi Pass, or across the southern Sierra are not known.

Tracer Studies. Some regional haze episodes in the vicinity of the Grand Canyon have been qualitatively linked to transport from southern California using a tracer of opportunity, methylchloroform (White et al. 1999), thus indicating the potential impact of southern California's emissions on wilderness areas in California's southeast desert. Planned tracer studies using releases from the San Fernando Valley have shown the influence of that area on the eastern edge of the San Gabriel Mountains (Reible et al. 1982).

Tracers released from the SJV during studies conducted from 1979 to 1992 (see earlier discussion) have been detected at Mojave, China Lake, and Edwards Air Force Base in the Mojave Desert. Because those sites were the only sampling locations, subsequent dispersion of the air masses from the SJV are not well understood (Lehrman et al. 1999). Tracer released from Tehachapi Pass was transported east and north of the eastern side of the pass (i.e., Barstow, Baker, China Lake), but seldom appeared as far south as JOTR (Green 1998).

III. JOSHUA TREE NATIONAL PARK

A. GENERAL DESCRIPTION

Joshua Tree National Park (JOTR) is located along the East-West Transverse Ranges of the Little San Bernardino Mountains. The southern boundary of the park lies along the base of these mountains on the northern edge of the Coachella Valley. The northern boundary of the park is the Morongo Basin. The park includes 321,332 ha, of which 240,186 ha are designated wilderness. In 1950, the original monument boundary was revised and the park was reduced in size; almost 121,410 ha that were known to contain significant mineral reserves were deleted from the park. More recently, 94,700 ha were added to the original Joshua Tree National Monument and its status was changed from national monument to national park.

The East-West Transverse Ranges include examples of both the Mojave and Colorado Desert ecosystems. Elevation ranges from near sea level in the Pinto Basin to 1,829 m at Inspiration Peak, which provides a compressed transition zone between the two desert types. The park contains early Pinto cultural sites and evidence of other prehistoric and historic Native American cultures. There are also sites associated with Euro-American goldmining, homesteading, and subsistence cattle ranching. The surrounding land use has changed dramatically since creation of the original national monument. There are subdivisions, utility corridors, mining sites, military facilities, and agricultural activities, in some cases right along the border of the park. Changes in the desert adjacent to the park have been dramatic during the last half-century. The park is rapidly becoming an ecological island surrounded by lands clearly impacted by human activities (Joshua Tree National Park 1999).

The purpose of the park is to preserve and interpret the biologically diverse examples of Mojave and Colorado Desert ecosystems; to preserve and interpret sites, structures, and objects associated with occupation by prehistoric, historic, and contemporary Native American groups, miners, and subsistence cattle ranchers; and to provide visitors the opportunity to experience and enjoy the natural and cultural resources of the park through compatible recreational activities (Joshua Tree National Park 1993).

JOTR in some ways provides a regional park for the Los Angeles metropolitan area, largely because of the 160-km distance to the city. Most visitors come from the Los Angeles area and visitation is particularly high on weekends. Total annual visitation has been rising steadily and is now over 1.2 million people per year. This trend is expected to continue with population expansion in the Los Angeles Basin. The Los Angeles County population is close to 9 million people and other counties within 160 km of the park add an additional 7 to 8 million people.

In 1994, JOTR was included as part of a Biosphere Reserve. These are set aside by the United Nations Man and the Biosphere Program (MAP), which is an international program of scientific cooperation that deals with interactions between people and the environment throughout all geographic and climatic areas of the world. The purpose of the Biosphere Reserves is to establish a network of protected samples of the world's major ecosystem types. The Colorado and Mojave Deserts Biosphere Reserve was created to include JOTR, Death Valley National Monument, Anza-Borrego State Park, and the Philip L. Boyd Deep Canyon Reserve.

JOTR is rich in archaeologic and historic resources. Archaeological resources include village sites, rock alignments, rock art sites, and sacred and ceremonial sites. Most are located in the Wonderland of Rocks area within JOTR. Historically, the monument reflects the cattle and ranching period and the mining period. Six sites representative of mining and ranching operations are on the National Register of Historic Places, and many other sites in JOTR have potential for National Register status. Much of JOTR has been affected by past human land uses.

There are more than one thousand known mining disturbance areas in the park, as well as hundreds of km of old roads.

The world's largest landfill has recently been proposed for the Eagle Mountain open pit iron mine site, which is located 2.5 km from the park's southeast corner. The proposed landfill could have significant impact on park resources, including air pollution, dust fallout, and visibility degradation.

Baseline information on natural resources is highly fragmented, incomplete, and largely outdated (Joshua Tree National Park 1993). One of the most significant resource values in JOTR is its spectacular vistas. Visibility of scenic features within the park and clear long distance vistas outside the park are considered to be primary attractions of JOTR and essential to visitors' enjoyment of the park. The quality of some views has diminished in past decades because of a deterioration of air quality. National Ambient Air Quality Standards for ozone have been exceeded in the park, especially during summer months (Joshua Tree National Park 1995).

1. Geology and Soils

The park's major scenic elements are largely a consequence of two rock formations. The white-colored monzogranite resulted from a Mesozoic granitic intrusion. It has eroded to form the dramatic rock formations found in the valley and the Jumbo Rocks area. The darker rocks are composed of Precambrian Pinto gneiss. Many of the outstanding geological features within the park have resulted from repeated uplift, lava flows, and erosion. About a hundred million years ago, the monzogranite occurred as a molten mass that cooled and crystalized several kilometers underground. During the crystallization process, the mass contracted slightly and formed countless joints or fractures within the rock. This mass of rock rose towards the earth surface because it was less dense than the rock surrounding it. At the same time, older layers of overlying Pinto gneiss were eroding away. The dark Pinto gneiss still covers many of the high mountain areas in the park, while the younger white, pink, and tan monzogranite is found as rock structures extending up along the sides of these mountains and protruding above the alluvium-filled intermountain basins and plateaus. Depending on whether the joints and cracks are predominantly vertical or horizontal, rock formations consist of columns and spires or horizontally stacked pancake or loaf structures.

The western part of the park contains mountainous areas with peaks that extend to over 1,500 m elevation, as well as medium-elevation plateaus and valleys. To the east, the plateaus drop off into the large Pinto Basin. Large alluvial fans radiate outward from the canyon mouths throughout the park.

The area around the park is tectonically active. The Pinto Mountains Fault runs along the north border of the park and through the Oasis of Mara. Seismic activity in the park is high because of the fault zones in the vicinity, which include the San Andreas Fault located to the west. The oasis at Cottonwood Springs was formed as a consequence of fault activity, as was the Oasis of Mara.

Most soils in the park today are poorly developed. In the eastern portion, soils are mostly alluvial, lacking any true soil structure. They consist largely of granitic fill, ranging from boulders to gravel and coarse sands that were deposited by drainage systems.

The highland areas are composed mostly of complex granitic rock materials (Figure III-1), including quartz monzonite, granodiorite, and gabbro. There are also metamorphic rocks in the highland areas of Paleozoic and Precambrian age, especially Precambrian Pinto gneiss and some

(*Felis rufus*) and badgers (*Taxidea taxus*) occur throughout the park. Ringtails (*Bassariscus astutus*) are rare in rocky areas. Three species of woodrat also occur: the desert woodrat (*Neotoma lepida*), white throated woodrat (*N. albigula*), and dusky footed woodrat (*N. fuscipes*; Van Gelder 1982). A bat photographed in 1998 by Chris Holbeck (Physical Sciences, JOTR) in an abandoned mine within the park was identified as a Townsend's big eared bat (*Plecotus townsendii*). The presence of ten other bat species has been confirmed within the park, including the spotted bat (*Euderma maculatum*).

Several species of amphibian are known from the region around JOTR, but only the red spotted toad (*Bufo punctatus*), California toad (*B. boreas halophilus*) and California treefrog (*Hyla cadaverina*) have been reported in the park. There are 23 species of snake reported in the park, including 5 species of rattlesnake. A large number of bird species live in or migrate through the park, which is adjacent to a major migratory flyway in the Coachella Valley. More than 270 different bird species have been reported.

Wildlife monitoring is conducted for desert tortoise (*Gopherus agassizii*), big horn sheep, and rare species of bat. Monitoring has focused on these species because of their rare, sensitive, or threatened status.

JOTR represents the largest protected area that contains habitat of the Mojave Desert population of the desert tortoise, which was federally-listed as threatened in 1990. Desert bighorn sheep were provided state protected status because of habitat loss and diseases introduced by domestic sheep. Although the species is relatively undisturbed within the park, the importance of genetic linkages with distant populations is not well understood.

B. EMISSIONS

JOTR is located on the boundary of two air basins, Mojave Desert and Salton Sea. The eastern edge of the heavily populated SoCAB lies approximately 50 km to the west of the park. Emissions from portions of Riverside and San Bernardino Counties within the Mojave Desert and Salton Sea air basins, as well as other counties within 140 km of JOTR, are shown in Table III-1. As previously discussed, pollutant transport occurs from the SoCAB to the Mojave Desert air basin; thus, counties within the SoCAB that are within 140 km of JOTR are included in the

Table III-1. 1995 Emissions from counties within 140 km of JOTR. (Source: CARB Almanac 1999b; SO_x from CARB Emissions Website 1999a.) Units are 1000 tons/year.

County	NO_x	ROG*	PM_{10}	CO	SO_x
Riverside [1] (Desert)	1.1	0.7	1.1	5.5	0.0
Riverside [1,2] (Salton Sea)	13.9	10.6	14.2	71.5	0.4
San Bernardino [1] (Desert)	59.5	19.3	85.4	118.3	4.4
San Bernardino [1,2] (S. Coast)	52.9	45.3	24.8	249.3	1.5
Riverside[1,2] (S. Coast)	53.7	45.3	55.5	280.0	1.5
Los Angeles [1,2] (S. Coast)	264.6	268.6	77.0	1565.5	22.6
Orange [2]	71.9	90.2	24.1	537.3	1.8

* Reactive organic gases
[1] Portion of the county in the air basin
[2] County (or portion) is in adjacent air basin in an area where transport between air basins occurs

table. The principal species of concern are ozone precursors (NO_x and ROG) and PM. SO_2 emissions are not high. With the exception of PM, emissions from Los Angeles County are much higher than emissions from the desert immediately surrounding JOTR. Because regulatory programs have reduced both emission levels and ambient pollutant concentrations within the SoCAB (see Section IIB), the levels of pollutant transport should also be declining; however, confirmatory data are not available. With one exception (Twenty-nine Palms), point sources that emit at least 100 tons/year of ROG, NO_x, PM_{10}, or SO_2 in the Mojave Desert air basin portions of Riverside and San Bernardino counties are located near communities that are not adjacent to JOTR (Victorville, Barstow, Trona, Needles, Blythe; Figures II-3 through II-6). Within the portions of Riverside and San Bernardino Counties in the Mojave Desert and Salton Sea air basins, stationary sources accounted for 13% of ROG emissions, 44% of NO_x emissions, and 13% of PM_{10} emissions in 1996 (CARB 1998b).

An inventory of in-park emissions has recently been compiled by the NPS-Air Resources Division. The results are presented in Table III-2.

Table III-2 . Summary of 1998 mobile source emissions (tons/yr) at JOTR.

Activity	Particulates	Sulfur Dioxide	Nitrogen Oxides	Carbon Monoxide	VOCs	HAPs[1,2]
Mobile Source Emissions						
Road Vehicles						
Visitor Vehicles	13.13	0.00	6.73	91.98	13.21	–
NPS/GSA Road Vehicles[2]	–	–	–	–	–	–
Concessioner Vehicles[2]	–	–	–	–	–	–
Vehicle Emission Subtotal	13.13	0.00	6.73	91.98	13.21	–
Nonroad Vehicles						
NPS Nonroad Vehicles [2]	–	–	–	–	–	–
TOTALS	13.13	0.00	6.73	91.98	13.21	–

[1] Hazardous air pollutants, based on the list compiled by EPA
[2] Data not available at this time

C. MONITORING AND RESEARCH ACTIVITIES

1. Air Quality

Ozone, SO_2, and fine particulate have been monitored within JOTR; PM_{10} is sampled at one location within 20 km of the park boundary (Table III-3). In addition, a biomonitoring garden has provided data on ozone effects on some of the plant species. A CASTNet site is located within the park for monitoring dry deposition. Deposition is not monitored directly, but is rather calculated from ambient concentration measurements. An NADP wet deposition site was recently installed in the park.

Table III-3. Air quality monitoring at JOTR.		
Species	Site within park	Site within 50 km
Ozone, hourly	CASTNet	
Ozone, passive	NPS*	
SO_2		
PM_{10}	IMPROVE*	
$PM_{2.5}$	IMPROVE*	
Wet deposition	NADP*	
Dry deposition	CASTNet	
Visibility		
* New site		

a. Wet Deposition

Blanchard et al. (1996) estimated annual and 10-year wet deposition rates throughout California by interpolating the observations from all NADP/NTN, CADMP, and special-studies monitoring locations for the period 1985 through 1994; they also estimated interpolation uncertainties. Upper-bound deposition values for unmonitored areas may be estimated as the sum of the interpolated values plus twice the interpolation uncertainty (i.e., mean plus two standard deviations). The results indicated that the 10-year mean total wet N deposition rates were less than 3 (+/- 3, at 2 sigma) kg/ha/yr (as N) throughout the state. Wet S deposition was less than 1.3 (+/- 1, at 2 sigma) kg/ha/yr (as S) throughout the state.

At the nearest CADMP sites (San Bernardino and Victorville, ~80 km west and northwest, respectively), wet S deposition ranged from 0.2 to 0.3 kg/ha/yr as S (Victorville) and 0.3 to 1.5 kg/ha/yr (San Bernardino) during the period 1986 through 1994 (Blanchard et al. 1996). The annual NO_3^- deposition rates were in the range of 0.3 to 0.8 kg/ha/yr as N (Victorville) and 0.6 to 2.1 kg/ha/yr as N (San Bernardino). The annual NH_4^+ deposition rates were in the range of 0.2 to 0.7 kg/ha/yr as N (Victorville) and 0.7 to 3.9 kg/ha/yr as N (San Bernardino). The multi-year mean total wet N deposition rates were 1.0 kg/ha/yr (Victorville) and 3.0 kg/ha/yr (San Bernardino; Blanchard et al. 1996). During the period 1984 through 1990, the annual-average H^+ concentration in precipitation at Victorville and San Bernardino was 6.0 µeq/L and 3.5 µeq/L (pH 5.22 and 5.46), respectively (Blanchard and Tonnessen 1993).

b. Occult/Dry Deposition

The CASTNet dry-deposition monitoring site began operating February 16, 1995. The monitoring instrument measures ambient concentrations of gases and particles, and EPA uses a computer model to calculate the dry-deposition rates from the measurements. The first calculations of dry-deposition rates were released by EPA in November 2000. For the years 1995 through 1998, the calculated annual dry-deposition rates of N and S ranged from 3.4 to 4.2 kg N/ha/yr and 0.42 to 0.52 kg S/ha/yr, respectively. When combined with the wet deposition measurements from the nearest NADP/NTN site (located at Tanbark Flat, 126 km west of the dry-deposition monitor), the data indicate that the annual total deposition rates of N and S ranged from 5.2 to 7.5 kg N/ha/yr and 1.1 to 2.0 kg S/ha/yr, respectively, over the period 1995 through

1998. The average total N and S deposition rates over the four-year period were 6.2 kg N/ha/yr and 1.5 kg S/ha/yr, respectively.

c. *Gaseous Monitoring*

Air quality has been monitored in the nearby community of Twenty-nine Palms. An ozone analyzer was initially installed at the Lost Horse Ranger Station in April 1984 and subsequently moved to Black Rock. The monitoring site was upgraded in 1988 with the addition of automated meteorological monitoring and again in 1991 with the addition of an instrument shelter and remote power supply equipment to improve data collection.

At Black Rock, ozone concentrations and exposure fluctuated from year to year from 1992 to 1997 without clear evidence of trend, as shown in Table III-4. The NAAQS for 1-hour maximum ozone was exceeded for all years of reported data at JOTR. The maximum 9AM to 4PM average ranged over this time period from 85 ppb to 109 ppb. SUM60 indexes were well over 100,000 ppb-hours during this reporting period. JOTR thus experiences extremely high levels of ozone, higher than any other Class I park in California and some of the highest in the NPS system. There is therefore a great deal of concern regarding the potential for ozone damage to plants in the park.

Table III-4. Summary of ozone concentrations and exposure from JOTR. Bold-face values of the 3-year average number of exceedances indicate violations of the federal 1-hour ozone standard. (Source: Joseph and Flores 1993, National Park Service, Air Resources Division 2000).

Year	Maximum Daily 1-hour Value (ppbv)	2nd Highest Daily 1-hour Value	Number of Daily Maximum 1-hour Values # to 125 ppb	3-year Average Number of Exceedances	Maximum 9am-4pm Average (ppbv)	Sum06 (ppbv-hour)[a]	Number of Valid Hours of Ozone Measurements
1987	148	147	10	na	104	52,000	4585
1988	134	131	3	na	84	48,000	6220
1989	139	126	2	**5.0**	93	15,000	4606
1990	132	120	1	2.0	92	21,000	5690
1991	135	133	6	3.0	107	65,000	6845
1992	138	131	7	**4.7**	95	57,000	6726
1993	138	121	1	**4.7**	85	16,000	6052
1994	165	147	20	**9.3**	102	72,000	8183
1995	151	148	4	**8.3**	90	45,000	8250
1996	146	139	5	**9.7**	109	69,000	8290
1997	149	142	9	**6.0**	101	65,000	8190
1998	142	138	9	**7.7**	92	43,000	7362
1999	137	127	2	**6.7**	95	64,000	7929

[a] Maximum 8 am - 8 pm 90-day rolling average.

A network of passive ozone samplers provides average ozone values during the summer at JOTR. Table III-5 displays summaries of the available data, which cover six sites in 1998 and three sites in 1997. These data show that some spatial variation does occur in mean ozone concentrations within the park. The 1998 data suggest that higher ozone concentrations can be found toward the northern and western parts of the park (Black Rock, Lost Horse, North Entrance and Keys View), areas potentially more exposed to pollutants transported from the SoCAB. The highest annual mean ozone levels in California national parks were seen at JOTR (see Table II-12). Hourly ozone concentrations measured at the Black Rock site (Table III-5) had the second highest mean ozone among the six passive ozone monitoring locations in 1998.

Table III-5. Summer average hourly ozone concentrations at passive sampling sites within JOTR. (Source: Dr. John D. Ray, National Park Service, Air Resources Division, NPS Passive Ozone website 1999). Units are ppb.

Sample Locations	Elevation (m)	1997	1998	1999
Sunrise Well	152	58.3	57.6	91.8
Black Rock	1231	60.9	65.5	77.7
Lost Horse	1256	56.3	59.2	
N. Entrance	866		63.7	
Keys View	1585		71.4	
Cottonwood	305		56.2	

Maximum and mean SO_2 are reported for JOTR for 1991 and 1992, based on daily averages (Table III-6). SO_2 measurements were discontinued after 1996 due to concerns about their accuracy. The measurements are considered sufficiently accurate to show that the SO_2 concentrations were well below the levels at which plant injury has been documented, ~40 to 50 ppb 24-hour average and 8-12 ppb annual average (Peterson et al. 1992).

Table III-6. Maximum and mean SO_2, from 24-hour-resolution samples at JOTR. Samples are collected every 3-4 days, unless noted. (Source: NPS Air Resources Division). Units are ppb.

SO_2	1988	1989	1990	1991	1992	1993	1994	1995	1996
Maximum	na	na	na	0.43*	0.16*	na	na	na	na
Mean	na	na	na	0.16*	0.08*	na	na	na	na

na Not available
* Less than 50 samples collected for the year

2. Aquatic Resources

JOTR is a desert park and has only one short perennial stream. Other freshwater sources occur as springs, wells, and seeps. Most of the springs flow from fractures and joints in the bedrock. Monitoring data suggest that discharge at some of the springs may be decreasing compared to historical records (Joshua Tree National Park 1995). There are three artificial impoundments within the park, at Barker Dam, Cow Camp, and Keys Lake. These are considered historic features; they were constructed to supply water for early ranching activities.

There is not a comprehensive water management plan for JOTR. Little information is available regarding water quality and quantity in the park or about the water sources that the vegetation and wildlife depend upon. There are concerns about potential hydrological problems within the park, including lowering of the water table at the Oasis of Mara and a reduction in flow of historic springs in some portions of the park. Continued withdrawal of water may result in decreased water levels throughout the Pinto Basin. The quantity of water available at any one site in the park is generally small or intermittent. Water monitoring has been limited to groundwater depth in one of the five oases in the park.

Surface waters that occur in JOTR do not exhibit any of the characteristics normally associated with sensitivity to acidification. In the desert environment of JOTR , water residence time (the time the water is in contact with soils and geologic materials) is normally quite long. This would allow base cations from mineral weathering to be effectively transported to surface waters in sufficient quantity as to neutralize any amount of atmospheric deposition of NO_3^- or SO_4^{2-} that could reasonably be expected to occur in the park. Water quality is therefore not considered to be an important AQRV in JOTR.

3. Vegetation

Limited assessments of potential effects of air pollutants on vegetation in JOTR have not documented any symptoms on plants growing naturally in the field (Temple 1989). However, experimental studies using controlled exposures have identified species that are potentially sensitive to ozone, SO_2, and NO_2 (Thompson et al. 1980,1984a,b; Temple 1989).

In a study completed by Patrick J. Temple in December 1985, data were presented which suggested that skunkbush sumac (*Rhus trilobata*) could be a useful indicator plant for assessing the effects of ozone on the native woody plants of JOTR. A biomonitoring plot (ozone indicator plant garden) was established in 1987 at Lost Horse Ranger Station and plants were surveyed on a routine basis for symptoms consistent with ozone injury. The garden was irrigated, in contrast with the naturally-growing populations in the park. Pathology samples of skunkbush sumac were collected during the 1987, 1988, and 1989 monitoring seasons. Symptoms were found that were consistent with those caused by ozone in laboratory fumigation tests.

In a controlled-exposure study, Thompson (1980) tested the sensitivity of five perennial species and five annual species native to the Mojave Desert and Colorado Desert to SO_2 and NO_2, and found a wide range of responses in terms of visible injury symptoms and growth. Thompson et al. (1984a,b) used controlled exposures, including exposures well above normal ambient concentrations, to determine the relative sensitivity of 49 Mojave Desert species to ozone and SO_2. They found considerable variation in sensitivity, with evening primrose and catseye species having some of the highest levels of injury. Results for individual species studied by Thompson et al. (1980,1984a,b) are not summarized here, but were used to compile the table of species sensitivities for JOTR (Table III-7).

Table III-7. Plant and lichen species of JOTR with known sensitivities to sulfur dioxide, ozone, and N oxides (H=high, M=medium, L=low, blank=unknown).

Scientific Name	Common Name	Sensitivity		
		SO_2	O_3	NO_x
Angiosperms				
Acacia greggii	Catclaw acacia	L	L	
Ambrosia dumosa	White burrobush	L		L
Artemisia dracunculus	Wormwood		M	
Artemisia tridentata	Big sagebrush	M	L	
Atriplex canescens	Fourwing saltbush	L		L
Baileya multiradiata	Desert marigold	H		M
Baileya pleniradiata	Woolly desert marigold	M	L	
Bromus rubens	Foxtail brome	M	M	
Caulanthus cooperi	Cooper's wild cabbage	M	M	
Chaenactis carphoclinia	Pebble pincushion	M	M	L
Chaenactis fremontii	Pincushion flower	M	M	
Chaenactis stevioides	Steve's dustymaiden	M	M	
Chilopsis linearis	Desert willow	M		L
Chorizanthe brevicornu	Brittle spineflower	M	M	
Coreopsis bigelovii	Bigelow's tickseed	M	M	
Cryptantha angustifolia	Panamint catseye	M	M	
Cryptantha circumscissa	Cushion catseye	M	M	
Cryptantha nevadensis	Nevada catseye	H	H	
Cryptantha pterocarya	Wingnut catseye	M	M	
Descurainia pinnata	Western tansymustard	M	M	
Encelia farinosa	Goldenhills	M		M
Eriogonum pusillum	Yellowturbans	M		
Erodium cicutarium	Redstem stork's bill	M	M	L
Eschscholzia parishii	Parish's goldenpoppy	M	H	
Eucrypta micrantha	Dainty desert hideseed	M	M	
Festuca octoflora	Sixweeks fescue	L	L	
Gilia stellata	Star gilia	M	M	
Gutierrezia sarothrae	Broom snakeweed	M		
Helianthus annuus	Common sunflower	H	L	
Langloisia schottii	Schott's calico	M	M	
Larrea tridentata	Creosotebush	M		M
Lepidium lasiocarpum	Shaggyfruit pepperweed	L	L	
Lupinus concinnus	Scarlet lupine	M	M	
Malacothrix glabrata	Smooth desert dandelion	M	H	
Mentzelia albicaulis	Whitestem blazingstar	M	H	
Mimulus guttatus	Seep monkeyflower		L	
Oenothera californica	California eveningprimrose	M	M	
Oenothera claviformis	Browneyes	H	H	
Oryzopsis hymenoides	Indian ricegrass	M		
Pectocarya heterocarpa	Chuckwalla combseed	M	H	
Pectocarya platycarpa	Broadfruit combseed	M	H	

Table III-7. Continued.

Scientific Name	Common Name	Sensitivity		
		SO_2	O_3	NO_x
Perityle emoryi	Emory's rocklily	M	L	
Phacelia campanularia	Desertbells	M	M	
Phacelia crenulata	Cleftleaf wildheliotrope	M		L
Plantago insularis var. fastigiata	Desert Indianwheat	M		M
Rhus trilobata var. anisophylla	Skunkbush sumac	L	H	
Rumex crispus	Curly dock		L	
Sambucus mexicana	Blue elder		H	
Schismus barbatus	Common Mediterranean grass	M		
Sphaeralcea ambigua	Desert globemallow	M	L	
Stephanomeria exigua	Small wirelettuce	M	M	
Streptanthella longirostris	Longbeak streptanthella	M	M	

Temple (1989) tested the sensitivity of four woody perennials (catclaw acacia, desert willow, skunkbush sumac, and Goodding's willow [*Salix gooddingii*]) to controlled exposures of ozone, ranging up to 200 ppbv over 4-hour periods. This study found that skunkbush sumac exhibited foliar injury at concentrations as low as 100 ppbv, while Goodding's willow exhibited foliar injury at 200 ppbv; other species had no injury symptoms. Surprisingly, skunkbush sumac exposed to elevated levels of ozone also had greater growth, but the fact that it had such clear damage symptoms at relatively low exposures indicates high sensitivity and good potential as a bioindicator.

Thus, several plant species that occur at JOTR have been documented as being sensitive to ozone, and ozone levels are very high in the park. Although no ozone damage has been confirmed on plants growing naturally in the field, some damage was found on skunkbush sumac grown under irrigated conditions in the biomonitoring garden. Under normal conditions, most desert plants are probably dormant during summer, when ozone levels are highest. During a wet year or following a strong summer monsoon season, some native species at JOTR might develop ozone injury, but this has not been demonstrated (P. Temple, pers. comm., 2000).

4. Visibility

Air pollution frequently causes impaired visibility in the park, particularly in the summer. In general, winter air quality is better when the prevailing air flows are not from the direction of Los Angeles basin. The park contains several critical desert vistas, including the 360º panorama from the top of Ryan Mountain. Mountain peaks and other geographic features, including Signal Peak 160 km to the south in Mexico and the Salton Sea, are easily seen from vista sites within the park when visibility is good.

As part of the Interagency Monitoring of Protected Visual Environments (IMPROVE) network, visual air quality in JOTR has been monitored using an aerosol sampler and camera. The aerosol sampler ran only for a short period during the 1992 MOHAVE (Measurement Of Haze and Visual Effects) study. The IMPROVE protocol site was located at the Lost Horse Ranger Station and operated from September, 1992 through September, 1993. Due to the special

4. Visibility

No IMPROVE aerosol, optical, or view monitoring has been conducted at JOTR since the conclusion of the MOHAVE intensive study in 1993. However, as part of the IMPROVE Network expansion, a new IMPROVE aerosol sampler was located at JOTR during the winter 2000-2001 season.

We do not recommend any additional visibility research or monitoring at this time.

IV. LASSEN VOLCANIC NATIONAL PARK

A. GENERAL DESCRIPTION

Cinder Cone and Lassen Peak National Monuments were established by presidential proclamation in 1907. Lassen Volcanic National Park (LAVO) was subsequently established by an Act of Congress in 1916, and the existing national monuments were incorporated into the new park. In 1972, 31,965 ha of the park's 43,018 ha were designated as wilderness. The Lassen Wilderness is bordered on the east by the Caribou Wilderness of Lassen National Forest. The park preserves a variety of volcanic landforms, and relatively undisturbed natural resources, including forests, lakes, and streams. LAVO also includes the most extensive, intact network of geothermal resources west of Yellowstone National Park.

The park is located at the southern end of the Cascade Mountain Range, only a few km from the northern terminus of the Sierra Nevada. LAVO is within a one-day drive of two major California metropolitan areas: Sacramento and the San Francisco Bay area. About 85% of LAVO's visitors are from California, with the majority from the Bay area. LAVO is famed for its outstanding geological resources and scenery, including forests, mountain peaks, volcanic features, and numerous lakes.

Elevations range from 1,585 m to 3,190 m at Lassen Peak, the most pronounced peak on the rim of the collapsed volcano, Mount Tehama. Topography is extremely diverse. Volcanic mountain peaks are found in the western half of the park, and there is a lava plateau to the east, a deep valley with steep, glaciated walls in the south, and peaks and ridges of lava modified by erosion in the north (Figure IV-1). Up until the time of the eruption of Mount St. Helens in 1980, Lassen Peak was the only recently-active volcano in the lower 48 states. Volcanic activity is still evident in the form of hot springs at Bumpass Hell, Devil's Kitchen, and Sulphur Works.

Annual visitation ranges approximately between 325,000 and 500,000 people, depending on the date that the park road opens. The road is impassable due to snow at the higher elevations as early as September and remains closed until crews are able to clear it, sometimes as late as mid-July.

The primary land use activities in the region around the park are logging and wood products manufacturing, cattle ranching, agriculture, and tourism. Cultural resources at LAVO are also diverse and include an archaeological record dating back at least 4,000 yrs, ethnographic sites, structures, historic wagon trails, and museum collections. Ten sites in the region have been listed on the National Register of Historic Places.

Air pollution in the Sacramento Valley and surrounding foothill and mountain areas is discernible from vistas in the park on many days of the year. Rapid population growth in the northern Sacramento Valley, especially in the Chico and Redding areas, raises additional concerns about future levels of air pollution and potential impacts on the resources of the park.

1. Geology and Soils

LAVO provides outstanding examples of geological processes and is of unquestioned national geological significance. Lassen Peak is one of the largest plug dome volcanoes in the world. The low broad plateau in the eastern portion of the park was formed largely by andosite and basaltic lava flows. These are the oldest rock formations in the park. About 50,000 YBP, Mount Tehama, a major stratovolcano of the Pacific Rim of Fire volcanic and seismic zone, developed at the southern end of this plateau. By 33,000 YBP, Mount Tehama had collapsed. The caldera was breached and therefore a central lake did not form (as Crater Lake formed in the collapsed caldera of Mount Mazama to the north). Before Mount Tehama collapsed, dacitic plug volcanoes began to form on its northeastern flank. The extrusion of Lassen Peak began about 11,000 YBP. A period of major volcanism began again in the area to the north of Lassen Peak

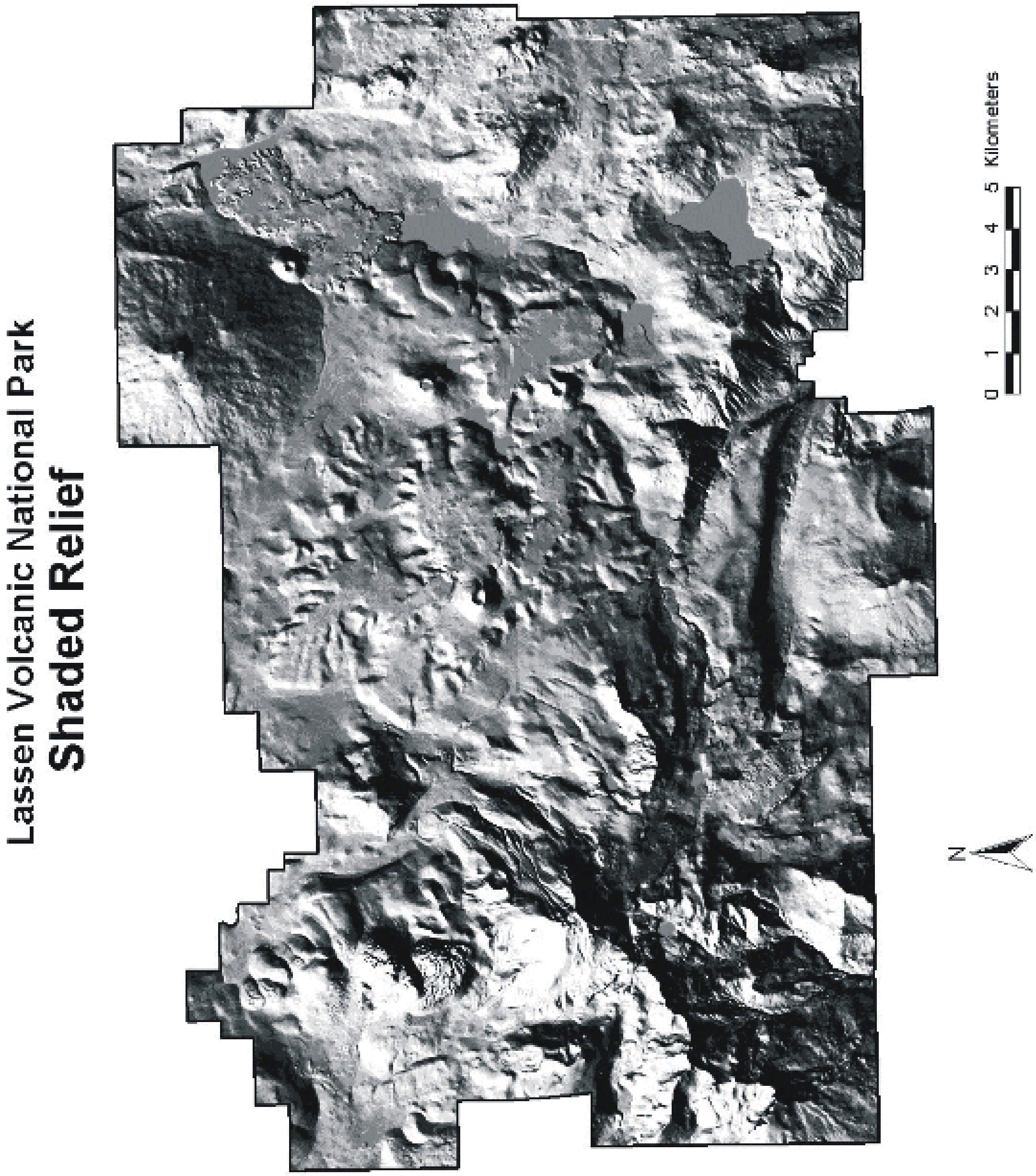


Figure IV-1. Shaded relief map of LAVO.

about 1,200 YBP, in the vicinity of what is now Chaos Crags. The steep northwestern side of Chaos Crags collapsed about 300 years ago in a huge rockfall-avalanche that formed Chaos Jumbles.

Lassen Peak is the most southerly of the volcanic cones in the Cascade Range. Other peaks along the remnants of Mount Tehama's original flanks include Brokeoff Mountain, Mount Diller, Pilot Pinnacle, and Mount Conard. Cinder Cone and Fantastic Lava Flows are more recent volcanic features, having been created during the last thousand years. A large eruption of Lassen Peak occurred on May 19, 1915, during which lava flowed over the northeast and southwest rims of the summit and created an avalanche mudflow which devastated an area about 0.4 km wide and 16 km long. Subsequently, a cloud of super-heated gas and debris burned remaining vegetation in the same area (Rowe 1974). This fan-shaped region is now called the devastated area.

Four types of volcanoes are found within the park: volcanic dome, shield volcano, cinder or tephra cone, and composite volcano. There is extensive evidence of glaciation throughout LAVO, including moraines, glacial erratics, tarns, cirques, and U-shaped valleys. Lassen Peak was extensively modified by glacial action. Lakes formed in some of the cirque basins, including Emerald Lake and Lake Helen. Other lakes of glacial origin include Summit, Feather, Silver, Big Bear, Little Bear, Swan, Echo, and Twin Lakes (Harris and Tuttle 1983).

There are four types of lava found in the park, and there is evidence suggesting that the lava types can cause distribution of certain plant species to vary from the life zones in which they would normally be expected. For example, Emerald and Helen Lakes are located within 0.4 km of each other at approximately the same elevation. The former is located in a basin of Brokeoff andosite and has well developed lake margin vegetation, including a number of species that do not occur at Lake Helen.

Boiling waters, mud pots, and steam vents are reminders of the active volcanism still present at LAVO. Bumpass Hell is the largest geothermal feature in LAVO. It probably lies directly over the major plume of heated water rising above the active magma chamber. Boiling Springs Lake may be the largest body of hot water in the world.

Soils in LAVO are volcanic in origin. They are typically young, stony, and well drained. The soils are also moderately to strongly acidic and contain little organic content. Soil depth varies from more than a meter in some places at lower elevations to virtually non-existent at higher elevations. Soils in the southern part of the park, which exhibit moderate to high erosion potential, include the Chumy, Lyonsville-Jibbs, and Windy series. Lytton and Donner Soil Series are found in the southwest portion of the park. Deeper and more permeable soils are found in the northern part of the park in the Cohasset, Lyonsville-Cohasset, and Jibbs soil series (Lassen Volcanic National Park 1981).

Important resource issues related to geology in LAVO include a lack of basic data on soils, geothermal, and water resources, and potential disruption of geothermal resources due to exploration and development outside of the park.

2. Climate

Summers are typically cool, and winters are cold with heavy snowfall. Mean summer temperatures range from 9 °C in May to 20 °C in July. Mean annual high temperature in winter is about 4 °C and in summer about 27 °C. The average minimum temperature is -6 °C in January and 7 °C in July. Annual average precipitation at Mineral is approximately 135 cm, most of which occurs in the form of snow. The average annual snowfall is about 4 m, but some portions of the park may receive more than 9 m of snow per year. Summers are generally dry, and about 24 cm of precipitation can be expected from May through October (Lassen Volcanic

National Park 1981). Winds are generally from the west and southwest. Southwest winds blow across the lower Sacramento Valley prior to reaching LAVO, and bring air pollutants from that location into the park.

3. Biota

Biological diversity in the park is relatively high, with over 700 species of flowering plants and 250 species of vertebrates. This great diversity is due, at least in part, to the geographical location of the park and the abundance of habitats that occur there. LAVO lies at the crossroads of three biological provinces: the Cascade Mountains to the north, the Sierra Nevada to the south and the Great Basin Desert to the east. About 24 Sierran plant species are at their northern limit in LAVO, and about 15 Cascadian species are at their southern limit (Lassen Volcanic National Park 1981).

There are four principal types of vegetative assemblage in the park: yellow pine forest, red fir forest, subalpine forest, and alpine. The major vegetation types are shown in Figure IV-2. The dominant plant associations in the park are montane and subalpine forests. Yellow pine forests cover about 12% of the park, up to an elevation of about 2,000 m. Jeffrey pine (*Pinus jeffreyi*) is predominant in this forest type, but white fir (*Abies concolor*) is also locally common. Additional tree species include Shasta red fir (*Abies magnifica* var. *shastensis*), sugar pine (*P. lambertiana*), ponderosa pine (*P. ponderosa*), lodgepole pine (*P. contorta*), Douglas fir (*Pseudotsuga menziesii*), and incense cedar (*Libocedrus decurrens*). Open forests of ponderosa pine and Jeffrey pine are found in the Manzanita Lake area.

The most widespread plant community in the park is the red fir forest, which occurs at elevations between about 2,000 and 2,400 m and covers about 47% of the park. Dominant tree species include Shasta red fir, lodgepole pine, and western white pine (*Pinus monticola*). Other tree species present include Jeffrey pine and mountain hemlock (*Tsuga mertensiana*). Red fir occurs in nearly pure stands on wetter sites and is associated with Jeffrey pine on drier sites with shallow soils. A few shrub species are present, including pine mat manzanita (*Arctostaphylos nevadensis*), chinquapin (*Castanopsis sempervirens*), Sierra gooseberry (*Ribes roezlii*), and snowbrush (*Ceanothus cordulatus*). Scattered throughout this forest zone are stands of old growth red fir. This species grows to diameters of 1 m or more and can live more than 300 years. Such old growth forests provide habitat for specialized animal species, including spotted owl (*Strix occidentalis*) and northern flying squirrel (*Glaucomys sabrinus*). Lodgepole pine forests cover about 9% of the park. Most stands have resulted from intense fire. This species is eventually replaced by more shade-tolerant tree species.

Subalpine forests are found above 2,400 m elevation along the rim of the Mount Tehama caldera and the high plateau area. The subalpine forests cover about 8% of the park and include pure stands of mountain hemlock, as well as krummholz stands of whitebark pine (*Pinus albicaulis*) at elevations of up to 3,000 m. These forests generally have relatively low stem densities and often contain a mosaic of tree islands and meadows below treeline. Weather conditions can be extremely severe, with 10 m snowdrifts adjacent to wind-blasted ridges.

The alpine community is characterized by thin to non-existent soils, intense sunlight, desiccating wind conditions and cold temperatures. Water is generally in short supply, either because it is frozen or because of the low water-holding capacity of the thin soils. Alpine plant communities are found at the upper elevations of Lassen Peak, Loomis Peak, Crescent's Crater, and Ski Heil Peak. Distribution of snowpack, which is influenced by wind, aspect, and topography, determines the time available for plant growth as well as soil moisture. A wide range of plant species, many with low, spreading stature, are found in the alpine zone, with distribution and abundance affected by local variation in snowpack and soil moisture.

summer range for two herds of migrating deer: the Cow Creek herd in the north and the East Tehama herd in the south. The East Tehama deer herd of black-tailed deer is the largest migratory herd of deer in California (Lassen Volcanic National Park 1999).

Beaver (*Castor canadensis*) are common in the Warner Valley. Available evidence suggests that they are not native to the park (Fellers 1981) but were transplanted in the vicinity of the park between 1923 and 1949. Beavers are expanding into park streams along the southern boundary and into the Hat Creek drainage in the northern section of the park. Red fox (*Vulpes vulpes*), mountain lion (*Felis concolor*), bobcat (*Felis rufus*), and coyote (*Canis latrans*) all inhabit the park, but are not frequently seen. LAVO is in the range of the Sierra Nevada subspecies of the red fox, a threatened species in California, and there have been many observations of red fox within the park. It is not known, however, whether those observed have been pure breed Sierra Nevada red fox, an introduced red fox, or some hybrid of the two. Black bear (*Ursus americanus*) are commonly seen, and a bear management plan was approved for LAVO in 1992, with the goal of restoring and maintaining a wild, natural free-living black bear population while protecting public safety and minimizing property damage.

A variety of rodent species can be observed within LAVO. These include yellow-bellied marmot (*Marmota flaviventris*), which are common on rocky slopes, and golden mantle ground squirrel (*Spermophilus lateralis*) which are often seen around campgrounds and picnic areas. Three species of chipmunk have been observed, including the lodgepole chipmunk (*Eutamias speciosus*), yellow pine chipmunk (*E. amoenus*), and Townsend's chipmunk (*E. townsendii*). Tree squirrels include Douglas' squirrel (*Tamiasciurus douglasii*) and western gray squirrel (*Sciurus griseus*). Pika (*Ochotona princeps*) are common in talus slope areas at high elevation. Both Nuttall's cottontail (*Sylvilagus audubonii*) and snowshoe hare (*Lepus americanus*) occur, but neither is common. Other ground squirrels include the California ground squirrel (*Spermophilus beecheyi*), which is resident over much of the lower elevation portions of the park, and Belding's ground squirrel (*Spermophilus beldingi*), which is found in the high meadows. Porcupine (*Erethizon dorsatum*) are found mainly at lower elevations. Mustelids within the park include badger (*Taxidea taxus*), mink (*Mustela vison*), and river otter (*Lutra canadenis*; Van Gelder 1982). Both wolverine (*Gulo gulo*) and fisher (*Martes pennanti*) have been previously observed in the park, but neither has been reported recently.

Over 150 species of birds have been recorded in the LAVO area, including the threatened northern spotted owl, peregrine falcon (*Falco peregrinus*), and bald eagle (*Haliaetus leucocephalus*). Other raptors include golden eagle (*Aquila chrysaetos*), osprey (*Pandion haliaetus*), sharp shinned hawk (*Accipiter striatus*), and goshawk (*Accipiter gentilis*). Peregrine falcons utilize the park in late summer and fall during post-breeding dispersal from lower elevations. LAVO has marginal bald eagle nesting habitat because of scarce food supplies and relatively harsh climate during the nesting season. A nest site has been monitored, however, at Snag Lake and nesting success has occurred in 9 of the 18 years of nest monitoring. The bald eagle nest site in the park is one of the highest recorded in California and because of the potential for inclement weather conditions during the early stages of the breeding season, the nest site is considered marginal (Baldridge et al. 1982). The limited fishery resource within the park can probably only support one nesting pair of eagles (Baldridge et al. 1982). A pair of bald eagles also nests outside the park, but forages at Manzanita Lake. Waterfowl are common in park lakes. A small, isolated breeding population of bufflehead, (*Bucephala albeola*), California's rarest boreal duck species, occurs in northeastern California, the majority of which is located within LAVO and Lassen National Forest.

The willow fly catcher (*Empidonax trailii*), a state-endangered species, breeds in LAVO. All 18 species of land birds that were described as decreasing or likely decreasing in the Sierra Nevada Ecosystem Project area are found within the park.

Seventeen species of amphibian and reptile have been reported in the park, and additional species are of probable occurrence there. Recent studies by Gary Fellers and Charles Drost have confirmed the presence of the rough-skin newt (*Taricha granulosa*), long-toed salamander (*Ambystoma macrodactylum*), Pacific treefrog (*Pseudacris regilla*), Cascades frog (*Rana aurora cascadae*), sagebrush lizard (*Sceloporus graciousus*), western terrestrial garter snake (*Thamnophis elegans*), and common garter snake (*Thamnophis sirtalis*). In 1991, intensive searches were made at all 16 sites in LAVO where the Cascades frog had been recorded; only two frogs were found at a single location. Subsequently, approximately 200 sites have been visited within the park. Only one of the sites has produced Cascades frogs (Fellers and Drost 1993). The species may be on the verge of extinction in the vicinity of LAVO, which is the southern extension of its range. It is certain that the Cascades frog is much rarer than it has been in the recent past and the decline has been precipitous over a period of less than 20 years. Fellers and Drost (1993) hypothesized that this decline has been caused by multiple factors, including the presence of non-native predatory fish which have restricted habitat and limited dispersal of frogs, loss of breeding habitat due to an extended drought, and gradual loss of open meadows and associated aquatic habitats. The latter factor may have been associated with fire suppression and/or cessation of cattle grazing.

The park's list of amphibians and reptiles is incomplete. A recent project statement sheet from LAVO suggested that as many as 25% of the species of reptile and amphibian which are likely to be present within the park have not been documented or included on the official park list.

A comprehensive survey of the park's butterfly fauna was conducted between 1984 and 1993. All of the park's major ecological communities were surveyed. Eighty-three butterfly species were recorded during this effort, although none were recorded that are listed as threatened or endangered, and none were endemic to the park. An annotated list of moth species known in LAVO is available at the park. It includes 147 species, representing 27 families of moths.

Because much of the park was carved by glaciation and is isolated from major river drainages, most lakes in LAVO were originally fishless. However, Manzanita, Butte, and Soda Lakes have direct connections to native trout streams and therefore may have had native fish populations. Waters within the park that presumably could support fish populations were altered by stocking. Stocking records available for the park date back to 1928; earlier stocking locations and intensities are not known. From about 1928 to 1965, 41 lakes were stocked on a more or less regular basis. Stocking was reduced to 17 lakes in 1968 and then an average of two lakes were dropped each year until all stocking was stopped by 1980. Based on recent fisherman creel reports and observations by park staff, seven lakes in the park continue to have self-sustaining populations of trout. Species present include brown trout (*Salmo trutta*), rainbow trout (*Oncorhynchus mykiss*), and eastern brook trout (*Salvelinus fontinalis*). Only the rainbow trout is native to park waters. West (1976) concluded that at most, 10% of the lakes investigated in the park were capable of supporting reproducing salmonid populations.

Between 1992 and 1998, underwater observations and videography were conducted in more than 25 lakes and ponds in the park by John DeMartini, Humboldt State University. The occurrence, distribution, and ecology of a wide range of aquatic species were documented, including sponges, annelids, cladocerans, fairy shrimp, Coleoptera, molluscs, copepods, aquatic vertebrates, and aquatic plants.

B. EMISSIONS

LAVO is on the boundary of three air basins, Mountain Counties, Northeast Plateau, and Sacramento Valley (SV). Located near the northern end of the SV, the park is potentially exposed to pollutants transported from the SV and other areas. The SV air basin (SVAB) includes nine counties and portions of two others. Emissions in the SV are dominated by sources in the Sacramento metropolitan area, at the southern end of the SV (Alexis et al. 1999). Since 1980, population growth in the SV has been more rapid than in many other parts of California, partially offsetting the effects of emission-control programs (Alexis et al. 1999). The Mountain Counties air basin (MCAB) includes the western slope of the Sierra Nevada, an area with relatively low population (~1% of the state total) and emissions (~ 3% of the state total; CARB 1998b). The Northeast Plateau is also an area of low population and emissions (CARB 1998b). Emission levels from counties within 140 km of LAVO are listed in Table IV-1. The principal species of concern are ozone precursors (NO_x and ROG) and PM. SO_2 emissions are not high.

LAVO is located within Lassen, Shasta, Tehama, and Plumas counties. Major point sources are not numerous in these counties, nor in other nearby counties (Figures II-3 through II-6). Sources within Lassen, Shasta, Tehama, and Plumas counties that emit at least 100 tons/year of ROG, NO_x, PM_{10}, or SO_2 are located near the communities of Chester, Quincy, Wendel, Burney, Redding, Anderson, and Red Bluff. As of 1996, stationary sources accounted for 9% of ROG emissions, 14% of NO_x emissions, and 4% of PM_{10} emissions in Lassen, Shasta, Tehama, and Plumas counties (CARB 1998b). Mobile sources dominated NO_x emissions, while area sources (road dust, construction, and farming operations) dominated PM emissions.

An inventory of in-park emissions has recently been compiled by the NPS-Air Resources Division. The results are presented in Table IV-2.

Table IV-1. 1995 Emissions from counties within 140 km of LAVO. Source: CARB Almanac 1999b; SO_x from CARB Emissions Website 1999a. Units are 1000 tons/year.

County	NO_x	ROG*	PM_{10}	CO	SO_x
Shasta	12.0	11.3	10.6	91.3	0.7
Tehama	6.2	4.0	5.1	27.7	0.4
Butte	7.7	10.6	10.2	60.2	0.4
Sutter	4.0	5.5	6.9	35.8	0.0
Yuba	3.3	4.0	3.3	23.4	0.4
Glenn	3.7	4.0	6.6	25.9	0.4
Lassen	3.7	4.7	8.4	30.7	0.4
Modoc	1.8	1.1	6.6	9.1	0.0
Siskiyou	7.7	9.1	11.7	109.1	0.4

* Reactive Organic Gases

Table IV-2 . Summary of 1998 stationary and area, and mobile source emissions (tons/yr) at LAVO.

Activity	Particulates	Sulfur Dioxide	Nitrogen Oxides	Carbon Monoxide	VOCs	HAPs[1]
Stationary and Area Source Emissions						
Stationary Combustion Sources						
Heating units	0.01	0.00	0.25	0.03	0.01	0.00
Generators	0.13	0.01	1.84	0.40	0.15	0.00
Woodstoves	1.49	0.02	0.14	11.25	2.58	0.00
Combustion Emission Subtotal	1.63	0.03	2.23	11.68	2.74	0.00
Fuel Storage Tanks						
Gasoline/Diesel Fuel Tanks	0.00	0.00	0.00	0.00	0.77	0.10
Area Sources						
Campfires	1.13	0.00	0.27	9.30	1.25	0.00
Prescribed Burning	384.21	0.00	1.83	4451.30	113.95	0.00
Area Source Emission Subtotal	385.34	0.00	2.09	4460.60	115.20	0.00
TOTALS	386.96	0.03	4.32	4472.28	118.71	0.11
Mobile Source Emissions						
Road Vehicles						
Visitor Vehicles	4.89	0.00	2.61	17.18	1.32	–
NPS/GSA Road Vehicles	0.83	0.00	1.76	5.88	0.52	–
Vehicle Emission Subtotal	5.72	0.00	4.37	23.06	1.84	–
Nonroad Vehicles						
NPS Nonroad Vehicles	0.12	0.00	1.01	0.42	0.15	–
TOTALS	5.84	0.00	5.37	23.48	1.99	--

[1] Hazardous air pollutants, based on the list compiled by EPA

C. MONITORING AND RESEARCH ACTIVITIES

1. Air Quality

Ozone, SO_2, fine particulate, and visual range have been monitored within the park (Figure IV-3 ,Table IV-3). Until recently, no wet deposition site has been located within 50 km of the park boundary. However, data from some of the CADMP sites, such as Quincy (~ 65 km southeast of the park boundary) are of some relevance. An NADP site was installed in 2000, but data are not yet available.

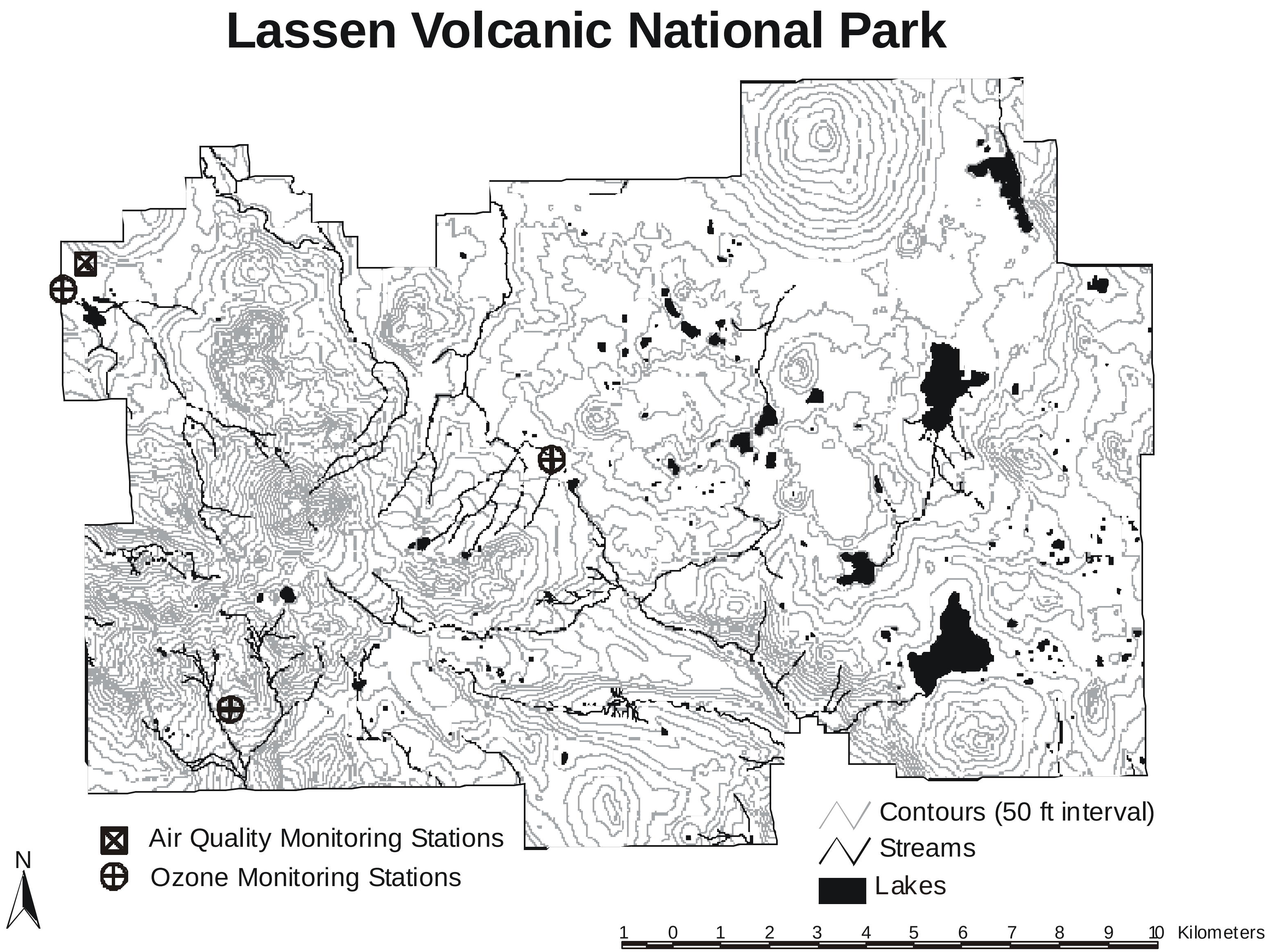


Figure IV-3. Hydrography and major contours of LAVO. Also shown are the locations of air quality and ozone monitoring stations.

Table IV-3. Air quality monitoring at LAVO.		
Species	Site within park	Site within 50 km
Ozone, hourly	CASTNet	
Ozone, passive		
SO_2	NPS	
PM_{10}	IMPROVE	
$PM_{2.5}$	IMPROVE	
Wet deposition	NADP*	
Dry deposition	CASTNet	
Visibility		
* New or soon to be installed site		

a. Wet Deposition

Blanchard et al. (1996) estimated annual and 10-year wet deposition rates throughout California by interpolating the observations from all NADP/NTN, CADMP, and special-studies monitoring locations for the period 1985 through 1994; they also estimated interpolation uncertainties. Upper-bound deposition values for unmonitored areas may be estimated as the sum of the interpolated values plus twice the interpolation uncertainty (i.e., mean plus two standard deviations). The results indicated that the 10-year mean total wet N deposition rates were less than 3 (+/- 3, at 2 sigma) kg/ha/hr (as N) throughout the state. Wet S deposition was less than 1.3 (+/- 1, at 2 sigma) kg/ha/hr (as S) throughout the state.

At the nearest CADMP site (Quincy), wet S deposition ranged from 0.4 to 0.9 kg/ha/hr as S (equivalently, 1.3 to 2.7 kg/ha/hr as SO_4^{2-}) during the period 1986 through 1994 (Blanchard et al. 1996). The annual NO_3^- and NH_4^+ deposition rates at Quincy were each in the range of 0.3 to 0.8 kg/ha/hr as N, yielding a multi-year mean total wet N deposition rate of 1.2 kg/ha/hr (Blanchard et al. 1996).

During the period 1984 through 1990, the annual-average H^+ concentration in precipitation at Quincy was 6.0 µeq/L (pH 5.22; Blanchard and Tonnessen 1993). At three other northern California locations (Eureka, Gasquet, and Montague), mean precipitation pH during that period was 5.32 to 5.34.

b. Occult/Dry Deposition

The CASTNet dry-deposition monitoring site located within LAVO began operating July 25, 1995. The monitoring instrument measures ambient concentrations of gases and particles, and EPA uses a computer model to calculate the dry-deposition rates from the measurements. The first calculations of dry-deposition rates for this site were released by EPA in November 2000. For the years 1996 through 1998, the calculated annual dry-deposition rates of N and S ranged from 0.47 to 0.57 kg N/ha/yr and 0.13 to 0.15 kg S/ha/yr, respectively. When combined with wet deposition measurements from the NADP/NTN network, the data indicate that the annual total deposition rates of N and S ranged from 3.0 to 5.5 kg N/ha/yr and 0.7 to 1.4 kg S/ha/yr, respectively, over the period 1996 through 1998 (a new NADP/NTN site is located 2.8 km from the dry-deposition monitor, but did not begin operating until June 2000). The average

total N and S deposition rates over the three-year period were 1.7 kg N/ha/yr and 0.51 kg S/ha/yr, respectively.

c. Gaseous Monitoring

Data from the hourly ozone monitor within the park show that ozone concentrations occasionally exceeded the state hourly ozone standard (90 ppb) but not the federal hourly standard (120 ppb; Table IV-4). The maximum 7-hour averages were in the range of 64 to 81 ppb, suggesting that compliance with the federal 8-hour ozone standard (80 ppb, 3-year average of fourth-highest annual values) will not be problematic. No passive ozone samplers have been sited within LAVO (source: Dr. John D. Ray, National Park Service, Air Resources Division, NPS Passive Ozone website 1999).

Table IV-4. Summary of ozone concentrations and exposure from LAVO. (Source: Joseph and Flores 1993; National Park Service, Air Resources Division 2000).

Year	Maximum Daily 1-hour Value (ppbv)	2nd Highest Daily 1-hour Value	Number of Daily Maximum 1-hour values # to 125 ppb	3-year Average Number of Exceedances	Maximum 9am-4pm Average (ppbv)	Sum06 (ppbv-hour)[a]	Number of Valid Hours of Ozone Measurements
1988	89	82	0	na	75	13,000	6493
1989	92	85	0	na	76	18,000	8023
1990	98	92	0	0	84	14,000	7792
1991	80	77	0	0	71	9,000	8106
1992	80	80	0	0	64	11,000	8125
1993	84	72	0	0	67	6,000	7741
1994	92	90	0	0	81	31,000	8192
1995	90	89	0	0	73	12,000	7337
1996	93	83	0	0	77	19,000	7352
1997	81	79	0	0	74	9,000	7948
1998	92	89	0	0	75	20,000	8217
1999	109	108	0	0	82	25,000	7449

[a] Maximum 8 am - 8 pm 90-day rolling average

Table IV-5 shows maximum and mean values for the 24-hour-resolution SO_2 samples for the period 1988-1993. SO_2 measurements were discontinued after 1996 due to concerns about their accuracy. The measurements are considered sufficiently accurate to show that the SO_2 concentrations were well below the levels at which plant injury has been documented, ~40 to 50 ppb 24-hour average and 8-12 ppb annual average (Peterson et al. 1992).

2. Aquatic Resources

LAVO contains over 200 lakes and ponds and 15 perennial streams (Figure IV-4). Most of the lakes in the park are less than about 1 ha in area, and many of those contain water only in spring and early summer. Lakes in the park are generally of glacial origin, although a few of the

Table IV-5. Maximum and mean SO_2, from 24-hour integrated samples at LAVO. Samples are collected every 3-4 days, unless noted. (Source: NPS Air Resources Division). Units are ppb.

SO_2	1988	1989	1990	1991	1992	1993
Maximum	0.63*	0.83*	18.23**	0.11*	0.05**	0.02*
Mean	0.05*	0.03*	0.45**	0.04*	0.01**	0.01*

na Not available
* Less than 50 samples collected for the year
** 50-75 samples collected for the year

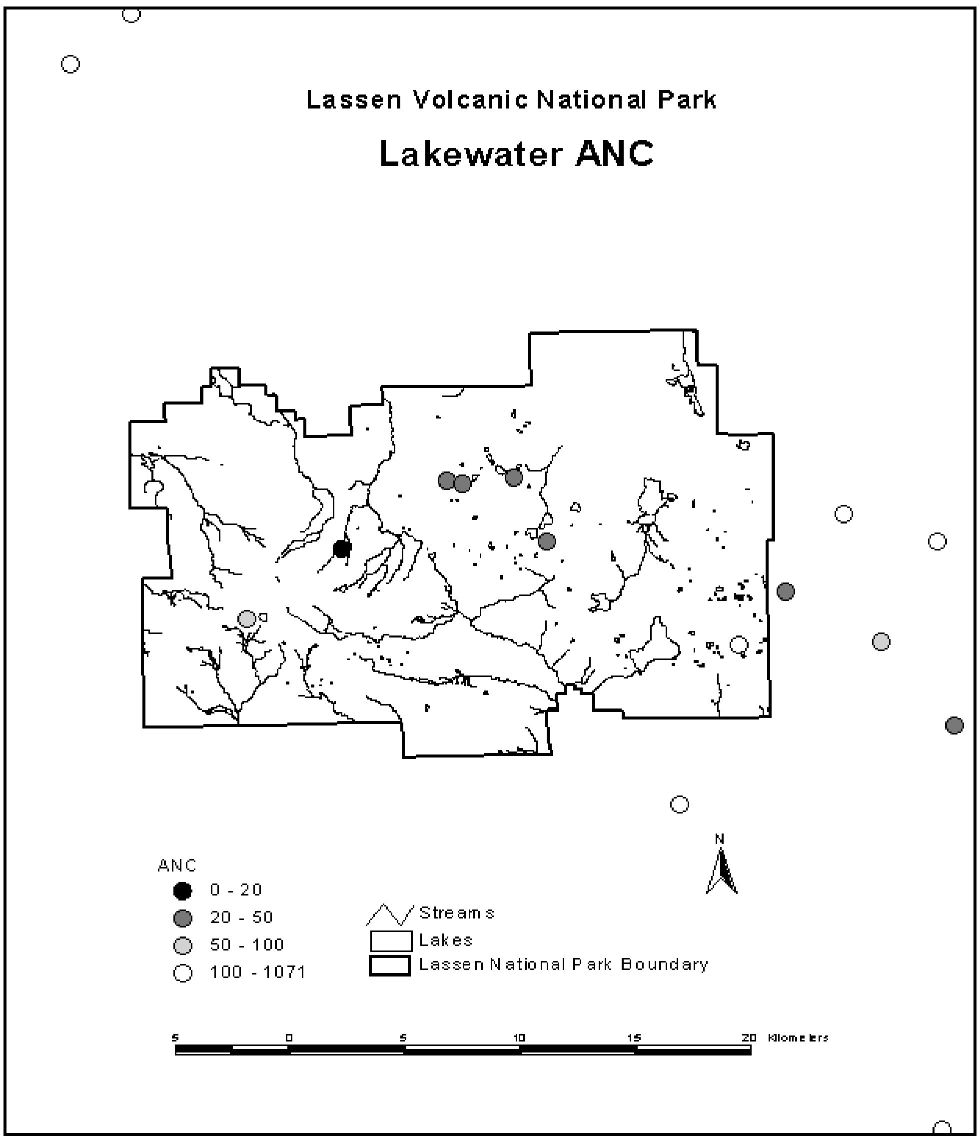


Figure IV-4. Acid neutralizing capacity (µeq/L) of lakes sampled in and near LAVO by the Western Lakes Survey.

smaller ones are found in the craters of extinct volcanoes. Inlet and outlet streams are generally swift-flowing and intermittent, flowing only during the spring runoff period. Aquatic ecosystems have been altered by concentrated visitor use and the diversion and damming of free-flowing streams, as well as stocking of non-native fish in lakes which were historically fishless.

LAVO contains portions of four drainage systems, all of which eventually drain into the Sacramento River. The area to the northeast of Lassen Peak drains into Hat Creek and Butte Creek, tributaries to the North Fork of the Sacramento River. The southeastern portion drains into the Feather River. The southwestern portion drains to Battle Creek or Mill Creek.

Manzanita Lake was created when Manzanita Creek was dammed in the mid 1800s for a small hydropower operation. Water was diverted from Manzanita Creek to Reflection Lake, which was originally a closed-basin lake, in order to provide water power and to improve fish production.

There is a lack of basic data on the composition and function of aquatic ecosystems in LAVO. Other important aquatic resource issues in the park include degradation of aquatic resources due to the presence of non-native fish, potential impacts on sensitive amphibian and reptile species, degradation of water quality by concentrated visitor use, and alteration of natural flow regimes in the diversion of water from Manzanita Creek to Reflection Lake.

Ecological conditions and suitability of lakes in the park for game fish production was evaluated in a field study of eight lakes conducted in 1960 (Hubbell 1960). The study included two large lakes, Butte Lake (87 ha) and Juniper Lake (232 ha); reported pH values for these two lakes were 7.6 and 6.9, respectively. The other six lakes studied by Hubbell (1960) had reported pH values that ranged from below 6 for Crumbaugh Lake to 6.6 for several of the other lakes. Lakes in this later group ranged in size from 3 to 9 ha.

The STORET database includes pH measurements at 27 sites within LAVO, which exhibited a median pH of 7.0. Median conductivity (n=27), Ca^{2+} (n=15), and SO_4^{2-} (n=19) concentrations were 54 µS/cm, 399 µeq/L, and 52 µeq/L, respectively. These data do not suggest acid sensitivity, but rather suggest substantial geological sources of S and base cations, perhaps associated with geothermal activity. Many of the sample site locations included within STORET are identified as springs, rather than lakes or streams. Six of the sample lakes in STORET are lakes that had conductivity < 5 µS/cm and pH reported as 6.0, but most were lakes that had also been included in the Western Lake Survey (WLS; Landers et al. 1987).

The WLS sampled seven lakes within LAVO, plus an additional nine lakes within 25 km of the park. Measured values of selected water quality parameters are listed in Table IV-6 for these 17 lakes. Five of the seven lakes sampled within LAVO had ANC < 50 µeq/L, and four of those had ANC values between 18 and 27 µeq/L. These lakes are highly sensitive to potential acidification from acidic deposition. Fewer highly sensitive lakes were sampled outside and adjacent to the park, although this latter group did include two lakes having ANC < 50 µeq/L (Table IV-6; Figure IV-4). It appears that highly sensitive lakes are more prevalent inside LAVO than in the surrounding areas. The sensitive lakes were small (< 8 ha) and had watershed areas between 31 and 114 ha. All had pH approximately equal to 6.5 and DOC # 2.1 mg/L. Sulfate concentrations in the sensitive lakes ranged from 2 to 6 µeq/L, confirming the very low levels of S deposition that LAVO receives. Lakewater NO_3^- concentrations were also very low, less than 1 µeq/L in all of the lakes sampled in and around the park by the WLS. Concentrations of Ca^{2+} and other base cations were very low (sum of base cation concentrations, C_B < 35 µeq/L) in the most sensitive lakes.

The WLS data suggest that a very high percentage of the lakes, and presumably also the streams, in LAVO are highly sensitive to potential acidification from acidic deposition. These sensitive lakes tend to be small, are situated in rather small watersheds, and are clearwater

Table IV-6. Results of lakewater chemistry analyses by the Western Lake Survey for selected variables in LAVO and adjacent areas.

Lake Name	Lake ID	Lake Area (ha)	Watershed Area (ha)	Elevation (m)	pH	ANC (µeq/L)	SO_4^{2-} (µeq/L)	NO_3^- (µeq/L)	Ca^{2+} (µeq/L)	C_B (µeq/L)	DOC (mg/L)
Lakes within LAVO											
Glen Lake	4A2-003	3.6	98	2110	7.3	151	1.4	0.4	69.7	155.2	2.9
(No Name)	4A3-010	2.6	31	2098	6.5	27	2.2	0.2	9.9	33.8	1.8
Feather Lake	4A3-011	4.8	41	2007	6.5	25	3.5	0.1	6.2	31.4	2.1
Emerald Lake	4A3-012	2.3	34	2476	7.1	88	17.6	0.2	49.4	102.8	0.9
Swan Lake	4A3-013	7.3	114	2019	6.5	23	6.2	0.4	6.5	31.1	1.1
Shadow Lake	4A3-054	7.6	70	2330	6.4	18	4.7	0.2	8.4	21.7	0.3
Little Bear Lake	4A3-072	1.8	106	2068	6.8	43	2.0	0.3	10.6	44.9	1.7
Lakes within 25 km of LAVO											
Black Lake	4A2-002	5	490	2146	7.5	194	0.8	0.2	71.7	195.5	2.0
Hidden Lakes (Eastern)	4A2-005	1.7	166	2095	6.9	62	0.7	0.1	19.8	65.2	2.2
Silver Lake	4A2-006	40	526	1974	7.2	205	2.1	0.5	95.1	204.7	1.6
(No Name)	4A2-060	2.2	57	2217	6.7	43	3.1	0.4	9.6	49.8	2.6
Box Lake	4A3-008	8.1	73	1960	7.1	130	0.5	0.2	78.4	141.5	3.8
Star Lake	4A3-016	4.1	73	1940	6.4	34	3.0	0.1	10.0	43.6	2.2
Lake Almanor	4A3-017	10010.7	126770	1372	8.2	1071	7.6	0.6	469.6	1108.0	1.6
Blue Lake	4A3-055	8.3	75	1643	7.0	104	1.6	0.1	30.7	116.8	2.6
Magee Lake	4A3-070	2	130	2201	7.3	175	1.0	0.1	107.8	176.5	1.5

systems with very low concentrations of Ca^{2+} and other base cations. None have ANC values approaching zero or pH < 6.0 because DOC concentrations are low and the concentrations of SO_4^{2-} and NO_3^- in the lakes are extremely low, owing to the low levels of S and N deposition received, and there appears to be an absence of geothermal contributions to the most acid-sensitive lakes.

There was one lake sampled by the WLS in LAVO (Emerald Lake) that had relatively high SO_4^{2-} concentration (18 µeq/L). This cannot be attributed to atmospheric deposition, and likely results from a minor watershed source of S, perhaps associated with geothermal activity. This SO_4^{2-} was accompanied by moderate concentrations of Ca^{2+} (49 µeq/L) and C_B (103 µeq/L), and the pH was above 7.0. This lake was also at the highest elevation (2,476 m) of the lakes that were sampled in and around LAVO. There are undoubtedly many other lakes and streams in the park that are chemically influenced by geothermal activity. Most thermal waters in the park are acidic (pH < 3.5) hot springs (Thompson 1983). The major thermal upflow from the Lassen geothermal system is at Bumpass Hell, which contains about 75 major fumaroles, acid sulfate hot springs, and mudpots. Such features are also common in Little Hot Springs Valley and Devil's Kitchen (Muffler et al.1982). Hot springs at Devil's Kitchen are acidic to neutral and have very high SO_4^{2-}/Cl^- ion ratios, likely derived from the mixing of meteoric water and magmatic gases rich in H_2S, SO_2, SO_3, and alkali halides (Ghiorso 1976).

On the whole, the lakes within LAVO sampled by the WLS reflected slightly greater acid-sensitivity than did the lakes sampled in YOSE or SEKI. Lakes in LAVO are among the most acid-sensitive in the National Park system.

A chemical survey of 101 high-altitude lakes in seven national parks in the western United States was conducted by the USGS during the fall of 1999 (Clow et al. 2000); 72 of the lakes were previously sampled during the fall of 1985 as part of the WLS. A strong effort was made to use methods and protocols similar to those used in the WLS. The objective of the 1999 lake survey was to provide information on water quality in the parks and assess whether there were significant differences in lake chemistry in the 1985 and 1999 data sets. Lakes in the three California parks (SEKI, YOSE, LAVO) and in Rocky Mountain National Park (Colorado) were extremely dilute; median specific conductances were # 12 µS/cm and median alkalinities were # 75 µeq/L. Specific conductances and alkalinities generally were substantially higher in Grand Teton and Yellowstone National Parks (Wyoming), and Glacier National Park (Montana), probably due to the prevalence of more reactive bedrock types. Concentrations of base cations and alkalinity were lowest in lakes in the alpine zone, probably because of minimal vegetation and soil development, and because of fast hydrologic flow rates. These conditions make alpine lakes highly sensitive to atmospheric deposition of pollutants. This is evidenced by relatively high NO_3^- concentrations in high-elevation lakes in Rocky Mountain National Park (0 to 29 µeq/L), which are subject to moderate levels of N deposition (3 to 5 kg N/ha/yr; Figure IV-5; Clow et al. 2000). Seven lakes were sampled in LAVO. Water chemistry was generally similar to the results of the WLS. There was not a consistent pattern of change in ANC. Nitrate concentration was 0 in all of the lakes.

One challenge that will need to be addressed in evaluating these survey data is separating effects of trends in water quality from variations due to differences in hydroclimatic conditions. A qualitative evaluation of the effects of climatic variations will be done by looking at variations in water chemistry and climate at several intensively monitored research watersheds in the Sierra Nevada and Rocky Mountains. This research is ongoing (D.W. Clow, U.S.G.S., Denver, pers. comm.).

Overall, there has been very little research conducted within LAVO on the potential effects of air pollutants on water quality. This is surprising in view of the high degree of sensitivity to

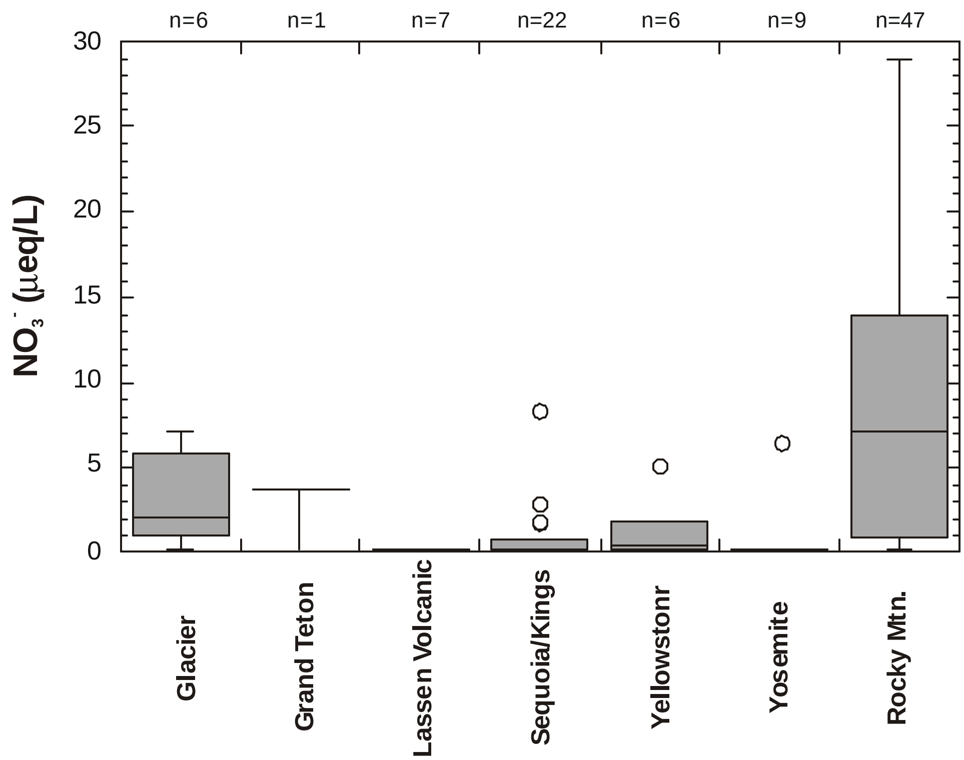


Figure IV-5. NO_3^- concentrations in high-elevation lakes sampled in 1999 in seven national parks by Clow et al. (2000).

acidification suggested by the available monitoring data. However, a great deal of research has been conducted on the acid base chemistry of high-elevation lakes in SEKI and their sensitivity to acidification from atmospheric inputs of N and S. Much of this work has been accomplished in conjunction with the Integrated Watershed Study (IWS; Tonnessen 1991) of the Emerald Lake watershed and various efforts to regionalize IWS findings. Specific studies have focused on such aspects as snowmelt hydrology, long-term monitoring, episodic acidification and neutralization processes, biological effects, and model predictions of future change. The results of these research efforts are described at length in Chapter IX of this report, because much of the work has been conducted within SEKI. However, the results of these studies are likely also applicable to high-elevation watersheds in LAVO. The reader interested in LAVO is therefore advised to refer extensively to the information presented in Chapter IX.

LAVO's Natural/Cultural Resources Management Plan (Lassen Volcanic National Park 1993) identified a critical need for assembling basic data on the aquatic resources of the park. Proposed actions included:

1. conduct a parkwide survey to determine the composition and interrelationships of park aquatic ecosystems,
2. conduct a parkwide inventory to determine and monitor the distribution status of amphibian and reptile populations and habitats,

3. determine cause and effect relationships associated with the diversion of water from Manzanita Creek to Reflection Lake,
4. monitor changes in aquatic ecosystems resulting from management and land use,
5. monitor and control alien fish populations, and
6. prepare a detailed water management action plan for the park.

To this list, we would also like to add obtaining a more complete characterization of the most acid-sensitive waters in the park and instituting a long-term aquatic monitoring program for three or four of the most sensitive systems.

3. Vegetation

Changes have occurred in the vegetation of LAVO since the park was established in 1916. Factors that may have been associated with the observed vegetation change include climatic change, natural disturbance regimes including fire, fluctuations in herbivore populations, and changes in land use. Taylor (1997) conducted a study to identify and describe patterns of vegetation change in LAVO, to identify mechanisms and factors that may have been responsible for such change, and to identify structural and compositional shifts in the vegetation. Taylor (1997) also attempted to identify and describe fire regimes across a montane forest gradient on Prospect Peak in LAVO. The fire history was reconstructed using fire-scarred partial cross-sections of live and dead trees and increment cores from fire-scarred and other live trees. The length of the fire record and frequency of fire occurrence varied among forest cover types. Jeffrey pine forests had the greatest fire frequency and also the longest fire record on Prospect Peak, with recorded fires ranging from the year 1507 to 1932. Natural fires have been effectively suppressed since the park was formed. Most natural fires have been the result of lightning strikes in the chaparral, Jeffrey pine, and red fir communities. Through implementation of the park's fire management plan, LAVO is attempting to restore the role of natural fire in these plant communities (Lassen Volcanic National Park and Caribou Wilderness Unit, Lassen National Forest, 1993).

Formal assessments of air pollution effects on vegetation in LAVO were first conducted in the 1990s. As part of the FOREST study, ozone injury in Jeffrey pine (dominant species) and ponderosa pine was evaluated at Manzanita Lake for three years between 1992 and 1994 (Arbaugh et al. 1998). Surveys indicated that 26.8% of the trees sampled had visible ozone injury, with mean whorl retention of 6.5 and mean Ozone Injury Index of 6.3. Although this level of injury is lower than at other locations in California in the FOREST study, it indicates a significant impact on vegetation in the park.

The sensitivities of plant species at LAVO to air pollutants are summarized in Table IV-7. In addition to Jeffrey pine and ponderosa pine, there are other potential ozone bioindicators in LAVO, including quaking aspen and black cottonwood (*Populus trichocarpa*), which are also highly sensitive to ozone. Skunkbush sumac is perhaps the most sensitive understory species in the park. Jeffrey pine and ponderosa pine are also regarded as good bioindicators for SO_2 and NO_2, neither of which poses a concern at the present time.

Table IV-7. Plant and lichen species of LAVO with known sensitivities to sulfur dioxide, ozone, and nitrogen oxides (H=high, M=medium, L=low, blank=unknown).

Scientific Name	Common Name	Sensitivity		
		SO_2	O_3	NO_x
Gymnosperms				
Abies concolor	White fir	H	M	H
Calocedrus decurrens	Incense cedar		M	
Pinus jeffreyi	Jeffrey pine	H	H	H
Pinus monticola	Western white pine	M	M	
Pinus ponderosa	Ponderosa pine	H	H	H
Pseudotsuga menziesii	Douglas fir	H	M	H
Taxus brevifolia	Pacific yew	L		
Tsuga mertensiana	Mountain hemlock	H	L	
Angiosperms				
Achillea millefolium	Common yarrow		L	
Alnus tenuifolia	Thinleaf alder	M		
Artemisia douglasiana	Douglas' sagewort		H	
Artemisia tridentata	Big sagebrush	M	L	
Bromus carinatus	California brome		L	
Bromus tectorum	Cheatgrass		M	
Cassiope mertensiana	Western moss heather		L	
Ceanothus velutinus	Snowbrush ceanothus	L		
Cercocarpus ledifolius	Curlleaf mountain mahogany	M		
Cichorium intybus	Chicory		L	
Collomia linearis	Narrowleaf mountaintrumpet		L	
Convolvulus arvensis	Field bindweed	H		
Cornus stolonifera	Redosier dogwood	M	L	
Elymus glaucus	Blue wildrye		H	
Epilobium angustifolium	Fireweed		L	
Epilobium brachycarpum	Autumn willowweed		L	
Erodium cicutarium	Redstem stork's bill	M	M	
Festuca idahoensis	Idaho fescue	H		
Galium bifolium	Twinleaf bedstraw		L	
Gayophytum diffusum	Spreading groundsmoke		H	
Gayophytum racemosum	Blackfoot groundsmoke		L	
Gentiana amarella	Autumn dwarfgentian		M	
Hackelia floribunda	Manyflower stickseed	L		
Lemna minor	Common duckweed	L		
Lolium perenne	Perennial ryegrass		M	
Lonicera involucrata	Twinberry honeysuckle	L		M
Mimulus guttatus	Seep monkeyflower		L	
Oenothera elata	Hooker’s evening primrose		H	
Osmorhiza chilensis	Sweetcicely		M	
Osmorhiza occidentalis	Western sweetroot		L	
Poa annua	Annual bluegrass	H	L	
Poa pratensis	Kentucky bluegrass		L	
Polygonum douglasii	Douglas' knotweed		L	
Populus tremuloides	Quaking aspen	H	H	
Populus trichocarpa	Black cottonwood	M	H	
Potentilla flabellifolia	High mountain cinquefoil		H	
Potentilla glandulosa	Gland cinquefoil		M	

Table IV-7. Continued.

Scientific Name	Common Name	Sensitivity		
		SO_2	O_3	NO_x
Prunus emarginata	Bitter cherry	M		
Prunus virginiana	Common chokecherry	M	M	
Quercus kelloggii	California black oak		M	
Rhus trilobata	Skunkbush sumac	L	H	
Ribes viscosissimum	Sticky currant	M		
Rosa woodsii	Woods' rose	M	L	
Rubus parviflorus	Thimbleberry		M	
Rumex crispus	Curly dock		L	
Salix scouleriana	Scouler's willow		M	
Saxifrage arguta	Stream saxifrage		L	
Symphoricarpos vaccinioides	Utah snowberry		L	
Taraxacum officinale	Common dandelion		L	
Thalictrum fendleri	Fendler's meadowrue		L	
Trifolium pratense	Red clover	L		
Trifolium repens	White clover		H	
Trisetum spicatum	Spike trisetum	M		
Viola adunca	Hookedspur violet		L	
Lichens				
Hypogymnia imshaugii		M	M	
Letharia vulpina		L	L	

4. Visibility

As part of the Interagency Monitoring of Protected Visual Environments (IMPROVE) network, visual air quality in LAVO has been monitored using an aerosol sampler and camera. The aerosol sampler began operation in March of 1988 and is located south of Table Mountain near the northwestern park boundary. The automatic 35mm camera was initially located atop a lift tower at the LAVO ski area. Photographs of the Butte Mountain vista to the south were captured from June 1987 through August 1993. The camera was relocated to the Windy Point Overlook west of the Park and east of the town of Paynes Creek in December 1993. Photographs of Lassen Peak were captured from the second location until April 1995. Data from this IMPROVE site have been summarized to characterize the full range of visibility conditions for the March 1988 through February 1999 period, based on seasonal periods (Spring: March, April, and May; Summer: June, July, and August; Autumn: September, October, and November; and Winter: December, January, and February) and annual periods (March through February of the following year, e.g., the annual period of 1998 includes March 1998 through February 1999). Complete descriptions of visibility characterization, mechanisms of sources and visibility impacts, and IMPROVE monitoring techniques and rationale are provided in Chapter I.

a. Aerosol Sampler Data - Particle Monitoring

A graphic and tabular summary of average reconstructed extinction values by season and year for the March 1988 through February 1999 period are provided in Table IV-8 and Figure IV-6, respectively. Reconstructed extinction budgets generated from aerosol sampler data

Table IV-8. Seasonal and annual average reconstructed extinction (b_{ext}; Mm^{-1}) at LAVO, March 1988 through February 1999.

Year	Spring (Mar, Apr, May)	Summer (Jun, Jul, Aug)	Autumn (Sep, Oct, Nov)	Winter (Dec, Jan, Feb)	Annual (Mar - Feb)[a]
1988	23.4	38.2	25.4	20.6	26.3
1989	25.3	42.7	25.0	17.9	27.1
1990	28.4	40.9	21.6	21.1	27.1
1991	21.0	30.7	24.1	17.3	23.0
1992	25.4	35.4	- - -	- - -	25.2
1993	25.6	31.1	26.9	18.4	25.1
1994	27.8	35.3	23.0	18.4	25.9
1995	23.2	33.5	23.8	16.3	24.0
1996	24.9	38.5	20.9	14.8	24.1
1997	23.2	30.5	20.4	16.8	22.3
1998	21.8	39.1	33.7	17.2	27.8
Mean[b]	24.5	36.0	24.5	17.9	25.2[c]

[a] Annual period data represent the mean of all data for each March through February annual period.
[b] Combined season data represent the mean of all seasonal means for each season of the March 1988 through February 1999 period.
[c] Combined annual period data represent the mean of all combined season means.

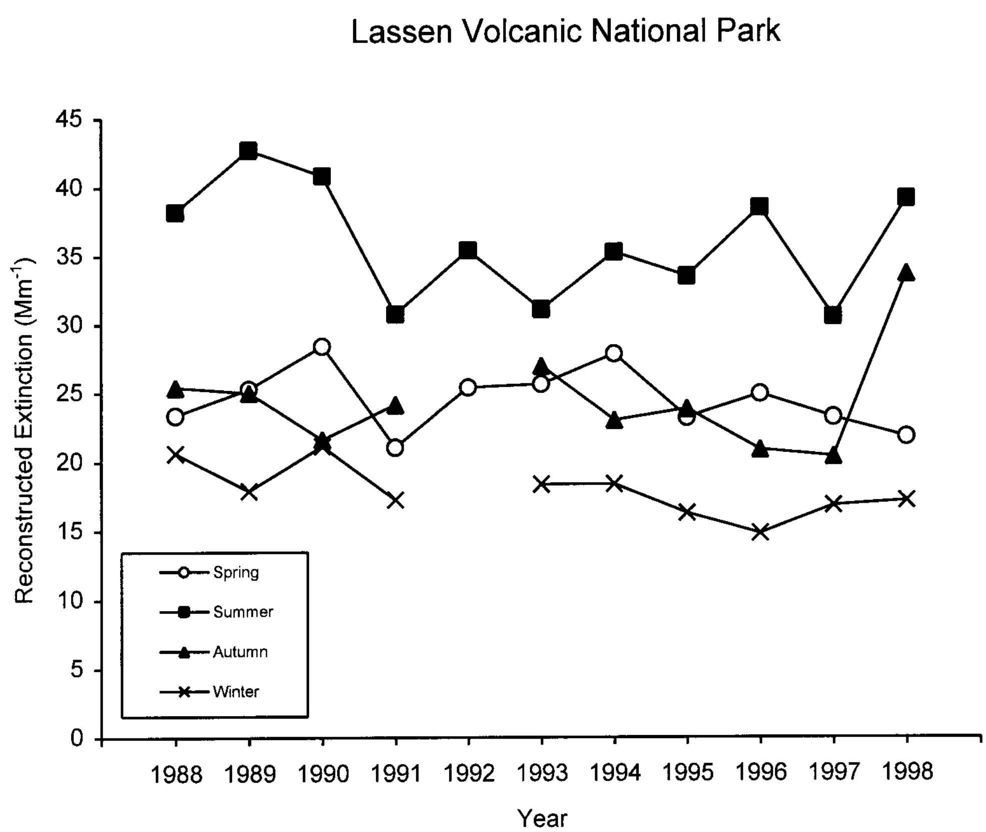


Figure IV-6. Seasonal average reconstructed extinction (Mm^{-1}) at LAVO, March 1988 through February 1999.

apportion the extinction at LAVO to specific aerosol species (Figure IV-7). The species shown are Rayleigh, sulfate, nitrate, organics, elemental (light absorbing) carbon, and coarse mass. The sum of these species account for the majority of non-weather related extinctions. Extinction budgets are listed by season and by mean of cleanest 20%, mean of the median 20%, and mean of the dirtiest 20%. The "cleanest" and "dirtiest" signify lowest fine mass concentrations and highest fine mass concentrations respectively, with "median" representing the 20% of days with fine mass concentrations in the middle of the distribution. Each budget includes the corresponding extinction coefficient, standard visual range (km), and deciview (dv).

The segment at the bottom of each stacked bar in Figure IV-7 represents Rayleigh scattering, which is assumed to be a constant 10 Mm^{-1} at all sites during all seasons. Rayleigh scattering is the natural scattering of light by atmospheric gases. Higher fractions of extinction due to Rayleigh scattering indicate cleaner conditions.

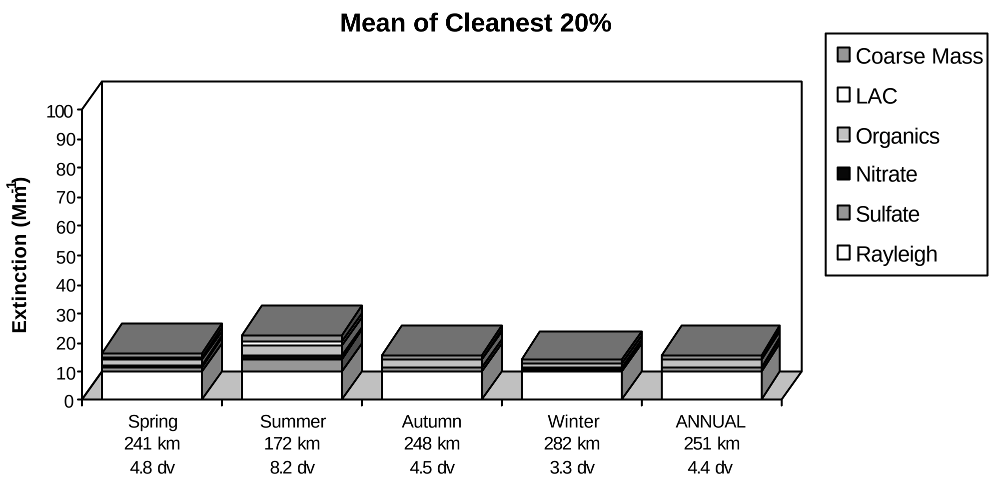
b. Camera Data - View Monitoring

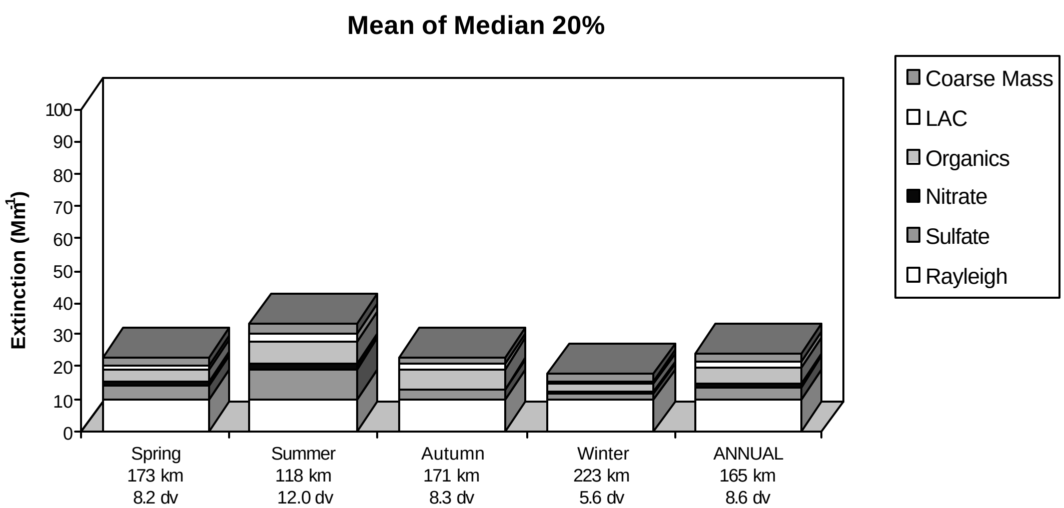
View monitoring slides document visual conditions and are an effective tool for interpreting the visual effects of measured optical and aerosol parameters or presenting monitoring program goals, objectives, and results to decision-makers and the public. The Lassen Peak vista photographs presented in Figure IV-8 were chosen to provide a feel for the range of visibility conditions possible and to help relate the extinction/SVR/haziness data to the visual sense.

c. Site-Specific Data Interpretation

Data from other IMPROVE visibility sites around the country have been presented graphically (Figures I-3 and I-4) so that spatial trends in visual air quality for the Lassen and Pacific Coast regions can be understood in perspective. Figures IV-6 and IV-7 have been provided to summarize LAVO visual air quality during the March 1988 through February 1999 period. Seasonal variances in the mean of the dirtiest 20% fractions are driven primarily by organic and sulfate extinctions. Non-Rayleigh atmospheric light extinction at LAVO, like many rural western areas, is largely due to organics and sulfates. Historically, visibility varies with patterns in weather, winds (and the effects of winds on coarse particles) and smoke from fires. No information is available on how the distribution of visibility conditions at present differs from the profile under "natural" conditions, but the cleanest 20% of the days probably approach natural conditions (GCVTC 1996). Smoke from frequent fires is suspected to have reduced presettlement visibility below current levels during some summer months.

Long-term trends fall into three categories: increases, decreases, and insignificant changes. The characterization of long-term trends can be a highly subjective exercise in that slopes and their significance can vary depending on the technique employed. Recently the IMPROVE aerosol network, initiated in March 1988, matured to a point where long-term trends of average ambient aerosol concentrations and reconstructed extinction can be assessed. In the recent IMPROVE report (Malm et al. 2000), the authors applied the Theil (1950) approach to describe trends for IMPROVE sites with eleven years of data. The distribution of $PM_{2.5}$ mass concentrations, reconstructed extinction expressed as deciview, and associated constituents were examined for each site. The data were sorted into three groups based on the cumulative frequency of occurrence of $PM_{2.5}$: lowest fine mass days, 0-20%, median fine mass days, 40-60%, and highest fine mass days, 80-100%. After sorting each group's average concentrations of $PM_{2.5}$ and selecting the associated principal aerosol species, scattering and/or absorption of each species, reconstructed light extinction and deciview were calculated. Figure IV-9 shows plots of the 10, 50, and 90 percentile groups at LAVO for both $PM_{2.5}$ and deciview.





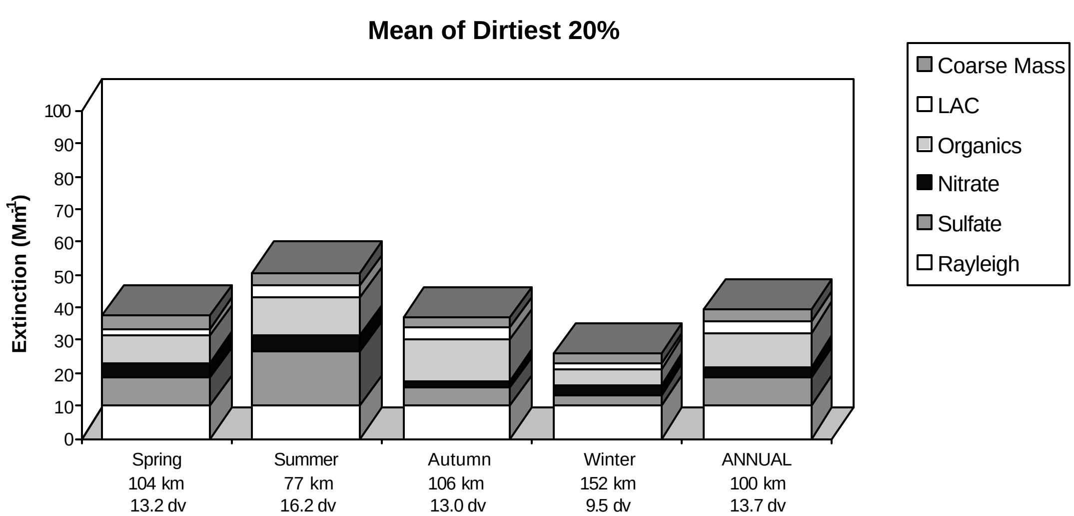


Figure IV-7. Reconstructed extinction budgets for LAVO, March 1988 through February 1999.

Given the visibility data summarized for LAVO, the majority of data show insignificant change over the period. It should be noted that the 90 percentile (highest fine mass days) $PM_{2.5}$ and deciview appear to have been highly variable over the eleven year summary period. In addition, analysis of the long-term five-year average deciview at LAVO also showed no significant increasing or decreasing trend (Malm et al. 2000).

D. RESEARCH AND MONITORING NEEDS

1. Deposition

We recommend continuation of CASTNet monitoring in the park in view of the apparent extreme acid-sensitivity of many lakes in the park. Continued collection of wet deposition data in the park would also be advisable, especially if lake monitoring in the future suggests any appreciable changes in acid-base chemistry.

2. Gases

We recommend that ozone monitoring be continued. In addition, a small network of passive ozone samplers would help to calibrate with the existing analyzer at Manzanita Lake and quantify spatial variation in ozone exposure at LAVO. One summer of sampling with samples located with the analyzer and at various elevations in the park should be sufficient.

3. Aquatic Systems

LAVO contains a very high percentage of acid-sensitive lakes. Fifty-seven percent of the lakes sampled within LAVO by the WLS had ANC # 27 µeq/L, and 71% had ANC # 50 µeq/L. These data suggest a higher percentage of acid-sensitive lakes than any other Class I national park in the western United States (c.f., Eilers et al. 1994, Peterson and Sullivan 1998). However, it is not known if other lakes exist within LAVO that are more sensitive than those sampled by the WLS. Furthermore, the seasonal and annual variability of the chemistry of the known acid-sensitive lakes has not been documented. Such information is critical if future monitoring data will be useful for documenting water chemistry trends over time. Some information regarding annual variability will soon be available with completion of the analyses of WLS lakes that were resurveyed in 1999 by the USGS (D. Clow, USGS, pers. comm.), but more data are required.

We recommend a screening survey to ascertain the distribution of ultrasensitive lakes within LAVO. Such a survey could be conducted during a single fall season. Field measurement of conductivity should be used to determine a suite of about 10 lakes to be sampled for full ion chemistry. The results of the chemical analyses, together with locational information and sampling logistics should be used to select a group of three or four lakes for long-term monitoring. The selected lakes should include some WLS lakes and perhaps other more acid-sensitive lakes identified in the screening survey. Monitoring should include annual sampling during July for water chemistry, with occasional (every 3 to 5 yr) additional (i.e., monthly) sampling during the period from ice-out to early fall. The latter effort will require access via skis to the sample sites. Some biological monitoring could also be conducted in conjunction with long-term chemical monitoring. Zooplankton monitoring may be the simplest and most cost-effective way to track potential changes in biological communities of high-elevation lakes in response to potential future increases in S or N deposition (c.f., Melack et al. 1989). Because of their seasonal variations, however, zooplankton monitoring would require a concerted effort.

We believe that immediate implementation of long-term monitoring of several ultra-sensitive lakes in LAVO is a critical need because:

1. LAVO appears to contain the highest percentage of ultrasensitive lakes in the NPS system,
2. there is a real danger of increased air pollution in future decades due to population growth up-wind of the park, and
3. there is a general scarcity of data on high-elevation aquatic ecosystems in the park.

Without the future availability of appropriate monitoring data, it will not be possible for the NPS to discern the extent to which any future increases in air pollution cause adverse chemical or biological changes in these outstanding aquatic resources.

4. Terrestrial Systems

An ongoing program of monitoring should be considered in order to detect the potential effects of ozone on vegetation in LAVO. Ozone injury has already been documented in Jeffrey pine and ponderosa pine in the park, and these species will likely continue to sustain injury. The challenge will be to quantify ozone injury with sufficient precision and temporal continuity to detect changes in injury over time. Studies of long-term growth patterns in trees with various injury levels could also be considered.

It is recommended that monitoring continue at Manzanita Lake at the same locations as the FOREST plots, preferably including the same trees, to allow long-term analysis. This location is ideal because it also has an ozone analyzer. Annual evaluations in late summer are recommended. Monitoring protocols should be the same as those used in the FOREST study (Arbaugh et al. 1998), including the use of the Ozone Injury Index (Duriscoe et al. 1996; Schilling and Duriscoe 1996) to quantify injury.

If LAVO wishes to determine if species other than the pines are being damaged by ozone, quaking aspen is the next best species for monitoring. Skunkbush sumac is the best understory bioindicator. If these or other species are evaluated, monitoring protocols from the Forest Health Monitoring Manual (USDA Forest Service 1999) are recommended for establishing plots and collecting data (see Appendix).

5. Visibility

We have no recommendations for visibility monitoring and research other than to continue IMPROVE monitoring.

V. LAVA BEDS NATIONAL MONUMENT

A. GENERAL DESCRIPTION

The 18,843 ha Lava Beds National Monument (LABE) is located about 250 km northeast of Redding, California and 80 km southeast of Klamath Falls, Oregon. LABE was established by presidential proclamation in 1925. The monument, which includes 11,518 ha of designated wilderness (Figure V-1), was first managed as part of Modoc National Forest; the NPS assumed responsibility for management in 1933. A second presidential proclamation in 1951 transferred lands at Petroglyph Point to LABE from the Bureau of Land Management. This detached unit (77 ha) is about 3 km east of the main body of the monument.

LABE is located at the interface between the Cascade Range and the Great Basin Geologic Provinces on the lower northern section of the Medicine Lake Shield Volcano. The monument contains examples of recent lava flows, cinder and spatter cones, lava tube caves, collapsed lava tubes and craters. The northern boundary of the monument is along the limit of the lava flows, which also corresponds with the historic shoreline of Tule Lake. The lake level was significantly lowered by water resources development activities that were initiated in 1905.

Elevation ranges from 1,200 m at Tule Lake to 1,674 m on Hippo Butte. The southwestern portion of the monument is about 1,525 m elevation, with many of the volcanic cones extending higher. The topography is generally flat to gently sloping, but it is dotted with numerous cinder cones that rise 60 to 150 m.

The national monument boundary is adjoined by Modoc National Forest, Klamath National Forest, Lower Klamath Basin National Wildlife Refuge, Bureau of Reclamation, Bureau of Land Management, and private lands. Annual visitation is about 125,000 visitors per year, most of which occurs during the summer months.

LABE has the greatest concentration of lava tube caves in the continental United States. The caves range in size from small grottos to the very large Catacombs Cave, with 2,100 m of surveyed passage. Many of the caves are highly complex both horizontally and vertically, with many interconnected branches and/or different levels. Depths range from surface tubes located above the land surface to 46 m below the surface (Larson and Larson 1990). Pictographs decorate the walls of several of the caves, including Fern Cave, which was and is a holy site in the Modoc culture. The paintings are believed to have been connected with visions associated with vision quests of the Modoc people.

There is an extensive array of cultural resources in the monument due to the 10,000 years of human occupation of the area. The Modoc Indians were hunters and gatherers who lived in semi-permanent villages along the shores of Tule Lake. With the arrival of European settlement, conflicts between cultures escalated and culminated in the Modoc Indian War of 1872-1873. Cultural resources include archaeological sites, rock art, historic structures, and an extensive array of fortifications from the Modoc Indian War. Petroglyph Point contains over 5,000 petroglyphs and represents the largest concentration of rock art in California. The entire monument has been named an archaeological district due to the richness and diversity of the area's pre-history.

1. Geology and Soils

The monument is of recent geologic origin. It is covered with volcanic rock, of which about two-thirds is basaltic lava that erupted over 11,000 years ago. Much of the lava was distributed by lava tubes, leaving flows with terrace-like borders ranging up to 9 m high separated by valley-like depressions in between (Larson and Larson 1990). There are a variety of cinder cones that rise above the general surface, as well as smaller spatter cones and chimneys. There are also several craters, the deepest of which is about 115 m.

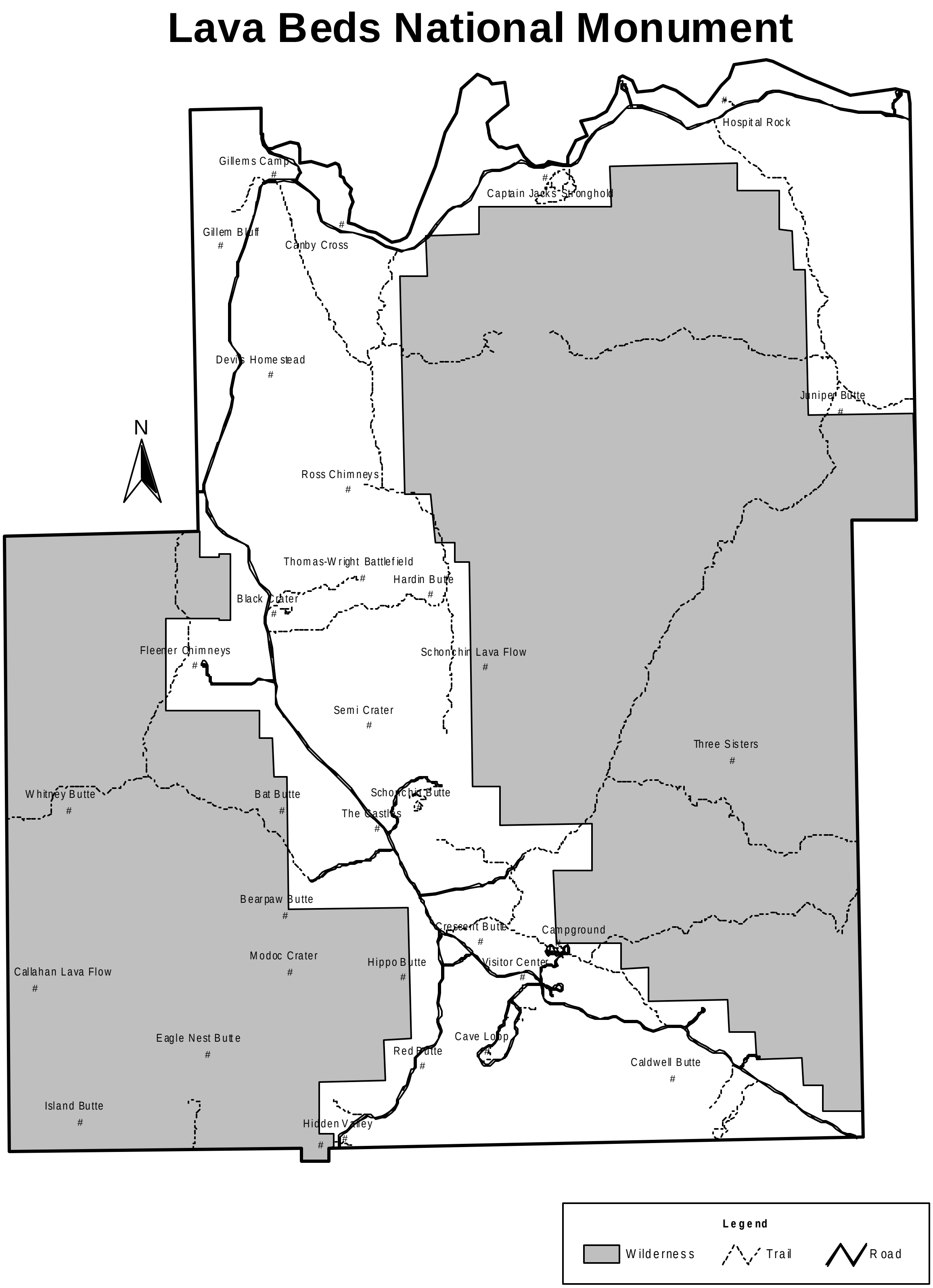


Figure V-1. Map of LABE, showing the major features and area of designated wilderness.

scrub are comprised of lava flows, with minimal vascular vegetation present except in small microsites where soil development and water accumulation allow plant growth.

Moving farther south and to higher elevations, juniper chaparral dominates the landscape, with western juniper (*Juniperus occidentalis*) the most important species. Other shrubs found between the larger junipers include curl-leaf mountain mahogany and antelope bitterbrush, and grasses that comprise the rest of the vegetation matrix include Idaho fescue and bluebunch wheatgrass. These juniper-chaparral areas had been ponderosa pine woodlands previously, but drought and western pine beetle combined to weaken and kill many of these stands in the 1920s (Martin and Johnson 1979).

Open ponderosa pine (*Pinus ponderosa*) stands are found at some of the higher elevations in the southern end of the park. Relatively low densities of trees are accompanied by antelope bitterbrush and greenleaf manzanita (*Arctostaphylos patula*), and by incense cedar (*Calocedrus decurrens*) and white fir (*Abies concolor*) at the highest elevations. The most common shrubs are snowbrush ceanothus (*Ceanothus velutinus*) and green leaf manzanita. The ponderosa pine-dominated communities probably had a greater grass component prior to fire exclusion, when fire frequencies ranged from 7 to 17 years (Martin and Johnson 1979). Higher native grass cover was probably found throughout LABE prior to grazing by domestic livestock.

Microhabitats associated with lava tube cave entrances and collapsed structures in LABE harbor a fern flora that is remarkable for both species diversity and the number of populations (Richard et al. 1994). The occurrence of ferns within and around some of the caves is a unique vegetative feature of the monument. These relatively stable protected environments have attenuated light levels and greater moisture availability than the Modoc Plateau above and therefore more closely resemble conditions typical of coastal forests. Thus, the cave entrances and collapsed structures provide island refuges for species of fern that are not found in the surrounding semi-desert landscape. A survey by Richard et al. (1994) in LABE found ferns at 20 cave entrances and 9 other sites within the monument.

The vegetation of the monument has been altered by grazing, western pine beetle infestation, and fire control (Martin and Johnson 1979). Land within the monument was heavily grazed by sheep and some cattle, continuing through 1974. Grazing had a significant impact on the native vegetation, including the intrusion of exotic plant species, such as cheatgrass and woolly mullein (Lava Beds National Monument 1994). Most of the native bunchgrass stands were selectively grazed out in the northern one half of the monument and non-native plants filled in the niche. The intrusion of exotic plant species such as cheatgrass is well integrated within the system (Lava Beds National Monument 1996).

Lightning fires are common in the LABE area. Fire records indicate 1.8 lightning fires per year on the monument, about half of which occur in July (Martin and Johnson 1979). With the establishment of the monument in the 1920s, wildfires were actively suppressed. This policy continued until the late 1980s. As a consequence of fire suppression, fuel loadings became much higher than the normal situation, and these high fuel loadings support wildfires that have greater intensity and are more resistant to control (Lava Beds National Monument 1996).

The monument supports a considerable diversity of animals. There are 51 species of mammals, 217 species of birds, 12 species of reptiles, and 2 species of amphibians known in the monument.

Snow depth at the upper elevations of the Medicine Lake Volcano drive mule deer (*Ododoileus hemionus*) to move down the eastern slope and into the monument during winter. Pronghorn (*Antilocapra americana*) occupy the northern and eastern portions. The monument provides habitat for mountain lions (*Felis concolor*), which have been sighted much more frequently in recent years.

The monument provides particularly important habitat for Townsend's big-eared bat (*Corynorhinus townsendii*), which makes use of monument caves for winter hibernation and also for summer maternity roosts. The monument is home to the largest hibernaculum and some of the largest maternity colonies of Townsend's big-eared bat, as well as the northern-most maternity colony of Brazilian free-tail bats (*Tadarida brasiliensis*) in the United States.

LABE provides suitable habitat for two federally-listed bird species, the bald eagle (*Haliaeetus leucocephalus*) and the American peregrine falcon (*Falco peregrinus anatun*). Although peregrine falcons have not been documented in the monument since the 1930s, they have been sighted recently in adjacent areas and it is reasonable to expect that they will reappear in the monument (Lava Beds National Monument 1996).

Tule Lake and nearby Lower Klamath Lake are home to North America's greatest concentration of waterfowl. Each spring and fall, millions of migrating birds converge on those lakes (Lamb 1991). The waterfowl populations in winter support the largest concentration of bald eagles in the lower 48 states.

There are two areas within the monument that are used extensively by bald eagles for roosting and protection from cold winter weather. These are the Caldwell/Cougar Butte and Eagle Nest Butte. The former is one of the major winter roosts of bald eagles located in the Klamath Basin. More than 900 bald eagles annually winter in the Klamath Basin, representing nearly half of the California bald eagle winter population.

Two species of amphibian, the boreal toad (*Bufo boreas*) and the Pacific treefrog (*Pseudacris regilla*), occur in LABE, which is surprising given that the monument is located more than 16 km from standing water. Frogs have been found at various caves located far from above-ground water sources. There are also four species of lizard and eight species of snake known to live within LABE. The latter include the western rattlesnake (*Crotalus viridis*) and desert night snake (*Hypsiglena torquata deserticola*).

B. EMISSIONS

LABE is located on the border of Siskiyou and Modoc counties, within the Northeast Plateau air basin. The Northeast Plateau is an area of low population and emissions (CARB, 1998b). Emission levels from Siskiyou and Modoc counties, as well as Lassen County, which is also within 140 kilometers of Lava Beds NM, are listed in Table V-1. The principal species of concern are ozone precursors (NO_x and ROG) and PM. SO_2 emissions are not high. No major point sources (those that emit at least 100 tons/year of ROG, NO_x, PM_{10}, or SO_2) are located in Siskiyou and Modoc counties, and they are not numerous in other nearby counties (Figures II-3 through II-6). As of 1996, stationary sources accounted for 5 percent of ROG emissions, 0.4

Table V-1. 1995 Emissions from California counties within 140 km of LABE. (Source: CARB Almanac 1999b; SO_x from CARB Emissions Website 1999a.) Units are 1000 tons/year.

County	NO_x	ROG*	PM_{10}	CO	SO_x
Lassen	3.7	4.7	8.4	30.7	0.4
Modoc	1.8	1.1	6.6	9.1	0.0
Siskiyou	7.7	9.1	11.7	109.1	0.4

* Reactive Organic Gases

percent of NO_x emissions, and 1 percent of PM_{10} emissions in Siskiyou and Modoc counties (CARB, 1998b). Mobile sources dominated NO_x emissions, while area sources (road dust, construction, and farming operations) dominated PM emissions.

The city of Klamath Falls, located 84 km northwest of the monument, faces significant air quality problems due to cool weather inversions that trap large volumes of smoke from the burning of agricultural fields and the burning of wood for heating. This on-going air quality problem affects the viewsheds of the monument (Lava Beds National Monument 1994).

An inventory of in-park emissions has recently been compiled by the NPS-Air Resources Division. The results are presented in Table V-2.

Table V-2 . Summary of 1998 stationary and area, and mobile source emissions (tons/yr) at LABE.

Activity	Particulates	Sulfur Dioxide	Nitrogen Oxides	Carbon Monoxide	VOCs	HAPs[1]
Stationary and Area Source Emissions						
Stationary Combustion Sources						
Heating units	0.00	0.02	0.04	0.01	0.00	0.00
Woodstoves	0.73	0.01	0.07	5.45	1.96	0.00
Combustion Emission Subtotal	0.73	0.02	0.10	5.46	1.96	0.00
Fuel Storage Tanks						
Gasoline/Diesel Fuel Tanks	0.00	0.00	0.00	0.00	0.32	0.04
Area Sources						
Campfires	0.14	0.00	0.03	1.16	0.16	0.00
Prescribed Burning	46.57	6.39	2.56	349.30	49.28	0.00
Area Source Emission Subtotal	46.71	6.39	2.59	350.46	49.44	0.00
TOTALS	47.44	6.41	2.69	355.92	51.72	0.04
Mobile Source Emissions						
Road Vehicles						
Visitor Vehicles	5.97	0.00	1.74	18.08	1.36	–
NPS/GSA Road Vehicles	0.17	0.00	0.16	0.64	0.05	–
Vehicle Emission Subtotal	6.13	0.00	1.90	18.72	1.40	–
Nonroad Vehicles						
NPS Nonroad Vehicles	0.06	0.00	0.41	0.19	0.06	–
TOTALS	6.19	0.00	2.31	18.90	1.47	–

[1] Hazardous air pollutants, based on the list compiled by EPA

C. MONITORING AND RESEARCH ACTIVITIES

1. Air Quality

Air quality monitoring within LABE is limited to ozone (passive ozone sampler) and camera data which were collected 1986-1991. No wet deposition site has been located within 50 km of the park boundary. However, data from some of the CADMP or NADP/NTN sites, such as Montague (~ 85 km west of the park boundary) are of some relevance.

During windy conditions, which are frequent in the area of the monument, unconsolidated lake bed sediments from Tule Lake and exposed topsoil are lofted and scour the petroglyph-bearing surfaces. This has caused erosion damages to the petroglyphs in recent years (Lava Bed National Monument 1996).

a. Wet Deposition

Blanchard et al. (1996) estimated annual and 10-year wet deposition rates throughout California by interpolating the observations from all NADP/NTN, CADMP, and special-studies monitoring locations for the period 1985 through 1994; they also estimated interpolation uncertainties. Upper-bound deposition values for unmonitored areas may be estimated as the sum of the interpolated values plus twice the interpolation uncertainty (i.e., mean plus two standard deviations). The results indicated that the 10-year mean total wet N deposition rates were less than 3 (+/- 3, at 2 sigma) kg/ha/yr (as N) throughout the state. Wet S deposition was less than 1.3 (+/- 1, at 2 sigma) kg/ha/yr (as S) throughout the state.

At the nearest monitoring site (Montague, having co-located NADP/NTN and CADMP samplers), wet S deposition ranged from 0.2 to 0.4 kg/ha/yr as S (equivalently, 0.5 to 1.1 kg/ha/yr as SO_4^{2-}) during the period 1986 through 1994 (Blanchard et al., 1996). The annual NO_3^- and NH_4^+ deposition rates at Montague were each in the range of 0.2 to 0.5 kg/ha/yr as N, yielding a multi-year mean total wet N deposition rate of 0.6 kg/ha/yr (Blanchard et al. 1996).

During the period 1984 through 1990, the annual-average H^+ concentration in precipitation at Quincy was 6.0 µeq/L (pH 5.22) (Blanchard and Tonnessen, 1993). At three other northern California locations (Eureka, Gasquet, and Montague), mean precipitation pH during that period was 5.32 to 5.34.

b. Occult/Dry Deposition

The CADMP co-located wet and dry deposition samplers at ten locations in California (Blanchard et al. 1996). At the three nonurban locations (YOSE, SEQU, and Gasquet, located near REDW), mean dry deposition rates of oxidized N species were roughly comparable to the rates of wet NO_3^- deposition. The nonurban dry S deposition rates were in the range of 0.7 to 1.3 times the rate of wet SO_4^{2-} deposition, while dry NO_3 plus aerosol NH_4^+ deposition rates were approximately 0.4 to 1.7 times the rate of wet NH_4^+ deposition (Blanchard et al., 1996). Because precipitation at LABE is very low, dry deposition of S and N may be proportionately higher at LABE than at the other nonurban sites. Lacking more specific information, it may be concluded that dry deposition rates in LABE are likely of the same order of magnitude or somewhat higher than wet deposition rates.

c. Gaseous Monitoring

One passive ozone sampling site has been established within LABE at an elevation of 1,451 m (Table V-3). The reported mean ozone concentrations are among the lower values recorded by passive ozone samplers within California parks (Table II-13). Statistics for annual hourly maxima, 7-hour maxima, and ozone exposure are not available for LABE. Similarly, SO_2 has not been monitored within LABE, so maximum and mean SO_2 statistics are not available, but are expected to be low.

Table V-3. Summer average hourly ozone concentrations at passive sampling sites within LABE (source: Dr. John D. Ray, National Park Service, Air Resources Division, NPS Passive Ozone website, 1999). Units are ppb.					
	1995	1996	1997	1998	1999
Ozone concentration	41.1	41.6	39.3	42.8	46.6

2. Aquatic Resources

There are no surface waters within LABE. There are, however, a few seeps at some caves and some caves develop ice layers during winter. Most of the scant rainfall and melting snow passes directly through the porous volcanic soils into the water table, which is about 700 ft below the surface.

A water quality study of the ice within caves in the monument was conducted between May and June of 1999. Thirteen caves known to contain ice were selected for study. Water chemistry measurements included alkalinity, pH, NH_4^+, and NO_3^-. Because naturally-occurring free-flowing water is not present within the monument, ice caves are valuable water resources for wildlife. There is some concern that wildlife and humans visiting these caves may be compromising the water quality (Cannon 1999). Measured pH values varied from 6.5 to 8 in the caves that were studied. Minimum alkalinity values reported were 160 µeq/L in four of the caves. Concentrations of NO_3^- and NH_4^+ were generally below detection limits (Cannon 1999).

3. Vegetation

There has been virtually no research or monitoring on the potential effects of air pollution on vegetation in LABE. This is despite the fact that the park has three good ozone bioindicators, Jeffrey pine (*Pinus jeffreyi*), ponderosa pine, and quaking aspen (*Populus tremuloides*), and that ozone injury has been documented at LAVO only 120 km to the south. Although SO_2 and NO_2 are currently not a concern, the pines are also good bioindicators for SO_2 and NO_2, and quaking aspen is a good bioindicator for SO_2.

The sensitivities of plant species at LABE to air pollutants are summarized in Table V-4.

Table V-4. Plant and lichen species of LABE with known sensitivites to sulfur dioxide, ozone, and nitrogen oxides (H=high, M=medium, L=low, blank=unknown).				
Scientific Name	Common Name	Sensitivity		
		SO_2	O_3	NO_x
Gymnosperms				
Abies concolor	White fir	H	M	H
Calocedrus decurrens	Incense cedar		M	
Pinus contorta	Lodgepole pine	H	M	H
Pinus jeffreyi	Jeffrey pine	H	H	H
Pinus monticola	Western white pine	M	M	
Pinus ponderosa	Ponderosa pine	H	H	H
Angiosperms				
Achillea millefolium	Common yarrow		L	
Agastache urticifolia	Nettleleaf giant hyssop		M	

Table V.4. Continued.

Scientific Name	Common Name	Sensitivity		
		SO_2	O_3	NO_x
Amaranthus retroflexus	Redroot amaranth	M		
Artemisia tridentata	Big sagebrush	M	L	
Bromus carinatus	California brome		L	
Bromus tectorum	Cheatgrass		M	
Ceanothus velutinus	Snowbrush ceanothus	L		
Cercocarpus ledifolius	Curlleaf mountain mahogany	M		
Convolvulus arvensis	Field bindweed	H		
Cryptantha circumscissa	Cushion catseye	M	M	
Epilobium brachycarpum	Autumn willowweed		L	
Erodium cicutarium	Redstem stork's bill	M	M	
Festuca idahoensis	Idaho fescue	H		
Gayophytum diffusum	Spreading groundsmoke		H	
Gayophytum racemosum	Blackfoot groundsmoke		L	
Helianthus annuus	Common sunflower	H	L	
Mentzelia albicaulis	Whitestem blazingstar	M	H	
Oryzopsis hymenoides	Indian ricegrass	M		
Phlox hoodii	Spiny phlox	L		
Poa pratensis	Kentucky bluegrass		L	
Populus tremuloides	Quaking aspen	H	H	
Potentilla glandulosa	Gland cinquefoil		M	
Prunus emarginata	Bitter cherry	M		
Prunus virginiana	Common chokecherry	M	M	
Rosa woodsii	Woods' rose	M	L	
Salix scouleriana	Scouler's willow		M	
Stipa comata	Needleandthread	L		
Symphoricarpos oreophilus	Whortleleaf snowberry	M		
Symphoricarpos vaccinioides	Utah snowberry		L	
Tragopogon dubius	Yellow salsify	M		
Vicia americana	American vetch		L	
Lichens				
Rhizoplaca melanophthalma		H		

4. Visibility

As part of the Interagency Monitoring of Protected Visual Environments (IMPROVE) network, visual air quality in LABE was monitored using an automatic 35-mm camera for a five-year period. The camera was located on a butte 1.6 km northwest of the monument headquarters. The camera site path to the northwest included Hamaker Mountain, south of Klamath Falls, Oregon. A summary of the view monitoring photographic data follows.

The camera system operated from August 1986 through September 1991. Color 35mm slide photographs of the Hamaker Mountain vista were taken three times per day until the site was removed.

View monitoring slides document visual conditions and are an effective tool for interpreting the visual effects of measured optical and aerosol parameters or presenting monitoring program goals, objectives, and results to decision-makers and the public. The LABE photographs presented in Figure V-3 were chosen to provide a feel for the range of visibility conditions observed during the period of record.

VI. PINNACLES NATIONAL MONUMENT

A. GENERAL DESCRIPTION

Pinnacles National Monument (PINN) was established by Presidential proclamation in 1908, stating that “the natural formations known as the pinnacle rocks with a series of caves underlying them ... are of scientific interest, and it appears that the public interest would be promoted by reserving these formations and caves as a national monument, with as much land as may be necessary for the proper protection thereof.” The impetus for creation of PINN was preservation of the geologic resources. However, it is recognized that the natural setting and cultural history are also important and integral to management of the monument (Pinnacles National Monument, Draft Research Management Plan). The monument included about 835 ha when it was established. Subsequently, presidential proclamation and land acquisitions increased its size to 5,865 ha by 1941. In 1976 there was an additional legislative change with an act of Congress that designated 5,370 ha of land within PINN as wilderness, 407 ha as potential wilderness, and added about 695 ha to the monument. Adjacent BLM lands have been under formal review for wilderness classification, and for addition to the monument.

In early 2000, President Clinton invoked the Antiquities Act of 1906 to expand the boundary of PINN. The expansion transferred approximately 3,200 ha of BLM land to the NPS, including both Wilderness Study Areas and other lands. This land will improve the park’s ability to protect a functioning and representative chaparral and valley oak savanna ecosystem and allow a variety of visitor use facilities and recreational opportunities. The boundary also extended to the east to include the Pinnacles Ranch. The purchase of the Pinnacles Ranch is expected to occur by early 2001 (Chad Moore, Pinnacles National Monument, pers. comm.)

The monument is located at the southern end of the Gabilan Mountains, part of the inner coastal ranges, 60 km from the Pacific Ocean. The Santa Lucia Mountains to the west prevent PINN from receiving much oceanic influence. Topography is generally steep. Elevation within the monument ranges from about 245 m where Chalone Creek exits the monument to 1,007 m at North Chalone Peak (Figure VI-1).

PINN is managed as a natural area where the principal objective is to provide an environment in which ecological processes are allowed to interact with a minimum of human intervention. Important resources within PINN include the volcanic pinnacle formation, the rock fall or talus caves, and the chaparral covered mountainous terrain with narrow wooded valleys and small streams. Wildflower displays are abundant in spring and many species of mammal, bird, and insect are native to the monument.

Visibility is an important attribute in the visitor experience in PINN. Views of the Pinnacles themselves are at relatively close range within the monument boundaries, thereby minimizing the effect of air pollutants on the quality of the viewing (Pinnacles National Monument 1980). However, the NPS has identified two integral vistas from the monument towards the San Andreas Rift Zone. These integral vistas are affected by pollution.

An important consideration regarding visitor use of this monument is that it is located close to a major metropolitan area and therefore can provide back country experience for the public on a day-use or weekend visit basis. Such a short-term wilderness experience is difficult to achieve throughout most of California. The San Francisco/Oakland/San Jose metropolitan region lies only about two hours driving time to the north of the monument. PINN provides a unique opportunity for millions of metropolitan residents to experience wilderness within a short commuting distance. The highly mobile human population is putting increasingly heavy recreational pressure on the resources in this monument. Annual visitation ranges from about 170,000 to 230,000, about 80% of which is local and regional day travelers.

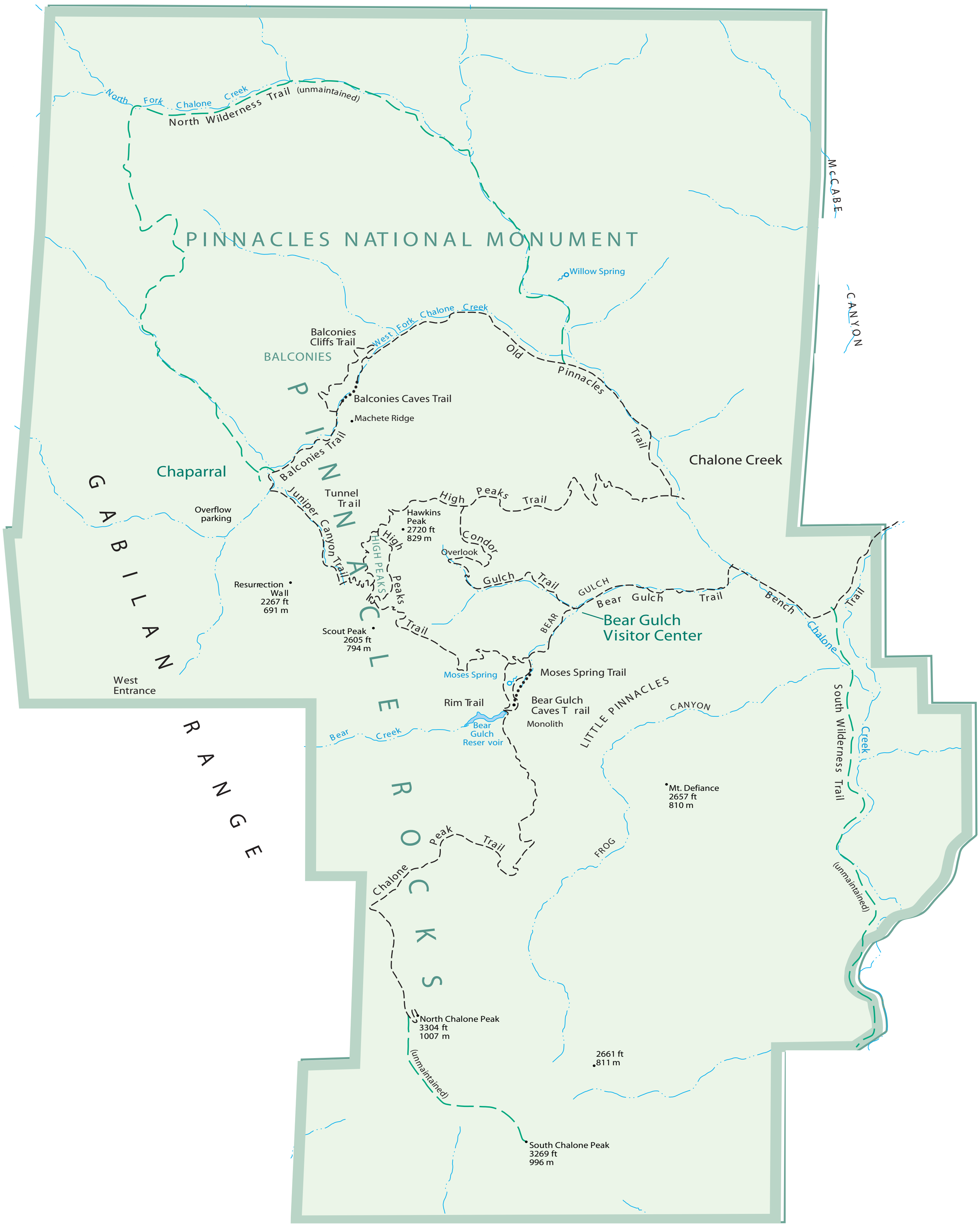


Figure VI-1. Location map for PINN.

The monument contains archaeological, architectural, and historic resources. In pre-historic times it was used by indigenous Costanoans seasonally for hunting, plant gathering, and acorn collecting, but apparently not for permanent habitation.

1. Geology and Soils

PINN occupies an area of both volcanic and earthquake activity. An ancient volcano apparently once rose to an elevation of about 2,400 m, about 1,400 m higher than the monument's current highest point, North Chalone Peak (Rowe 1974). Two large faults developed, one on each side of the monument running roughly in a north-south direction. The central block dropped downward, particularly on the eastern side, and produced a tilt in that direction. Erosional processes and sliding of loose rocks downhill from the rim of the volcano, combined with joint fractures which continued to develop, eventually formed the spires and pinnacles that can be observed in the monument today.

The Pinnacles formation is thus the eroded remnants of a Miocene volcano. Cliffs are composed of rhyolite, dactite, and breccia. Adjacent to chaparral covered granitic mountains, which form most of the Gabilan Range, there are also unconsolidated sandstone cliffs on the eastern edge of the monument. Most of PINN is underlain by volcanic rocks, except the western and southeastern portions of the monument, which are underlain principally by granitic rocks.

The pinnacles are in an advanced stage of decomposition. They represent the effects of weathering, block faulting, and continuous earthquakes over the past 23 million years. The rock that forms the pinnacles is well consolidated breccia, with formations that rise vertically up to several hundred meters.

The monument is located along the eastern edge of the Pacific Plate. The rock within the monument is believed to have originated about 300 km to the south of the monument's present location and then moved northward along the San Andreas Fault at a rate of approximately 3.8 cm per year. Three large faults (Miner's Gulch, Pinnacles, and Chalone Creek) occur within the area of the monument. The Chalone Creek Fault is believed to have been the ancestral main line of the San Andreas Fault, which is now located about 6 km to the east. Because of the faulting and earthquake activity that has occurred within the monument, deep narrow gorges were created where huge boulders have toppled from higher formations and become wedged at various heights above the canyon floor, creating the Bear Gulch and Balconies talus caves (Pinnacles National Monument 1980).

Gabilan limestone, a white coarse-grained marble with some quartzites and schist, is found in thin, isolated bodies within the granitic rocks on the west side of the monument. The granitic rocks consist mainly of granodiorite, with some granite and gneiss on the west and southeast sides of the monument. This unit is intruded by sills and dikes of rhyolitic porphyry. These rocks tend to be moderately to deeply weathered and fractured, and the fractures are commonly filled with calcite.

Caves found in Pinnacles are excellent examples of talus caves. These caves provide important habitat, especially for bats, and also are an important part of the visitor experience to the monument. Upper Bear Gulch Cave is about 520 m long and both Middle Bear Gulch Cave and Balconies Cave are around 160 m long.

The soil at Pinnacles is divided into three soil types that correspond to differences in parent material. Igneous rockland soils are found in the central volcanic block area. East of Chalone Creek and north of Bear Valley Creek is an area characterized as the Laniger series, which is derived from the conglomerates east of the Chalone Creek fault. The granitics that underlie the pinnacles formation are exposed around the periphery of the volcanic block. Soils derived from this parent material are in the Sheridan series. Alluvial soils are found in creek bottoms and

valleys. All soils in the monument are described as well to excessively drained, with erosion potentially severe. Textures range from coarse, sandy loam in the Sheridan series to gravelly, sandy loam in the Laniger series (Pinnacles National Monument 1986). Soils in the monument have little ability to retain nutrients and water.

Soils derived from coarse-grained granites are highly erodible because they contain very little binding material. As a consequence, relatively large volumes of sediment can be deposited at the uphill edge of vegetation barriers or buffer strips following wildfire in chaparral vegetation. These mounds of sediment remain uncompacted for relatively long periods of time. If an intense rainstorm follows a wildfire, very large volumes of sediment can be deposited and transported within the stream channels (Heede 1988). Significant changes to stream channel geometry can occur with a 10-year flood event or larger.

2. Climate

The climate is Mediterranean, with hot dry summers and cool winters. Average maximum temperature ranges from about 16°C in January to above 35°C in July. Average minimum temperatures are about 0.7°C in January and minimum July temperature is 10.3°C . Average daily maxima for the period 1956 through 1982 exceeded 32°C in July, August, and September (Pinnacles National Monument 1986). Summer daytime temperatures over 38°C and nighttime temperatures below 10°C are not uncommon, making for very large temperature swings.

Nearly all of the monument's annual rainfall occurs between October and May. Average rainfall is 42 cm per year and it is not uncommon for the majority to fall during several severe storms. Although most of the rainfall occurs during the winter months, high intensity thunderstorms of short duration occasionally occur during summer (Meyer 1995). There is moderate to severe lack of moisture during the summer months, which has a profound influence on the aquatic and vegetative communities of the monument (Pinnacles National Monument 1999). The soils have poor water holding capacity and most rainfall runs off immediately. Tributary streams are ephemeral and only Chalone Creek contains surface water year-round; portions of its stream bed are also dry from July through September.

The seasonal variability in precipitation is due to the east Pacific high, which acts to shut off precipitation for long periods. Changes in the east Pacific high shift on approximately a six-year cycle, which results in persistent drought for five to eight years, followed by a wet period (Pinnacles National Monument 1999).

3. Biota

The pinnacles rock formations, which cover about 5% of the monument, do not provide good habitat for most vegetation, with the exception of lichens, which grow on the rocks in profusion. About 80% of the monument is covered with coastal chaparral vegetation, 10% is oak woodland, and the remaining 5% is riparian communities. Perhaps the most significant botanical resource in the monument is the extensive coastal chaparral plant community. The coastal chaparral ecosystem provides a rich education and research opportunity for the study of extremely specialized plant and animal communities of types not well preserved elsewhere in the National Park system. Preservation of the scientific value of the chaparral community is equally as important as the pinnacles formations themselves, which historically have been considered the primary resource (Pinnacles National Monument 1976). Preservation of this chaparral community becomes more important as additional areas of California's native vegetation disappear with suburban expansion and grazing pressure. The chaparral vegetation community consists mainly of specialized plants such as chamise (*Adenostoma fasciculatum*), buck brush (*Ceanothus cuneatus*), manzanita (*Arctostaphylos*), and gray pine (*Pinus sabiniana*).

Surveys of the plant communities at PINN were conducted in March 1985 and June and August 1987 by Halvorson and Clark (1989). Physical features of the habitat were described as well as the composition and structure of the vegetation at 87 sites. Of the 560 plant taxa known to occur within the monument, 241 were encountered in this study. Vegetation community descriptions are provided below and are taken from the report of Halverson and Clark (1989).

There are 13 plant communities described within the monument: grassland, coastal sage scrub, chamise chaparral, manzanita chaparral, mixed chaparral, hollyleaf cherry chaparral, California buckeye chaparral, blue oak woodland, gray pine woodland, southern oak woodland, valley oak woodland, riparian woodland, and herbaceous pockets in the rocky uplands. Riparian sites are the most diverse in terms of number of plant taxa, while coastal sage scrub and manzanita chaparral are the least diverse. The existing vegetation map for the monument differs somewhat from the classification of Halvorson and Clark (1989), although the two classifications correspond for some communities.

The grassland community occurs intermingled with oak woodlands and chaparral throughout the monument at elevations between 300 and 830 m. In most places the soils are deep with highly differentiated horizons, although soils are shallower and poorly differentiated on the steeper slopes. Fifty-nine plant taxa were recorded. The dominant taxa were non-native grasses, including *Avena barbata, Bromus mollis,* and *Bromus rubens*. The coastal sage scrub community is restricted to widely scattered, dry, west facing slopes at mid elevations, between about 440 and 890 m. Thirty-four taxa were recorded. The dominant taxa were wild buckwheat (*Eriogonun fasciculatum*) and black sage (*Salvia mellifera*), with various understory plants between the shrubs. The mixed chaparral plant community is found on steep north-facing slopes. It is also found on slopes of all orientations where deeper soils have developed. The elevation range of the sites sampled was 353 to 829 m. It is one of the most diverse plant communities sampled, with a total of 74 taxa encountered. The dominant shrubs were chamise, buck brush, wild buckwheat, bigberry manzanita (*Arctostaphylos glauca)*, and hollyleaf cherry (*Prunus ilicifolia*). The chamise chaparral community occurs primarily on south-facing slopes between 350 and 900 m elevation. A total of 52 taxa were observed, dominated by chamise. The species comprises a dense, apparently single-aged stand covering about 70% of the ground surface with a mean height of 2.3 m. The Manzanita chaparral community occurs on east and south-facing slopes at elevations between 305 and 585 m. Twenty-six plant taxa were observed. The community is characterized by manzanita (*Arctostaphylos* spp.), but the dominant shrub was chamise, with a mean cover of 58.5%. Buck brush was also an important associate. Hollyleaf cherry chaparral occurs in minor drainages on north-facing slopes and on talus slopes below large rock formations at lower to mid-elevations between about 320 and 720 m. Sixty-two plant taxa were observed, dominated by hollyleaf cherry, buck brush, and scrub oak (*Quercus dumosa*). There were scattered gray pine and coast live oak (*Quercus agrifolia*) in the hollyleaf cherry chaparral community. They accounted for a mean cover of 17%.

The California buckeye woodland community is found in the northeastern section of the monument, on steep, north-facing slopes that are relatively moist and cool. The elevational range of this community is narrow, and is restricted to below 400 m. Sixty-three plant taxa were observed. The dominant tree species were California buckeye (*Aesculus californica*) and scrub oak. Dominant shrubs included hollyleaf cherry, redberry (*Rhamnus crocea*) and buck brush. Blue oak woodland community is found throughout the monument, between about 330 and 940 m elevation, primarily on north and west-facing slopes on gentle terrain and on alluvial material near the valley bottoms. It has an open canopy of blue oak (*Quercus douglasii*) and gray pine. It is floristically one of the more diverse plant communities in the monument. The southern oak community was found only at two locations, at the Chalone Creek picnic area and the Bear

Gulch picnic area. A total of only 20 plant taxa were observed in this community, which is dominated by interior live oak (*Quercus wislizenii*). Gray pine woodland is found throughout the monument, on east, southeast, and occasionally northeast facing slopes between elevations of about 300 and 760 m. It is found in a variety of ecological situations and is dominated by an open canopy of gray pine. There is also a well-developed understory of chaparral or coastal sage scrub taxa. The most common shrubs in this community were buck brush, chamise, and wild buckwheat.

The valley oak plant community was found at one site on the bench at the confluence of Sandy Creek and Chalone Creek. It was dominated by valley oak (*Quercus lobata*), coast live oak, and non-native grasses. It comprises an open savanna with widely spaced oak trees having large canopies. The riparian woodland community was found along water courses between 290 and 450 m elevation. A number of tree species were found, including willow (*Salix* spp.), California sycamore (*Platanus racemosa*), and gray pine. The most important shrub taxa were poison oak (*Toxicodendron diversilobum*) and wild buckwheat. This was the most species-rich plant community within the monument, with a total of 81 plant taxa recorded.

Outside the riparian zone, all plant species in the monument have adapted to a four to five month drought period. Most species grow during late winter and spring and are dormant during the hot and dry summer. Many perennial shrubs have leathery, evergreen leaves to reduce water loss. Some species, including California buckeye, mountain mahogany (*Cercocarpus betuloides*), black sage, deerweed (*Lotus scoparius*), and coastal sage (*Artemesia californica*) minimize water loss by dropping all or part of their leaves during the drought period. Herbaceous plant species typically complete their life cycle before June each year.

Significant ground disturbance in California buckeye, blue oak woodland, and riparian areas is caused by rooting by feral pigs, with less pig disturbance in other communities. Fire exclusion may also have contributed to some changes in species distribution and abundance and to fuel loading (Biswell 1976), especially in the woodland communities. While exotic species comprise only 15% of the taxa present, they account for up to 70% of plant cover in some communities, with the greatest effect in woodlands and grasslands.

The monument contains an abundant lichen flora, both in number of species present and biomass. Ninety-three lichen species were listed from the monument by Smith (1987). These were generally relatively widespread common species. Forty-two of these are fruticose species, nine are foliose, and 42 are crustose. None of the specimens collected by Smith (1987) showed signs of bleaching or other symptoms of air pollution damage. Also because some species known to be particularly sensitive to air pollution were well developed in the monument, Smith concluded that the air quality was excellent.

The monument supports a wide variety of mammals, reptiles, birds, amphibians, and insects. About 30 mammal species occur in the monument and there are about 75 resident bird species. Mammals include the coast black-tailed deer (*Odocoileus hemionus*), bobcat (*Felis rufus*), racoon (*Procyon lotor*), grey fox (*Urocyon cinereoargenteus*), and coyote (*Canis latrans*). Infrequent sightings of mountain lion (*Felis concolor*) and black bear (*Ursus americanus*) are also reported. Twenty-two species of reptile have been recorded within the monument, including the western pond turtle (*Clemmys marmorata*), eight species of lizard, and thirteen species of snake (Fellers 1986). The latter include the Pacific coastal rattlesnake (*Crotalus viridis*) and the gopher snake (*Pituophis melanoleucus*).

Federally-listed threatened and endangered (or formerly endangered) animal species within the monument include the California red-legged frog (*Rana aurora draytoni*) and peregrine falcon (*Falco peregrinus*). In addition, there are 21 other animal species that are California Species of Special Concern.

Available information on mammalian fauna within the monument focuses primarily on Townsend's big-eared bat (*Corynorhinus townsendii*) and wild pig (*Sus scrofa*). Townsend's big-eared bat was designated as a species of special concern in California in 1986 (Williams 1986). There are 37 known colonies in California with a total population of about 4,250 adult females (Pierson and Rainey 1998). A maternity colony was discovered in a cave within the monument in June of 1997. There are an additional 15 species of bat suspected to reside at PINN, but only 13 additional species have been verified to date.

Wild pigs constitute a significant ecological problem in PINN, mainly because of habitat disturbance in the riparian communities of the monument. Pigs were introduced by European settlers as livestock in California in the mid 1700s. In the early 1920s, European wild boars were imported into Monterey County. Some animals escaped and dispersed into central coastal areas where they bred with feral domestic pigs. Wild pigs have subsequently increased in numbers and expanded their range, especially in coastal counties from Humboldt to Santa Barbara (Waithman 1995). Currently, a fence is under construction around PINN to completely exclude pigs from the monument.

A small mammal study between November, 1984 and May 1986 noted 15 species of small mammal in PINN, including 14 species of rodent and 1 species of shrew (Fellers and Arnold 1988).

Over 140 bird species have been recorded in the monument. An estimated 60 species nest in PINN and about 45 species are considered year-round residents. Raptor nesting at PINN during the period of 1984 to 1992 was summarized by Rechtin (1992). PINN has one of the more dense concentrations of prairie falcons (*Falco mexicanis*) within California. A large number of raptors nest within the monument, including red-tailed hawk (*Buteo jamaicensis*), red-shouldered hawk (*Buteo lineatus*), black-shouldered kite (*Elanus caeruleus*), coopers hawk (*Accipiter cooperii*), sharp-shinned hawk (*A. striatus*), American kestrel (*Falco sparverius*), long-eared owl (*Asio otus*), barn owl (*Tyto alba*), great horned owl (*Bubo virginianus*), northern pygmy owl (*Glaucidium gnoma*), western screech owl (*Otus kennicottii*), and turkey vulture (*Cathartes aura*). Sightings have also been made of other raptor species, which have not been recorded to nest in the monument. These include osprey (*Pandion haliaetus*), bald eagle (*Haliaetus leucocephalus*), Swainson's hawk (*Buteo swainsoni*), ferruginous hawk (*Buteo regalis*), rough legged hawk (*Buteo lagopus*), merlin (*Falco columbarius*), and northern harrier (*Circus cyaneus*). Cliffs at PINN provide excellent nesting habitat for raptors.

Banta and Morafka (1968) provided an annotated check list of recent amphibians in PINN and Bear Valley. Oak salamanders (*Andeides lugubris lugubris*) occur in the coast live oak and gray pine forests and in riparian and moist chaparral environments. The northern slender salamander (*Batrachosets attenuatus attenuatus*) is the most common salamander at PINN. It is abundant in strips of riparian woodland along Bear Gulch, Chalone Creek, West Fork Creek, and along Sandy Creek between the monument entrance and California Highway 25. It is also found in moist chaparral above the riparian strips. Other amphibian species recorded for the monument include the Monterey salamander (*Ensatina eschscholtzi eschscholtzi*), Hammond's spade foot toad (*Scaphiopus hammondi hammondi*; no recent sightings), California toad (*Bufo boreas halophilus*), desert tree frog (*Pseudacris regilla desertitola*), California red-legged frog, and foothill yellow-legged frog (*Rana boylei*; Banta and Morafka 1968).

The three spine stickleback (*Gasterosteus aculeatus*) is abundant in Chalone and Bear Creeks. Sacramento perch (*Archopiltes interruptus*) was also observed at several locations along Chalone Creek by Ely (1993). Although this species is native to the Salinas and Pajaro River systems, its presence in the monument may have been aided by man (Ely 1993). Mosquito fish (*Gambusia*) were also found in Chalone Creek downstream from its junction with Grassy

Canyon. Minor populations of green-eared sunfish exist in isolated pools along Chalone Creek (Pinnacles National Monument 1999).

Bees are abundant in chaparral (Force 1990). This is likely due to the absence of low ground cover in mature chaparral, which leaves room for nesting sites of ground nesting bees, and also because frequent fires have historically provided dead branches for wood nesting types.

Preliminary samples of the bee fauna were collected in 1996 at PINN as part of a larger bee survey in western North America. The results suggested remarkable diversity of bees within the monument and prompted a more intensive and systematic effort in 1997. This effort nearly doubled the total collected bee fauna for PINN, bringing the total number of species to 410. Thirteen of those were previously undescribed and a few are known only from PINN. All bee families present within the United States are represented within the boundaries of the monument. Also, a plot of the cumulative number of species collected versus cumulative collection effort suggested that a large number of species are present in the monument and have not yet been collected. PINN may be near the top of the list for bee diversity in North America (Pinnacles National Monument, undated summary of bee survey; Pinnacles National Monument 1999; Terry Griswold, USDA Bee Systematic and Biology Lab, Logan, UT, pers. comm. [via Chad Moore]). The potential effects of air pollution on this outstanding resource are not known.

4. Fire

Fire is an integral part of California chaparral vegetation dynamics. In the past, lightning was the primary source of ignition. In addition, Native Americans burned chaparral for a variety of reasons, including to improve access and conditions for wildlife and hunting, and to preserve the oak trees in the foothill woodland areas for their acorns (Biswell 1976). Due to fire suppression activities, the mosaics of age classes disappeared from the chaparral vegetation type and fuels became more uniformly heavy and widespread.

The substantial fire regime at PINN was historically regulated by flamability patterns in the chaparral and oak woodland communities and by lightning frequency in the Gabilan and Diablo Mountains. Because lightning is an infrequent occurrence within PINN, most natural fires burned into the monument from outside (Greenlee and Moldenke 1982). However, modern practices have resulted in the establishment of barriers to the spread of natural fire in the region. Modification of surrounding vegetation through agriculture and range management, road construction, and fire management policies, have virtually eliminated the source of natural fire to the monument. Prescribed fire must therefore be applied by park managers in order to maintain fire's role in the ecological development of PINN's vegetational communities (Pinnacles National Monument 1986). The management plan for restoring fire in chaparral vegetation at PINN was summarized by Biswell (1976).

B. EMISSIONS

PINN is located on the boundary of San Benito and Monterey counties, in the North Central Coast Air Basin (NCCAB). The NCCAB is not heavily urbanized, but does include the cities of Monterey, Santa Cruz, Salinas, and other communities within the Monterey Bay region. This area is growing rapidly. Emissions from counties within 140 km of PINN are shown in Table VI-1. Emissions from counties in the San Francisco Bay Area Air Basin (SFBAAB) that are within 140 km of PINN are also included in the table, as pollutant transport is known to occur between the SFBAAB and the NCCAB. The principal species of concern are ozone precursors (NO_x and ROG) and PM. SO_2 emissions are not high. The major point sources of emissions (those that emit at least 100 tons/year of ROG, NO_x, PM_{10}, or SO_2) in the NCCAB are located

Table VI-1. 1995 Emissions from counties within 140 km of PINN. (Source: CARB Almanac, 1999b; SO_x from CARB Emissions Website, 1999a.) Units are 1000 tons/year.

County	NO_x	ROG*	PM_{10}	CO	SO_x
Monterey	20.8	19.7	14.2	102.9	1.1
San Benito	2.6	1.8	4.4	11.3	0.0
Santa Cruz	6.9	10.6	4.7	51.8	0.4
San Mateo**	22.6	23.0	8.4	165.3	0.7
Santa Clara**	46.0	48.5	17.9	333.2	1.5
Alameda**	43.1	43.1	13.1	283.2	3.3

* Reactive Organic Gases
** County (or portion) is in adjacent air basin in an area where transport between air basins occurs

near communities that are not adjacent to PINN (Davenport, Santa Cruz, Moss Landing, and San Ardo; Figures II-3 through II-6). Within Monterey and San Benito counties, stationary sources accounted for 18% of ROG emissions, 28% of NO_x emissions, and 5% of PM_{10} emissions in 1996 (CARB 1998b).

An inventory of in-park emissions has recently been compiled by the NPS-Air Resources Division. The results are presented in Table VI-2.

C. MONITORING AND RESEARCH ACTIVITIES

1. Air Quality

Air quality monitoring within PINN includes particulate matter ($PM_{2.5}$ and PM_{10}), hourly ozone, and SO_2 (Table VI-3). An NADP sampler was added in 1999. Meteorological data are collected at Bear Gulch. From mid-1984 through mid-1988, a CADMP wet deposition site was located at Salinas, approximately 40 km northwest of the park boundary. A CASTNet site is located within the park for monitoring dry deposition. Deposition is not monitored directly, but is rather calculated from ambient concentration measurements. The nearest CADMP dry deposition monitor was located in Fremont, in the San Francisco Bay area; data from that urban location are unlikely to be representative of deposition rates within PINN.

a. Wet Deposition

An NADP/NTN wet-deposition monitoring site began operating within PINN in November 1999. At present, the available data are limited. Blanchard et al. (1996) estimated annual and 10-year wet deposition rates throughout California by interpolating the observations from all NADP/NTN, CADMP, and special-studies monitoring locations for the period 1985 through 1994; they also estimated interpolation uncertainties. Upper-bound deposition values for unmonitored areas may be estimated as the sum of the interpolated values plus twice the interpolation uncertainty (i.e., mean plus two standard deviations). The results indicated that the 10-year mean total wet N deposition rates were less than 3 (+/- 3, at 2 sigma) kg/ha/yr (as N) throughout the state. Wet S deposition was less than 1.3 (+/- 1, at 2 sigma) kg/ha/yr (as S) throughout the state.

Table VI-2 . Summary of 1998 stationary and area, and mobile source emissions (tons/yr) at PINN.

Activity	Particulates	Sulfur Dioxide	Nitrogen Oxides	Carbon Monoxide	VOCs	HAPs[1]
Stationary and Area Source Emissions						
Stationary Combustion Sources						
Heating units	0.00	0.00	0.04	0.01	0.00	0.00
Generators	0.00	0.00	0.00	0.00	0.00	0.00
Woodstoves	0.41	0.00	0.04	3.06	1.15	0.00
Combustion Emission Subtotal	0.41	0.00	0.08	3.07	1.15	0.00
Fuel Storage Tanks						
Gasoline/Diesel Fuel Tanks	0.00	0.00	0.00	0.00	0.38	0.05
Area Sources						
Prescribed Burning	60.75	0.00	0.17	305.25	37.88	0.00
Area Source Emission Subtotal	60.75	0.00	0.17	305.25	37.88	0.00
TOTALS	61.16	0.00	0.25	308.32	39.40	0.05
Mobile Source Emissions						
Road Vehicles						
Visitor Vehicles	0.42	0.00	0.13	1.30	0.10	–
NPS/GSA Road Vehicles	0.05	0.00	0.03	0.16	0.01	–
Vehicle Emission Subtotal	0.47	0.00	0.16	1.47	0.11	–
Nonroad Vehicles						
NPS Nonroad Vehicles	0.01	0.00	0.01	1.73	0.21	–
TOTALS	0.48	0.00	0.17	3.20	0.32	–

[1] Hazardous air pollutants, based on the list compiled by EPA

Table VI-3. Air quality monitoring at PINN.

Species	Site within park	Site within 50 km
Ozone, hourly	CASTNet	
Ozone, passive		
SO_2	NPS	
PM_{10}	IMPROVE	
$PM_{2.5}$	IMPROVE	
Wet deposition	NADP*	ARB**
Dry deposition	CASTNet	
Visibility	IMPROVE	

* New site
** Closed before 1994

At the nearest CADMP site to PINN (Salinas), wet S deposition ranged from 0.4 to 0.5 kg/ha/yr as S (equivalently, 1.1 to 1.7 kg/ha/yr as SO_4^{2-}) during the period 1984 through 1988 (Blanchard et al., 1996). Because Salinas is located near the ocean, some SO_4^{2-} derives from marine aerosol; the amounts of wet SO_4^{2-} deposition not of marine origin ranged from 0.2 to 0.3 kg/ha/yr as S (Blanchard et al. 1996).

The annual NO_3^- and NH_4^+ deposition rates at Salinas were in the ranges of 0.2 to 0.3 kg/ha/yr and 0.4 to 0.6 kg/ha/yr as N, respectively, yielding a multi-year mean total wet N deposition rate of 0.8 kg/ha/yr (Blanchard et al., 1996). During the period 1984 through 1988, the annual-average H^+ concentration in precipitation at Salinas was 3.2 µeq/L (pH 5.49) (Blanchard and Tonnessen, 1993).

b. Occult/Dry Deposition

The CASTNet dry-deposition monitoring site began operating May 16, 1995. The monitoring instrument measures ambient concentrations of gases and particles, and EPA uses a computer model to calculate the dry-deposition rates from the measurements. The first calculations of dry-deposition rates were released by EPA in November 2000. For the years 1996 through 1999, the calculated annual dry-deposition rates of N and S ranged from 1.3 to 1.6 kg N/ha/yr and 0.30 to 0.34 kg S/ha/yr, respectively. When combined with wet deposition measurements from the NADP/NTN network, the data indicate that the annual total deposition rates of N and S ranged from 3.0 to 5.5 kg N/ha/yr and 0.7 to 1.4 kg S/ha/yr, respectively, over the period 1996 through 1999 (a new NADP/NTN site is located 0.3 km from the dry-deposition monitor, but did not begin operating until November 1999). The average total N and S deposition rates over the four-year period were 4.1 kg N/ha/yr and 1.0 kg S/ha/yr, respectively.

c. Gaseous Monitoring

Data from the hourly ozone monitor within the park showed that ozone concentrations occasionally exceeded the federal hourly ozone standard (120 ppb) and consistently violated the state hourly standard (90 ppb) (Table VI-4). Among the ten ozone monitors in the NCCAB, both the highest hourly ozone maximum and the highest mean daily maximum ozone occurred at PINN in 1997 (CARB 1997a). The maximum 9 am - to - 4 pm ozone concentrations at PINN were in the range of 77 to 95 ppb (Table VI-4). Between 1990 and 1997, 8-hour ozone maxima at PINN were 88 to 109 ppb, suggesting that compliance with the federal 8-hour ozone standard (80 ppb, 3-year average of fourth-highest annual values) may be problematic; the annual 4th-highest 8-hour ozone values during that time period were 77 to 95 ppb (Alexis et al., 1999). No passive ozone samplers have been sited within PINN.

Table VI-5 displays 24-hour integrated sample maximum and mean SO_2 for the period 1988-1996. SO_2 levels at PINN are low, with annual means of 0.05 to 0.14 ppb and maxima ranging from 0.26 to 0.59 ppb. SO_2 measurements were discontinued after 1996 due to concerns about their accuracy. The measurements are considered sufficiently accurate to show that SO_2 concentrations were well below the levels at which plant injury has been documented, ~40 to 50 ppb 24-hour average and 8-12 ppb annual average (Peterson et al, 1992).

Table VI-4. Summary of ozone concentrations and exposure from PINN monitoring sites (Source: Joseph and Flores, 1993; National Park Service, Air Resources Division 2000).

Year	Maximum Daily 1-hour Value (ppbv)	2nd Highest Daily 1-hour Value (ppbv)	Number of Daily Maximum 1-hour Values Greater Than or Equal to 125 ppb	3-Year Average Number of Exceedences	Maximum 9am-4pm Average (ppbv)	Sum06 (ppbv-hour)[a]	Number of Valid Hours of Ozone Measurements
1987	146	140	5	na	103	45,000	6099
1988	127	121	1	na	84	25,000	8005
1989	138	107	1	2.3	80	25,000	7599
1990	121	112	0	.7	94	21,000	7719
1991	139	103	1	.7	96	33,000	7451
1992	108	108	0	.3	82	14,000	6904
1993	110	104	0	.3	83	25,000	7954
1994	97	95	0	0	77	23,000	8150
1995	138	110	1	.3	82	30,000	8050
1996	120	118	0	.3	95	41,000	8102
1997	112	92	0	.3	86	21,000	8125
1998	124	113	0	0	85	32,000	7918
1999	107	105	0	0	80	29,000	7855

[a] Maximum 8 am - 8 pm 90-day rolling average

Table VI-5. Maximum and mean SO_2, from 24-hour resolution samples at PINN. Samples are collected every 3-4 days, unless noted. (Source: NPS Air Resources Division). Units are ppb.

SO_2	1988	1989	1990	1991	1992	1993	1994	1995	1996
Maximum	0.26*	0.34	0.59	0.37	0.33	0.34	0.47	0.32*	na
Mean	0.09*	0.1	0.14	0.1	0.1	0.1	0.1	0.05*	na

na Not available
* Less than 50 samples collected for the year

2. Aquatic Resources

a. Water Quality

Chalone Creek is the predominant drainage in the southern Gabilan Range and flows the length of PINN from the northwest to the southeast corners. Most of the monument occurs within the Chalone Creek watershed and its tributaries include the North and West Forks of Chalone Creek, Sandy Creek, and Frog Creek. The West Fork of Chalone Creek and Frog Creek are intermittent streams and a limited amount of surface water can be found during the dry

season. Sandy Creek is perennial, although some stretches of it lack surface water during the dry season and it has very low flow during late summer. Chalone Creek is perennial in some stretches and intermittent in others (Ely 1993). Bear Gulch Reservoir is stagnant for much of the summer and is subject to eutrophication (Pinnacles National Monument 1999).

Periodically heavy rains cause extensive flooding within the monument. There have been three large floods in the Chalone Creek watershed during the past two decades, including a recent 40-year flood event in 1998. These have caused millions of dollars in damage to park facilities. Some sections of stream reach experienced considerable erosion, whereas others experienced high sedimentation. Some of the physical characteristics that make this watershed prone to erosion and flash flooding also make it susceptible to water quality degradation. Alluvial sands conduct water, and also potentially pollutants, quickly with little buffering. Also, the narrow canyons have forced human facilities to be located very close to surface waters.

Several water quality samples have been collected from Chalone Creek and analyzed for a number of water quality variables. Available data illustrate that Chalone Creek is very well buffered, with specific conductance values in excess of 200 μS/cm. Calcium concentrations are higher than 1,000 μeq/L (Chad Moore, Pinnacles National Monument, pers. comm.). In 2000, the park began monitoring stream macroinvertebrates as an indicator of water quality and stream health.

PINN does not exhibit any of the characteristics typically associated with acid-sensitive surface waters. Available data, although limited, support the likelihood that surface waters in PINN are not sensitive to potential acidification from any amount of S and/or N deposition that could reasonably occur in the foreseeable future.

b. Aquatic Biota

Ely (1993) conducted field surveys to determine the current status and distribution of the California red-legged frog and the foothill yellow-legged frog in PINN. An additional objective of the surveys was to locate California toads on an incidental basis. Ely (1993) did not observe any foothill yellow-legged frogs or larvae during the field surveys. Widespread habitat of this species is not present within the monument and its presence is in doubt. The western toad appears to be present along the length of Chalone Creek, although certain reaches are more suitable for reproduction than others.

The California red-legged frog was relatively abundant along Chalone Creek and was observed in locations along the entire length of this drainage where surface water and proper pool formation were present (Ely 1993). This species was also documented on the West Fork of Chalone Creek, Frog Canyon, and Lower Bear Creek. Despite numerous historical records and voucher specimens of this species from Bear Gulch Reservoir and Bear Creek, however, Ely (1993) saw no California red-legged frogs or larvae at those locations. Pacific tree frog *(Pseudacris regilla*) was the only amphibian species observed at the reservoir.

The status of the California red-legged frog, which is now a federally listed species, within the monument is currently an issue of concern. Surveys in 1994 yielded a total of 70 adults and over 3,000 tadpoles in the monument. Repeat surveys in 1998, however, found only 7 adults and no eggs or tadpoles. Data are not available to determine whether these differences were attributable to climatic fluctuations, natural variability, or actual population decline. Riparian disturbance by feral pigs and the destruction or elimination of shoreline vegetation was judged to heavily increase the predation potential on the California red-legged frog, particularly the young of the year (Ely 1993).

3. Vegetation

There has been virtually no research or monitoring on the effects of air pollution on vascular plants in PINN. A study of the lichen flora was conducted by Smith (1987), who documented 93 species of lichens in the park. As part of this study, the health of lichens in PINN were evaluated in order to determine if there was any evidence of injury from air pollution. All lichen species examined were judged to have robust growth with high fertility and no bleaching, indicating that there was no injury. The high diversity of species and abundance of individual species also suggested that there had been no selective effects on sensitive species.

Little work has been done on the potential effects of N deposition on semi-arid, nonforested ecosystems, such as those found at PINN. There is concern, however, that atmospheric deposition of plant-available forms of N (NH_4^+, NO_3^-) may play an important role in the ongoing replacement of native plant species with exotic, invasive annuals in some coastal sage scrub vegetation communities of California (c.f., Allen et al. 1996, Padgett et al. 1999). Dry deposition of N may be a substantial contributor of inorganic N to such ecosystems, which makes quantification of N input difficult. Padgett et al. (1999) documented changes in soil inorganic N along a depositional gradient from an estimated 35-45 kg N/ha/yr adjacent to Los Angeles to sites about 70 km to the south that exhibited atmospheric concentrations of N (and presumably N deposition) about one third of the levels inland of Los Angeles. The summer soil surface NO_3^- concentrations were near detection limits at the less polluted sites, but in the range of 50-60 µg N/g soil under highly polluted conditions (Padgett et al. 1999). Such changes in available N may have large impacts on vegetation communities, including the coastal sage scrub community at PINN. Dry (and therefore total) N deposition has not been well-quantified at PINN, but it is believed to be only a small fraction of the total N deposition received in the vicinity of Los Angeles.

The sensitivities of plant species at PINN to air pollutants are summarized in Table VI.6.

Table VI.6. Plant and lichen species of PINN with known sensitivities to sulfur dioxide, ozone, and nitrogen oxides (H=high, M=medium, L=low, blank=unknown).

Scientific Name	Common Name	Sensitivity		
		SO_2	O_3	NO_x
<u>Angiosperms</u>				
Achillea millefolium	Common yarrow		L	
Ailanthus altissima	Tree of Heaven		H	
Amaranthus retroflexus	Redroot amaranth	M		
Artemisia douglasiana	Douglas' sagewort		H	
Artemisia dracunculus	Wormwood		M	
Bromus rubens	Foxtail brome	M	M	
Clematis ligusticifolia	Western white clematis	M		
Convolvulus arvensis	Field bindweed	H		
Descurainia pinnata	Western tansymustard	M	M	
Elymus glaucus	Blue wildrye		H	
Erodium cicutarium	Redstem stork's bill	M	M	
Lemna minor	Common duckweed	L		
Lupinus concinnus	Scarlet lupine	M	M	
Mimulus guttatus	Seep monkeyflower		L	
Platanus racemosa	California sycamore		M	

Table VI.6. Continued.

Scientific Name	Common Name	Sensitivity		
Platystemon californicus	California creamcups	M	M	
Poa annua	Annual bluegrass	H	L	
Potentilla glandulosa	Gland cinquefoil		M	
Rumex crispus	Curly dock		L	
Salvia columbariae	Chia	M	M	
Sambucus mexicana	Blue elder		H	
Taraxacum officinale	Common dandelion		L	
Thysanocarpus curvipes	Sand fringepod	M	M	
Veronica anagallis-aquatica	Water speedwell		L	
Vulpia octoflora	Sixweeks fescue	L	L	
<u>Lichens</u>				
Acarospora chlorophana		H		
Buellia punctata		M		
Caloplaca holocarpa		M		
Candelaria concolor		H		
Candelariella vitellina		M		
Cladonia chlorophaea		M		
Evernia prunastri		M	H	
Hypogymnia imshaugii		M	M	
Lecanora muralis		M		
Leptogium californicum			M	
Letharia vulpina		L	L	
Melanelia glabra			L	
Parmelia sulcata		M	H	
Peltigera canina		L	H	
Physcia adscendens		M		
Physcia aipolia		M		
Physcia biziana			L	
Physcia stellaris		M		
Physconia detersa		H	L	
Physconia grisea			L	
Ramalina farinacea			H	
Ramalina menziesii			H	
Rhizocarpon geographicum		L		
Tuckermannopsis merrillii			M	
Usnea subfloridana		H	H	
Xanthoparmelia cumberlandia		H		
Xanthoria candelaria		H	H	
Xanthoria fallax		H	L	
Xanthoria polycarpa		H	L	

4. Visibility

As part of the Interagency Monitoring of Protected Visual Environments (IMPROVE) network, visual air quality at PINN has been monitored using an aerosol sampler, transmissometer, and camera. The aerosol sampler has operated from March 1988 through the present at its location approximately one-half mile southwest of the east entrance to the monument. The transmissometer operated from March 1988 through August 1993, when monitoring was discontinued due to IMPROVE network funding limitations. The transmissometer site path viewed from north to southeast along the Chalone Creek valley. The automatic 35mm camera operated from August 1986 through March 1988, viewing east towards the San Benito Mountains. The vista was changed to view Hawkins Peak to the north in March of 1988. Pictures of the new vista continued to be taken until January of 1995. Data from this IMPROVE site have been summarized to characterize the full range of visibility conditions for the March 1988 through February 1999 period, based on seasonal periods (Spring: March, April, and May; Summer: June, July, and August; Autumn: September, October, and November; and Winter: December, January, and February) and annual periods (March through February of the following year, e.g., the annual period of 1998 includes March 1998 through February 1999). Complete descriptions of visibility characterization, mechanisms of sources and visibility impacts, and IMPROVE monitoring techniques and rationale are provided in Chapter I.

a. Aerosol Sampler Data - Particle Monitoring

A tabular and graphic summary of average reconstructed extinction values by season and year for the March 1988 through February 1999 period are provided in Table VI-7 and Figure VI-2, respectively.

Table VI-7. Seasonal and annual average reconstructed extinction (b_{ext}; Mm^{-1}) at PINN, March 1988 through February 1999.

Year	Spring (Mar, Apr, May)	Summer (Jun, Jul, Aug)	Autumn (Sep, Oct, Nov)	Winter (Dec, Jan, Feb)	Annual (Mar - Feb)[a]
1988	49.2	47.3	46.0	46.3	47.6
1989	44.6	45.7	58.3	58.0	52.0
1990	52.6	47.1	48.1	64.5	52.5
1991	43.3	44.0	56.3	44.5	47.4
1992	53.6	44.7	49.3	34.4	45.7
1993	44.2	43.2	52.8	40.8	45.8
1994	44.8	43.1	41.4	38.2	42.2
1995	42.0	44.9	51.5	35.2	44.0
1996	43.7	45.0	37.6	33.4	40.3
1997	42.2	37.9	39.9	27.8	37.8
1998	37.9	42.2	44.4	38.7	40.6
Mean[b]	45.3	44.1	47.8	42.0	45.1[c]

[a] Annual period data represent the mean of all data for each March through February annual period.
[b] Combined season data represent the mean of all seasonal means for each season of the March 1988 through February 1999 period.
[c] Combined annual period data represent the mean of all combined season means.

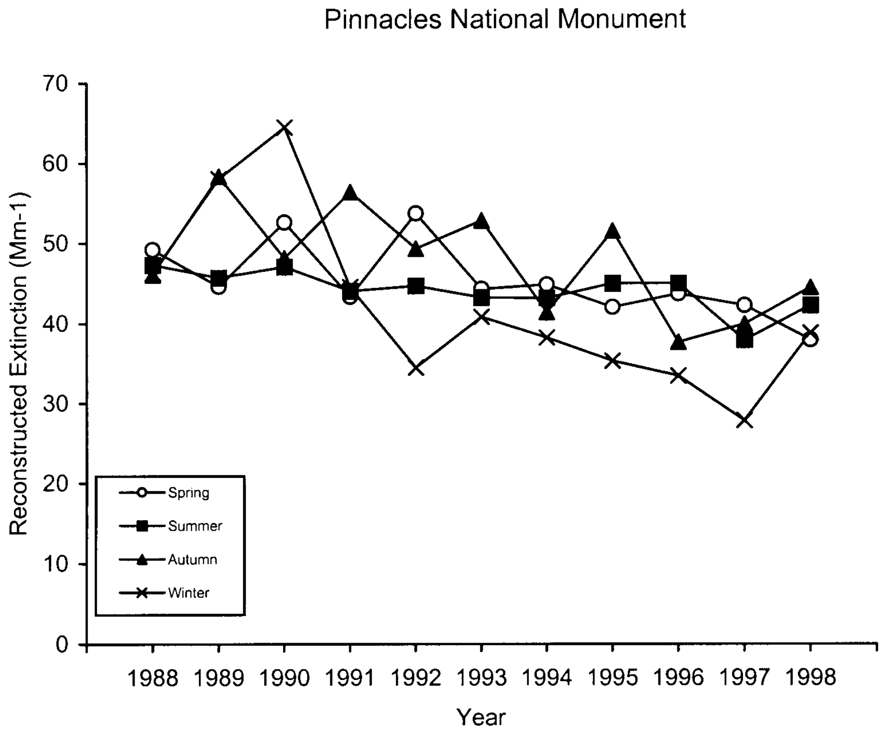


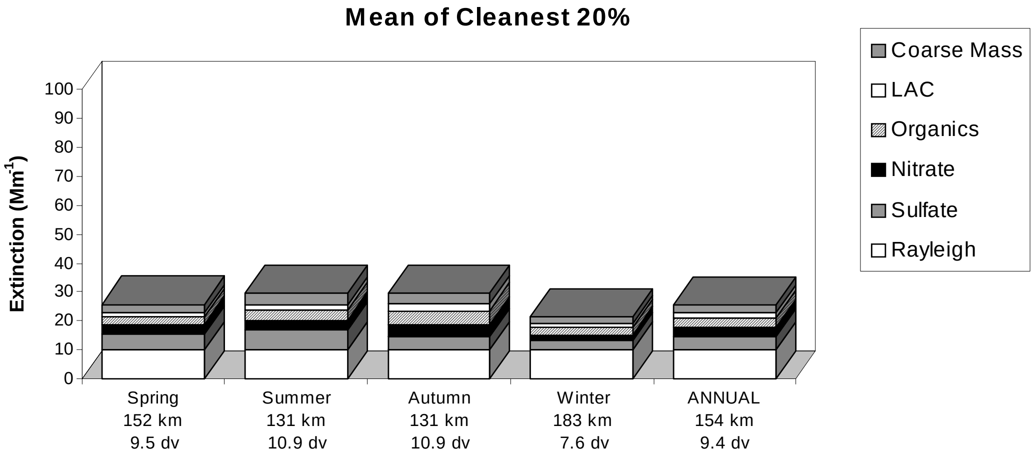
Figure VI-2. Seasonal average reconstructed extinction (Mm^{-1}) at PINN, March 1988 through February 1999.

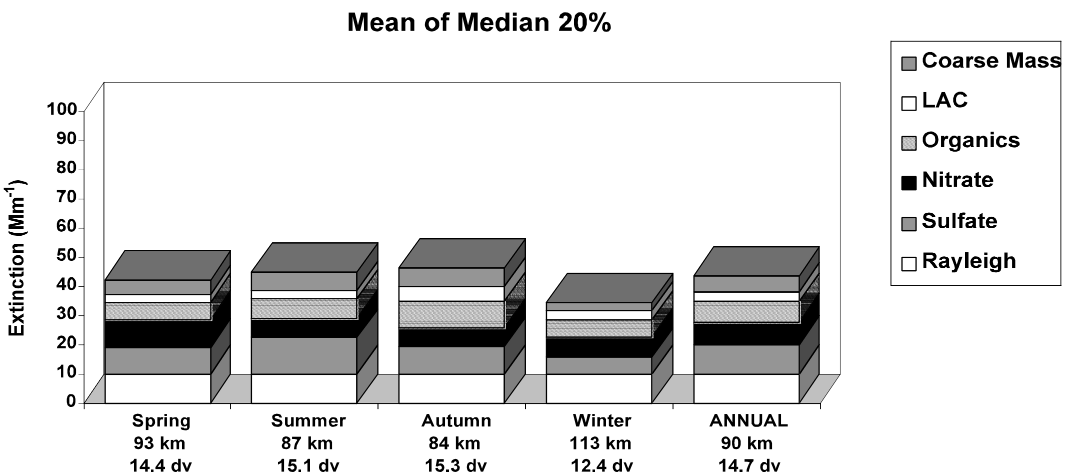
Reconstructed extinction budgets generated from aerosol sampler data apportion the extinction at PINN to specific aerosol species (Figure VI-3). The species shown are Rayleigh, sulfate, nitrate, organics, elemental (light absorbing) carbon, and coarse mass. The sum of these species account for the majority of non-weather related extinctions. Extinction budgets are listed by season and by mean of cleanest 20%, mean of the median 20%, and mean of the dirtiest 20%. The "cleanest" and "dirtiest" signify lowest fine mass concentrations and highest fine mass concentrations respectively, with "median" representing the 20% of days with fine mass concentrations in the middle of the distribution. Each budget includes the corresponding extinction coefficient, standard visual range (km), and deciview (dv).

The segment at the bottom of each stacked bar in Figure VI-3 represents Rayleigh scattering, which is assumed to be a constant 10 Mm^{-1} at all sites during all seasons. Rayleigh scattering is the natural scattering of light by atmospheric gases. Higher fractions of extinction due to Rayleigh scattering indicate cleaner conditions.

b. Transmissometer Data - Optical Monitoring

The transmissometer system consists of two individually-housed primary components: a transmitter (light source) and a receiver (detector). The light extinction coefficient (b_{ext}) at any time can be calculated based on the intensity of light emitted from the source and the amount of light measured by the receiver (along with the path length between the two). Transmissometers provide continuous, hourly b_{ext} measurements. Meteorological or optical interference factors (such as clouds, rain, or a dirty optical surface) can affect transmissometer measurements.





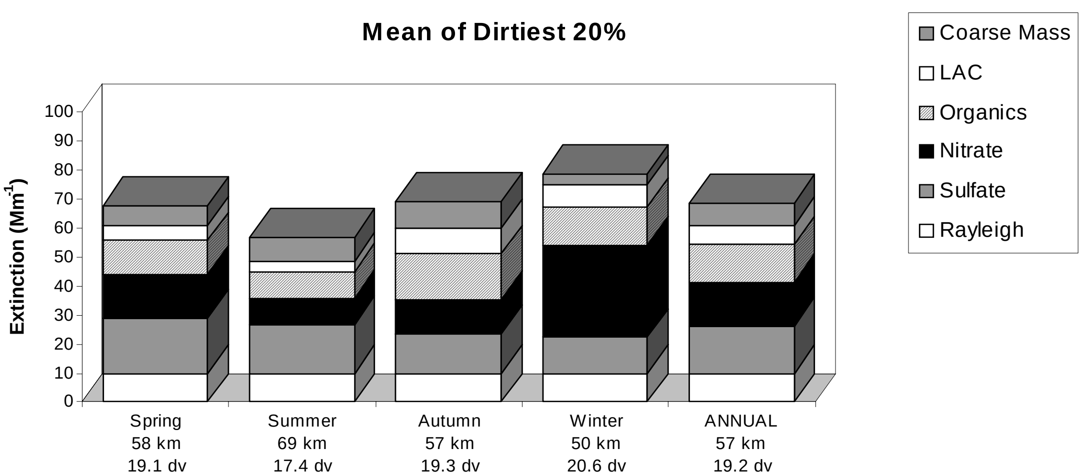


Figure VI-3. Reconstructed extinction budgets for PINN, March 1988-February 1999.

Collected data that may be affected by such interferences are flagged invalid, "filtered". Seasonal and annual data summaries are typically presented both with and without weather-influenced data. Visibility metric presentations include mean values of filtered b_{ext} data. The best, worst, and average conditions using the arithmetic means of the 20th percentile least impaired visibility, the 20th percentile most impaired visibility, and for all data for the season are presented. Detailed descriptions of the transmissometer system and data reduction and validation procedures used can be found in Standard Operating Procedures and Technical Instructions for Optec LPV-2 Transmissometer Systems (ARS, 1996).

Table VI-8 provides a tabular summary of the "filtered" seasonal and annual mean extinction values. Combined season data represent the mean of all valid seasonal b_{ext} means. Extinction values are also presented in units of standard visual range (in km) and dv. Tables VI-9 and VI-10 summarize the 20% clean and 20% dirty visibility metric statistics respectively. Data are represented according to the following conditions:

- No data are reported for seasons when the percentage of valid hourly averages (including weather) compared to total possible hourly averages, was less than 50%.
- Annual data represent the mean of all valid seasonal b_{ext} values for each March through February annual period. No data are reported for years that had one or more invalid seasons.
- Combined season data represent the mean of all valid seasonal b_{ext} values for each season (spring, summer, autumn, winter) of the March 1988 through August 1993 period.
- Combined annual period data represent the unweighted mean of all combined seasonal b_{ext} values.

Figure VI-4 provides a graphic representation of the "filtered" annual mean, 20% clean, and 20% dirty, visibility metric statistics. No data are reported for annual periods with one or more invalid seasons.

When comparing reconstructed (aerosol) extinction (Table VI-7) with measured (transmissometer) extinction (Table VI-8) the following differences/similarities should be considered:

- Data Collection - Reconstructed extinction measurements represent 24-hour samples collected twice per week. Transmissometer extinction estimates represent continuous measurements summarized as hourly means, 24 hours per day, seven days per week.
- Point versus Path Measurements - Reconstructed extinction represents an indirect measure of extinction at one point source. The transmissometer directly measures the irradiance of light (which calculated gives a direct measure of extinction) over a finite atmospheric path.

Reconstructed extinction is typically 70%-80% of the measured extinction. With a ratio over 80%, this relationship shows good agreement for PINN.

c. Camera Data - View Monitoring

View monitoring slides document visual conditions and are an effective tool for interpreting the visual effects of measured optical and aerosol parameters or presenting monitoring program goals, objectives, and results to decision-makers and the public. The Hawkins Peak vista photographs presented in Figure VI-5 were chosen to provide a feel for the range of visibility conditions possible and to help relate the extinction/SVR/haziness data to the visual sense.

Table VI-8. Seasonal and Annual Arithmetic Means, Pinnacles National Monument, California Transmissometer Data (Filtered), March 1988 through August 1993.

YEAR	Spring (Mar, Apr, May)			Summer (Jun, Jul, Aug)			Autumn (Sep, Oct, Nov)			Winter (Dec, Jan, Feb)			Annual (March - February)[a]		
	SVR (km)	b_{ext} (Mm^{-1})	dv	SVR (km)	b_{ext} (Mm^{-1})	dv	SVR (km)	b_{ext} (Mm^{-1})	dv	SVR (km)	b_{ext} (Mm^{-1})	dv	SVR (km)	b_{ext} (Mm^{-1})	dv
1988	85.9	49.6	15.6	83.7	50.9	15.9	100.6	44.9	14.3	124.6	38.7	12.3	98.7	46.0	14.5
1989	75.2	55.4	16.8	66.5	62.7	18.0	67.2	64.5	18.1	85.1	52.9	15.9	73.5	58.9	17.2
1990	100.7	45.1	14.3	87.2	48.6	15.4	84.3	51.0	15.8	82.3	53.9	16.2	88.6	49.7	15.4
1991	93.5	45.9	14.8	85.4	48.9	15.6	87.5	50.1	15.5	106.2	40.4	13.5	93.2	46.3	14.9
1992	81.8	53.5	16.2	74.2	57.8	17.1	77.8	55.4	16.6	89.8	46.1	15.0	80.9	53.2	16.2
1993	81.6	51.6	16.0	76.0	54.3	16.6	---	---	---	---	---	---	***	***	***
Mean[b]	86.5	50.2	15.6	78.8	53.9	16.4	83.5	53.2	16.1	97.6	46.4	14.6	86.6	50.9[C]	15.7

-- No data are reported for seasons with <50% valid data.

*** No annual data are reported for periods with one or more invalid seasons.

a Annual period data represent the mean of all valid seasonal b_{ext} means for each March through February annual period.

b Combined season data represent the mean of all valid seasonal b_{ext} means for each season of the March 1988 through August 1993 period.

c Combined annual period data represent the mean of all combined seasonal b_{ext} means.

Table VI-9. Seasonal and Annual 20% Clean Visibility Metric Statistics, Pinnacles National Monument, California Transmissometer Data (Filtered), March 1988 through August 1993.

YEAR	Spring (Mar, Apr, May)			Summer (Jun, Jul, Aug)			Autumn (Sep, Oct, Nov)			Winter (Dec, Jan, Feb)			Annual (March - February)[a]		
	SVR (km)	b_{ext} (Mm^{-1})	dv	SVR (km)	b_{ext} (Mm^{-1})	dv	SVR (km)	b_{ext} (Mm^{-1})	dv	SVR (km)	b_{ext} (Mm^{-1})	dv	SVR (km)	b_{ext} (Mm^{-1})	dv
1988	123.6	32.1	11.6	124.9	32.0	11.5	153.3	26.0	9.5	195.5	20.5	7.1	149.3	27.7	9.9
1989	98.6	39.8	13.8	91.0	43.6	14.7	98.7	40.3	13.9	131.5	31.0	11.1	105.0	38.7	13.4
1990	156.7	25.6	9.3	126.2	31.5	11.4	125.7	31.6	11.4	123.7	32.0	11.6	133.1	30.2	10.9
1991	140.2	28.5	10.4	119.7	33.2	11.9	133.2	30.5	11.0	147.2	27.0	9.9	135.1	29.8	10.8
1992	125.1	31.9	11.5	108.1	37.0	13.0	111.6	35.4	12.6	116.9	33.6	12.1	115.4	34.5	12.3
1993	112.9	35.0	12.5	102.3	38.5	13.4	---	---	---	---	---	---	***	***	***
Mean[b]	126.2	32.2	11.5	112.0	36.0	12.7	124.5	32.8	11.7	143.0	28.8	10.4	126.4	32.4[c]	11.6

-- No data are reported for seasons with <50% valid data.
*** No annual data are reported for periods with one or more invalid seasons.
a Annual period data represent the mean of all valid seasonal b_{ext} means for each March through February annual period.
b Combined season data represent the mean of all valid seasonal b_{ext} means for each season of the March 1988 through August 1993 period.
c Combined annual period data represent the mean of all combined seasonal b_{ext} means.

Table VI-10. Seasonal and Annual 20% Dirty Visibility Metric Statistics, Pinnacles National Monument, California Transmissometer Data (Filtered), March 1988 through August 1993.

YEAR	Spring (Mar, Apr, May)			Summer (Jun, Jul, Aug)			Autumn (Sep, Oct, Nov)			Winter (Dec, Jan, Feb)			Annual (March - February)[a]		
	SVR (km)	b_{ext} (Mm^{-1})	dv	SVR (km)	b_{ext} (Mm^{-1})	dv	SVR (km)	b_{ext} (Mm^{-1})	dv	SVR (km)	b_{ext} (Mm^{-1})	dv	SVR (km)	b_{ext} (Mm^{-1})	dv
1988	54.9	72.8	19.7	55.8	71.5	19.6	52.9	76.3	20.2	57.9	74.9	19.5	55.4	73.9	19.8
1989	49.7	80.6	20.8	44.9	88.4	21.7	40.3	99.2	22.8	48.0	88.2	21.3	45.7	89.1	21.7
1990	53.9	75.5	20.0	57.2	69.7	19.3	53.5	74.9	20.0	46.1	87.2	21.5	52.7	76.8	20.2
1991	58.8	67.5	19.0	58.2	68.1	19.1	53.0	77.2	20.2	65.0	63.2	18.2	58.8	69.0	19.1
1992	49.4	81.0	20.8	47.2	85.7	21.3	47.8	84.6	21.2	61.3	65.7	18.7	51.4	79.3	20.5
1993	53.6	74.4	20.0	53.7	74.2	19.9	---	---	---	---	---	---	***	***	***
Mean[b]	53.4	75.3	20.1	52.8	76.3	20.2	49.5	82.4	20.9	55.7	75.8	19.8	52.8	77.5[c]	20.2

-- No data are reported for seasons with <50% valid data.

*** No annual data are reported for periods with one or more invalid seasons.

a Annual period data represent the mean of all valid seasonal b_{ext} means for each March through February annual period.

b Combined season data represent the mean of all valid seasonal b_{ext} means for each season of the March 1988 through August 1993 period.

c Combined annual period data represent the mean of all combined seasonal b_{ext} means.

IMPROVE sites with eleven years of data. The distribution of $PM_{2.5}$ mass concentrations, reconstructed extinction expressed as dv, and associated constituents were examined for each site. The data were sorted into three groups based on the cumulative frequency of occurrence of $PM_{2.5}$: lowest fine mass days, 0-20%, median fine mass days, 40-60%, and highest fine mass days, 80-100%. After sorting each group's average concentrations of $PM_{2.5}$ and selecting the associated principal aerosol species, scattering and/or absorption of each species, reconstructed light extinction and deciview were calculated. Figure VI-6 shows plots of the 10, 50, and 90 percentile groups at PINN for both $PM_{2.5}$ and deciview.

Given the visibility data summarized for PINN, the data show a decreasing trend for $PM_{2.5}$ and deciview which indicates improving visibility over the 10 year period analyzed. Malm et al. (2000) also discovered a decreasing trend in the long term 5-year average deciview at PINN.

D. RESEARCH AND MONITORING NEEDS

1. Deposition

CASTNet monitoring should be continued.

2. Gases

It is recommended that passive ozone samplers be installed at three locations within PINN, one adjacent to the ozone analyzer and two at remote locations, one on the east side and one on the west side of the monument. These samplers will help to quantify spatial variation in ozone at PINN. One summer of data would provide sufficient calibration.

3. Aquatic Systems

There are no research or monitoring needs with regard to potential aquatic effects of air pollution at PINN.

4. Terrestrial Systems

It is recommended that long-term monitoring for the potential effects of ozone be initiated at PINN, especially given high SUM06 values. The best ozone bioindicator at the park is blue elder. Monitoring protocols from the Forest Health Monitoring manual (USDA Forest Service 1999) are recommended for establishing plots and collecting data (see Appendix). Plots should be evaluated in the late summer on an annual basis. Although there are other potential bioindicators for ozone (Douglas sagewort, blue wildrye [*Elymus glaucus*]), blue elder has symptoms that are more easily identified.

Smith (1987) recommended a three-phase monitoring program for lichens in PINN: (1) measuring the community structure of lichen flora on tree bark, (2) measuring the growth of *Physconia detersa* (which is a bioindicator for SO_2), and (3) chemical analysis of *Ramalina menziesii* and *Hypogymnia imshaugii*. Given the lack of current symptoms and the difficulty of identifying injury in lichens (see Appendix), we do not recommend lichen monitoring at this time. However, measuring community structure as suggested by Smith (point #1) would be a relatively low-cost effort if conducted every three years or so, such that changes in the diversity of lichen flora over time could be quantified.

5. Visibility

Aside from continuing IMPROVE aerosol monitoring, no additional visibility monitoring or research are recommended at this time.

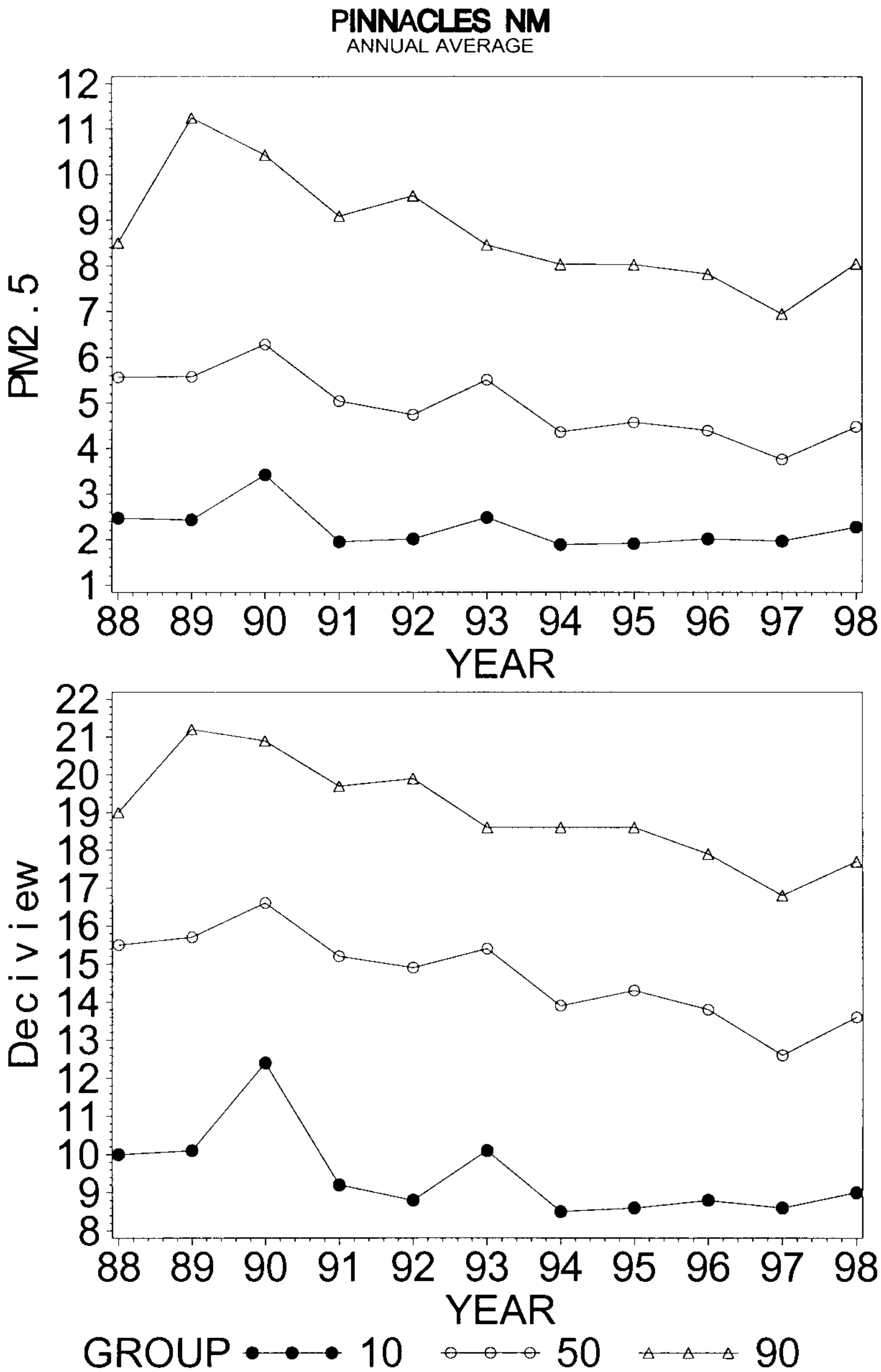


Figure VI-6. Trends in annual averages for $PM_{2.5}$ ($\mu g/m^3$) and visibility (deciview) for PINN.

VII. POINT REYES NATIONAL SEASHORE

A. GENERAL DESCRIPTION

Point Reyes National Seashore (PORE), located 48 km north of San Francisco, is comprised of slightly more than 28,735 ha of land, including 72 km of seashore. PORE was established in 1962 in order to "save and preserve, for the purpose of public recreation, benefit and inspiration, a portion of the diminishing seashore of the United States that remains undeveloped."

Physiographically, PORE is a long, narrow, steep-ridged peninsula with a broad flat arm that projects 32 km into the Pacific Ocean and ends in the white cliffs of the Point Reyes headlands. The San Andreas Fault zone separates the peninsula from the general California coastal area. The Point Reyes Peninsula is a picturesque and highly diverse natural region with great scenic, historic, scientific, and recreational value. It includes high cliffs battered by the ocean; secluded coves populated by marine mammals and seabirds; long, windswept beaches exposed to the high winds and waves of the Pacific Ocean; quiet sheltered beaches and estuaries; rolling grassy hills; and a high forested ridge. Each of the landscapes of the peninsula is a product of the geological composition and geological history of that area. The main features of the peninsula are directly related to the underlying geology. The Point Reyes Peninsula is separated from the Coast Range of the mainland by a long, straight depression called the Olema Trough that extends from Tomales Bay to Bolinas Lagoon along the San Andreas Fault.

The southern portion of the seashore is rugged and densely vegetated. The Douglas fir forest that covers Inverness Ridge gives way on the lower slopes to chaparral and grasslands, which contain a collection of small lakes in close proximity to the ocean. Miles of beaches and pocket coves are bordered by steep cliffs and broken by rocky headlands. Drake's Beach and the long sand spit that protects Limantour Estero are relatively protected and have gentle surf conditions. In contrast, the 16-km long Point Reyes beach faces directly into the prevailing winds and the full force of the Pacific Ocean.

In 1988, the UNESCO Man and the Biosphere Program approved the designation of the Central California Coast Biosphere Reserve, under the International Biosphere Reserve Program. It includes the entire seashore and other public lands in the area.

In 1976, Congress established the 10,265-ha Point Reyes Wilderness (PL 94-544 and PL 94-567) and designated 3,240 ha as potential wilderness. Located near the San Francisco metropolitan area, this wilderness area is one of the most accessible within the United States wilderness system. In 1985, Congress changed the name of the wilderness area to the Phillip Burton Wilderness (PL 99-68; Point Reyes National Seashore 1999).

Although PORE contains superb natural resources and a substantial wilderness area, much of the seashore allows commercial uses, which include dairy operations, cattle grazing, and commercial oyster cultivation.

Land use within the seashore is managed in four distinct zones. These include the natural zone, special use zone, historic zone, and development zone. The natural zone consists mainly of designated wilderness or potential wilderness area (13,235 ha). It also includes two marine reserves (Point Reyes Headlands Reserve and Estero de Limantour Reserve). The special use zone includes pre-existing pastoral uses such as dairy and beef ranching (about 6,880 ha), a 2-ha oyster farm, and a small radio range station. The historic zone includes a number of historic buildings or groups of buildings such as the Point Reyes Lighthouse complex, the Olema lime kiln, the Point Reyes lifeboat station area, and a number of ranch complexes and archaeological sites. The development zone includes park visitor and administrative facilities.

The enabling legislation and subsequent laws, the Point Reyes National Seashore Act (PL87-657) and the National Parks and Recreation Act of 1978 (PL 95-625) provided that owners of lands acquired for park establishment may "retain for himself and his/her heirs and

assigns, a right of use and occupancy for a definite term of not more than twenty-five years...or a term ending in death... whichever is later." Approximately 8,335 ha of PORE have been retained in agricultural production within a "pastoral" zone." The Northern District of Golden Gate National Recreational Area (GOGA), which is administered by PORE, contains an additional 4,250 ha that are in beef cattle grazing. State and federal enclaves and commercial activities within the seashore were excluded from purchase. These inholdings comprise 2,000 ha (Point Reyes National Seashore 1998).

PORE supports a unique and varied landscape that has been subject to a broad range of human and natural events. Saved from development by its inclusion within the NPS system, Point Reyes is unique not only in its assemblage of natural and cultural features, but also in its proximity to a major urban population. This juxtaposition makes the seashore's resources and recreational opportunities readily accessible to a large number of people, and enhances the importance of the special qualities for which it was set aside. This diversity, complexity, rarity, and proximity to the Bay Area make balancing the NPService mandates of use and preservation difficult (Point Reyes National Seashore 1999).

The enabling legislation mandate to continue agricultural activities within the seashore has had significant effects on both terrestrial and aquatic resources in PORE, although a range management and monitoring program was implemented in 1987 and serves to control grazing activities and their associated impacts. Historic land use practices at PORE have significantly altered the landscape and its stability and also the vegetative communities and wildlife dependent upon them. Clearing, logging, cultivation, cropping, road building, commercial development, and agricultural grazing have markedly altered the environment. A large number of exotic plant and animal species have invaded PORE, which in some cases threaten to alter the character of the habitat and wildlife values associated with them. Some streams no longer support anadromous fisheries because of dam building, siltation, and loss of riparian habitat; other streams, where fish survive, have been adversely affected by siltation and loss of riparian habitat.

Exotic animals, such as the red fox (*Vulpes vulpes*) and house cat (*Felis domesticus*), have the potential to significantly disrupt native plant and animal species. Municipal water development, additional residential and community development, and adjacent agricultural activities have serious implications for water quantity and quality in affected streams and estuaries.

1. Geology and Soils

The Point Reyes Peninsula is roughly triangular in shape and projects offshore from the valley of the San Andreas Fault to an apex at Point Reyes itself. The peninsula has displaced towards the northwest along the fault line at a rate of about 5 cm per year, and that has apparently continued for about the last 80 million years (Rowe 1974). Depressions were created along the fault line, forming Tomales and Bolinas Bays.

The backbone of Inverness Ridge is granitic (Figure VII-1). The oldest rocks in the peninsula are schist, quartzite, and crystalline limestone. They are found in small isolated patches surrounded by the granite of Inverness Ridge. These rocks were originally shale, sandstone, and limestone, respectively, that were engulfed by the molten granite about 80 million years ago and metamorphosed. From Mt. Whittenberg to the south, the granite of Inverness Ridge is covered by an increasing thickness of younger, lighter colored marine shales. Similar shales and sandstones cover the lower western slopes of the ridge. On the west side of Inverness Ridge and extending south to the promontory of Point Reyes, the open pastureland is underlain by light colored siltstones and claystones.

pine and Douglas-fir forests. In some places it also occurs in a complex pattern with the coastal prairie grasslands. The coastal scrub community is dominated by coyote bush on north-facing slopes and a mix of species on south-facing slopes, including California coffeeberry (*Rhamnus californica*), coast sagebrush (*Artemisia californica*), and poison oak (*Toxicodendron diversilobium*). The Inverness bishop pine forest covers about 2,024 ha along Inverness Ridge. The distribution of this plant community is generally coincident with the occurrence of granitic rock materials which are found between Mt. Whittenberg and Tomales Point. Along Inverness Ridge and Mount Whittenberg south to the seashore boundary, is a Douglas-fir forest. It is found in association with soils derived from marine shales that overlay the granitic bedrock of the ridge. The forest includes pure stands of Douglas-fir and also stands that include Douglas fir and a secondary canopy of mixed hardwoods such as tanoak (*Lithocarpus densiflora*), madrone, and California bay. Understory shrub species include California coffeeberry, blue blossom (*Ceanothus thryseflorus*) and monkey flower (*Mimulus* spp.).

The coastal strand, which is desiccated by wind and salt spray, is sparsely vegetated, mostly with low, prostrate, and succulent plants. European beach grass (*Ammophila arenaria*), a widespread exotic introduced in the late 19th century, is the most common species in this area, and has replaced much of the American dune grass (*Elymus mollis*). Dune stabilization by European beach grass disrupts the natural dynamics of the coastal strand community. A wide range of native species, such as beach morning glory (*Convolvulus soldanella*) and sand verbena (*Abronia* spp.), receive strong competition from the exotic ice plant (*Carpobratus edulis*). Many rare native dune plant species occur in the coastal strand and on coastal bluffs.

Coastal prairie is a narrow strip of habitat just inland from the coastal strand. Most of the native perennial grasses have been displaced by exotics as a result of livestock grazing, plowing, burning, and planting. Coastal rangeland encompasses rolling pasturelands that cover much of the western and northern sections of the Point. Most of this area is subject to livestock grazing, interspersed with patches of various plant communities. Coastal scrub, or soft chaparral, occurs between the coastal and forest communities. Coyote bush (*Baccharis pilularis*) and bush lupine (*Lupinus arboreus*) are dominant species, occurring in various densities with sword fern (*Polystichum munitum*), salal (*Gaultheria shallon*), pink flowering currant (*Ribes sanguineum glutinosum*), and other species.

Salt, brackish, and fresh water marsh all occur in PORE, and whereas the boundaries between these marshlands were formerly gradual, they are now mostly defined by roads, levees, culverts, and fill. Salt marsh is located at the edge of estuaries, and has been impacted considerably by accelerated sedimentation from logging and other watershed disturbances and natural erosion. Freshwater marsh is limited in distribution, although Olema Marsh is an excellent example of this community with high species diversity. Intertidal plant communities are dominated by various green, brown, and red marine algae, and eelgrass. Sea lettuce (*Ulva* spp.) is a particularly prominent green algal species in shallow estuaries, while brown algae are found in extensive kelp beds farther offshore.

Within the coastal prairie vegetation community of PORE, there are about 25 dairy and beef cattle ranches, 1,415 ha of wilderness, and populations of 22 of the rare plants known from Point Reyes. There are only a few remaining remnants of the native coastal prairie plant community. Most have been altered significantly by 150 years of ranching and cultivation and the introduction of more than 180 species of exotic plants.

Because of the diversity of habitat and plant species in PORE, wildlife species are also highly diverse. The seashore contains 47 listed animal species, 14 of which have federal status as endangered, 8 as threatened, and 24 species of concern (Point Reyes National Seashore, 1999). Among these are the endangered or formerly endangered brown pelican (*Pelecanus*

occidentalis), American peregrine falcon (*Falco peregrinus anatum*), and Myrtle's silverspot butterfly (*Speyeria zerene myrtleae*), and the federally threatened northern spotted owl (*Strix occidentalis*), western snowy plover (*Charadrius montanus*), California red-legged frog (*Rana aurora*), coho salmon (*Oncorhynchus kisutch*), and steelhead trout (*O. mykiss*; Point Reyes National Seashore 1994, 1999). Native land mammal species number about 37 species, including mountain lion (*Felis concolor*) and an endemic race of mountain beaver (*Aplodontia rufa*). The mountain beavers at PORE constitute a disjunct population which represents the southern-most extension of the species along the coast. In addition, there are about two dozen species of marine mammal, three of which breed at PORE. Large ungulates within the seashore include tule elk (*Cervus elaphus nannodes*), black-tailed deer (*Odocoileus hemionus columbianus*) and the introduced Axis deer (*Cervus axis*) and fallow deer (*Cervus fallow*).

Because the Point Reyes Peninsula is located along the Pacific flyway and juts prominently from the coast, the peninsula has an extremely high number of resident and migratory bird species. Over 480 bird species have been recorded in PORE, representing about 50% of the bird species in North America. The Point Reyes Peninsula is one of the best bird watching and ornithological research areas in the United States. PORE contains the Point Reyes Bird Observatory, the first bird observatory and one of the pre-eminent avian research institutions in the country.

PORE contains a particularly high density of both northern spotted owl and red-legged frogs. There are highly significant seabird colonies, including rare species such as Ashy storm-petrel (*Oceanodroma homochroa*), and 10 other species.

The ocean waters adjacent to PORE are rich in nutrients and support an abundant fishery and associated marine fauna. PORE contains a number of significant seabird rookeries and haul-out and pupping areas for pinnipeds. Marine mammals that inhabit or migrate past the Point Reyes Peninsula include the southern sea otter (*Enhydra lutris nereis*), stellar sea lion (*Eumetopais jubatus*), California gray whale (*Eschrichtius robustus*), humpback whale (*Megaptera novaeangliae*), and blue whale (*Balaenoptera musculus*). PORE is one of the premier locations along the Pacific coast for whale watching and observation of other marine mammals. Northern elephant seals (*Mirounga angustirostris*) breed on the Point Reyes headlands. Point Reyes accounts for about 20% of the entire California population of harbor seals (*Phoca vitulina*).

PORE includes a number of significant estuarine resources. Tomales Bay, Drake's Estero, Estero de Limantour, and Abbott's Lagoon are considered some of the best examples of ecologically intact west coast estuaries in the United States. Tomales Bay parallels much of the eastern portion of the seashore and has been the focal point for community and area interest in issues of water quality and quantity, anadromous fisheries, agricultural land practices, and land development issues (Point Reyes National Seashore 1994). Drake's Estero has been characterized as possibly the most pristine estuary on the Pacific coast.

B. EMISSIONS

PORE is located in western Marin County, within the San Francisco Bay Area Air Basin (SFBAAB), a predominantly urban air basin with substantial emissions of air pollutants. However, its coastal location near the northwestern edge of the air basin, combined with prevailing northwesterly winds off the Pacific Ocean, place PORE in a generally upwind position relative to the urbanized portions of the SFBAAB. Approximately 20% of the state's population lives in the nine counties of the SFBAAB, and emission sources within the SFBAAB account for about 20% of total statewide emissions (Alexis et al., 1999). Since 1980, both population in the SFBAAB and the number of vehicle miles traveled have increased more slowly

than in other urban areas of California, growing by 27 and 43% respectively (Alexis et al., 1999). Both emission levels and ambient pollutant concentrations have shown clear, downward trends in the SFBAAB (Alexis et al., 1999).

Emissions from counties within 140 km of PORE are shown in Table VII-1. The principal species of concern are ozone precursors (NO_x and ROG) and PM. SO_2 emissions are not high. No major point sources are located in Marin County. Within the SFBAAB, the closest sources that emit at least 100 tons/year of ROG, NO_x, PM_{10}, or SO_2 are located in Solano (Benecia), Contra Costa (Rodeo, Hercules, Richmond, Martinez, Pittsburg, and Antioch), and San Francisco counties (Figures II-3 through II-6). However, prevailing winds tend to blow emissions from these sources eastward through the Carquinez Strait or southward along the length of San Francisco Bay. As of 1996, stationary sources accounted for 22 % of ROG emissions, 19 % of NO_x emissions, and 11 % of PM_{10} emissions in the SFBAAB (CARB 1998b). Mobile sources dominated NO_x (76 %) and ROG emissions (59 %), while area sources (road dust, construction, and residential fuel combustion) dominated PM emissions (81 %).

Table VII-1. 1995 Emissions from counties within 140 km of PORE. Source: CARB Almanac 1999b; SO_x from CARB Emissions Website 1999a. Units are 1000 tons/year.

County	NO_x	ROG*	PM_{10}	CO	SO_x
Alameda	43.1	43.1	13.1	283.2	3.3
Contra Costa	50.4	36.1	12.0	219.0	14.6
Marin	7.3	8.8	3.7	64.6	0.4
Napa	3.3	4.4	1.5	27.4	0.0
San Francisco	15.7	18.3	6.2	110.2	2.9
San Mateo	22.6	23.0	8.4	165.3	0.7
Santa Clara	46.0	48.5	17.9	333.2	1.5
Solano [1] (SF Bay Area)	12.4	10.6	3.7	66.4	6.6
Sonoma [1] (SF Bay Area)	10.2	11.3	4.4	84.3	0.4

* Reactive Organic Gases
[1] Portion of the county in the air basin

Air quality within the national seashore is generally good due to the prevailing westerly marine air flows. However, during periods when atmospheric conditions displace the east Pacific high pressure system, air flows from the San Francisco Bay area and can degrade the air quality of the seashore. This mainly occurs during the late summer and early fall, when the major atmospheric systems undergo a seasonal change. During this time, the seashore is often impacted by a general haze, which significantly impairs visibility.

An inventory of in-park emissions has recently been compiled by the NPS-Air Resources Division. The results are presented in Table VII-2.

Table VII-2 . Summary of 1998 stationary and area, and mobile source emissions (tons/yr) at PORE.

Activity	Particulates	Sulfur Dioxide	Nitrogen Oxides	Carbon Monoxide	VOCs	HAPs[1]
Stationary and Area Source Emissions						
Stationary Combustion Sources						
Heating units	0.00	0.01	0.07	0.01	0.00	0.00
Generators	0.00	0.00	0.03	0.01	0.00	0.00
Woodstoves	0.44	0.00	0.04	3.31	0.76	0.00
Combustion Emission Subtotal	0.44	0.01	0.14	3.32	0.76	0.00
Fuel Storage Tanks						
Gasoline/Diesel Fuel Tanks	0.00	0.00	0.00	0.00	0.00	0.00
Area Sources						
Prescribed Burning	10.45	1.44	0.58	78.50	0.00	0.00
Area Source Emission Subtotal	10.45	1.44	0.58	78.50	0.00	0.00
TOTALS	10.89	1.45	0.71	81.82	0.76	0.00
Mobile Source Emissions						
Road Vehicles						
Visitor Vehicles	24.69	0.00	7.32	76.00	5.70	–
Visitor Shuttle Buses	0.02	0.00	0.08	0.24	0.02	–
NPS/GSA Road Vehicles	0.96	0.00	0.70	3.09	0.22	–
Vehicle Emission Subtotal	25.66	0.00	8.09	79.32	5.95	–
Nonroad Vehicles						
NPS Nonroad Vehicles	0.21	0.00	1.25	7.60	3.78	–
TOTALS	25.87	0.00	9.34	86.92	9.73	–

[1] Hazardous air pollutants, based on the list compiled by EPA

C. MONITORING AND RESEARCH ACTIVITIES

The seashore is in the process of conducting a comprehensive baseline resource inventory. Many ecosystem components have been inventoried and a few are monitored but key components are lacking and the programs are not yet integrated. There is therefore need to complete baseline inventories of basic resources and provide background information upon which to build a monitoring program (Point Reyes National Seashore 1999).

1. Air Quality

Air quality monitoring within Point Reyes has included particulate matter ($PM_{2.5}$ and PM_{10}), hourly ozone, and SO_2 (Table VII-3). Passive ozone samplers have not been located in the

Table VII-3. Air quality monitoring at PORE.		
Species	Site within park	Site within 50 km
Ozone, hourly	NPS**	
Ozone, passive	NPS*	
SO_2	NPS	
PM_{10}	IMPROVE	
$PM_{2.5}$	IMPROVE	
Wet deposition		ARB**
Dry deposition		
Visibility		

* New site
** Closed before 1994

seashore. From mid-1984 through mid-1988, a CADMP wet deposition site was located at San Rafael, approximately 25 km east of the seashore boundary. No CASTNet site is located within the seashore for monitoring dry deposition. The nearest CADMP dry deposition monitor was located in Fremont; data from that urban location are unlikely to be representative of deposition rates within the seashore.

a. Wet Deposition

Blanchard et al. (1996) estimated annual and 10-year wet deposition rates throughout California by interpolating the observations from all NADP/NTN, CADMP, and special-studies monitoring locations for the period 1985 through 1994; they also estimated interpolation uncertainties. Upper-bound deposition values for unmonitored areas may be estimated as the sum of the interpolated values plus twice the interpolation uncertainty (i.e., mean plus two standard deviations). The results indicated that the 10-year mean wet total N deposition rates were less than 3 (+/- 3, at 2 sigma) kg/ha/yr (as N) throughout the state. Wet S deposition was less than 1.3 (+/- 1, at 2 sigma) kg/ha/yr (as S) throughout the state.

At the nearest CADMP site (San Rafael), wet S deposition ranged from 1.1 to 2.2 kg/ha/yr as S (equivalently, 3.2 to 6.7 kg/ha/yr as SO_4^{2-}) during the period 1984 through 1988 (Blanchard et al. 1996). Because San Rafael is located near the ocean, some SO_4^{2-} derives from marine aerosol; the amounts of wet SO_4^{2-} deposition not of marine origin ranged from 0.7 to 1.7 kg/ha/yr as S (Blanchard et al. 1996). Located ~25 km east of Point Reyes, San Rafael is within the urbanized portion of the SFBAAB. At the more rural NADP/NTN site of Hopland, ~80 km north of Point Reyes, nonmarine wet SO_4^{2-} deposition ranged from 0.2 to 1.0 kg/ha/yr as S between 1984 and 1995 (Blanchard et al. 1996). The annual NO_3^- and NH_4^+ deposition rates at San Rafael were in the ranges of 0.5 to 1.2 kg/ha/yr and 0.7 to 1.8 kg/ha/yr as N, respectively, yielding a multi-year mean total N wet deposition rate of 1.9 kg/ha/yr (Blanchard et al., 1996). During the period 1984 through 1988, the annual-average H^+ concentration in precipitation at San Rafael was 8.7 µeq/L (pH 5.06) (Blanchard and Tonnessen, 1993).

b. Occult/Dry Deposition

The CADMP co-located wet and dry deposition samplers at ten locations in California (Blanchard et al., 1996). At the three nonurban locations (YOSE, SEQU, and Gasquet, located near REDW), mean dry deposition rates of oxidized N species were roughly comparable to the

rates of wet NO_3^- deposition. The nonurban dry S deposition rates were in the range of 0.7 to 1.3 times the rate of wet SO_4^{2-} deposition, while dry NH_3 plus aerosol NH_4^+ deposition rates were approximately 0.4 to 1.7 times the rate of wet NH_4^+ deposition (Blanchard et al., 1996). Lacking more specific information, it may be concluded that dry deposition rates in PORE are likely of the same order of magnitude as wet deposition rates.

c. Gaseous Monitoring

Ozone concentrations and exposures are shown in Table VII-4. Concentrations and exposure were low compared to other parks in the state. Low ozone concentrations are typical of coastal locations in the San Francisco Bay Area.

Table VII-4. Summary of ozone concentrations and exposure from PORE monitoring sites (Source: Joseph and Flores, 1993; National Park Service, Air Resources Division 2000).

Year	Maximum Daily 1-hour Value (ppbv)	2nd Highest Daily 1-hour Value (ppbv)	Number of Daily Maximum 1-hour Values Greater Than or Equal to 125 ppb	3-Year Average Number of Exceedences	Maximum 9am-4pm Average (ppbv)	Sum06 (ppbv-hour)[a]	Number of Valid Hours of Ozone Measurements
1989	80	73	0	na	63	2,000	7870
1990	78	67	0	na	55	1,000	8072
1991	72	63	0	0	65	1,000	7983
1992	66	66	0	0	52	0	6924

[a] Maximum 8 am - 8 pm 90-day rolling average

Annual maxima and means of the 24-hour resolution samples of SO_2 are shown in Table VII-5 for the period 1988-1993. SO_2 measurements were discontinued after 1996 due to concerns about their accuracy. The measurements are considered sufficiently accurate to show that the measured SO_2 concentrations were well below the levels at which plant injury has been documented, ~40 to 50 ppb 24-hour average and 8-12 ppb annual average (Peterson et al. 1992).

Table VII-5. Maximum and mean SO_2, from 24-hour-resolution samples at PORE. Samples are collected every 3-4 days, unless noted. (Source: NPS Air Resources Division). Units are ppb.

SO_2	1988	1989	1990	1991	1992	1993
Maximum	0.48*	1.04**	1.25	0.37**	0.78**	0.43*
Mean	0.06*	0.12**	0.15	0.05**	0.07**	0.03*

na Not available
* Less than 50 samples collected for the year
** 50-75 samples collected for the year

2. Aquatic Resources

Hydrography on the Point Reyes Peninsula is governed by the Inverness Ridge. Small streams flow down the east slope of this ridge into Olema Valley with a flow either northward into Tomales Bay or southeast into Bolinas Lagoon. On the west side of the ridge there are larger streams such as the Arroya Honda and Coast Creek, which flow directly into the ocean (Figure VII-3). There are also a number of drainage anomalies associated with faults, particularly the San Andreas Fault. Mean annual runoff is estimated to range between about 15 and 40 cm across the peninsula (Dale and Rantz 1966).

Freshwater resources within PORE include 36 streams, several natural lakes, and numerous springs. There are also substantial wetland areas and riparian areas. There are many man-made impoundments, most of which have been used for stock watering.

No thorough inventory has been conducted of park water resources, and monitoring data are generally not available. The STORET database only contains water quality data for three sampling locations within PORE. All had very high conductivity (> 200 µS/cm) and base cation concentrations (Ca^{2+} > 400 µeq/L). None would be sensitive to acidification from acidic deposition.

Pastoral land use at PORE, including dairy and beef cattle operations, and also oyster cultivation, have the potential to adversely impact water quality. Overgrazing may contribute to erosional problems, increasing sedimentation of streams, lakes, and estuaries. Many land uses can also contribute to N enrichment, eutrophication, and bacterial contamination of surface water resources within PORE. Six dairies operate in the seashore under leases, special use permits, or through reservation of possession. There are also 14 beef cattle ranches.

The seashore contains a number of streams which historically had significant spawning runs for coho salmon (*Oncorhynchus kisutch*) and steelhead trout (*Oncorhynchus mykiss*). Virtually all of the smaller streams within the seashore have been altered by impoundments and diversions, some of which have eliminated salmon and steelhead migration and spawning throughout much of the park. The streams have also been impacted by years of grazing and past timber harvest activities. Olema Creek, within the seashore, is tributary to Lagunitas Creek, which historically was one of the most productive anadromous spawning streams on the central California coast. It is also the major source of freshwater to Tomales Bay. It is considered a prime opportunity to enhance salmon and steelhead spawning. Pine Gulch Creek is similar in size to Olema Creek and also represents an important opportunity for salmonid restoration within the seashore.

A draft Water Resources Management Plan was prepared in 1995 to identify water resource-related issues and management concerns, to provide a summary of existing hydrological information, and to provide park management with a recommended action plan for addressing water-related issues within the seashore (National Park Service and Point Reyes National Seashore 1995). Management actions recommended in this action plan included inventory and assessment activities; implementation of a limited water quality monitoring program; and implementation of a number of resource management activities developed to address resource protection, watershed management, fisheries, and related needs.

Both the quantity and quality of freshwaters that flow into Tomales Bay are significant concerns. Impacts of nonpoint source pollutants of urban and agricultural source areas also remain a significant concern for Tomales Bay. Streams within the watershed once supported substantial runs of anadromous fish, including steelhead trout and coho salmon. These fisheries resources and their associated aquatic habitat have been severely depleted throughout California including the Tomales Bay watershed (Smith 1986). This watershed also provides habitat for an endangered species of freshwater shrimp (*Syncaris pacifica*), and it includes a number of

Figure VII-3. Hydrography of PORE. Also shown are the locations of air quality and water quality sampling sites monitored by the NPS.

reservoirs which provide a significant proportion of the water supply for the Marin Municipal Water District. Elevated nutrient levels, turbidity, and coliform bacteria concentrations have been observed, presumably associated with agricultural activities in the watershed (National Park Service and Point Reyes National Seashore 1995).

The central and southern areas of the seashore drain into Drakes Bay. The most prominent feature of this area is the Drakes Estero, which consists of a large central estuary area and four finger embayments. A total of six streams flow into Drakes Estero, four of which are perennial. Land use consists mainly of ranching. Several man-made freshwater ponds have been created as part of the ranching operations. The Estero de Limantour (331 ha) is designated as a State Marine Reserve. Drainage into this estero includes several small impoundments associated with ranching activities that are no longer active.

The Tomales Bay watershed comprises almost half of the area of Marin County. About 80% of the watershed is in agricultural landuse and nearly 30,000 head of cattle and 20,000 sheep graze the hillsides of the Tomales Bay watershed outside the park (Point Reyes National Seashore 1995).

Little information is available on the fresh water resources of PORE. However, an unpublished limnological survey of Bass Lake was conducted in 1976 by Carl Widmer, University of the Pacific, Stockton, California. Bass Lake is a clear, slightly saline, slightly alkaline, warm monomictic lake that exhibits eutrophic characteristics. This 3 ha lake has a pH near 8.0 and would not be at all sensitive to the impacts of acidic deposition. The summer hypolimnion of both Bass Lake and Wildcat Lake produces high concentrations of hydrogen sulfide, which mixes throughout the lakes during fall turnover, causing extensive fish kills (Johnston 1985a, Widmer 1976).

The Bolinas Bay and Lagoon drainage is located at the southeastern end of the seashore. Logging, farming, development, and over-grazing in this watershed during the 19th and early 20th centuries created devastating patterns of erosion and sedimentation within the watershed (National Park Service and Point Reyes National Seashore 1995).

Water quality is an important concern at PORE and there are a number of water quality problems associated with past and current land use within and adjacent to the seashore. The most significant issues relate to erosional inputs, eutrophication, damage to riparian communities, and amphibian and anadramous fish habitat. Acidification from acidic deposition is not, however, a significant threat to the aquatic ecosystems of PORE. Freshwater ecosystems are not expected to be highly sensitive to acidification. In addition, because the prevailing winds are largely from the direction of the Pacific Ocean, anthropogenic S and N deposition are low and are expected to stay low in the future.

3. Vegetation

Baseline vegetation transects for long-term monitoring were established in 1989 within habitat types identified by the Fire Management Plan. They will be monitored to detect vegetative changes attributed to fire effects and successional processes. Baseline condition and trend transects have been established and monitored on agricultural rangelands since 1987. A number of these transects are in areas which have been withdrawn from agricultural production (Point Reyes National Seashore 1994). A recent vegetation map led to the establishment of over 1200 plots, some of which will be monitored over time.

There has been virtually no research or monitoring on the effects of air pollution on vegetation at PORE. The sensitivities of plant species at PORE to air pollutants are summarized in Table VII-6.

Table VII-6. Plant and lichen species of PORE with known sensitivities to sulfur dioxide, ozone, and nitrogen oxides (H=high, M=medium, L=low, blank=unknown).

Scientific Name	Common Name	Sensitivity		
		SO_2	O_3	NO_x
Gymnosperms				
Pseudotsuga menziesii	Douglas fir	H	M	H
Taxus brevifolia	Pacific yew	L		
Angiosperms				
Acer macrophyllum	Bigleaf maple		L	
Achillea millefolium	Common yarrow		L	
Arctostaphylos uva-ursi	Kinnikinnick	L	L	
Artemisia douglasiana	Douglas' sagewort		H	
Bromus carinatus	California brome		L	
Bromus tectorum	Cheatgrass		M	
Cichorium intybus	Chicory		L	
Clematis ligusticifolia	Western white clematis	M		
Conium maculatum	Poison hemlock		L	
Convolvulus arvensis	Field bindweed	H		
Corylus cornuta var. californica	California hazelnut	H		
Elymus glaucus	Blue wildrye		H	
Erodium cicutarium	Redstem stork's bill	M	M	
Festuca octoflora	Sixweeks fescue	L	L	
Gaultheria shallon	Salal		M	
Holodiscus discolor	Oceanspray	H	M	
Lemna minor	Common duckweed	L		
Lepidium lasiocarpum	Shaggyfruit pepperweed	L	L	
Lolium perenne	Perennial ryegrass		M	
Medicago sativa	Alfalfa		M	
Mimulus guttatus	Seep monkeyflower		L	
Osmorhiza chilensis	Sweetcicely		M	
Physocarpus capitatus	Pacific ninebark		M	
Pinus radiata	Monterey pine		L	
Platystemon californicus	California creamcups	M	M	
Poa annua	Annual bluegrass	H	L	
Poa pratensis	Kentucky bluegrass		L	
Potentilla glandulosa	Gland cinquefoil		M	
Rubus parviflorus	Thimbleberry		M	
Rubus spectabilis	Salmonberry	M	M	
Taraxacum officinale	Common dandelion		L	
Thysanocarpus curvipes	Sand fringepod	M	M	
Trifolium pratense	Red clover	L		
Trifolium repens	White clover		H	
Viola adunca	Hookedspur violet		L	

4. Visibility

As part of the Interagency Monitoring of Protected Visual Environments (IMPROVE) network, visual air quality in PORE has been monitored using an aerosol sampler and camera. The aerosol sampler began operation in March of 1988. It is located at the North District Ranger Station, south of Tomales Bay State Park and north of Point Reyes Hill. The automatic 35mm camera was located on a peninsula at the south-west corner of Drakes Bay and operated from June 1987 through April 1995. The camera viewed east across Drakes Bay towards the Point Reyes Wilderness area. Data from this IMPROVE site have been summarized to characterize the full range of visibility conditions for the March 1988 through February 1999 period, based on seasonal periods (Spring: March, April, and May; Summer: June, July, and August; Autumn: September, October, and November; and Winter: December, January, and February) and annual periods (March through February of the following year, e.g., the annual period of 1998 includes March 1998 through February 1999). Complete descriptions of visibility characterization, mechanisms of sources and visibility impacts, and IMPROVE monitoring techniques and rationale are provided in Chapter I.

a. Aerosol Sampler Data - Particle Monitoring

A tabular and graphic summary of average reconstructed extinction values by season and year for the March 1988 through February 1999 period are provided in Table VII-7 and Figure VII-4, respectively.

Reconstructed extinction budgets generated from aerosol sampler data apportion the extinction at PORE to specific aerosol species (Figure VII-5). The species shown are Rayleigh, sulfate, nitrate, organics, elemental (light absorbing) carbon, and coarse mass. The sum of these species account for the majority of non-weather related extinctions. Extinction budgets are listed

Table VII-7. Seasonal and annual average reconstructed extinction (b_{ext}; Mm^{-1}) at PORE, March 1988 through February 1999.

Year	Spring (Mar, Apr, May)	Summer (Jun, Jul, Aug)	Autumn (Sep, Oct, Nov)	Winter (Dec, Jan, Feb)	Annual (Feb-Mar)[a]
1988	52.5	59.0	42.1	59.6	54.5
1989	48.5	71.5	72.1	93.7	74.2
1990	55.6	66.7	59.2	76.2	65.7
1991	48.4	59.6	59.2	79.6	62.2
1992	59.0	62.8	53.2	54.7	58.3
1993	46.5	63.8	57.0	91.3	67.7
1994	42.4	57.8	50.6	76.3	57.1
1995	43.7	69.1	44.2	43.2	49.2
1996	53.1	68.1	45.1	43.2	52.6
1997	47.7	67.7	45.8	49.3	52.0
1998	40.0	68.3	48.0	56.9	54.0
Mean[b]	48.8	64.9	52.4	65.8	58.9[c]

[a] Annual period data represent the mean of all data for each March through February annual period.
[b] Combined season data represent the mean of all seasonal means for each season of the March 1988 through February 1999 period.
[c] Combined annual period data represent the mean of all combined season means.

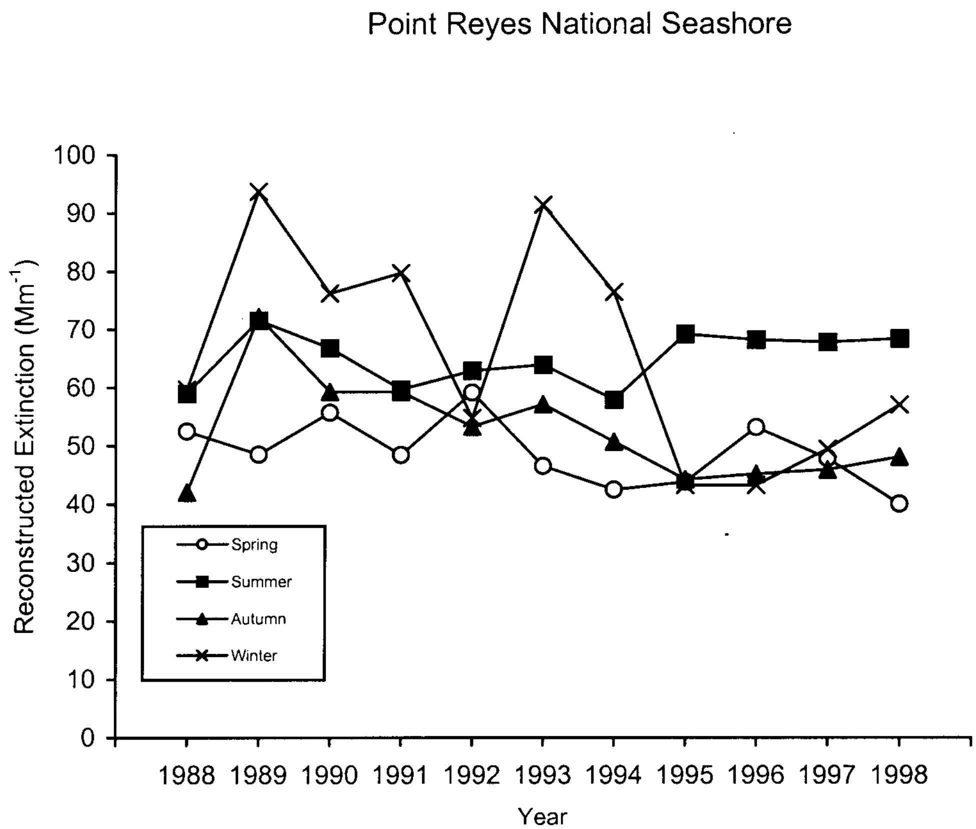


Figure VII-4. Seasonal average reconstructed extinction (Mm^{-1}), PORE, March 1988 through February 1999.

by season and by mean of cleanest 20%, mean of the median 20%, and mean of the dirtiest 20%. The "cleanest" and "dirtiest" signify lowest fine mass concentrations and highest fine mass concentrations respectively, with "median" representing the 20% of days with fine mass concentrations in the middle of the distribution. Each budget includes the corresponding extinction coefficient, standard visual range (km), and deciview (dv).

The segment at the bottom of each stacked bar in Figure VII-5 represents Rayleigh scattering, which is assumed to be a constant 10 Mm^{-1} at all sites during all seasons. Rayleigh scattering is the natural scattering of light by atmospheric gases. Higher fractions of extinction due to Rayleigh scattering indicate cleaner conditions.

b. Camera Data - View Monitoring

View monitoring slides document visual conditions and are an effective tool for interpreting the visual effects of measured optical and aerosol parameters or presenting monitoring program goals, objectives, and results to decision-makers and the public. The Point Reyes vista photographs presented in Figure VII-6 were chosen to provide a feel for the range of visibility conditions possible and to help relate the extinction/SVR/haziness data to the visual sense.

c. Site-Specific Data Interpretation

Data from other IMPROVE visibility sites around the country have been presented graphically (Figures I-3 and I-4) so that spatial trends in visual air quality for the Point Reyes and Pacific Coast regions can be understood in perspective. Figures VII-4 and VII-5 have been provided to summarize PORE visual air quality during the March 1988 through February 1999 period. Seasonal variances in the mean of the dirtiest 20% fractions are driven primarily by nitrate and sulfate extinctions. Non-Rayleigh atmospheric light extinction at PORE is largely due to sulfates, nitrates, and coarse mass. Unlike many parts of the western United States, a small fraction of extinction can be attributed to light absorbing carbons. Historically, visibility varies with patterns in weather, winds (and the affects of winds on coarse particles) and coastal fog.

Long-term trends fall into three categories: increases, decreases, and insignificant changes. The characterization of long-term trends can be a highly subjective exercise in that slopes and their significance can vary depending on the technique employed. Recently the IMPROVE aerosol network, initiated in March 1988, matured to a point where long-term trends of average ambient aerosol concentrations and reconstructed extinction can be assessed. In the IMPROVE report (Malm et al. 2000), the authors applied the Theil (1950) approach to describe trends for IMPROVE sites with eleven years of data. The distribution of $PM_{2.5}$ mass concentrations, reconstructed extinction expressed as deciview, and associated constituents were examined for each site. The data were sorted into three groups based on the cumulative frequency of occurrence of $PM_{2.5}$: lowest fine mass days, 0-20%, median fine mass days, 40-60%, and highest fine mass days, 80-100%. After sorting each group's average concentrations of $PM_{2.5}$ and selecting the associated principal aerosol species, scattering and/or absorption of each species, reconstructed light extinction and deciview were calculated. Figure VII-7 shows plots of the 10, 50, and 90 percentile groups at PORE for both $PM_{2.5}$ and deciview.

Given the visibility data summarized, large fluctuations in the highest (90th percentile) fine mass days were observed. The majority of data show insignificant change over the 1988 through 1999 period. However, Malm et al. (2000) discovered a decreasing trend in the long term 5-year average deciview at PORE.

D. RESEARCH AND MONITORING NEEDS

1. Deposition

Because the prevailing winds at PORE come off the Pacific Ocean and deposition of S and N are expected to remain low, additional monitoring of wet and/or dry deposition at PORE is not a high priority.

2. Gases

No gaseous pollutant monitoring is recommended.

3. Aquatic Systems

There are no needs relative to air quality.

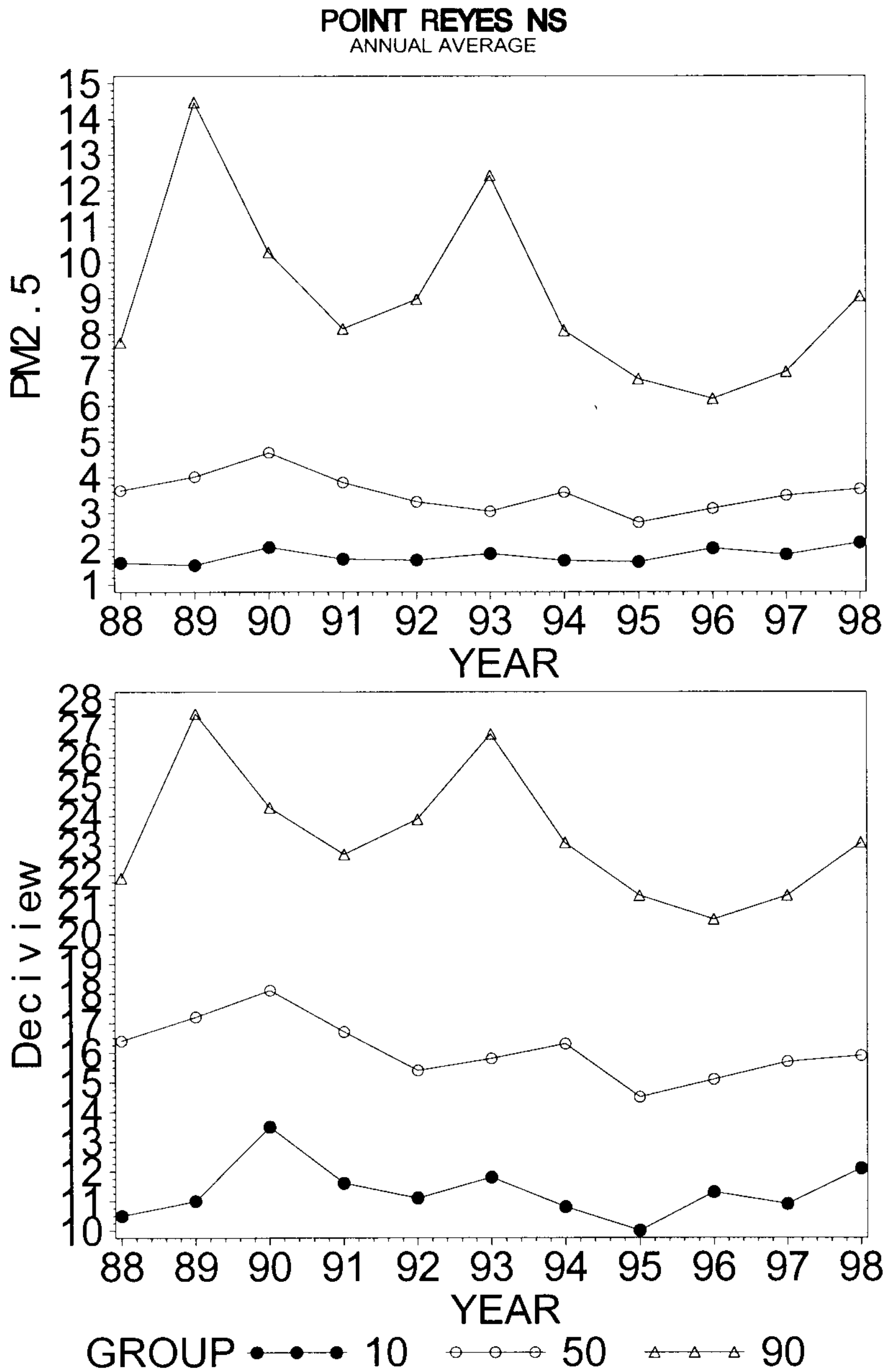


Figure VII-7. Trends in annual averages for $PM_{2.5}$ ($\mu g/m^3$) and visibility (deciview) for PORE.

4. Terrestrial Systems

Because ambient ozone levels at PORE are currently quite low, oxidant injury in vegetation is unlikely at the present time, and a long-term monitoring program for vegetation is probably not warranted at this time. However, ascertaining current foliar condition could be valuable for establishing a reference point in time, even if it only confirms that no injury is present. This could be done rather quickly and at low cost. There is often uncertainty associated with the evaluation of foliar injury. If ozone concentrations increase dramatically in the park in the future, it will be useful to have baseline foliar condition data under the currently low ozone exposure regime of the park. The existence of such data may later help to determine the likelihood that park vegetation has actually been damaged by increased ozone concentration, especially if such future damage is only of moderate severity.

If NPS feels that a reference survey is needed, the focus should be on the ozone bioindicator Douglas sagewort. Monitoring protocols from the Forest Health Monitoring manual (USDA Forest Service 1999) are recommended for establishing plots and collecting data (see Appendix). If ozone injury is detected in initial surveys, evaluations should be done on an annual basis in the late summer. If no injury is detected, additional surveys are probably not merited unless ambient ozone levels increase significantly. White clover (*Trifolium repens*) is another potential bioindicator for ozone, although it is not a native species.

5. Visibility

Other than continued IMPROVE aerosol monitoring, no additional visibility monitoring or research is recommended at this time.

VIII. REDWOOD NATIONAL PARK

A. GENERAL DESCRIPTION

Congress passed legislation in 1968 which established the Redwood National Park (REDW). The original designation encompassed 23,500 ha and was intended to preserve significant examples of virgin coastal redwood (*Sequoia sempervirens*) forests. Most of the land designated to be included within the national park was located in the lower one third of the Redwood Creek drainage basin. Shortly after establishment of the park, concern was expressed that timber harvesting on private lands upstream and upslope of the newly created park was threatening resources within the park. In response to this concern, a number of studies were initiated in 1973 on erosion, sediment transport, and aquatic habitat throughout the Redwood Creek drainage basin. The results of much of this research were summarized in a number of papers included in the volume by Nolan et al. (1995a). The results suggested that timber harvesting could increase the naturally high rates of erosion in Redwood Creek and have adverse impacts on redwood groves in the lower watershed. As a consequence, Congress authorized expansion of the park in 1978. In addition to the area that was added to the park, 100 km^2 of land directly upstream of the park were also designated as a park protection zone, in which timber harvest operations were to be reviewed by the NPS (Nolan et al. 1995b).

REDW was established to protect significant examples of the primeval coastal redwood forest and the streams and seashores with which they are associated for the purpose of public inspiration, enjoyment, and scientific study. Of the 42,900 ha within the park boundary, 11,330 ha are owned and managed by the State of California as three state parks: Jedediah Smith, Del Norte, and Prairie Creek State Parks (Figure VIII-1). State parks within the national park have some cooperative programs, but administration is accomplished as separate units. Within the boundaries of the national park and the three state parks, 15,375 ha of old-growth redwood forests are preserved. The park also preserves portions of the Smith and Klamath Rivers and Redwood Creek, which feature outstanding fishery and recreational resources. Fifty-six km of relatively undisturbed Pacific coastline include sandy beaches, rocky cliffs, and tidepools. About 500,000 visitors enter the park each year (Redwood National Park 1998).

REDW preserves the largest contiguous section of old-growth coast redwood forest. It includes some of the world's tallest trees and is renowned for its biotic diversity and inspirational atmosphere. The forest contains a number of rare and endangered species that are dependent on the integrity and old-growth character of the forest for their survival (Redwood National Park 1998).

Major changes have occurred in the natural resources of the park since the arrival of European culture. Half of the parkland has been directly altered by logging, plowing, pasturing, and other modifications of the vegetation. Alteration of the watershed by logging has resulted in exaggerated erosion with periodic large storms and floods. Erosion and sediment transport have resulted in widened stream channels, which in turn have destroyed or threatened streamside redwood groves and damaged aquatic resources. The occurrence of natural fires was limited by increasingly effective fire control programs.

The coastal strip of northern California in the vicinity of REDW once contained extensive dense stands of redwood trees. Logging activities from the mid-1800s to mid 1900s removed about 85% of the original redwood stands. A survey by the NPS in 1963 and 1964 revealed that, of the original redwood forest area of over 800,000 ha, only 15% remained, and only 2.5% was protected in state parks (Harris and Tuttle 1983).

Air quality can affect a number of the resource values of the park. Jeffrey pine woodland is found on serpentine soils in the Little Bald Hills area in the northeast corner of the park. Jeffrey pine (*Pinus jeffreyi*) is particularly susceptible to potential damage from ozone. There are also a

Redwood National Park and Surrounding Region

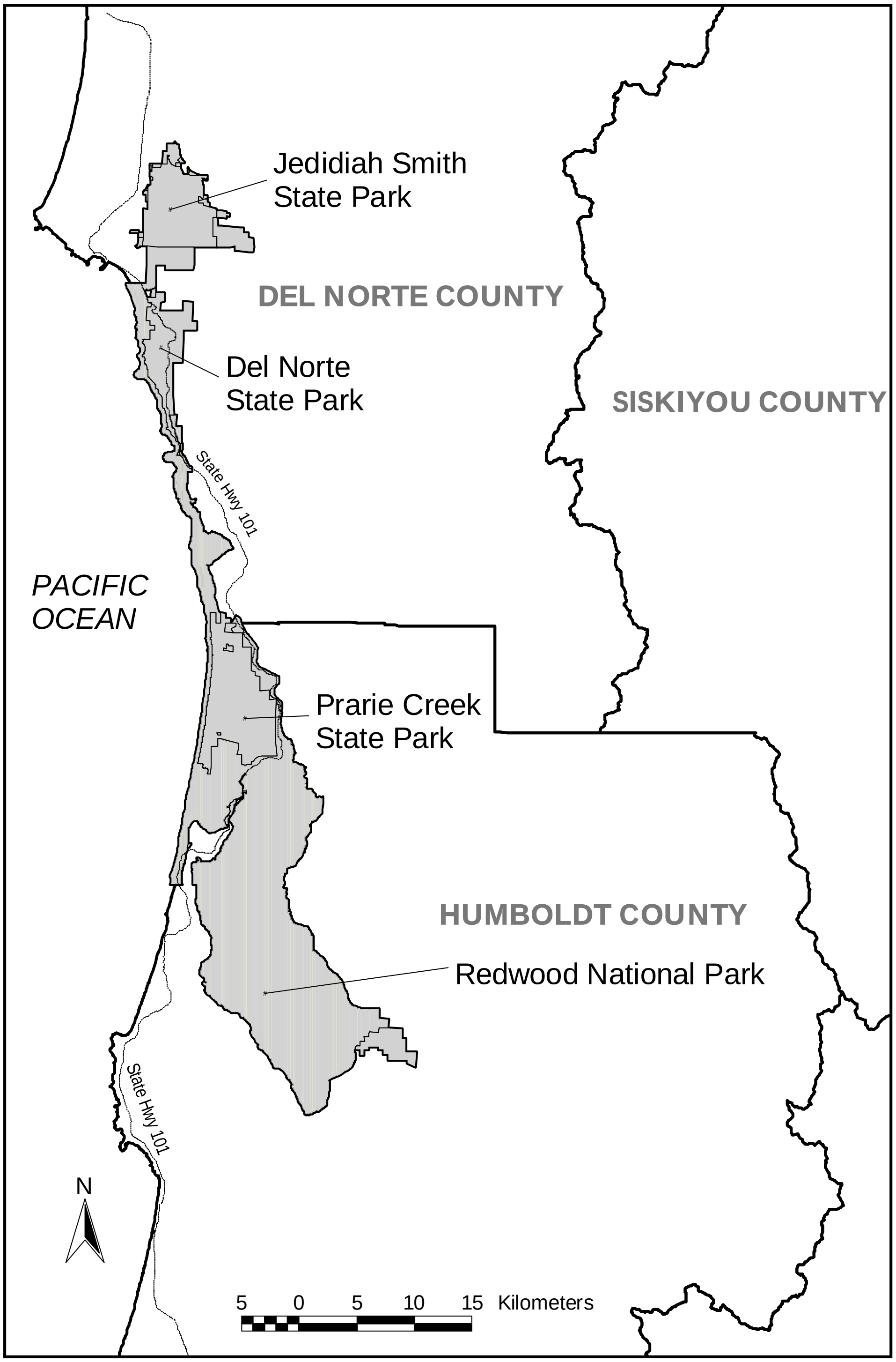


Figure VIII-1. Locational map for REDW.

number of overlooks and scenic vistas within the park that depend upon good air quality for their visibility. These include coastal scenes and inland vistas. Air quality in the region is generally considered to be good to excellent. There are no major sources of air pollution located adjacent to the park. The area is relatively isolated and access is limited. Although the climate is mild, summer fog and winter rain discourage some potential residents. The major industries within the region are timber harvesting, tourism, and fishing. Agriculture is also locally important.

The park has two distinctive physiographic environments, the coastline and the mountains of the Coast Range. The coastline is rugged with sections of steep rocky cliffs, broken by rolling slopes covered with grass and brush. The tidal zone is generally rocky, except for the Gold Bluffs Beach area, a seven-mile expanse of dunes and sandy beach adjacent to the nearly vertical Gold Bluffs, which rise 30 to 120 m in height. Major streams and ridge lines in the Coast Range trend north-west. Rounded mountain summits contrast with the steep side slopes that have been deeply incised by stream erosional processes. Elevation in the park ranges from sea level to 1,000 m at an unnamed peak in the Bald Hills.

Four Native American cultures with ties to REDW lands (Tolowa, Yurok, Chilula, and Hupa) represent a diverse indigenous presence, which includes three distinct languages and traditional arts, ceremonies, and methods of subsistence. The archaeological record of these peoples extends more than 4,500 YBP on REDW lands (Redwood National Park 1998).

1. Geology and Soils

Mesozoic age rocks of the Franciscan complex underlie most areas of the park. This complex was laid down on the ocean floor as deposits of sand and mud about 150 to 100 million YBP. The deposits were then carried eastward on the oceanic plate, accreted to the North American continent, and eventually uplifted in the formation of the Coast Range Mountains. Folding and faulting of the rock has further complicated the geology of the area. The Franciscan assemblage is bounded on the west by the San Andreas Fault which is several miles off the coast, and on the east by the South Fork Mountain Fault, which runs through the northeast corner of the park. There are also isolated exposures of chert and volcanic greenstones. The other major geologic formation in the park is a deposit of loosely consolidated sediments that are primarily fluvial in origin and known as the Gold Bluffs formation. These materials are thought to have been deposited as a river delta by the Klamath River more than 2 million YBP.

Studies of the geology and soils of REDW have largely been limited to the Redwood Creek watershed, Bald Hills, and Little Bald Hills areas. A substantial amount of geological mapping and research has been conducted in the Redwood Creek watershed (c.f., Cashman et al. 1995). Composition and distribution of bedrock units within the watershed and the distribution of major faults are key features in the geomorphic development of the Redwood Creek basin. Slope gradients, slope profiles, and drainage patterns reflect the properties of the underlying bedrock. The main channel of Redwood Creek generally follows the trace of the Grogan Fault, and other linear topographic features are also developed along major fault lines.

Much of the Redwood Creek watershed is underlain by virtually unmetamorphosed, Franciscan sedimentary rocks, including graywacke sandstone, mudstone, and conglomerate. Soil profiles developed from these geologic materials include the Hugo, Kneeland, Melbourne, Mendicino, and Tyson soil series. The west side of the watershed contains medium, gray, well-foliated quartz-mica schist, quartz-mica-feldspar schist, and quartz-graphite schist. These schists have weathered mostly to the Orick, Masterson, and Sites soil series (Iwatsubo et al. 1975).

Geological research conducted within the Redwood Creek watershed has included studies of hillslope erosional processes that remove material from upland slopes, interactions between streamside hillslopes and stream channels, and processes within the stream channel itself. A

large rainstorm in 1964 caused considerable erosion and sedimentation in this watershed. One suspected reason for such widespread effects from storm events is that channel and hillslope processes are likely connected by a positive feedback loop; alluvium introduced by a single landslide may be capable of initiating additional streamside landslides downstream. In addition, hillslope destablization can occur throughout long reaches of the channel as a result of a small number of failures in upstream locations (Nolan et al. 1995b).

Erosional processes have been extensively studied within the Redwood Creek watershed (Nolan et al. 1995b), which occurs within a geologic province characterized by some of the highest rates of erosion in the United States (Brown and Ritter 1971, Milliman and Meade 1983). The observed high rates of erosion along Redwood Creek are partly the result of inherently weak rock units situated in a tectonically active area with a Mediterranean climate (Nolan et al. 1995b).

Nolan and Janda (1995) reported water and suspended sediment discharge data from eight tributary streams to the downstream third of the Redwood Creek basin. Average annual rainfall in the eight tributary watersheds was about 180 cm. The study basins were forested primarily by redwoods, but prairie grass, brush and oak woodlands were found in up to 10% of the drainage areas. Ground-disturbing activities associated with logging were highly variable across the study watersheds, ranging up to 87% of the watershed area. The results suggested that timber harvest and associated activities caused tributary streams to become major sediment sources at times of high water discharge. Sampling conducted during nine storms suggested that water discharge per unit area in streams draining harvested terrain was about twice as high as from streams draining unharvested terrain under similar hydrologic conditions. Measurements of suspended sediment discharge were about ten times greater from harvested terrain as compared with unharvested terrain (Nolan and Janda 1995).

Soils within the park are generally well developed because of the mild, wet climate, which promotes a high degree of weathering of the underlying geologic materials. Most soils have strongly developed surface horizons that are rich in organic matter and nutrients, especially in areas that have coniferous forests, oak woodlands, and prairies (Redwood National Park 1998). The rate of erosion for the north coast region ranges from 6.4 to 16 tons/ac/yr, which is 10 to 100 times the rate for other river basins in the United States. Systematic mapping of soils in the lower Redwood Creek basin began in 1981. The soils inventory was conducted primarily in support of watershed rehabilitation, vegetation studies, and other resource management activities. An interim soil survey report of the Lower Redwood Creek basin was prepared by REDW in 1998.

Popenoe (1987) described ten forest soil series used in the REDW soil survey that was conducted in the Redwood Creek basin and also summarized laboratory data for 31 pedons that were sampled in the Redwood Creek watershed and the Little Bald Hills area of the park. Geologic units within the park include the mountains underlain by sedimentary and schist rocks of the Franciscan assemblage, serpentine rocks in the Little Bald Hills east of Crescent City, coastal plains sediments which are of Klamath origin, and alluvial soils of mixed lithology.

The substrate of the Bald Hills area is partly residuum from highly sheared Franciscan siltstone, sandy siltstone, and graywacke, and partly colluvium from these materials (Gordon 1980). The soils have low base saturation, probably due to leaching from the high amounts of rainfall received (Hektner et al. 1983). Erosion rates are high due to the high relief, the incoherent Franciscan bedrock, and rainy climate (Popenoe 1987).

There are two major soil groups in the Bald Hills area. The most common group in the prairies and oak woodland is the Xeralfs, formerly called Kneeland soils, which are characterized by a distinct increase in clay content with depth. These soils have impaired

drainage, which is likely a major cause for the resistence of these soils to colonization by Douglas fir. The second most common group of soils in the prairies and oak woodlands is the Umbrepts, many of which were formerly called Wilder soils. These soils are well drained throughout, and are confined mostly to upper slopes and ridges. They are highly vulnerable to surface erosion. Douglas fir has been actively invading the Umbrepts in the Bald Hills area (Redwood National Park 1992). Comparison of soils in the Bald Hills area with adjacent forest soils showed major differences associated with vegetation, relief, and drainage. The significant differences observed in color and chemistry indicate that the Bald Hills soils formed under different vegetation types because soil formation requires many centuries. Thus, vegetation patterns must have been stable for a long time to allow the observed soil differences to develop (Redwood National Park 1992).

2. Climate

The climate of the northern coast of California is mesothermal, characterized by heavy annual precipitation, which is concentrated in the winter months, and dry summers. The coastal strip is subject to frequent fog, which results from the cooling of the Pacific air masses over the cold coastal waters and the capping of this layer by the subsiding air from aloft. The fog moves inland along river canyons and frequently occurs in the redwood groves during the night and early morning hours. Fog is the dominant climatic feature, generally occurring on a daily basis during summertime and fairly frequently during the rest of the year. Fog occurs mostly within about 5 km of the coast, but may extend considerable distances up river corridors.

The climate of REDW is coastal Mediterranean, characterized by high precipitation amounts during winter, mild temperatures, and short, dry summers with frequent fog. Most precipitation occurs as rain, although snow falls occasionally at the higher elevations (Bradford and Iwatsubo 1978).

Climate varies with elevation and distance from the coast. Mean temperature is 8.2 ^{o}C in January and 14.9 ^{o}C in July in Klamath. Mean January temperature is 6.5 ^{o}C and mean July temperature is 15.2 ^{o}C at Prairie Creek Redwood State Park, north of Orick. Further from the coast line, winters are colder and summers are warmer. The mean daily high temperature in July is about 25^{o} C, with temperatures rarely exceeding 38^{o} C in the Bald Hills area. Mean daily low temperature in January is 2^{o} C (Sugihara and Reed 1987).

Annual precipitation in the park ranges from less than 150 cm to over 250 cm, 95% of which falls between October and May (Popenoe 1987). In general, the lower values occur near sea level and the higher values at the higher elevations (Redwood National Park 1998). Average annual rainfall in the Redwood Creek watershed ranges from about 178 cm per year at Orick to 254 cm per year or more near the head of the watershed (Iwatsubo et al. 1975).

Prevailing winds are from the northwest or south southwest. Occasionally in the fall, a warm dry wind comes from the east. Weather data for the park are available from the National Weather Service stations in Eureka and Crescent City.

Dawson (1998) reported the results of a three-year study of fog inputs and the use of fog water by plants inhabiting redwood forests in and near REDW. On average, 34% of the annual hydrologic input was from fog drip off of the redwood trees. Stable H and O isotope analyses of water from fog, rain, soil water, and xylem water were used to characterize the water sources used by the dominant plant species. During summer, when fog was most prevalent, ~19% of the water within the redwood trees and ~66% of the water within the understory plants came from fog water inputs to the ecosystem. These results confirm the importance of fog for the water relations of the plants and the hydrology of the redwood forest.

3. Biota

Vegetation distribution within the park follows the gradient of climate from the coast to the inland areas. Sitka spruce (*Picea sitchensis*) predominates along the coastline, within 1 or 2 km inland. With increasing elevation and distance from the coast, sitka spruce trees drop out of the forest, and Douglas fir (*Pseudotsuga menziesii*) become co-dominant with redwood. In areas that have been logged within the last 50 years, red alder (*Alnus rubra*) dominates the canopy, especially in areas underlain by schist.

Redwood trees grow on a variety of soil and rock types, but most of the range of the redwood is underlain by rocks of the Franciscan formation, which includes sandstone, schist, and other sedimentary and metamorphic rocks that were deposited under the Pacific Ocean and since have been uplifted by movements of the earth's crust. They are the predominant rocks found in REDW. There are also geological substrates where redwoods do not thrive. For example, redwoods are absent from the greenish serpentine bedrock and reddish soils of the Little Bald Hills adjacent to Jedidiah Smith Redwood State Park. Redwoods are uncommon on the steep ridges of the Bald Hills area. Along most of the coast, sitka spruce out-compete the more salt-sensitive redwood trees. The spruce forests of the coast buffer redwood forests from salt and wind.

Redwood forests are best developed on cool, moist, lower elevation sites near the coast. Redwood trees eventually drop out of the forest at the higher elevation sites which tend to be hotter and drier. This is probably a function of both climate and fire frequency. The species and its fossil predecessors were widely distributed throughout North America in the past and redwood fossils have been found across the United States. However, the natural range of the coastal redwood has been reduced to California and southern Oregon because of climatic changes, the rise of mountain ranges, and other events. They are less drought-resistant than most other conifers. In fact, water stress and light intensity at the tops of the tall redwood trees reduce the size of the needles, change the shape and orientation of the needles, and also reduce growth and the length of the stems. On the lower branches, however, needles assume a feather-like arrangement in response to the shade and higher relative humidity near the floor of the forest. Thus, the foliage varies from shade to sun in response to physiological stress.

The coast redwood grows along a narrow coastal strip from near Brookings, Oregon to just south of Monterey. The species grows best where fog moderates the typical summer drought conditions. Winters are mild with abundant rain; frost is common, but hard freezes and snow are unusual (Eifert 1993). Coast redwoods grow best only as far inland as the extent of the coastal maritime climate. The fog protects the trees from desiccating conditions during summer. Some of the tallest measured trees on earth grow on an alluvial terrace along Redwood Creek. One tree in the tall trees grove was measured at 112 m. These alluvial terraces and lower elevation areas near the coast provide optimal growing conditions for redwood trees.

The redwood forest is dominated by coast redwood, with Douglas-fir, grand fir, Sitka spruce, western hemlock, and associated species varying according to whether a site is upland, riparian, alluvial, or near the ocean. Associated hardwoods include tanoak (*Lithocarpus densiflorus)*, Pacific madrone (*Arbutus menziesii*), bigleaf maple (*Acer macrophyllum*), California bay (*Umbellularia californica*), and red alder. Pacific rhododendron (*Rhododendron macrophyllum*), sword fern (*Polystichum munitum*), redwood sorrel (*Oxalis oregana*), and numerous other herbaceous species are found in the understory. There are about 16,000 ha of old-growth forest remaining in the park. Redwood is a striking dominant in these stands, growing to be over 100 meters tall, up to 6 meters in diameter, and 400 to 800 years old (occasionally over 2,000 years).

1989). There are no threatened or endangered plant species known to occur in the Bald Hills area (Redwood National Park 1992).

The Jeffrey pine/chaparral/knobcone pine vegetation type includes several distinct vegetation communities that are grouped together because they are localized in the Little Bald Hills area, an area of about 525 ha southeast of Jedidiah Smith Redwood State Park. Vegetation in this area is sparse, due to the presence of serpentine soils. The driest ridgetops here are occupied by widely scattered Jeffrey pine. The chaparral vegetation type is located downslope, dominated by manzanita, golden chinquapin, rhododendron, huckleberry oak (*Quercus vaccinfolia*), and other evergreen shrubs interspersed with stands of knobcone pine. Knobcone pines are restricted to the Little Bald Hills area and require frequent fires to survive.

Logging has had the greatest impact on the vegetation in the park. Most timber harvesting in parklands occurred between 1950 and 1978. The park contains between 18,000 and 20,000 ha of second growth forest that are in varying stages of regrowth (Redwood National Park 1998).

In prehistoric times, lightning and Native Americans were the ignition sources for fires in the park. Veirs (1981, 1982) suggested, based on tree age distribution in old-growth redwood forests, that fires that significantly influenced stand composition occurred at 250 to 500 year intervals in moist coastal sites, 100 to 250 year intervals in intermediate sites, and 33 to 50 year intervals in high-elevation interior sites. Such fires were generally surface fires that burned understory fuels and had relatively little canopy involvement (Redwood National Park 1995).

Fire suppression has probably not had a large effect on bottomland redwood forests or on the fire potential in those areas, except in drought years. Fire suppression has had a greater influence on upland mixed conifer and Douglas fir dominated forests where drier conditions allow lightning-caused fires to spread.

REDW offers opportunity for observing both marine and terrestrial mammal species. It is one of the better parks in the state in which to see elk (*Cervus elaphus*) and brush rabbits (*Sylvilagus bachmani*). Elk are common in the central part of the park and can almost always be seen along Prairie Creek. The opportunity to see elk is one of the main attractions of the park. In particular, Boyes Prairie, located in Prairie Creek Redwood State Park, is well known for its elk herd. Black-tail deer (*Ododoileus hemionus*) are also common. Rodent species include Douglas' squirrels (*Tamiasciurus douglasii*), California gray squirrels (*Sciurus carolinensis*), Townsend's chipmunk (*Eutamias townsendii*), and perhaps the golden mantle ground squirrel (*Spermophilus lateralis*). Bobcats (*Felis rufus*) are common and mountain lions (*Felis concolor*) are also resident in the park, but are rarely encountered. Both gray fox (*Urocyon cinereoargenteus*) and coyote (*Canis latrans*) may be encountered almost anywhere within the park. Black bears (*Ursus americanus*) are also common. Mustelids include both the spotted skunk (*Spilogale putorius*) and striped skunk (*Mephitis mephitis*), long tailed weasel (*Mustela frenata*), ermine (*Mustela erminea*), and marten (*Martes americana*). Mink (*Mustela vison*) are uncommon, but occur in the Redwood Creek area along with river otters (*Lutra canadenis*). Other terrestrial mammal species that may be observed within the park include the dusky-footed woodrat (*Neotoma fuscipes*), beaver (*Castor canadensis*), ringtail (*Bassariscus astutus*), and muskrat (*Ondatra zibethicus*). The only marine mammal that is resident to park waters is the harbor seal (*Phoca vitulina*). However, other seal and sea lion species are frequently observed, including the northern elephant seal (*Mirounga angustirostris*), northern sea lion (*Eumetopias jubatus*), and California sea lion (*Zalophys californianus*). The northern fur seal (*Callorhinus ursinus)* has also been observed. Gray whales (*Eschrichtius robustus*) are frequently sighted during migration and a number of other whale and dolphin species have been observed along the park's coastline (Van Gelder 1982).

The park contains suitable habitat for a number of small mammals, whose range is confined to moist, dense coniferous forest and associated coastal habitats in the Pacific Northwest. These include the marsh shrew (*Sorex bendirii*), shrew-mole (*Neurotrichus gibbsii*), coast mole (*Scapanus orarius*), mountain beaver (*Aplodontia rufa*), California red-backed vole (*Cleithronomys californicus*), red tree vole (*Arborimus longicaudus*), Townsend's vole (*Microtus townsendii*), and Pacific jumping mouse (*Zapus trinotatus*).

More than 400 species of birds have been reported in the park, 200 of which are known to breed there. Many state and federal threatened, endangered, or candidate bird species occur in the park. Two bird species associated with old-growth forests, the northern spotted owl (*Strix occidentalis caurina*) and the marbled murrelet (*Brachyramphus marmoratus marmoratus*), are the most prominent listed terrestrial species. Other listed and formerly listed species include brown pelican (*Pelecanus occidentalis*), western snowy plover (*Charadrius alexandrinus nivosus*), American peregrine falcon (*Falco peregrinum anatum*), bald eagle (*Haliateus leucocephalus*), bank swallow (*Riparia riparia*), great gray owl (*Strix nebulosa*), and Aleution Canada goose (*Branta canadensis leucopareia*).

The REDW region is rich in herpetofauna, especially amphibians, both in species diversity and abundance. Several species found in the northcoast region are listed as California Species of Concern (CSC), including the federally threatened California red-legged frog and two candidates for federal listing: Del Norte salamander (*Plethodon elongatus*) and western pond turtle (*Clemmys marmorata*). In addition, the tailed frog (*Ascaphus truei*), foothill yellow-legged frog (*Rana boylii*), and Olympic salamander (*Rhyacotriton olympicus*) are listed as CSC but are not candidates for federal endangered species status. Current knowledge regarding species distributions within the park is limited. There has been no survey of representative habitats throughout the park, although site-specific studies have been conducted to compare amphibian distribution and abundance in logged versus old-growth forest and to examine the effects of sedimentation on amphibian habitat.

One fish species, the tidewater goby (*Eucyclogobius newberryi*), which inhabited the coastal estuaries in the park as recently as 1980, is listed as endangered. In addition, three salmonid species that inhabit park waters are listed or proposed for listing under the Federal Endangered Species Act. They include the chinook (*Oncorhynchus tshawytscha*) and coho salmon (*Oncorhynchus kisutch*) and steelhead (*Oncorhynchus mykiss*). During winter and spring, steelhead and chinook salmon constitute major sport fisheries in the Smith River. Because of the large size of Smith River salmon and steelhead, this river has national prominence among sport fishing enthusiasts. Coho salmon are also widely distributed in the Smith River Basin, but the most significant population is in Jedidiah Smith Redwood State Park. Mill Creek in Del Norte Coast Redwood State Park and Jedidiah Smith Redwood State Park, and also Prairie Creek in Prairie Creek Redwood State Park, are important spawning grounds for chinook and coho salmon and for steelhead and coastal cutthroat trout. A variety of other fish species occur in the park (Table VIII-1).

B. EMISSIONS

REDW is located in Humboldt and Del Norte counties, within the North Coast Area Air Basin (NCAB). Its coastal location, combined with prevailing northwesterly winds off the Pacific Ocean, place REDW in a generally upwind position relative to most emission sources within the NCAB.

Emissions from counties within 140 km of REDW are shown in Table VIII-2. The principal species of concern are ozone precursors (NO_x and ROG) and PM. SO_2 emissions are not high. No major point sources are located in Del Norte County. In Humboldt County, the closest

Table VIII-1. Fish species known to occur in Redwood Creek and tributaries. (Source: Redwood National Park, unpublished fish species list.)

Common Name	Species Name
Chinook salmon	*Oncorhynchus tshawytscha*
Coho salmon	*Oncorhynchus kisutch*
Steelhead trout	*Oncorhynchus mykiss*
Coastal cutthroat trout	*Oncorhynchus clarki*
Chum salmon[a]	*Oncorhynchus keta*
Eastern brook trout[b]	*Salvelinus fontinalis*
Prickly sculpin	*Cottus asper*
Riffle sculpin	*Cottus qulosus*
Coastrange sculpin	*Cottus aleuticus*
Pacific lamprey	*Lampetra tridentata*
Humboldt sucker	*Catostomus occidentalis humboldtianus*
Three spine stickleback	*Gasterosteus aculeatus*

[a] Uncommon, one reported occurrence
[b] Alien species (Coyote Creek spring pond)

Table VIII-2. 1995 Emissions from counties within 140 km of REDW. Source: CARB Almanac, 1999b; SO_x from CARB Emissions Website, 1999a. Units are 1000 tons/year.

County	NO_x	ROG*	PM_{10}	CO	SO_x
Del Norte	1.5	2.2	3.3	36.5	0.0
Humboldt	8.4	8.8	7.7	72.3	1.1
Trinity	1.1	2.2	6.2	23.0	0.0

* Reactive Organic Gases

sources that emit at least 100 tons/year of ROG, NO_x, PM_{10}, or SO_2 are located near Arcata, Eureka, and Scotia (Figures II-3 through II-6). As of 1996, stationary sources accounted for 21% of ROG emissions, 22% of NO_x emissions, and 7% of PM_{10} emissions within Humboldt and Del Norte counties (CARB, 1998b). Mobile sources dominated NO_x (73%), while area sources (road dust, residential fuel combustion, waste burning, and construction) dominated PM emissions (86%).

A significant air pollutant in the park is PM_{10}. In the park, particulates are derived primarily from burning of timber harvest slash piles. There are two pulp mills near Eureka about 40 km southwest of the southern end of the park and also a saw mill near Crescent City, within 2 km of the park boundary. Sawmill smoke, burning of debris piles, smoke from woodstoves, and prescribed burns are potential sources of air pollution that originate in or near the park. The most significant air pollution point sources in the vicinity of the park are industrial sites along

Humboldt Bay, 80 km to the south. Local visibility is often impaired by fog, rain, low clouds, and salt spray haze. (Redwood National Park 1998).

An inventory of in-park emissions has recently been compiled by the NPS-Air Resources Division. The results are presented in Table VIII-3.

Table VIII-3. Summary of 1998 stationary and area, and mobile source emissions (tons/yr) at REDW.

Activity	Particulates	Sulfur Dioxide	Nitrogen Oxides	Carbon Monoxide	VOCs	HAPs[1]
Stationary and Area Source Emissions						
Stationary Combustion Sources						
Space and Water Heating units	0.00	0.00	0.10	0.01	0.00	–
Generators	0.00	0.00	0.00	0.00	0.00	–
Woodstoves	0.18	0.00	0.02	1.35	0.31	–
Combustion Emission Subtotal	0.18	0.00	0.12	1.36	0.31	–
Area Sources						
Campfires	0.23	0.00	0.06	1.92	0.26	–
Prescribed Burning	125.10	5.48	2.20	915.00	29.58	–
Miscellaneous Area Sources	0.00	0.00	0.00	0.00	0.09	–
Area Source Emission Subtotal	125.33	5.48	2.25	916.92	29.93	–
TOTALS	125.51	5.48	2.37	918.28	30.24	–
Mobile Source Emissions						
Road Vehicles						
Visitor Vehicles	1.93	–	2.73	10.87	0.94	–
NPS/GSA Road Vehicles	1.39	–	3.50	10.77	0.97	–
Vehicle Emission Subtotal	3.32	–	6.21	21.64	1.90	–
Nonroad Vehicles						
NPS Nonroad Vehicles	0.26	–	1.32	7.91	4.88	–
TOTALS	3.57	–	7.53	29.55	6.78	–

[1] Hazardous air pollutants, based on the list compiled by EPA

C. MONITORING AND RESEARCH ACTIVITIES

1. Air Quality

In conjunction with the NPS Air Resources Division, the park established an air quality and meteorological monitoring station in 1987 at the Requa Maintenance Operations Center. Visibility, ambient air quality, and meteorological monitoring have been conducted at the park. Visibility monitoring included an automated camera and fine particulate sampler. Measured

meteorological parameters included wind speed and direction, temperature, dew point, solar radiation, and precipitation. Air quality monitoring within REDW has included particulate matter ($PM_{2.5}$, PM_{10}), hourly ozone, and SO_2 (Table VIII-4). Passive ozone samplers have not been located in the park. An ozone and meteorological monitoring site operated at Requa near the mouth of the Klamath River between 1987 and 1995. Since mid-1984, a CADMP wet deposition site has been located at Gasquet, approximately 10 km northeast of the northernmost park boundary. CADMP also co-located a dry-deposition monitor at Gasquet from 1988 through 1992. There is not a CASTNet site within the park for monitoring dry deposition.

Table VIII-4. Air quality monitoring at REDW.

Species	Site within park	Site within 50 km
Ozone, hourly	NPS**	ARB
Ozone, passive		
SO_2	NPS	
PM_{10}	IMPROVE	
$PM_{2.5}$	IMPROVE	
Wet deposition		ARB**
Dry deposition		ARB**
Visibility		

* New site
** Closed before 1996

Because of the prevailing westerly ocean winds, scarcity of local pollution sources, and low population in the area surrounding the park, the air quality in REDW is considered good to excellent. All federal standards for air quality are consistently achieved.

a. Wet Deposition

At the nearest CADMP site (Gasquet), wet S deposition ranged from 1.3 to 3.9 kg/ha/yr as S (equivalently, 3.9 to 11.7 kg/ha/yr as SO_4^{2-}) during the period 1990 through 1998 (Table VIII-5). Because Gasquet is located near the ocean, some SO_4^{2-} derives from marine aerosol; the amounts of SO_4^{2-} deposition not of marine origin ranged from 0.3 to 1.2 kg/ha/yr as S (Blanchard et al., 1996). The annual NO_3^- and NH_4^+ deposition rates at Gasquet were in the ranges of 0.3 to 0.7 kg/ha/yr and 0.1 to 0.6 kg/ha/yr as N, respectively, yielding a multi-year mean total wet N deposition rate of 0.8 kg/ha/yr (Table VIII-5). During the period 1990 through 1998, the annual-average H^+ concentration in precipitation at Gasquet ranged from 2.7 to 5.8 µeq/L (pH 5.57 to 5.24; Table VIII-6).

Summary statistics on rainfall chemistry were reported by Iwatsubo et al. (1975) and Bradford and Iwatsubo (1978) using rainfall samples collected within REDW. Median SO_4^{2-} concentration was reported to be 0.7 mg/L (15 µeq/L) and total NO_2^{2-} plus NO_3^- concentration was reported as 0.03 mg/L of N (2 µeq/L). It is not known why the SO_4^{2-} concentrations reported by Iwatsubo et al. (1975) and Bradford and Iwatsubo (1978) were twice as high as concentrations reported in more recent years by CADMP (Table VIII-6). There may have been

Table VIII-5. Wet deposition of S and N at the CADMP site of Gasquet, ~10 km. from REDW (Source: Blanchard et al, 1996; Dwight Oda and Brent Takemoto, CARB, 1999, personal communication). Units are kg/ha/yr.

Water Year*	Sulfur	NO_3-N	NH_4-N	Total Inorganic N
1985	2.7	0.5	0.4	0.9
1986	2.4	0.8	0.5	1.4
1987	2.1	0.8	0.4	1.2
1988	1.6	0.8	0.6	1.3
1989	2.6	0.5	0.7	1.2
1990	2.0	0.4	0.3	0.7
1992	1.4	0.4	0.2	0.6
1993	3.4	0.7	0.2	0.9
1994	1.3	0.3	0.1	0.4
1995	2.9	0.7	0.3	1.0
1996	3.9	0.7	0.3	1.0
1997	3.0	0.7	0.6	1.3
1998	2.6	0.5	0.2	0.7
Average	2.5	0.6	0.4	1.0

* The CARB water year is July 1 through June 30. For instance, water year 1995 is for July 1, 1994 through June 30, 1995.

Table VIII-6. Wetfall chemistry at Gasquet, ~10 km from REDW. Units are µeq/L, except precipitation (cm). Source: Blanchard et al, 1996; Dwight Oda and Brent Takemoto, CARB, 1999.

WATER YEAR*	Prec	H^+	SO_4^{-2}	NH_4^+	NO_3^-	Ca^{+2}	Mg^{+2}	Na^+	K^+	Cl^-
1985	181.1	4.1	9.3	1.8	1.6	7.7	11.5	46.2	1.7	55.8
1986	236.8	4.3	6.4	2.5	1.6	2.1	7.7	30.7	1.2	37.6
1987	185.4	5.1	7.2	2.9	1.7	5.3	8.3	36.1	0.9	43.2
1988	173.3	4.2	5.7	3.1	2.4	4.5	7	29.1	0.8	35.1
1989	213.5	5.2	7.5	1.7	2.4	4.3	7.1	35.7	1.3	38.6
1990	169.9	5.8	7.4	1.3	1.5	6.8	5.1	25.0	0.8	29.1
1992	138.0	3.9	6.3	0.8	2.2	5.6	10.6	26.4	1.0	32.4
1993	268.0	2.7	8.0	0.6	1.9	8.9	13.7	37.6	1.0	43.1
1994	133.7	3.4	6.0	0.5	1.5	11.7	15.1	29.9	0.9	33.7
1995	281.1	4.2	6.5	0.7	1.7	14.1	19.8	35.2	1.5	39.6
1996	296.7	4.9	8.1	0.7	1.8	5.6	14.6	49.7	1.3	42.3
1997	294.2	4.4	6.4	1.5	1.6	6.3	11.4	34.1	1.2	38.2
1998	224.4	4.8	7.3	0.6	1.6	10.2	13.9	43.2	1.2	51.7
Ave.	225.7	4.3	7.0	0.8	1.7	8.7	13.0	35.1	1.1	38.8

* The CARB water year is July 1 through June 30. For instance, water year 1995 is for July 1, 1994 through June 30, 1995.

methodological differences or the site of the earlier sampling may have received higher marine aerosol contributions and therefore higher concentrations of SO_4^{2-} derived from seawater.

b. Occult/Dry Deposition

The CADMP co-located wet and dry deposition samplers at Gasquet (Blanchard et al., 1996). Dry deposition was calculated from measurements of the ambient concentrations of both gas-phase and particulate species (Table VIII-7). Mean dry deposition rates of oxidized N species summed to ~0.8 kg/ha/yr (as N), compared with the multiyear mean wet NO_3^- deposition of 0.5 kg/ha/yr (as N) at the same location. The CADMP wet and dry deposition samplers also indicated that dry S (SO_2 plus aerosol SO_4^{2-}) deposition rates were about 0.2 kg/ha/yr compared with 2.6 kg/ha/yr wet SO_4^{2-} deposition (uncorrected for marine SO_4^{2-}, Table VIII-5), or 0.9 kg/ha/yr wet SO_4^{2-} deposition (corrected for marine SO_4^{2-}, Blanchard et al. 1996). The mean dry NH_3 plus aerosol NH_4^+ deposition was 0.8 kg/ha/yr (as N) compared with 0.3 kg/ha/yr (as N) wet NH_4^+ deposition (Tables VIII-5 and VIII-7). Total N deposition was 1.0 kg/ha/yr wet and 1.6 kg/ha/yr dry (Tables VIII-5 and VIII-7), which is lower than expected vegetation injury thresholds.

Table VIII-7. Long-term annual averages of calculated dry deposition fluxes at Gasquet using data from 1988-94. Units are kg/ha/yr as SO_2, ozone, NO_2, etc. The averages were constructed by weighting four seasons equally. Source: Blanchard et al (1996).

Gas-Phase Species					Particulate		
SO_2	Ozone	HNO_3	NO_2	NH_3	NO_3^-	SO_4^{2-}	NH_4^+
0.25	20.57	1.03	1.9	0.9	0.11	0.29	0.06

c. Gaseous Monitoring

Data from the hourly ozone monitor show that ozone concentrations and exposures for the period 1992-1995 were low relative to other areas of the state (Table VIII-8). The monitoring equipment was removed in June, 1995. No passive ozone samplers are sited within REDW.

Annual maxima and means compiled from 24-hour resolution SO_2 samples are displayed in Table VIII-9 for the period 1988-1995. SO_2 measurements were discontinued after 1996 due to concerns about their accuracy. The measurements are considered sufficiently accurate to show that the measured SO_2 concentrations were well below the levels at which plant injury has been documented, ~40 to 50 ppb 24-hour average and 8-12 ppb annual average (Peterson et al, 1992).

Table VIII-8. Summary of ozone concentrations and exposure from REDW. (Source: Joseph and Flores 1993; National Park Service, Air Resources Division 2000).

Year	Maximum Daily 1-hour Value (ppbv)	2nd Highest Daily 1-hour Value	Number of Daily Maximum 1-hour values ≤ to 125 ppb	3-year Average Number of Exceedances	Maximum 9am-4pm Average (ppbv)	Sum06 (ppbv-hour)[a]	Number of Valid Hours of Ozone Measurements
1988	68	60	0	na	44	0	8007
1989	47	47	0	na	43	0	7521
1990	61	53	0	0	45	0	7488
1991	54	52	0	0	44	0	7677
1992	64	55	0	0	49	0	7925
1993	54	50	0	0	41	0	7827
1994	51	50	0	0	47	0	8079
1995	52	50	0	0	45	0	2771

[a] maximum 8 am - 8 pm 90-day rolling average

Table VIII-9. Maximum and mean SO_2, from 24-hour-resolution samples at REDW. Samples are collected every 3-4 days, unless noted. (Source: NPS Air Resources Division). Units are ppb.

SO_2	1988	1989	1990	1991	1992	1993	1994	1995
Maximum	0.37*	0.15*	0.15**	0.30*	0.25**	0	0.28	0.03*
Mean	0.07*	0.05*	0.03**	0.05*	0.02**	0	0	0.01*

na Not available
* Less than 50 samples collected for the year
** 50-75 samples collected for the year

2. Aquatic Resources

a. Water Quality

Surface water resources within the park include salt water, streams and rivers, estuaries, and lagoons. Three large river systems drain most of the park and have cut deep gorges through the forested, mountainous terrain. Redwood Creek in the southern portion of the park has a total drainage area of 720 km^2, one-third of which is located within the national park. For much of its length, Redwood Creek follows the Grogan fault, which cuts between unmetamorphosed to slightly metamorphosed clastic sedimentary rocks on the east and metamorphic schistose rocks to the west. Most tributary streams are short and steep and drain small areas. Thus, the tributaries to Redwood Creek form a trellised drainage pattern of low order, high gradient streams.

Klamath River is the largest river in the north coast area, with a drainage area of 39,000 km^2 in northern California and southern Oregon. It flows through a narrow strip of parkland in the central portion of the park. Three major hydroelectric dams in California regulate flow in the

Klamath and divert water to other uses. The lower Klamath is part of the federal Wild and Scenic River System.

In the northern portion of the park is the Smith River, which drains 1,600 km^2 of southern Oregon and northern California. Its watershed is very steep and prone to landslides. It includes 5,000 km of prime habitat for salmon and steelhead (Redwood National Park 1985). It is part of the state and federal Wild and Scenic River System. Both the Smith and Klamath Rivers drain large mountainous areas and are significantly influenced by snowmelt. In contrast, snowmelt has only a minor impact on the runoff in Redwood Creek.

There are no natural ponds or lakes within the park, although lagoons, sloughs, and marshes occur. There are also several ponds adjacent to former mill sites within the park, as well as fire suppression ponds and sediment catchment basins. There is a dense network of coastal streams (Figure VIII-3).

Subsequent to the Redwood National Park Expansion Act in 1978, the park initiated a hydrologic monitoring program in cooperation with the U.S. Geological Survey. A network of rainfall measurement, stream gaging, and sediment sampling stations was established along Redwood Creek and several of its tributaries. Some of the gages were operated for a few years and others continue to be operated (Klein 1998). A second network of stream gaging stations with suspended sediment sampling was also established in 1990 in Prairie Creek, a tributary to Lower Redwood Creek, whose drainage area is mostly within REDW. About 66% of the annual precipitation in the Redwood Creek watershed occurs as runoff, producing an average daily flow of about 30 m^3/sec at Orick (Nolan and Marron 1995).

Streams are generally small and steep and do not have well developed floodplains. Rivers and streams throughout the park are subject to frequent flooding due to the heavy amounts and seasonal concentration of precipitation. Flooding near the mouths of the rivers is also common in response to high tides in conjunction with heavy rains and it is often augmented by high winds. Following the 1964 flood, the U.S. Army Corps of Engineers constructed flood control devices on the lower portion of Redwood Creek, from the confluence with Prairie Creek downstream to within 300 m of the Pacific Ocean.

Within the park's boundaries, there are estuaries at the mouths of the Klamath River and Redwood Creek. During construction of the Highway 101 bypass around Prairie Creek Redwood State Park from 1987 to 1992, about 164,000 m^3 of gravel was mined from between the levies of the lower portion of Redwood Creek. This mining activity removed point bars and pools, widened the baseflow channel, spread the flow out, and reduced water depth at lower flows. Construction of the bypass caused substantial increases in sediment in Prairie Creek and its tributaries (Redwood National Park 1998).

Overall, the water quality in the park meets or exceeds the water quality objectives established by the North Coast Regional Water Quality Control Board. The major water quality concern is nonpoint source pollution, which is widespread within the park and includes sedimentation, increased streamwater temperature, and nutrient enrichment. Activities that have contributed to nonpoint source pollution include logging, mining, construction, ranching, sand and gravel operations, and wastewater effluent disposal. Land use activities have had their greatest impact towards the southern portions of the park, with the Smith River in the north having the least amount of water quality degradation.

Averett and Iwatsubo (1975) reported the results of water quality measurements in Redwood Creek and selected tributaries. Their results did not suggest a high degree of sensitivity of the water chemistry to adverse effects of acid deposition. Average specific conductance measurements were consistently greater than 35 μS/cm and total alkalinity, as bicarbonate, averaged greater than 10 mg/L (ANC ≈ 200 μeq/L). pH values were variable,

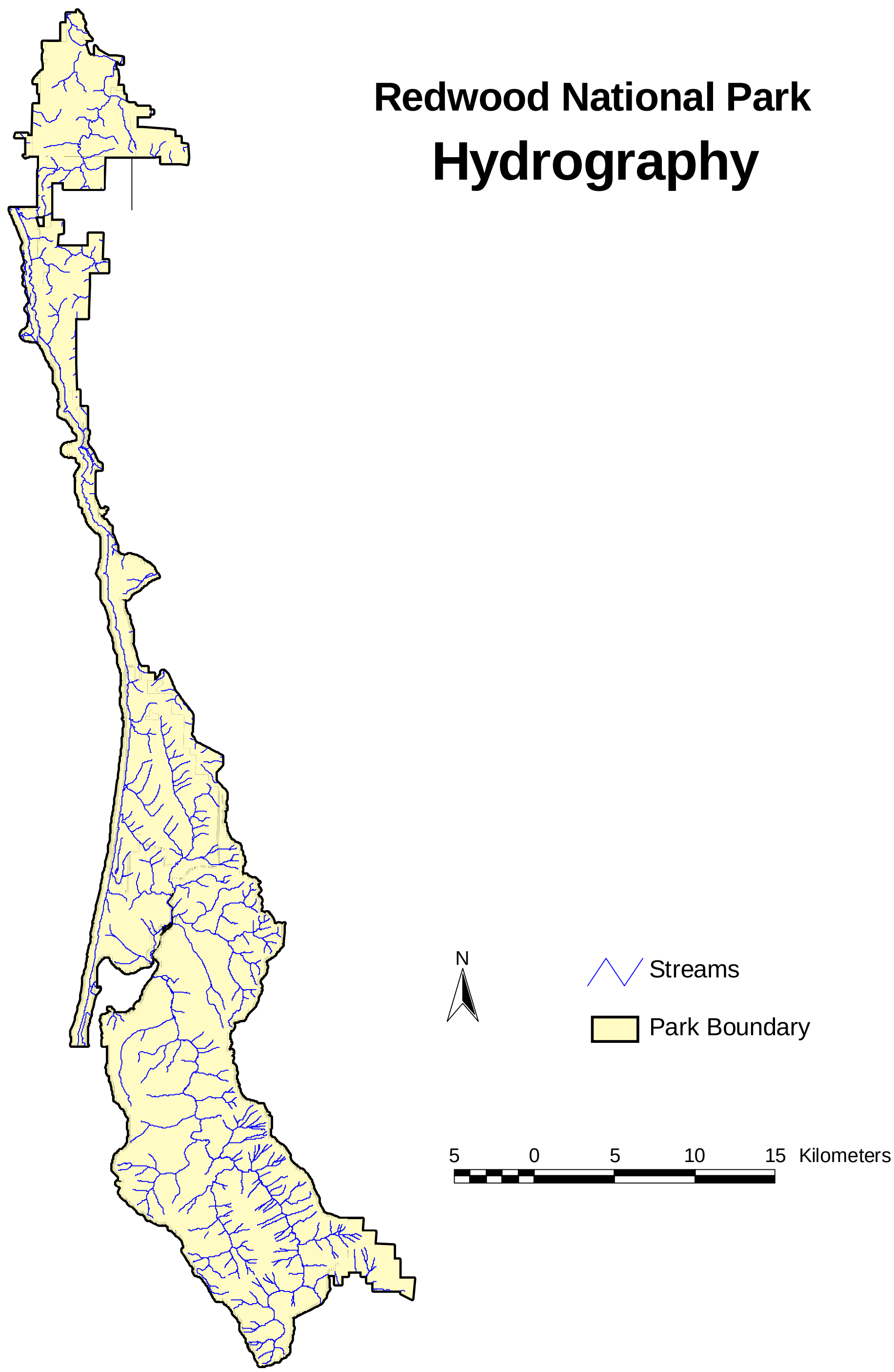


Figure VIII-3. Hydrography of REDW.

ranging between about 6.5 and 7.5, although pH measurements below 6 were reported for some of the tributary streams on some of the sampling occasions.

Water chemistry for Redwood Creek and Mill Creek watersheds were reported by Bradford and Iwatsubo (1978). Specific conductance values measured at all stations ranged from 30 to 300 μS/cm. Median pH values measured during each synoptic study at each station ranged from 6.1 to 7.9. In the Mill Creek drainage basin, alkalinity reported as $CaCO_3$ ranged from 6 to 35 mg/L with a mean of 17.5 mg/L. Specific conductance ranged from 26 to 88 μS/cm with a mean of 50.2 μS/cm. These data suggest that water chemistry is not particularly sensitive to potential adverse effects of acid deposition, although the streamwater chemistry is relatively dilute.

The purpose of the study by Bradford and Iwatsubo (1978) of the water quality in Redwood Creek and Mill Creek was to document existing water quality conditions and attempt to identify effects of logging on water quality in the mainstems of these two creeks and their tributary streams. pH was seasonally variable with a median of 6.8 during the rainy season, increasing to 7.4 during the dry season.

Configuration of the Redwood Creek channel has changed markedly since the 1950s. Most of the changes prior to about 1973 apparently occurred mainly upstream of Copper Creek. Observed changes included aggradation of the channel, erosion of the stream banks, and widening of the stream to occupy greater areas of the floodplain. Changes in hydraulic geometry were towards increased width, decreased depth, and increased velocity for a given water discharge. Increase in velocity likely reflected burial of coarse channel roughness elements by sedimentation from upstream erosion. In addition, the closed vegetation canopy was largely removed by bank erosion and landslides along the streamside. The primary causes of these changes included vegetation removal, alteration of hillside drainages, development of an extensive logging road network, and construction of thousands of kilometers of tractor log skidding trails. These changes also appear to have occurred, at least in part, in response to a sequence of major storms (Nolan and Marron 1995). Concern about the effects of such changes on aquatic and riparian resources of the park led to a number of studies to characterize the magnitude and cause of the changes (c.f., Iwatsubo et al. 1975, 1976; Janda et al. 1975; Nolan 1980; and Nolan and Marron 1995). Similar changes have also been observed in nearby watersheds (c.f., Kelsey 1980, Lisle 1981), although the most complete record of channel response in the region has been provided for Redwood Creek. Increase in bank to bank width in excess of 100% and channel fill in excess of 4.5 m have been observed subsequent to major storms in 1964 and 1972. Such storms can drastically change channel geometries because they trigger land slides throughout the unstable terrain of the watershed, and these landslides can introduce large volumes of sediment that may be sufficient to overload channel transport capacities for decades (Nolan and Marron 1995).

There is concern that water discharge has been altered as a consequence of land use activities. For example, Lee et al. (1975) constructed a rainfall-runoff model for Redwood Creek which showed a 20% increase in storm runoff from an intensively logged basin as compared with a basin where no intensive logging occurred. In addition, local residents have noted lower summer flows in aggraded reaches and extensively disturbed tributaries. Such trends towards higher peak flows and lower summer flows in logged basins have been documented in other areas (c.f., Hornbeck et al. 1970, Harris 1973).

Eighty-one percent of the virgin coniferous forest in the Redwood Creek watershed has been logged. Timber harvesting started during the 19^{th} century, but it was most intensive in the 15 years prior to a large storm event in December of 1964. This combination of intensive logging activities, followed by a large storm event is believed to be responsible for the much-

accelerated erosion that has been observed within the watershed and also contributed to increased streamwater temperature.

Higher streamwater temperature influences fish egg development, rearing success, species competition, and other factors important to salmonid fish (Beschta 1987). REDW, in cooperation with the U.S. Geological Survey, Biological Resources Division, initiated a pilot study in 1997 to characterize variation in summer water temperature regimes basin-wide in the Redwood Creek watershed. During the August, 1998 sampling, maximum stream temperature at the sample site locations ranged from 20.2ºC in the Redwood Creek estuary to 26.8 ºC at Redwood Valley. In addition, four tributary creeks were sampled, and maximum temperature in those ranged from 16.9 ºC in Prairie Creek to 21.8 ºC in Lacks Creek (Ozaki et al. 1999). Some of these temperatures are in the range of upper lethal temperatures for steelhead trout (23.9 ºC) and chinook salmon (26.2 ºC). Snorkel surveys conducted since 1981 in Redwood Creek have shown declining numbers of adult summer steelhead, and the entire population is small (< 40; Redwood National Park, Research Project Statement, February 1999).

Triska et al. (1995) quantified fluxes of inorganic N in Little Lost Man Creek in REDW between 1974 and 1982, and also in experimental channels during 1979. The work was conducted during low-flow periods between the months of May and November over a 1,500 m reach of the stream that flowed through an area that had been clear-cut in 1965. The period of study coincided with development of the riparian canopy which was dominated by alder, an N-fixing species. The study demonstrated that development of the alder riparian zone is of long-term importance in N cycling within the Little Lost Man Creek watershed. Little Lost Man Creek is a third-order pool and riffle stream, which is covered by old-growth coastal redwood forest on about 92% of the watershed. The forest includes associated Douglas fir and western hemlock. The other 8% of the watershed was clearcut at two sites, between 1962 and 1965, prior to incorporation into REDW. Following the clearcutting, alder has dominated the riparian vegetation. This is typical for coastal habitats in northern California and the Pacific Northwest.

A number of different factors that operate on a variety of time scales influence N uptake in aquatic and terrestrial systems, and therefore N transport in surface waters. Uptake by photosynthetic algae can produce significant fluctuation in N concentration in streamwater, especially during low-flow conditions. Fluctuations in light, temperature, and discharge introduce substantial variability within a stream reach (Triska et al. 1995). In addition, canopy development in the riparian zone has a major impact on solar input and therefore N uptake by autotrophs (Swanson et al. 1982).

The park includes a wide variety of aquatic habitats and wetlands, including headwater streams, large rivers, ocean shoreline, deep ocean waters, estuaries, ocean beaches and rocky intertidal zones. These habitats support a diversity of aquatic biota.

b. Aquatic Biota

The aquatic biota in the Redwood Creek drainage basin have been well studied. An assessment of benthic invertebrates, fish, periphyton, and phytoplankton was reported by Averett and Iwatsubo (1975, 1995) and by Iwatsubo and Averett (1981). When the study of the aquatic biota in Redwood Creek was begun in 1973, much of the land that was upstream from the present REDW was privately owned and contained virgin redwood forest, which was commercially clearcut. Logging in the upper watershed resulted in landslides, mass movement, fluvial erosion and deposition of sediments, increased water temperature, and changes in water quality (Averett and Iwatsubo 1995). By 1975 about 65% of the basin had been logged.

The benthic invertebrate community in Redwood Creek was diverse and consisted of 144 taxa from the mainstem and tributaries of the creek and also 30 taxa from the estuary (Averett

and Iwatsubo 1995). The dominant order was Diptera, which constituted 22% of the benthic invertebrates. This was followed by Ephemoroptera (17.7%), Coleoptera (13.2%), Tricoptera (11.3%), Plecoptera (4.3%), as well as lesser numbers of other orders. Variations in the number of benthic invertebrate species collected during the study reflected seasonal changes and these were also greatly influenced by stream discharge and movement of the stream bed subsequent to large storm events. Of the 1,066 fish that were captured during the study, 69.6% were steelhead trout. Next in abundance were Humboldt sucker (10.6%), three-spined stickleback (10.3%), Coast Range sculpin (5.8%), coho salmon (3.4%), and chinook salmon 0.3%). Other fish species that have been reported to inhabit Redwood Creek drainage include rainbow trout (*Salmo gairdneri*), coastal cutthroat trout, and eulachon (*Thaleichthys pacificus*) (DeWitt 1964).

Summer steelhead have been monitored in Redwood Creek by snorkelers since the summer of 1981 (Anderson 1993). About 25 populations of summer steelhead are known in California and the species is classified as a Sensitive Species by the U.S. Forest Service and as a Species of Special Concern by the California Department of Fish and Game. Summer steelhead are vulnerable to a number of potential water quality problems because they hold in freshwater for about 8 to 10 months. Adult fish migrate upriver during spring, hold in pools during summer, and spawn during winter. They are therefore sensitive to high water temperatures, low summer flows, and other habitat related problems.

The numbers of fish observed by Anderson (1993) peaked at 44 during 1984 and 1985, but have since declined. Summer steelhead in Redwood Creek face a number of problems, including habitat degradation, poor water quality, sport fishing and poaching, and small population size. The combined effects of timber harvest and large storms have deposited large amounts of sediment in the creek and degraded fish habitat. Both erosion and hillslope mass wasting caused sedimentation of the main stem that filled the deep pools, which are the preferred habitat of summer steelhead. Major flood events then caused significant channel adjustments including channel widening, aggradation, and bank erosion. The resultant widened stream bed and accompanying shallow riffles provided little or no cover for the fish (Anderson 1993). Temperatures in the mainstem during summer were well above the preferred temperature range for steelhead which is 7.3 to 14.6^o C (Reiser and Bjornn 1979). Schools of fish were rare and were observed only in pools that were adjacent to tributaries where cooler water entered Redwood Creek.

Cold pools provide extremely important summer habitat for anadromous fish in Redwood Creek and similar creeks within the park. These pools maintain water temperature that is often several degrees C cooler than the main stream temperatures. Cold pools form where cooler groundwater or mainstream water cooled by intergravel flow seeps from the channel and does not rapidly mix with the warmer mainstream water (Keller et al. 1995). The occurrence and morphology of these pools is controlled by a variety of factors, including geologic variability, and also the availability of large organic debris within the stream channel.

Park staff conduct long-term monitoring of salmonid winter spawning and salmonid carcass surveys in the Redwood Creek watershed (Anderson 1999). The surveys determine presence or absence of adult fish and redds in index streams, timing of runs, distribution of spawning salmonids within the streams, size and sex ratios of returning spawners, and redd characteristics. Park staff also monitor water temperature and fish utilization in the Redwood Creek Estuary.

Seasonal analysis of species diversity and functional group organization of aquatic invertebrates in two coastal streams within REDW was conduced by Strange (1989). Benthic invertebrate samples were collected at five-week intervals from Prairie Creek and Lost Man Creek for a period of one year from March, 1986 through March, 1987. A total of 265, 955

aquatic invertebrates were collected, representing 125 taxa. Aquatic insects comprised 89% of the total number and the remainder included crustaceans, molluscs, and annelids.

Redwood Creek Estuary provides critical habitat for chinook salmon and steelhead trout, as well as a variety of estuarine species, including starey flounders (*Platichtys stellatus*) and staghorn sculpin (*Leptocottus armatus*). Construction of the flood control project during the mid-1960s along the lower 5.1 km of Redwood Creek severely altered the hydrology and impaired the physical and biological functioning of the estuary (Hofstra and Sacklin 1987). Whole sections of the creek were channelized and flood control levies were constructed subsequent to damaging floods that had occurred in the 1950s and again in 1964. Restructuring of the channel caused loss of riparian vegetation, accumulation of sediment in areas which were previously viable fish rearing habitat, and isolation of productive side channel areas behind the constructed levies. Estuary restoration efforts have been ongoing in recent years (Hofstra and Sacklin 1987).

Amphibians are especially sensitive to environmental pollutants and UV radiation because of their permeable skin (Stebbins and Cohen 1995), and are therefore useful as bioindicators of ecosystem health. Studies of the distribution of Plethodontid salamanders in the redwood region of northern California were reported by Bury and Martin (1973). Logging of redwood forests appeared to alter species composition of these salamanders, which included the arboreal salamander (*Aneides lugubris*), clouded salamander (*A. ferreus*), ensatina (*Ensatina eschscholtzii*), and California slender salamander (*Batrachoseps attenuatus*).

Bury (1983) reported differences in amphibian populations in logged vs. old-growth redwood forests. Amphibian surveys were conducted on eight 0.125 ha plots in or near REDW. Amphibian populations on the old-growth sites had more individuals and greater biomass as well as different species composition as compared with the logged plots. For example, the Olympic salamander occurred only in a rivulet through one of the old-growth sites, whereas the Pacific treefrog (*Pseudacris regilla*) was found only in a logged plot. Pacific giant salamanders (*Dicamptodon tenebrosus*) occurred on half of the old-growth sites, but none were found in logged areas. Ensatina and California slender salamander were found principally in old-growth stands. Also, the clouded salamander was more prevalent in second-growth than in old-growth forests (Bury 1983).

Selected forest stands were sampled within REDW for herpetofauna relative abundance in 1997 and 1998. The goal of the surveys was to develop a baseline inventory of the amphibian and reptile species found within the park, estimate relative abundance of species and selected habitats, and determine differences in species composition and relative abundance within old-growth and second-growth redwood stands (Falvey 1998). In 1998, the surveys were conducted in old-growth redwood/conifer habitat, and in second-growth stands that were stratified by cutting history into greater than 40 years and greater than 30 years post logging. A total of seven amphibian species were found during the 1998 surveys. Old-growth stands showed greater numbers of individuals recorded per sampling effort than did the second growth stands. At five old-growth sample sites, there was an average of 36 reptiles and amphibians observed per site, whereas at 10 second-growth sites, there was an average of 13 individuals observed per site. There was no significant difference between the second growth sites based on their cutting age. Species observed are listed in Table VIII-10, the most common of which were California slender salamander and ensatina (Salvey 1998).

Welsh and Ollivier (1998) documented lower densities of amphibians in streams that had been impacted by sedimentation in response to road construction of the highway bypass. Construction activities resulted in a large accidental infusion of fine sediments into streams in Prairie Creek State Park during an October 1989 storm event. Amphibian species studied were

Table VIII-10. Amphibian species observed in REDW during 1998 surveys of old growth and second growth redwood stands (Source: Falvey 1998).	
Common Name	Species Name
California slender salamander	*Batrachoseps attenuatis*
Ensatina	*Ensatina eschscholtzii*
Del Norte salamander	*Plethodon elongatus*
Pacific giant salamander	*Dicamptodon tenebrosus*
Clouded salamander	*Aneides ferreus*
Black salamander	*Aneides flavipunctatus*
Roughskin newt	*Taricha granulosa*

tailed frogs (*Ascaphus truei*), Pacific giant salamanders, and southern torrent salamanders (*Rhyacotriton variegatus*). Although sedimentation effects were species-specific, reflecting differential use of stream microhabitats by the three species investigated, the vulnerability of all of these species to infusions of fine sediments was probably the result of their common reliance on interstitial spaces in the streambed matrix for critical life requisites, such as cover and foraging (Welsh and Ollivier 1998).

Aquatic biota of REDW can be assumed to be highly sensitive to the potential impacts of surface water acidification from acidic deposition. Anadromous salmonid fish are especially sensitive to low pH and slightly elevated concentrations of Al (c.f., Baker et al. 1990) and constitute an important resource in REDW. The park also contains a rich diversity of amphibians, many of which are likely highly sensitive to acidification. However, the stream water chemistry measured to date in the park suggests that park waters have moderate buffering capacity and would not be subject to acidification unless the deposition of S or N increased quite dramatically. In addition, the location of the park relative to pollution sources and prevailing winds makes future acidification highly unlikely.

3. Vegetation

There have been minimal efforts to identify the potential effects of air pollution on vascular plants in REDW. There have been several studies of lichens, which were motivated by a proposed nickel mine on Gasquet Mountain approximately 15 km north of the park. Gough et al. (1987, 1988) collected samples of *Hypogymnia entermorpha* and *Usnea* sp. from 29 trees in two areas of the Little Bald Hills and analyzed them for a wide range of elements. They determined that the biogeochemisty of these species appeared to reflect the geochemistry of the ultramafic terrain over which they were growing. The samples were collected primarily as a baseline to which future samples could be compared.

An additional study by del Moral et al. (1984) documented the lichen flora on each of 100 Douglas-fir and Jeffrey pine trees in the Little Bald Hills. They identified 27 lichen species and determined that lichen vigor was excellent. In 1986, a follow-up study by Denison (1987) again found the condition of lichens on these trees to be excellent. An additional study by Denison and Sillett (1989) surveyed epiphytic lichens on Oregon white oak, and again found a diverse lichen flora with no evidence of pollutant injury. Finally, Sillett et al. (1989) surveyed canopy

epiphytes of an old-growth Douglas-fir and an old-growth coast redwood, and found a diverse lichen flora with no evidence of pollutant injury.

The sensitivities of plant species at REDW to air pollutants are summarized in Table VIII-11.

Table VIII-11. Plant and lichen species of REDW with known sensitivities to sulfur dioxide, ozone, and nitrogen oxides (H=high, M=medium, L=low, blank=unknown).

Scientific Name	Common Name	Sensitivity		
		SO_2	O_3	NO_x
Gymnosperms				
Juniperus communis	Common juniper	L	L	
Picea sitchensis	Sitka spruce	M		
Pinus contorta	Lodgepole pine	H	M	H
Pinus jeffreyi	Jeffrey pine	H	H	H
Pseudotsuga menziesii	Douglas fir	H	M	H
Taxus brevifolia	Pacific yew	L		
Thuja plicata	Western redcedar	L		
Tsuga heterophylla	Western hemlock	M	M	
Angiosperms				
Acer circinatum	Vine maple		L	
Acer macrophyllum	Bigleaf maple		L	
Achillea millefolium	Common yarrow		L	
Apocynum androsaemifolium	Spreading dogbane		M	
Arctostaphylos uva-ursi	Kinnikinnick	L	L	
Artemisia douglasiana	Douglas' sagewort		H	
Berberis nervosa	Cascade Oregongrape		L	
Bromus carinatus	California brome		L	
Ceanothus velutinus	Snowbrush ceanothus	L		
Cirsium arvense	Canadian thistle		L	
Conium maculatum	Poison hemlock		L	
Convolvulus arvensis	Field bindweed	H		
Corylus cornuta var. *californica*	California hazelnut	H		
Crataegus douglasii	Black hawthorn	L		
Elymus glaucus	Blue wildrye		H	
Epilobium angustifolium	Fireweed		L	
Epilobium brachycarpum	Autumn willowweed		L	
Erodium cicutarium	Redstem stork's bill	M	M	
Festuca idahoensis	Idaho fescue	H		
Gaultheria shallon	Salal		M	
Holodiscus discolor	Oceanspray	H	M	
Lolium perenne	Perennial ryegrass		M	
Lonicera involucrata	Twinberry honeysuckle	L		M
Lupinus latifolius	Broadleaf lupine		M	
Mimulus guttatus	Seep monkeyflower		L	
Osmorhiza chilensis	Sweetcicely		M	
Osmorhiza occidentalis	Western sweetroot		L	

Table VIII-11. Continued.

Scientific Name	Common Name	Sensitivity		
		SO_2	O_3	NO_x
Phacelia heterophylla	Varileaf phacelia		L	
Physocarpus capitatus	Pacific ninebark		M	
Poa annua	Annual bluegrass	H	L	
Poa pratensis	Kentucky bluegrass		L	
Populus trichocarpa	Black cottonwood	M	H	
Potentilla glandulosa	Gland cinquefoil		M	
Prunus emarginata	Bitter cherry	M		
Prunus virginiana	Common chokecherry	M	M	
Quercus kelloggii	California black oak		M	
Ribes bracteosum	Stink currant		H	
Robinia pseudoacacia	Black locust	H	L	
Rubus parviflorus	Thimbleberry		M	
Rubus spectabilis	Salmonberry	M	M	
Rumex crispus	Curly dock		L	
Taraxacum officinale	Common dandelion		L	
Trifolium pratense	Red clover	L		
Trifolium repens	White clover		H	
Vaccinium parvifolium	Red huckleberry		M	
Vicia americana	American vetch		L	
Viola adunca	Hookedspur violet		L	
Lichens				
Alectoria sarmentosa			H	
Bryoria abbreviata			H	
Bryoria fremontii			H	
Bryoria glabra		M	H	
Cladonia chlorophaea		M		
Cladonia coniocraea		M		
Cladonia fimbriata		H		
Evernia prunastri		M		
Hypogymnia enteromorpha		M	H	
Hypogymnia imshaugii		M	M	
Hypogymnia physodes		M	M	
Hypogymnia tubulosa		H		
Letharia vulpina		L		
Lobaria pulmonaria		H	L	
Parmelia saxatilis		M	M	
Parmelia sulcata		M	L	
Platismatia glauca		M	H	
Usnea hirta		H	H	
Usnea longissima		H	H	
Usnea subfloridana		H	H	

More than 25% of the area within the park that was prairie and oak woodland in 1850 is now coniferous forest. Widespread invasion of Douglas fir into the oak woodlands and prairies is considered to be a major problem, not only in the park but also elsewhere in California. An estimated 336 ha of prairies and oak woodlands have converted to Douglas fir-dominated forests during the last 130 to 140 years (Redwood National Park 1992). The Bald Hills Vegetation Management Plan identified three primary vegetation management problems in the Bald Hills area: (1) encroachment of Douglas fir, (2) replacement of native herbaceous species with non-native species, and (3) declining wildlife habitat in response to the loss of oak woodland and prairie vegetation.

The Bald Hills vegetation management plan (Redwood National Park 1992) also established protocols for the Bald Hills area to maintain the diversity of plants and animals that prevailed when the area was first visited by Europeans. Management strategies were proposed to restore, mimic, and perpetuate natural processes that will maintain vigorous prairies and oak woodlands. The plan calls for a combination of resource protection, restoring fire through prescribed burning, manually removing Douglas fir, and reseeding and replanting with native species.

4. Visibility

As part of the Interagency Monitoring of Protected Visual Environments (IMPROVE) network, visual air quality in REDW has been monitored using an aerosol sampler and camera. The aerosol sampler began operation in March 1988. Located near the Pacific coast, north of the Klamath River, the system is still operational today. The automatic 35mm camera operated from June 1987 through March 1995. Located one-half mile south of the aerosol sampler, the camera photographed the coastline of REDW to the south. Data from this IMPROVE site have been summarized to characterize the full range of visibility conditions for the March 1988 through February 1999 period, based on seasonal periods (Spring: March, April, and May; Summer: June, July, and August; Autumn: September, October, and November; and Winter: December, January, and February) and annual periods (March through February of the following year, e.g., the annual period of 1998 includes March 1998 through February 1999). Complete descriptions of visibility characterization, mechanisms of sources and visibility impacts, and IMPROVE monitoring techniques and rationale are provided in Chapter I.

a. Aerosol Sampler Data - Particle Monitoring

A tabular and graphic summary of average reconstructed extinction values by season and year for the March 1988 through February 1999 period are provided in Table VIII-12 and Figure VIII-4, respectively.

Reconstructed extinction budgets generated from aerosol sampler data apportion the extinction at REDW to specific aerosol species (Figure VIII-5). The species shown are Rayleigh, sulfate, nitrate, organics, elemental (light absorbing) carbon, and coarse mass. The sum of these species account for the majority of non-weather related extinctions. Extinction budgets are listed by season and by mean of cleanest 20%, mean of the median 20%, and mean of the dirtiest 20%. The "cleanest" and "dirtiest" signify lowest fine mass concentrations and highest fine mass concentrations respectively, with "median" representing the 20% of days with fine mass concentrations in the middle of the distribution. Each budget includes the corresponding extinction coefficient, standard visual range (km), and deciview (dv).

The segment at the bottom of each stacked bar in Figure VIII-5 represents Rayleigh scattering, which is assumed to be a constant 10 Mm^{-1} at all sites during all seasons. Rayleigh scattering is the natural scattering of light by atmospheric gases. Higher fractions of extinction due to Rayleigh scattering indicate cleaner conditions.

Table VIII-12. Seasonal and annual average reconstructed extinction(b_{ext}; Mm^{-1}), REDW, March 1988 through February 1999.

Year	Spring (Mar, Apr, May)	Summer (Jun, Jul, Aug)	Autumn (Sep, Oct, Nov)	Winter (Dec, Jan, Feb)	Annual (Feb-Mar)[a]
1988	57.0	56.2	52.6	35.8	57.3
1989	52.1	63.3	63.0	29.2	59.2
1990	66.9	59.5	63.2	41.4	64.2
1991	57.9	50.3	64.8	37.9	58.5
1992	73.3	52.1	56.6	28.8	58.6
1993	43.3	47.2	63.3	30.8	51.0
1994	50.5	48.3	52.7	32.4	51.5
1995	50.9	47.1	63.5	29.3	54.9
1996	47.5	56.4	45.3	24.6	51.6
1997	53.6	52.3	55.0	22.2	51.6
1998	44.4	47.5	42.7	27.4	45.7
Mean[b]	54.3	52.7	56.6	30.9	54.9[c]

[a] Annual period data represent the mean of all data for each March through February annual period.
[b] Combined season data represent the mean of all seasonal means for each season of the March 1988 through February 1999 period.
[c] Combined annual period data represent the mean of all combined season means.

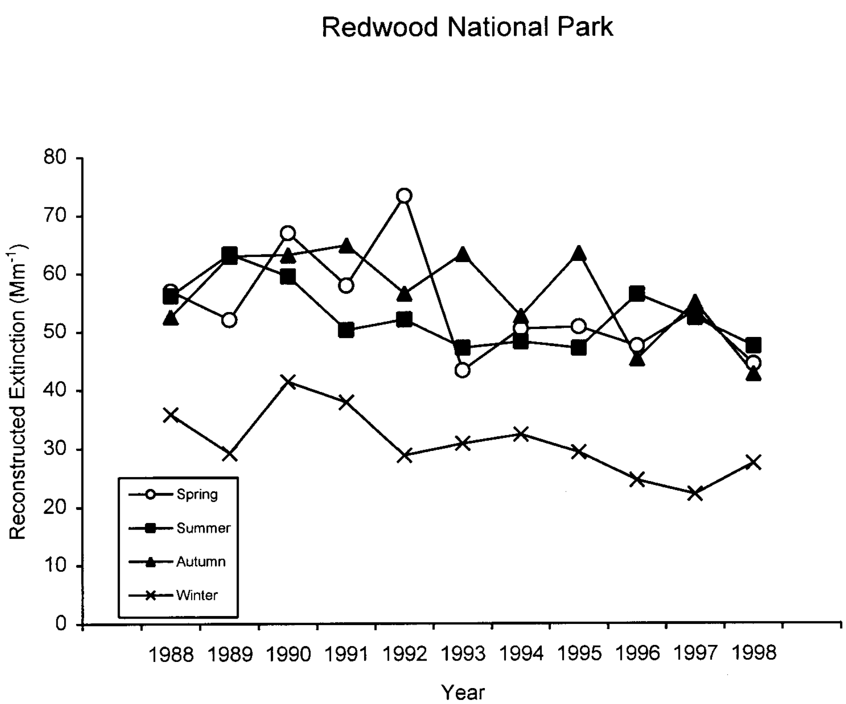
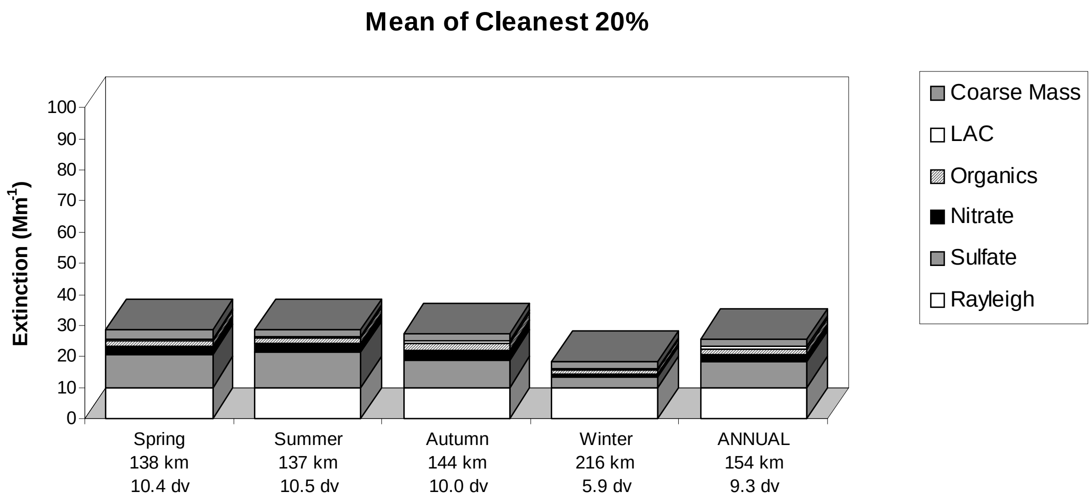
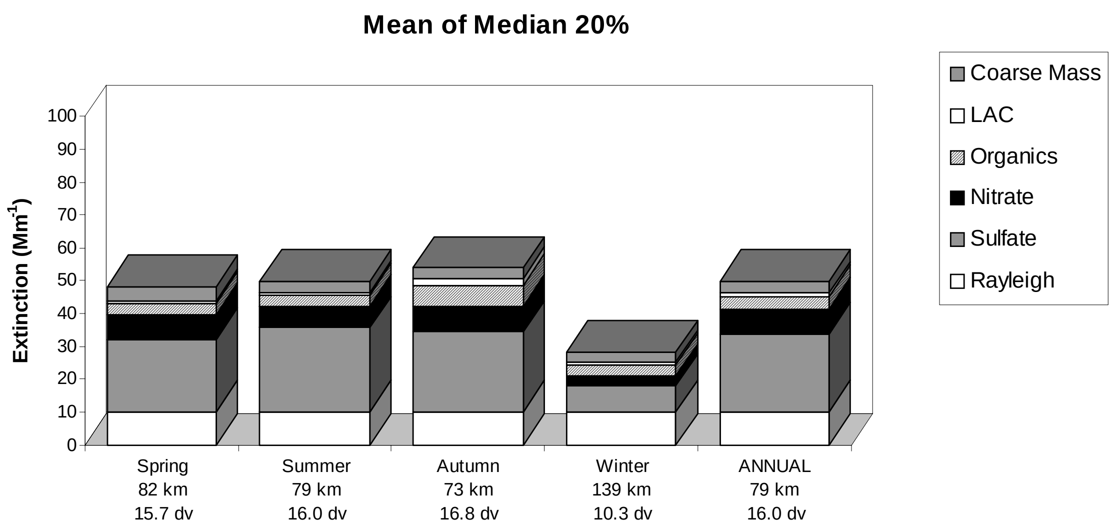


Figure VIII-4. Seasonal average reconstructed extinction (Mm^{-1}) in REDW, March 1988 through February 1999.





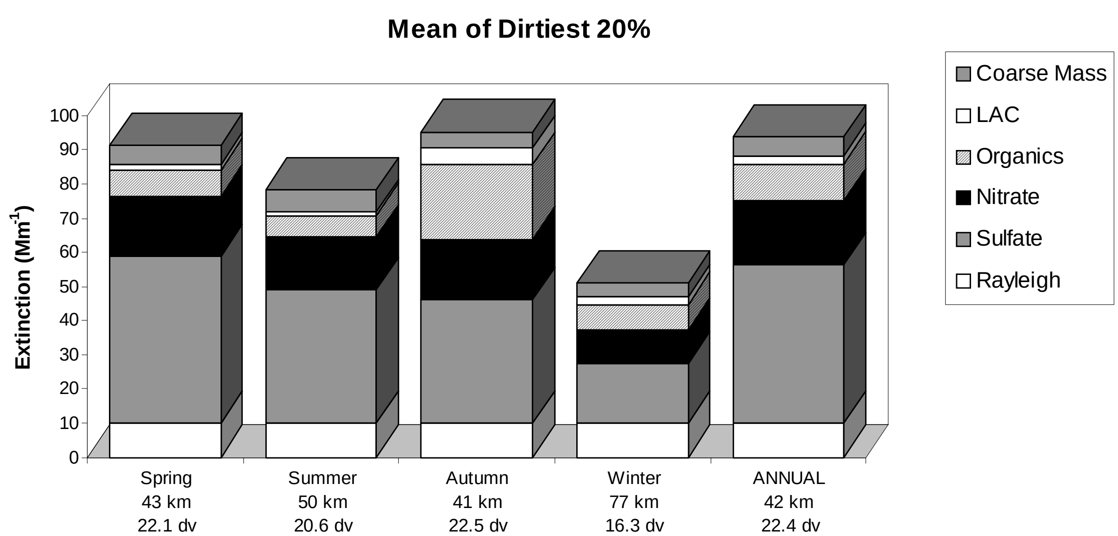


Figure VIII-5. Reconstructed extinction budgets for REDW, March 1988-February 1999.

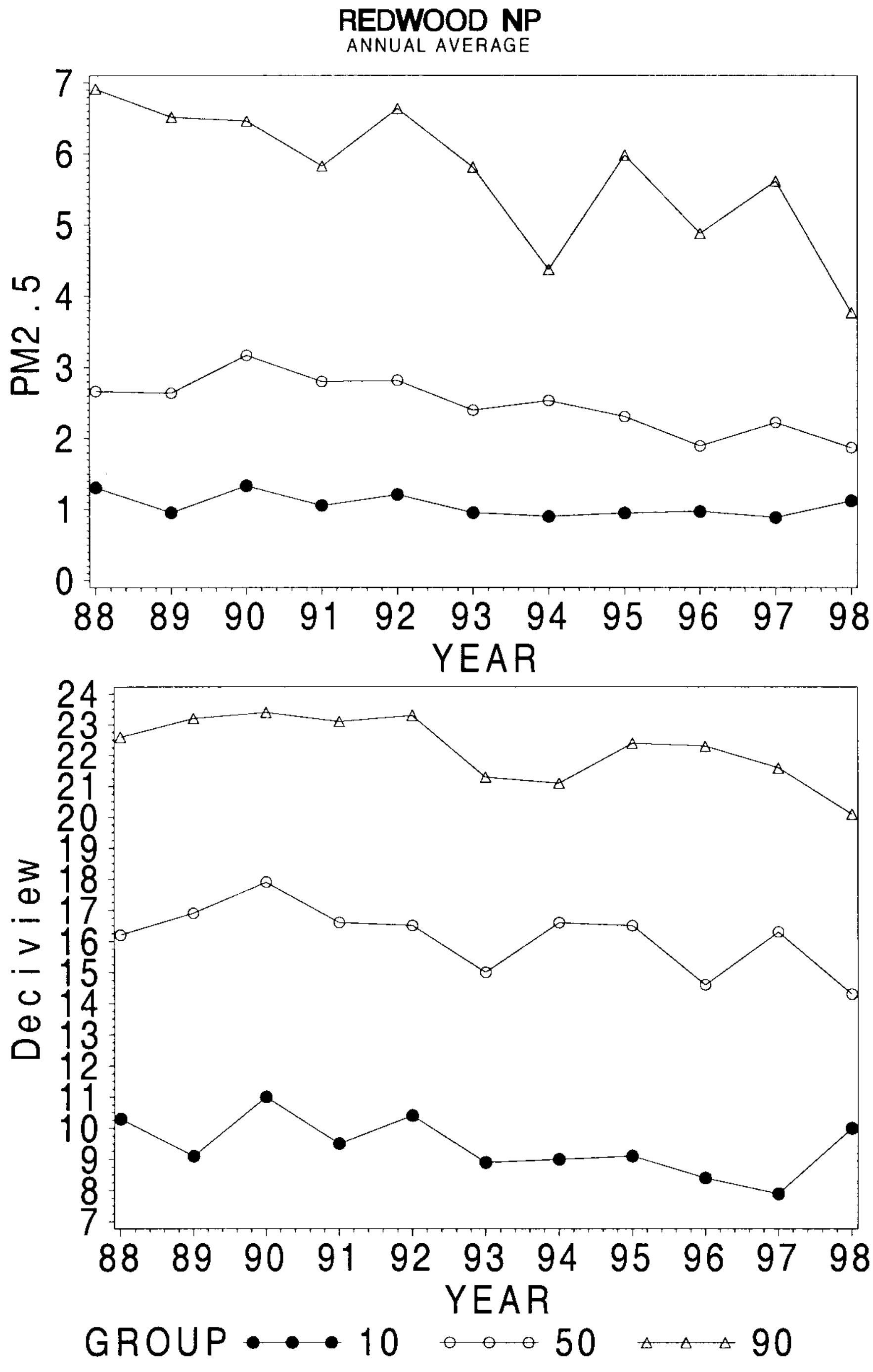


Figure VIII-7. Trends in annual averages for $PM_{2.5}$ ($\mu g/m^3$) and visibility (deciview) for REDW.

3. Aquatic Systems

There are no research or monitoring needs with respect to potential effects of air pollution on aquatic resources in REDW. Aquatic resources within the park are not believed to be highly sensitive to acidification; acidic deposition is low, and it is not expected to increase dramatically in the future.

4. Terrestrial Systems

Given the northern coastal location of REDW and current low ambient ozone levels, oxidant injury in vegetation is unlikely, and a long-term monitoring program for vegetation is probably not warranted at this time. However, ascertaining current foliar condition could be valuable for establishing a reference point in time, even if it only confirms that no injury is present. This could be done rather quickly and at low cost. There is often uncertainty associated with the evaluation of foliar injury. If ozone concentrations increase dramatically in the park in the future, it will be useful to have baseline foliar condition data under the currently low ozone exposure regime of the park. The existence of such data may later help to determine the likelihood that park vegetation has actually been damaged by increased ozone concentration, especially if such future damage is only of moderate severity.

If NPS feels that a reference survey is needed, it is recommended that monitoring for potential ozone injury in Jeffrey pine be initiated in the Little Bald Hills area. Monitoring protocols should be the same as those used in the FOREST study (Arbaugh et al. 1998), including the use of Ozone Injury Index (Duriscoe et al. 1996, Schilling and Duriscoe 1996) to quantify injury (see Appendix). If ozone injury is detected in initial surveys, evaluations should be done on an annual basis in the late summer. If no injury is detected, additional surveys are probably not merited unless ambient ozone levels increase significantly.

5. Visibility

Except for continuing IMPROVE monitoring, no additional visibility monitoring or research is recommended at this time.

IX. SEQUOIA/KINGS CANYON NATIONAL PARKS

A. GENERAL DESCRIPTION

Sequoia National Park, America's second national park, encompasses over 1,500 km^2 and includes not only the world famous giant sequoias but also part of the largest uncut mixed conifer forest remaining in the Sierra Nevada. Below the conifer belt are important tracts of California chaparral and threatened oak woodland. Above lies an extensive alpine zone culminating in Mt. Whitney, the highest peak in the lower 48 states. In the 3,960 m of vertical relief of the park (Figure IX-1) are pristine examples of every environment from Mediterranean chaparral to arctic tundra. Over 1,300 plant species occupy the park together with a wide range of animals, including species of special concern such as peregrine falcon and Sierra bighorn sheep. There are about a million visitors to the park each year.

Kings Canyon National Park encompasses the most rugged portion of the Sierra Nevada. Numerous glacial canyons, including the deepest in the country, thread their way between ice-sculpted summits (Figure IX-1). The Kings River flows through the largest major watershed that remains undammed in the southwestern United States. The world's largest grove of giant sequoias and California's most extensive cave system are protected in the Grant Grove portion of the park. Established in 1890 as General Grant National Park, this area became part of the newly created Kings Canyon National Park in 1940. Ecological significance of Kings Canyon National Park lies in its extensive alpine areas, its incredible cave systems, and the sequoia/mixed conifer forest. Six hundred thousand people visit Kings Canyon National Park each year.

Together, Sequoia and Kings Canyon National Parks (SEKI) encompass about 350,000 ha of contiguous parkland, and are managed as one unit by the NPS. They form an international biosphere reserve and are bounded on three sides by national forest wilderness areas. Since 1982, SEKI has been the site of broad interdisciplinary research on the effects of acidic deposition on park ecosystems. Using a watershed approach, this program developed a cooperative effort that included the California Air Resources Board (CARB) and helped to provide the infrastructure necessary to support interdisciplinary ecosystem studies. These studies have included meteorology, precipitation chemistry, snow deposition and hydrology, dry deposition, stream hydrology and chemistry, aquatic and terrestrial productivity, soil characterization and solution chemistry, elemental input-output budgets, and mathematical modeling of major ecosystem processes (c.f., California Air Resources Board 1989, Engle and Melack 1997). The results of many of these studies are applicable to other Class I parks (and also wilderness areas) in the state, including YOSE and LAVO.

SEKI is significant for a variety of reasons, including the following. These parks contain:

- the largest giant sequoia trees and groves in the world, including the world's largest tree;
- an extraordinary continuum of ecosystems arrayed along the greatest vertical relief (418 to 4,418 m elevation) of any protected area in the lower 48 states;
- the highest, most rugged portion of the High Sierra which is part of the largest continuous alpine environment in the lower 48 states;
- magnificent, deep, glacially-carved canyons including Kings Canyon, Tehipite Valley, and Kern Canyon;
- the core of the largest area of contiguous designated wilderness in California, and the second largest in the lower 48 states;
- the largest preserved southern Sierran foothills ecosystem;
- about 180 known marble caverns, many inhabited by endemic cave fauna; and
- a wide spectrum of prehistoric and historic sites.

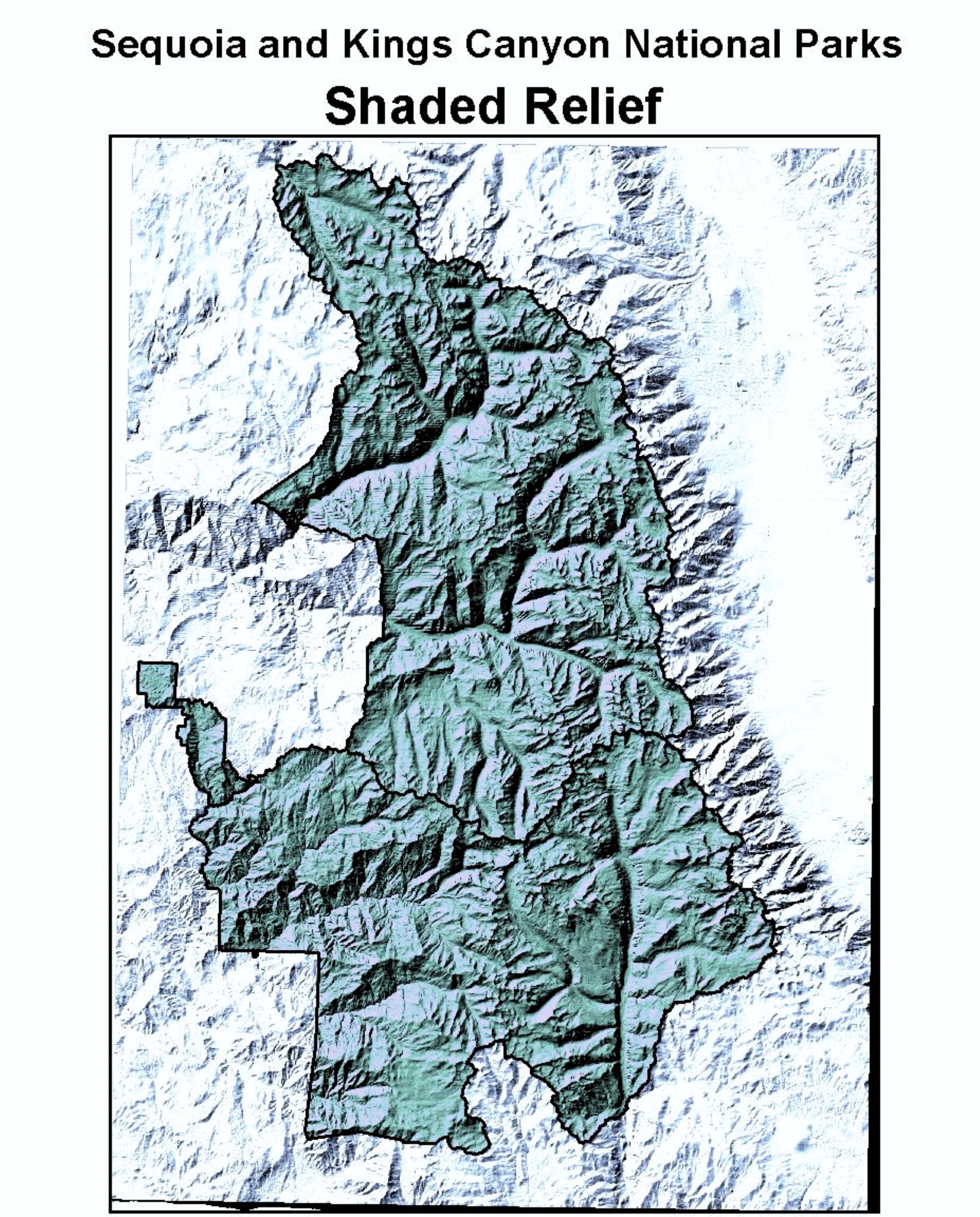


Figure IX-1. Shaded relief map of SEKI.

SEKI was established as national parks because of the unique values of their natural resources, especially their wilderness character and vegetation, with emphasis on giant sequoia forests. They were also established as public parks for the enjoyment and benefit of the people. High mountain areas contain hundreds of lakes in basins carved out of granite by ancient glaciers. Thousands of kilometers of mountain streams are found in the parks, gathering into major forks of the Kaweah, Kern, Kings, and San Joaquin Rivers. Vegetation is highly diverse, beginning in the lower elevation areas and foothill slopes as open oak savannah and chaparral brush fields and progressing upward through climatically-influenced bands of different forest types, including ponderosa pine forest and mixed conifer forests. The latter include giant sequoia groves, fir forests, extensive lodgepole pine forests, and high-elevation foxtail pine. The parks provide habitat for a variety of fish and wildlife species. Existence of some species is potentially threatened, including the California bighorn sheep and the little kern golden trout.

SEKI also has a significant cave resource. There are about 180 known caves within the parks, and the potential for several times that many. SEKI contains the longest cave (23 km) in California and also contains about a quarter of the solution caves in California. The parks therefore attract cave explorers from all over the country. Some of the caves in SEKI contain extremely rare fauna.

There are a number of air quality and air pollution concerns for SEKI. They include the following:

- the San Joaquin Valley, which lies upwind of SEKI, is one of the most polluted areas in California and is a primary source of air pollutants transported to these parks; projected population growth may result in increasing emissions in the future;
- ozone levels frequently exceed health-based state and occasionally federal air quality standards;
- ozone has caused injury to yellow pines, black oaks, and giant sequoia seedlings;
- late summer precipitation and spring snowmelt are moderately acidic and associated impacts have been suggested or documented;
- visibility in these parks is often obscured, most frequently during summer.

A planning effort was initiated for SEKI in the summer of 1997. When it concludes with publication of the new General Management Plan, the parks will have defined a broad management framework that will guide operations of these two parks for the next 15 to 20 years.

1. Geology and Soils

As the mountain block rose in the Sierra Nevada, erosion intensified and eventually exposed igneous rocks of the batholith. It has been estimated that a layer of material more than 11 km thick was eroded from the top of the Sierra Nevada (Harris and Tuttle 1983). This uncovering of the batholith also exposed roof pendants of metamorphous marine sedimentary rock. These were downward projections into the batholith. Some of the granites in SEKI are massive and have few joints or fractures; therefore exfoliation processes have shaped many of the exposed bedrock surfaces and there are a number of exfoliation domes similar to those in YOSE. These include Moro Rock, Alla Peak, Beetle Rock, and Sunset Rock (Harris and Tuttle 1983).

Fissures were more extensive in Kings Canyon National Park than they were in Sequoia, especially towards the northern end. Moraines of various ages mark the glacial recessions of the Pleistocene. Hanging glaciers occupied the canyons of the Kings River and its tributaries in the upper reaches of the San Joaquin River Basin in the northern part of SEKI. Consequent glacial erosion was severe.

Glacial features and exfoliation domes are conspicuous in Kings Canyon National Park. Exfoliation domes include Tehipite and Kettle Domes, Obelisk, Centinel Dome, and North

Dome. All are formed in massive granite. Kings Canyon glacier was the largest of the valley glaciers at about 71 km long; it reached down to a relatively low elevation of 765 m on the western Sierra slope (Harris and Tuttle 1983). The U-shaped Tokapah Valley is a good example of a canyon cut by glacial action.

Mesozoic granitic rocks of the Sierra Nevada Batholith underlie 95% of the mapped area, including 14 discrete granitic intrusions. Nine of the igneous plutons are granodiorites that contain minor to conspicuous amounts of hornblende, three are hornblende-free lucogranites, one is biotite granite and granodiorite, and one is mafic diorite.

Parkwide soils data are generally lacking. However, a soil survey was conducted of the central portion of Sequoia National Park during the period 1982-1984 by the University of California, Department of Land, Air, and Water Resources, Davis, working in cooperation with the NPS. The resulting soil map data were depicted on an orthophotograph prepared from 1:80,000-scale aerial photographs in 1985. Soils data available in the park GIS database are depicted in Figure IX-2.

Soil properties were evaluated by Dahlgren et al. (1997) at seven sites along an elevational transect in the central Sierra Nevada, from near Millerton Lake (elevation 198 m) to Kaiser Pass (elevation 2,865 m). The objective of the study was to document soil properties and pedogenic processes that are influenced by climatic factors, particularly precipitation and temperature. All of the soils that were sampled had formed in residuum weathered from granitic bedrock (non-glaciated) and occurred on sites with southerly aspect and slope ranging from 9 to 15%. Mean annual air temperature decreased from 16.7 to 3.9^{o}C and mean annual precipitation increased from 33 to 127 cm along the transect. Soil pH decreased by about two units and soil base saturation decreased from about 90% to 10% with increasing elevation, likely in response to climatic and vegetational changes and other site factors (Figure IX-3). Some soil properties, including pH, soil color, clay and secondary Fe oxide concentrations, showed a pronounced threshold change over a short distance at about 1,600 m elevation (c.f., pH in Figure IX-3). This change coincided with the present elevation of the average effective winter snow line (Dahlgren et al. 1997).

2. Climate

Climate is highly variable throughout the parks and is strongly influenced by elevation and orographic effects. Distinct elevational vegetation zonation is correlated with precipitation and temperature. In the chaparral community, at relatively low elevation, climatic conditions include approximately 64-89 cm of annual precipitation, whereas at the higher elevations, precipitation typically exceeds 100 cm/yr. Summers in the park are generally dry at all elevations. Ninety five percent of annual precipitation falls from October through May. In the Kaweah watershed at 500 m, abut 99% of annual precipitation falls as rain, and moving up in elevation to 2,000 m, 50% of annual precipitation falls in the form of snow. Above 3,000 m, 80 to 100% of annual precipitation is in the form of snow. Mean July and January temperatures decrease by 5.6 and 4.7^{o}C, respectively, for each km increase in elevation in both the Kaweah and Kern watersheds (Stephenson 1988).

Studies of snow hydrology and chemistry were initiated in the Emerald Lake watershed in 1984 as part of the CARB's Integrated Watershed Study (IWS). Four sites were instrumented in the watershed, and meteorological parameters were continuously monitored during the 1986 water year. Snowfall volume and chemistry were sampled on an event basis from snowboards, and at intervals from snow pits at several locations. Several detailed snow surveys were done to estimate the volume and distribution of snow throughout the watershed during the spring. An

energy balance snowmelt model was developed and tested. Calculated sublimation and evaporation accounted for the loss of 20 to 25% of the snowfall (Dozier et al. 1987).

Dozier et al. (1989) continued measurements of snow water equivalents (SWE) by taking hundreds of depth measurements and depth profiles at six locations during the 1986, 1987, and 1988 water years. Snow dominated the calculated water balance, and accounted for 95% of the precipitation and subsequent streamflows. Evaporation from the snowpack was the principal route of water loss to the atmosphere, accounting for about 80% of the total evaporation (which constituted about 22% of the total precipitation input).

Zones within the watershed were mapped, based on similar snow properties and topographic parameters that accounted for variations in both accumulation and ablation by Elder et al. (1991). Elevation, slope, and radiation values estimated from a digital elevation model were used to determine the zones. The authors found that partitioning the watershed on the basis of topographic and radiation variables improved the results over a simple random sample (Elder et al. 1991).

A detailed evaluation of climatic conditions of the Emerald Lake watershed was conducted by Marks et al. (1992). Measurements of snowfall, meteorological, and snow cover conditions were used to characterize the climate at several locations within the watershed during that snow season. The data were then integrated into continuous hourly time series estimates of solar and thermal radiation; air, soil, and snow temperature; humidity; and wind at two representative sites in the watershed for the entire snow season. The two sites were selected to represent topographic extremes. One was a ridge site at an exposed location 10 m above Emerald Lake. The second site was chosen to represent lakeshore conditions. The resulting micrometeorological data are some of the most detailed of any alpine watershed in North America (Marks et al. 1992). Snow deposition was the most difficult parameter to monitor in the watershed because major snow deposition events severely increased avalanche danger and restricted access to the watershed for extended periods. Although 17 deposition events were monitored during the snow season, accumulating over 2.6 m of SWE, 70% of the deposition was accounted for by just three snowfall events. Over 40% occurred during a single snowstorm event when over 1 m of SWE was deposited.

3. Biota

Because of the pronounced gradients in elevation, temperature, and precipitation found within SEKI, vegetation communities are diverse and occur in distinct elevation zones, ranging from chaparral to alpine communities. The distribution and ecology of vegetation in SEKI have been relatively well studied. According to Vankat (1982), there are 15 discrete vegetation types in the park, with their boundaries described along elevational and topographic-moisture gradients. These vegetation types range between 400 and 3,600 meters, and include lower elevation chaparral and woodlands, mid to high elevation forests, and high elevation meadows and alpine vegetation (Figure IX-4).

Lowland live oak woodland and blue oak woodland are found at the lowest elevations in the park, with the former dominated by interior live oak (*Quercus wislizenii*) and California buckeye (*Aesculus californica*) on moist sites, and the latter dominated by blue oak (*Quercus douglasii*) on drier sites. Within Sequoia National Park, there are about 16,000 ha of chaparral, ranging in elevation from about 300 to 1,525 m. The chaparral is found mainly on the steep slopes of the southwestern portion of the park. The chaparral in Sequoia National Park and PINN may be the last significant protected remnants of California chaparral. The predominant chaparral species in the park is chamise (*Adenostoma fasciculatum*), which is the most widely distributed of the chaparral species; it dominates the steep, hot, dry, south and west facing slopes at lower

elevations, just above the oak woodland types on sites with intermediate soil moisture. In some places it forms very dense stands of nearly impenetrable brush, with species cover approaching 100%. Herbaceous understory is sparse. Occasional shrub associates include whiteleaf manzanita (*Arctostaphylos viseida*) and buck brush (*Ceanothus cuneatus*).

Spanish bayonet (*Yucca whipplei*) is also conspicuous on rocky outcrops. On more mesic sites, such as north and east facing slopes in the higher elevations, the chamise is replaced by a broad leaf mixed chaparral, which exhibits greater diversity and a more uneven cover. It is characterized by mountain mahogany (*Cercocarpus betuloides*), deerbrush (*Ceanothus integerrimus*) and whiteleaf manzanita (*Arctostaphylos viscida)*, but is in some places associated with tree species, including California buckeye and interior live oak. At the higher elevations, greenleaf manzanita (*Arctostaphylos patula*), snowbrush (*Ceanothus cordulatus*), and chinquapin (*Castanopsis sempervirens*) are characteristic of a lower growing chaparral which is more highly adapted to cool temperatures (Parsons 1976). At slightly higher elevation, upland live oak woodland, dominated by interior live oak and canyon live oak (*Quercus chrysolepis*) occurs on moist sites, whereas black oak woodland, dominated by California black oak (*Q. kelloggii*) and occasional conifers, occurs on drier sites.

The change from black oak woodland to ponderosa pine forest is abrupt at relatively dry sites at mid elevation. This vegetation type is widespread on the west side of the park and is dominated by ponderosa pine (*Pinus ponderosa*), with incense cedar (*Calocedrus decurrens*), white fir *(Abies concolor*), and sugar pine (*P. lambertiana*) as the subdominants; incense cedar and white fir are later successional species in this situation and are often found in the understory. Early successional ponderosa pine is very tolerant of fire, whereas the co-occurring species are fire-sensitive. At slightly higher and wetter sites, white fir forest dominates; this is the forest type in which giant sequoia (*Sequoiadendron giganteum*) is found in scattered groves in the park. This charismatic park namesake species is known for its immense biomass and potential old age, and is one of the most dominant canopy trees in the world (Harvey et al. 1980, Aune 1994). It has very thick fire-resistant bark, and fire scars record dozens of fire events over thousands of years in older trees.

Thirty-five of the 75 sequoia groves in the Sierra are found in SEKI. Giant Forest has more than 20,000 sequoia trees, including the General Sherman tree which is believed to be the largest living tree in the world, with a circumference of 31 m and a height of almost 84 m.

The boundaries of the giant sequoia groves in the Sierra Nevada appear to have been very stable for at least the past 500 years. A majority of the groves studies by Rundel (1971) showed evidence of a gradual decrease in stand density due to low levels of regeneration. Vegetation in the giant sequoia groves is often dominated by white fir, with sugar pine as a characteristic associate. Giant sequoia constitutes a minor portion of the tree density, but includes the majority of the basal area of the canopy species (Rundel 1971).

At somewhat higher elevation on moist sites, white fir gives way to red fir (*Abies magnifica*) forest, with the latter species often occurring in nearly pure stands. On drier sites that often have shallow soils, Jeffrey pine forest gradually replaces ponderosa pine. A band of lodgepole pine forest dominates above 2,700 m, and subalpine forest comprises the highest arboreal zone, with a transition from lodgepole pine (*P. contorta*) to foxtail pine (*P. balfouriana*) and whitebark pine (*P. albicaulis*). Meadows are intermixed with wetter forest types above 2,000 m, and alpine vegetation containing a variety of forbs and grasses occurs above treeline.

High-elevation meadows constitute an important vegetation resource within the parks. The vegetation of seven subalpine meadows within SEKI was classified into 19 plant associations and described by Benedict (1983). Vegetation data were also analyzed to elucidate potential relationships between plant associations and 16 environmental factors. The environmental factor

al. (1991) investigated phytoplankton productivity in subalpine Eastern Brook Lake. Comparisons were also made of primary production and near-surface chlorophyll levels among other high-elevation lakes in the Sierra Nevada. Eastern Brook Lake was generally similar to other high-elevation lakes with respect to phytoplankton productivity, chlorophyll levels, and algal taxa present (Thomas et al. 1991). Additional data on algal primary production and chlorophyll concentrations in lakes in SEKI were presented by Melack et al. (1987, 1989) and Sickman and Melack (1991).

A number of rodent species can be observed within SEKI. The California ground squirrel (*Spermophilus beecheyi*) is the common species below about 2,400 m, whereas at higher elevation Belding's ground squirrels (*Spermophilus beldingi*) are frequently encountered. The species of chipmunks present vary with elevation, and include Merriam's chipmunk (*Eutamias merriami*), lodgepole chipmunk (*E. speciosus*), and alpine chipmunk (*E. alpinus*); the Colorado chipmunk (*E. quadrivittatus*) may also occur on the eastern edge of the parks. Golden mantled ground squirrels (*Spermophilus lateralis*) frequent picnic areas and woodland and campsite areas at elevations up to about 3,300 m. Tree squirrels present in the parks include Douglas' squirrel (*Tamiasciurus douglasii*) and western gray squirrel (*Sciurus griseus*). Yellow-bellied marmots (*Marmot flaviventris*) and pikas (*Ochotona princeps*) are frequently observed in high mountain meadows and rocky areas.

Mule deer (*Ododoileus hemionus*) are widespread, and mountain sheep (*Ovis canadensis*) can be observed in the eastern and southern portions of Kings Canyon National Park and in the high country of Sequoia National Park. Mustelids include both the spotted skunk (*Spilogale putorius*) and striped skunk (*Mephitis mephitis*), long tailed weasel (*Mustela frenata*), ermine (*Mustela erminea*), badger (*Taxidea taxus*), and wolverine (*Gulo gulo*). Fishers (*Martes pennanti*) may also occur within the parks. Both coyote (*Canis latrans*) and gray fox (*Urocyon cinereoargenteus*) are common. Red fox (*Vulpes vulpes*) is rare near timberline. Black bears (*Ursus americanus*) are found throughout the parks. Bobcats (*Felis rufus*) are relatively common and mountain lions (*F. concolor*) are also present, but seldom seen. Other species of mammal which occur within the parks include ringtail (*Bassariscus astutus*), whitetail jack rabbit (*Lepus townsendii*), porcupine (*Erethizon dorsatum*), dusky-footed woodrat (*Neotoma fuscipes*), and mountain beaver (*Aplodontia rufa*; Van Gelder 1982).

Golden beaver (*Castor canadensis suburatus*) are found in the upper Kern River Canyon of Sequoia National Park. It has been difficult to resolve the question of whether these beavers are native or exotic, but the bulk of the evidence suggests that they are exotic (Townsend 1979).

Seventy-five high-elevation lakes in the Sierra Nevada were sampled for microcrustacean species and major ion chemistry by Stoddard (1987b). Five common microcrustacean community types were delineated with cluster analysis, and these were related by stepwise multiple regression to fish presence and to various chemical, morphometric, topographic, and geologic variables. The distribution of four of the community types could be well-predicted using fish presence and elevation or fish absence and lake depth. The distribution of the fifth community type was independent of fish presence or absence, but phosphate concentration was a significant predictor of the presence of this community (Stoddard 1987b).

Melack et al. (1989a) surveyed fish populations in lakes, streams, and ponds in the Marble Fork of the Kaweah River watershed, including Emerald, Aster, Heather, and Pear Lakes, associated inlet and outlet streams, and ponds in the vicinity of Emerald Lake. They found only two common aquatic vertebrate species in the study area, the brook trout (*Salvelinus fontinalis*) and the Pacific tree frog (*Pseudacris regilla*). Brook trout were the only fish present in the study lakes and associated outlet streams. They were stocked in the lakes from the 1920s to 1960s, but the current populations are maintained by natural reproduction.

Emerald Lake contained an estimated 1,000 brook trout older than one year and there was little change in population size from 1985 to 1987. Other surveyed lakes had similar estimates for brook trout population size, ranging from about 660 adult fish in Pear Lake to 1200 adult fish in Aster Lake. The number of underyearlings of brook trout was found to be variable from year to year, but the data of Melack et al. (1989) suggested that populations of brook trout in Emerald Lake and associated waters remained relatively constant from year to year. The interannual variability in success of reproduction was probably related to local climatic conditions whereas the constancy of the adult population levels was probably related to adult longevity and dominance by a few age classes (Melack et al. 1989).

Physical disturbances such as flooding, drought, avalanche, and ice formation can cause trout mortality, limit recruitment, and affect growth rates. For example, heavy snowfall, extended winter conditions, and flooding during snowmelt in 1986 reduced young-of-the-year to 18-28% of the average for 1985, 1987, and 1988 (Soiseth 1992).

The Little Kern golden trout (*Oncorhynchus aquabonita whitei*) is recognized by the state of California as a threatened species. This subspecies of the golden trout is endemic to the Little Kern River drainage in Sequoia National Park and Sequoia National Forest. The principal cause of the threatened status of this subspecies is believed to be interbreeding with rainbow trout (*Oncorhynchus mykiss*) introduced into the drainage in the past (Christenson 1978).

The Sacramento-San Joaquin River system is hydrologically isolated from other drainage systems in California. As a result of this isolation, there are a variety of fish species in the river and the fish fauna is over 75% endemic (Miller 1958). Since the late 19th century, intensive agriculture, mining, industry, and the development of population centers in the Sacramento-San Joaquin Valley have drastically changed the quality and distribution of the water, particularly on the valley floor. These changes, combined with widespread introductions of non-native fish, have had serious impacts on the native fish fauna (Moyle and Nichols 1973).

In 1970, a study of fish species distributions was conducted in the San Joaquin River system, including tributaries that flow from YOSE and SEKI. The purpose of the study was to describe the foothills fish associations and to analyze ecological relationships between fish species distributions and environmental characteristics. The study was carried out on streams between elevations of approximately 90 and 1,100 m. Twenty species of fish were collected, nine of which were native. The fish were classified into four distinct fish associations, each generally found under a distinctive set of environmental conditions. The rainbow trout association was found in cold, clear, permanent streams of the higher elevations that were included in the study. The California roach association was found in the small, warm, intermittent tributaries to the larger streams. The native cyprinid-catostomid association was restricted to the larger, low elevation streams, and the introduced fishes association was found in low elevation intermittent streams that had been heavily modified by human activities (Moyle and Nichols 1973).

Moyle and Nichols (1974) conducted a study of streams in the foothills of the Sierra Nevada above the San Joaquin Valley. The study area was confined to streams between the elevations of 90 and 1,100 m. Twenty-four species of fish were collected in the study, 12 native, 12 introduced. The present distribution of those fish species were compared to the inferred pre-1900 distribution from available historical records. Overall, the ranges of introduced species have expanded while the ranges of native species, especially the California roach (*Hesperoleucus symmetricus*), hardhead (*Mylopharodon conocephalus*), Sacramento squawfish (*Ptychocheilus grandis*), and Sacramento sucker (*Catostomus occidentalis*), have contracted (Moyle and Nichols 1974). The authors found healthy populations of native fishes only in a rather narrow middle elevation band of comparatively undisturbed sections of foothill streams.

The native fish populations in different foothill stream systems now appear to be isolated from each other and are thus in danger of local extinction as foothill development proceeds.

The bullfrog (*Rana catesbeiana*) was introduced into California several times in the period between about 1914 and 1920 (Storer 1922). After the initial introduction, it spread rapidly throughout the state. In the San Joaquin Valley and the Sierra Nevada foothills, one of the results of the introduction of bullfrogs seems to have been the elimination of the red legged frog (*R. aurora*) from the valley floor and foothill ponds and also the reduction of populations of the foothill yellow-legged frog (*R. boylei*) in foothill streams (Moyle 1973).

4. Fire

The overall goal of the SEKI fire management program within the giant sequoia/mixed conifer forest is now to restore and maintain the range of fire behavior and effects present prior to European settlement to the maximum extent possible (Piirto et al. 1998). Changes in fire management policies within SEKI were summarized by Vankat (1977). In September 1967, the NPS reversed its policy of suppression of all fires. Burning to achieve vegetation or wildlife management objectives was approved as a substitute for natural fire. In 1968, an experimental let-burn area was established, more recently known as a fire management zone. The first such zone included all land above 2,400 m elevation in the drainage of the middle fork of the Kings River in Kings Canyon National Park (Vankat 1977). All lightning-caused fires in this zone were allowed to burn under careful observation. Also, the first relatively large prescribed burn was carried out in that same year. The fire management zone was subsequently enlarged in 1969 and the prescribed burning program was expanded to include middle-elevation sequoia groves and other vegetation types. The high-elevation fire management zone was further expanded in 1971 and 1972 to include approximately 71% of the area of SEKI. The initial objective of the fire management program was to eliminate unnatural accumulations of dead and downed vegetation debris and dense stands of young trees which were filling in formerly open forest, so that naturally-occurring fire could be allowed to carry out its natural role (Vankat 1977).

A fire management plan was originally prepared for SEKI in 1979. The current version was prepared in 1992. The principal goals are to (1) protect public safety, natural resources, and developments from wildfire through the use of management-ignited prescribed fires, as well as various prevention, suppression, and pre-suppression activities and (2) to restore and maintain the pre-20th century fire regime to the maximum extent possible.

The prescribed fire program in SEKI allows naturally-occurring fires to burn in certain areas, generally above 2,450 m elevation, although these fires are closely monitored and actively suppressed if fire threatens to burn beyond zone boundaries. Management-ignited prescribed burns are carried out to reduce fuel hazards in areas where fuels are deemed to be unacceptably high. SEKI is developing one of the most detailed fire-effects databases in the NPS. Information on fire and fuels is also being collected to help to develop computer models that will better predict the behavior of wildfire and prescribed natural fire.

Before the arrival of Europeans, frequent fires characterized the chaparral environment. These served to preserve a mosaic of different age stands of brush, preventing a large buildup of fuels, and benefitting wildlife by providing continuous supply of tender shoots for browse. The vegetation patterns had been selected for as a consequence of periodic fire regimes over a period of thousands of years. Following fire exclusion throughout most of the 20th century, the chaparral vegetation in the park became more prone to catastrophic fire because of heavy fuel accumulation (Parsons 1976). More recently, park management has focused on determining the role of fire in the chaparral ecosystem and evaluating the feasibility of utilizing different manipulative means for accomplishing increased fire frequency. Park staff and cooperating

scientists initiated a research program directed at obtaining a better understanding of the ecology of the foothill zone of Sequoia National Park. Because fire plays such an integral role in the dynamics of the foothill ecosystems, much of this effort was directed towards gaining a better understanding of the role of fire in the area (Parsons and Stohlgren 1986, Rundell 1982). Fire exclusion during the 20th century has also resulted in increased stem densities and fuel loadings in the understory of mid-elevation forests throughout the park, with the potential for increased overstory damage should a contemporary fire occur. Park managers are using limited applications of prescribed burning to reduce these fuel loadings and increase germination of giant sequoia.

A study was conducted by Caprio and Swetnam (1995) to document fire occurrence patterns in montane forest stands in Sequoia National Park for the last 300 to 400 years using dendrochronological analysis. Data were reported from 12 sites along an elevational gradient in the Kaweah River watershed. Estimated fire histories documented the occurrence of widespread fires, with temporally variable fire frequencies. The highest frequencies were observed in the mid to late 1700s, followed by a decline in fire occurrence. That decline continued into the settlement era with nearly a complete cessation of fires by the start of the 20^{th} century. Major fire years were recorded at many sites as synchronous fire dates across most of the elevational transect in the years 1729, 1755, 1770, 1795, 1812, 1856, and 1873 (Caprio and Swetnam 1995).

Swetnam et al. (1991) reconstructed a 1400-year history of wildfire in the Mariposa Grove of giant sequoias in Sequoia National Park. Partial cross-sections were taken from 18 dead and fire-scarred trees and the tree rings and fires scars were dated. The resulting chronology suggested that fires recurred at intervals ranging from 1 to 15 years. Changes in fire frequency over scales of centuries were also suggested by the data. In the early 1960s, ecologists became concerned about the observation that there were relatively few sequoia seedlings or saplings within the giant sequoia groves, whereas the density of some other shade-tolerant trees seemed to be increasing. Research also suggested that elimination of episodic fires during the past century may have played a role in eliminating necessary conditions for sequoia regeneration. Sequoia seeds germinate and establish best in mineral soils that have been exposed by surface burns (Harvey et al. 1980).

This concern about changes in the structure of sequoia/mixed conifer stands within the park was an important stimulus to the use of prescribed fire in some groves as early as 1968. Many of the prescribed burn areas within the groves now have abundant sequoia seedlings and saplings (Swetnam et al. 1991).

Fire scars in giant sequoia were used by Swetnam (1993) to reconstruct the spatial and temporal patterns of surface fires that burned episodically through five giant sequoia groves during the past 2,000 years. Comparisons with independent dendroclimatic reconstructions suggested that regionally synchronous fire occurrence was inversely related to annual precipitation amount and directly related to decadal to centennial mean temperature. The regionally synchronous fire histories indicated the importance of climate in maintaining nonequilibrium conditions in these giant sequoia groves. Fire scar records also revealed the importance of fuel accumulation processes at the local scale. The record indicated that some fires probably burned throughout individual groves and some were smaller and burned only around a single tree or group of trees. Regional synchrony of fire dates does not indicate continuous burns between groves. It rather suggests that large areas burned throughout the region during some years. Fire frequencies and sizes constantly change through time and therefore Swetnam (1993) inferred that many of the properties and components of these ecosystems were also non-equilibrial.

The Giant Sequoia prescribed fire program at SEKI has been widely recognized as one of the more aggressive fire programs in existence. However, during the first 26 years of prescribed burning in SEKI, only about 1,820 of the 4,370 ha of giant sequoia forest in those parks were burned, including management-ignited fires, prescribed fires, and wildfires. Swetnam (1993) extrapolated data for Giant Forest which suggested an average of 1,050 ha would need to be burned each year to achieve the 4.1 year mean fire interval that was estimated to occur between about 1,300 and 100 YBP. This compares to a mean of 70 ha per year that was actually burned between 1968 and 1994 (Parsons and Botti 1996). The area burned under the current fire management program seems to not be sufficient to simulate pre-settlement fire regimes.

A significant increase in the use of prescribed fire in the future may be necessary within SEKI to reverse unhealthy forest conditions that have developed due to fire suppression. To promote ecosystem health through increased prescribed burning, it will be necessary to more effectively coordinate the prescribed fire program with air quality specialists and regulators. Following the extreme fire season of 1994 there has been increasing public interest in, and perhaps support for, the use of prescribed fire as a way of promoting ecosystem health and reducing the risk of catastrophic loss from wildfire.

B. EMISSIONS

SEKI is located within the San Joaquin Valley (SJV) air basin (SJVAB), and is potentially exposed to pollutants transported from the SJV and other areas. Approximately nine percent of the state's population lives in the eight counties of the SJVAB and emission sources within the SJVAB account for about 14 percent of total statewide emissions (Alexis et al., 1999). Emissions in the SJV derive from a number of moderate-sized urban areas, primarily located along the Highway 99 corridor. Since 1980, population growth in the SJV has been more rapid than in other parts of California, partially offsetting the effects of emission-control programs (Alexis et al., 1999).

Emission levels from counties within 140 km of SEKI are found in Table IX-1. The principal species of concern are ozone precursors (NO_x and ROG) and PM. SO_2 emissions are not high. SEKI is located within Fresno and Tulare counties. No major point sources are located in Tulare County; major point sources are not numerous in Fresno County, nor in most

Table IX-1. 1995 Emissions from counties within 140 km of SEKI. (Source: CARB Almanac, 1999b; SO_x from CARB Emissions Website, 1999a.) Units are 1000 tons/year.

County	NO_x	ROG*	PM_{10}	CO	SO_x
Merced	15.3	10.2	17.5	75.9	0.7
Madera	11.3	7.7	8.8	41.2	0.4
Fresno	39.8	39.4	45.3	205.1	3.7
Tulare	16.8	18.3	19.3	113.9	0.4
Kern [1] (San Joaquin)	60.2	50.0	24.8	186.5	2.9
Kings	9.5	8.0	13.5	36.5	0.4

* Reactive Organic Gases
[1] Portion of the county in the air basin

other counties of the SJV (Figures II-3 through II-6). Within Fresno County, the closest sources that emit at least 100 tons/year of ROG, NO_x, PM_{10}, or SO_2 are located near Fresno and Kingsburg; three additional point sources are located in the western portion of the county. As of 1996, stationary sources accounted for 22 % of ROG emissions, 34 % of NO_x emissions, and 4 % of PM_{10} emissions in Fresno County (CARB, 1998b). In Tulare County, stationary sources accounted for 8 % of ROG emissions, 11 % of NO_x emissions, and 5 % of PM_{10} emissions in 1996 (CARB, 1998b). In both Fresno and Tulare counties, mobile sources dominated NO_x emissions, while area sources (road dust, construction, and farming operations) dominated PM emissions. Mobile sources accounted for 46 % of ROG emissions in Fresno County and 41 % in Tulare County.

Besides criteria pollutants (ozone, PM, NO_2, SO_2, and CO), pesticides, which are used extensively in California's heavily-farmed SJV, are of potential concern. Pesticides vary as to their levels of volatility. Potential impacts are also affected by the degree to which a particular pesticide breaks down in the environment and the levels of toxicity of the breakdown products. An overview of pesticide usage in the SJV is shown in Table IX-2. The amount of pesticide use suggests that further study is warranted. For example, in Fresno County, the estimated pesticide VOC emissions potential of 17.6 million pounds (8.8 thousand tons; Table IX-2) is a substantial portion of the total ROG emissions of 39.4 thousand tons (Table IX-1); however, the California Department of Pesticide Regulation (CDPR) estimate of VOC emissions potential exceeds that used in the CARB emissions inventory (CARB, 1998b) by a factor of two. Pesticides are, of course, of concern for their potential toxicity, and not just for their ozone-forming potential. Some research indicates that pesticides used in the SJV may be transported into Sequoia National Park, though the observed concentrations do diminish substantially between the Ash Mountain entrance station and Lower Kaweah (Zabik and Seiber, 1993).

Table IX-2. Pesticide applications in the SJV by county in 1994. (Source: California Department of Pesticide Regulation website, 1999).

Air Basin	Counties	Pesticide application (million lbs.)	VOC Emissions Potential (million lbs.)
San Joaquin Valley	Fresno	55.1	17.6
	Kern	32.4	10.9
	Kings	11.5	6.5
	Madera	13.3	2.0
	Merced	12.5	4.5
	San Joaquin	15.8	3.8
	Stanislaus	9.4	3.1
	Tulare	23.1	6.0

Pesticides applied to crops that are intensively produced in the Central Valley can volatilize under warm valley temperatures, and subsequently be transported via upslope air movement to be deposited in the cooler, high-elevation regions of Sequoia National Park. Zabik (1993) demonstrated atmospheric transport of organophosphate pesticides from the Central Valley to two sites within Sequoia National Park. The first site was located at Ash Mountain, at an

elevation of 533 m. The second site was located in Giant Forest at an elevation of 1,920 m. A third site outside the park was located at the base of the Sierra Nevada foothills. Samples were collected of air, dry deposition, and wet deposition during the winters of 1990-91 and 1991-92, and also during the spring of 1992. Zabik (1993) concluded that organo-phosphate compounds were detected in dry deposition but it appeared that wet deposition accounted for the majority of organophosphate insecticide deposition at all three of the sites during the winter months.

Air and dry deposition samples were collected during the period May to September, 1996 at three elevations in Sequoia National Park and analyzed for eight pesticides by LeNoir et al. (1999). Air samples showed a decrease in pesticide concentration with elevation only at the lower elevations. A dilution of 1.7 was observed from 200 m to 533 m elevation, but no significant dilution was found between 533 and 1,920 m.

An inventory of in-park emissions has recently been compiled by the NPS-Air Resources Division. The results are presented in Table IX-3.

Table IX-3. Summary of 1998 stationary and area, and mobile source emissions (tons/yr) at SEKI.

Activity	Particulates	Sulfur Dioxide	Nitrogen Oxides	Carbon Monoxide	VOCs	HAPs[1]
Stationary and Area Source Emissions						
Stationary Combustion Sources						
Heating units	0.01	0.02	0.44	0.07	0.14	0.00
Generators	0.04	0.09	0.68	0.15	0.05	0.00
Woodstoves	1.01	0.00	0.09	7.66	1.76	0.00
Combustion Emission Subtotal	1.07	0.11	1.22	7.88	1.95	0.00
Fuel Storage Tanks						
Gasoline/Diesel Fuel Tanks	0.00	0.00	0.00	0.00	1.35	0.22
Area Sources						
Campfires	6.72	0.00	1.58	55.30	7.50	0.00
Prescribed Burning	850.00	0.00	2.31	8030.00	260.35	0.00
Wastewater Treatment Plant	0.00	0.00	0.00	0.00	0.04	0.00
Area Source Emission Subtotal	856.72	0.00	3.89	8085.30	267.89	0.00
TOTALS	857.78	0.11	5.11	8093.18	271.19	0.22
Mobile Source Emissions						
Road Vehicles						
Visitor Vehicles	45.50	0.00	15.11	133.25	9.65	–
NPS/GSA Road Vehicles	1.90	0.00	1.45	6.40	0.47	–
SEQU Concessioner Vehicles	0.21	0.00	0.19	0.76	0.06	–
Vehicle Emission Subtotal	47.61	0.00	16.74	140.41	10.18	–
Nonroad Vehicles						
NPS Nonroad Vehicles	0.33	0.00	2.15	1.54	0.78	–
TOTALS	47.94	0.00	18.89	141.95	10.95	–

[1] Hazardous air pollutants, based on the list compiled by EPA

C. MONITORING AND RESEARCH ACTIVITIES

1. Air Quality

Ozone, SO_2, fine particulate, PM_{10}, wet deposition, and dry deposition have all been monitored within Sequoia National Park (Table IX-4). The CADMP dry-deposition site is no longer operational, but a CASTNet site is now located within the park for monitoring dry deposition. For both the CADMP and CASTNet monitors, deposition is not monitored directly, but is rather calculated from ambient concentration measurements.

Table IX-4. Air quality monitoring at SEKI.

Species	Site within park	Site within 50 km
Ozone, hourly	CASTNet	
Ozone, passive	NPS	
SO2	NPS	
PM_{10}	IMPROVE	
$PM_{2.5}$	IMPROVE	
Wet deposition	NADP	
Dry deposition	CASTNet	
Visibility		

a. Wet Deposition

Wet S deposition averaged less than 1 kg/ha/yr as S (equivalently, 3 kg/ha/yr as SO_4^{2-}), with individual years ranging as high as 1.2 to 1.4 kg/ha/yr in some monitoring locations (Table IX-5). Multi-year NO_3^- and NH_4^+ wet deposition rates were each in the range of 0.5 to 1.6 kg/ha/yr as N, yielding total wet N deposition rates of 1.1 to 3.0 kg/ha/yr (Table IX-5). Annual wet N deposition rates were as high as 4.4 and 5.7 kg/ha/yr during some years at some locations in Sequoia National Park (Table IX-5). Annual-average H^+ concentration in precipitation ranged from 1.8 to 4.5 µeq/L (pH 5.74 to 5.35; Table IX-6). However, individual storm events and early spring meltwater often exhibit higher acidity, posing questions about potential transient impacts (see later discussions).

Stohlgren and Parsons (1987) reported wet deposition of N and S for the NADP/NTN monitoring site at Giant Forest for the period 1981-1984. They documented an extremely high variability of ion concentrations in the wet deposition when sample volumes were low, thereby making statistical interpretation of the data problematic. Sample volume alone did not explain much of the variability in the distributions of ion concentrations. Some of the variability may have been due, at least in part, to the length of the dry period between storm events (McColl et al. 1982) and/or collecting samples in the middle of events. For example, if a winter storm event with low ion concentrations was bisected by scheduled sampling, it would artificially create two low-volume, low-concentration storm samples. Summer concentrations of NO_3^- and SO_4^{2-} averaged two and five times higher, respectively, as compared with concentrations reported for remote areas of the world (Stohlgren and Parsons 1987). Though it is uncertain what minimum concentrations of acid may cause direct or indirect foliar damage to plant species in SEKI, Stohlgren and Parsons (1987) emphasized that it is important to note that concentrations are highest during the primary growing season and the period of maximum drought stress, both of which occur during the summer.

Table IX-5. Wet deposition of S and N at CADMP, NADP and UCSB sites in SEKI. (Source: Blanchard et al. 1996; Melack et al. 1998; Dwight Oda and Brent Takemoto, CARB, 1999, pers. comm.). Units are kg/ha/yr.

Site	Water Year*	Sulfur	NO_3-N	NH_4-N	Total Inorganic N
EMERALD LAKE UCSB (2824 m.)	1991	0.4	0.6	0.6	1.2
	1992	0.5	0.7	1.1	1.8
	1993	1.2	1.0	1.1	2.1
	Average	0.7	0.8	0.9	1.7
MINERAL KING UCSB (2694 m.)	1991	0.6	0.9	1.3	2.3
	1992	0.5	0.7	0.8	1.4
	1993	0.6	0.6	0.7	1.3
	Average	0.6	0.7	0.9	1.7
ONION VALLEY UCSB (2800 m.)	1991	0.5	0.6	0.7	1.3
	1992	0.5	0.5	0.4	1.0
	1993	0.4	0.5	0.5	1.0
	Average	0.5	0.5	0.6	1.1
SEQ-GF/NADP (1902 m.)	1986	1.0	1.2	1.0	2.2
	1990	0.8	1.2	1.6	2.7
	1991	0.8	1.4	1.6	3.0
	1992	0.5	0.8	1.1	1.9
	1993	1.4	1.7	2.7	4.4
	1994	0.7	1.2	1.8	3.0
	Average	0.9	1.2	1.6	2.9
SEQUOIA ASH MT CADMP (548 m.)	1985	0.8	1.2	1.5	2.8
	1986	0.9	1.7	1.2	3.0
	1987	0.7	1.6	1.1	2.8
	1988	0.7	1.8	1.3	3.1
	1989	0.6	1.1	0.8	1.9
	1990	0.9	1.6	2.1	3.6
	1991	0.5	1.0	1.7	2.7
	1992	0.4	0.6	0.9	1.5
	1993	0.8	1.1	2.0	3.1
	1994	0.5	1.0	1.7	2.6
	1995	1.2	2.0	3.6	5.7
	1996	0.4	0.6	0.7	1.3
	1997	0.5	1.1	1.8	2.9
	1998	0.4	0.6	0.5	1.1
	Average	0.7	1.2	1.5	2.7
SEQUOIA GF CADMP (1890 m.)	1986	1.1	1.6	1.2	2.8
	1987	0.7	1.9	1.3	3.1
	1988	0.8	1.6	1.2	2.8
	1989	0.6	0.8	0.7	1.5
	1990	0.9	1.2	1.5	2.7
	1991	0.7	1.4	2.0	3.4
	1992	0.4	0.7	0.8	1.5
	1993	1.2	1.5	2.4	3.9
	1994	0.7	1.1	1.9	3.1
	1995	1.0	1.5	2.2	3.7
	1996	0.5	0.5	0.8	1.3
	1997	0.4	0.6	0.7	1.3
	1998	0.3	0.4	0.4	0.8
	Average	0.7	1.2	1.3	2.5

* The CARB water year is July 1 through June 30. For instance, water year 1995 is for July 1, 1994 through June 30, 1995.

Table IX-6. Wetfall chemistry at CADMP, NADP and UCSB sites in SEKI. Units are μeq/ L, except precipitation (cm). Source: Blanchard et al. 1996; Melack et al. 1998; Dwight Oda and Brent Takemoto, CARB, 1999; Melack et al. 1997.

NAME	Water Year*	Prec	H^+	SO_4^{-2}	NH_4^+	NO_3^-	Ca^{+2}	Mg^{+2}	Na^+	K^+	Cl^-
EMRLD LAKE UCSB (2824 m.)	1991	112.2	3.5	2.5	3.9	3.5	2.1	0.7	1.6	1.6	1.5
	1992	78.4	3.7	4.3	10.3	6.5	3.9	1.0	1.3	1.5	2.0
	1993	256.3	5.1	3.0	3.0	2.8	1.1	0.5	1.7	1.1	2.8
	Ave.	149.0	4.1	3.3	5.7	4.3	2.4	0.7	1.5	1.4	2.1
MINERAL KING UCSB (2694 m.)	1991	64.3	5.1	6.1	14.6	10.5	3.4	1.4	2.3	1.4	2.2
	1992	56.0	3.9	5.2	9.8	8.5	4.3	0.9	1.7	1.2	2.3
	1993	108.4	3.9	3.4	4.4	3.9	2.6	0.5	1.2	1.1	1.6
	Ave.	76.2	4.3	4.9	9.6	7.6	3.4	0.9	1.7	1.2	2.0
ONION VALLEY UCSB (2800 m.)	1991	70.0	6.1	4.2	7.2	6.2	3.3	0.8	1.3	1.0	1.7
	1992	52.8	4.0	5.8	6.0	6.9	4.4	0.9	1.8	1.1	2.0
	1993	86.2	5.6	3.1	4.1	4.0	2.8	0.7	1.0	1.1	2.9
	Ave.	69.7	5.2	4.3	5.8	5.7	3.5	0.8	1.4	1.1	2.2
SEQ-GF/NADP (1902 m.)	1981	50.8	6.8	16.9	13.7	14.1	4.6	1.8	4.4	0.9	5.6
	1982	141.0	5.0	8.3	4.8	4.1	2.1	1.4	2.8	0.4	8.5
	1983	182.9	4.0	4.5	4.4	4.2	1.7	1.3	3.8	0.4	4.5
	1984	84.3	3.3	10.8	9.0	8.1	3.3	1.7	3.4	1.0	4.1
	1986	103.6	3.7	6.1	8.2	6.8	1.7	1.8	3.4	0.4	3.6
	1987	53.0	3.8	7.1	15.3	13.5	2.2	1.3	3.8	0.9	3.5
	1988	77.0	3.3	7.3	8.2	9.4	4.0	1.3	3.4	1.2	2.6
	1990	60.9	2.4	8.5	18.6	13.6	2.8	1.1	4.5	0.4	4.5
	1991	64.0	2.1	7.6	18.1	15.3	5.8	1.9	5.3	0.9	5.1
	1992	55.1	2.1	5.6	13.7	10.6	4.7	1.2	3.8	0.5	3.3
	1993	130.1	2.9	6.6	15.0	9.3	1.7	0.9	3.2	0.4	3.1
	1994	70.0	2.8	5.9	18.7	12.2	1.9	1.1	4.0	0.5	3.9
	Ave.	76.0	2.5	6.8	16.8	12.2	3.4	1.2	4.1	0.5	4.0
SEQUOIA ASH MT CADMP (548 m.)	1985	55.9	7.4	8.6	15.9	19.5	4.8	2.0	15.4	1.2	11.5
	1986	89.9	6.0	6.3	13.6	9.9	3.3	1.8	6.3	0.9	7.1
	1987	35.9	6.1	11.7	32.5	22.4	6.9	2.7	5.8	1.1	6.8
	1988	46.9	5.0	9.5	28.1	19.8	10.0	1.6	3.7	0.6	2.9
	1989	41.8	3.9	8.8	19.0	12.7	6.0	2.2	5.5	0.9	8.2
	1990	39.7	3.1	13.6	37.2	28.2	9.1	3.6	6.0	0.8	4.8
	1991	39.4	1.9	8.6	30.1	18.8	6.7	2.3	5.7	0.9	5.8
	1992	36.2	1.8	6.1	18.2	11.3	12.0	4.6	5.2	1.6	4.7
	1993	79.4	2.5	5.9	17.7	9.8	8.9	4.0	4.2	0.9	4.7
	1994	41.0	1.8	7.4	29.0	16.6	18.8	7.4	4.8	1.0	5.5
	1995	102.0	3.2	7.3	25.2	14.4	12.1	6.2	4.5	1.4	4.6
	1996	52.2	3.4	4.4	10.2	7.6	6.8	3.6	5.0	0.9	5.0
	1997	82.2	3.8	4.0	15.5	9.9	7.0	2.8	2.7	1.2	2.8
	1998	71.5	4.5	3.3	5.4	5.5	13.3	2.9	6.5	0.9	7.9
	Ave.	60.4	2.9	6.7	20.9	13.6	10.5	4.1	5.0	1.1	5.1

Table IX-6. Continued.

NAME	Water Year*	Prec	H^+	SO_4^{-2}	NH_4^+	NO_3^-	Ca^{+2}	Mg^{+2}	Na^+	K^+	Cl^-
SEQUOIA GF CADMP (1890 m.)	1986	166.7	5.2	4.1	6.8	5.3	1.8	1	4.5	0.5	4.6
	1987	61.3	6.7	7.4	22	14.5	4.7	1.3	2.5	0.5	3.9
	1988	76.3	4.6	6.5	15.4	10.8	9.4	1.1	2.5	0.5	1.8
	1989	70.8	4.2	5.4	7.9	7.2	2.8	1.1	2.1	0.5	3.8
	1990	61.0	4.5	9.2	17.9	13.9	6.0	2.1	4.1	1.0	3.0
	1991	57.2	3.0	8.0	24.9	17.8	7.0	2.3	4.0	1.0	4.0
	1992	55.8	2.6	4.3	10.8	8.7	7.9	3.1	2.8	0.7	2.5
	1993	128.1	2.5	5.6	13.5	8.5	8.2	3.1	2.3	0.7	2.8
	1994	67.6	2.3	6.3	20.4	11.8	13.2	5.9	3.8	0.7	4.3
	1995	162.7	3.5	3.9	9.7	6.6	7.6	5.6	2.9	1.2	2.5
	1996	95.2	3.2	3.3	5.7	4.0	5.2	2.7	3.0	0.8	3.2
	1997	114.1	2.6	2.1	4.1	3.8	6.0	2.5	1.6	0.8	1.4
	1998	75.2	4.0	2.4	4.0	3.8	7.9	1.6	2.3	0.9	2.9
	Ave.	90.8	3.1	5.0	12.3	8.8	7.7	3.2	3.0	0.9	3.0

* The CARB water year is July 1 through June 30. For instance, water year 1995 is for July 1, 1994 through June 30, 1995.

The Emerald Lake Basin has been the focus of considerable research on the effects of N and S deposition on soils, forests, and surface waters. Wet deposition was monitored near treeline (elevation 2,800 m) during the water years 1985 through 1987 by Williams and Melack (1991). Precipitation amounts ranged from one of the wettest years on record (1986) to one of the driest (1987). Volume-weighted pH was 4.9 for rainfall and 5.3 for snowfall. Volume-weighted mean annual concentrations of SO_4^{2-}, NO_3^-, and NH_4^+ throughout the study were all about 4 to 5 µeq/L. Average total wet deposition of N and S were 2.3 and 2.1 kg/ha/yr, respectively. Low Cl^- and high NH_4^+ concentrations in rain, compared with snow, suggest that localized convective systems (as opposed to oceanic frontal systems during winter) are the main sources of ions in rainfall. Afternoon upslope air flow, induced by heating of air along the mountain slopes, transports air masses from the SJV to the upper reaches of Sequoia National Park on a daily basis during summer (Williams and Melack 1997b).

Extensive monitoring of wet deposition to high elevations of the Sierra Nevada was initiated in 1990 at nine sites (Melack et al. 1998). The upper Marble Fork of the Kaweah River was added to the monitoring program in 1992. Snow chemistry was summarized by Melack et al. (1998) for eight of the (mainly alpine and subalpine) watersheds. Snow chemistry was dilute and similar among the watersheds. Mean concentrations of NO_3^- and NH_4^+ in snow were 2.4 and 2.7 µeq/L, respectively. Mean SO_4^{2-} concentration was 2.0 µeq/L (range about 1.0 to 3.0 µeq/L). However, NO_3^- and NH_4^+ concentrations in non-winter precipitation were eight to nine times greater than in the snowpack (mean values, 20.7 and 23.4 µeq/L, respectively). The SO_4^{2-} concentration in non-winter precipitation was also high, with a mean of 15.1 µeq/L. In contrast, the mean Cl^- level measured in non-winter precipitation (4.2 µeq/L) was only slightly higher than the mean Cl^- concentration in winter snowfall.

Mean annual wet NH_4^+ deposition was 0.70 kg/ha NH_4^+-N and mean annual wet NO_3^- deposition was 0.63 kg/ha NO_3^--N for the 36 water years of record. For both ions, the maximum wet loading rates were measured at Emerald Lake during water year 1987 (3.6 kg N/ha).

Concentrations of N measured in winter snow in the Emerald Lake watershed were among the most dilute measurements of N recorded in wet precipitation (Williams et al. 1995). Nitrogen concentrations in winter snow of about 2 µeq/L each for NH_4^+ and NO_3^- were comparable to measurements from central Alaska (Galloway et al. 1982). However, mean concentrations of N in rainwater of about 55 µeq/L for NH_4^+ and 42 µeq/L for NO_3^- were comparable to N concentrations in rainfall in areas having considerable anthropogenic sources of N, such as the Adirondack and Catskill Mountains of New York (Stoddard 1994).

b. Occult/Dry Deposition

The CADMP co-located wet and dry deposition samplers at Giant Forest (Blanchard et al., 1996). Dry deposition was calculated from measurements of the ambient concentrations of both gas-phase and particulate species (Table IX-7). Mean dry deposition rates of oxidized N species summed to ~0.7 kg/ha/yr (as N), compared with the multiyear mean wet NO_3^- deposition 1.0 kg/ha/yr (as N) at the same location. The CADMP wet and dry deposition samplers also indicated that dry S (SO_2 plus aerosol SO_4^{2-}) deposition rates were about 0.7 times the rate of wet SO_4^{2-} deposition (Blanchard et al., 1996). Dry NH_4 plus aerosol NH_4^+ deposition was about 0.4 times the rate of wet NH_4^+ deposition (Blanchard et al., 1996). Similarly, Bytnerowicz et al. (1991) measured dry deposition of ions to branches of lodgepole pine and western white pine (*Pinus monticola*) in Emerald Lake watershed during the summer of 1987. Extrapolated out to the entire year, their estimates of dry NO_3–N and dry SO_4–S deposition were each near 0.5 kg/ha/yr (as N or S). At Eastern Brook Lake in the eastern Sierra Nevada, Bytnerowicz et al. (1992) estimated lower dry deposition of N, in the range of 0.4 kg/ha/yr (as NO_3^-N plus NH_4^+-N).

The CASTNet dry deposition monitoring site located at Lookout Point within Sequoia National Park began operating February 4, 1997. The monitoring instrument measures ambient concentrations of gases and particles, and EPA uses a computer model to calculate the dry deposition rates from the measurements. The first calculations of dry-deposition rates for this site were released by EPA in November 2000, for 1999 only. The calculated annual dry deposition rates of N and S were 2.5 kg N/ha/yr and 0.7 kg S/ha/yr, respectively, for 1999. When combined with the wet deposition measurements from the nearby NADP/NTN site (located at Giant Forest, 15.5 km from the dry deposition monitor), the data indicate that the 1999 annual total deposition rates of N and S were 4.7 kg N/ha/yr and 1.3 kg S/ha/yr, respectively.

Table IX-7. Long-term annual averages of calculated dry deposition fluxes at SEKI (Giant Forest) using data from 1988-94. Units are kg/ha/yr as SO_2, ozone, NO_2, etc. The averages were constructed by weighting four seasons equally. For NO_2, no fourth-quarter (October-December) periods met the completeness criteria. Source: Blanchard et al (1996).

Gas-Phase Species					Particulate		
SO_2	Ozone	HNO_3	NO_2	NH_3	NO_3^-	SO_4^{2-}	NH_4^+
0.3	28.07	2.76	0.05	0.49	0.35	0.3	0.14

Brown and Lund (1994) studied the influence of dry deposition and foliar interactions on the chemical composition of throughfall in the Emerald Lake watershed. Summer dry deposition was a substantial component of total annual deposition and generally was in excess of summer wet deposition. In contrast, dry deposition during winter does not appear to be a significant contributor to S and N fluxes in high-elevation watersheds in SEKI. Dozier et al. (1989) developed a water and solute balance for Emerald Lake watershed during the 1986, 1987, and 1988 water years. The annual volume-weighted concentration of solutes in precipitation was less than or equal to 5 µeq/L for each of the major ions and dry deposition to the winter snowpack was not an important contributor of S or N. Additional solute budget data for Emerald Lake was presented by Melack et al. (1998).

Cloudwater deposition in SEKI was compared with measurements at similar locations in other parts of the United States by Hoffman et al. (1989). Average concentrations of NO_3^- measured in cloudwater in SEKI were intermediate between the concentrations measured by Lovett et al. (1982) at Mt. Mooselauke, NH and measurements by Castillo et al. (1985) at Whiteface Mountain, NY, although NH_4^+ concentrations were more than double those found at either of the northeastern sites. In contrast, cloudwater SO_4^{2-} concentrations were lower at SEKI. Cloudwater pH in Sequoia National Park averaged about 5, compared with pH about 3.5 at the northeastern sites (Hoffman et al. 1989).

Collett et al. (1990) selected two sites in the Sierra Nevada to serve as locations for monitoring the chemical composition of cloud water. The sites were situated at Lower Kaweah at an elevation of 1,856 m in SEKI and at Turtleback Dome, elevation 1,590 m, in YOSE. Both sites were located in open areas which easily intercept approaching clouds. Over 250 hours of cloudwater interception was recorded at Lower Kaweah between September 1987 and August 1988. The two peak months for cloudwater interception were November and April. The number of hours that Lower Kaweah was immersed in clouds during the study year was much less, however, than has been estimated for some other sites in the eastern U.S. and Canada (Collett et al. 1990). pH of the cloudwater samples collected in SEKI ranged from 3.9 to 6.5, with fall samples tending to be somewhat more acidic than those collected during the winter or spring. The inorganic composition of the cloudwater was dominated by NO_3^-, SO_4^{2-}, and NH_4^+. Samples with large excesses of NH_4^+ relative to the sum of NO_3^- and SO_4^{2-} had pH values generally greater than 5, whereas those with excesses of NO_3^- and SO_4^{2-} tended to be more acidic. Cloudwater samples at Turtleback Dome in YOSE tended to be more acidic than those collected at Lower Kaweah during the spring of 1988. In SEKI, the neutralization of cloudwater acidity by NH_3 was largely responsible for maintaining the pH of cloudwater above levels commonly seen at sites in the eastern United States.

Collett et al. (1989) had previously suggested that the deposition of NO_3^-, SO_4^{2-}, and NH_4^+ by cloudwater interception at Lower Kaweah might be comparable to amounts introduced by precipitation, which was because the average concentrations of NH_4^+ and NO_3^- in cloudwater were more than 10 times those observed in precipitation, and the average concentrations of SO_4^{2-} were more than three times those observed in precipitation. Based on the data that were collected by Collett et al. (1990), those earlier estimates were revised. Estimates of total deposition from cloudwater interception appear to be much lower than previously predicted, especially for SO_4^{2-}. Revised cloudwater deposition estimates were, however, still significant compared to precipitation inputs, and for NH_4^+ and NO_3^- the estimated cloudwater deposition input was greater than 50% of the respective wet deposition input.

c. Gaseous Monitoring

Ozone concentrations and exposures for the period 1992-1997 are shown in Table IX-8. The federal hourly ozone standard (120 ppb) was often violated. The state standard (90 ppb) was violated at all locations all years. SUM60 indexes were well over 100,000 ppb-hour for all sites for most years. One-hour maxima, daily mean, and maximum 9AM-4PM average are also higher than at most other parks in the state. As a consequence of such high ozone exposure in SEKI, a great deal of effort has focused on documentation of ozone damage to park vegetation.

Table IX-9 shows summer averages from seven passive ozone sampling sites from 1996-1998, with unavailable data indicated by blanks. During 1996, the only year with complete data for all sites, mean ozone concentrations were greatest at the sites in the western portions of the park, which are potentially more exposed to pollutants transported from the SJV. As part of an effort to monitor ozone exposure and vegetation effects in the Sierra Nevada, the USDA Forest Service and California Air Resources Board conducted passive ozone sampling at Giant Forest and Grant Grove in 1999.

Maximum and mean 24-hour integrated samples for SO_2 are listed in Table IX-10 for the period 1993-1996. SO_2 measurements were discontinued after 1996 due to concerns about their accuracy. The measurements are considered sufficiently accurate to show that the SO_2 concentrations were well below the levels at which plant injury has been documented, ~40 to 50 ppb 24-hour average and 8-12 ppb annual average (Peterson et al. 1992).

2. Aquatic Resources

a. Water Quality

SEKI contains an exceptional array of pristine surface water resources (Figure IX-5), including scores of alpine and subalpine lakes, many of which are extremely dilute, large river systems, mountain streams, and impressive waterfalls. Many of the aquatic resources in the park are highly sensitive to air quality degradation, especially the high-elevation lakes and streams.

The Great Western Divide, a north-south range of rugged mountains located a short distance to the west of the main crest of the Sierra, separates the watersheds of the westward flowing Kaweah River from the southward flowing Kern River in SEKI. The western part of Sequoia National Park is drained by the Kaweah River. It is deeply dissected by branching canyons and ridges. The main canyon of the Kaweah at Ash Mountain Park Headquarters is more than 1,200 m deep and several of the branch canyons are deeper. The Kern River originates in the eastern part of the parks between the Great Western Divide and the main crest of the Sierra. The Kern River is less deeply entrenched than the Kaweah. Its valley is incised about 600 m in the upland areas between the Boreal Plateau on the east side and the Chagoopa Plateau on the west. Tributary streams flowing into the Kern River occur in many places as hanging valleys cut into the canyon walls.

The potential for chronic, and especially episodic, acidification of lakes and streams in SEKI from N and S deposition is an important water quality concern. This concern is based largely on the moderate levels of N deposition that occur in the parks, extreme sensitivity to acidification of many high-elevation surface waters, potential short-term effects of acidic rain-on-snow events during the snowmelt period, and documented importance of episodic hydrological processes in alpine and subalpine watersheds within the parks.

Table IX-8. Summary of ozone concentrations and exposure from SEKI monitoring sites (Source: Joseph and Flores, 1993; National Park Service, Air Resources Division 2000). Bold-face values of the 3-year average number of exceedences indicate violations of the federal 1-hour ozone standard.

Site	Year	Maximum Daily 1-hour Value (ppbv)	2nd Highest Daily 1-hour Value (ppbv)	Number of Daily Maximum 1-hour Values Greater Than or Equal to 125 ppb	3-Year Average Number of Exceedences	Maximum 9am-4pm Average (ppbv)	Sum06 (ppbv-hour)[a]	Number of Valid Hours of Ozone Measurements
Ash Mountain	1982	150	140	10	na	120	85,000	2630
	1983	120	120	0	na	110	61,000	2625
	1984	130	120	1	**3.7**	93	17,000	1057
	1985	140	130	3	1.3	107	90,000	3237
	1986	140	120	1	1.7	109	73,000	3626
	1987	138	127	4	2.7	116	77,000	8266
	1988	124	121	0	1.7	104	63,000	7849
	1989	116	114	0	1.3	91	71,000	8015
	1990	120	119	0	0	93	68,000	8111
	1991	122	120	0	0	97	73,000	8216
	1992	120	117	0	0	100	72,000	8081
	1993	128	127	2	.7	109	75,000	8195
	1994	132	126	4	2.0	108	87,000	8192
	1995	128	119	1	2.3	107	71,000	7933
	1996	124	117	0	1.7	100	75,000	3438
Grant Grove	1990	121	119	0	na	96	57,000	8253
	1991	110	108	0	na	97	62,000	8164

Table IX-8. Continued.

Site	Year	Maximum Daily 1-hour Value (ppbv)	2nd Highest Daily 1-hour Value (ppbv)	Number of Daily Maximum 1-hour Values Greater Than or Equal to 125 ppb	3-Year Average Number of Exceedences	Maximum 9am-4pm Average (ppbv)	Sum06 (ppbv-hour)	Number of Valid Hours of Ozone Measurements
Grant Grove	1992	124	116	0	0	100	61,000	8169
	1993	128	125	2	.7	99	64,000	8018
	1994	125	123	1	1.0	100	79,000	8210
	1995	115	112	0	1.0	91	54,000	5513
Lower Kaweah	1984	110	110	0	na	93	63,000	3085
	1985	130	120	1	na	100	32,000	1722
	1986	130	120	1	.7	94	31,000	2760
	1987	118	109	0	.7	89	46,000	7938
	1988	117	112	0	.3	95	67,000	8126
	1989	112	110	0	0	87	61,000	8051
	1990	121	112	0	0	92	58,000	8221
	1991	116	112	0	0	97	68,000	8186
	1992	121	119	0	0	102	70,000	8182
	1993	129	125	2	.7	99	78,000	8212
	1994	125	123	1	1.0	104	81,000	7721
	1995	115	110	0	1.0	93	55,000	8033
	1996	123	122	0	.3	96	76,000	8213
	1997	112	111	0	0	90	60,000	8111

Table IX-8. Continued.

Site	Year	Maximum Daily 1-hour Value (ppbv)	2nd Highest Daily 1-hour Value (ppbv)	Number of Daily Maximum 1-hour Values Greater Than or Equal to 125 ppb	3-Year Average Number of Exceedences	Maximum 9am-4pm Average (ppbv)	Sum06 (ppbv-hour)[a]	Number of Valid Hours of Ozone Measurements
Lower Kaweah	1998	131	116	1	.3	94	58,000	7016
	1999	115	108	0	.3	91	62,000	7377
Lookout Point	1992	117	115	0	na	105	74,000	6221
	1993	119	113	0	na	94	30,000	5865
	1997	120	115	0	0	98	69,000	6166
	1998	119	117	0	0	99	66,000	6245
	1999	124	122	0	0	101	77,000	7771

[a] Maximum 8 am - 8 pm 90-day rolling average

Table IX-9. Summer average hourly ozone concentrations at passive sampling sites within SEKI (source: Dr. John D. Ray, National Park Service, Air Resources Division, NPS Passive Ozone website, 1999). Units are ppb.

Sample Locations	Elevation (m.)	1996	1997	1998	1999
Crabtree	3261	48.4	57.4	53.8	59.4
Le Conte Canyon	2652	49	53.6	52.1	59.9
Bear Paw Meadow	2177	62.8		60.4	
Mineral King	2314	47.4			
Cedar Grove	1432	39.4			
Lower Kaweah	1865	64.6			
Grant Grove	1981	59			

Table IX-10. Maximum and mean SO_2, from 24-hour-resolution samples at SEKI. Samples are collected every 3-4 days, unless noted. (Source: NPS Air Resources Division). Units are ppb.

SO_2	1988	1989	1990	1991	1992	1993	1994	1995	1996
Maximum	na	na	na	na	na	0.91*	0.82	0.48	0.89**
Mean	na	na	na	na	na	0.20*	0.17	0.13	0.19**

na Not available
* Less than 50 samples collected for the year
** 50-75 samples collected for the year

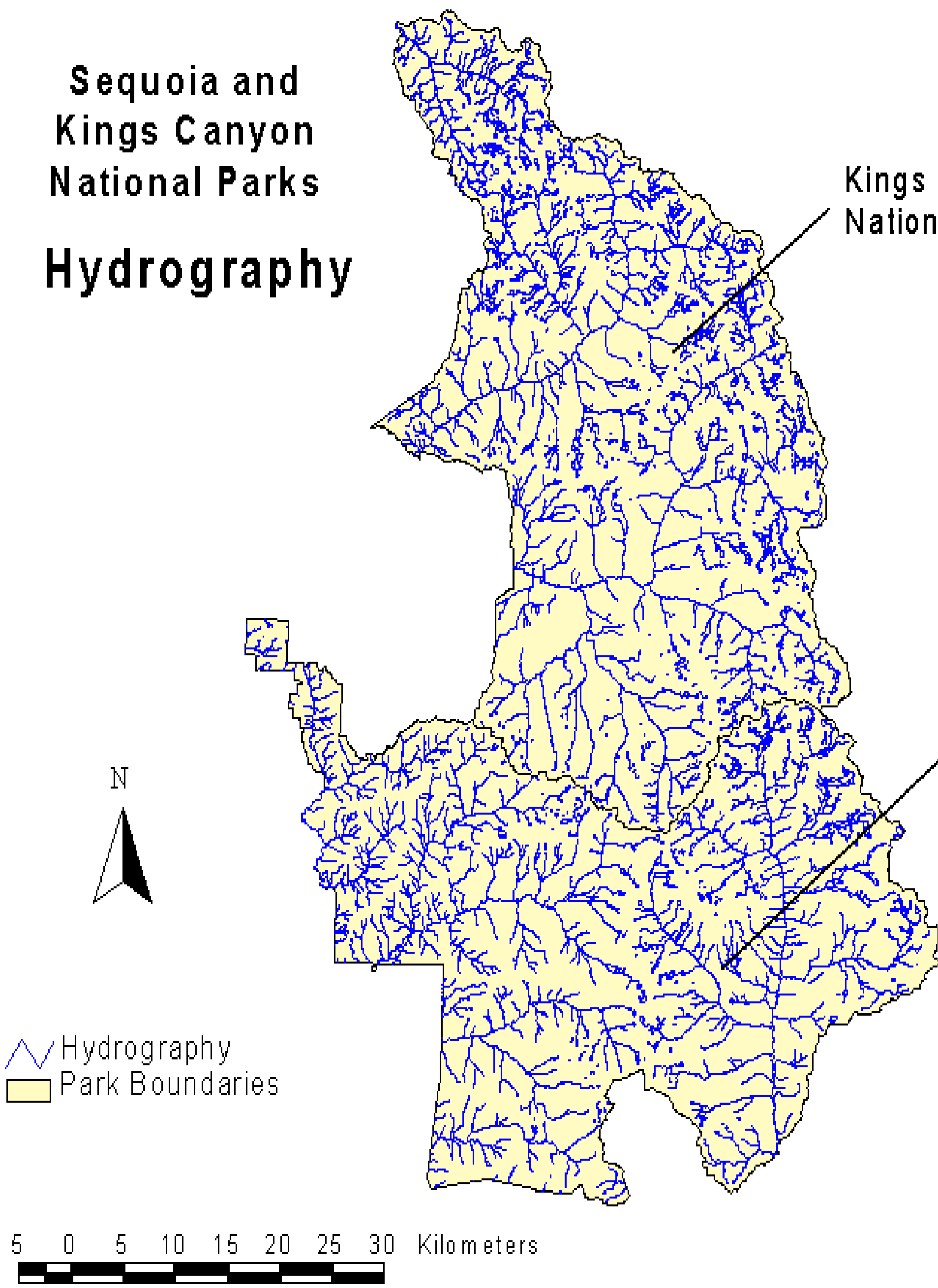


Figure IX-5. Hydrography of SEKI.

Chronic Chemistry

High-elevation lakes and streams in the Sierra Nevada, including those within SEKI, are among the most dilute, poorly-buffered waters in the United States (Landers et al. 1987, Melack and Stoddard 1991). The catchments that supply runoff to these waters are underlain primarily by granitic bedrock and have poorly-developed soils and sparse vegetation. The hydrologic cycle is dominated by the annual accumulation and melting of a dilute, mildly acidic (pH 5.5) snowpack (Melack et al. 1998). A large number of water quality studies have been conducted within the parks, and are summarized below.

Water chemistry was measured in four high-elevation lakes in Kings Canyon National Park during the summer of 1975. These lakes were located in Rae Lake basin and Sixty Lakes basin, both near timberline at elevations of approximately 3,100-3,400 m. Vegetation is sparse in both watersheds (Silverman and Erman 1979). Although major ion chemistry was not presented, the specific conductance and pH data did not suggest an unusually high degree of sensitivity to acidification from atmospheric deposition of S or N.

The Western Lakes Survey (WLS, Landers et al. 1987) sampled 11 lakes in Kings Canyon National Park and 9 lakes in Sequoia National Park, plus an additional 14 lakes within 25 km of SEKI (Table IX-11). About one third of the lakes sampled in and near SEKI had ANC < 50 µeq/L, and two of the lakes in the park had ANC < 20 µeq/L (Figure IX-6). These lakes, especially those having ANC < 20 µeq/L, are sensitive to potential acidification from acidic deposition. All of the sensitive lakes had pH between 6.1 and 7.0, and most had sum of base cation concentrations between about 25 and 50 µeq/L (Table IX-11). Most were small (< about 7 ha), occurred in watersheds less than about 100 ha in area, and were situated at elevations ranging from about 2,800 m to 3,700 m.

The concentrations of SO_4^{2-} in the most acid-sensitive lakes tended to be relatively low; lakewater SO_4^{2-} concentration ranged between about 3 and 10 µeq/L in most cases. Such concentrations are approximately what would be expected, assuming average SO_4^{2-} concentrations in precipitation of about 3 to 5 µeq/L, negligible dry deposition, and less than 50% evapotranspiration. However, many of the WLS lakes in SEKI, including two of those having low ANC (< 50 µeq/L), had relatively high concentrations of SO_4^{2-} (>> 10 µeq/L), which are likely the result of watershed sources of S. Nitrate concentrations were variable in the acid-sensitive lakes, ranging from near zero to 10 µeq/L.

Clow et al. (1996) presented a comparison of median major ion chemistry from four surface water surveys in the Sierra Nevada (Table IX-12). Water chemistry measured in the various surveys was similar, but granitic watersheds were more acid-sensitive than non-granitic watersheds, and were less likely to exhibit high SO_4^{2-} concentrations suggestive of watershed sources of S.

Atmospheric deposition and stream discharge and chemistry were measured during the period of 1984-1987 in two mixed conifer watersheds in Sequoia National Park by Stohlgren et al. (1991). Over the three year period, SO_4^{2-}, NO_3^-, and Cl^- were the major anions in wet deposition, with volume-weighted average concentrations of 13, 12, and 10 µeq/L, respectively. The two study watersheds, Log Creek and Tharp's Creek, are both located at elevations ranging from about 2,000-2,400 m. The pH values of both streams were typically circumneutral, although pH excursions below 6.0 were found in both streams during snowmelt. ANC values were typically in the range of 150-450 µeq/L. Neither of these streams would be expected to be sensitive to acidification from acidic deposition. Stream SO_4^{2-} concentrations were typically in the range of 2-6 µeq/L in both streams (Stohlgren et al. 1991), suggesting that geologic S sources do not exist in these watersheds.

Table IX-11. Results of lakewater chemistry analyses by the Western Lake Survey for selected variables in SEKI and adjacent areas.

Lake Name	Lake ID	Lake Area (ha)	Watershed Area (ha)	Elevation (m)	pH	ANC (μeq/L)	SO_4^{2-} (μeq/L)	NO_3^- (μeq/L)	Ca^{2+} (μeq/L)	C_B (μeq/L)	DOC (mg/L)	Park
Lakes within SEKI												
(No Name)	4A1-029	4.7	65	3263	6.9	47	3.4	0.2	20	53	1.1	KICA
(No Name)	4A1-033	4.3	93	3251	6.7	28	13.6	2.8	29	42	0.3	KICA
(No Name)	4A1-034	1.7	28	3715	7.0	36	9.7	4.3	33	48	0.3	KICA
(No Name)	4A1-035	12.1	148	3521	6.8	44	37.1	1.6	63	82	0.6	KICA
Horseshoe Lakes (Middle)	4A1-036	3	249	3208	7.2	83	7.3	0.3	49	84	1.0	KICA
Swamp Lakes (Western)	4A1-037	6.7	114	2916	6.8	34	4.1	0.5	21	36	0.7	KICA
(No Name)	4A1-038	2.5	155	3257	7.2	104	10.9	0.3	67	109	2.2	KICA
(No Name)	4A1-040	9.5	119	3355	7.4	80	9.2	0.5	64	81	0.6	KICA
(No Name)	4A1-041	2.1	49	3184	6.9	60	6.4	0.4	46	66	1.9	KICA
(No Name)	4A1-053	5.8	236	3306	7.3	98	5.1	0.8	78	96	0.5	KICA
(No Name)	4A1-056	5.2	88	3666	6.1	15	7.2	10.3	20	33	0.2	KICA
(No Name)	4A1-042	8.4	370	3574	8.0	178	45.7	0.0	178	228	1.5	SEQU
(No Name)	4A1-043	2.6	41	3281	6.5	17	4.9	2.5	15	25	0.7	SEQU
(No Name)	4A1-044	9.4	660	3330	7.5	101	11.2	0.0	82	111	0.8	SEQU
(No Name)	4A1-045	1.8	176	3525	7.1	97	9.3	0.7	70	109	0.7	SEQU
(No Name)	4A1-046	2.6	60	3379	6.9	52	31.1	4.1	64	87	0.4	SEQU
(No Name)	4A1-047	3.8	194	3550	7.1	57	8.3	0.2	34	62	0.7	SEQU
Big Five Lakes (Small N)	4A1-048	3.4	28	3111	7.1	140	4.9	0.2	85	153	2.2	SEQU
Wright Lakes (NW)	4A1-060	2	26	3525	7.3	106	7.3	0.0	72	114	1.4	SEQU
Hockett Lakes (Center)	4A2-045	2.4	85	2598	6.7	69	3.7	0.0	45	95	6.6	SEQU
Lakes Within 25 km of SEKI												
Neil Lake	4A1-023	1.8	91	3240	6.8	46	7.3	8.2	30	60	0.6	
Vee Lake	4A1-024	20.8	383	3404	7.3	67	4.0	0.0	43	68	0.9	
(No Name)	4A1-025	1.5	36	2816	6.7	72	4.4	0.1	38	77	3.1	
Merriam Lake	4A1-026	13.6	624	3334	6.9	50	5.0	0.4	33	52	0.4	

Table IX-11. Continued.

Lake Name	Lake ID	Lake Area (ha)	Watershed Area (ha)	Elevation (m)	pH	ANC (µeq/L)	SO_4^{2-} (µeq/L)	NO_3^- (µeq/L)	Ca^{2+} (µeq/L)	C_B (µeq/L)	DOC (mg/L)	Park
Wahoo Lakes (NW)	4A1-027	3.6	220	3446	7.1	53	5.2	3.5	36	56	0.3	
Heather Lake	4A1-028	3	44	3063	6.8	37	8.9	0.4	26	47	1.3	
Upper Lamarck Lake	4A1-030	16.8	433	3330	7.5	110	6.7	3.3	91	116	0.4	
Big Pine Lakes (Sixth L.)	4A1-031	3.4	168	3382	8.9	238	68.6	0.5	202	310	1.1	
Long Lake	4A1-049	3.5	287	3396	7.3	98	5.9	0.0	43	102	1.6	
Big Pine Lakes (Second L.)	4A1-052	12	1891	3062	7.4	134	32.3	5.1	109	169	0.4	
Horton Lake	4A1-054	7.1	1013	3031	7.4	140	23.8	2.2	116	163	0.6	
Little Lake	4A2-042	4.6	163	2810	6.5	25	4.5	0.1	7	32	1.3	
Little Spanish Lake	4A2-044	1.9	67	2598	6.8	69	2.1	0.1	49	77	4.6	
Duck Lake	4A2-053	1.9	60	2794	6.9	60	2.2	0.2	29	67	3.4	

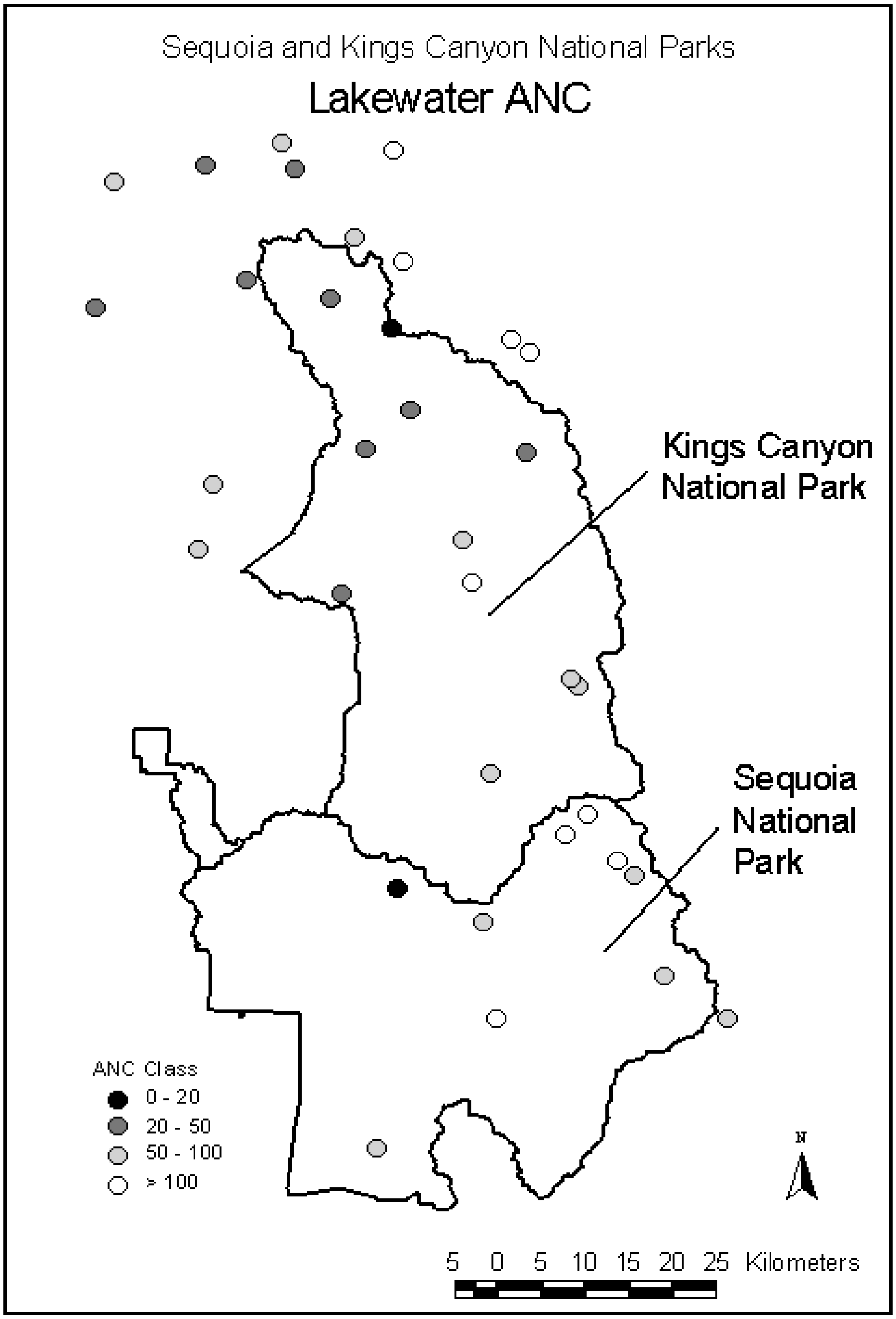


Figure IX-6. Map depicting lakewater ANC in SEKI, based on results of the Western Lakes Survey.

Table IX-12. Comparison of major ion chemistry data from four surface water synoptics in the Sierra Nevada. Median values (first and third quartiles in parentheses) given in µeq/L, except SiO_2 in µmol/L. (Source: Clow et al. 1996)

	Elevation (m)	pH	Ca^{2+}	Mg^{2+}	Na^{+}	K^{+}	Cl^{-}	NO_3^-	SO_4^{2-}	HCO_3^-	SiO_2
Clow et al. (1996), (*n* = 27)	2164	6.8 (6.7-7.0)	39 (34-83)	7 (5-12)	28 (22-59)	4.2 (4-6)	6 (5-18)	2.1 (0.0-3.9)	10 (7-12)	42 (36-64)	56 (39-84)
Melack et al. (1985), granitic (*n* = 44)	3389	6.9 (6.4-7.2)	37 (22-54)	5 (3-7)	16 (10-24)	6 (4-15)	8 (5-16)	0.4 (0.1-2.4)	13 (9-20)	44 (25-65)	
Melack et al. (1985), non-granitic (*n* = 29)	3121	7.2 (7.0-7.8)	70 (34-308)	10 (5-25)	26 (18-48)	12 (7-20)	12 (8-21)	0.2 (0.1-0.9)	31 (18-91)	71 (41-258)	
WLS (population) (*N* = 2119)	3008	6.9 (6.6-7.3)	43 (29-104)	6 (3-9)	19 (11-21)	4	2 (1-3)	0.4 (0.1-1.7)	7 (4-7)	60 (37-130)	33 (22-60)

A chemical survey of 101 high-altitude lakes in 7 national parks in the western United States was conducted by the USGS during the fall of 1999; 72 of the lakes were previously sampled during the fall of 1985 as part of the WLS (Clow et al. 2000). A strong effort was made to use methods and protocols similar to those used during the WLS. The objective of the 1999 lake survey was to provide information on water quality in the parks and assess whether there were significant differences in lake chemistry in the 1985 and 1999 data sets. Lakes in the three California parks (SEKI, YOSE, LAVO) and in Rocky Mountain National Park (Colorado) were extremely dilute; median specific conductances were ≤ 12 µS/cm and median alkalinities were ≤ 75 µeq/L. Specific conductances and alkalinities generally were substantially higher in Grand Teton and Yellowstone National Parks (Wyoming), and Glacier National Park (Montana), probably due in part to the prevalence of more reactive bedrock types. Concentrations of base cations and ANC were lowest in lakes in the alpine zone, probably because of minimal vegetation and soil development, and because of fast hydrologic flow rates. These conditions make alpine lakes highly sensitive to atmospheric deposition of pollutants. This is evidenced by relatively high NO_3^- concentrations in high-elevation lakes in Rocky Mountain National Park (0 to 29 µeq/L), which are subject to moderate levels of N deposition (3 to 5 kg N/ha/yr; Figure IV-5; Clow et al. 2000).

This study sampled 24 lakes in SEKI, including all of the WLS lakes. Five lakes had ANC < 30 µeq/L, with the lowest being 7 µeq/L. That same lake (4A1-056) was the only one having relatively high NO_3^- concentration (9 µeq/L). Eleven lakes had ANC < 50 µeq/L (Clow et al., USGS, Denver, per. comm.).

One challenge that will need to be addressed is separating effects of trends in water quality from variations due to differences in hydroclimatic conditions. A qualitative evaluation of the effects of climatic variations will be done by looking at variations in water chemistry and climate at several intensively monitored research watersheds in the Sierra Nevada and Rocky Mountains. This research is ongoing (D.W. Clow, USGS, Denver, pers. comm.).

Thirty lakes were selected at random for study by Jenkins et al. (1994) from the subset of Sierra lakes above approximately 2,400 m elevation. The lakes were selected using EPA's EMAP area-based sampling technique which allowed extrapolations to be performed to a population of 1,404 lakes that are greater than 1 ha in surface area. The lakes were sampled for their water chemistry and to assess populations of fish and macroinvertebrates that were present in the lakes and their associated streams. The results of water chemistry analyses were generally

similar to those found in the WLS, although lakes in the survey of Jenkins et al. (1994) were higher in elevation and smaller in surface area overall as compared with the WLS lakes. Median pH and ANC values were slightly lower than the medians reported in the WLS, and NO_3^- concentrations were higher (Table IX-13). Several lakes at the southeastern border of Kings Canyon National Park were acidic from watershed sources of S. These lakes had high SO_4^{2-} and Al concentrations. In the lakes represented by the study, 8% had pH less than 6. Frequency distributions for selected chemical parameters are shown in Figure IX-7.

Table IX-13. Comparison of median values for some chemical measurements from the study of Jenkins et al. (1994) and from the Western Lake Survey (Landers et al. 1987). (Source: Jenkins et al. 1994).

Parameter	WLS	Jenkins et al. (1994) study
N	2119	1404
pH	6.93	6
ANC (µeq/L)	60	51
Ca^{2+} (µeq/L)	43	55
Na^+ (µeq/L)	19	17
Mg^{2+} (µeq/L)	6	5
SBC (µeq/L)	76	93
Cl^- (µeq/L)	2	3
NO_3^- (µeq/L)	0.4	2
SO_4^{2-} (µeq/L)	7	8
Elevation (m)	3008	3292

Jenkins et al. (1994) divided their study lakes into categories (A through D) based on concentrations of SO_4^{2-}, ANC, and the relationship between (Ca^{2+} + Na^+) and ANC. Lakes with Ca^{2+} and Na^+ as the dominant cations were classified as category A lakes. Category B and C lakes were high in Ca^{2+} and SO_4^{2-}. Category B lakes were moderate in ANC, whereas Category C lakes were low in pH and ANC relative to the concentrations of other ions. Category C lakes were located in basins that contained roof pendant metamorphics and they appeared to be restricted to an area about 10 km in diameter at the eastern edge of Kings Canyon National Park. In lakes 8 and 45, ANC was near or below 0. The SO_4^{2-} concentration was very high. Category D lakes were those with high Mg^{2+}, K^+, Ca^{2+}, and Na^+ concentrations, and low concentrations of SO_4^{2-}. All four category D lakes were located in areas dominated by volcanic soils.

<u>Process Studies</u>

During the 1980s, an Integrated Watershed Study (IWS) was conducted at the Emerald Lake watershed (2,800 to 3,400 m elevation) in Sequoia National Park, the purpose of which was to investigate the possibility of acid-induced damage to the watershed and to determine the consequences of acidification on Sierran surface waters (Tonnessen 1991). The IWS included studies of deposition, terrestrial systems and aquatic systems.

Emerald Lake is granitic and composed mainly of granodiorites with some mafic inclusions, aplite dikes, and pegamite veins (Sisson and Moore 1984). The soils in the Emerald Lake watershed provide a large capacity for neutralization of incoming acidity. However, a

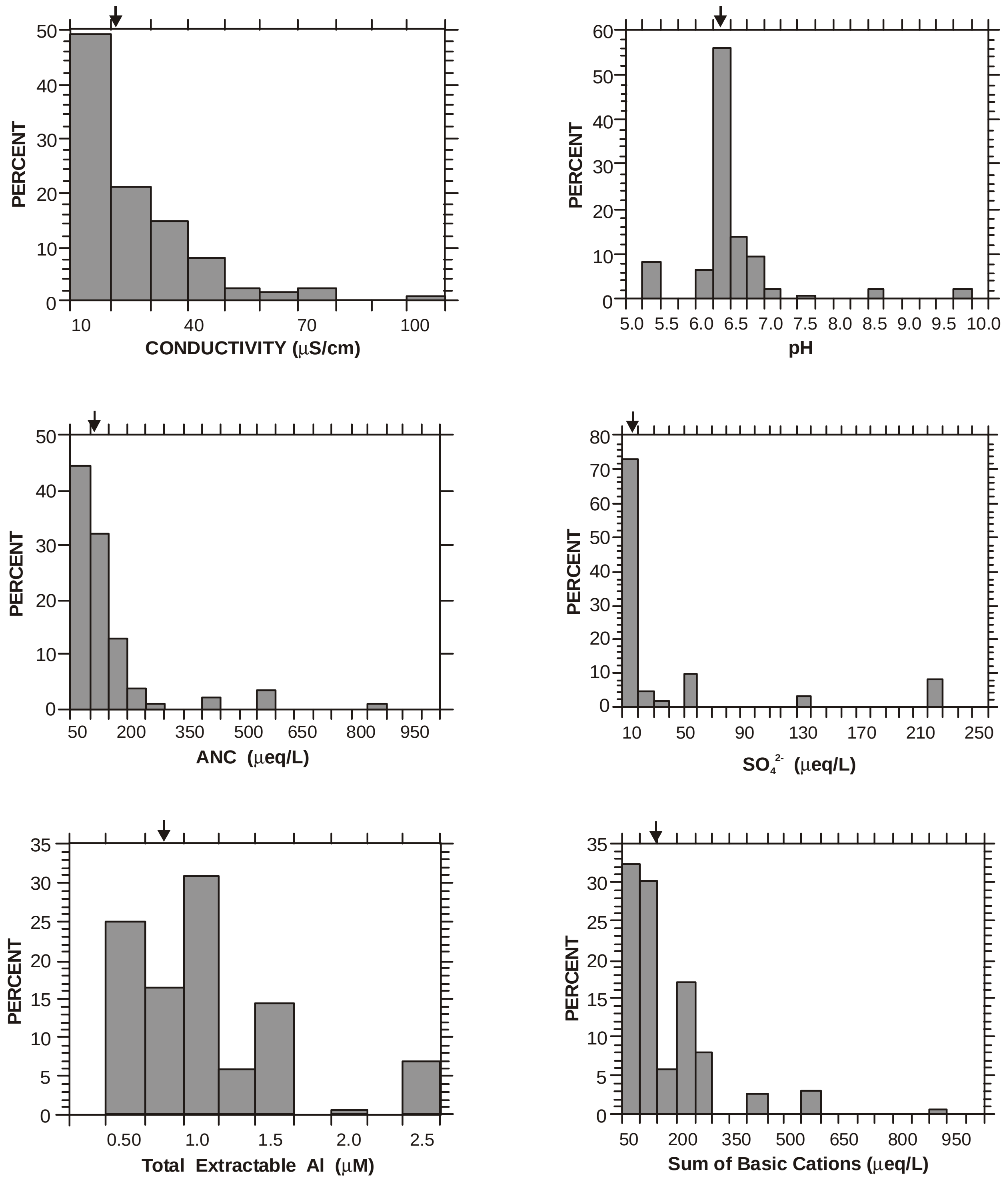


Figure IX-7. Frequency distributions for some chemical characteristics of the target population of lakes in the study of Jenkins et al. (1994). Arrows indicate median values.

significant fraction of the snowmelt and rainfall in the watershed has insufficient contact with soils to allow that neutralization to occur.

Emerald Lake does not represent an extreme case of acid sensitivity relative to the Sierra Nevada in general. About 15% of the lakes included in EPA's Western Lakes Survey had lower ANC and about 30% had flashier hydrology (as reflected by the watershed area to lake area ratio).

Williams and Melack (1989) examined variation in the onset of snowmelt in portions of the Emerald Lake watershed and effects on the magnitude and timing of ionic pulses in the various sub-basins. Source areas of snowmelt runoff varied spatially and temporally in response to slope and aspect. Concentrations of NO_3^- and SO_4^{2-} in streams were correlated with the amount of snowmelt in each sub-basin. Inflows to Emerald Lake had elevated concentrations of NO_3^- (~18 μeq/L) and SO_4^{2-} (~9 μeq/L) at initiation of snowmelt, and then decreased as snowmelt progressed. The onset of snowmelt, and accompanying ionic pulses in streamwater, shifted from sub-basins with southwesterly aspect to those with northerly aspect (Williams and Melack 1989).

Kattelmann and Elder (1991) developed a water balance for two years for the Emerald Lake watershed, which provides insight into the hydrology of headwater catchments in the Sierra Nevada. Snow dominated the water balance and accounted for 95% of the precipitation. Direct short-term runoff from snowmelt accounted for more than 80% of the streamflow in both years. Additional water balance data for Emerald Lake were provided by Melack et al. (1998).

Snowmelt typically dilutes lake outflow solute concentrations in the Sierra Nevada by 30 to 40%, as measured by decreases in Na^+, Cl^-, or silica (Melack et al. 1998). In contrast, SO_4^{2-} concentrations are only reduced by about 10%. Relative to dilution, episodic SO_4^{2-} concentrations showed an increase of 128 to 150% (Melack et al. 1998). Nitrate is also relatively increased in comparison with other ions in a similar fashion.

Williams et al. (1993) utilized hydrologic, mineralogic and soils data to determine the sources and geochemical controls on the composition of surface waters in the Emerald Lake watershed. Preferential weathering of the anorthite component of plagioclase accounted for about 60% of the annual export of Ca^{2+} from the watershed in streamwater. There was a large excess of Si after accounting for weathering processes of granitic bedrock, consistent with the results from other alpine basins underlain by crystalline bedrock (e.g., Mast et al. 1990). This underscores the lack of adequate understanding of the sources and sinks of Si in granitic basins.

Temporal variation in the geochemical controls on solute composition of surface waters were attributed by Williams et al. (1993) to a combination of changes in hydrologic routing, relative contributions of water from different hydrologic reservoirs, and residence time within these reservoirs. Stoichiometric weathering products of surface waters were not consistent over time and were divided into three distinct periods:

- snowpack runoff,
- summer transition period, and
- low flow period from late summer through winter.

About half of the snowpack runoff was estimated to become streamflow as Hortonian overland flow and the other half infiltrated soils and talus before becoming streamflow. Applied tracers illustrated that hydrologic residence time in subsurface reservoirs was on the order of hours to days during snowpack runoff. Streamwater was not in equilibrium with weathering products during snowmelt.

Discharge from the soil reservoirs appeared to be the primary source of streamflow during the summer transition period. In contrast, processes below the soil zone appeared most important during the low flow period (Williams et al. 1993).

Despite the short residence time of meltwater in the soil compartment during snowmelt, acid neutralization via cation exchange appeared to be important. Improved understanding of cation exchange processes in both soil and talus compartments of these watersheds is needed. It will provide the foundation for further refinement of models to predict the response of these systems to future changes in acidic deposition (Sullivan 2000). Acid neutralization via base cation flux from lake sediments does not seem to be important in high-elevation Sierra Nevada lake watersheds. Michaels (1989) studied the solid phase and interstitial water of the sediments of Eastern Brook Lake, Emerald Lake, and Mosquito Lake. Annual average base cation fluxes from deep sediments were roughly 10 neq/cm^2/day, which represented less than 2% of the total watershed sources of alkalinity.

Fluxes and transformations of N were studied from 1985 to 1987 at Emerald Lake watershed and reported by Williams et al. (1995). The results of this study indicated that up to 90% of annual wet N deposition was stored in the seasonal snowpack. NO_3^- and NH_4^+ were released from storage as an ionic pulse, with the first fractions of meltwater having concentrations of NO_3^- and NH_4^+ as high as 28 µeq/L, compared with bulk snow concentrations < 5 µeq/L. The soil reservoir of organic N (81 keq/ha) was much greater than N storage in litter and biomass (12 keq/ha). Assimilation of N by vegetation was balanced by the release of N from soil mineralization, nitrification, and litter decay. Mineralization and nitrification processes in the watershed produced 1.1 keq/ha/yr of inorganic N, which represented 3.5 times the atmospheric N loading. During early snowmelt runoff, streamwater NO_3^- concentrations reached their maximum levels (20 µeq/L). During the growing season, streamwater NO_3^- concentrations were near zero (Williams et al. 1995).

Focus shifted in the late 1980s from the IWS at Emerald Lake to a larger group of watersheds. Research on the catchments of Pear, Topaz, Crystal, and Ruby Lakes was initiated in 1986 (Sickman and Melack 1989). Additional lakes were added to the research and monitoring program in 1990 (Melack et al. 1993).

Sickman and Melack (1989) reported results of two years of study at Pear, Topaz, Crystal, and Ruby Lakes, the first two of which are located in Sequoia National Park. Snow chemistry was very similar among the four watersheds, with pH ranging from 5.3 to 5.5. Most H^+ was deposited by winter snow, but acid anion loading (NO_3^-, SO_4^{2-}) was higher during the summer to autumn period than it was during winter in the Pear Lake and Topaz Lake watersheds because of high concentration of these solutes in rainfall. Ruby Lake is located in the eastern escarpment of the Sierra Nevada in the John Muir Wilderness. It is the largest (12.6 ha) and deepest (35 m) of the lakes and lies at the highest elevation (3,426 m); it also has the largest watershed (424 ha). The watershed contains a large amount of mineralized granite and a glacier in the upper cirque. Crystal Lake is located in the eastern Sierra Nevada near Mammoth Lakes. It is the smallest lake (4 ha) and relatively shallow (14 m).

Topaz Lake, in Sequoia National Park, is 5.2 ha in area and is located about 6 km NNW of Emerald Lake at 3,219 m elevation. It is the shallowest of the lakes (5 m) and smallest in volume. The entire watershed is granitic and is situated above treeline in the alpine zone. Pear Lake, located 0.5 km from Emerald Lake, in Sequoia National Park, has a surface area of 8 ha and depth of 27 m. It lies at 2,904 m elevation in an entirely granitic watershed.

The four lakes were sampled approximately bimonthly from October, 1986 through June, 1988 by Sickman and Melack (1989) and spring snowpack samples were obtained. The extent of vertical mixing varied from lake to lake. The deeper lakes, Pear and Ruby, were strongly stratified during both winter and summer. Summer surface temperatures in Pear Lake were more than 12 ºC higher than bottom water. The latter showed oxygen depletion below about 18 m. Anoxic conditions were associated with high ANC, which ranged from ~100 µeq/L during the

summer of 1987 to ~600 µeq/L during February, 1988. Ruby Lake also had low dissolved oxygen concentrations in deep waters, but little or no increase in ANC was observed. Topaz Lake was sufficiently shallow that summer stratification did not occur and winter stratification was weak. ANC patterns showed minima during snowmelt and maxima under ice. Crystal Lake showed slight oxygen depletion in winter and ANC levels of bottom waters about 25 µeq/L higher than surface waters in winter. ANC concentrations were homogenous during summer. The entirely granitic basins (Topaz and Pear) contributed less ANC per ha to their representative lakes than did the watersheds with some volcanic rocks (Crystal) or with a glacier and more readily weatherable rocks (Ruby; Sickman and Melack 1989).

The eight water quality research and monitoring sites reported by Melack et al. (1993) and Sickman and Melack (1998) were located in alpine and subalpine settings across a majority of the north-south extent of the Sierra Nevada. Four were located on the western slope, all within Sequoia National Park (Emerald, Pear, and Topaz Lakes, Marble Fork of the Kaweah River). The other four were located to the north, and along the eastern slope of the Sierra Nevada range.

Melack et al. (1993) reported two years of intensive research at the seven high-elevation lakes (excluding the Marble Fork of the Kaweah River). Solute concentrations, particularly ANC and base cation concentrations, were greatest during winter, declined to minima during snowmelt, and gradually increased during summer and autumn. SO_4^{2-} concentrations varied most in lakes with lowest volumes. Nitrate concentrations increased during snowmelt in most lakes due to inputs of streamwater enriched with NO_3^-. They conducted chemical and biological sampling and also made hydrological measurements at regular intervals throughout a two-year period with most intensive study during snowmelt. Results of measurements of selected water chemistry parameters are presented in Figure IX-8. Zooplankton species known to be intolerant of acidification were found in all seven lakes, and Melack et al. (1993) concluded that their presence is evidence that Sierra Nevada lakes are not currently showing chronic biological effects of acidic deposition. Results of these studies were summarized by Melack et al. (1998).

The volume-weighted mean pH for lake outlet streamflow during the 36 water years of record examined by Melack et al. (1998) for seven lakes in the Sierra Nevada was 6.05, and ranged from 5.6 to 6.7. Three lakes (including Pear and Emerald in Sequoia National Park) had volume-weighted mean ANC in the range of 15 to 30 µeq/L and were classified by Melack et al. (1998) as low in ANC. Moderate ANC waters (including the Marble Fork) exhibited mean ANC in the range of 30 to 50 µeq/L.

Sulfate concentrations were most consistent of the ions measured. With the exception of Ruby and Spuller Lakes, annual average SO_4^{2-} concentration ranged from 5 to 7 µeq/L. A paired catchment study by Brown et al. (1997) near Pear Lake demonstrated that soils can be important in regulating the chemical composition of surface runoff. The authors found significant relationships between SO_4^{2-} and base cation concentrations and between SO_4^{2-} and Si concentrations in runoff, and were able to reconcile Ca^{2+} residuals in stoichometric weathering reactions by subtracting Ca^{2+} in the equivalent amount of SO_4^{2-} present. Such findings suggest that SO_4^{2-} is partially a product of weathering processes in these watersheds (Brown et al. 1997).

Some water quality research has also been conducted on lower-elevation streams in SEKI, which tend to be much less sensitive to potential acidification from N and S deposition than are the alpine and subalpine surface waters. Williams and Melack (1997a) reported solute concentrations in the streamwater of Log Creek in 1984-1995, a period which included a six-year drought. Altered climatic conditions have been shown to affect biogeochemical processes, and can increase solute concentrations in streamwater during drought periods (Swank and Waide 1988, Bayley et al. 1992b). Drier climatic conditions can also, however, lower the water table and result in the oxidation of stored sulfides, which in some cases can cause acidification of

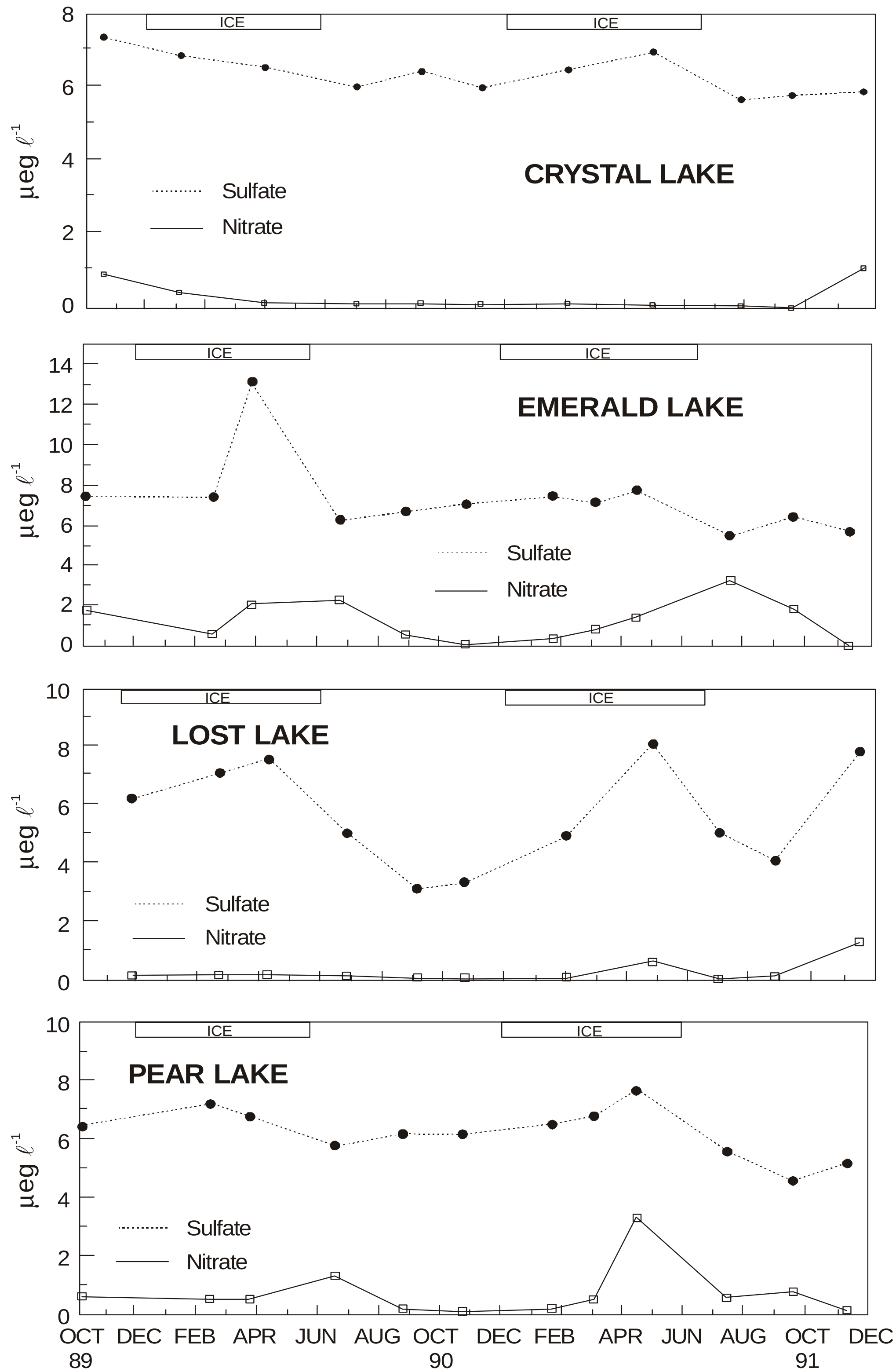
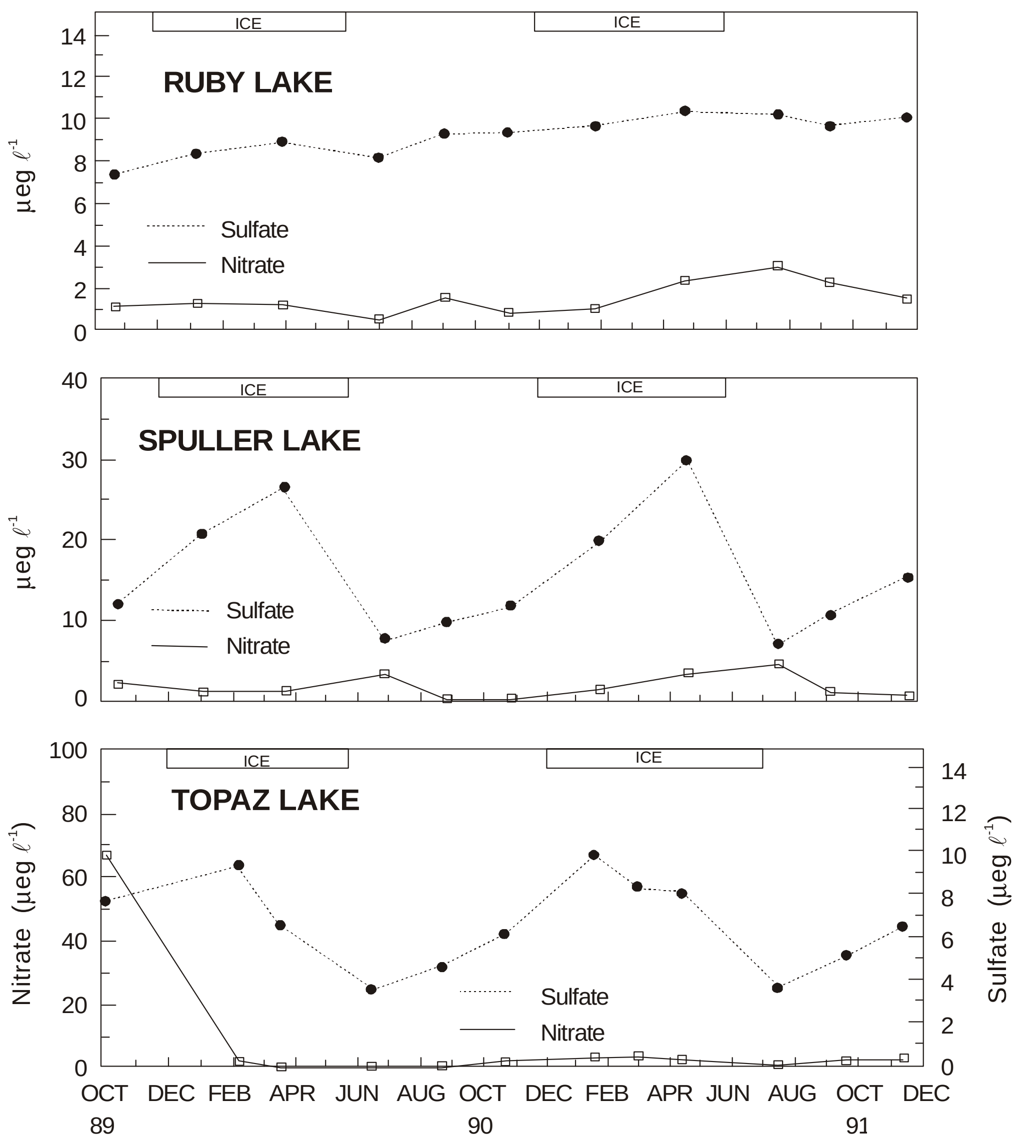


Figure IX-8. Volume-weighted mean lake chemistry for the period October, 1989 to November, 1991 in 7 high-elevation Sierra Nevada lakes. Emerald, Pear, and Topaz Lakes are located in SEKI. (Source: Melack et al. 1993)



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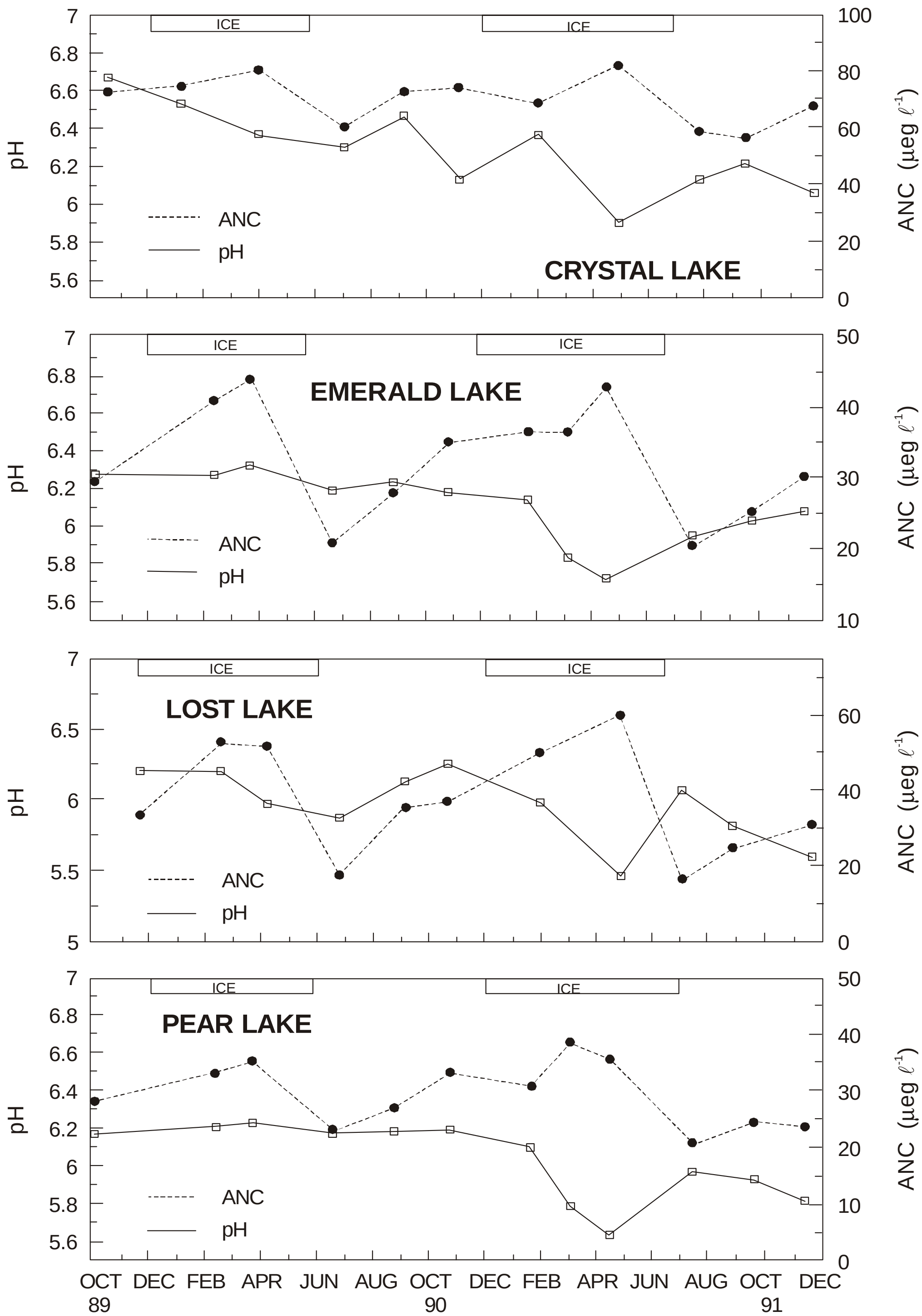

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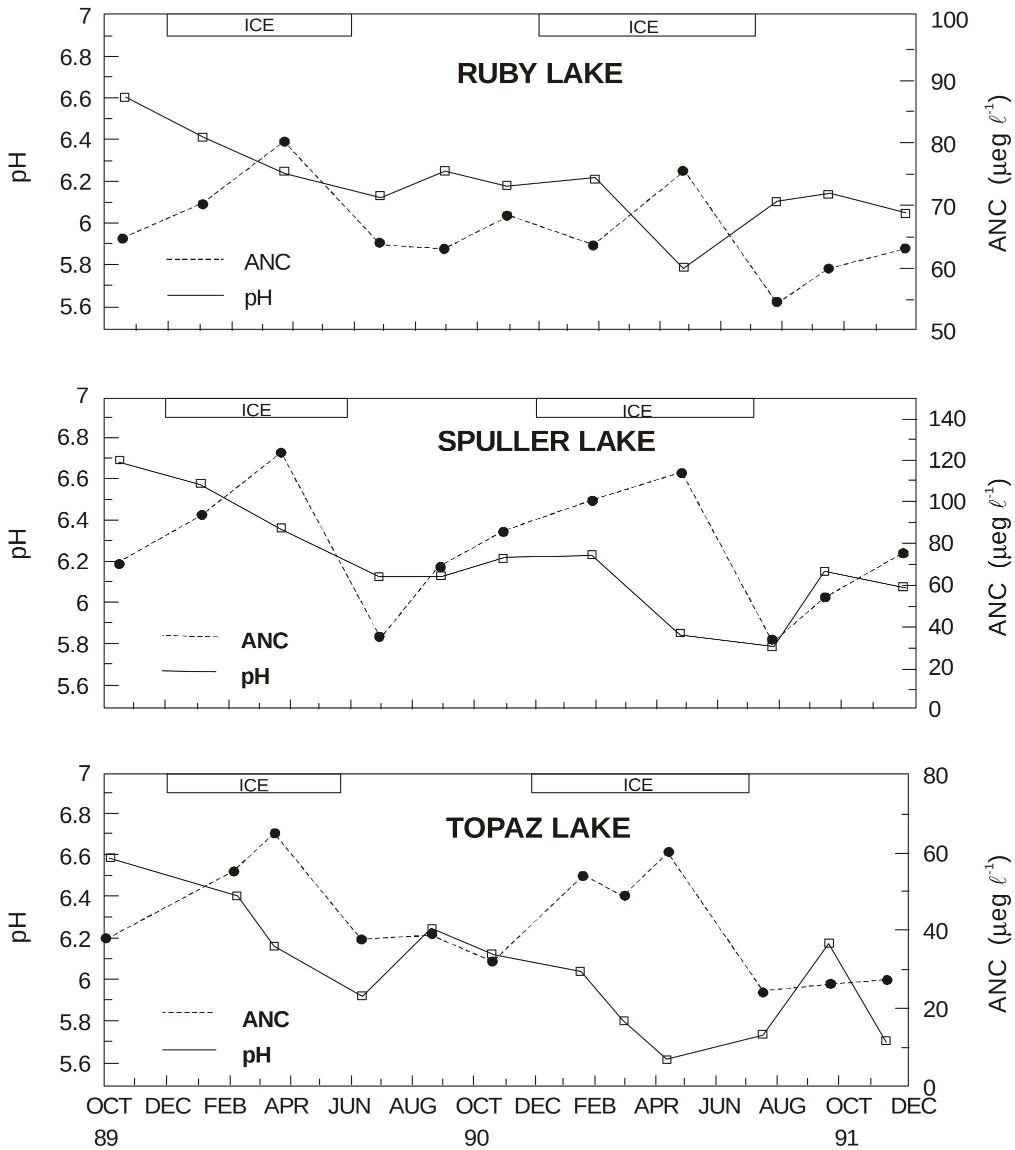

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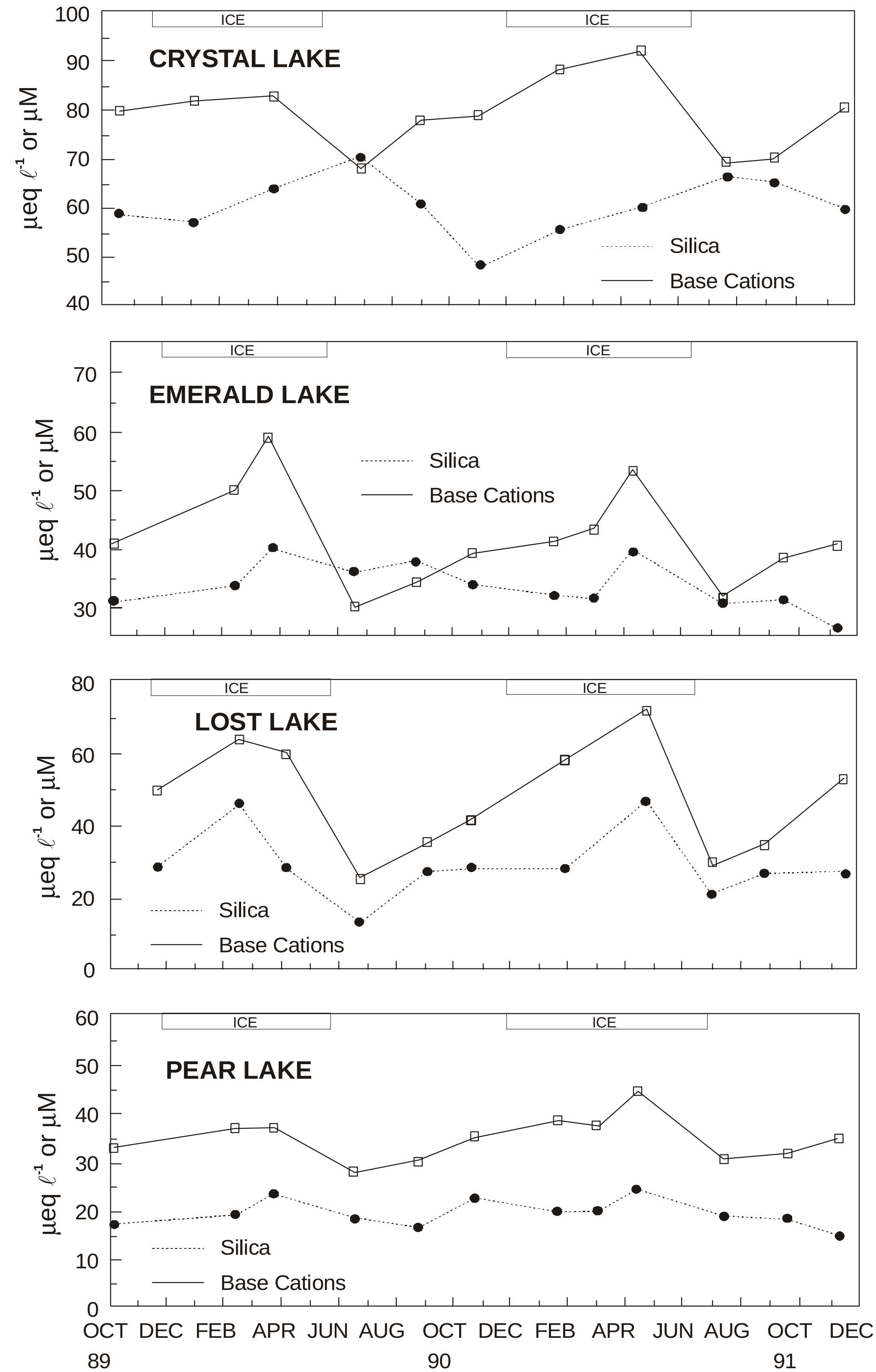

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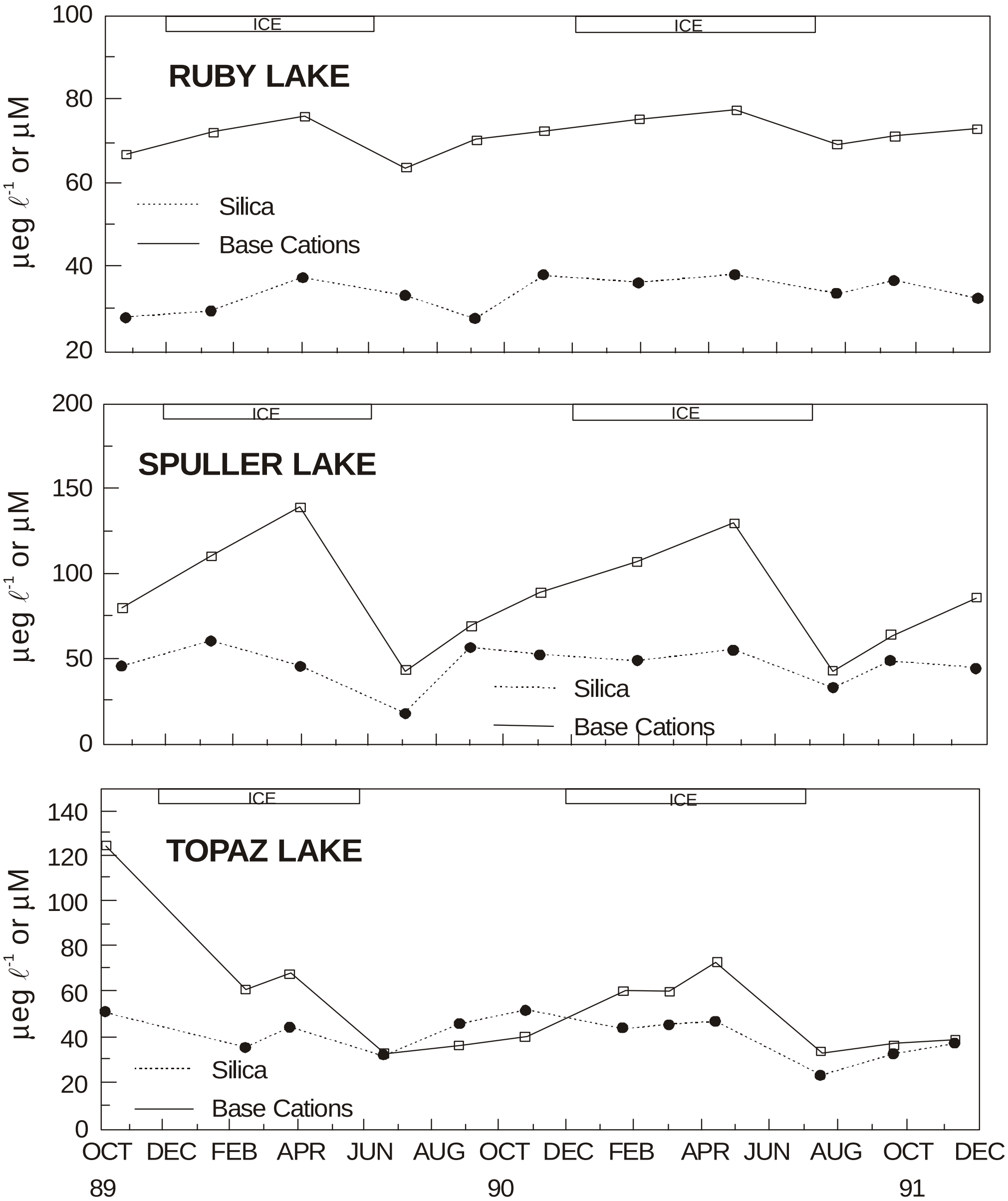

Figure IX-8. Continued.

surface waters (Bayley and Schindler 1987). Drought was bracketed by a wet period of three years at the Log Creek watershed, both before and after the six-year drought. The drought resulted in quantities of precipitation as low as 53% (during 1987) of the long-term average for the Giant Forest area, which is 110 cm.

Only the volume-weighted mean concentration of Ca^{2+} was significantly higher ($p < 0.01$) during the drought period as compared with the other six years of the study using a t-test. Although the data suggested that there may have been a concentrating effect for Ca^{2+}, Na^{+}, and SO_4^{2-} in Log Creek, concentration increases were inconsistent among solutes and years and may not have been entirely related to drought conditions. For example, SiO_2 concentration decreased, presumably a result of suppressed weathering related to reduced soil moisture during the drought. However, the concentration of SiO_2 continued to decline after the drought had ended in 1993. The observed lack of a strong correlation between precipitation amount and streamwater solute concentrations suggested that the regulatory mechanisms influencing solute concentrations in streamwater were not strongly affected by the drought (Williams and Melack 1997a).

Streamwater solute concentrations in Tharp's Creek during the period 1984-1995 were reported by Williams and Melack (1997a). The study documented the effects of prescribed burning in the Tharp's Creek catchment, which significantly increased the concentrations of most solutes in streamwater. In the first year after the prescribed burning, volume-weighted mean concentrations of all the base cations and acid anions increased significantly and the ANC more than doubled. Volume-weighted mean concentrations of SO_4^{2-} and NO_3^- in the stream were about 16- and 2,000-fold above pre-burn baselines, respectively, during the first year after burning. All solutes remained above pre-disturbance conditions at the end of the third year after burning. Large pulses in NO_3^- concentrations in streamwater were observed after burning, which was attributed by Williams and Melack (1997a) to increased organic matter mineralization and subsequent rapid nitrification in the catchment. Streamwater also showed a large increase in NH_4^+ concentration (about 25 µeq/L) immediately after burning. Ammonium decreased to the pre-burn baseline several months later. Concentrations of NO_3^- exceeded 60 µeq/L in January, 1991, the first month of streamwater runoff after the burn, and in March of that year NO_3^- concentrations exceeded 140 µeq/L. During the following spring, 1992, NO_3^- concentrations increased above 120 µeq/L and persisted above 60 µeq/L for several weeks. In contrast, the pre-fire concentrations seldom exceeded 0.5 µeq/L (Williams and Melack 1997a). Gross NO_3^- export in the Tharp's Creek watershed increased from an annual average of less than 0.03 kg/ha/yr during the seven-year pre-burn period to about 1.6 kg/ha/yr during the first three years after the burn (Williams and Melack 1997b). Volume-weighted mean concentrations of both Ca^{2+} and Mg^{2+} in Tharp's Creek increased by a factor of about three during the first year after the burn, compared to pre-fire baseline conditions, whereas K^+ increased by about a factor of two and Na^+ increased by a factor slightly greater than one.

Disturbances such as forest fire can alter the timing, magnitude, and form of elemental fluxes to receiving waters (Likens et al. 1970, Vitousek et al. 1979, Williams and Melack 1997a). Effects of fire can include erosion, solute leaching, alkalinity reduction or generation, and nitrification of soils (Schindler et al. 1980; Bayley and Schindler 1987; Bayley et al. 1992a,b; Chorover et al.1994; Williams and Melack 1997a). Prescribed burning can be used to moderate the intensity of wildfire and therefore mitigate the effects of wildfire on the quality of receiving waters (Richter et al. 1982). Other disturbances such as wind storms, clear cutting, and slash burning generally increase solute losses in forested catchments (Schindler et al. 1980; Bayley et al. 1992a,b). Burning of the Tharp's Creek catchment killed most of the younger trees and the understory vegetation, whereas the larger trees were scarred but left alive. Most forest

litter was combusted in the fire and an ash layer was left throughout the catchment, which remained in the litter zone of the forest floor. The observed increase in streamwater ANC reported by Williams and Melack (1997a) contrasted with the results reported by Likens et al. (1970) and Bayley et al. (1992b). Likens et al. (1970) reported acidification of streamwater due to nitrification and increased NO_3^- export, which counteracted the increased base cation export after clear cutting. Bayley et al. (1992b) reported a decrease in ANC and an increase in streamwater H^+ concentration subsequent to forest fire.

Monitoring

Several studies have been conducted in SEKI to determine the extent to which lakewater chemistry has changed over time. These have included the resurvey of WLS lakes by Clow et al. (2000), for which analyses are on-going, and also a resurvey in the early 1980s of a group of lakes originally sampled in 1963 (Bradford et al. 1983). In addition, long-term monitoring has been conducted for eight high-elevation surface waters, several of which are located in SEKI (Sickman and Melack 1998).

Water samples were collected by helicopter from 170 lakes in SEKI and YOSE in September 1965. pH values ranged from 4.7 to 7.3, with an arithmetic mean and median both equal to pH 6.0. Specific conductance measurements ranged from 2.4 to 110 µS/cm, with an arithmetic mean of 8.0 and a median of 7.0 (Bradford et al. 1968). A follow-up study was conducted by randomly sampling 124 lakes in July 1980, 1981, 1982, and 1983 during the spring thaw, and also in October 1980 before the winter freeze (Bradford et al. 1983). The results suggested that lake water acidity, total electrolyte concentration, and the concentrations of a number of trace elements did not increase between 1965 and 1983. Median values for lakewater SO_4^{2-} in July 1980, October 1980, and July 1981 were 8.3, 8.8, and 11.0, respectively. Similarly, the median concentrations of lakewater NO_3^- were 7.1, 2.9, and 5.7, respectively. Bradford et al. (1983) also sampled feeder streams and snow. These samples were analyzed for pH, conductivity, and bicarbonate concentration. Median pH value of the feeder streams was equal to that of the lakewater at pH 6.4. In contrast snowmelt water had a median pH of 5.6.

Similarly, no long-term trends in the pH or ANC of surface water were identified for the eight waters studied by Melack et al. (1998). This was despite the fact that one lake (Emerald) had 12 years of monitoring data. Melack et al. (1998) reported volume-weighted average concentrations of major ions in the Emerald Lake outflow for the period 1983 to 1994. No trend was found in either pH or ANC during this time period, although NO_3^- concentrations declined after about 1990. The cause of the decline in NO_3^- in Emerald Lake was not clear. Peak NO_3^- values in the lake outlet during 1983 through 1987 were above 10 µeq/L in nearly all years, but only 8 to 9 µeq/L during water years 1990 through 1994.

Sickman and Melack (1998) reported the results of long-term monitoring of the water chemistry of the seven high elevation Sierra Nevada lakes and catchments during the period 1983-1996. They described temporal variations in the concentrations of NO_3^- and SO_4^{2-} in the lakewaters. Long-term trends in water quality were only evident in two of the lakes: an increase in SO_4^{2-} concentration in Ruby Lake and a lowering of the annual maxima and minima of NO_3^- concentrations in Emerald Lake. From October, 1987 through April, 1994 the concentrations of SO_4^{2-} in Ruby Lake increased from about 6 µeq/L to about 12 µeq/L. In Emerald Lake, the NO_3^- maxima declined by about 25-50% during the period of study. The lower concentrations of NO_3^- in Emerald Lake during the spring and summer seasons are likely a result of greater retention of N in the terrestrial portions of the catchment, perhaps in response to the 1987-1992 drought (Melack et al. 1998). There is no definitive explanation for the increase in N retention in Emerald Lake watershed in the 1990s. However, recent studies in forested watersheds in the

northeastern United States (Mitchell et al. 1996) and high-elevation catchments of the Alps (Sommaruga-Wögrath et al. 1997) suggest that climate may be an important factor affecting N cycling within high-elevation catchments. In particular, the drought of 1987 to 1992 may have increased N uptake in the Emerald Lake watershed by lengthening the growing season. Snowpacks were shallower and melted faster than in the relatively wet years of 1983-1986 (Sickman and Melack 1998). Williams and Melack (1997a) also measured a decline in NO_3^- concentration in Log Creek which was contemporaneous with the changes that were measured in Emerald Lake. Log Creek drains a small mixed conifer catchment at 2,067 m elevation within 15 km of the Emerald Lake watershed. The correspondence of temporal changes in NO_3^- concentrations between Emerald Lake and Log Creek supports the hypothesis that N dynamics in these watersheds are susceptible to climatic factors (Sickman and Melack 1998). Drought conditions may also have been responsible for increasing the proportion of runoff derived from shallow groundwater in the Ruby Lake basin as evidenced by the increased SO_4^{2-} concentrations.

Topaz Lake is located in a region of Sequoia National Park known as the table lands, about 6 km NNW of Emerald Lake. The geology of the watershed is dominated by fine grained granodiorite containing abundant mafic inclusions. Because of the low relief around the lake, it tends to expand during snowmelt and floods the meadow, forming a shallow bay. As the summer progresses, the lake level declines and the water retreats from the bay (Melack et al. 1993). Topaz Lake showed slightly elevated NO_3^- concentrations, in the range of 2 to 8 μeq/L, during late summer and autumn of both 1991 and 1993. Sickman and Melack (1998) interpreted these data as an indication that biological processes are largely responsible for the accumulation and release of NO_3^- during periods of low runoff. The shallow lake and extensive seasonally flooded meadow at Topaz Lake was believed to have accentuated the influence of these biological processes. In the other lakes, NO_3^- was exported only during the non-growing season, primarily during snowmelt. However, given the short growing season of the vegetation in these watersheds and the possibility that much of the NO_3^- exported during snowmelt was derived from catchment sources (Melack et al. 1998, Kendall et al. 1995), Sickman and Melack (1998) concluded that the N export during the non-growing season probably represented a natural, undisturbed condition for most of these watersheds.

Thus, available resurvey and monitoring data do not suggest changes in surface water chemistry in SEKI, or other high-elevation Sierra Nevada watersheds, that could be attributable to N and S deposition. There is a good database extending back to the 1980s for several acid-sensitive lakes, and this will provide an important benchmark against which potential future changes in chemistry could be evaluated. Because SEKI receives moderate levels of N deposition and contains many lakes that are highly acid-sensitive, it is extremely important that water quality monitoring continues. Without a continued long-term monitoring record, it will be difficult or impossible to determine if and when lakewater acidification occurs or the extent of such acidification. At present, funding for continued monitoring of water quality in SEKI is not assured (J. Melack, UC Santa Barbara, pers. comm.).

Seasonality and Episodic Acidification

The hydrologic cycle in the Sierra Nevada is dominated by snowfall and snowmelt, with over 90% of the annual precipitation falling as snow between November and April. Through the process of preferential elution (Johannessen and Henriksen 1978), the relatively small loads of acidic deposition in Sierran snowpacks can supply relatively high concentrations of SO_4^{2-} and NO_3^- during the early part of snowmelt (Stoddard 1995). In most cases, lakewater pH and ANC decrease with increasing runoff, reaching minima near peak snowmelt discharge (Figure IX-9; Melack and Sickman 1995). Most other solutes exhibit temporal patterns identified by Melack

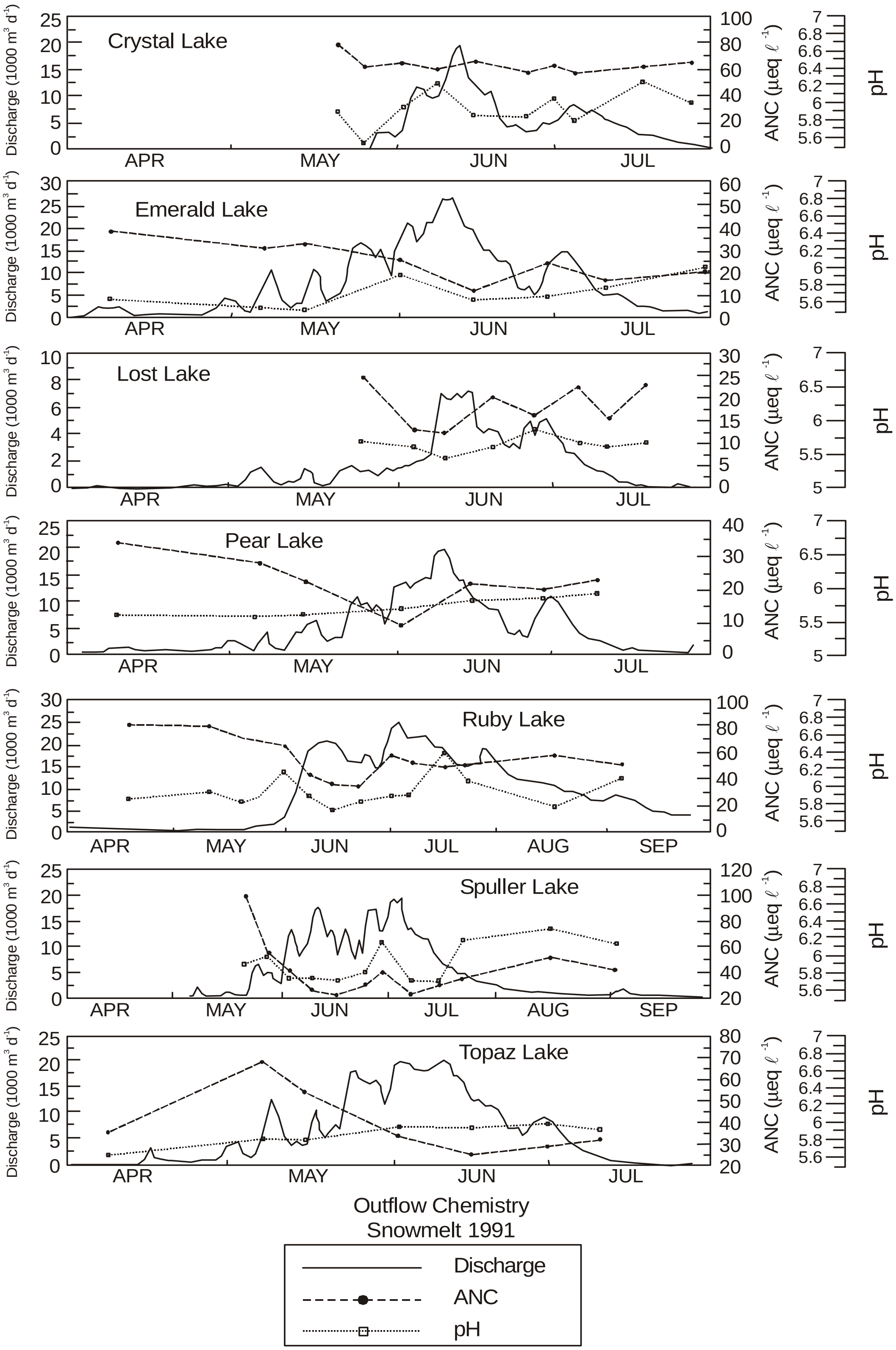

Figure IX-9. Outflow discharge, pH, and ANC during snowmelt in 1991 at 7 high-elevation Sierra Nevada lakes. (Source: Melack and Sickman 1995)

et al. (1998) either as dilution, or a pulse of increased concentration followed by dilution (pulse/dilution) or biological uptake (pulse/depletion). Nitrate and SO_4^{2-} concentrations often declined at peak runoff. Nitrate peaks of 5 to 15 µeq/L were common, although they were usually less than 2 µeq/L in the N-limited lakes (Chrystal and Lost Lakes). Patterns of change in SO_4^{2-} concentration were similar to NO_3^- patterns but much smaller in magnitude. Except in watersheds thought to have significant bedrock sources of S (Spuller and Ruby Lakes), the differences between SO_4^{2-} maxima and minima were generally within a few µeq/L (c.f., Figure IX-10).

The concentrations of base cations and ANC generally exhibited a dilution pattern and reached minima near peak runoff. Outflow ANC declined by 24 to 80% during the spring, with an average decline of 50%. Lowest ANC was generally about 15 µeq/L (e.g., Pear Lake). Seasonal ANC depressions were greatest during years with deep snowpacks and high snowmelt runoff.

In some catchments, NO_3^- concentration declined throughout the snowmelt period (dilution). A second pattern was observed as a NO_3^- pulse during Stage 2 of snowmelt (i.e., 25-50% of cumulative runoff). This was seen in the Emerald Lake watershed and was described by Melack et al. (1998) as a pulse of NO_3^- early in the melt followed by depletion caused by biological uptake (pulse/depletion).

The hydrology of alpine watersheds that are seasonally covered with snow differs from hydrology of lower elevation watersheds in that more precipitation occurs as snow and most runoff occurs during the late spring/early summer melt season (Kattelmann and Elder 1991). Because of the snowmelt-dominated hydrological cycle and related subtle chemical changes of such watersheds, these systems may be particularly sensitive to the combined effects of acid deposition and climate warming (Wolford et al. 1996). The ionic pulse often observed during the first few days of snowmelt can result in rapid changes in the chemical composition of lakes and streams, particularly in watersheds that have limited capacity to neutralize acids. These short-term changes can also have adverse effects on in-lake and in-stream biota (Barmuta et al. 1990, Wolford et al. 1996).

Concentrations of NO_3^- in the Emerald Lake outlet increased from 2-3 µeq/L in the fall to 10-13 µeq/L during spring runoff. The observed increases in NO_3^-, and also SO_4^{2-}, were attributed to preferential elution from the snowpack and low retention rates in the watershed. In-lake reduction of NO_3^- and SO_4^{2-} within Emerald Lake was relatively small, and most of the acid anions passed through the lake outlet (Melack et al. 1998).

Williams and Melack (1991) documented an ionic pulse in meltwater in the Emerald Lake watershed two-fold to twelve-fold greater than the snowpack average. SO_4^{2-} and NO_3^- concentrations in meltwater decreased to below the initial bulk concentrations after about 30% of the snowpack had melted. The ionic pulse was variable spatially, depending on the rate of snowmelt. At a site with relatively rapid snowmelt, the pulse lasted only two days, whereas at a site with a slow rate of melt, the pulse lasted about 10 days. The first fraction of meltwater draining from the snowpack had concentrations of both NO_3^- and NH_4^+ as high as 28 µeq/L, compared to bulk snowpack concentrations < 5 µeq/L (Williams et al. 1995). Streamwater NO_3^- concentrations reached an annual peak during the first part of snowmelt runoff, with maximum streamwater concentrations of 18 µeq/L. During the summer growing season, streamwater NO_3^- concentrations were always near or below detection limits (0.5 µeq/L).

Melack et al. (1989b) presented time series limnological data for Emerald Lake that spanned a five year period (1983-1988). Lakewater chemistry data were presented for bottom water at 9.5 m depth and surface water at 1 m depth. The chemistry of surface and bottom waters were similar in pH during mixing periods, and then tended to diverge during stratification when pH

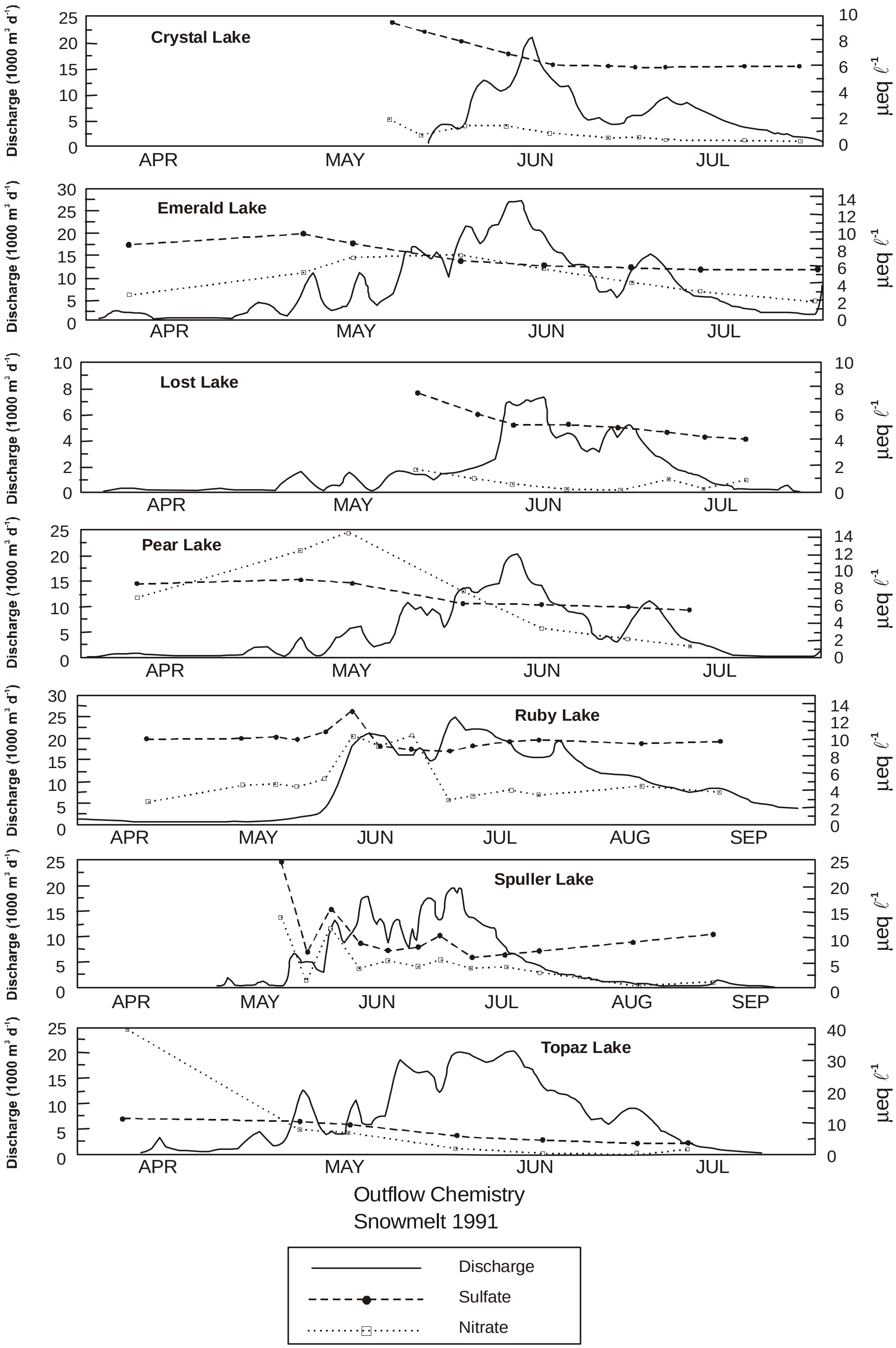

Figure IX-10. Outflow discharge, and concentrations of SO_4^{2-} and NO_3^- during snowmelt in 1991 at 7 high-elevation Sierra Nevada lakes. (Source: Melack and Sickman 1995)

was lower at depth. There was also a seasonal pattern of increased lake pH during the ice-free season followed by decreased pH under ice cover. The seasonal pattern in the concentration of both ANC and pH in the surface water of the lake resembled the ANC of the inflow stream (Melack et al. 1989). The ANC concentration in the bottom water of the lake was similar to that in the surface water during mixing and summer stratification, but ANC of the bottom water was often higher than that of the surface water during inverse stratification under ice cover (Figure IX-11).

The concentration of NO_3^- in lakewater also showed a pronounced seasonal pattern that resembled the pattern of NO_3^- in the inflow stream. Concentration of NO_3^- in the lakewater increased as the inflow discharge increased during snowmelt, reflecting the flushing of the epilimnion during that period. Subsequently, complete mixing of the water column during spring increased the concentration of NO_3^- in the bottom water. As the inflow discharge dropped, the concentration of NO_3^- decreased throughout the water column and then during summer stratification NO_3^- was lower in concentration in the hypolimnion than in the epilimnion (Melack et al. 1989b). In contrast to the observed patterns of NO_3^- concentration in Emerald Lake, SO_4^{2-} concentrations changed very little in the lake between seasons.

Leydecker et al. (1999) reported the results of episodic process studies during the period 1990 to 1994 in seven high-elevation watersheds that ranged in elevation from 2,475 m (Lost Lake) to 3,390 m (Ruby Lake). Dilution of baseflow base cation concentrations accounted for 75 to 97% of the observed episodic ANC reductions. In lakes where increasing anion concentrations (acidification) were noted during episodes (including Emerald and Pear Lakes in SEKI), NO_3^- and SO_4^{2-} were equally important during the first half of the snowmelt. Sulfate dominated the latter half.

Research at Gem Lake, a 2.8 ha, high-elevation (3,341 m) lake situated at timberline on the eastern slope of the Sierra Nevada, illustrated the importance of episodic processes in high elevation Sierran lakes. It undergoes an annual cycle of low ANC (30-40 µeq/L) during the ice-free season and high ANC (> 240 µeq/L) during fall and winter. Both the lakewater pH and ANC decrease dramatically during snowmelt. Episodic pH declined by up to a full pH unit and the mean summer pH (6.43) was nearly half a pH unit lower than the mean winter pH (6.89) (Stoddard 1987a). Based on comparisons among the chemistry of snow, snowmelt, and runoff, Stoddard (1987a) concluded that the declines in ANC were due to dilution rather than acidification. Increases in ANC in fall and winter could be attributed to mass balance estimates of weathering in the watershed.

In the Sierra Episodes Study, ten lakes and their watersheds were selected for study. Results for two of the lakes were reported by Stoddard (1995), one of which typified the response of the majority of high elevation lakes in the study (Treasure Lake) and one whose response was most extreme (High Lake). At Treasure Lake, ANC began to decline at the onset of snowmelt and reached a minimum at peak runoff, corresponding with minimum base cation, NO_3^-, and SO_4^{2-} concentrations (Figure IX-12). At no point did Treasure Lake become acidic. High Lake watershed contained a deeper snowpack, and began melting later in the season. ANC fell to zero and below twice during the first 10 days of snowmelt. The ANC minimum corresponded with maximum concentrations of base cations, NO_3^- and Al (Figure IX-13). The High Lake watershed produced snowmelt that was both later and more rapid than other lakes included in the Sierra Episodes Study, perhaps due to its high elevation and small watershed area (Stoddard 1995). This caused increases in NO_3^- concentration to values greater than 40 µeq/L, exceeding concurrent increases in base cations and causing the lake to become acidic for brief periods. High Lake appears to be representative of the most extreme conditions of episodic acid-sensitivity in the Sierra Nevada.

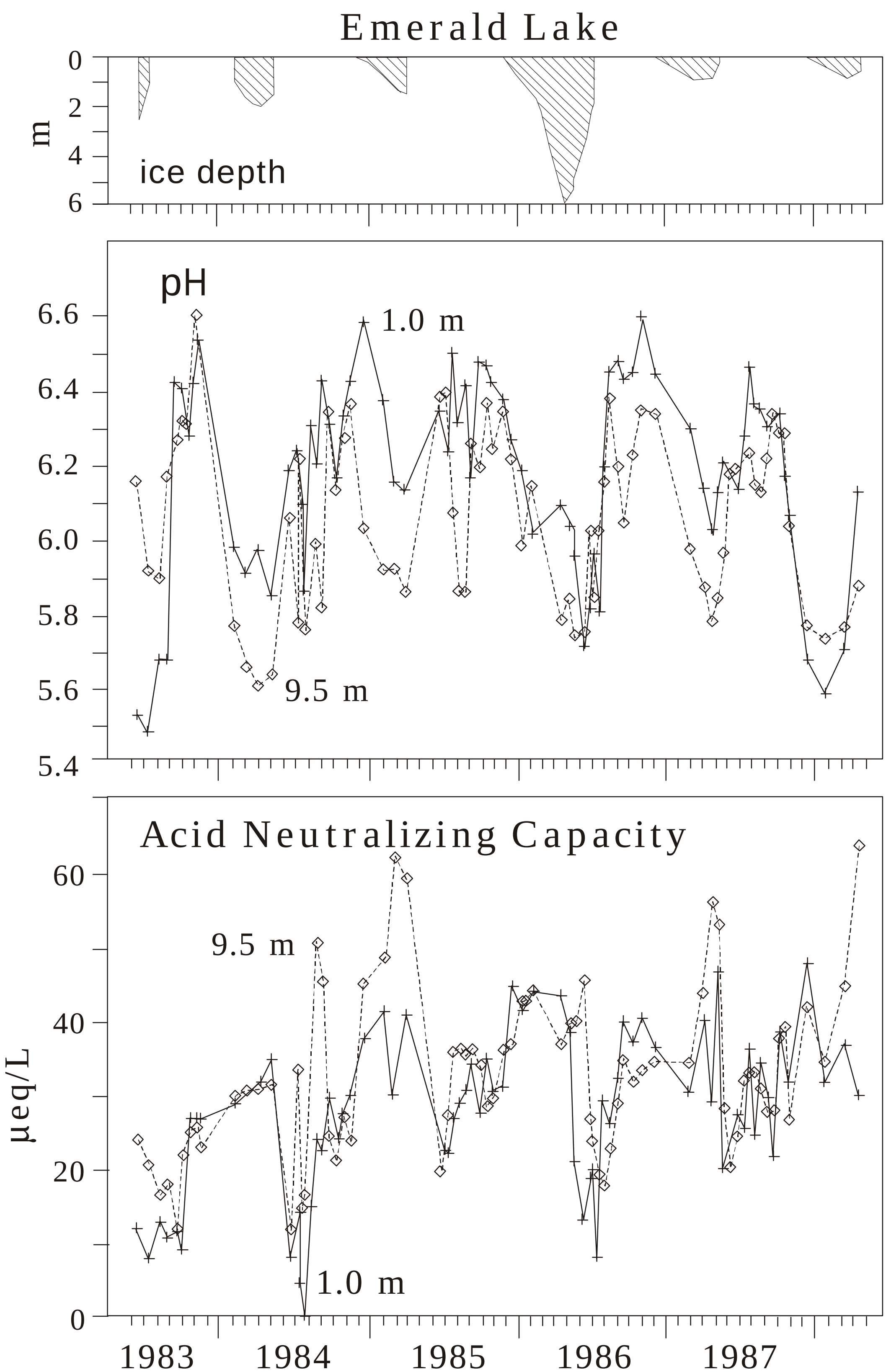

Figure IX-11. pH and ANC of Emerald Lake at 1.0 m and 9.5 m depth during the period 1983 to 1988. (Source: Melack et al. 1989)

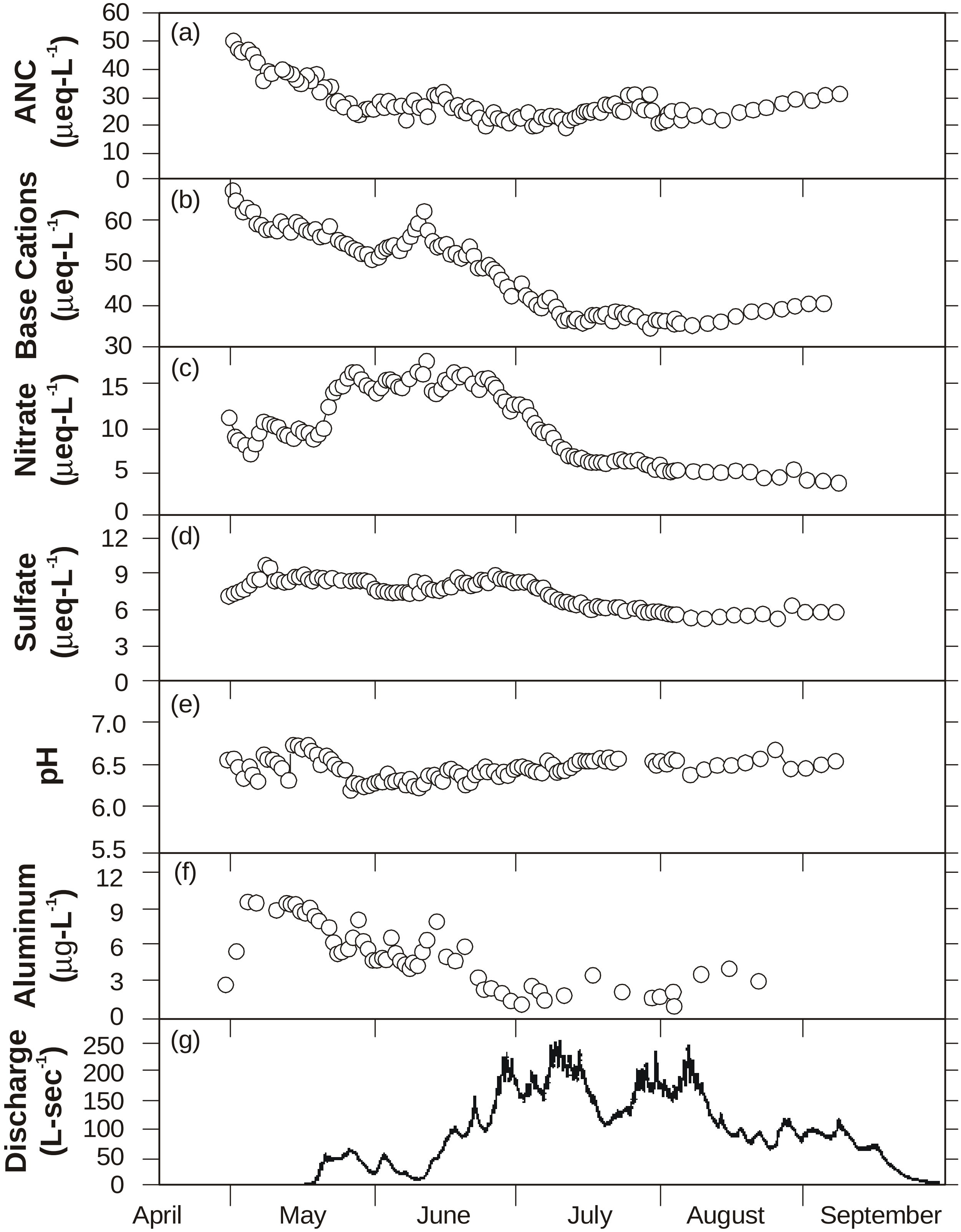

Figure IX-12. Time series of major ions and discharge in Treasure Lake during snowmelt in 1993. (Source: Stoddard 1995)

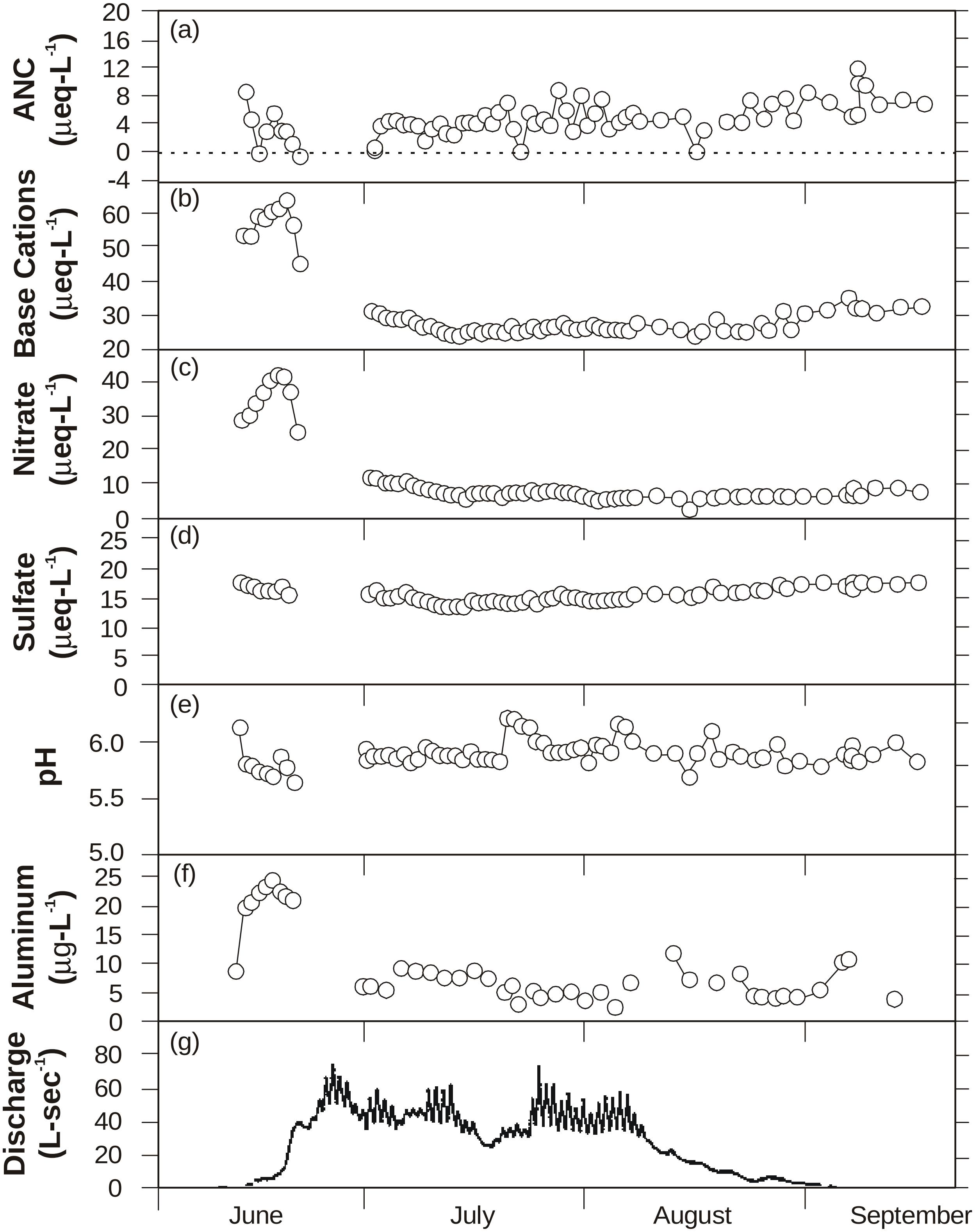

Figure IX-13. Time series of major ions and discharge in High Lake during snowmelt in 1993. (Source: Stoddard 1995)

Dilution of base cation concentrations during snowmelt is the primary factor responsible for seasonal ANC depressions in lakes in the Sierra Nevada. Melack et al. (1998) classified the lakes into two classes according to snowmelt response. In shallower lakes having short residence times (i.e., lakes that are rapidly flushed during snowmelt), NO_3^- and SO_4^{2-} accounted for < 10% of the ANC decline (Lost, Topaz, Spuller Lakes). In lakes having larger water volumes or lower snowmelt rates, NO_3^- and SO_4^{2-} accounted for 25% to 35% of the ANC decrease (Emerald, Pear, Ruby, Crystal Lakes). In the latter group, NO_3^- and SO_4^{2-} contributed about equally during the first half of snowmelt, but SO_4^{2-} was more important during the latter half of snowmelt.

Leydecker et al. (1999) found that the relationship between minimum ANC during snowmelt and fall overturn ANC was linear, and the equation remained unchanged with the addition of data from earlier surveys. Application of this equation to the population of lakes in the Sierra Nevada that were included in the WLS statistical frame suggested that none of the lakes represented by the WLS are acidified by snowmelt under current levels of acidic deposition. However, the confidence limit of this empirical relationship allowed the possibility that up to 1.8% of lakes in the Sierra Nevada are acidified to ANC ≤ 0 during snowmelt.

Leydecker et al. (1999) tabulated pre-melt and fall index ANC and episodic ANC values for each of their study watersheds and years using both lake outflow and in-lake chemistry samples. They found that the in-lake sampling every other month captured the snowmelt ANC depression almost as well as the more intensive outflow sampling. This was because the interval of minimum ANC was so extended that, typically, any sample collected from late May to early August captured most of the ANC depression. They developed an equation for minimum lakewater ANC as a function of fall overturn ANC, using a simplified version of the two-compartment chemical mixing model proposed by Eshleman et al. (1995):

$$\text{minimum ANC} = m\,\{\text{index ANC}\} - b) \tag{9-1}$$

The parameter values of the model were estimated by Melack et al. (1998) using seven of the eight monitoring lake sites[1], plus 15 additional lakes in the Sierra Nevada. The resulting equation ($r^2 = 0.98$, se = 8.0 µeq/L) was

$$\text{minimum ANC} = (0.88 \pm 0.03)\,\{\text{overturn ANC}\} - (8.6 \pm 2.8) \tag{9-2}$$

<u>Paleolimnological Studies</u>

Whiting et al. (1989) developed a diatom calibration set of 45 lakes in the Sierra Nevada. The calibration lakes ranged in pH from 5.16 to 9.53 and in alkalinity from -4 to 1,051 µeq/L. Diatom-inferred pH profiles were developed for four lakes, including Harriet Lake in YOSE, Emerald Lake in Sequoia National Park, Eastern Brook Lake in Inyo National Forest, and Lake Forty-Five in Kings Canyon National Park. The diatom-inferred pH profile from Harriet Lake (pH 6.52) suggested no significant trends over the last 250 years. Emerald Lake (pH 6.10) was inferred to have experienced a very small pH increase (< 0.1 pH units) over the past 60 years, and perhaps another small increase (about 0.15 pH units) since 1976. Eastern Brook Lake (pH 7.06) showed indication of long-term alkalinization of about 0.3 pH units over the past 200

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[1] Data from Spuller Lake were omitted from the analysis. This lake, and other similar shallow lakes with appreciable year-round groundwater inflows have unusually large snowmelt ANC depressions.

years. The pH has fluctuated significantly since about 1970. Lake Forty-Five (pH 5.16) was inferred to have acidified about 0.2 pH units over the past 60 years (Whiting et al. 1989, Holmes et al. 1989). Whiting et al. (1989) emphasized, however, that Lake Forty-Five is anomalous, and has apparently had pH around 5.2-5.4 since before 1900. This is a whole pH unit lower than nearly all other Sierra Nevada lakes. Also, the SO_4^{2-} concentration in Lake Forty-Five is extremely high, 194 µeq/L. This suggests a natural geological source of S and acidity in this watershed. The authors provided a plausible explanation for the inferred further lake acidification in recent years. They hypothesized that landslides dating to about 1920 may have exposed fresh weatherable sulfides within the watershed or within the lake itself and consequently resulted in increased lakewater SO_4^{2-} concentrations and increased acidity.

Modeling

Several hydrochemical models have been developed and/or applied using lake, snow, and watershed data collected in the intensively-studied watersheds in Sequoia National Park. Such models have been used to predict the extent of chronic and episodic acidification in response to current and potential future levels of acidic deposition.

It has long been recognized that a hydrogeochemical model was needed that could be applied successfully to areas with variable snow cover. The alpine hydrochemical model (AHM), described by Wolford et al. (1996), incorporates a detailed description of watershed processes in alpine settings and simulates events critical to biota such as the ionic pulse associated with spring snowmelt. The model computes integrated water and chemical balances for multiple terrestrial, stream, and lake subunits within the watershed, each of which can have a unique and variable snow covered area. Wolford et al. (1996) used two years of data from the Emerald Lake watershed for fitting and testing the model by comparing observations with modeled daily output. The ability of AHM to describe a second year of data with parameters estimated from the first year provided some confidence that AHM can be applied to estimate changes in response to anthropogenic perturbations. Additional model testing will be required, however, to confirm the general applicability of this model for describing the behavior of seasonally snow-covered alpine catchments.

Potential episodic ANC depressions due to acidic deposition were simulated with the Episodic Event Model (EEM) for lakes in the Sierra Nevada by Nikolaidis et al. (1989). EEM is a hydrologic mixing model which simulates the dilution of epilimnion water with snowmelt or precipitation runoff. Because the model did not include any in-watershed acid neutralization of storm or snowmelt runoff, it represents a worst case scenario. A Monte-Carlo simulation technique was used for each lake, to represent the assumed range of values for each input parameter. Two hundred and fifty simulations were run in order to obtain an estimate of the mean and standard deviation of the simulated episodic lake ANC. The study divided the Sierra Nevada into three geographic regions. Data from the Giant Forest deposition monitoring station were used to characterize the southern Sierra region. Data from the Yosemite and Mammoth deposition monitoring stations were applied to the central Sierra region.

Spring snowmelt events were simulated to have greater impacts on lakewater quality than summer rainstorms. Under current loading rates, the average ANC for the 28 lakes in the southern Sierra region after a 20-day early spring snowmelt event was projected to be 16 ± 17 µeq/L. If the event occurred in late spring, the projected average ANC declined only to 30 ± 22 µeq/L, from the pre-episode baseline average ANC of 60 µeq/L (Nikolaidis et al. 1989).

Hooper et al. (1990) developed a simple process-based model called the Alpine Lake Forecaster (ALF), using IWS data from Emerald Lake watershed. During a heavy snow year, the model accurately simulated chemical changes in surface water in response to the initial ionic

pulse from the snowmelt and the dilution that occurred at peak snowmelt. Hooper et al. (1990) found evidence that cation exchange in the watershed may be an important source of alkalinity during early snowmelt in a light snow year, and the model therefore overpredicted acidification during that early snowmelt period. The model predicted a 2 to 5 µeq/L change in minimum ANC of the lake outflow if the deposition of both NO_3^- and S were doubled and a moderate acidic pulse was released from the snowpack. The model further suggested that deposition of both NO_3^- and S would have to increase 2 to 18-fold over current loadings to reduce lakewater ANC to zero. The modeled deposition increase depended on hydrologic conditions and the pattern of solute release from the snowpack.

In a follow-up study using the ALF model, Hooper and Peters (1993) simulated the chemistry of Topaz and Pear Lakes in Sequoia National Park and also of Ruby and Crystal Lakes, located north of the park on the eastern slope of the Sierra Nevada in Inyo and Mono Counties. Geochemical processes represented in the model seemed to be applicable to three of the lakes. However, Pear Lake turns over infrequently because of its geomorphology, and its ANC seems to be controlled largely by biological rather than geochemical processes, thereby rendering ALF unsuitable. Hooper and Peters (1993) found that parameter estimation for ALF required collection of calibration data over a large range of discharge, from baseflow to peak snowmelt discharge.

A sensitivity analysis of the geochemical formulations of the model indicated that the streamwater chemical response to acidic snowmelt was highly dependent upon the proportion of more readily weatherable minerals contained in the bedrock. The authors suggested that the relationship between the sum of base cation concentration compared with the silica concentration might serve to quantify the relative sensitivity of the watershed to acidification.

Cosby and Sullivan (2001) evaluated the sensitivity of six alpine and subalpine lakes in four Class I national parks in the Sierra Nevada, Cascade, and Rocky Mountains to increased atmospheric loading of S and N, using the Model of Acidification of Groundwater in Catchments (MAGIC). A model evaluation was conducted of dose/response relationships and critical loads for S and N deposition (on their own and combined) for headwater lakes and their catchments in Rocky Mountain, Grand Teton, Sequoia, and Mt. Rainier National Parks. The effects of future increases in S and N deposition on the chemistries of Loch Vale, Surprise Lake, Amphitheatre Lake, Emerald Lake, Pear Lake, and Eunice Lake were evaluated for 50-year simulations. The results of the MAGIC model simulations suggested that the chemistry of the study lakes is moderately to highly sensitive to changes in S, and to a lesser extent, N deposition. The loads of S deposition that would drive chronic lakewater ANC to below 20 µeq/L were estimated to range from 1 to 10 kg S/ha/yr, respectively. Comparable loads for N deposition were estimated to be 7 to 32 kg N/ha/yr, respectively. Slightly higher loads of S and N were required to produce chronic ANC values below zero. In general, acid sensitivity was greatest for Pear and Emerald Lakes in Sequoia National Park, although Loch Vale in Rocky Mountain National Park, Colorado was more sensitive than any of the other lakes to simulated changes in N deposition. This modeling exercise is a prelude to a proposed regional analysis of critical loads for the larger population of acid-sensitive aquatic resources. Such an analysis will provide the scientific foundation for federally-mandated land management decisions that are required to protect sensitive resources in Class I national parks and wilderness areas from adverse impacts of acidic deposition.

b. Aquatic Biota

Emerald Lake has been the focus of a considerable amount of work on the potential biological effects of S and N deposition. Melack et al. (1989a) concluded that Emerald Lake is

not showing serious chemical or biological effects of acidification. Many species of aquatic biota known to be acid-sensitive are found in Emerald Lake and associated streams, and the brook trout population does not show signs of acid-induced stress. However, Melack et al. (1989a) also concluded that Emerald Lake and associated streams are extraordinarily sensitive to acidification because of their extremely dilute ionic chemistry. They postulated, based on their experimental work, that even slight acidification of the lake would result in changes in the species composition of the zooplankton assemblage. If acidic deposition was to increase in the future, streams would also likely be impacted. Seasonal patterns in the chemistry of Emerald Lake are controlled largely by thermal stratification, flushing during snowmelt, and nutrient uptake by phytoplankton during the ice-free season (Melack et al. 1989a). Interannual variation in the chemistry of Emerald Lake and the streams in its watershed were explained by variation in the annual quantity of snowfall, which in turn determines the degree of flushing of the lake during snowmelt and also the duration of the subsequent ice-free season.

Of the aquatic invertebrate groups, zooplankton typically show the clearest and most consistent response to acid-induced stress, commonly including reduction of species diversity, and changes in the relative abundance of species (Økland and Økland 1986, Melack et al. 1989a). The zoobenthos also include species that are acid-sensitive. Surveys and experiments in streams have documented in particular the sensitivity to acidification of mayflies (*Ephemeroptera*) and stoneflies (*Plecoptera*) (Hall and Ide 1987, MacKay and Kersey 1985, Ormerod et al. 1987). Melack et al. (1989a) conducted detailed sampling of the invertebrate communities of Emerald Lake and its outflow beginning in July of 1984. The aim of these monitoring studies was to produce a baseline of biological data which could then be used to establish the magnitude of seasonal and year-to-year variations against which to judge future changes in aquatic biota that may occur in response to acidic deposition. They examined two major biological communities in detail: the zooplankton of Emerald Lake itself and the zoobenthos of its outflow.

The observed typical pattern of seasonal change in species composition of zooplankton in Emerald Lake was as follows. From January until late March all species were at low densities except for the cladoceran *Bosmina longirostris*. In late May or early June *Holopedium gibberum* dominated the crustaceans, and was followed later in summer and early fall by high numbers of *Daphnia rosea* and *Diaptomus signicauda*. Concomitantly, the colonial rotifer *Conochilus unicornis* reached its peak abundance together with *Polyarthra vulgaris*. *Keratella taurocephala* became more dominant later in fall, and *Bosmina longirostris* became dominant during winter (Melack et al. 1989a).

Composition of zoobenthos was diverse with dipteran taxa, especially Chironomidae, making up the bulk of the species. Zooplankton in Emerald Lake were typical of those found in other oligotrophic high altitude lakes in the Sierra Nevada that contain brook trout (Stoddard 1987). Species composition of the stream benthos was similar to that found in other streams in the Kaweah catchment. The presence of taxa that are known to be acid intolerant in Emerald Lake (e.g., *Daphnia rosea, Diaptomus signicauda*) and in the stream (e.g., *Pisidium, Baetis*, other Ephemeroptera and some Plecoptera) suggested that neither the lake nor stream habitat is currently suffering chronic acid stress (Melack et al. 1989a).

There was a marked seasonality to the zooplankton assemblage in Emerald Lake, which was repeated regularly from year to year except when long winter conditions delayed life cycles and diminished the abundance of species that were characteristic of late summer. The data also suggested that the extreme climatic conditions that were observed in 1986 may have reduced the density of *Bosmina longirostris* in comparison with the previous winters. Most species seemed to have recovered in 1987, however, indicating that the zooplankton were resilient to this sort of

climatic stress. Stream benthos also showed some evidence of reduced densities in 1986, consistent with the observation of scouring of the outflow resulting from the 1986 avalanche and the poor recruitment of brook trout in that year. Melack et al. (1989a) concluded that interannual variation in climate and disturbance regime could significantly affect zooplanktonic and benthic community structure and such effects would have to be considered in interpretation of long-term monitoring data.

Melack et al. (1989a) concluded from their field experiments that four zooplankton species would be good indicators of future acidity in Emerald Lake: *Daphnia rosea, Diaptomus signicauda, Bosmina longirostris,* and *Keratella taurocephala*. The former two declined with experimental acidification, while the latter increased, at least down to pH 5.0. Melack et al. (1989a) also concluded that the zooplankton of Emerald Lake was the most reliable and easiest group of aquatic invertebrates to use for continued biological monitoring. This was because the major species are taxonomically well known, samples are easy to collect, and they can be processed more rapidly and inexpensively than the benthic samples.

Of the 1,404 Sierra lakes located above 2,400 m elevation that were represented by the statistical design of Jenkins et al. (1994), the authors estimated that 881 contained one or two species of salmonid fish, 127 contained mountain yellow-legged frogs (*Rana muscosa*) but no other vertebrates, 284 contained only invertebrates, and 112 contained no fish and almost no invertebrates. The most commonly collected fish species were golden trout (*Oncorhynchus mykiss aquabonita*) and rainbow trout (*O. mykiss*). Brook trout, and brown trout (*Salmo trutta*) ranked third and fourth in frequency of occurrence, respectfully. Few lakes contained cutthroat trout (*O. clarki*). Jenkins et al. (1994) judged that nearly all of the present high-elevation fish populations were established by humans and that the vast majority (possibly all) of the lakes located above 2,439 m in the Sierra Nevada were originally devoid of fish due to natural barriers to fish movement.

Trout belonging to the genus *Oncorhynchus* (i.e., golden, rainbow, and cutthroat trout) appear to be more sensitive to acid input than trout belonging to the genera *Salvelinus* (brook trout) or *Salmo* (brown trout) and would be expected to be the first fish to respond to substantial increases in acidic deposition in the high Sierra.

Clear distinctions in water chemistry between lakes that contained fish versus those which did not were generally not found in the study. However, Al concentration tended to be lower where golden trout were present versus absent, and pH tended to be higher where brook trout were present versus absent. Both of these chemical factors were correlated with elevation, however, and the observed relationships may have been confounded by patterns of stocking related to elevation.

The number of macroinvertebrate taxa identified in streams and lakes was positively correlated with pH, and in lakes it was negatively correlated with NO_3^- concentration and elevation. Similarly, pH was higher and NO_3^-, SO_4^{2-}, and elevation tended to be lower where the common macroinvertebrates *Callibaetis* and *Pisidium* were present compared to where they were absent. The authors concluded, however, that all fish and most invertebrate species were able to live and reproduce in lakes and streams with pH as low as 6.

Jenkins et al. (1994) conducted a dose-response experiment during snowmelt in channels next to the outlet stream of Spuller Lake, considered to be a representative high-altitude Sierra Lake. Buried eggs of golden trout were exposed to a gradient of six pH levels that ranged from pH 4.8 to 6.6. The experiment was carried out for a period of 40 days and the survivorship of eggs was determined 9-10 days after the exposure period. No significant impact was observed of acid inputs on egg survival. Based on review of the literature and results of this study, Jenkins et al. (1994) suggested that the eggs of trout species would not be affected by acid inputs until the

pH was lowered to below pH 4.5. They also indicated, however, that literature data suggest that later stages such as sac and swim-up fry are quite sensitive to high Al concentrations and would therefore be negatively affected by pH depressions in the range of 5.0 to 5.5 if these depressions were accompanied by high concentrations of Al.

The dominant cations in most of the study lakes were Ca^{2+} and Na^{+}, and about half of the lakes had Ca^{2+} less than 50 µeq/L. Total Al concentrations were generally less than 1.9 µM (50 µg/L). The eggs of golden, rainbow, and cutthroat trout (spring spawning species) and the fry of brook and brown trout (fall spawning species) are present at snowmelt in the high Sierra, the time when acid pulses are most likely to occur from acidic deposition. It is difficult to predict trout population responses to different acidity and Al levels from available literature data, largely because different studies have used different methods, approaches, dosing levels, acclimation periods, water chemistry, and trout stages and strains. In addition, many studies have used combinations of acid and Al concentrations which are unlikely to occur in High Sierra waters (Jenkins et al. 1994). Adverse effects of episodic pulses of acidity on the biota of aquatic systems in the Sierra Nevada will likely depend mostly on changes in the concentrations of H^{+}, Al^{n+}, and Ca^{2+} ions, and also on the locations and developmental stages of the sensitive aquatic organisms that are present.

Between 1995 and 1997, a biological survey was conducted of approximately 2,200 lakes in the John Muir Wilderness (JMW) and Kings Canyon National Park. Information was collected on fish, amphibians, invertebrates, and physical attributes of lakes and ponds. Preliminary results were reported by Matthews and Knapp (1999). The authors suggested a linkage between widespread introduction of non-native trout and the decline of a native amphibian, the mountain yellow-legged frog (*Rana muscosa*). The majority of the larger lakes at the upper elevations of the Sierra Nevada, which were historically fishless, now have one or more species of non-native trout. Matthews and Knapp (1999) compared the aquatic fauna of lakes in JMW to that found in Kings Canyon National Park. The areas are generally similar except regarding fish stocking, which historically has been much more intensive in JMW than in Kings Canyon National Park. The more limited fish stocking in Kings Canyon National Park was terminated in the late 1970s. Matthew and Knapp (1999) found that 29% of the lakes in JMW had introduced trout, compared to 19% in Kings Canyon National Park. For lakes larger than 1 ha in area, 80% contained trout in JMW as compared with 40% in Kings Canyon National Park. The authors found more populations of mountain yellow-legged frogs in Kings Canyon National Park compared to JMW. Thirty-five percent of the lakes in Kings Canyon National Park contained the frog species compared to only 5% in JMW. In addition, the total number of frogs that were observed in Kings Canyon National Park was much higher: 69,000 compared with 9,000 adults, sub-adults, and larvae in Kings Canyon National Park and JMW, respectively. The yellow-legged frog appears to be particularly sensitive to the effects of trout introductions because it is highly aquatic in all life stages and the tadpoles overwinter two to three times before metamorphosing into sub-adult frogs. This overwintering requirement restricts successful breeding to bodies of water that do not dry up during summer. These are the bodies of water into which trout were most commonly introduced. Rowan (1996) also reported that benthic invertebrate species diversity, mean abundance, and mean size were all lower in trout-containing lakes compared to fishless lakes.

A survey was conducted in the early summer of 1992 of 104 lakes in the Bench Lake/ Mount Pinchot area of Kings Canyon National Park by Bradford et al. (1998). They surveyed the chemical conditions and the presence and absence of vertebrate populations. The location of the watersheds and lakes studied by Bradford et al. (1998) are provided in Figure IX-14. The results of this study are highly relevant to questions regarding potential biological effects of

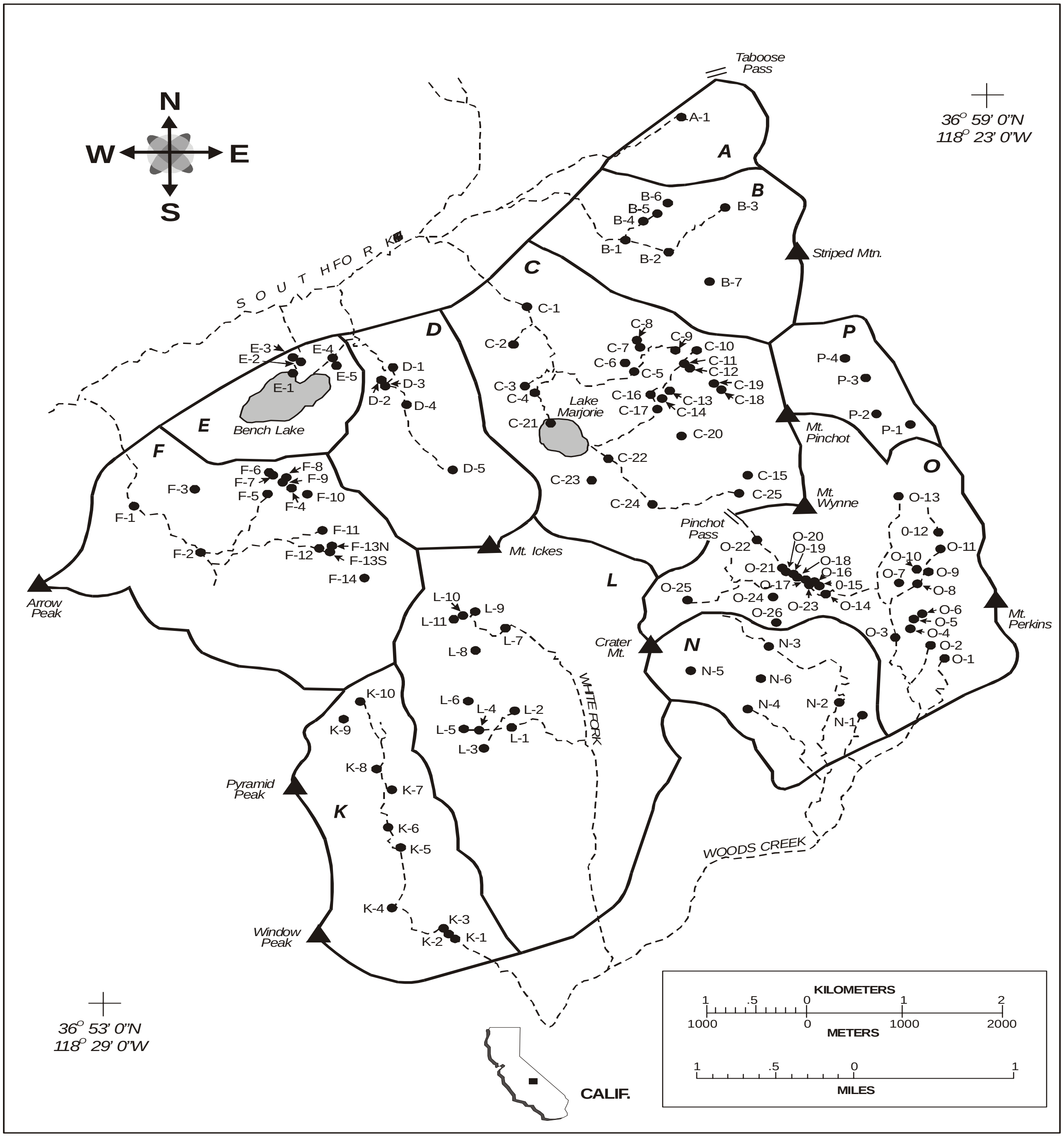


Figure IX-14. Study area and major watersheds surveyed by Bradford et al. (1994, 1998) in Kings Canyon National Park for water chemistry and the distribution of aquatic biota. Closed, labeled circles correspond to lakes shown on base map (U.S. Geological Survey, 7 ½' quadrangle, Mt. Pinchot, Calif., Provisional Edition, 1985). The 33 lakes in the detailed survey are as follows: acidic lakes (C-22, -24; F-1, -2, -13N, -14; L-7, -11), fish lakes (B-1; C-4, -5, -21; D-4; E-1; N-3), and non-acidic fishless lakes (B-5; C-2, -10, -17, -23; D-5; E-4; F-4, -11, -12, -13S; L-1, -8, -9; O-7, -8, -21; P-3). (Source: Bradford et al. 1998)

acidic deposition in high-elevation aquatic ecosystems in the Sierra Nevada. Although the natural acidity of many of the study lakes was accompanied by rather high concentrations of SO_4^{2-} (> 400 µeq/L, which could have direct biological effects in some cases), the observed relationships between lakewater chemistry and the distribution of aquatic biota in these lakes should generally correspond with anticipated changes if acidic deposition was to increase dramatically in the future. Lakes ranged in pH from 5.0 to 9.3 and included 10 lakes that had pH less than 6. On the basis of that survey, Bradford et al. (1998) selected 33 lakes for more detailed analyses of their chemical and biological characteristics. Ten of the lakes had measured ANC ≤ 20 µeq/L. The results of selected chemical parameter values for these low-ANC lakes are provided in Table IX-14. SO_4^{2-} was the dominant anion in a number of the low-ANC lakes. Many of these had pH values below 6 and ANC near or below 0. The source of the SO_4^{2-} and acidity was sulfuric acid produced by the oxidation of pyrite that is found in metamorphic and granitic rocks in that area (Bradford et al. 1998). Neutralization of the internal watershed source of acidity occurs downstream, primarily by dilution with water that has low ionic strength. Bradford et al. also conducted faunal surveys in the study lakes. Yellow-legged frog tadpoles and limnethilid cadus larvae (*Hesperophylax*) and large microcrustaceans of the genera *Daphnia* and *Diaptomus* were rare or absent in lakes that had pH values less than 6 but were commonly found in lakes with pH above 6 (c.f., Figures IX-15 and IX-16). Four species of trout were found and trout distribution appeared to be related to historical patterns of fish stocking. Distribution of trout also appeared to have a large impact on the distribution and abundance of amphibian and invertebrate taxa. Large, mobile, and conspicuous taxa were rare or absent in lakes that contained trout and relatively common in lakes that lacked trout.

Table IX-14. Selected water chemistry parameter values of the 10 lakes in Kings Canyon National Park that were sampled by Bradford et al. (1998) and that had ANC ≤ 20 µeq/L. See Figure IX-14 for lake locations.

Code	Date	Conductivity (µS/cm)	pH	ANC (µeq/L)	NO_3^- (µeq/L)	SO_4^{2-} (µeq/L)
L-7	7/2/92	62	4.82	-7	7.3	460
F-1	6/30/92	60	4.86	-1	9.2	436
F2	6/30/92	73	4.80	0	10.8	581
F-11	6/30/92	29	5.87	11	14.0	198
F-14	8/23/92	32	5.30	1	25.8	250
C-22	7/3/92	49	5.11	3	11.7	350
C-24	6/29/92	51	4.83	-2	12.4	396
B-5	8/15/92	5	6.33	15	1.2	13
F13NA	8/23/92	45	5.77	5	15.0	338
L-11	8/30/92	40	5.96	6	26.2	278

Presence of yellow-legged frog tadpoles in lakes was determined at least in part by the acid-base status of the lakes. Tadpoles were not observed in any of the lakes that had pH less than 6. They were commonly observed in lakes that had pH above 6 and that also lacked fish. In contrast, the adult frogs were often found in lakes with pHs as low as 5. Bradford et al. (1998)

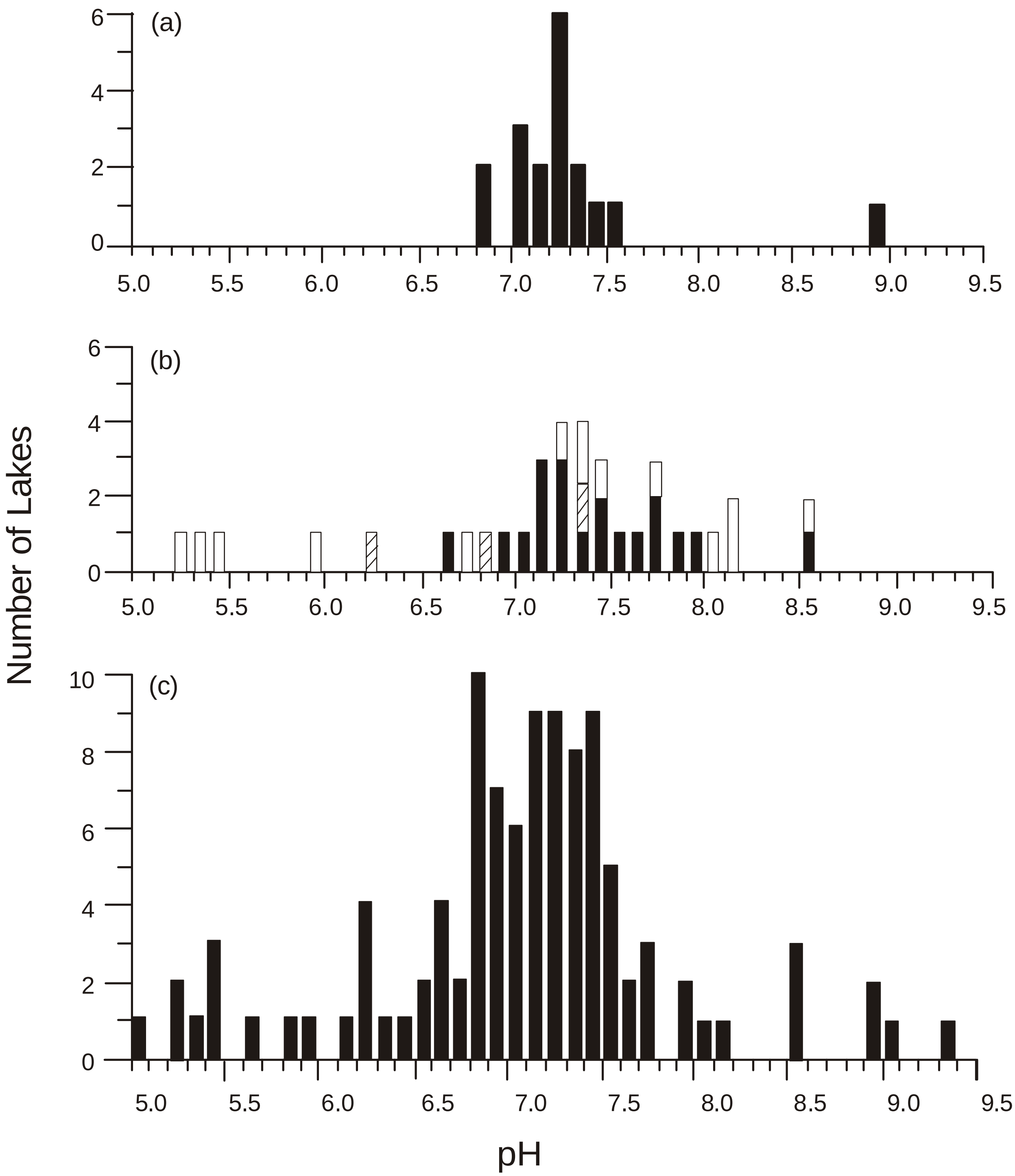


Figure IX-15. Frequency distribution of pH in the synoptic survey of Bradford et al. (1998) in the Mt. Pinchot area of Kings Canyon National Park. (a) Lakes where trout were observed (n=18). (b) Lakes where yellow-legged frog tadpoles and adults were observed (n=36). Solid bars represent lakes containing both tadpoles and adults (n=19); hatched bars represent lakes with tadpoles only (n=3); and open bars represent lakes with adults only (n=14). (c) All lakes surveyed (n=104). (Source: Bradford et al. 1998)

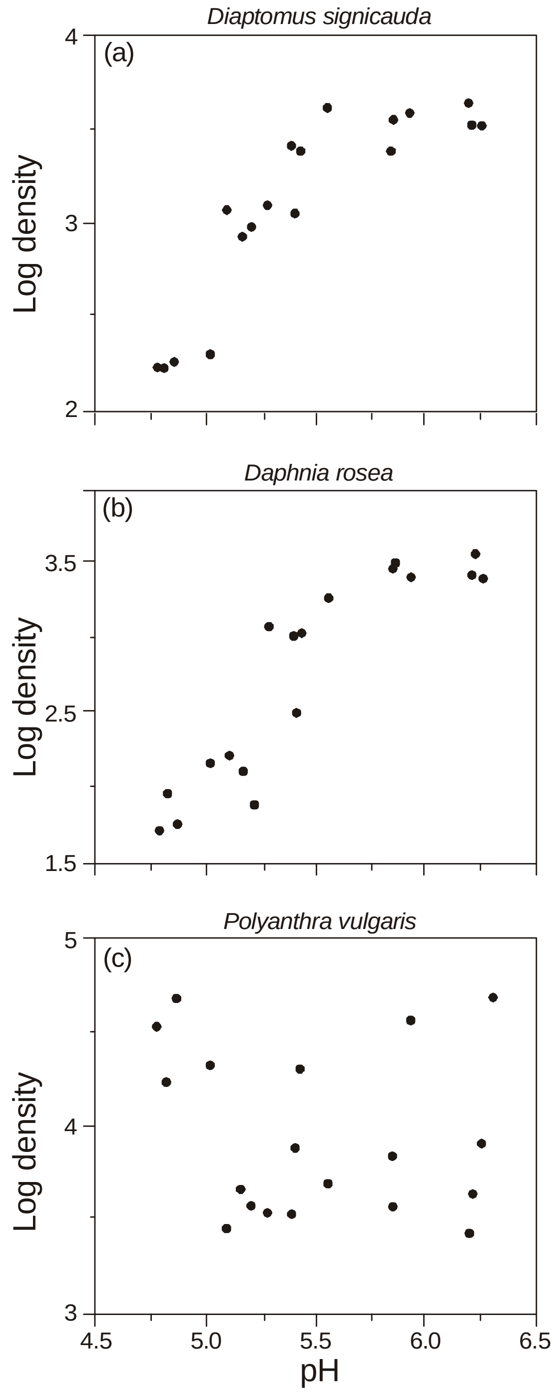


Figure IX-16. Relationships between log-transformed abundances (no./m^3) of three zooplankton taxa and pH in a field experiment (Barmuta et al. 1990) for comparison with data for the same or related taxa in the detailed survey of Bradford et al. (1998). The field experiment was conducted for 35 days in 18 polyethylene enclosures (each 7.6 m^3) that extended from the bottom to the surface of a High Sierra lake. Nitric and sulphuric acid were added in varying amounts to enclosures to create six pH levels, with three replicate enclosures at each level. Data shown are mean abundances of designated zooplankton in each enclosure versus mean pH of each enclosure after acidification over the duration of the experiment. (Source: Bradford et al. 1998)

found this difference between tadpole and adult distribution relative to lake pH to be particularly striking where lakes having pH less than and above 6 occurred in close proximity to each other. The mountain yellow-legged frog is relatively common in the study area, compared to other places in the Sierra Nevada. The species was recorded in 36 of the 104 lakes that were surveyed. Bradford et al. (1998) noted that tadpoles appeared to be adversely affected at pH levels below 6 in the field whereas survival of embryos and tadpoles in the laboratory for seven-day periods was not significantly affected by pH values as low as 4.75 (Bradford et al. 1992, 1993). There are a number of possibilities for the observed difference in sensitivity in the field versus laboratory data. For example, the low pH lakes studied by Bradford et al. (1998) contained concentrations of SO_4^{2-} and Al that were often many times greater than levels used in the laboratory experiments, and both SO_4^{2-} and Al can be toxic to amphibian embryos and larvae. Bradford et al. (1998) suggested other possible explanations for the apparent greater sensitivity in the field. These included the possibility that the sampling missed extreme pH depressions, that only a restricted part of the mountain yellow-legged frog life cycle was examined in the laboratory experiments, and finally that laboratory experiments do not recreate the host of conditions faced by embryos and larvae in the field.

Only two amphibian species were found in the Marble Fork of the Kaweah River watershed by Melack et al. (1989a). Mount Lyell salamanders (*Hydromantes platycephalus*) were sighted on two occasions during the study period. Tadpoles of the Pacific tree frog (*Pseudacris regilla*)were found in 8 of 10 ponds surveyed in 1985, 10 of 15 surveyed in 1986, and 11 of 15 surveyed in 1987 (Melack et al. 1989a, Soiseth 1992). In addition, tree frog tadpoles were collected in Topaz Lake. Of these ponds and lakes, only Hidden Pond contained fish. Occurrence of larvae of the Pacific tree frog in ponds appeared to be unrelated to pond size, degree of permanence, elevation, or pond pH. The latter varied from pH 5.3 to 7.2 for ponds which contained tadpoles and varied from 5.7 to 7.0 for ponds which did not. Tadpole population densities for 1985 varied from 0 to over 50/ m^2. Again, there appeared to be little or no relationship between tadpole density and pond size, degree of permanence, elevation, or pH.

Lakes and streams in the Marble Fork basin contained dense reproducing populations of brook trout that were characterized by slow growth rates (particularly after age 3 years), small adult body size, and long life span. Slow growth rate and small body size were likely the result of crowding relative to the available food supply. Melack et al. (1989a) also surveyed the chemistry and biology of eight lakes and a vernal pond in the Kaweah River drainage beginning in 1984. The study lakes included Emerald, Heather, Pear, Hidden, Topaz, Frog, Lyness, Aster, and a vernal pond. Five lakes had large reproducing populations of brook trout; the others were fishless. Lakes were sampled in August and September of 1984 through 1987. All of the lakes sampled in the survey were generally similar in chemistry to Emerald Lake. With the exception of Pear Lake, the total ionic content of the other survey lakes was slightly higher than that of Emerald Lake, however. Twenty-three species of zooplankton were collected, ranging from 9 in Hidden Lake to 17 in Aster and Heather Lakes over all four years. The number of species collected in the lakes tended to decline with increasing elevation across all lakes ($p < 0.05$). There were also differences observed in the species composition of zooplankton assemblages in lakes with and without trout. The number of macroinvertebrate taxa that were collected in sweep samples from the lakes ranged from 10 in Heather Lake to 18 in Frog Lake, with a mean of 11. Again, species richness tended to decline with increasing elevation, both across all lakes and in those containing trout ($p \leq 0.05$). Melack et al. (1989a) concluded that elevation and the presence of trout were the most important determinants of zooplankton community composition in lakes in the Marble Fork catchment. These results agreed with those of Stoddard (1986) who studied 75 lakes in the Sierra Nevada, including Emerald, Pear, and Heather Lakes. The

zooplankton assemblage of Emerald Lake was concluded to be typical of those found in high elevation lakes containing brook trout (Stoddard 1986). The most apparent pattern in the distribution of macroinvertebrates in the Kaweah drainage appeared to be related to the presence of brook trout. Such studies of lake macroinvertebrate communities are rare, but suggest that lakes containing few or no fish generally contain conspicuous macroinvertebrate taxa which are largely absent from lakes with greater numbers of fish (Melack et al. 1989a).

In addition to increasing the potential for episodic acidification, increased atmospheric deposition of N has the potential of altering the algal biomass of lakes in the Sierra Nevada to the extent that N is the growth-limiting nutrient in those lakes. To examine this issue, Sickman and Melack (1989) conducted experimental manipulations of the plankton in Emerald Lake *in situ* using artificial enclosures. Experiments were conducted using five microcosms (volume 10 L) and four mesocosms (volume about 3,500 L) during the summers of 1983 and 1984. These experiments were designed to determine the limiting nutrients for phytoplankton growth in Emerald Lake and separate out the effects of increased acidity from those of increased nutrient concentration on the lake's phytoplankton.

The results suggested that phytoplankton growth in Emerald Lake is strongly limited by phosphorus. Five out of seven experiments indicated that chlorophyll levels of the seston (algae and particulate matter in the water column) increased with phosphorus additions. Nitrogen was never found to be the sole limiting element and was co-limiting in only one experiment. The extent to which these results are applicable to other lakes throughout the Sierra Nevada is unclear. Nutrient limitation has been studied in three other montane lakes in California: Castle, Tahoe, and Gem. In Castle Lake and Lake Tahoe, phytoplankton productivity was demonstrated to be limited by the availability of N (Goldman 1966). At high-altitude Gem Lake, algal biomass was shown to be limited by P in combination with copper or iron (Stoddard 1987). For Emerald Lake, however, given the role of P in limiting phytoplankton biomass in the lake, it's unlikely that increased N loading associated with acidic deposition will have a fertilizing effect on the lake plankton (Sickman and Melack 1989).

A recent study was conducted of pesticide residues in brook trout and Pacific tree frogs in the Kaweah River watershed (Datta et al. 1998). Samples were collected along Sycamore Creek, at 610 m elevation, and compared with results for brook trout from Lake Tahoe and tree frog egg masses from Upper Meadows in LAVO. Concentrations of polychlorinated biphenyls (PCBs) in brook trout taken from Sycamore Creek (4.8 to 8.1 ppb, wet weight) were similar to those found in lake trout in Lake Tahoe and rainbow trout from Marlette Lake. However, tree frog tadpoles and egg masses showed enriched levels of tri- and tetrachlorinated biphenyl congeners. Analyses for currently used pesticides upwind from SEKI revealed the presence of chlorpyrifos, an organophosphate pesticide, and chlorothalanil, a chloronitrile fungicide. This may have implications for the observed decline in populations of some amphibian species.

c. Summary of Current Status of Surface Water Resources and Magnitude of Effects

Thus, there has been a great deal of research on the chemistry and biology of streams, and especially lakes, within SEKI, and to some extent elsewhere within high-elevation areas of the Sierra Nevada. Significant effort has been expended to determine the extent to which current levels of acidic deposition may have contributed to chronic and/or episodic acidification and the extent of acidification that would be required to elicit adverse biological impacts. The results of these studies paint a rather complete picture of the current acid-base status and biological conditions of freshwater resources in SEKI. These results are also very relevant to YOSE, and likely to acid-sensitive high-elevation aquatic resources in LAVO.

Alpine and subalpine lakes and streams in SEKI are extremely acid-sensitive. A high percentage have ANC < 50 µeq/L and many have ANC below 30 µeq/L. It is possible that some of the most acid-sensitive lakes and streams have been chronically acidified by current levels of acidic deposition, but such acidification, if it has occurred, has been quite small in magnitude. It is likely that many of the lakes and streams that exhibit lowest ANC experience episodic acidification in response to current levels of S, and especially N, deposition. Such an effect is superimposed on natural episodic acidification processes, especially base cation dilution.

There is no evidence to suggest that current levels of acidic deposition have caused either chronic or episodic biological effects. Available data suggest that such effects are unlikely. It is very likely, however, that adverse biological effects would be manifested under moderately increased deposition of S or N. Baseline conditions have been rather well described. Continued monitoring of aquatic chemistry and biology would be able to document if and when future changes occur. It is important to continue long-term monitoring efforts so that future acidification, if it should occur, will be detected quickly.

3. Vegetation

Graber and Haultain (1993) reported the results of a plot-based parkwide survey of vascular plants in SEKI that was conducted between 1985 and 1991. They employed a randomized systematic sampling design based on 1 km-grid intersections. They conducted an exhaustive enumeration of plant species on 0.1 ha circular plots, took additional measurements of plant species dominance, and recorded abiotic environmental factors. This Natural Resources Inventory was designed to gather and organize data about the resources of the parks and to array them in a relational computer database and within the framework of a geographic information system. Over 500 plots were surveyed and these were generally physiographically representative of the terrain within the park. An acceptable representation was found for elevation, and the distribution of plot aspects was also similar to that of the park as a whole. Steep slopes (> 30^o) were somewhat undersampled, however, due to safety concerns associated with conducting field work on such slopes. The survey encountered 860 vascular plant species, 68% of the 1,268 species known to occur in the parks.

There have been more vegetation monitoring and research activities related to air pollution at SEKI than at any other western national park. A significant threat to natural resources by high oxidant levels combined with a supportive scientific environment have motivated numerous intensive and extensive studies at the park over the past 20 years. Many of these scientific activities have focused on the effects of elevated ozone concentrations on vegetation.

The mixed conifer forest at SEKI is part of a broad region of the Sierra Nevada that has significant impacts to vegetation from chronic exposure to ozone. Spatial patterns of ozone injury and spatial and temporal patterns of tree growth in SEKI have been quantified by several studies. In addition, intensive physiological studies have provided evidence to support clear cause-effect relationships.

One of the first studies to quantify ozone injury to ponderosa pine and Jeffrey pine in SEKI was completed in the early 1980s (Warner et al. 1983). This study documented widespread foliar injury as part of a broader regional pattern encompassing adjacent national forests (Pronos et al. 1978, Pronos and Vogler 1981, Allison 1982), and motivated periodic surveys of oxidant injury to these bioindicator species at SEKI and at YOSE to the north (Duriscoe and Stolte 1989, Duriscoe 1990, Stolte et al. 1992, Ewell and Gay 1993). An extensive survey of sites along the western edge of SEKI (where ozone exposure is highest) indicated that approximately 90% of pines had some evidence of foliar injury (including high injury on first-year needles), with symptomatic trees having lower needle retention than asymptomatic trees (Peterson and Arbaugh

1988, 1992, Peterson et al. 1991). Injury at SEKI was higher than any other location in the Sierra Nevada (See Chapter I for more discussion of FOREST study). The recent definitive FOREST study (Arbaugh et al. 1998) corroborated this pattern, determining that 93% of pines had visible injury, again higher than any other location in the Sierra Nevada. Physiological studies in and near SEKI in the field (Patterson and Rundel 1989, Bytnerowicz and Grulke 1992, Grulke 1999) and in controlled exposure studies of seedlings (Temple et al. 1992, 1993) have confirmed that the observed injury is related to physiological effects in pines. The potential physiological effects of elevated ozone on giant sequoia have also been documented (Evans and Leonard 1991, Grulke et al. 1996); seedlings less than 5 years old are moderately susceptible to ozone injury, whereas older trees (including the overstory trees) are apparently resistant (Grulke et al. 1989, Grulke and Miller 1994, Miller and Grulke 1994).

Even more compelling evidence of the effects of ozone on mixed conifer forest has been provided by data that indicate that growth has been reduced in injured trees. Ewell et al. (1989b) and Duriscoe and Stolte (1992) found that the level of ozone injury in ponderosa pine foliage was inversely correlated with foliar biomass, which indicates a significant change in productivity relationships of mature trees. Reduced stem growth has been documented for both ponderosa pine and Jeffrey pine with ozone injury in SEKI. Open-grown Jeffrey pine with injury symptoms were found to have 11% lower radial growth since 1950 than uninjured trees from nearby locations (Peterson et al. 1987). As part of a study that examined tree growth at 56 sites ranging from the Tahoe National Forest to the Sequoia National Forest, 8 sites (4 symptomatic, 4 asymptomatic) were examined at SEKI (Peterson et al. 1991, Peterson and Arbaugh 1992). The symptomatic sites at SEKI had the largest number of trees with decreases in radial growth since 1950 of any location in the Sierra Nevada. Older trees and trees with lower needle retention tended to have the lowest growth.

Repeated documentation of severe ozone injury in dominant trees, reduced tree growth, and associated physiological data for mature trees in the western portion of SEKI present a clear picture of widespread impacts from prolonged exposure to air pollutants. Placed within the context of larger data sets for the Sierra Nevada (Peterson et al. 1991, Arbaugh et al. 1998), SEKI consistently has the highest level of damage, with the "hot spot" occurring within the mixed conifer zone.

Permanent plots established in forest stands (Warner et al. 1983, Parsons et al. 1992) and chaparral (Parsons and Stohlgren 1986) are a valuable resource for monitoring the effects of air pollution on plants in the context of temporal variation in vegetation dynamics. As part of an effort to monitor ozone exposure and vegetation effects in the Sierra Nevada, the USDA Forest Service and California Air Resources Board have recently established permanent monitoring plots for ponderosa pine and Jeffrey pine at Giant Forest and Grant Grove.

Some research on the lichen flora of SEKI has also been conducted. In 1984, Wetmore (1985) collected lichens at several locations in the park in order to determine if there were any indications of adverse effects of air pollutants. Although he noted the presence of species known to be sensitive to ozone and SO_2, he did not observe any injury symptoms, and chemical analyses did not reveal abnormally high concentrations of any elements.

Additional studies are examining the potential for elevated N deposition to mitigate the effects of ozone on yellow pines, as part of the joint NPS-USEPA program on Park Research and Intensive Monitoring of Ecosystems (PRIMENet). Research directed by the USGS Western Ecological Research Center on the long-term effects of climatic variability has established a series of 20 1-to-2-ha plots that quantify vegetation characteristics from lower treeline to upper treeline.

The sensitivities of plant species at SEKI to air pollutants are summarized in Table IX-15. In addition to the well-known ozone bioindicators ponderosa pine and Jeffrey pine, SEKI contains 10 other vascular plant species with high ozone sensitivity.

Table IX-15. Plant and lichen species of Sequoia National Park and Kings Canyon National Park with known sensitivities to sulfur dioxide, ozone, and nitrogen oxides (H=high, M=medium, L=low, blank=unknown).

Scientific Name	Common Name	Sensitivity		
		SO_2	O_3	NO_x
<u>Gymnosperms</u>				
Abies concolor	White fir	H	M	H
Calocedrus decurrens	Incense cedar		M	
Pinus contorta	Lodgepole pine	H	M	H
Pinus flexilis	Limber pine	L	M	
Pinus jeffreyi	Jeffrey pine	H	H	H
Pinus monticola	Western white pine	M	M	
Pinus ponderosa	Ponderosa pine	H	H	H
Sequoiadendron giganteum	Giant sequoia		M	
Tsuga mertensiana	Mountain hemlock	H	L	
<u>Angiosperms</u>				
Acer macrophyllum	Bigleaf maple		L	
Achillea millefolium	Common yarrow		L	
Agastache urticifolia	Nettleleaf giant hyssop		M	
Agropyron smithii	Western wheatgrass	M		
Alnus rhombifolia	White alder	H	M	
Alnus tenuifolia	Thinleaf alder	M		
Artemisia douglasiana	Douglas' sagewort		H	
Artemisia dracunculus	Wormwood		M	
Artemisia tridentata	Big sagebrush	M	L	
Betula occidentalis	Water birch	M		
Bromus carinatus	California brome		L	
Bromus rubens	Foxtail brome	M	M	
Bromus tectorum	Cheatgrass		M	
Cassiope mertensiana	Western moss heather		L	
Ceanothus velutinus	Snowbrush ceanothus	L		
Cercocarpus ledifolius	Curlleaf mountain mahogany	M		
Chenopodium fremontii	Fremont's goosefoot		L	
Collomia linearis	Narrowleaf mountaintrumpet		L	
Cordylanthus rigidus ssp. brevibracteatus	Stiffbranch bird's beak		H	
Coreopsis bigelovii	Bigelow's tickseed	M	M	
Cornus stolonifera	Redosier dogwood	M	L	
Corylus cornuta var. californica	California hazelnut	H		
Cryptantha circumscissa	Cushion catseye	M	M	
Descurainia californica	Sierran tansymustard		L	
Elymus glaucus	Blue wildrye		H	
Epilobium angustifolium	Fireweed		L	
Epilobium brachycarpum	Autumn willowweed		L	
Erodium cicutarium	Redstem stork's bill	M	M	
Festuca octoflora	Sixweeks fescue	L	L	

Table IX-15. Continued.

Scientific Name	Common Name	Sensitivity		
		SO_2	O_3	NO_x
Galium bifolium	Twinleaf bedstraw		L	
Gayophytum diffusum	Spreading groundsmoke		H	
Gayophytum racemosum	Blackfoot groundsmoke		L	
Gentiana amarella	Autumn dwarfgentian		M	
Geranium richardsonii	Richardson's geranium	M	M	
Hackelia floribunda	Manyflower stickseed	L		
Helianthus annuus	Common sunflower	H	L	
Lepidium virginicum var. pubescens	Hairy pepperweed		H	
Lolium perenne	Perennial ryegrass		M	
Lonicera involucrata	Twinberry honeysuckle	L		M
Lupinus latifolius	Broadleaf lupine		M	
Mimulus guttatus	Seep monkeyflower		L	
Oenothera elata	Hooker’s evening primrose		H	
Oryzopsis hymenoides	Indian ricegrass	M		
Osmorhiza chilensis	Sweetcicely		M	
Phacelia heterophylla	Varileaf phacelia		L	
Physocarpus capitatus	Pacific ninebark		M	
Platanus racemosa	California sycamore		M	
Platystemon californicus	California creamcups	M	M	
Poa annua	Annual bluegrass	H	L	
Poa pratensis	Kentucky bluegrass		L	
Polygonum douglasii	Douglas' knotweed		L	
Populus tremuloides	Quaking aspen	H	H	
Populus trichocarpa	Black cottonwood	M	H	
Potentilla flabellifolia	High mountain cinquefoil		H	
Potentilla fruticosa	Shrubby cinquefoil		L	
Potentilla glandulosa	Gland cinquefoil		M	
Prunus emarginata	Bitter cherry	M		
Quercus kelloggii	California black oak		M	
Ribes viscosissimum	Sticky currant	M		
Rosa woodsii	Woods' rose	M	L	
Rubus parviflorus	Thimbleberry		M	
Rumex crispus	Curly dock		L	
Salix scouleriana	Scouler's willow		M	
Salvia columbariae	Chia	M	M	
Sambucus melanocarpa	Black elderberry		M	
Sambucus mexicana	Blue elder		H	
Schismus barbatus	Common Mediterranean grass	M		
Senecio serra	Butterweed groundsel		H	
Symphoricarpos vaccinioides	Utah snowberry		L	
Taraxacum officinale	Common dandelion		L	
Thalictrum fendleri	Fendler's meadowrue		L	
Thysanocarpus curvipes	Sand fringepod	M	M	
Tragopogon dubius	Yellow salsify	M		
Trifolium repens	White clover		H	
Trisetum spicatum	Spike trisetum	M		
Viola adunca	Hookedspur violet		L	
Vitis californica	California wild grape		L	

Table IX-15. Continued.				
Scientific Name	Common Name	Sensitivity		
		SO_2	O_3	NO_x
Lichens				
Acarospora chlorophana		H		
Aspicilia caesiocinerea		L		
Bryoria abbreviata			H	
Bryoria fremontii			H	
Bryoria fuscescens		M		
Buellia punctata		M		
Calicium viride		M	H	
Caloplaca cerina		H		
Candelaria concolor		H		
Candelariella vitellina		M		
Cladonia chlorophaea		M		
Cladonia coniocraea		M		
Cladonia fimbriata		H		
Collema nigrescens			M	
Evernia prunastri		M	H	
Hypocenomyce scalaris		M		
Hypogymnia enteromorpha		M	M	
Hypogymnia imshaugii		M	M	
Lecanora carpinea		M		
Lecanora chlarotera		M		
Lecanora hagenii		M		
Lecanora muralis		M		
Lecanora saligna		M		
Lecidea atrobrunnea		L		
Leptogium californicum			M	
Letharia columbiana		L	L	
Letharia vulpina		L	L	
Melanelia glabra			L	
Melanelia multispora			L	
Melanelia subaurifera			H	
Melanelia subolivacea			L	
Normandina pulchella		H		
Ochrolechia androgyna		H		
Parmelia saxatilis		M	L	
Parmelia sulcata		M	H	
Parmeliopsis ambigua		M		
Peltigera canina		L	H	
Peltigera collina			H	
Phaeophyscia nigricans		M		
Phaeophyscia orbicularis		M	H	
Physcia adscendens		M		
Physcia aipolia		M		
Physcia biziana			L	
Physcia caesia		M		
Physcia dubia		M		
Physcia stellaris		M		
Physcia tenella		M	L	
Physconia detersa		H	L	

Table IX-15. Continued.

Scientific Name	Common Name	Sensitivity		
		SO_2	O_3	NO_x
Platismatia glauca		M	H	
Pseudephebe minuscula			H	
Pseudephebe pubescens			H	
Ramalina farinacea			H	
Rhizocarpon geographicum		L		
Rhizoplaca melanophthalma		H		
Tuckermannopsis merrillii			M	
Umbilicaria polyphylla		M		
Xanthoparmelia cumberlandia		H		
Xanthoria candelaria		H	H	
Xanthoria elegans		M		
Xanthoria fallax		H	L	
Xanthoria polycarpa		H	L	

4. Visibility

As part of the Interagency Monitoring of Protected Visual Environments (IMPROVE) network, visual air quality in SEKI has been monitored using an aerosol sampler. A full IMPROVE aerosol sampler (modules A, B, C, and D) began operation in March of 1994. The sampler is located near the Ash Mountain Park Headquarters at the southwest entrance to Sequoia National Park. Data from this IMPROVE site have been summarized to characterize the full range of visibility conditions for the March 1994 through February 1999 period, based on seasonal periods (Spring: March, April, and May; Summer: June, July, and August; Autumn: September, October, and November; and Winter: December, January, and February) and annual periods (March through February of the following year, e.g., the annual period of 1998 includes March 1998 through February 1999). Complete descriptions of visibility characterization, mechanisms of sources and visibility impacts, and IMPROVE monitoring techniques and rationale are provided in Chapter I.

a. Aerosol Sampler Data - Particle Monitoring

A tabular and graphic summary of average reconstructed extinction values by season and year for the March 1994 through February 1999 period are provided in Table IX-16 and Figure IX-17, respectively. Reconstructed extinction budgets generated from aerosol sampler data apportion the extinction at SEKI to specific aerosol species (Figure IX-18). The species shown are Rayleigh, sulfate, nitrate, organics, elemental (light absorbing) carbon, and coarse mass. The sum of these species account for the majority of non-weather related extinctions. Extinction budgets are listed by season and by mean of cleanest 20%, mean of the median 20%, and mean of the dirtiest 20%. The "cleanest" and "dirtiest" signify lowest fine mass concentrations and highest fine mass concentrations respectively, with "median" representing the 20% of days with fine mass concentrations in the middle of the distribution. Each budget includes the corresponding extinction coefficient, standard visual range (km), and deciview (dv).

The segment at the bottom of each stacked bar in Figure IX-18 represents Rayleigh scattering, which is assumed to be a constant 10 Mm^{-1} at all sites during all seasons. Rayleigh scattering is the natural scattering of light by atmospheric gases. Higher fractions of extinction due to Rayleigh scattering indicate cleaner conditions.

Table IX-16. Seasonal and annual average reconstructed extinction (b_{ext}; Mm^{-1}) at SEKI, March 1994 through February 1999.

Year	Spring (Mar, Apr, May)	Summer (Jun, Jul, Aug)	Autumn (Sep, Oct, Nov)	Winter (Dec, Jan, Feb)	Annual (Feb-Mar)[a]
1994	92.0	94.8	53.3	50.5	73.7
1995	53.1	55.6	103.3	75.4	70.1
1996	57.3	- - -	65.5	59.9	61.3
1997	61.8	56.1	52.4	40.3	53.4
1998	45.9	62.5	55.5	63.1	57.2
Mean[b]	62.0	67.3	66.0	57.8	63.1[c]

[a] Annual period data represent the mean of all data for each March through February annual period.
[b] Combined season data represent the mean of all seasonal means for each season of the March 1988 through February 1999 period.
[c] Combined annual period data represent the mean of all combined season means.

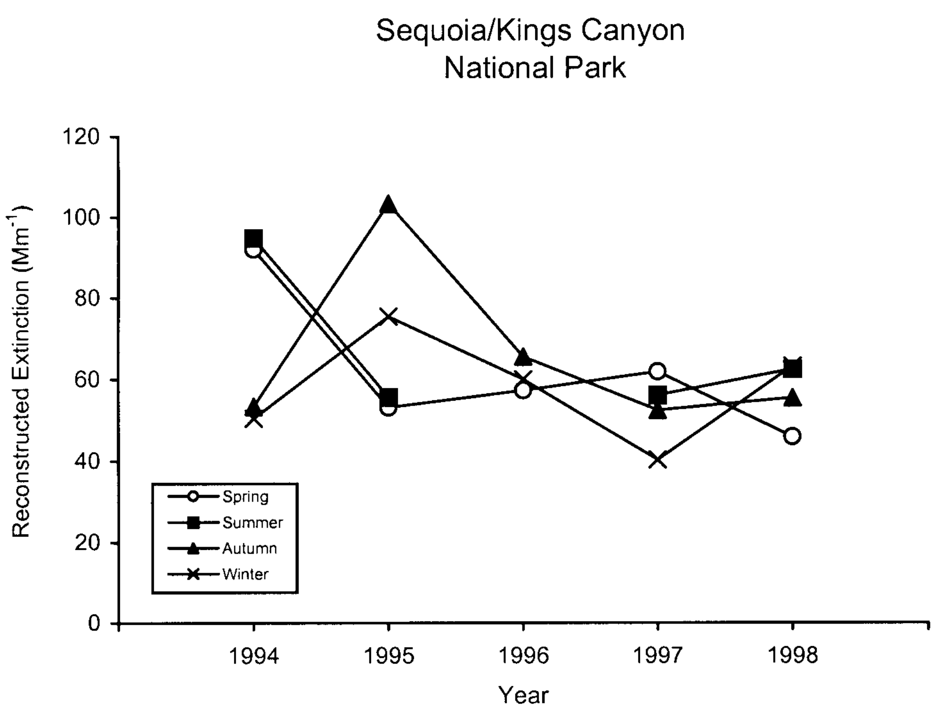


Figure IX-17. Seasonal average reconstructed extinction (Mm^{-1}) SEKI, March 1994 through February 1999.

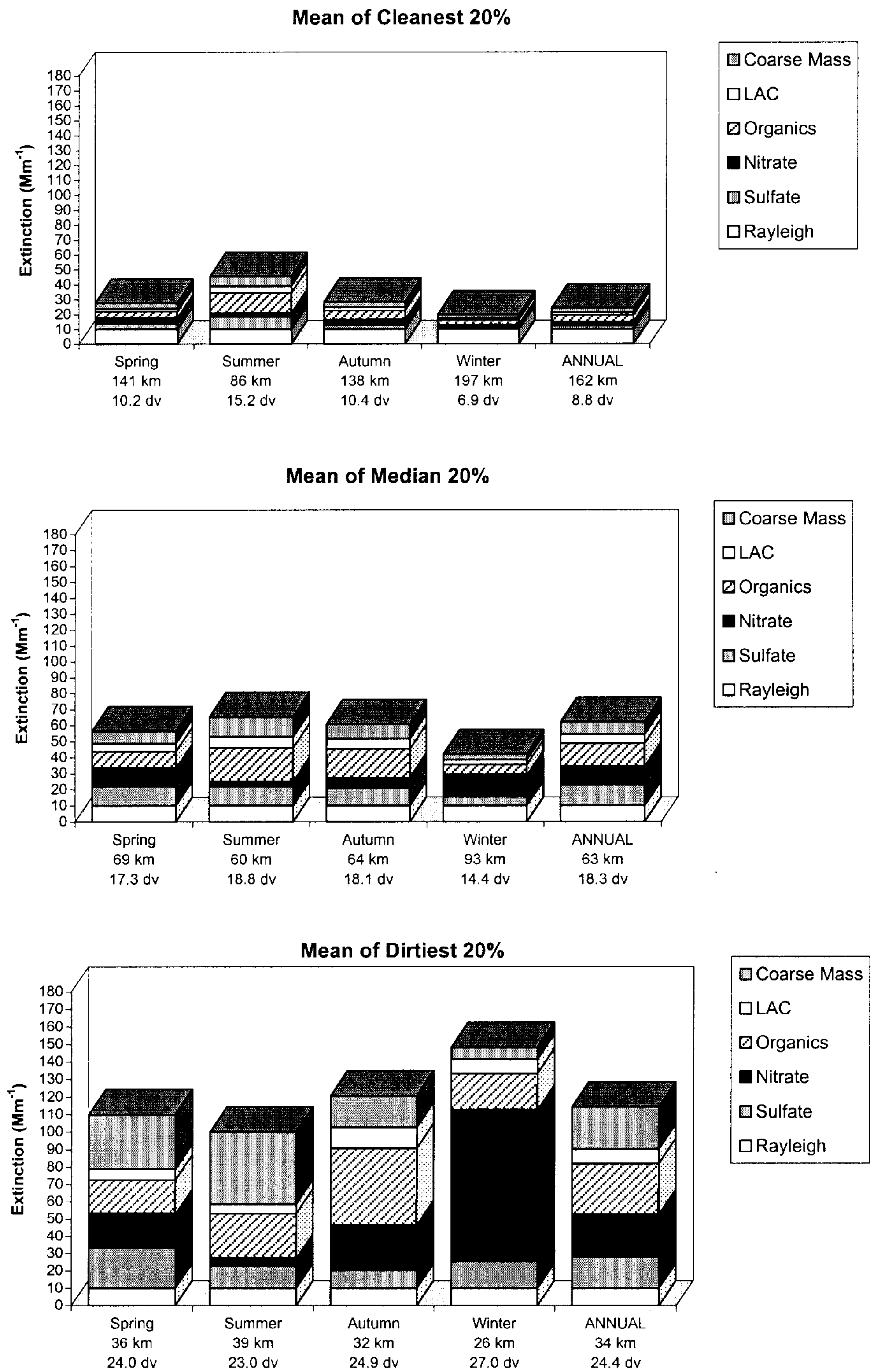


Figure IX-18. Reconstructed extinction budgets for SEKI, March 1994 through February 1999.

b. Site-Specific Data Interpretation

Data from other IMPROVE visibility sites around the country have been presented graphically (Figures I-2 and I-3) so that spatial trends in visual air quality for the Sierra Nevada region can be understood in perspective. It should be noted that data summaries for the Sierra Nevada region only include YOSE. The period of record for SEKI (3/1994 - 2/1999) differed too much from the YOSE period of record (3/1988 - 2/1999) to represent an accurate combined regional summary. Figures IX-17 and IX-18 have been provided to summarize SEKI visual air quality during the March 1994 through February 1999 period. Seasonal variances in the mean of the dirtiest 20% fractions are driven by organic and nitrate extinctions as well as variations in coarse mass. Non-Rayleigh atmospheric light extinction at SEKI is largely due to organics, sulfates, and nitrates. Historically, visibility varies with patterns in weather, winds (and the affects of winds on coarse particles) and smoke from fires. No information is available on how the distribution of visibility conditions at present differs from the profile under "natural" conditions, but the cleanest 20% of the days probably approach natural conditions (GCVTC, 1996). Smoke from frequent fires is suspected to have reduced presettlement visibility below current levels during some summer months.

Long-term trends fall into three categories: increases, decreases, and insignificant changes. The characterization of long-term trends can be a highly subjective exercise in that slopes and their significance can vary depending on the technique employed. Recently the IMPROVE aerosol network, initiated in March 1988, matured to a point where long-term trends of average ambient aerosol concentrations and reconstructed extinction can be assessed for sites with an eleven year period of record. Because full IMPROVE aerosol monitoring was not initiated at SEKI until March 1994, the monitoring period is not considered long enough to adequately summarize long-term trends. Once additional data is obtained, through continued monitoring, long-term trends can be considered.

D. RESEARCH AND MONITORING NEEDS

1. General

In addition to monitoring efforts, resources should also be devoted to acquiring and integrating existing data, and to analyzing and presenting findings from the available data bases. Real-time data displays (e.g., daily particulate levels) are potentially of significant interest to park visitors. Existing meteorological and air-quality data could be more extensively analyzed to better characterize patterns of transport of air pollutants (e.g., fine particulate), including the nature of upslope and downslope flows.

Better characterization of pollutant transport is also needed to support the park’s controlled burning program. While emissions from burning are not a critical component of deposition, fine particulate is a health concern potentially affecting park employees, visitors, and residents of nearby communities.

Additional efforts should be devoted to characterizing the ambient atmospheric levels of non-criteria air pollutants (e.g., pesticides) and comparing these levels with their concentrations in aquatic systems. This research effort would be enhanced by linking it to any ongoing studies of atmospheric transport.

Carbon monoxide (CO) should be monitored whenever burning occurs. Monitoring locations should include sites near campgrounds, offices, and residences.

2. Deposition

Continued monitoring of wet and dry deposition is needed to provide estimates of spatial patterns and temporal variability of deposition. The long-term data records should be

maintained by continuing wet-deposition monitoring at Ash Mountain (CARB) and Giant Forest (NADP). Dry deposition monitoring should continue at Ash Mountain (IMPROVE), Lookout Point (CASTNet), and Wolverton (NOAA).

The long-term deposition record that is presently available lends itself to analyses of trends. The data should be analyzed to determine the statistical power of trend tests (i.e., the length of monitoring needed to detect trends and the levels of change that can be discerned within a specified time).

3. Gases

Continued monitoring of ozone is needed to provide estimates of spatial patterns and temporal variability. The long-term data records should be maintained by continuing ozone monitoring at the existing sites at Giant Forest, Ash Mountain, and Lookout Point. Passive ozone monitors suggest that ozone levels vary significantly within SEKI, and additional efforts to characterize the spatial patterns of ozone concentrations should be supported. Summary statistics documenting violations of the state and federal ozone standards are also of interest.

The use of existing passive sampler data should be continued to provide on-site data relevant for co-located ozone-injury vegetation plots. The value of previous ozone data collected with passive samplers should be assessed for developing calibrated spatial relationships relative to analyzers, so that large-scale ozone exposure in the park can be quantified. If the spatial distribution (especially variation in elevation) has not been sufficient, it is recommended that a one-time study with well-distributed passive samplers be used for spatial calibration.

4. Aquatic Systems

Long-term monitoring of high-elevation surface water resources in SEKI should continue. SEKI has some of the best baseline data on aquatic acid-base chemical status of any western park. Of the Class I parks in California that contain highly sensitive aquatic resources (SEKI, YOSE, LAVO), deposition of S and N is highest in SEKI. A variety of biological studies have documented the occurrence of acid-sensitive aquatic biota in low-ANC lakes within SEKI and characterized associated biological communities. A continued aquatic monitoring program should therefore be a top priority.

A great deal of effort and expense has been committed to aquatic resource characterization in SEKI with respect to potential lake acidification from acidic deposition. It is critical that lakewater chemical monitoring be continued so that any future acidification can be documented in a timely manner, thereby allowing remedial actions to be implemented before any damage to aquatic ecosystems becomes severe. Long-term monitoring should be continued in at least two or three high-elevation lakes. Emerald and Topaz Lakes are good candidates because they already have long periods of record. Pear Lake is also a good candidate because it is slightly more acid-sensitive, although the sampling logistics are more difficult for Pear Lake and it contains seasonally anoxic bottom water, which complicates interpretation of limnological data. Continued long-term monitoring should include, at a minimum, annual sampling in the fall at each lake and biweekly or monthly sampling during snowmelt (April through July) at least once every three years. Monitoring of snow chemistry at initiation of snowmelt should also be considered a high priority.

Long-term monitoring of acid-sensitive high-elevation lakes in SEKI should include a biological component. Zooplankton constitutes the group of aquatic invertebrates that is easiest and most reliable for detection of biological impacts associated with acidification (Melack et al. 1989a). Zooplankton monitoring could be conducted at the lakes selected for continued chemical monitoring, but it will entail a significant effort.

Acid-sensitive lakes in the park should be included in park-wide and/or regional modeling efforts to determine the levels of S or N deposition that would be likely to cause chronic or episodic acidification in the future. Such modeling should be statistically-based so that results and conclusions are applicable to the population of acid-sensitive lakes in the park or throughout the region. There are several process-based watershed models available for this effort.

5. Terrestrial Systems

It is strongly recommended that monitoring of ozone injury be continued at existing permanent plots in SEKI (Grant Grove, Giant Forest). This is critical in order to measure temporal changes in foliar condition and to compare SEKI to other locations in the Sierra Nevada as part of a regional network of monitoring sites. Having these monitoring data associated with ozone analyzer data at SEKI is an important link between ozone exposure and ozone effects in mature trees. Although the current monitoring plots are likely sufficient to measure broad spatial patterns in pines, SEKI should consider additional long-term monitoring in the Deer Ridge area, because this site seems to have particularly severe impacts. Monitoring protocols should be the same as those used in the FOREST study (Arbaugh et al. 1998), including the use of Ozone Injury Index (Duriscoe et al. 1996, Schilling and Duriscoe 1996) to quantify injury.

Because SEKI has such high levels of ozone exposure and already has widespread injury in pines, it is likely that other sensitive plant species are also being affected. Therefore, it is recommended that SEKI consider monitoring plots that include a broader range of ozone bioindicator plants in order to provide a more comprehensive description of the effects of air pollution on vegetation. These plots can include quaking aspen, black cottonwood, and blue elder, whose symptoms are well documented, but would ideally include some understory species of the mixed conifer forest that are co-located with injured pines. Information on the effects of ozone on sequoia seedlings is primarily from controlled fumigation chamber studies. Field plots should be established to better quantify the effects of ozone on sequoia regeneration and recruitment. Monitoring protocols from the Forest Health Monitoring manual (USDA Forest Service 1999) are recommended for establishing plots and collecting data (see Appendix).

It would be helpful to have more information on plant species in SEKI that are sensitive to ozone. This information could be obtained by screening several species in controlled-exposure studies. In addition, expanded studies of not only ozone injury, but other measures of ecosystem patterns and processes, would help to determine if reduced vigor in pines is affecting other components. These studies could include various aspects of regeneration, decomposition, resource allocation, and physiological response. Linkages with studies conducted by the USGS Western Ecological Research Center on these topics should be formalized where possible.

6. Visibility

IMPROVE aerosol monitoring should continue at SEKI. As part of the IMPROVE Network expansion, the EPA and NPS have relocated and upgraded the aerosol sampler in the parks. The site at Ash Mountain (550 m) inadequately represented the average elevation of the parks. The new site is located at a higher elevation (2,250 m). This new location may also better represent conditions for John Muir Wilderness to the north.

Aerosol concentrations can vary at different elevations throughout the parks. Aerosol and optical monitoring at both high and low elevations would provide information concerning the extent and nature of visibility impairment at SEKI. Emissions from industrial processes and wildland fires also contribute to visibility impairment at SEKI. Therefore, monitoring should be developed and conducted to identify carbon emissions from industrial processes and wildland fires.

X. YOSEMITE NATIONAL PARK

A. GENERAL DESCRIPTION

Yosemite National Park (YOSE) covers 26,390 km^2, with elevations ranging from 648 m at El Portal at the western edge of the park to 3,997 m at the summit of Mount Yell. The landscape is dominated by branching canyons of two large river systems which originate in glaciated headwater areas that are dotted with alpine lakes, ridges, and peaks of exposed granite. The Merced and the Tuolumne Rivers flow through large glaciated canyons that are deeply incised into the surrounding forested uplands.

Sedimentary rocks of the Yosemite area were folded and uplifted by granitic intrusions between about 125 and 150 million years ago. Uplift formed a series of northwesterly ridges overlying a westward sloping batholith. Over the next 125 million years, gradual uplifting and continued erosion occurred, thereby increasing the slope of the batholith. The greatest period of uplift began about 2 million YBP, during which time severe faulting and major downthrusting occurred east of the Sierra crest. This created the spectacular eastern Sierra escarpment. The character of the major canyons was subsequently altered by several periods of mountain glaciation.

Visitation is heavily concentrated in Yosemite Valley, which is about 11 km long and 1.6 to 3.2 km wide. It represents a widened part of the canyon of the Merced River, with parallel sides that are boldly sculpted and hung with waterfalls. The valley floor is generally level and lies at an elevation of about 1,220 m. The uplands on both sides rise an additional 900 to 1,200 m. Yosemite Canyon is distinguished from most other canyons in the Sierra Nevada by its great width relative to its depth, its exceptionally sheer walls, and by its level, almost gradeless floor (Matthes 1950).

An area map for the park is shown in Figure X-1.

1. Geology and Soils

Yosemite's dramatic scenery has resulted from a variety of geological processes, including sedimentation, igneous intrusion, folding, faulting, erosion, and glaciation. About 2 million years ago, a series of uplifts raised the Sierra Nevada block to its present height, about 4,200 m above sea level, and the Merced River Valley was cut to a depth of about 450 m. Tributary valleys are not typically cut as deeply by their smaller streams as are larger river valleys, and were therefore left hanging high on the canyon walls in the Merced River Valley. During the Pleistocene, three distinct periods of glacial ice invasion occurred, each lasting for many thousand years. Yosemite Valley was filled rim to rim with ice, covering Glacier Point to a 150-m depth, and leaving the higher peaks such as El Capitan, Half Dome, and Sentinel Dome protruding above the surface of the ice (Rowe 1974). The Merced River Valley was cut deeper by the ice, whereas tributary valleys containing less ice were not deepened as much. Therefore, waterfalls, such as Yosemite Falls and Bridal Veil Falls, were left hanging even higher on the canyon walls. Glacial moraines formed in some places, damming the rivers and streams behind them. Ancient Lake Yosemite was formed in such a fashion, about 9 km long and 600 m deep at the head of the valley. The lake filled with sediment, leaving the present level of the valley raised on top of nearly 100 m of gravel, sand, and silt (Rowe 1974). There are still a few minor glaciers within the park, including Lyell Glacier and Maclure Glacier.

Bedrock geology in YOSE is dominated by granitic intrusive rock that has a relatively uniform composition and there have been no large-scale metamorphic events since it was intruded (Huber 1987). Despite the relatively uniform bedrock composition, however, regional water quality sampling surveys have indicated that there is considerable variation in water

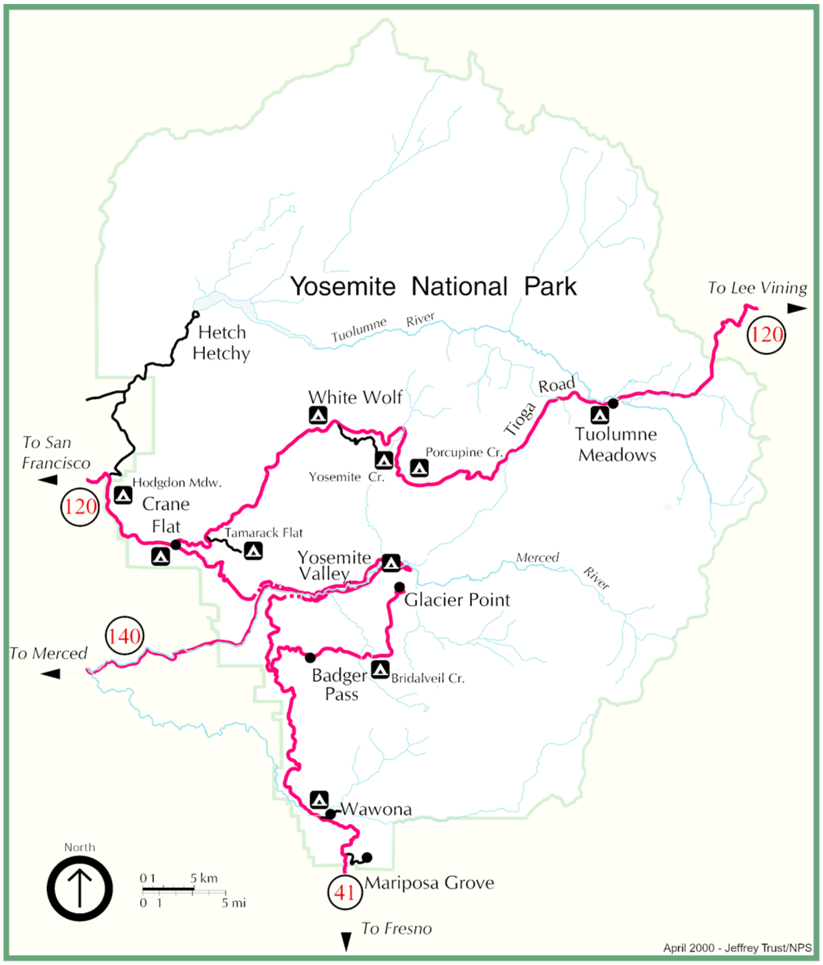


Figure X-1. Map of YOSE.

chemistry throughout the park (Hoffman et al. 1976, Melack et al. 1985, Landers et al. 1987, Clow et al. 1996).

The geology of the Merced River watershed was described by Huber (1987). Most of the rocks in the upper Merced River basin are part of the Tuolumne Intrusive Suite, which is a group of four concentrically arranged plutonic bodies. The rocks within this suite are all granites and granodiorites, although they differ somewhat in texture and composition across the suite. The oldest are found along the margin of the suite and are dark, fine-grained, and composed mostly of plagioclase, biotite, and hornblende. Younger rocks are found towards the center of the suite, in the northern part of the watershed. These rocks are lighter colored, coarser grained, and composed mostly of plagioclase, quartz, and K-feldspar. Other important rock units in the watershed include granites and granodiorites belonging to the intrusive suites of Washburn Lake, Buena Vista Crest, and Yosemite Valley. In general, granites in these suites contain plagioclase, quartz, K-feldspar, and biotite, and the granodiorites contain plagioclase, quartz, biotite, and hornblende. There are also relatively small outcrops of metavolcanic rocks, and limited exposures of metasedimentary rocks and diorite in some portions of the watershed. About 20% of the upper Merced River watershed is underlain by surficial deposits, mostly glacial tills in the

valley bottoms and in the form of lateral and recessional moraines. The till generally has a mineralogy similar to the granitic bedrock present at the higher elevations of the watershed and was probably derived from it.

Massive granitic domes of the Yosemite region were formed by the slow loss of curving shells or scales in a process called exfoliation. The scale thickness can vary from much less than 1 m to over 50 m on different domes. Inducement of exfoliation is caused at least in part by very slow expansion of the granite. The domes at YOSE are not unique to the region, but they are found in greater abundance in the park than anywhere else in the world.

Metamorphosed sedimentary rocks occur within YOSE mainly as remnants. Layers of schist, slate, and limestone that once covered the region's granite layers are now mostly eroded. Some volcanic rocks also occur in the Tuolumne Meadows area.

Soils in the YOSE region are primarily derived from the underlying granitic bedrock. More than 50 soil series are found in the park. General or local variations in soils characteristics are dependent upon such factors as glacial history and more recent influence of weathering and stream erosional processes. Local variations also result from micro climate and vegetation. Most of the high country soils are glacial or residual; alluvial soils are found along streams in some areas. Above 1,830 m elevation are found extensive areas of glacial moraine material, which includes a mixture of sand, glacial flour, and various sized pebbles and boulders.

Limited soils information is available for some portions of the park. The high country meadow soils, such as those found at Tuolumne Meadows, have developed to a 1 m layer of dark brown topsoil as a consequence of the abundance of plant materials and associated decomposition. Yosemite Valley soils are deep sandy soils containing a mixture of colluvial, alluvial, residual, and glacial materials. Colluvial soils include the variously sorted talus slopes. Alluvial soils are associated with stream or flood plain deposits. Residual soils are found throughout the valley and glacial soils are associated mainly with terminal moraines. Detailed soils information is also available for soils in the El Portal and Wawona areas. Soils maps and associated data are not available, however, for most other areas of the park.

Zinke and Alexander (1963) reported on a study of the soils of the Yosemite Valley including their mode of formation and presented correlations between the occurrence of vegetation types and soil types. More recently, the NPS and Natural Resources Conservation Service (NRCS) have been involved in a cooperative effort to conduct a comprehensive soil survey of portions of YOSE. The purpose has been to map previously unmapped areas in YOSE, located in parts of Madera, Mariposa, and Tuolumne Counties. The study area is about 300,000 ha and ranges in elevation from about 550 m to 4,100 m (Taskey 1996).

Soil samples were collected by Parker (1989) at 10-15 cm depth from five points located in each of 95 sites along elevational and topographic gradients in YOSE. Eight forest types were represented in the sampling, from lower montane (1,200 to 1,900 m) to subalpine (2,500 to 3,300 m). Available Ca, Mg, K, and P contents were estimated for pooled soil samples from each site by plasma emission spectroscopy of filtered solutions extracted by mixing 5 g of soil with 20 ml of dilute hydrochloric acid and sulfuric acid. Soil acidity was also measured in a 1:1 paste of soils and distilled water. Both pH and the extracted base cations decreased with elevation ($p < 0.01$), with an abrupt change in the soil acidity and levels of extracted base cations at about 1,900 m.

2. Climate

Climate is variable throughout YOSE and is influenced by elevation and orographic effects. At the high elevations, precipitation occurs mostly as winter snow. Summer temperatures are cool and below-freezing temperatures are common during most months.

The monitoring station for collection of precipitation volume and chemistry has been operating as part of the NADP/NTN network since 1981. It is in Hodgden Meadow at an elevation of 1,800 m. Average annual precipitation at the NADP station was 110 cm/yr between 1982 and 1991. In Yosemite Valley, at an elevation of 1,200 m, the annual average precipitation is 95 cm and increases to 120 cm near Tuolumne Meadows at an elevation of 2,650 m (Clow et al. 1996). Discharge has been measured at the Happy Isles gaging station on the Merced River since 1915. Average annual runoff is 64 cm.

The California Department of Water Resources conducts snow survey measurements at a variety of locations in California. Data from the 1995 California Snow Survey at Tuolumne Meadows indicated average water equivalence in inches on April 1 of 57 cm. At Snow Flat in the Merced River drainage, the average water equivalence on April 1 of that year was 110 cm.

3. Biota

Distribution of plant species and forest types within YOSE is strongly influenced by elevation, climate, soils, and physiography. The large elevational gradient and diversity of environmental conditions within the park contribute to highly diverse plant ecosystems. About 1,400 native vascular plant species occur within the park and they are distributed throughout the major vegetation zones.

The vegetation of YOSE varies along elevational and moisture-topographic gradients. Hall (1997) identified seven general vegetation classes within the park: (1) scrub and chaparral communities, (2) broadleaved upland woodlands, (3) broadleaved upland forests, (4) lower montane coniferous forests, (5) upper montane coniferous forests, (6) subalpine coniferous forests, and (7) grassland, meadow, and other herb communities.

The most common form of chaparral is the montane chaparral community, which covers about 2% of the park in disjunct areas from 1,200 to 3,300 m elevation. The lower portions of this zone, for example in the El Portal area, include blue oak (*Quercus douglasii*), canyon live oak (*Q. chrysolepsis*), gray pine (*Pinus sabiniana*), and mountain mahogany (*Cercocarpus betuloides*), as well as a number of shrub chaparral species, including chamise (*Adenostoma fasciculatum*), manzanita (*Arctostaphylos* spp.), and ceanothus (*Ceanothus* spp.). At higher elevations within this zone, gray pine and canyon live oak are found in open grassy woodlands, and northfacing slopes are characterized by ponderosa pine (*P. ponderosa*) and California black oak (*Q. kelloggii*). The dominant shrub species are greenleaf manzanita (*Arctostaphylos patula*), whitethorn ceanothus (*Ceanothus cordulatus*), and huckleberry oak (*Quercus vaccinifolia*) at higher elevations. This and several other chaparral communities generally form low, dense canopies with a large, volatile fuel component that carries fast-moving, hot fires.

Broadleaved upland woodlands contain broadleaved tree species and a generally discontinuous canopy. Overstory species consist of blue oak, interior live oak (*Q. wislizenii*), gray pine, and western juniper (*Juniperus occidentalis*) on rocky soils. Understory species consist of many grasses and forbs, most of which are non-native. Interior live oak and California black oak woodlands are often found in fairly continuous stands as the only overstory species. Broadleaved upland forests range widely in the park from 600 to 2,700 m. At lower elevations, canyon live oak forest is found on steep, rocky slopes between lower-elevation woodland and higher-elevation forest. Forests on north slopes have a more diverse understory with several shrub species, including California buckeye (*Aesculus californica*), California bay (*Umbellularia californica*), poison oak (*Toxicodendron diversilobium*), and curl-leaf mountain mahogany (*Cercocarpus ledifolius*). At higher elevation, quaking aspen (*Populus tremuloides*) is found in dense stands near streams and other areas with high soil moisture.

Lower montane coniferous forests are widespread at lower to mid elevations in YOSE, and are often generically referred to as mixed conifer forest. The mixed conifer zone occurs from about 900 to 2,100 m elevation. It is dominated by ponderosa pine, sugar pine (*P. lambertiana*), incense cedar (*Calocedrus decurrens*), white fir (*Abies concolor*), and Douglas fir (*Pseudotsuga menziesii*), and it occupies 21% of the park. Ponderosa pine dominates lower slopes and drier south-facing aspects, with typical high, open canopies and a sparse understory. In many areas, it is found mixed with the more shade tolerant incense cedar and white fir, as well as sugar pine, California black oak, and various shrub species. Prior to the 20th century, these stands burned more frequently and had a more open structure and lower stem density with a greater dominance by ponderosa pine. On cooler sites, white fir is increasingly dominant within this forest type. Groves of towering giant sequoia (*Sequoiadendron giganteum*) are found at scattered locations mixed with various combinations of the other lower montane coniferous species.

Upper montane coniferous forests vary considerably in species and stand structure. The upper montane zone is found between about 2,000 and 2,600 m elevation and occupies 23% of the park. Red fir (*A. magnifica*) is dominant on well-drained sites, whereas lodgepole pine (*Pinus contorta*) predominates on moist sites. Dry and rocky areas, especially on south-facing slopes, are occupied by Jeffrey pine (*P. jeffreyii)*, but western white pine (*P. monticola*) and juniper are also common. At the highest elevations, red fir is dominant in nearly pure stands with sparse understory. Western white pine occurs sporadically mixed with red fir and can have dwarf montane chaparral in the understory. Jeffrey pine is found at somewhat lower elevations on well-drained soils and granitic ridges. It can intergrade with both white fir and red fir, and typically has montane chaparral species in the understory. The only deciduous tree that is common in the upper montane zone is quaking aspen. This is also the zone of heaviest snowfall.

Subalpine coniferous forests cover about 40% of the park above about 2,400 m, and are dominated by lodgepole pine, mountain hemlock (*Tsuga mertensiana*), and whitebark pine (*Pinus albicaulis*) in various combinations. Lodgepole pine forest is by far the most common, and occurs in areas with deep snowpack and cool, dry summers, generally with rocky, well-drained soils. It is also found in cold air drainages at lower elevation. Mountain hemlock is found on similar sites but on northerly aspects and in concavities where snowpack is deeper. Whitebark pine is found up to treeline on thin soils and in exposed areas subject to strong winds. It is often found in krummholz form. Subalpine meadows are mixed with these forests at higher elevations and in wet areas, with a wide variety of sedges, grasses, and forbs.

The alpine zone occurs above tree line, which is at about 3,200 m, and it occupies 14% of the park. It is characterized by short grasses, sedges, alpine willow (*Salix* spp.), and a variety of dwarfed and matted flowering plants. The alpine zone has four major types of vegetation communities. These include barren areas such as bedrock, boulder fields, and permanent snow and ice areas; rocky fell fields, which include scattered mixtures of rocks and dwarfed plants; alpine tundra; and riparian areas along streams and lakeshores. Alpine meadows, which are found continuously above treeline, typically have a rich flora that varies according to the amount of snowpack.

The lower-elevation forests and woodlands of YOSE have been altered considerably by human activities during the 20th century. Grazing by livestock has been accompanied by the introduction of numerous exotic grasses and other plant species that now dominate the biomass of the understory at many locations. Fire exclusion has reduced fire frequency, resulting in higher stem densities and altered species distributions in some areas. Prescribed burning has been used to address the concern of increased fuel loading, with most of the treatments in the Yosemite Valley area. In addition, past cutting practices in the Valley have facilitated the spread

of annosus root rot (*Heterobasidion annosum*) and greatly reduced the vigor and growth of ponderosa pine (Peterson et al. 1991).

The park contains three federal category 2 candidate plant species: wooly sunflower (*Eriophyllum nubigenum*), Bolander's clover (*Trifolium bolanderi*), and Hetch Hetchy monkey flower (*Mimulus filicaulis*). The wooly sunflower occurs only in YOSE and the adjacent Stanislaus National Forest, and grows in isolated groups on granite slabs and domes within the upper mixed conifer forest. Bolander's clover is found in isolated locations in wet mountain meadows. In addition, 83 species of plants are listed on the California Native Plant Society list for sensitive species due to their limited distributed or potential for future habitat disturbance from human activities (Yosemite National Park 1993).

Because of the diversity of vegetation zones in the park, a large variety of wildlife species occur, including about 77 species of mammal. Among the most conspicuous mammals are black bear (*Ursus americanus*) and mule deer (*Odocoileus hemionus*), both of which are widespread throughout the park. Black bear populations in YOSE have expanded in recent decades, which was thought to be associated with a greater abundance of unnatural food sources throughout the park. Mule deer are common at the lower elevations and occur up to the higher mountain meadows during summer. California bighorn sheep (*Ovis canadensis*) are rare in the park, although at one time they were relatively common.

A large number of mammal species are commonly observed within YOSE. Ground squirrels and chipmunks are locally common and include the California ground squirrel (*Spermophilus beecheyi*), Belding's ground squirrel (*Spermophilus beldingi*), and golden mantle ground squirrel (*Spermophilus lateralis*). Several species of chipmunks, which are indistinguishable to most viewers, occur in the park. Most are common. These include Merriam's chipmunk (*Eutamias merriami*), alpine chipmunk (*E. alpinus*), lodgepole chipmunk (*E. speciosus*), as well as long-eared (*E. quadrimaculatus*), yellow pine (*E. amoenus*), and Townsend's (*E. townsendii*) chipmunks. Two species of diurnal tree squirrels occur in the park: Douglas' squirrel (*Tamiasciurus douglasii*) and western gray squirrel (*Sciurus griseus*). Both the yellow-bellied marmot (*Marmota flaviventris*) and pika (*Ochotona princeps*) are common in rocky areas and talus slopes at high elevation. Racoons (*Procyon lotor*) are common within the park and the ringtail (*Bassariscus astutus*) is also present but is seldom seen. Mustelids include both the striped (*Mephitis mephitis*) and spotted (*Spilogale putorius*) skunks, marten (*Martes americana*), long-tailed weasel (*Mustela frenata*), ermine (*Mustela erminea*), fisher (*Martes pennanti*), wolverine (*Gulo gulo*), badger (*Taxidea taxus*), and river otter (*Lutra canadensis*). Both coyote (*Canis latrans*) and gray fox (*Urocyon cinereoargenteus*) are common and red fox (*Vulpes vulpes*) may also occur in the higher meadows. Both mountain lion (*Felis concolor*) and bobcat (*Felis rufus*) also occur throughout the park, but are seldom seen. Other mammal species which may be encountered within YOSE include the brush rabbit (*Sylvilagus bachmani*), black-tailed jackrabbit (*Lepus californicus*), white-tailed jackrabbit (*Lepus townsendii*), snowshoe hare (*Lepus americanus*), porcupine (*Erethizon dorsatum*), dusky-footed woodrat (*Neotoma fuscipes*), bushy-tailed woodrat (*Neotoma cinerea*) and mountain beaver (*Aplodontia rufa*; Van Gelder 1982). Thirteen bat species are found in the lower elevation chaparral and mixed conifer ecosystems, as are five species of shrew and one species of mole. About 224 bird species have been observed in the park. YOSE has a particularly large number of native reptile and amphibian species compared to most mountain regions of the West. There are 13 species of snake, 9 lizards, 1 turtle, 2 toads, 1 tree frog, 3 frogs, and 3 salamanders.

Some wildlife species within YOSE are, or have been, classified as threatened and endangered. The American peregrine falcon (*Falco peregrinus annatum*) and the southern bald

eagle (*Haliaetus leucocephalus leucocephalus*) can be sighted in some areas. The California spotted owl (*Strix occidentalis*) has also been observed in the Yosemite Valley area.

Fish found in YOSE have largely been introduced. Prior to widespread stocking for improvement of sport fishing, the native fish fauna was restricted both in number of species and in range. The last period of glaciation eliminated all fish from the high country. The waterfalls remaining on all the rivers after retreat of the glaciers prevented repopulation by upstream migration. Thus, only the lower drainage systems of the Merced and Tuolumne Rivers were populated with fish when Europeans first arrived. Rainbow trout (*Oncorhynchus mykiss*) and Sacramento sucker (*Catostomus occidentalis*) were abundant and Sacramento squaw fish (*Ptychocheilus grandis*), hardhead (*Mylopharodon conocephalus*), California roach (*Hesperoleucus symmetricus*), and riffle sculpin (*Clinocottus*) were also present. Chinook salmon (*Oncorhynchus tshawytscha*) occurred in these lower drainages.

Most of the lakes and streams in YOSE were historically fishless due to the presence of natural barriers to fish migration. Fish stocking occurred in many of the park's fresh waters throughout much of the 20th century, and resulted in alteration of the natural ecological communities.

Conflicts between the basic NPS mandate to preserve and restore natural conditions within the national parks and the continued maintenance of an unnatural sport fishery in YOSE had long been recognized. Phasing out of fish stocking programs was adopted as a policy for natural areas of the NPS in 1969. YOSE began a five-year phase out of fish stocking in 1972 (Elliott and Laughlin 1992). A list of 127 lakes in YOSE that were believed to contain fish populations as of 1985 was provided by Johnston (1985b). A summer fish population survey was conducted in 1992 of the upper Merced River watershed (Bertetta 1992). Over 42 lakes, streams, and sections of river reach were surveyed for fish populations. Five species of trout existed within the survey area: rainbow trout, eastern brook trout (*Salvelinus fontinalis*), golden trout (*Oncorhynchus aquabonita*), brown trout (*Salmo trutta*), and cutthroat trout (*Oncorhynchus clarki*).

Distribution and abundance of lotic invertebrates were determined from five habitats of the Merced River in Yosemite Valley during the period of 1992 to 1995 by Carter and Fend (1997). Lotic benthic invertebrates were collected from riffles, runs, pools, wood, and margin habitat from eight different sites. A total of 338 taxa were collected during the four year study.

Several species of amphibians have experienced significant declines in the Sierra Nevada, including YOSE, in recent years. Category 2 candidate amphibian species found in YOSE include the California red-legged frog (*Rana aurora draytoni*), which is now listed as threatened; limestone salamander (*Hydromantes brunus*); Mount Lyell salamander (*Hydromantes platycephalus*); foothill yellow-legged frog (*Rana boylei*); mountain yellow-legged frog (*Rana muscosa*); and Yosemite toad (*Bufo canorus*).

Introduced predatory fish, possibly interacting with drought-induced loss of refuge habitat, have contributed to the decline of some amphibian species in the Sierra Nevada. However, the overall cause of the dramatic losses of amphibians remains unknown (Drost and Fellers 1996).

4. Fire

Within YOSE, prescribed fires, wildfires, and campfires can have significant impacts on air quality and cause temporary reductions in visibility and violations of state and federal fine particulate standards. However, curtailment of prescribed natural fires or prescribed burns for the purpose of reducing smoke impacts on visitors and residents may have significant long-term ecological impacts. An integrated approach to smoke management and innovative approaches to

controlling smoke levels are required in order to balance the requirements of the Clean Air Act with NPS mandates for managing natural ecosystems.

It is currently acknowledged that Native Americans in the Yosemite area used fire to modify the environment, especially in Yosemite Valley and around major village sites. Subsequently, state land grant guardians and federal administrators of the national parks followed the traditions of prescribed burns, at least in Yosemite Valley. These fires were intended to clear underbrush, keep the forest open, and prevent conifer invasion of the meadows. Burning continued intermittently until 1930, but there was no prescribed burning within the park between 1930 and 1970. The modern prescribed burning program was initiated in 1970, and was concentrated in Yosemite Valley, Wawona, Foresta, and the sequoia groves. The management of natural fires in YOSE began in 1972 with the establishment of natural fire zones comprising about 25% of the park. These zones were initially largely in the alpine and subalpine portions of the park and a few lodgepole pine areas. The natural fire zone was gradually expanded as it became evident that natural fire could be restored to the upper elevation vegetative communities without significant danger of very intense, uncontrollable fires that might endanger personnel safety or property. Since about 1978, the program goal has expanded to burn ecologically significant areas on a parkwide basis.

All of the major forest and chaparral plant communities in the park have evolved under the continual influence of fire regimes. A multitude of adaptations to these fire regimes are evident at the species, community, and ecosystem levels (Yosemite National Park 1990). Fire is also important in the ecology of the lowland, grass, and herb communities. Estimated historical fire return intervals suggest that about 6,500 ha of the park may have burned annually under natural conditions in the past. Fire regimes vary dramatically, however, between vegetation types. Lightning strike occurrence generally varies directly with elevation, but the percentage of strikes that start fires and the size and extent of individual fires vary inversely with elevation. This is because lower elevation vegetation communities tend to be much more flammable than the higher elevation communities. As a consequence, the lower chaparral community typically only supports a small number of very large fires, whereas the subalpine forest supports a large number of smaller fires. Lightning is an important ignition source in the middle elevations, and is a major phenomenon in YOSE during the summer months. During the period from 1931 to 1976, the average number of fires per year was 36.5, with a range from one fire in 1954 to 121 fires in 1967. Less than 1% of the fires occurred below 1,300 m elevation in the chaparral zone; 37% occurred between 1,300 and 2,000 m in the mixed conifer zone; 40% occurred between 2,000 and 2,700 m in the red fir zone; 20% occurred between 2,200 and 3,300 m in the lodgepole pine zone; and the remaining 2% occurred above 3,300 m in the subalpine and alpine zones (Van Wagtendonk 1977).

B. EMISSIONS

YOSE is located adjacent to the San Joaquin Valley (SJV), which experiences high levels of ozone during summer months. Concentrations of gases and particulate air pollutants are transported to the park from the valley by prevailing westerly winds. It is not uncommon during the summer to document several days when ozone levels exceed the state health standard. Additionally there are many other sources of air pollutants within the park, including automobiles, buses, construction activities, campfires, wood-burning stoves, natural and prescribed fire, and road dust. Often these pollution sources within the park may combine with weather conditions and cause PM_{10} levels to exceed state and sometimes federal health standards.

Ozone, acid precipitation, particulates, and visibility are currently monitored in the park. Acid deposition is sampled at Hodgdon Meadow and Turtleback Dome. Ozone monitoring occurs at Wawona, Turtleback Dome, Yosemite Valley, near the Hetch Hetchy entrance, at Crane Flat, and at four sites along the Tioga Road up to Tioga Pass. Additional gaseous pollutant sampling is done on Turtleback Dome with dry deposition and Improve samplers. Precipitation, temperature, and wind data are collected throughout the year at Hetch Hetchy, South Entrance, Big Oak Flat, Yosemite Valley, and Turtleback Dome. Snow depth and water content are measured monthly at several locations in the park.

The goal of the air program is to preserve, protect and enhance air quality and air quality related values. Clean air is one of the most important natural resources in the park. In particular, viewing natural scenery and park features is a primary activity of most visitors to YOSE.

YOSE is located on the boundary of two air basins, Mountain Counties and SJV, and is potentially exposed to pollutants transported from the SJV and other areas. Approximately 9% of the state's population lives in the eight counties of the SJV air basin (SJVAB), and emission sources within the SJVAB account for about 14% of total statewide emissions (Alexis et al., 1999). Emissions in the SJV derive from a number of moderate-sized urban areas, primarily located along the Highway 99 corridor. Since 1980, population growth in the SJV has been more rapid than in other parts of California, partially offsetting the effects of emission-control programs (Alexis et al., 1999). The Mountain Counties air basin (MCAB) includes the western slope of the Sierra Nevada, an area with relatively low population (~1% of the state total) and emissions (~ 3% of the state total; CARB 1998b). Emission levels from counties within 140 km of YOSE are listed in Table X-1. The principal species of concern are ozone precursors (NO_x and ROG) and PM. SO_2 emissions are not high.

YOSE is located within Madera, Mariposa, and Tuolumne counties. Major point sources are not numerous in these counties, nor in most other counties of the SJV or Mountain Counties air basins (Figures IIB-3 through IIB-6). Sources that emit at least 100 tons/yr of ROG, NO_x, PM_{10}, or SO_2 are located near Jamestown and Standard in Tuolumne County, and further away, near Madera and Chowchilla in western Madera County; none are located within Mariposa

Table X-1. 1995 Emissions from counties within 140 km of YOSE. (Source: CARB Almanac 1999b; SO_x from CARB Emissions Website, 1999a.) Units are 1000 tons/year.

County	NO_x	ROG*	PM_{10}	CO	SO_x
Amador	3.3	3.7	3.3	21.2	0.4
Calaveras	1.5	3.7	3.7	23.4	0.0
El Dorado [1] (Mountain)	5.5	6.9	5.5	54.8	0.4
Placer [1] (Mountain)	1.5	1.5	2.6	9.9	0.0
Mariposa	0.7	1.8	2.6	10.6	0.0
Madera	11.3	7.7	8.8	41.2	0.4
Tuolumne	2.9	4.7	3.7	34.3	0.4
Fresno	39.8	39.4	45.3	205.1	3.7
Merced	15.3	10.2	17.5	75.9	0.7
San Joaquin	26.6	28.1	16.4	143.1	1.8
Stanislaus	17.2	16.8	14.2	102.9	1.1
Tulare	16.8	18.3	19.3	113.9	0.4
Kings	9.5	8.0	13.5	36.5	0.4

* Reactive Organic Gases
[1] Portion of the county in the air basin

County. As of 1996, stationary sources accounted for 12% of ROG emissions, 36% of NO_x emissions, and 7% of PM_{10} emissions in Madera, Mariposa, and Tuolumne counties (CARB 1998b). Mobile sources dominated NO_x emissions, whereas area sources (road dust, construction, and farming operations) dominated PM emissions.

An inventory of in-park emissions has recently been compiled by the NPS-Air Resources Division. The results are presented in Table X-2.

Table X-2. Summary of 1998 stationary and area, and mobile source emissions (tons/yr) at YOSE.

Activity	Particulates	Sulfur Dioxide	Nitrogen Oxides	Carbon Monoxide	VOCs	POM
Stationary and Area Source Emissions (1)						
Stationary Combustion Sources						
Heating units	0.31	9.90	6.53	1.45	0.14	0.0823
Generators	0.06	0.06	0.87	0.19	0.07	0.0000
Woodstoves	1.52	0.02	0.11	11.11	10.08	0.0001
Commercial Food Preparation (2)	2.40	–	–	–	–	1.6800
Combustion Emission Subtotal	4.30	9.97	7.51	12.75	10.29	0.0824
Area Sources						
Campfires	8.36	–	1.97	68.88	9.35	–
Prescribed Burning	31.47	–	6.79	292.04	12.83	–
Wastewater Treatment Plant	–	–	–	–	0.55	–
Area Source Emission Subtotal	39.83	–	8.75	360.92	22.73	–
TOTALS	44.13	9.97	16.26	373.67	33.01	0.0824
Mobile Source Emissions at Yosemite Valley						
Road Vehicles						
Visitor Vehicles	–	–	–	–	–	–
NPS/GSA Road Vehicles	–	–	–	–	–	–
YCS Concessioner Vehicles	–	–	–	–	–	–
Vehicle Emission Subtotal	21.91	0.00	12.76	450.95	42.80	–
Mobile Source Emissions at Yosemite NP						
Road Vehicles						
Visitor Vehicles	154.78	0.00	90.05	3184.99	303.92	–
NPS/GSA Road Vehicles (1)	0.00	0.00	0.00	0.00	0.00	–
Concessioner Vehicles (1)	0.00	0.00	0.00	0.00	0.00	–
Vehicle Emission Subtotal	154.78	0.00	90.05	3184.99	303.92	–
Nonroad Vehicles						
NPS Nonroad Vehicles	0.00	0.00	0.00	0.00	0.00	–
TOTALS	154.78	0.00	90.05	3184.99	303.92	–

(1) Park-wide data not available
(2) PM2.5 from charbroiling

C. MONITORING AND RESEARCH ACTIVITIES

1. Air Quality

a. Monitoring Sites

Ozone, SO_2, fine particulate, wet deposition, and dry deposition have all been monitored within YOSE (Table X-3). The CADMP dry-deposition site is no longer operational, but a CASTNet site is now located within the park for monitoring dry deposition. For both the CADMP and CASTNet monitors, deposition is not monitored directly, but is rather calculated from ambient concentration measurements.

Table X-3. Air quality monitoring at YOSE.

Species	Site within park	Site within 50 km.
Ozone, hourly	CASTNet	
Ozone, passive	NPS	
SO_2	NPS	
PM_{10}	IMPROVE	
$PM_{2.5}$	IMPROVE	
Wet deposition	NADP	
Dry deposition	CASTNet	
Visibility	IMPROVE	

b. Wet Deposition

Wet S deposition in YOSE averaged 0.6 to 0.8 kg/ha/yr as S (equivalently, 1.8 to 2.4 kg/ha/yr as SO_4^{2-}), with individual years ranging as high as 1.1 kg/ha/yr in some monitoring locations (Table X-4). Multi-year NO_3^- and NH_4^+ deposition rates were each in the range of 0.7 to 1.0 kg/ha/yr as N, yielding total wet N deposition rates of 1.5 to 1.9 kg/ha/yr (Table X-4). For individual years, total wet N deposition rates were as high as 2.6 kg/ha/yr in YOSE (Table X-4).

Annual-average H^+ concentration in precipitation ranged from 3.2 to 4.4 µeq/L (pH 5.49 to 5.36; Table X-5). These values are not substantially more acidic than the expected value of water in equilibrium with atmospheric CO_2 (pH ~ 5.7). However, individual storm events or meltwater may exhibit higher acidity, posing questions about potential transient impacts (see later discussions).

c. Occult/Dry Deposition

The CADMP co-located wet and dry deposition samplers at Turtleback Dome (Blanchard et al., 1996). Dry deposition was calculated from measurements of the ambient concentrations of both gas-phase and particulate species (Table X-6). Mean dry deposition rates of oxidized N species summed to ~0.8 kg/ha/yr (as N), comparable to the multiyear mean wet NO_3^- deposition of 0.8 kg/ha/yr (as N) at the same location. The CADMP wet and dry deposition samplers also indicated that dry S (SO_2 plus aerosol SO_4^{2-}) deposition rates (~0.3 kg/ha/yr) were about half the rate of wet S deposition (Blanchard et al., 1996). Dry NH_3 plus aerosol NH_4^+ deposition was about 0.4 times the rate of wet NH_4^+ deposition (Blanchard et al., 1996). The total dry N

Table X-4. Wet deposition of S and N at CADMP, UCSB and NADP sites in YOSE. (Source: Blanchard et al. 1996; Dwight Oda and Brent Takemoto, CARB, 1999, personal communication). Units are kg/ha/yr.

Site	Water Year*	Sulfur	NO_3-N	NH_4-N	Total Inorganic N
Tioga Pass UCSB (2993 m)	1991	0.6	0.7	0.7	1.4
	1992	0.5	0.6	0.6	1.2
	1993	1.1	0.9	1.0	2.0
	Average	0.7	0.7	0.8	1.5
Yosemite CADMP (1395 m.)	1985	0.9	1.2	1.1	2.3
	1987	1.0	1.3	1.3	2.6
	1988	0.5	0.9	0.9	1.8
	1989	0.4	0.6	0.6	1.1
	1991	0.4	0.5	0.6	1.2
	1992	0.5	0.7	0.8	1.5
	1993	0.7	0.8	0.8	1.6
	1994	0.4	0.7	0.8	1.5
	1995	0.9	1.2	1.3	2.5
	Average	0.6	0.9	0.9	1.8
Yosemite NADP (1408 m)	1990	0.9	1.2	1.3	2.4
	1992	0.6	0.8	0.6	1.4
	Average	0.8	1.0	0.9	1.9

* The CARB water year is July 1 through June 30. For instance, water year 1995 is for July 1, 1994 through June 30, 1995.

Table X-5. Wetfall chemistry at CADMP, NADP and UCSB sites in YOSE. Units are µeq/L, except precipitation (cm). Source: Blanchard et al. 1996; Dwight Oda and Brent Takemoto, CARB, 1999; Melack et al., 1997.

Site	Water Year*	Prec. Amt.	H^+	SO_4^{-2}	NH_4^+	NO_3^-	Ca^{+2}	Mg^{+2}	Na^+	K^+	Cl^-
Tioga Pass UCSB (2993 m)	1991	97.0	5.5	3.6	5.1	5.2	3.3	0.7	2.4	1.4	1.8
	1992	81.1	5.4	3.8	5.3	5.3	2.9	0.8	1.2	1.1	1.4
	1993	224.0	4.8	3.0	3.2	3.0	2.4	0.6	1.1	1.0	1.3
	Ave.	134.0	5.2	3.5	4.5	4.5	2.9	0.7	1.5	1.2	1.5
Yosemite CADMP (1395 m.)	1985	79.7	4.3	7.1	10.7	10.2	8.4	2.9	6	1.6	6.4
	1987	68	8	9	13.8	13.6	3.4	1.5	4.3	1.3	5
	1988	65.5	5.3	4.9	9.4	9.8	4.9	2	2.7	1.1	3
	1989	51.1	5.2	5.1	7.8	7.9	3.2	1.2	2.5	0.5	4.5
	1991	45.5	3.6	5.6	9.7	8.4	3.9	1.9	3.9	0.7	4.6
	1992	57.0	3.4	5.9	9.9	9.0	5.8	2.8	2.6	0.7	2.5
	1993	85.0	3.2	5.0	6.5	6.6	9.4	3.2	2.7	1.0	3.0
	1994	45.1	4.0	5.3	12.4	11.1	17.5	3.6	2.2	0.7	2.4
	1995	143.0	4.4	4.1	6.6	6.0	18.3	3.9	3.4	1.3	3.1
	Ave.	75.1	3.7	5.2	9.0	8.2	11.0	3.1	3.0	0.9	3.1
Yosemite NADP (1408 m.)	1990	90.3	3.8	6.3	9.9	9.1	1.6	0.7	2.3	0.2	2.7
	1992	74.1	4.1	5.1	5.7	7.6	2.4	0.9	3.6	0.2	2.9
	Ave.	82.2	4.0	5.7	7.8	8.4	2.0	0.8	3.0	0.2	2.8

* The CARB water year is July 1 through June 30. For instance, water year 1995 is for July 1, 1994 through June 30, 1995.

Table X-6. Long-term annual averages of calculated dry deposition fluxes at YOSE (Turtleback Dome) using data from 1988-94. Units are kg/ha/yr as SO_2, ozone, NO_2, etc. The averages were constructed by weighting four seasons equally. Since no first-quarter (January-March) or fourth-quarter (October-December) periods met the completeness criteria, the annualized flux rates represent annual deposition occurring if the deposition rates during the period April through September persisted throughout the year. Source: Blanchard et al (1996).

Gas-Phase Species					Particulate		
SO_2	Ozone	HNO_3	NO_2	NH_3	NO_3^-	SO_4^{2-}	NH_4^+
0.28	46.82	3.36	0.14	0.37	0.15	0.41	0.1

deposition rate of 1.2 kg/ha/yr (Table X-6) was somewhat less than the wet N deposition rate of 1.8 kg/ha/yr (Table X-4).

The CASTNet dry-deposition monitoring site located at Turtleback Dome began operating September 25, 1995. The monitoring instrument measures ambient concentrations of gases and particles, and EPA uses a computer model to calculate the dry-deposition rates from the measurements. The first calculations of dry-deposition rates for this site were released by EPA in November 2000. For the years 1996 through 1999, the calculated annual dry-deposition rates of N and S ranged from 0.9 to 1.1 kg N/ha/yr and 0.24 to 0.28 kg S/ha/yr, respectively. When combined with the wet deposition measurements from the nearby NADP/NTN site (located 16.5 km from the dry-deposition monitor), the data indicate that the annual total deposition rates of N and S ranged from 2.1 to 6.0 kg N/ha/yr and 0.58 to 1.5 kg S/ha/yr, respectively, over the period 1996 through 1999. The average total N and S deposition rates over the four-year period were 3.6 kg N/ha/yr and 0.96 kg S/ha/yr, respectively.

Collett et al. (1990) selected two sites in the Sierra Nevada to serve as locations for monitoring the chemical composition of cloud water. The sites were situated at Lower Kaweah at an elevation at 1,856 m in SEKI and at Turtleback Dome, elevation 1,590 m, in YOSE. Both sites were located in open areas which easily intercept approaching clouds. Cloudwater interception was recorded between September 1987 and August 1988. The two peak months for cloudwater interception were November and April. The number of hours that the sites were immersed in clouds during the study year was much less, however, than has been estimated for some other sites in the eastern U.S. and Canada (Collett et al. 1990). The pH of the cloudwater samples collected in SEKI ranged from 3.9 to 6.5, with fall samples tending to be somewhat more acidic than those collected during the winter or spring. The inorganic composition of the cloudwater was dominated by NO_3^-, SO_4^{2-}, and NH_4^+. Samples with large excesses of NH_4^+ relative to the sum of NO_3^- and SO_4^{2-} had pH values generally greater than 5, whereas those with excesses of NO_3^- and SO_4^{2-} tended to be more acidic. Cloudwater samples at Turtleback Dome in YOSE tended to be more acidic than those collected at Lower Kaweah during the spring of 1988. In SEKI in particular, the neutralization of cloudwater acidity by NH_3 was largely responsible for maintaining the pH of cloudwater above levels commonly seen at sites in the eastern United States.

Collett et al. (1989) had previously suggested that the deposition of NO_3^-, SO_4^{2-}, and NH_4^+ by cloudwater interception at Lower Kaweah might be comparable to amounts introduced by precipitation. This suggestion was based on the observations that the average concentrations of NH_4^+ and NO_3^- in cloudwater were more than 10 times those observed in precipitation, and the average concentrations of SO_4^{2-} were more than three times those observed in precipitation. Based on the data that were collected by Collett et al. (1990), those earlier estimates were revised. Estimates of total deposition from cloudwater interception appear to be much lower than previously predicted, especially for SO_4^{2-}. Revised cloudwater deposition estimates were, however, still significant compared to precipitation inputs, and the estimated cloudwater deposition inputs of NH_4^+ and NO_3^- at this site were greater than 50% of the respective wet deposition inputs.

d. Gaseous Monitoring

Mean summer ozone concentrations from passive samplers at five sites are shown in Table X-7. During 1998, the only year with data for all sites, the highest ozone levels occurred at the Tioga Road and Tioga Pass sites, located at 2,393 m and 3,030 m elevation, respectively. Yet, at an intermediate location (Tuolumne Meadows, 3,002 m), mean ozone was nearly 20 ppb lower (Table X-7). The lowest mean ozone concentration occurred at the southernmost site (Wawona

Table X-7. Summer average hourly ozone concentrations (ppb) at passive sampling sites within YOSE (source: Dr. John D. Ray, National Park Service, Air Resources Division, NPS Passive Ozone website, 1999).

Sample Locations	Elevation (m)	1997	1998	1999
Yosemite NP-Camp Mather	1432	42.4	40.2	45.6
Yosemite NP-Wawona Valley	1280	35.4	31.7	38.3
Yosemite NP-Tioga Road	2393		51.2	53.0
Yosemite NP-Tuolumne Meadows	3002		33.4	35.9
Yosemite NP-Tioga Pass	3030		51	48.0

Valley), and Camp Mather (western side) recorded an intermediate mean ozone level. Further data will be needed to more definitively establish the spatial patterns of ozone within the park, including a possible correlation between elevation and ozone level.

Ozone concentrations and exposures from continuous samplers for the period 1992-1997 are shown for three sites in YOSE in Table X-8. The maxima were below the federal one-hour ozone standard (120 ppb) during all years, but the California hourly-ozone standard of 90 ppb was exceeded at all sites. Like the passive ozone samplers, the means at Wawona Valley were lower than those at Camp Mather; however, the maxima were greater at Wawona Valley. Turtleback Dome exhibited means comparable to Camp Mather and maxima comparable to those at Wawona Valley.

Maximum and mean 24-hour integrated samples for SO_2 are listed in Table X-9 for the period 1991-1996. SO_2 measurements were discontinued after 1996 due to concerns about their accuracy. The measurements are considered sufficiently accurate to show that the SO_2 concentrations were well below the levels at which plant injury has been documented, ~40 to 50 ppb 24-hour average and 8-12 ppb annual average (Peterson et al, 1992).

2. Aquatic Resources

a. Water Quality

Surface waters of YOSE comprise a significant natural resource because many have not been altered from their natural condition. Numerous lakes occur within the park along with an intricate network of rivers, streams, and waterfalls, which together comprise a variety of aquatic habitats throughout an elevational gradient from about 600 to 4,000 m.

There are about 268 lakes within the park and 92 rivers and streams (1,415 km). Stream runoff varies considerably throughout the year, with most runoff occurring from mid-April through July. Two major reservoirs are located within the park, both of which provide water for San Francisco. Hetch Hetchy Reservoir was impounded by O'Shaughnessy Dam in 1913 and Lake Eleanor was impounded by Eleanor Dam in 1918. The U.S. Geological Survey operates a number of stream flow gaging stations on the Tuolumne and Merced Rivers in or near the park.

An amendment passed in 1987 to the Wild and Scenic Rivers Act of 1968 designated 70 km of the mainstem of the Merced River and four miles within the El Portal administrative site as a Wild and Scenic River. The bill also designated 40 km of the South Fork within YOSE.

Table X-8. Summary of ozone concentrations and exposure from YOSE monitoring sites (Source: Joseph and Flores, 1993; National Park Service, Air Resources Division 2000).

Site	Year	Maximum Daily 1-hour Value (ppbv)	2nd Highest Daily 1-hour Value (ppbv)	Number of Daily Maximum 1-hour Values Greater Than or Equal to 125 ppb	3-Year Average Number of Exceedences	Maximum 9am-4pm Average (ppbv)	Sum06 (ppbv-hour)[a]	Number of Valid Hours of Ozone Measurements
Camp Mather	1988	96	95	0	na	84	26,000	5,865
	1989	87	82	0	na	74	21,000	7,284
	1990	86	86	0	0	81	19,000	4,835
	1991	88	86	0	0	76	31,000	4,958
	1992	86	81	0	0	na	21,000	7,970
	1993	91	91	0	0	na	38,000	8,085
	1994	97	95	0	0	na	49,000	6,843
	1995	97	92	0	0	na	29,000	4,243
	1996	94	91	0	0	na	32,000	1,956
Turtle-back Dome	1992	100	na	na	na	84	na	na
	1994	113	111	0	na	92	62,000	8,227
	1995	114	104	0	0	86	42,000	7,903
	1996	107	106	0	0	84	57,000	8,099
	1997	111	107	0	0	88	27,000	8,000
	1998	106	104	0	0	87	51,000	7,287
	1999	96	95	0	0	81	50,000	7,390

Table X-8. Continued.								
Site	Year	Maximum Daily 1-hour Value (ppbv)	2nd Highest Daily 1-hour Value (ppbv)	Number of Daily Maximum 1-hour Values Greater Than or Equal to 125 ppb	3-Year Average Number of Exceedances	Maximum 9am-4pm Average (ppbv)	Sum06 (ppbv-hour)[a]	Number of Valid Hours of Ozone Measurements
Wawona Valley	1987	133	116	1	na	105	55,000	6,128
	1988	119	113	0	na	90	52,000	7,958
	1989	111	99	0	0	87	37,000	8,142
	1990	120	116	0	0	94	53,000	7,334
	1991	105	98	0	0	88	30,000	7,528
	1992	111	108	0	0	na	38,000	8,129
	1993	98	95	0	0	na	26,000	6,733
	1994	95	92	0	0	na	21,000	7,890
	1995	110	103	0	0	na	32,000	8,051
	1996	103	101	0	0	na	36,000	6,694

[a] Maximum 8 am - 8 pm 90-day rolling average

Table X-9. Maximum and mean SO_2, from 24-hour-resolution samples at YOSE. Samples are collected every 3-4 days, unless noted. (Source: NPS Air Resources Division). Units are ppb.

SO_2	1991	1992	1993	1994	1995	1996
Maximum	0.63	0.04*	0.36*	0.45	0.32	0.34**
Mean	0.15	0.02*	0.13*	0.1	0.1	0.08**

na Not available
* Less than 50 samples collected for the year
** 50-75 samples collected for the year

Water quality surveys conducted during the early 1980s by the U.S. Geological Survey and in 1985 by the Environmental Protection Agency indicated that most waters sampled in YOSE represent relatively pristine water quality conditions. Water resource management within the park primary focuses on the development of a balance between regional population increases and park use dynamics and their associated effects on water resources on the one hand, and the preservation of these resources in their pristine condition on the other hand. Increased visitation and use has expanded the demands for domestic water supplies and associated waste water treatment within the park and also has increased the possibility of adverse effects on water quality in the back country. Major domestic water quality and waste water treatment plants are in compliance with both state and federal regulations covering water quality, but there are still several areas of concern, especially in the heavily-used back country and in outlying areas of the park.

The result of surface water quality data retrievals for YOSE from five national data bases were summarized by NPS-WRD (1994). The Storage and Retrieval Data Base Management System (STORET) covered 109 water quality sampling stations within the park boundary. pH was measured 973 times at 107 monitoring stations between 1968 and 1991. One hundred and seven observations at 58 monitoring stations had pH $\leq$ 6.5. Total alkalinity was determined by low-level Gran analysis 11 times at 11 lake monitoring stations in 1985. All observations were below 200 μeq/L.

The STORET database includes 10 lakes in YOSE that had reported conductivity $\leq$ 5 μS/cm. Full ion chemistry data are not available in STORET for these lakes, so it is not possible to judge if the reported conductivity values are reasonable. Several had reported conductivity of only 1 μS/cm. These data suggest that YOSE contains a number of highly dilute and presumably acid-sensitive lakes.

A 40-km stretch of the Merced River was studied by Hoffman et al. (1976) to evaluate its water quality. pH values were generally in the range of 6.5 to 7. Specific conductance was generally above 10 μS/cm. The authors reported alkalinity values of 2 to 24 mg/L as $CaCO_3$, which would correspond to ANC in the range of about 40 to 500 μeq/L).

A three-year water quality study was carried out by the U.S. Geological Survey during the period 1981 through 1983. pH and other physical and biological data were collected from 17 lakes and 31 streams throughout the park. The minimum and maximum pH values recorded were 5.6 and 7.9, respectively. In a separate study, 15 lakes and streams were sampled in 1983 by the USGS within the park, 6 of which had specific conductance values of about 1 or 2 μS/cm; pH varied from 5.4 to 7.9.

The surface water draining granitic bedrock in YOSE shows considerable variation in chemical composition despite the fact that bedrock chemistry is relatively homogeneous. Other geological factors, including jointing of the bedrock and the distribution of glacial till, appear to exert strong controls on water chemistry (Clow et al. 1996). Water chemistry data from three surface water surveys in the upper Merced River watershed conducted in August 1981, June 1988, and August 1991 were analyzed and compared by Clow et al. (1996) with mapped geological, hydrological, and topographic features to try to identify solute sources and major processes that control water chemistry within the watershed during base flow periods. A total of 23 sites were sampled during the three surveys. The water at most of the sample sites was dilute, with ANC values ranging from 26 to 77 µeq/L at most sites. ANC was considerably higher in two of the sampled subwatersheds, however, ranging from 51 to 302 µeq/L (Table X-10). Base cations and silica were also substantially higher in those same two subwatersheds. Concentrations of weathering products in surface water were correlated with the fraction of each subwatershed that was underlain by surficial material, primarily glacial till.

The Upper Merced River watershed contains over 100 lakes and ponds, most of which are located in headwater cirques. Vegetation covers about 45% of the watershed, including a red fir forest grading into a mixed subalpine forest above 2,750 m, and alpine vegetation areas above about 3,200 m. SO_4^{2-} concentrations in the Merced River at the Happy Isle gaging station tended to be lowest during snowmelt and increased gradually throughout the summer and autumn (Figure X-2). Clow et al. (1996) interpreted the amount of seasonal variability in SO_4^{2-}

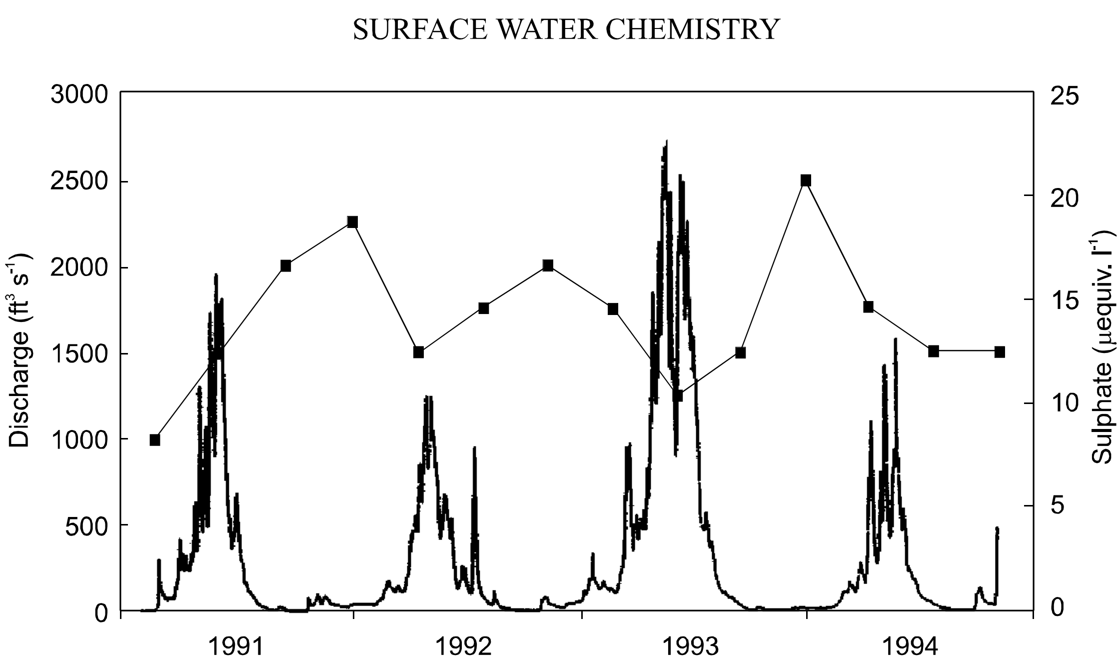


Figure X-2. Time series plot of SO_4^{2-} concentration and discharge at the Happy Isles gage from 1991 to 1994 (Source: Clow et al. 1996).

Table X-10. Major ion chemistry and discharge for samples collected during the 1991, 1988, and 1981 sampling synoptics. Discharge is given in m^3/s and all chemical constituents are expressed in units of µeq/L, except for SiO_2 which is in µmol/L. (Source: Clow et al. 1996)

Site No.	Sampling Site	Q	pH	Ca^{2+}	Mg^{2+}	Na^+	K^+	Cl^-	NO_3^-	SO_4^{2-}	HCO_3^-	SiO_2
1991 Synoptic												
1	Merced R. at Happy Isles guage	1.77	7.08	89.8	14.8	65.2	7.2	56.5	2.1	10.6		70.0
7	Merced R. below Merced HSC	1.56	6.92	79.8	9.1	52.2	4.3	62.1	2.1	12.1	59.4	36.7
8	Merced R. above Merced HSC	1.56	6.65	79.8	5.8	56.5	7.9	67.8	2.1	11.9	65.0	36.7
13	North Fork Lyell Creek	0.027	6.95	23.0	4.9	26.1	3.1	1.4	<1.0	4.2	38.2	83.3
11	Merced Peak Fork		6.53	30.4	4.9	13.0	2.6	4.8	2.9	10.2	25.6	25.0
14	Hutching Creek	0.099	6.82	36.4	8.2	17.4	3.6	1.7	7.1	10.0	33.2	31.7
16	Cathedral Fork	0.014	6.68	54.9	8.2	52.2	4.3	4.5	<1.0	14.4	77.4	90.0
15	Echo Creek above Cathedral Fork	0.023	7.12	33.9	11.5	39.1	4.1	25.4	<1.0	5.8		85.0
19	Illilouette Creek above Mono Meadow	0.218	7.61	139.7	29.6	91.3	12.3	25.4	<1.0	10.0	226.0	183.3
18	Clark Fork	0.003	6.11	43.4	10.7	21.7	4.1	5.9	3.6	4.0	51.0	61.7
20	Mono Meadow Tributary	0.022	7.11	164.7	38.7	113.0	14.8	3.7	<1.0	3.1	302.0	250.0
1988 Synoptic												
1	Merced R. at Happy Isles gage	14.20	6.90	43.4	4.9	35.7	5.6	15.8	2.8	8.3	56.7	68.3
3	Merced R. above Vernal Fall		6.88	36.4	3.3	29.1	4.6	13.0	3.8	7.9	43.0	56.7
4	Sunrise Creek		7.15	99.8	19.8	87.0	9.2	5.9	<1.0	2.5	185.6	216.7
5	Merced R. at Bunnel Cascade		6.97	35.9	6.6	26.5	3.6	11.6	3.3	8.1	38.2	55.0
6	Echo Creek above Merced R.		6.88	24.5	0.8	27.4	3.3	5.6	<1.0	4.6	39.4	71.7
7	Merced R. below Merced HSC		6.80	36.9	<0.8	26.5	3.6	14.4	4.7	9.4	35.6	48.3
9	Fletcher/Lewis Creek		6.78	33.9	<0.8	22.6	3.6	4.2	4.2	9.4	38.1	55.0
10	Merced R. below Washburn Lake		6.72	29.9	0.8	22.2	3.3	5.9	5.2	9.6	30.1	43.3
11	Merced Peak Fork		6.64	22.0	4.1	19.1	3.1	4.8	4.3	8.8	22.1	40.0
12	Lyell Fork		6.65	38.4	5.8	19.6	3.6	4.5	9.7	12.3	33.5	40.0
1981 Synoptic												
2	Illiluoette Creek above Merced R.	0.048	6.0	164.7	32.1	121.7	17.9	96.0	2.0	<20	200	199.7
17	Fletcher Creek above Bofelsang HSC	0.037	5.9	39.9	11.5	17.4	5.1	5.6	1.0	<20	40	36.6
7	Merced R. below Merced HSC	0.125	6.3	104.8	10.7	65.2	5.1	90.4	1.0	<20	60	31.6
21	Fletcher Creek above Fletcher Lake	1.9	5.9	54.9	18.9	17.4	5.1	2.8	2.0	<20		38.3
22	Nelson Lake		5.9	29.9	23.0	30.4	5.1	8.5	1.0	<20		33.3

concentrations at this site to be lower than might be expected from the effects of snowmelt dilution during the spring and subsequent increased evapotranspiration during the late summer. This observed small amount of variation in SO_4^{2-} concentration suggested that there is a process or perhaps a combination of processes which regulate SO_4^{2-} in the surface waters in this watershed, at least to some extent. Retention of S within the watershed can occur by a variety of processes. These include adsorption to soil surfaces, incorporation into soil organic matter, plant uptake, and SO_4^{2-} reduction and oxidation reactions in seasonally anoxic areas (Clow et al. 1996).

The presence of glacial till can exert an important control on the base cation concentrations and therefore the ANC of surface waters. The quantity of soil and other types of surficial materials in the watershed are important determinants of drainage water chemistry. This is largely because such materials can slow the movement of water through a watershed, increasing its residence time, and provide additional opportunities for weathering products to be contributed to drainage water. Also, because till can contain considerable amounts of fine material, it provides abundant mineral surfaces with which the drainage water can react (Peters and Murdoch 1985, Newton et al. 1987, Clow et al. 1996).

The Western Lakes Survey (WLS, Landers et al. 1987) sampled 9 lakes within YOSE and an additional 14 lakes within 25 km of YOSE (Table X-11). Two-thirds of the WLS lakes in YOSE, and nearly half of those near the park, had ANC < 50 µeq/L. Three of the sampled lakes in and near the park had ANC < 20 µeq/L. The locations of acid-sensitive lakes are shown in Figure X-3. Many of the sensitive low-ANC lakes were small (< 10 ha), as is typically the case for acid-sensitive western lakes, although several were considerably larger (16 to 30 ha); watershed areas were similarly variable. pH values in the sensitive lakes were generally between 6.2 and 7.0, although one lake outside the park had pH < 6. Lakewater SO_4^{2-} concentrations were generally between about 3 and 12 µeq/L in the sensitive lakes, which is approximately what would be expected on the basis of average measured SO_4^{2-} concentrations in precipitation of about 3 to 5 µeq/L, negligible dry deposition of S, and less than 50% evapotranspiration. There were also three lakes within YOSE with slightly higher SO_4^{2-} concentration (14 and 18 µeq/L), which could be partly attributable to watershed sources of S, and one lake outside the park with very high SO_4^{2-} (92 µeq/L) which is almost certainly attributable to geologic S. The three lakes having higher-than-expected SO_4^{2-} concentration were not among the most acid-sensitive of the sampled lakes (ANC = 71, 57, 38 µeq/L, respectively). Although most of the sampled lakes had very low NO_3^- concentration (< 2 µeq/L), there was one lake in YOSE with high NO_3^- (8 µeq/L), and two lakes outside the park with high NO_3^- (6,10 µeq/L). All of the lakes having high NO_3^- were low in ANC (27 to 38 µeq/L) and were over 3,100 m elevation. These relatively high-elevation, acid-sensitive lakes likely experience much higher NO_3^- concentrations during snowmelt, as compared with the values reported by the WLS for the fall sampling (Table X-11). It is therefore likely that N-driven episodic acidification is significant in these lakes.

The most acid-sensitive lakes in and near YOSE, i.e., those having ANC less than about 30 µeq/L, had very low concentrations of base cations (C_B about 20 to 35 µeq/L), about half of which was Ca^{2+}, and low DOC (< 2 mg/L). All are presumably highly-sensitive to chronic, and especially episodic, acidification if S deposition increased substantially. In view of the relatively high NO_3^- measured by the WLS in some of the lakes in the fall of 1986, it is likely that some are sensitive to chronic acidification under increased N-deposition. Many may be experiencing episodic acidification to ANC values near zero under current deposition levels.

The limited data available regarding temporal trends in surface water quality within YOSE suggest that recent acidification has not occurred. Water samples were collected by helicopter

Table X-11. Results of lakewater chemistry analyses by the Western Lake Survey for selected variables in YOSE and adjacent areas.

Lake Name	Lake ID	Lake Area (ha)	Watershed Area (ha)	Elevation (m)	pH	ANC (µeq/L)	SO_4^{2-} (µeq/L)	NO_3^- (µeq/L)	Ca^{2+} (µeq/L)	C_B (µeq/L)	DOC (mg/L)
Lakes within YOSE											
(No Name)	4A1-006	3.6	34	2720	6.3	15	4.7	0.8	9	21	0.6
Mary Lake	4A1-007	26	298	2940	7.0	71	17.5	0.1	44	87	0.6
(No Name)	4A1-008	29.8	619	2232	6.5	29	3.1	0.0	14	37	1.5
Lake Vernon	4A1-009	26.1	2960	1988	6.6	57	14.0	0.4	43	83	1.7
Roosevelt Lake	4A1-013	25	440	3106	6.8	38	4.2	0.1	26	49	0.6
Bingaman Lake	4A1-015	5.2	98	3403	7.0	67	3.9	0.0	38	77	0.6
(No Name)	4A1-016	6.3	1020	3111	6.6	27	12.1	8.1	36	55	0.5
(No Name)	4A1-055	1.7	60	2745	6.6	34	2.7	0.1	21	46	1.8
Vogelsang Lake	4A1-059	9.1	163	3154	6.8	46	2.6	0.1	32	57	0.8
Lakes Within 25 km of YOSE											
Lost Lake	4A1-003	4.1	93	2964	7.4	168	7.5	0.0	128	168	0.6
Leopold Lake	4A1-004	4.1	36	2696	6.3	16	7.2	1.5	9	26	1.5
(No Name)	4A1-005	2	31	2550	5.8	15	9.3	1.2	13	32	2.4
Hoover Lakes (NE)	4A1-012	2.5	629	2964	7.5	242	385.9	2.6	493	593	0.3
Kidney Lake	4A1-014	5.9	218	3184	7.0	38	92.9	6.3	86	135	0.3
Nydiver Lakes (Middle)	4A1-017	3.4	96	3086	6.6	26	5.2	1.2	18	34	0.6
Iceberg Lake	4A1-018	15.8	186	2989	6.7	26	6.5	2.2	23	34	0.4
Walton Lake	4A1-019	1.5	52	3159	6.6	34	8.5	9.5	38	60	1.1
Cow Meadow Lake	4A1-058	7.4	3693	2379	6.5	55	4.4	5.3	33	64	2.9
Star Lakes (North)	4A2-037	1.7	44	2428	6.9	80	4.5	0.2	37	88	2.3
Bare Island Lake	4A2-038	3.9	26	2550	6.9	80	10.7	0.0	48	85	1.7
Chiquito Lake	4A2-054	5.3	407	2428	6.6	92	1.0	0.1	52	123	10.0
Silver Lake	4A3-043	46.8	12054	2203	7.4	234	41.3	0.1	199	296	0.9
Twin Lakes (NE)	4A3-066	163.2	10127	2159	7.7	442	97.5	0.2	378	539	1.0

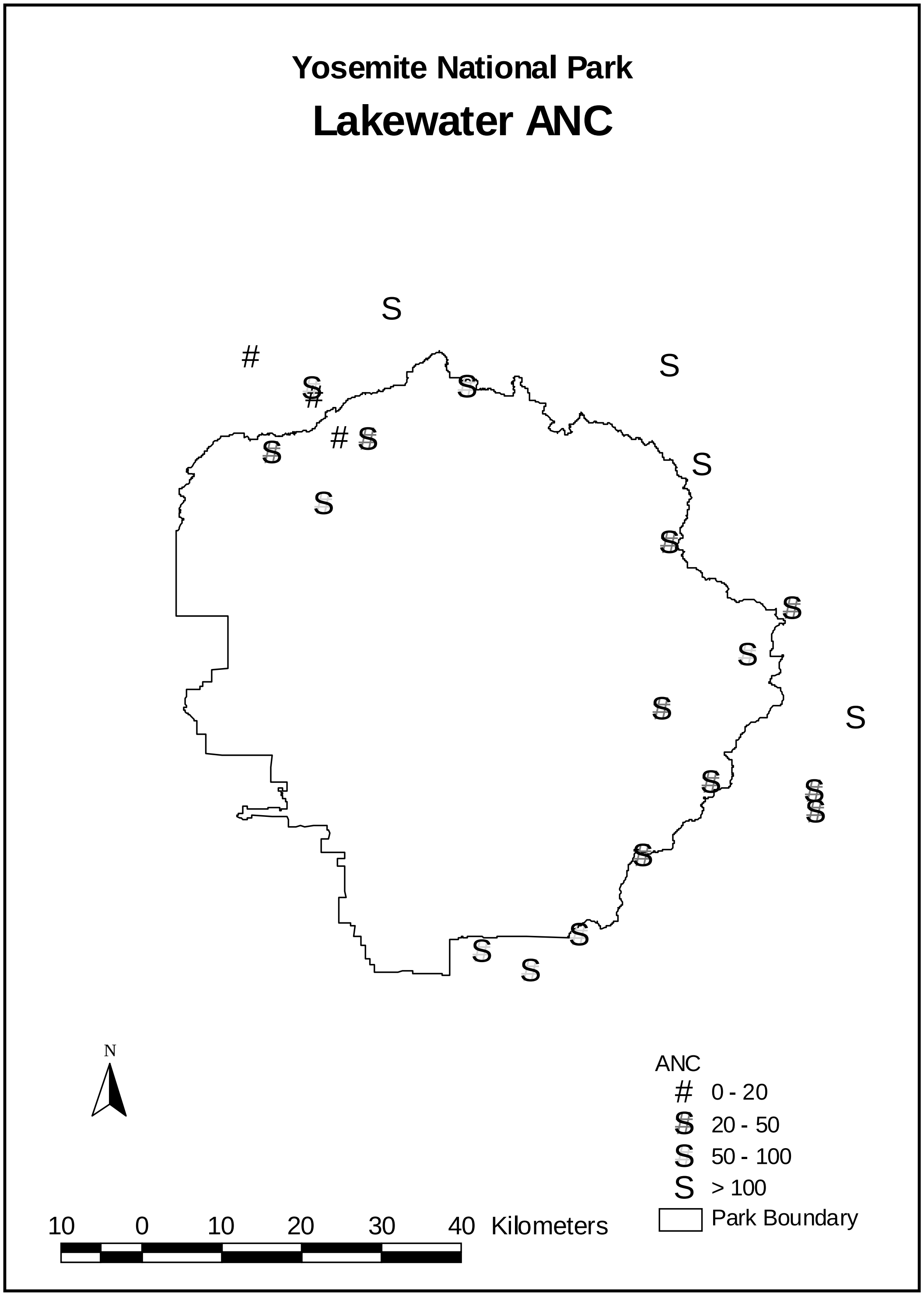


Figure X-3. Location of WLS lakes sampled in and near YOSE, coded by ANC.

from 170 lakes in SEKI and YOSE in September 1965. pH values ranged from 4.7 to 7.3, with an arithmetic mean and median both equal to pH 6.0. Specific conductance measurements ranged from 2.4 to 110 µS/cm, with an arithmetic mean of 8.0 and a median of 7.0 (Bradford et al. 1968). A follow-up study was conducted by randomly sampling 124 lakes in the three parks in July 1980, 1981, 1982, and 1983 during the spring thaw, and also in October 1980 before the winter freeze (Bradford et al. 1983). The results suggested that lake water acidity, total electrolyte concentration, and the concentrations of a number of trace elements did not increase between 1965 and 1983. Median values for lakewater SO_4^{2-} in July 1980, October 1980, and July 1981 were 8.3, 8.8, and 11.0 µeq/L, respectively. Similarly, the median concentrations of lakewater NO_3^- were 7.1, 2.9, and 5.7 µeq/L, respectively. Bradford et al. (1983) also sampled feeder streams and snow. These samples were analyzed for pH, conductivity, and bicarbonate concentration. Median pH value of the feeder streams was equal to that of the lakewater (6.4). In contrast, snowmelt water had a median pH of 5.6.

A chemical survey of 101 high-altitude lakes in 7 national parks in the western United States was conducted by the USGS during the fall of 1999; 72 of the lakes were previously sampled during the fall of 1985 as part of the WLS (Clow et al. 2000). A strong effort was made to use methods and protocols similar to those used during the WLS. The objective of the 1999 lake survey was to provide information on water quality in the parks and assess whether there were significant differences in lake chemistry in the 1985 and 1999 data sets. Lakes in the three California parks (SEKI, YOSE, LAVO) and in Rocky Mountain National Park (Colorado) were extremely dilute; median specific conductances were $\leq$ 12 µS/cm and median alkalinities were $\leq$ 75 µeq/L. Specific conductances and alkalinities generally were substantially higher in Grand Teton and Yellowstone National Parks (Wyoming), and Glacier National Park (Montana), probably due to the prevalence of more reactive bedrock types. Concentrations of base cations and alkalinity were lowest in lakes in the alpine zone, probably because of minimal vegetation and soil development, and because of fast hydrologic flow rates. These conditions make alpine lakes highly sensitive to atmospheric deposition of pollutants. This is evidenced by relatively high NO_3^- concentrations in high-elevation lakes in Rocky Mountain National Park (0 to 29 µeq/L), which are subject to moderate levels of N deposition (3 to 5 kg N/ha/yr; Figure IV-5; Clow et al. 2000).

Of the 10 lakes sampled in YOSE, 9 of which had been included in the WLS, all had ANC $\leq$ 75 µeq/L and 4 had ANC $\leq$ 30 µeq/L. Only one lake exhibited NO_3^- concentration higher than 0.1 µeq/L. Sulfate concentration varied from 2 to 13 µeq/L (D.W. Clow, U.S.G.S., Denver, pers. comm.).

One challenge that will need to be addressed is separating effects of trends in water quality from variations due to differences in hydroclimatic conditions. A qualitative evaluation of the effects of climatic variations will be done by looking at variations in water chemistry and climate at several intensively monitored research watersheds in the Sierra Nevada and Rocky Mountains. This research is ongoing (D.W. Clow, U.S.G.S., Denver, pers. comm.).

Other than the various synoptic surveys and monitoring studies that have been conducted in YOSE, and that are summarized above, there has not been much research conducted on the potential impacts of air pollutants on aquatic ecosystems within the park. However, a great deal of research has been conducted on the acid base chemistry of high-elevation lakes in SEKI and their sensitivity to acidification from atmospheric inputs of N and S. Much of this work has been accomplished in conjunction with the Integrated Watershed Study (IWS; Tonnessen 1991) of the Emerald Lake watershed and various efforts to regionalize IWS findings. Specific studies have focused on such aspects as snowmelt hydrology, long-term monitoring, episodic acidification and neutralization processes, biological effects, and model predictions of future

change. The results of these research efforts are described at length in Chapter IX of this report, because much of the work has been conducted within SEKI. However, the results of these studies are certainly also applicable to high-elevation watersheds in YOSE. The reader interested in YOSE is therefore advised to refer extensively to the information presented in Chapter IX.

b. Aquatic Biota

Extensive study of the vertebrate animals (including amphibians) that occurred along a transect across the Sierra Nevada through YOSE was conducted during the period 1914 to 1920 (Grinnell and Storer 1924). Sampling occurred and observations were made of vertebrate species at 40 stations along the 144-km stretch extending from LaGrande at the edge of the Central Valley to Mono Lake on the east side of the mountains. Sites visited ranged from 76 m elevation to 3,962 m on Mt. Lyell and covered all of the life zones in the area. To evaluate changes that have occurred in the amphibian fauna in the Yosemite area, Drost and Fellers (1994) resurveyed the Yosemite transect during the spring and summer of 1992. They revisited the original Grinnell and Storer sites and made intensive searches for the frog species known from the area. They also visited a number of other areas with previous records of frog presence and abundance and surveyed additional lakes, meadows and other sites along the transect that appeared, at least on field inspection and examination of maps and aerial photos, to offer suitable habitat for frogs. They were able to directly compare their data for 32 of the 40 originally designated collecting stations. Most frog species were present at far fewer sites in 1992 than during the period 1916 to 1920. Foothill yellow-legged frog was not found at any of the sites in 1992. The western toad (*Bufo boreas*) was reduced to one site in the 1992 survey and mountain yellow-legged frog to two. For the latter two species, this was a decline of over 80% in number of sites occupied. Yosemite toad was present at about half of the sites where it was found in the 1920 survey (Drost and Fellers 1994). Of all the frog species in the Yosemite section, only the Pacific tree frog (*Pseudacris regilla*) was still present in 1992 at most of the sites where it was found in the earlier survey. The California red-legged frog (*Rana aurora*), a low-elevation species, had been found at three sites by Grinnell and Storer (1924). Drost and Fellers (1994) did not find the species at any of the original sites, but did find California red-legged frog tadpoles at one of the supplemental sites that they visited. Possible changes in abundance were evaluated on the basis of designated abundance categories, such as rare, common, or abundant. This comparison showed a strong downward trend for some species at the sites where they occurred during both surveys. No species except Pacific tree frog showed apparent increases at any sites, and even for this species there were declines at nearly 80% of the sites where there was an evident change in numbers. Drost and Fellers (1994) evaluated possible causes of the decline and their findings are summarized in Chapter II. A variety of hypotheses have been proposed for world-wide amphibian declines in recent decades, including acidic deposition (c.f. Blaustein and Wake 1990, Wyman 1990). In the Yosemite area, as elsewhere, the declines are very much an enigma and their cause is in need of further study (Drost and Fellers 1994).

Widespread introduction of non-native fish has apparently limited distribution and abundance of some frog species, particularly mountain yellow-legged frog, but it did not seem to adequately explain the overall decline of the frog fauna for the Yosemite area and there were several reasons given by Drost and Fellers (1994) for this. Most all of the frog species in the area are capable of surviving and reproducing in waters that contain fish as long as there is emergent vegetation or other escape cover present. Furthermore, mountain yellow-legged frog seems to be the most susceptible to fish predation, but it remained the most numerous frog species in Westfall Meadows as late as 1977, long after fish were introduced (Yoon 1977) and it

has since disappeared from that area. Toad species are much less susceptible to fish predation. Toad species frequently breed in ephemeral bodies of water that do not harbor fish and they also produce toxic skin secretions and are therefore generally avoided by fish and other predators (c.f. Peterson and Blaustine 1991). In addition, frog populations have disappeared from sites that were never planted with fish or that were too small or ephemeral to support fish. Also, what is known about the timing of the population declines does not agree with introduced fish being the sole or primary cause. Large numbers of trout were planted in YOSE between 1932 and 1951. Subsequently, the numbers of fish planted has steadily declined, but the range of the planted fish expanded. However, significant declines for Yosemite toad, mountain yellow-legged frog, and probably other species seem to have occurred much later than this, in the 1970s and after. Isolation of remaining populations by fish may be a cause for some of these later declines (Bradford et al. 1993).

Based on the historical runoff record, it was estimated that the 1987 through 1992 period represented an approximately 1 in 300-yr drought event (Roos 1992). However, the Grinnell and Storer survey also occurred during a drought period, so comparisons between the two surveys should not be greatly biased in this regard. Also, Yoon (1977) reported that mountain yellow-legged frog was still common in Westfall Meadows in 1977 at the end of the drought that was shorter than the 1987-1992 drought, but had a much lower annual rainfall. Furthermore, the declines of Yosemite toad, red-legged frog, foothill yellow-legged frog, and possibly mountain yellow-legged frog seemed to have been occurring for a much longer period of time than was reflected in the 1987 to 1992 drought. In fact, most of the period had been wetter than normal (Drost and Fellers 1994).

The effect of increased acidity may warrant further study. Indirect or confounded effects might be possible. Drost and Fellers (1994) concluded that surface water acidification should not be discounted as a possible cause of amphibian declines. However, pesticides and other toxic chemicals can also kill adult and larval amphibians. A study by Zabik and Seiber (1993) illustrated that organophosphate insecticides used in the Central Valley can be transported in measurable quantities to over 1,900 m elevation in the Sierra Nevada. Other direct pathological studies of these agricultural chemicals on amphibians are not available.

3. Vegetation

Significant changes have occurred in the vegetative communities of YOSE since the arrival of European man. Some of these changes have been documented through a number of inventory, monitoring, and assessment projects. Such changes have likely been driven by a variety of causes, including introduction of exotic species, fire suppression, livestock grazing, and climatic fluctuations. The effects of air pollution are therefore superimposed on top of a suite of other changes.

An inventory of vegetative communities that occur in Yosemite Valley was reported by Acree (1994). The objectives were to identify areas of native and exotic vegetation within the valley to be used in planning decisions and monitoring schemes and also to serve as a basis for planting design and future restoration projects. A number of exotic plant species occur in Yosemite Valley, primarily in the disturbed areas such as roadsides and bike path margins. Exotic species have a particularly large impact in the meadow communities, where they may comprise up to 80 percent of the total vegetation, based on cover estimates.

Heady and Zinke (1978) evaluated vegetational changes in Yosemite Valley for the past approximately 100 years. The assessment was based on analysis of photographs, written materials in park files, structure of forest stands, botanical composition in relation to soil development, and vegetational measurements over a 10-year period. The study concluded that

forest stands in Yosemite Valley have thickened, with continuous establishment of trees over the past 100 years, and half of the area that was meadow in the late 1800s is now forest. Some trees must be removed continually from the meadows to maintain the vistas of waterfalls, granite cliffs, and mountain spires. Suggestions for vegetational management within the valley were also given.

Vegetation changes over the past 80 years in the high elevation areas of YOSE were evaluated by Vale (1987). These changes were identified from 59 scenes originally photographed at the turn of the century and then rephotographed in 1984-1985. Analysis of the resulting photograph pairs suggested that 1) krummholz has increased in height and density, 2) forests at the upper forest line have increased in density, 3) meadows have been invaded by trees, 4) local patches of thin forest have increased in density, and 5) trees on many domes and rock slopes have increased in number. Fire suppression, fluctuations in climate, and livestock grazing were explored as possible causes of the observed changes. Fires in the high elevation areas of YOSE were probably sufficiently uncommon during previous centuries that even with decades of fire suppression, the high elevation forest today has probably deviated little from what is thought of as natural conditions (van Wagtendonk 1977). Climatic conditions, however, can increase the likelihood of fire even in the high elevation areas of the park. For example, in Kings Canyon National Park, fires burned several high elevation meadows in 1977 during the most severe two-year drought in recorded history (Parsons 1981). In one of those burned meadows, some of the many invasive lodgepole pines were killed (DeBenedetti and Parsons 1979). Vale (1987) speculated that removal of Native Americans and their fires from YOSE in the late 19th century and the onset of fire suppression early in the 20th century seemed unable to account for most of the vegetation changes documented in the pairs of photographs. The one possible exception was the thickening of small patches of forest. The author concluded that it was conceivable that these areas represent areas previously thinned by local fires. Vale (1987) evaluated climatic fluctuation as a possible cause of the observed vegetation changes that were documented in the photographs by examining the climatological data for stations in and near the Yosemite high country, as well as published summaries of both recorded data and tree ring studies. The climatological data suggested that the Yosemite region was relatively dry from about 1910 through the mid 1930s, and that period was both preceded and followed by wetter conditions. There was a warm and dry period early in the 20th century but it was equaled or perhaps exceeded by an episode of similar intensity but longer duration in the first half of the 19th century. The author speculated that the increased vigor of krummholz stands and increased tree density at tree line likely represented response to the warmer temperatures of the period 1910 through 1930 and the lack of a recurrence of the colder temperatures of the late 1800s. Climatic fluctuation seemed an unlikely cause of other vegetation changes that were documented in the photographs, however.

Grazing by domestic livestock has been identified as a cause of invasions of meadows and brushlands by trees (Vale 1981). Grazing can produce such changes by reducing the cover of herbaceous plants and exposing mineral soil surfaces, both of which are conducive to germination of tree seeds and survival of tree seedlings. In YOSE, sheep were eliminated from the national park lands by 1905 (Holmes 1979). The germination date for invasive trees in Dana Meadows near Tioga Pass accelerated shortly thereafter, after about 1915 to 1925 (Vale 1981).

Vale (1987) postulated overall that fire suppression or insect-induced mortality could be responsible for local forest thickening; climate fluctuations seemed to be the most probable cause of change in krummholz and forest line vegetation; and changes in livestock grazing might explain the invasion of meadows by lodgepole pine. The author was not able to offer a reasonable explanation, however, for the observed increase in tree density on rock slopes.

Forest composition patterns in YOSE were related to environmental factors by Parker (1989) through numerical classification of forest types along elevational and topographic gradients and development of regression models relating basal area of common tree species to environmental variables. The eight forest types were differentiated primarily by elevation zone and secondarily by topographic setting. However, regression models consistently included elevation and extracted soil base cation levels (as reflected by extracted soil Mg) as explanatory variables of species basal area. The two *Abies* species (white fir and red fir) were negatively correlated with soil Mg, whereas other montane tree species were positively correlated with soil Mg. Topography and soil physical properties were only infrequently incorporated into species regression models (Parker 1989). These results suggest that Sierran forest composition patterns might respond to soil acidification from acidic deposition, although deposition of S or N would likely have to increase dramatically before any significant soil acidification would occur at YOSE.

Widespread ozone injury in ponderosa pine has been documented at YOSE in several studies. Ozone injury was first documented by Duriscoe and Stolte (1989), and was followed up by another survey that determined that about 30% of the pines at four symptomatic sites had ozone injury in needles that were two years old or less (Peterson et al. 1991, Peterson and Arbaugh 1992). A more recent study (Arbaugh et al. 1998) found that about 40% of pines at two sites had ozone injury.

An analysis of tree growth at YOSE (Peterson et al. 1991, Peterson and Arbaugh 1992) found that the growth of symptomatic ponderosa pine was significantly lower since 1950, including trees with some of the largest growth reductions measured in the Sierra Nevada. However, two sites included in the YOSE study had growth decreases throughout the 20th century, with the most probable cause being annosus root rot, which is common throughout Yosemite Valley (Parmeter et al. 1978). The differential effects of ozone and fungal pathogens on tree growth at YOSE are difficult to discern.

The sensitivities of plant species at YOSE to air pollutants are summarized in Table X-12.

Table X-12. Plant and lichen species of YOSE with known sensitivities to sulfur dioxide, ozone, and nitrogen oxides (H=high, M=medium, L=low, blank=unknown).

Scientific Name	Common Name	Sensitivity		
		SO_2	O_3	NO_x
Gymnosperms				
Abies concolor	White fir	H	M	H
Calocedrus decurrens	Incense cedar		M	
Pinus monticola	Western white pine	M	M	
Pinus ponderosa	Ponderosa pine	H	H	H
Pseudotsuga menziesii	Douglas fir	H	M	H
Sequoiadendron giganteum	Giant sequoia		M	
Taxus brevifolia	Pacific yew	L		
Tsuga mertensiana	Mountain hemlock	H	L	
Angiosperms				
Acer glabrum	Rocky Mountain maple	M		
Acer macrophyllum	Bigleaf maple		L	
Achillea millefolium	Common yarrow		L	
Agastache urticifolia	Nettleleaf giant hyssop		M	

Table X-12. Continued.

Scientific Name	Common Name	Sensitivity		
		SO_2	O_3	NO_x
Alnus rhombifolia	White alder	H	M	
Alnus tenuifolia	Thinleaf alder	M		
Artemisia dracunculus	Wormwood		M	
Artemisia ludoviciana	Louisiana sagewort	M		
Artemisia tridentata	Big sagebrush	M	L	
Betula occidentalis	Water birch	M		
Calochortus nuttallii	Sego lily		L	
Cassiope mertensiana	Western moss heather		L	
Ceanothus velutinus	Snowbrush ceanothus	L		
Cercocarpus ledifolius	Curlleaf mountain mahogany	M		
Clematis ligusticifolia	Western white clematis	M		
Fragaria virginiana	Virginia strawberry		H	
Galium bifolium	Twinleaf bedstraw		L	
Gayophytum diffusum	Spreading groundsmoke		H	
Geranium richardsonii	Richardson's geranium	M	M	
Hackelia floribunda	Manyflower stickseed	L		
Helianthus annuus	Common sunflower	H	L	
Lonicera involucrata	Twinberry honeysuckle	L		M
Oenothera elata	Hooker’s evening primrose		H	
Osmorhiza chilensis	Sweetcicely		M	
Osmorhiza occidentalis	Western sweetroot		L	
Platystemon californicus	California creamcups	M	M	
Polygonum douglasii	Douglas' knotweed		L	
Populus tremuloides	Quaking aspen	H	H	
Populus trichocarpa	Black cottonwood	M	H	
Potentilla flabellifolia	High mountain cinquefoil		H	
Potentilla fruticosa	Shrubby cinquefoil		L	
Prunus emarginata	Bitter cherry	M		
Prunus virginiana	Common chokecherry	M	M	
Quercus kelloggii	California black oak		M	
Rhus trilobata	Skunkbush sumac	L	H	
Ribes viscosissimum	Sticky currant	M		
Rubus parviflorus	Thimbleberry		M	
Rudbeckia hirta	Blackeyed susan		H	
Rumex crispus	Curly dock		L	
Salix scouleriana	Scouler's willow		M	
Salvia columbariae	Chia	M	M	
Sambucus mexicana	Blue elder		H	
Symphoricarpos oreophilus	Whortleleaf snowberry	M		
Taraxacum officinale	Common dandelion		L	
Thalictrum fendleri	Fendler's meadowrue		L	
Thysanocarpus curvipes	Sand fringepod	M	M	
Trifolium pratense	Red clover	L		
Trifolium repens	White clover		H	
Vaccinium ovalifolium	Ovalleaf blueberry		M	
Vicia americana	American vetch		L	
Vitis californica	California wild grape		L	

Table X-12. Continued.

Scientific Name	Common Name	Sensitivity		
		SO_2	O_3	NO_x
Lichens				
Acarospora chlorophana		H		
Aspicilia caesiocinerea		L		
Calicium viride		M	H	
Cladonia chlorophaea		M		
Cladonia fimbriata		H		
Lecidea atrobrunnea		L		
Leptogium californicum			M	
Letharia vulpina		L	L	
Melanelia glabra			L	
Melanelia multispora			L	
Melanelia subolivacea			L	
Parmelia saxatilis		M	L	
Peltigera canina		L	H	
Peltigera collina			H	
Peltigera didactyla			H	
Phaeophyscia ciliata			M	
Physcia aipolia		M		
Physcia caesia		M		
Physcia dubia		M		
Physcia stellaris		M		
Physconia detersa		H	L	
Pseudephebe pubescens			H	

4. Visibility

As part of the Interagency Monitoring of Protected Visual Environments (IMPROVE) network, visual air quality in YOSE has been monitored using an aerosol sampler, transmissometer, and camera. The aerosol sampler and transmissometer began operation in March and August of 1988, respectively. Both instruments were originally located on Turtleback Dome near the mouth of the Yosemite Valley. The transmissometer transmitter was moved in November of 1994 in efforts to increase the path length to its current distance of 4.47 km. The sampler and transmissometer receiver still reside in their original locations. The automatic 35mm camera operated from September 1983 through July 1985, taking photographs of the Yosemite Valley, including Half Dome. Due to repetitive vandalism, the system was relocated to the transmissometer receiver site where it photographed Telegraph Hill and the San Joaquin valley from September 1984 through April 1995. Data from this IMPROVE site have been summarized to characterize the full range of visibility conditions for the March 1988 through February 1999 period, based on seasonal periods (Spring: March, April, and May; Summer: June, July, and August; Autumn: September, October, and November; and Winter: December, January, and February) and annual periods (March through February of the following year, e.g., the annual period of 1998 includes March 1998 through February 1999). Complete descriptions of visibility characterization, mechanisms of sources and visibility impacts, and IMPROVE monitoring techniques and rationale are provided in the introduction of this document.

a Aerosol Sampler Data - Particle Monitoring

A tabular and graphic summary of average reconstructed extinction values by season and year for the March 1988 through February 1999 period are provided in Table X-13 and Figure X-4, respectively.

Reconstructed extinction budgets generated from aerosol sampler data apportion the extinction at YOSE to specific aerosol species (Figure X-5). The species shown are Rayleigh, sulfate, nitrate, organics, elemental (light absorbing) carbon, and coarse mass. The sum of these species account for the majority of non-weather related extinctions. Extinction budgets are listed by season and by mean of cleanest 20%, mean of the median 20%, and mean of the dirtiest 20%. The "cleanest" and "dirtiest" signify lowest fine mass concentrations and highest fine mass concentrations respectively, with "median" representing the 20% of days with fine mass concentrations in the middle of the distribution. Each budget includes the corresponding extinction coefficient, standard visual range (km), and deciview (dv).

The segment at the bottom of each stacked bar in Figure X-5 represents Rayleigh scattering, which is assumed to be a constant 10 Mm^{-1} at all sites during all seasons. Rayleigh scattering is the natural scattering of light by atmospheric gases. Higher fractions of extinction due to Rayleigh scattering indicate cleaner conditions.

Table X-13. Seasonal and annual average reconstructed extinction (b_{ext} in Mm^{-1}) in YOSE for the period March 1988 through February 1999.

Year	Spring (Mar, Apr, May)	Summer (Jun, Jul, Aug)	Autumn (Sep, Oct, Nov)	Winter (Dec, Jan, Feb)	Annual (Mar - Feb)[a]
1988	32.2	50.6	25.9	22.9	33.5
1989	31.7	40.0	30.7	19.0	31.4
1990	37.2	41.3	35.8	37.2	37.6
1991	33.4	33.3	36.0	21.7	31.7
1992	38.7	42.9	34.7	22.1	35.8
1993	32.1	32.3	36.5	21.2	31.4
1994	34.0	43.4	26.9	18.8	31.2
1995	26.4	32.8	47.7	22.7	34.9
1996	28.9	48.0	39.2	21.9	36.6
1997	35.0	32.7	32.3	19.5	30.9
1998	26.9	37.1	33.6	20.7	31.4
Mean[b]	32.4	39.5	34.5	22.5	33.3[c]

[a] Annual period data represent the mean of all data for each March through February annual period.
[b] Combined season data represent the mean of all seasonal means for each season of the March 1988 through February 1999 period.
[c] Combined annual period data represent the mean of all combined season means.

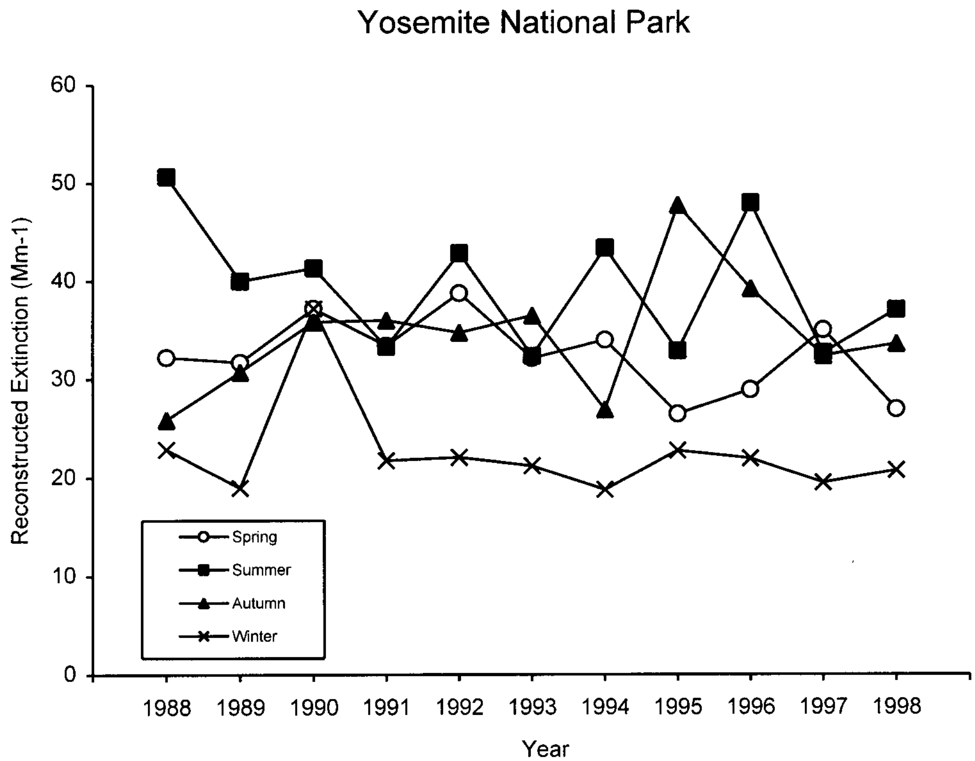


Figure X-4. Seasonal average reconstructed extinction (Mm^{-1}) at YOSE, March 1988 through February 1999.

b. Transmissometer Data - Optical Monitoring

The transmissometer system consists of two individually-housed primary components: a transmitter (light source) and a receiver (detector). The light extinction coefficient (b_{ext}) at any time can be calculated based on the intensity of light emitted from the source and the amount of light measured by the receiver (along with the path length between the two). Transmissometers provide continuous, hourly b_{ext} measurements. Meteorological or optical interference factors (such as clouds, rain, or a dirty optical surface) can affect transmissometer measurements. Collected data that may be affected by such interferences are flagged invalid, “filtered”. Seasonal and annual data summaries are typically presented both with and without weather-influenced data. Visibility metric presentations include mean values of filtered b_{ext} data. The best, worst, and average conditions using the arithmetic means of the 20th percentile least impaired visibility, the 20th percentile most impaired visibility, and for all data for the season are presented. Detailed descriptions of the transmissometer system and data reduction and validation procedures used can be found in Standard Operating Procedures and Technical Instructions for Optec LPV-2 Transmissometer Systems (ARS, 1996).

Table X-14 provides a tabular summary of the “filtered” seasonal and annual mean extinction values. Combined season data represent the mean of all valid seasonal b_{ext} means. Extinction values are also presented in units of standard visual range (in kilometers) and deciview (dv). Tables X-15 and X-16 summarize the 20% clean and 20% dirty visibility metric statistics respectively. Data are represented according to the following conditions:

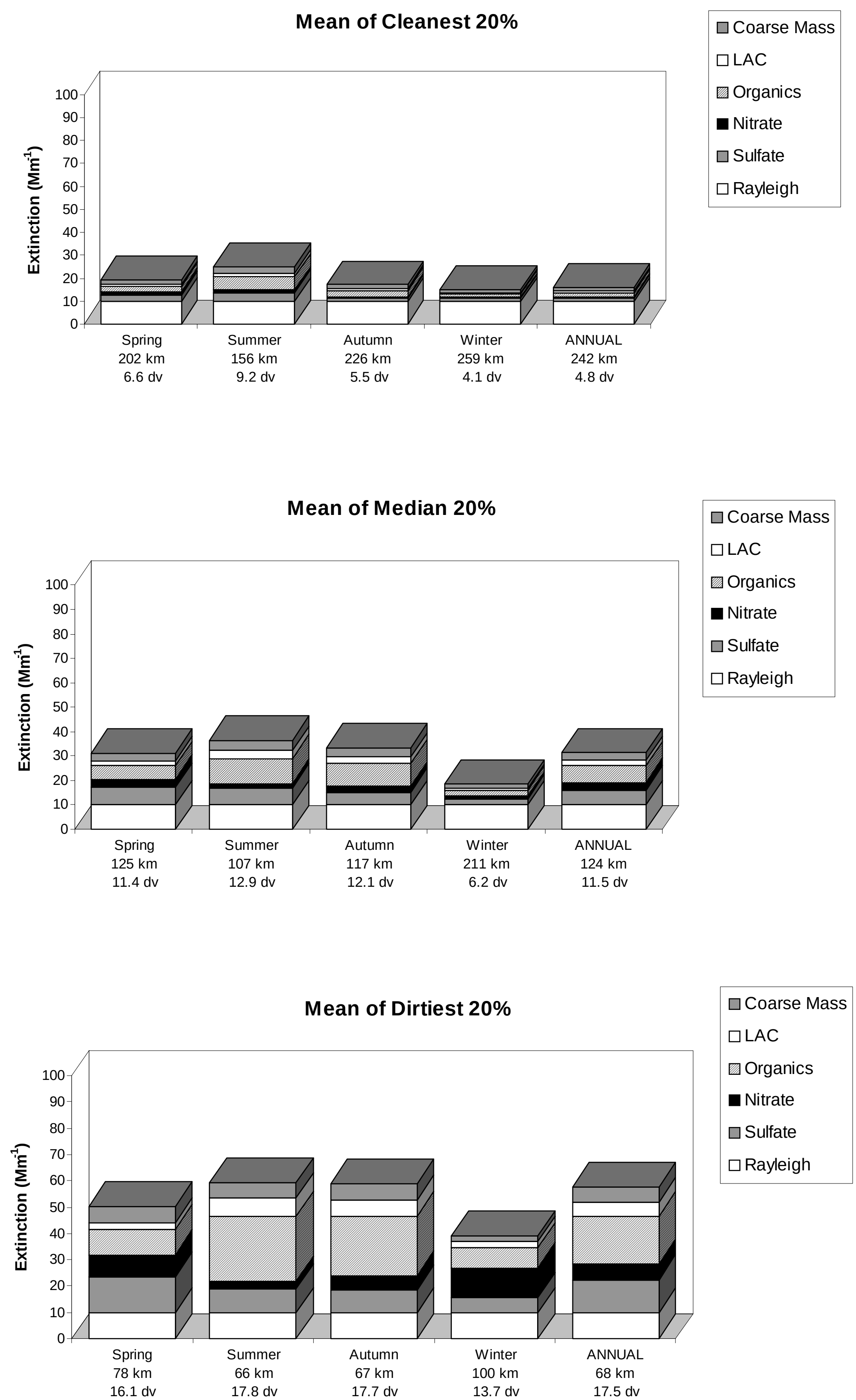


Figure X-5. Reconstructed extinction budgets for YOSE, March 1988 - February 1999.

Table X-14. Seasonal and Annual Arithmetic Means, YOSE, California Transmissometer Data (Filtered), September 1988 through February 1999.

YEAR	Spring (Mar, Apr, May)			Summer (Jun, Jul, Aug)			Autumn (Sep, Oct, Nov)			Winter (Dec, Jan, Feb)			Annual (March - February)[a]		
	SVR (km)	b_{ext} (Mm^{-1})	dv	SVR (km)	>b_{ext} (Mm^{-1})	dv	SVR (km)	b_{ext} (Mm^{-1})	dv	SVR (km)	b_{ext} (Mm^{-1})	dv	SVR (km)	b_{ext} (Mm^{-1})	dv
1988							—	—	—	109.7	118.9	19.5	***	***	***
1989	56.1	104.1	21.6	73.6	106.4	20.1	123.6	42.7	12.9	136.8	33.0	11.0	97.5	71.6	16.4
1990	100.5	44.4	14.2	79.7	98.8	17.9	107.4	49.3	14.1	148.0	33.3	10.7	108.9	56.5	14.2
1991	101.2	44.7	14.3	70.6	59.9	17.5	74.0	81.9	18.3	101.4	41.9	13.7	71.9	72.0	16.0
1992	73.5	58.8	17.2	81.5	57.2	16.4	87.7	53.1	15.7	83.3	54.1	15.9	81.5	55.8	16.3
1993	121.3	38.1	12.3	108.0	39.3	13.3	102.7	45.2	14.1	129.8	35.5	11.5	115.5	39.5	12.8
1994	91.5	47.7	15	69.1	74.1	18.3	107.9	43.6	13.7	181.9	31.6	8.6	112.6	49.3	13.9
1995	61.9	69.3	18.8	56.5	72.0	19.5	71.1	78.2	18.8	192.9	28.7	8.1	95.6	62.1	16.3
1996	118.0	41.1	12.9	66.4	73.6	18.6	68.4	75.5	18.7	---	---	---	***	***	***
1997	104.9	42.9	13.7	---	---	---	77.8	66.2	17.5	219.0	39.1	7.5	***	***	***
1998	123.4	54.8	13.5	85.8	48.3	14.8	96.9	53.5	15.1	143.4	42.9	11.4	112.4	49.9	13.7
Mean[b]	95.2	54.6	15.4	76.8	70.0	17.4	91.8	58.9	15.9	144.6	45.9	11.8	100.6	58.8[c]	15.1

-- No data are reported for seasons with <50% valid data.
*** No annual data are reported for periods with one or more invalid seasons.
a Annual period data represent the mean of all valid seasonal b_{ext} means for each March through February annual period.
b Combined season data represent the mean of all valid seasonal b_{ext} means for each season of the September 1988 through February 1999 period.
c Combined annual period data represent the mean of all combined seasonal b_{ext} means.

Table X-15. Seasonal and Annual 20% Clean Visibility Metric Statistics, YOSE, California Transmissometer Data (Filtered), September 1988 through February 1999.

YEAR	Spring (Mar, Apr, May)			Summer (Jun, Jul, Aug)			Autumn (Sep, Oct, Nov)			Winter (Dec, Jan, Feb)			Annual (March - February)[a]		
	SVR (km)	b_{ext} (Mm^{-1})	dv	SVR (km)	>b_{ext} (Mm^{-1})	dv	SVR (km)	b_{ext} (Mm^{-1})	dv	SVR (km)	b_{ext} (Mm^{-1})	dv	SVR (km)	b_{ext} (Mm^{-1})	dv
1988							---	---	---	311.1	12.8	2.4	***	***	***
1989	123.8	35.1	12.1	180.9	23.7	8.2	231.7	18.1	5.6	176.2	22.3	8.0	178.2	24.8	8.5
1990	142.3	27.6	10.1	126.4	31.4	11.4	180.6	22.0	7.8	227.4	17.5	5.5	169.2	24.6	8.7
1991	168.5	24.7	8.7	103.3	38.3	13.4	134.4	29.7	10.8	123.5	31.7	11.5	132.4	31.1	11.1
1992	107.1	37.4	13.1	123.5	32.2	11.6	132.6	30.0	10.9	106.0	37.2	13.1	117.3	34.2	12.2
1993	176.9	22.6	8.0	159.4	24.9	9.1	154.4	25.5	9.3	179.6	22.1	7.9	167.6	23.8	8.6
1994	129.9	30.3	11.1	103.8	38.1	13.3	176.0	22.5	8.0	261.8	15.2	4.1	167.9	26.5	9.1
1995	81.9	48.1	15.7	71.0	55.3	17.1	153.1	26.4	9.5	293.4	13.6	3.0	149.9	35.9	11.3
1996	189.7	21.0	7.3	98.9	40.0	13.8	123.5	32.5	11.7	---	---	---	***	***	***
1997	144.2	27.3	10.0	---	---	---	148.1	29.0	10.2	316.4	12.5	2.2	***	***	***
1998	221.4	18.0	5.8	150.6	26.6	9.7	157.9	25.0	9.1	199.5	19.7	6.8	182.4	22.3	7.9
Mean[b]	148.6	29.2	10.2	124.2	34.5	12.0	159.2	26.1	9.3	219.5	20.5	6.5	162.9	27.6[c]	9.5

-- No data are reported for seasons with <50% valid data.

*** No annual data are reported for periods with one or more invalid seasons.

a Annual period data represent the mean of all valid seasonal b_{ext} means for each March through February annual period.

b Combined season data represent the mean of all valid seasonal b_{ext} means for each season of the September 1988 through February 1999 period.

c Combined annual period data represent the mean of all combined seasonal b_{ext} means.

Table X-16. Seasonal and Annual 20% Dirty Visibility Metric Statistics, YOSE, California Transmissometer Data (Filtered), September 1988 through February 1999.

YEAR	Spring (Mar, Apr, May)			Summer (Jun, Jul, Aug)			Autumn (Sep, Oct, Nov)			Winter (Dec, Jan, Feb)			Annual (March - February)[a]		
	SVR (km)	b_{ext} (Mm^{-1})	dv	SVR (km)	>b_{ext} (Mm^{-1})	dv	SVR (km)	b_{ext} (Mm^{-1})	dv	SVR (km)	b_{ext} (Mm^{-1})	dv	SVR (km)	b_{ext} (Mm^{-1})	dv
1988							---	---	---	109.7	118.9	32.6	***	***	***
1989	20.1	204.6	29.9	16.4	264.6	32.2	50.6	86.1	20.9	83.3	59.1	16.3	42.6	153.6	24.8
1990	57.3	75.6	19.6	24.5	316.7	31.0	46.9	105.2	22.1	67.2	67.0	18.2	49.0	141.1	22.7
1991	60.0	68.9	19.0	47.7	83.8	21.1	26.5	201.5	28.1	72.9	63.9	17.2	51.8	104.5	21.4
1992	46.2	89.7	21.6	43.4	104.5	22.5	44.5	96.8	22.1	51.0	97.1	21.3	46.3	97.0	21.9
1993	66.7	69.5	18.3	71.8	55.3	17.0	55.6	80.5	20.0	81.8	63.7	16.5	69.0	67.3	18.0
1994	54.0	76.7	20.0	29.6	161.6	26.6	56.0	76.5	19.8	94.0	78.5	16.2	58.4	98.3	20.7
1995	39.9	109.5	23.3	41.8	95.3	22.4	32.1	142.6	25.5	89.5	68.1	16.2	50.8	103.9	21.9
1996	55.8	76.4	19.8	33.3	146.6	25.5	30.7	149.1	26.1	---	---	---	***	***	***
1997	63.5	74.8	18.9	---	---	---	35.1	123.5	24.5	81.6	132.0	19.7	***	***	***
1998	39.4	148.1	24.6	54.5	83.0	20.2	42.5	113.2	23.0	52.8	117.8	22.1	47.3	115.5	22.5
Mean[b]	50.3	99.4	21.5	40.3	145.7	24.3	42.1	117.5	23.2	68.9	101.3	19.6	50.4	116.0[c]	22.2

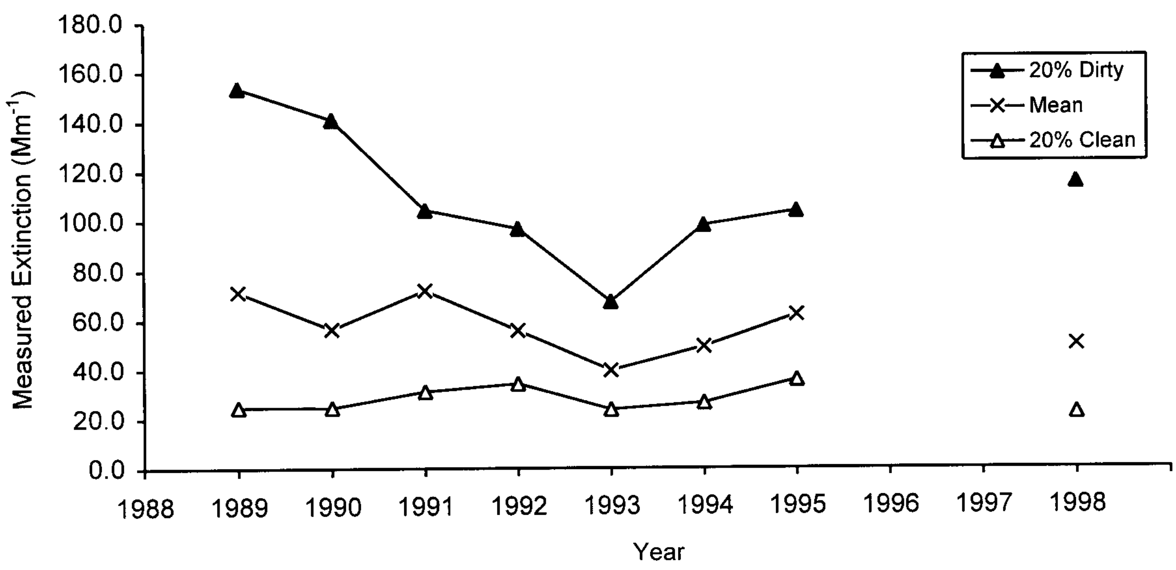
-- No data are reported for seasons with <50% valid data.
*** No annual data are reported for periods with one or more invalid seasons.
a Annual period data represent the mean of all valid seasonal b_{ext} means for each March through February annual period.
b Combined season data represent the mean of all valid seasonal b_{ext} means for each season of the September 1988 through February 1999 period.
c Combined annual period data represent the mean of all combined seasonal b_{ext} means.

- No data are reported for seasons when the percentage of valid hourly averages (including weather) compared to total possible hourly averages, was less than 50%.
- Annual data represent the mean of all valid seasonal b_{ext} values for each March through February annual period. No data are reported for years that had one or more invalid seasons.
- Combined season data represent the mean of all valid seasonal b_{ext} values for each season (spring, summer, autumn, winter) of the March 1988 through February 1999 period.
- Combined annual period data represent the unweighted mean of all combined seasonal b_{ext} values.

Figure X-6 provides a graphic representation of the "filtered" annual mean, 20% clean, and 20% dirty, visibility metric statistics. No data are reported for annual periods with one or more invalid seasons.

When comparing reconstructed (aerosol) extinction, Table X-13, with measured (transmissometer) extinction, Table X-14, the following differences/similarities should be considered:

- Data Collection - Reconstructed extinction measurements represent 24-hour samples collected twice per week. Transmissometer extinction estimates represent continuous measurements summarized as hourly means, 24 hours per day, seven days per week.



Note: For a specific year to be plotted, at least 50% of the data for each season must be valid.

Figure X-6. Annual arithmetic mean and visibility metric statistics, YOSE, transmissometer data (filtered).

- Point versus Path Measurements - Reconstructed extinction represents an indirect measure of extinction at one point source. The transmissometer directly measures the irradiance of light (which calculated gives a direct measure of extinction) over a finite atmospheric path.

Reconstructed extinction is typically 70%-80% of the measured extinction. With a ratio of 70%, this relationship shows fair agreement for YOSE.

c. Camera Data - View Monitoring

An automatic 35mm camera system operated at YOSE from September 1983 through April 1995. Color 35mm slide photographs of Half Dome were taken three times per day until July 1985, when the system was stolen for a second time. Photographic monitoring of the Telegraph Hill vista continued at a more secure location from September 1984 through April 1995.

View monitoring slides document visual conditions and are an effective tool for interpreting the visual effects of measured optical and aerosol parameters or presenting monitoring program goals, objectives, and results to decision-makers and the public. The Telegraph Hill vista photographs presented in Figure X-7 were chosen to provide a feel for the range of visibility conditions possible and to help relate the extinction/SVR/haziness data to the visual sense.

d. Site-Specific Data Interpretation

Data from other IMPROVE visibility sites around the country have been presented graphically (Figures I-3 and I-4) so that spatial trends in visual air quality for the Yosemite and Sierra Nevada regions can be understood in perspective. Figure X-4 and X-5, as well as Tables X-13, X-14, and X-15, have been provided to summarize YOSE visual air quality during the March 1988 through February 1999 period. Seasonal variances in the mean of the dirtiest 20% fractions are driven primarily by organic and nitrate extinctions. Non-Rayleigh atmospheric light extinction at YOSE, is largely due to organics and sulfates. Historically visibility varies with patterns in weather, winds (and the affects of winds on coarse particles) and smoke from fires. No information is available on how the distribution of visibility conditions at present differs from the profile under "natural" conditions, but the cleanest 20% of the days probably approach natural conditions (GCVTC, 1996). Smoke from frequent fires is suspected to have reduced presettlement visibility below current levels during some summer months.

Long-term trends fall into three categories: increases, decreases, and insignificant changes. The characterization of long-term trends can be a highly subjective exercise in that slopes and their significance can vary depending on the technique employed. Recently the IMPROVE aerosol network, initiated in March 1988, matured to a point where long-term trends of average ambient aerosol concentrations and reconstructed extinction can be assessed. In the recent IMPROVE report (Malm et al. 2000), the authors applied the Theil [1950] approach to describe trends for IMPROVE sites with eleven years of data. The distribution of $PM_{2.5}$ mass concentrations, reconstructed extinction expressed as deciview, and associated constituents were examined for each site. The data were sorted into three groups based on the cumulative frequency of occurrence of $PM_{2.5}$: lowest fine mass days, 0-20%, median fine mass days, 40-60%, and highest fine mass days, 80-100%. After sorting each group's average concentrations of $PM_{2.5}$ and selecting the associated principal aerosol species, scattering and/or absorption of each species, reconstructed light extinction and deciview were calculated. Figure X-8 shows plots of the 10, 50, and 90 percentile groups at YOSE for both $PM_{2.5}$ and deciview.

Given the visibility data summarized for YOSE, the majority of data show insignificant change over the period. In addition, the Malm et al. (2000) analysis of the long term five-year average deciview at YOSE also showed no significant increasing or decreasing trend.

D. RESEARCH AND MONITORING NEEDS

1. Deposition

Continue monitoring wet and dry deposition.

2. Gases

Ozone monitoring sites should be continued. In addition, an investigation of spatial variations in ozone exposure is recommended.

3. Aquatic Systems

Continued periodic monitoring of several acid-sensitive, high elevation lakes in YOSE should be a high priority. There are a number of highly sensitive candidates, including several lakes sampled in the WLS and again in 1999 by the USGS (most without names): WLS identification codes 4A1-006, 4A1-016, 4A1-008, 4A1-055, and 4A1-013 (Roosevelt Lake). A group of about three of these lakes should be selected on the basis of sampling logistical considerations for long-term monitoring. At a minimum, monitoring should include annual sampling during July. Additional (i.e., monthly) sampling should also be conducted at least every three years during the snowmelt period (April through July). The logistics of high-elevation lake sampling in YOSE are very difficult, especially during spring, when the lakes would have to be accessed by skis from the east side of the park. However, YOSE contains some of the most acid-sensitive aquatic ecosystems in the world. Small increases in N or S deposition would increase the severity of episodic acidification during snowmelt and cause some lakes and streams to become episodically (and perhaps chronically) acidic. Under such conditions, adverse biological effects would be expected to occur. The collection of monitoring data is required to determine if and when these lakes become acidic in response to potential future increases in N or S deposition. Biological monitoring would also be very useful and could be accomplished in conjunction with chemical monitoring. Zooplankton constitutes the group of aquatic invertebrates that is easiest and most reliable for detection of biological impacts associated with acidification of Sierra Nevada lakes (Melack et al. 1989a).

Acid-sensitive lakes in the park should be included in park-wide and/or regional modeling efforts to determine the levels of S or N deposition that would be likely to cause chronic or episodic acidification in the future. Such modeling should be statistically-based so that results and conclusions are applicable to the population of acid-sensitive lakes in the park or throughout the region. There are several process-based watershed models available for this effort.

Additional research is needed on the cause of the observed large declines in amphibian populations and distributions in YOSE and elsewhere in the Sierra Nevada. It is not clear whether or not air pollution has played a role in amphibian decline; at this point, it cannot be definitively ruled out. There is also a critical need for continued monitoring of known amphibian populations to assess trends. However, monitoring amphibian populations as an indication of acidification effects may be problematic, given the extent of population declines of the mountain yellow-legged frog and Yosemite toad. Biological surveys in the spring and summer may be sufficient for determination of population trends over a period of years (Drost and Fellers 1994).

4. Terrestrial Systems

Existing monitoring sites should be sufficient to quantify pine injury in YOSE for the time being. Monitoring protocols should be the same as those used in the FOREST study (Arbaugh et al. 1998), including the use of Ozone Injury Index (Duriscoe et al. 1996, Schilling and Duriscoe 1996) to quantify injury.

Because YOSE already has widespread ozone injury in pines, it is recommended that the park consider including some additional ozone bioindicators in a long-term monitoring plan in order to provide a more comprehensive description of the effects of air pollution on vegetation. Species such as quaking aspen, black cottonwood, and blue elder, whose symptoms are well documented, as well as selected understory species, could be examined in monitoring plots co-located with injured pines. Monitoring protocols from the Forest Health Monitoring manual (USDA Forest Service 1999) are recommended for establishing plots and collecting data (see Appendix).

5. Visibility

IMPROVE monitoring should continue at the park. In addition, aerosol concentrations can vary throughout the park. Therefore, the extent and nature of visibility impairment should be monitored at YOSE. The use of aerosol and optical monitoring could provide a profile of visibility impairment. Industrial processes and wildland fires contribute to visibility impairment at YOSE. Therefore, a program to monitor carbon should be developed and conducted to identify emissions from industrial processes and wildland fires.

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APPENDIX

Forest Health Monitoring protocols

The recently published Forest Health Monitoring Field Methods Guide (USDA Forest Service 1999) is an excellent source of information on quantifying the effects of air pollution and other biophysical factors on vegetation condition. This guide includes information on forest measurements, crown-condition classification, damage and mortality assessment, ozone bioindicator plants, lichens, and soil measurements and sampling. It is recommended that this guide be used to establish new monitoring plots and/or to modify older plots. The section on ozone bioindicator plants in the western United States is particularly relevant for Class I national parks included in this report. A consistent set of protocols among national parks and between national parks and national forests will facilitate comparisons at broad spatial scales and over time.

Protocols for measuring ozone injury in pines

Because several of the Class I national parks in California contain the ozone bioindicators ponderosa pine and Jeffrey pine, it is desirable to have uniformity of methods across parks and between agencies. The FOREST study (Arbaugh et al. 1998) demonstrated that a uniform methodology of assessing injury in pines applied across a large number of locations in California could produce consistent spatial patterns and statistically significant correlations with ozone exposure. Parks that are monitoring the condition of bioindicator pines and other tree species should consult USDA Forest Service (1999) and Miller et al. (1996) for an excellent summary of the technical aspects of field survey methods, including pollutant pathology (Duriscoe et al. 1996), descriptions of biotic and abiotic symptoms (Stolte 1996), and calculation of the Oxidant Injury Index (OII; Schilling and Duriscoe 1996). Ozone injury index is calculated based on those morphological characters that express the primary effects of ozone on pines: (1) visible injury expressed as chlorotic mottle (x_1), (2) number of needle whorls retained (x_2), (3) percent live crown (x_3), and (4) modal needle length (x_4), where $OII = 0.4x_1 + 0.4x_2 + 0.1x_3 + 0.1x_4$. The advantage of using OII is that it has been shown to be a quantitatively robust measure of ozone injury and is well correlated with other measurements such as the Forest Pest Management index (FPM; Pronos et al. 1978) and Oxidant Injury Score (Miller 1973), thereby facilitating regional comparisons in space and time.

Lichen Monitoring

Several species of lichens in California are known to be sensitive to air pollutants (Nash and Wirth 1988, D. Peterson et al. 1992, Stolte et al. 1993). As a taxonomic group, lichens are widespread in all of the parks, with a variety of habitats and growth forms. The sensitivity of lichen species to air pollutants is summarized in this report within the sections on individual parks.

However, most species of lichens are poor bioindicators for technical and practical reasons. First, they are difficult to identify and generally require the assistance of someone trained in lichen taxonomy to provide confident identification. Second, injury symptoms are difficult to diagnose and describe for all but a few genera. Again, a lichenologist is usually required to provide a good assessment. Visual lichen pathology is regarded even by lichenologists as an intractable field method. However, there has been considerable support for assessment of (1) spatial patterns of species diversity with emphasis on assumed bioindicator species, and (2) chemical analysis with emphasis on elements such as sulfur and lead (Jackson et al. 1993, McCune and Geiser 1997).

In a classic study of how air pollution can affect spatial patterns of lichen diversity, Sigal and Nash (1983) determined that forests in the San Bernardino Mountains had a depauperate lichen flora that was greatly depleted compared to botanical collections earlier in the century. This provided compelling evidence for the effects of elevated levels of air pollution in the Los Angeles Basin on numerous sensitive lichen taxa. This study inferred that ozone caused the reduction in lichen flora diversity, although it has also been suggested that SO_2 may have had some effect (J. Bennett, pers. comm.). High nitrogen loading in this region (Fenn and Poth 1999) may also be a factor. A general concern with lichen diversity assessments is the effects of climatic variation (precipitation, temperature, snowfall, ice, etc.) at both large and small scales on lichen abundance and distribution. The effects of this variation must be discriminated from any potential air pollution signal. In general, there are many potential causes for the abundance and distribution of lichens at various spatial scales.

While it has been suggested that chemical analysis of lichen tissue can be used to measure the effects of regional air pollution, it is probably an effective tool only for measuring the acute effects of point-source pollution. First, the phytotoxicity of various elements on individual lichen species is virtually unknown. Second, the cause-effect relationship of pollutant emission/dispersion with chemical composition of lichen tissue is extremely difficult to quantify.

Third, because most lichens are accumulators of chemicals (i.e., minimal tissue turnover) and because pollutant dispersion has complex relationships with climatic variation and pollutant deposition (especially in mountainous topography), it is difficult to correlate actual variation in deposition with lichen tissue chemistry. A sparse data base of deposition monitoring further compounds the problem of quantitative analysis.

For all of these reasons, the use of lichens as bioindicators is not generally a quantitatively robust or cost-effective means of detecting and measuring the effects of air pollutants. It is reasonable for most parks to conduct lichen surveys as a reference point of general floristic composition. These surveys—in cooperation with experienced lichenologists—can be conducted periodically at permanent plots over time to determine if there have been any changes in distribution and abundance of species. The cause of any changes must be carefully inferred and distinguished from normal variation. Parks wishing to undertake assessments of lichens should consult Stolte et al. (1993) and USDA Forest Service (1999) for protocols and field methods.

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