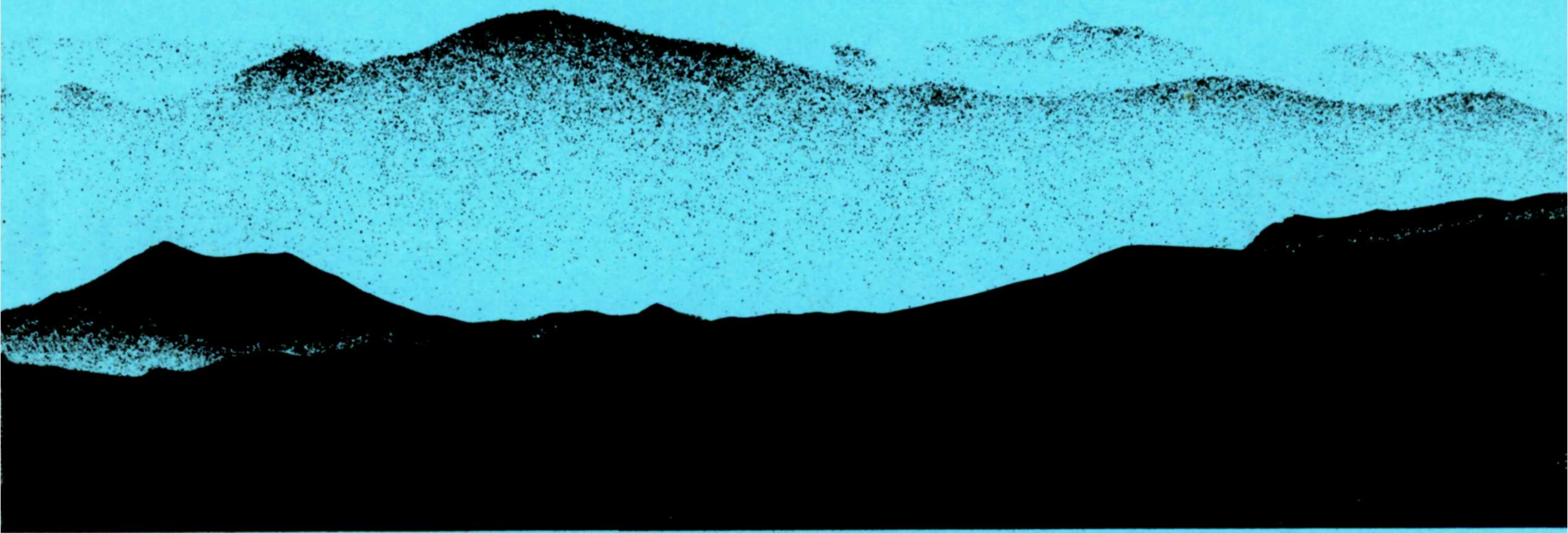
Assessment of Air Quality and Air Pollutant Impacts in National Parks of the Rocky Mountains and Northern Great Plains

August 1998



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EXECUTIVE SUMMARY

This report summarizes current and potential future conditions of air quality and air pollution impacts in national parks of the Rocky Mountains and northern Great Plains region, and recommends monitoring and research activities that could be implemented to acquire critical new data. The focus of this report is on the effects of criteria pollutants (ozone, SO_2, NO_x, particulates) on air quality and on air quality related values (terrestrial vegetation, aquatic systems, visibility). Other pollutants (e.g., H_2S, fluoride) are also discussed for specific locations where these pollutants may have potential impacts on natural resources.

In general, air quality in the Rocky Mountains and northern Great Plains region is considerably better than in most other areas of the continental United States. This is primarily due to the absence of high levels of fossil fuel combustion associated with metropolitan areas and because atmospheric conditions are not highly conducive to the formation and accumulation of ozone. Although current air quality is good in most areas except the Front Range of the Colorado Rockies, some pollutants may have potential impacts on national park resources in both the short- and long-term future.

Emissions in the Rocky Mountains and northern Great Plains come from a variety of sources. Nearly all SO_2 emissions and over 70% of NO_x emissions are produced from fuel combustion by electric utilities and other industrial processes. High emissions of NO_x and VOCs in Colorado, particularly in the greater Denver metropolitan area, result in synthesis of relatively high concentrations of tropospheric ozone.

Ozone appears to be an imminent concern only at Rocky Mountain NP, which is the recipient of air masses originating from the Denver-to-Fort Collins area during the summer. There are known sensitive vegetative bioindicators in the park, including ponderosa pine, quaking aspen and two lichen species. Because of the potential for increased ozone exposure in the park as the human population increases east of the Front Range, it is recommended that additional ozone monitoring be conducted and that vegetation monitoring plots be considered.

Atmospheric nitrogen deposition is an important environmental concern at the present time only in Rocky Mountain NP. Moderately high levels of N deposition, coupled with high watershed sensitivity to acidification, have likely contributed to some episodic acidification of surface waters in this park at current levels of nitrogen deposition. Relatively small amounts of chronic surface water acidification also cannot be ruled out. Continued or increased monitoring of surface water resources in Rocky Mountain NP is recommended.

A high degree of sensitivity to acidification was also demonstrated for surface waters in Grand Teton NP, although current levels of deposition are somewhat lower than in Rocky Mountain NP. Again, continued or increased monitoring of surface water resources is advised. In both Rocky Mountain and Grand Teton NPs, high elevation lakes constitute an important sensitive aquatic resource of concern.

Sulfur deposition currently is low at most of the parks in the region, and at current levels of deposition does not appear to be a major concern with respect to acidification of lakes and streams. However, emissions of SO_2, as well as H_2S, from many small oil and gas production facilities in North Dakota may have potential impacts on vegetation on a regional (for SO_2) and local (for H_2S) basis in Theodore Roosevelt NP. Sulfur deposition should continue to be monitored throughout the region. Emissions of sulfur compounds in North Dakota will likely increase in direct proportion to the number and productivity of active oil and gas wells. Sulfur dioxide concentrations may increase considerably throughout the Rocky Mountains and northern Great Plains if additional coal-fired power plants are built, because these new sources would have the potential to disperse pollutants long distances.

Visibility is a source of concern for all parks in the Rocky Mountains and northern Great Plains because of the magnificent vistas viewed from within and looking into the parks. In general, the best visibility in the conterminous United States typically occurs in the Great Basin, Colorado Plateau, and Rocky Mountains. Regional and local pollution sources that appear to contribute to visibility impairment throughout the Rocky Mountains and northern Great Plains include automobiles, coal- and oil-fired power plants, smelters, wildfires, and urban emissions. Average concentrations of sulfates, organics, and elemental carbon are highest in summer, while nitrate concentrations are generally highest in winter and spring.

National Park Service monitoring data from 1988 to 1995 show that the annual average standard visual range for the central Rocky Mountains (represented by monitoring data from Rocky Mountain NP and Yellowstone NP) is 123 km; organics are the largest contributor to annual aerosol extinction (29%), followed by sulfates (24%). The annual average standard visual range for the northern Rocky Mountains (represented by monitoring data from Glacier NP) is 69 km; sulfates are the largest contributor to annual aerosol extinction (32%), followed by organics (28%). The annual average standard visual range for the northern Great Plains (represented by monitoring data from Badlands NP) is 85 km; sulfates are the largest contributor to annual aerosol extinction (40%), followed by organics (18%).

Long-term future visibility impairment may result from increased organics and elemental carbon concentrations due to possible increased gas and mineral developments. In the central Rocky Mountain region, higher organic concentrations may result from increased population growth along the Colorado Front Range. Visibility may also be affected by prescribed fires and wildfires, with the specific effects of fire on visibility varying interannually depending on the frequency and intensity of fires adjacent to a specific park and on climatic conditions.

Because relatively little is known about the dispersion of air pollutants in the Rocky Mountains and northern Great Plains, the contribution of pollutants from distant metropolitan areas such as Salt Lake City or from coal-fired power plants is poorly quantified. In addition, there are few data on the pollutant sensitivity of the native flora of this region. Finally, there is little information on the aquatic chemistry of the thousands of lakes and streams in the mountainous parks. Having this sort of data would greatly enhance the ability of national park resource managers to anticipate and identify potential impacts. We encourage the National Park Service to participate in modeling and data collection efforts, in cooperation with other agencies when appropriate, that would augment current knowledge about the impacts of air pollution on terrestrial and aquatic resources in the region.

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 and SO_2 are more expensive to obtain, it 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, SO_2, and visibility. Similarly, additional measurements of acid neutralizing capacity and other characteristics of high mountain lakes 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, should there be sufficient interest and funding for such activities in the future.

I. INTRODUCTION

National parks in the Rocky Mountains and northern Great Plains 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 parks in the Rocky Mountains and northern Great Plains receive generally low levels of atmospheric pollutants (Sisterson et al. 1990, Smith 1990). Nevertheless, sensitive aquatic and terrestrial ecosystems, especially those at high elevations, can potentially be degraded by existing or future pollution (Peterson et al. 1992). Elevated emission levels of sulfur dioxide (SO_2) and nitrogen oxides and elevated concentrations of ozone (O_3) have been measured in some portions of these regions (Sisterson et al. 1990). Some areas also have impaired visibility.

Recognizing the valuable role that scientific research can play in proper management of a park, the NPS Director commissioned the National Research Council to review the Agency'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 national parks in the Rocky Mountains and northern Great Plains.

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 national parks. Knowledge of emissions inventories, coupled with scientific understanding of the effects of pollutants on natural resources, will provide land managers with a framework within which to protect sensitive resources within the parks 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 states 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 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 6,000 acres and national wilderness areas over 5,000 acres, 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 the Rocky Mountains (Colorado, Montana, and Wyoming) include (alphabetically) Glacier National Park (GLAC), Grand Teton National Park (GRTE), Rocky Mountain National Park (ROMO), and Yellowstone National Park (YELL). NPS Class I areas within the northern Great Plains (North Dakota and South Dakota) include Badlands National Park (BADL), Theodore Roosevelt National Park (THRO), and Wind Cave National Park (WICA).

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 the Rocky Mountains and northern Great Plains. 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 synthesis of knowledge on the ecological effects of atmospheric pollutants, and a park-specific assessment of pollution vulnerability.

B. SCOPE AND ORGANIZATION

This report is based on a concern for the ecological integrity of Class I areas. Thus, the scope is limited to addressing potential threats 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, fluoride, and hydrogen sulfide is covered in less detail, and radionuclides and organic toxins are not addressed. With regard to trace metals, an extensive analysis of the topic is not justified, because at present the problem is localized in proximity to industrial smelters. Also, information is lacking with which to evaluate the dispersal of trace metals in the Rocky Mountain and northern Great Plains regions via airborne particles.

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. For example, little mention is made of the direct effect of pollutants on fish or other important taxonomic groups. These omissions often reflect the lack of information on these topics rather than any reflection on their ecological significance.

It is hoped that this report will serve audiences including staff with the NPS Air Resources Division, regional air quality officers, individual park staff and organizations dealing with air quality issues in the Rocky Mountain and northern Great Plains regions. 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. Deposition

There are relatively few data on air quality in the Rocky Mountain and northern Great Plains regions, with most data being short-term and/or from urban areas remote from wildland locations. General patterns of N and S deposition can be inferred from national databases, but ozone distribution is more difficult to estimate. It is known that air pollution in urban areas adjacent to the Rockies, especially at locations east of the Front Range in Colorado, has increased considerably during the last 30 to 40 years. Dispersion and transport of pollutants vary locally, but there are clearly periods of high ozone exposure every summer. The challenge is to quantify the spatial distribution of this exposure from the limited database. 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 data on dry deposition are available at four sites in the Rocky Mountain region, collected within the Clean Air Status and Trends Network (CASTNet). The sites are located at Gothic, CO; Pinedale and Centennial, WY; and Glacier NP, MT. Sites have also been installed in ROMO and YELL, but data from these sites are not yet available. The CASTNet file contains weekly atmospheric concentration (from filter pack samplers), deposition velocity, and estimated weekly dry deposition fluxes of S and N. Dry deposition estimates were extremely low at all sites, well under 0.5 kg/ha/yr for each constituent. Based on these data, the general scarcity of significant point sources in close proximity to the parks, and the generally low levels of measured wet deposition, we conclude that dry deposition of S and N at parks considered in this report is low to negligible.

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 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.

2. 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

Table I-1. National Ambient Air Quality Standards.

Constituents	Averaging Time	Primary Standard	Secondary Standard
	Annual Arith. Mean	30 ppbv	none
Sulfur Dioxide	24-hour[a]	140 ppbv	none

	3-hour[a]	none	500 ppbv
PM-10	Annual Arith. Mean[b] 24-hour[b]	50 $\mu g/m^3$ 150 $\mu g/m^3$	same same
PM-2.5	Annual[b] 24-hour[b]	15 $\mu g/m^3$ 65 $\mu g/m^3$	same same
Carbon Monoxide	8-hour[a] 1-hour[a]	9,000 ppbv 35,000 ppbv	same same
Ozone	8-hour[c]	80 ppbv	same
Nitrogen Dioxide	Annual Arith. Mean	50 ppbv	same
Lead	Calendar Quarter	1.5 $\mu g/m^3$	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.

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 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, and particulate matter (Table I-2), and (4) not cause or contribute to adverse impacts to AQRVs in any Class I area (Peterson et al. 1992).

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 for 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 the identified critical dose to be exceeded?
- If the critical dose is exceeded, what amount of additional pollution is considered “insignificant”?

Table I-2. Prevention of significant deterioration increments (in $\mu g/m^3$); PSD increments are not defined for ozone.

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

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. 1992).

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. The level of ozone in a "pristine" area may be as low as 10 to 25 ppbv (weekly average) (Altshuller and Lefohn 1996, Cooper and Peterson 1996). Areas in the southeastern United States and 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 tend to increase with elevation (Loibl et al. 1994; Sandroni et al. 1994; Brace and Peterson 1996,1998).

Ozone causes injury to highly sensitive species of plants at concentrations as low as 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 photosynthesis rates, lowered sugar production, reduced growth and possibly death.

b. Sulfur Dioxide

Sulfur dioxide (SO_2) is a product of fossil fuel production. Some of the largest emitters are coal-fired electric power plants and smelters. 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 sulfur 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. Fluoride

Fluoride occurs naturally at low levels in soils, air, and water, and it also is emitted through industrial processes including aluminum manufacturing. Fluoride occurs in two forms, gaseous (HF) and particulate. Soil fluoride is mainly in particulate form and increases in concentration with increased soil acidity. The background level of fluoride in most soils ranges between 0.05% and 1%. Particulate fluoride is less readily absorbed by plants than the gaseous form. Background levels of fluoride in plants are around 2 to 20 ppm (Treshow and Anderson 1989). Plant tissues accumulate fluoride and sensitive species may begin to show symptoms of damage at levels around 30 to 50 ppm (Treshow and Anderson 1989). Marginal leaf necrosis is the classic symptom of fluoride toxicity. Other symptoms in plants include leaf tip burn, chlorosis, necrotic spotting, accelerated foliar senescence, and reduced growth rates (Treshow and Anderson 1989, Smith 1990).

d. Hydrogen Sulfide

Hydrogen sulfide (H_2S) is a highly toxic gas that is formed by the decomposition of organic matter containing sulfur. It is found in mines, wells, sewers, cesspools, natural gas, volcanic gas and in some spring waters. It is a colorless, flammable gas with a density greater than that of air. The “rotten egg” odor of H_2S gas is one of the most easily detectable odors to the human nose with an odor threshold less than 5 ppbv (Weiss 1986). Concentrations between 20 and 150 ppmv cause irritation of the eyes and slightly higher concentrations cause irritation of the upper respiratory tract (Sax and Lewis 1989). Exposures of between 600 and 1000 ppmv may be fatal to humans within 30 minutes. It is highly corrosive to standard metals and causes paint discoloration.

Air pollution by H_2S is not a widespread problem but rather is typically localized near emitters such as oil and gas wells, kraft paper mills, sewage treatment plants, petroleum refineries and coke ovens. Background levels for H_2S are estimated to be less than 0.2 ppbv (Smith 1990). Little work has been done on H_2S effects on wildlife and vegetation (Bicknell 1984). Research has focused on the criteria pollutants and therefore H_2S has received only occasional attention, perhaps because H_2S is readily converted to SO_2 and sulfate in the atmosphere. However, limited information indicates that H_2S can cause symptoms in some plant species. Douglas-fir was found to show slight symptoms in response to chronic concentrations as low as 0.1 ppmv and extensive foliar damage to concentrations of 0.3 ppmv (Thompson and Kats 1978). In general, at acute doses of 100 ppmv for 5 hours, angiosperms develop interveinal necrotic blotches while gymnosperms develop distal necrosis (Smith 1990).

e. 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.

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.

4. Vegetation and Bioindicators

Although acidic deposition was originally cited as a likely cause of reduced vigor in some forests, oxidants can reduce photosynthesis, growth, and productivity of sensitive plant species, even at relatively low exposure levels (Reich and Amundson 1985) and are a more likely stressor of vegetation in much of North America. Chronic exposure to ozone can cause substantial loss of vigor, which can in turn lead to greater susceptibility to additional stresses. The fact that ozone concentrations are often higher downwind from urban areas and that concentrations tend to increase with elevation, suggests that national park lands can be particularly vulnerable. Despite known general relationships between regional “sources” of ozone (and its precursors) with wildland “receptors”, it has been extremely difficult to estimate ozone exposure in those wildlands because of a sparse dataset collected primarily in urban areas. Only limited on-site data are available in mountainous locations. In addition, identification of possible symptoms of air pollutant injury is difficult, because visual symptoms are generally poorly documented for vegetation in western North America. The physiological functions of some lichen species are affected by air pollutants, but diagnostic symptoms are difficult to quantify in the field.

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. Much of this work has been conducted on species native to the northeastern and southwestern United States, and little work has been conducted on species of the Rocky Mountain and Great Plains regions.

Pollutants can cause injury to various plant tissues including leaves, stems and roots. Foliar injury is the most visible form of injury, although it often can be confused with pathogen or fungal diseases. Ozone symptoms in conifer and hardwood foliage that could be considered "typical" include chlorosis, stipple (uniform black spots) and accelerated needle and leaf loss (Miller et al. 1983, Hogsett et al. 1989, Treshow and Anderson 1989). Sulfur dioxide-induced injury includes dieback of leaf and needle tips and necrotic spots (Treshow and Anderson 1989). There are few data on the effects of NO_2 on plant species, and existing information stems from scattered studies in Europe and the United States.

Sensitivity classes can be used to relate potential foliar injury in trees to pollutant exposures for ozone (Tables I-3 and I-4). These sensitivity classes can be used to identify severity of injury as well as to indicate potential injury thresholds. Any visible injury symptom can be considered a significant impact, regardless of sensitivity class. If pollutant concentrations listed in the tables are reached, then special attention should be directed at identifying injury symptoms. Ozone-sensitivity tables are a rough guideline in which injury level and/or pollutant exposure are keyed to the general sensitivity of a sensitive receptor.

Table I-3. Sensitivity classes for conifers in relation to ozone exposure. (Source: after Peterson et al. 1992)

Sensitivity class	Needle age class with chlorotic mottle (years)	Needle retention as percent of normal (%)	Ozone concentration (7-hour growing season mean) (ppbv)[a]
Low	$\geq$ 5	71-80	61-70
Medium	3-4	41-70	71-90
High	1-2	< 40	> 90

[a] Highest continuous 7-hr period

Table I-4. Sensitivity classes for hardwoods in relation to ozone exposure.

(Source: after Peterson et al. 1992)		
Sensitivity class	Percent leaf area with stippling	Ozone concentration (7-hour growing season mean) (ppbv)[a]
Low	21-40	71-90
Medium	41-60	91-120
High	60-100	>120

[a] Highest continuous 7-hr period

Ozone-sensitivity classes for conifers are based primarily on symptomatology of ponderosa pine (*Pinus ponderosa*) 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). Because these data are from quantitative studies at a variety of locations, Table I-3 should be quite relevant for assessing ozone injury in ponderosa pine. Applicability to other coniferous species may vary considerably with respect to specific symptoms and pollutant exposure.

Ozone-sensitivity classes for hardwoods 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 under field and experimental 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 aspen includes chlorosis, stippling, necrotic spotting, and leaf margin burn. Symptoms generally vary seasonally, with stippling being most prominent in the spring and black, bifacial (both leaf surfaces) necrosis appearing in late summer (J.P. Bennett, pers. comm.). Great care must be taken in distinguishing ozone symptoms from various pathogens and insect herbivores commonly found on this species. Table I-4 should be relevant for assessing ozone injury in aspen. Applicability to other hardwood species may vary considerably with respect to specific symptoms and pollutant exposure.

There are few data on potential SO_2- and NO_2-induced injury to tree species of the Rocky Mountains and Great Plains. Sensitivity and general exposure guidelines for SO_2 and NO_2 are based on relatively few data and sources of information and include a large amount of subjective judgment based on expert opinion (Peterson et al. 1992). For this reason, we do not present such guidelines. Any evaluation of potential pollutant impacts should consider the fact that some species are more affected by episodic, high concentrations, while other species are more affected by long-term, chronic exposures.

There are no data on injury symptoms for lichens growing in the Rocky Mountain and Great

Plains regions. Data from studies done in other regions provide general information on sensitivity of lichens to ozone and SO_2 but very little information exists on lichen sensitivity to nitrogen. Three sensitivity classes of lichens can be defined as low, medium and high (Table I-5). These classes are based on limited experimental data and should be regarded as general when applied to field situations.

Table I-5. Sensitivity classes for lichens based on prolonged exposure. (Source: Peterson et al. 1992)

Sensitivity classes	Ozone concentration (ppbv, 24-hr growing season mean)	SO_2 concentration (ppbv, 24-hr growing season mean)
Low	>65	>30
Medium	15-70	10-35
High	<20	5-15

Nitrogen is a critical nutrient for many plant metabolic processes. Long-term deposition of elevated levels of N compounds may affect soil microbiological processes, resistance to insects and pathogens, winter injury in conifers, and foliar leaching. Perhaps more important are the potential effects of long-term N deposition on ecosystem structure and diversity. Nitrogen is a potential fertilizer that can be assimilated preferentially by some plant species (Miller et al. 1976); for example, increased N deposition could cause plant species in a N-poor system (e.g., a bog) to be replaced by species with higher N requirements.

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 nitrate (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)

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, SO_4^{2-} or NO_3^- concentrations are mainly balanced by increasing base cation concentrations and some decrease in alkalinity. 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 sulfur or nitrogen 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:

<u>Dilute</u> - 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.

<u>Acid neutralizing capacity</u> - 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 Rocky Mountain region, 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.

<u>Base cations</u> - 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) causes water to be naturally low in pH and ANC, or even to be acidic (ANC<0), but also imparts 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.

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. This is most likely to happen in groundwater recharge lakes where nutrient inputs are derived largely from deposition to the lake surface. 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).

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). 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). A new mechanistic model of N dynamics and leaching from forested ecosystems that is currently being extensively tested is the Model of Ecosystem Retention and Leaching of Inorganic Nitrogen (MERLIN; Cosby et al. in press).

6. Visibility

The best visibility in the conterminous United States typically occurs in the Great Basin, Colorado Plateau, and Rocky Mountains. 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 (IMPROVE) program. IMPROVE monitoring is conducted at four (4) NPS Class I areas in the central and northern Rocky Mountains, and northern Great Plains: BADL, GLAC, ROMO, and YELL. Table I-6 summarizes visibility monitoring across the Rocky Mountains and northern Great Plains since 1988.

Table I-6. Visibility monitoring in Class I National Parks of the central Rocky Mountain, northern Rocky Mountain, and northern Great Plains regions.

Region	National Park	Visibility Monitoring		
		Particle (Aerosol)	Optical	View
Central Rocky Mountains	GRTE	None	None	9/14/85 - 9/22/91
	ROMO	3/1/88 - Present	11/25/87 - Present	10/25/85 - 4/1/95
	YELL	3/1/88 - Present	7/18/89 - 7/28/93	9/4/86 - 4/1/95
Northern Rocky Mountains	GLAC	3/1/88 - Present	3/1/89 - Present	6/14/85 - 4/12/95
Northern Great Plains	BADL	3/1/88 - Present	1/13/88 - Present	8/5/87 - 4/11/95
	THRO	None	None	9/4/85 - 9/20/91
	WICA	None	None	10/2/85 - 8/11/91

Program descriptions and spatial distribution summaries presented in the following sections were extracted from “Spatial and Temporal Patterns and the Chemical Composition of the Haze in the United States: An Analysis of Data from the IMPROVE Network, 1988-1991” (Sisler et al., 1993) and “Spatial and Seasonal Patterns and Long Term Variability of the Composition of the Haze in the United States: An Analysis of Data from the IMPROVE Network” for 1992-1995 (Sisler et al., 1996).

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, hygroscopic 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 wavelengths of visible light (0.4 - 0.7 μm) 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.

Gases and particles also absorb light. 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. Primary sources of elemental carbon include diesel exhaust and smoke.

With few exceptions, average concentrations of sulfate, organics, and elemental carbon are highest in summer. Nitrate concentrations are generally highest in winter or spring. Soil concentrations are highest in spring or summer.

c. IMPROVE Station Description and Rationale

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

Particle Monitoring

Particle monitoring provides concentration measurements of atmospheric constituents that contribute to visibility impairment. Four independent IMPROVE sampling modules are used to automatically collect two 24-hour samples of suspended particles each week by drawing air through filters. Three of the four samplers (modules A, B, and C) collect fine particles with diameters <2.5 μm. Particles of <2.5 μm are especially efficient at scattering light. The fourth sampler (module D) collects coarse particles with diameters up to 10 μm. Coarse particles do not scatter light well, and therefore, typically do not contribute much to visibility impairment. 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, and organic and elemental carbon species. In addition, the module A filter is used to estimate the light absorption coefficient (b_{abs}). All data obtained from the IMPROVE sampling modules are used to determine chemical concentrations and to reconstruct extinction from known extinction efficiencies of certain species.

Optical Monitoring

Optical monitoring provides a 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. 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, however, 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 Rocky Mountains area, no nephelometer monitoring has been conducted for any individual NPS Class I area in the Central or Northern Rocky Mountains or Northern Great Plains regions.

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 five year (or greater) photographic monitoring record. No view monitoring has been conducted at any Central or Northern Rocky Mountains or Northern Great Plains region visibility monitoring site since April 1995.

The IMPROVE monitoring network currently consists of 55 sites with various configurations of particle, optical, and view monitoring equipment. Five IMPROVE sites are located in the Central Rocky Mountains Region, one site in the Northern Rocky Mountains Region, and one site in the Northern Great Plains Region. A detailed description of the entire IMPROVE visibility and particle monitoring network may be found in Sisler et al. (1996).

d. Overview of Visibility Conditions

Particle, optical, and view monitoring data for the Central Rocky Mountains, Northern Rocky Mountains, and Northern Great Plains regions were extracted from IMPROVE network data archives and summarized for the March 1988 through February 1995 period.

Figure I-1 summarizes the spatial distribution of total reconstructed light extinction (including Rayleigh) averaged by site over three years (March 1992 through February 1995), as presented in "Spatial and Seasonal Patterns and Long Term Variability of the Composition of the Haze in the United States: An Analysis of Data from the IMPROVE Network" for 1992-1995 (Sisler et al., 1996). Figure I-2 provides a graphic summary of measured light extinction by geographic region and by season for the period March 1988 through February 1995. 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 41 km (b_{ext} >95 Mm^{-1})
- Below Average - conditions represent a mean visual range from 41 km to 78 km (b_{ext} from 95 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 reconstructed or measured light extinction

(>100 Mm^{-1}) occurred in the eastern United States while the Colorado Plateau and the Great Basin regions had the lowest light extinction (< 27 Mm^{-1}).

IMPROVE network particle and optical (transmissometer) monitoring sites represented in Figures I-1 and I-2 respectively are listed by geographic region in Table I-7. Site-specific seasonal and annual averages of reconstructed and measured extinction for the period March 1988 through February 1995 are provided in Tables I-8 and I-9. 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%.

Tables I-8 and I-9 summarize the seasonal and annual averages of reconstructed light extinction coefficients for Central and Northern Rocky Mountains and Northern Great Plains regions (March 1988 through February 1995). It should be noted that these data are from one or two sites in each region and may not be representative of the entire regional area. Table I-8 also shows the breakdown of non-Rayleigh extinction among fine and coarse particle scattering, and absorption.

Figure I-2. 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 1995.

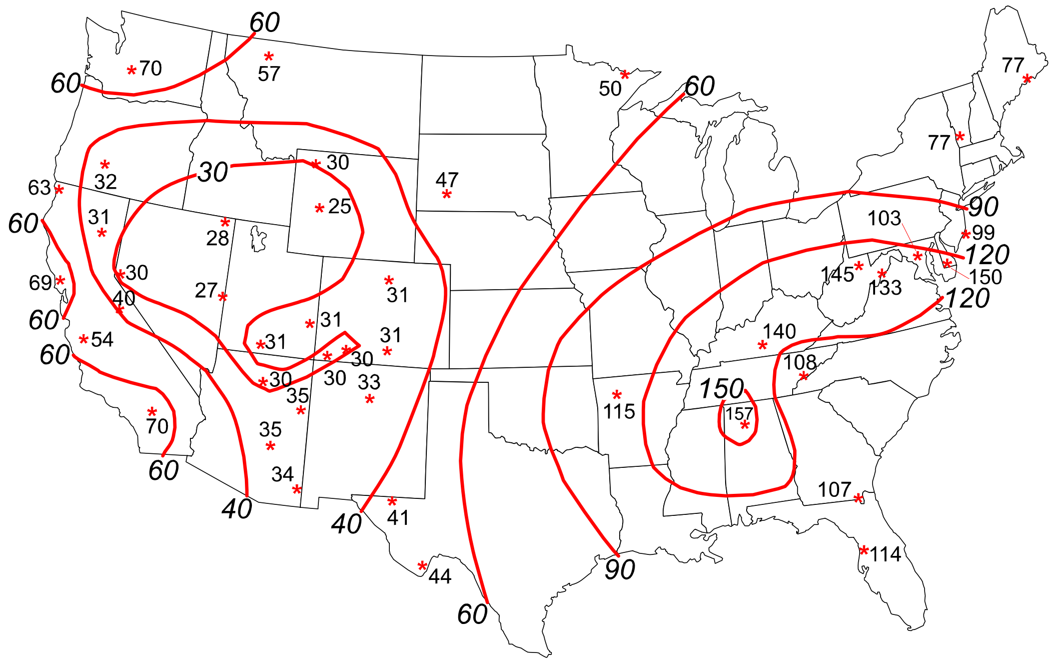


Table I-7. Operational particle and optical monitoring sites of the IMPROVE monitoring network March 1988 through February 1995 by geographic region.

Alaska
· Denali NP (A)

Appalachian Mountains
· Great Smoky Mountains NP (AN)
· Shenandoah NP (AT)
· Dolly Sods WA (AN)

Boundary Waters
· Boundary Waters Canoe Area (AN)

Cascade Mountains
· Mount Rainier NP (AN)

Central Rocky Mountains
· Bridger WA (AT)
· Great Sand Dunes NM (A)
· Rocky Mountain NP (AT)
· Weminuche WA (A)
· Yellowstone NP (AT)

Coastal Mountains
· Pinnacles NM (A)
· Point Reyes (A)
· Redwood NP (A)

Colorado Plateau
· Bandelier NM (AT)
· Bryce Canyon NP (A)
· Canyonlands NP (AT)
· Grand Canyon NP (ATN)
· Mesa Verde NP (A)
· Petrified Forest NP (AT)

Florida
· Chassahowitzka NWR (A)
· Okefenokee NWR (AN)

Great Basin
· Jarbidge WA (AN)
· Great Basin NP (AT)

Lake Tahoe
· D.L. Bliss State Park (ATN)
· South Lake Tahoe (AN)

Mid Atlantic
· Edmond B. Forsythe NWR (AN)

Mid South
· Upper Buffalo WA (AN)
· Sipsey WA (A)
· Mammoth Cave NP (AN)

Northeast
· Acadia NP (AN)
· Lye Brook WA (AN)

Northern Great Plains
· Badlands NP (AT)

Northern Rocky Mountains
· Glacier NP (AT)

Sierra Nevada
· Yosemite NP (AT)

Sierra-Humboldt
· Crater Lake NP (A)
· Lassen Volcanoes NP (A)

Sonoran Desert
· Chiricahua AZ (AT)
· Tonto NM (A)

Southern California
· San Gorgonio WA (AT)

Washington, D.C.
· Washington, D.C. (A)

West Texas
· Big Bend NP (AT)
· Guadelupe Mountains NM (AT)

NP = National Park
NM = National Monument
WA = Wilderness Area
NWR = National Wildlife Refuge
NS = National Seashore

A = Aerosol Sampler
T = Transmissometer
N = Nephelometer

Table I-8. Seasonal and annual average reconstructed light extinction (Mm^{-1}) apportioned by general category for the central Rocky Mountain, northern Rocky Mountain, and northern Great Plains regions (March 1988 through February 1995).

Season	Total Extinction	Natural Rayleigh Extinction	Non-Rayleigh Extinction	Fine Scattering	Coarse Scattering	Absorption
Central Rocky Mountains						
Spring	32.2	10.0	22.2	14.7	3.0	4.5
Summer	40.4	10.0	30.4	16.6	5.9	7.9
Autumn	30.8	10.0	20.8	12.8	3.2	4.8
Winter	23.6	10.0	13.6	9.2	1.9	2.5
Annual	31.7	10.0	21.7	13.2	3.6	4.9
Northern Rocky Mountains						
Spring	48.7	10.0	38.7	27.5	3.6	7.6
Summer	50.1	10.0	40.1	25.1	6.4	8.6
Autumn	65.8	10.0	55.8	38.6	5.1	12.2
Winter	66.9	10.0	56.9	45.8	2.2	8.8
Annual	57.1	10.0	47.1	33.5	4.3	9.3
Northern Great Plains						
Spring	50.4	10.0	40.4	29.6	4.1	6.7
Summer	48.3	10.0	38.3	25.3	5.4	7.6
Autumn	40.2	10.0	30.2	20.0	4.2	6.0
Winter	45.5	10.0	35.5	27.9	2.7	4.9
Annual	46.0	10.0	36.0	25.6	4.1	6.3

Table I-9. Contributions of various types of fine particles (Mm^{-1}) to the total seasonal and annual average non-Rayleigh aerosol light extinctions for the central Rocky Mountain, northern Rocky Mountain, and northern Great Plains regions (March 1988 through February 1995).

Season	Non-Rayleigh Aerosol Extinction	Sulfate	Nitrate	Organics	Elemental Carbon (Absorption)	Soil & Coarse Material
Central Rocky Mountains						
Spring	22.2	7.1	2.5	5.1	4.5	3.0
Summer	30.4	6.3	1.3	9.1	7.9	5.9
Autumn	20.8	5.0	1.4	6.4	4.8	3.2
Winter	13.6	3.2	1.5	4.5	2.5	1.9
Annual	21.7	5.3	1.6	6.3	4.9	3.6
Northern Rocky Mountains						
Spring	38.7	14.4	3.1	10.0	7.6	3.6
Summer	40.1	11.1	1.8	12.3	8.6	6.4
Autumn	55.8	15.0	4.9	18.7	12.2	5.1
Winter	56.9	21.2	12.4	12.3	8.8	2.2
Annual	47.1	15.3	4.9	13.2	9.3	4.3
Northern Great Plains						
Spring	40.4	18.1	5.6	5.9	6.7	4.1
Summer	38.3	15.2	1.4	8.7	7.6	5.4
Autumn	30.2	10.6	2.7	6.6	6.0	4.2
Winter	35.5	14.3	8.7	5.0	4.9	2.7
Annual	36.0	14.6	4.4	6.6	6.3	4.1

Table I-9 further identifies the contributions of sulfate, nitrate, organics, elemental carbon, and coarse particles (including fine soil) to the non-Rayleigh aerosol light extinction.

Figure I-3 graphically depicts the percentage of total light extinction (including Rayleigh) contributed by sulfate, nitrate, organics, elemental carbon, and coarse particles.

Central Rocky Mountains

Aerosol and optical measurements presented for this region were made at two locations in the mountainous Class I areas of Colorado and Wyoming at ROMO and YELL. The annual average total reconstructed extinction for the central Rocky Mountains for the March 1988 through February 1995 period is 31.7 Mm^{-1}, of which, 68% is due to aerosol extinction. The seasonal variation is significant with a maximum total extinction of 40.4 Mm^{-1} in summer and a minimum of 23.6 Mm^{-1} during winter. The seasonal variance is driven primarily by organic extinction and absorption. Organic extinction peaks at 9.1 Mm^{-1} in summer and drops to 4.5 Mm^{-1} in winter. Absorption (elemental carbon) ranges from 7.9 Mm^{-1} in summer to 2.5 Mm^{-1} in winter. Organics are the largest contributor to aerosol extinction during the summer months at 29.9%. Absorption follows closely

Figure I-3 Annual average percentage of total light extinction (including Rayleigh) contributed by sulfate, nitrate, organics, elemental carbon, and coarse particles for the central Rocky Mountain, northern Rocky Mountain, and northern Great Plains regions (March 1988 through February 1995).

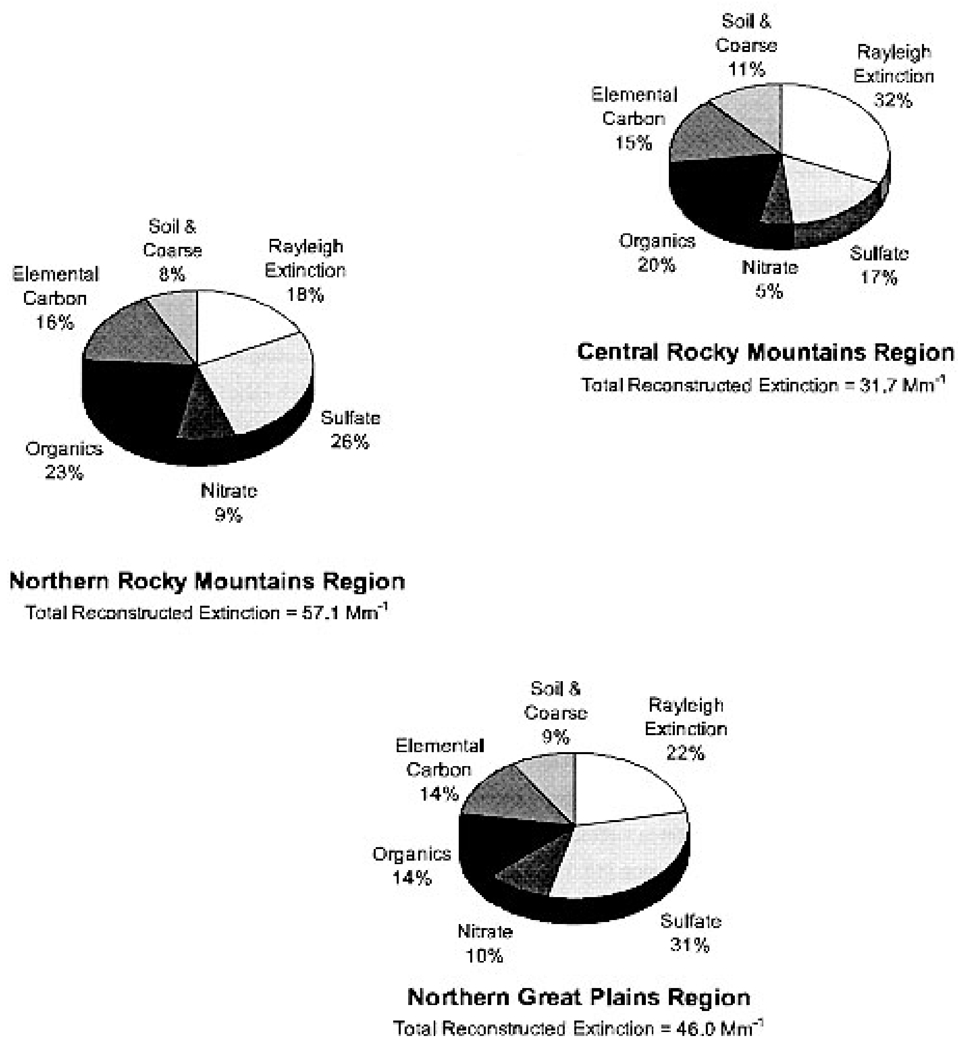


Figure I-3 Annual average percentage of total light extinction (including Rayleigh) contributed by sulfate, nitrate, organics, elemental carbon, and coarse particles for the central Rocky Mountain, northern Rocky Mountain, and northern Great Plains regions (March 1988 through February 1995).

behind at 25.9%. Organics are also the largest contributor to annual aerosol extinction (29%), followed by sulfates (24.4%), absorption (22.6%), coarse mass (16.6%), and nitrate (7.3%).

Northern Rocky Mountains

This region is represented by one site close to the U.S./Canada border, GLAC. Aerosol monitoring and optical monitoring was conducted at the site. The annual average total reconstructed light extinction for the March 1988 through February 1995 period is 57.1 Mm^{-1}, of which 82% is due to aerosol extinction. A modest seasonality of total extinction occurs, ranging between 66.9 Mm^{-1} in winter to 48.7 Mm^{-1} during spring. The seasonal variance is driven primarily by sulfate and nitrate extinction. Sulfate and nitrate extinctions peak during winter at 21.2 Mm^{-1} and 12.4 Mm^{-1}, respectively, and drop to 11.1 Mm^{-1} and 1.8 Mm^{-1} in summer. The largest contributor to annual aerosol extinction is sulfate (32.5%), followed by organics (28%), and absorption (elemental carbon, 19.7%).

Northern Great Plains

This region is represented by only one Class I area in South Dakota, BADL. Aerosol and optical monitoring were both conducted at this site. The annual average total reconstructed light extinction averaged 46.0 Mm^{-1} for the March 1988 through February 1995 period. Unlike most other regions, extinction was highest in spring and lowest in autumn. Seasonal variance is driven primarily by sulfate and nitrate extinction. Sulfate extinction peaks at 18.1 Mm^{-1} in spring and drops to 10.6 Mm^{-1} in autumn. Nitrate extinction in spring, at 5.6 Mm^{-1}, is more than four times its summer extinction of 1.4 Mm^{-1}. The maximum nitrate extinction of 8.7 Mm^{-1} occurs in winter. The main contributor to annual extinction is sulfate, which accounts for 40.5% of the aerosol extinction. The next highest contributor is organics (18.3%), followed closely by absorption (elemental carbon 17.5%), nitrate (12.2%), and coarse mass (11.4%).

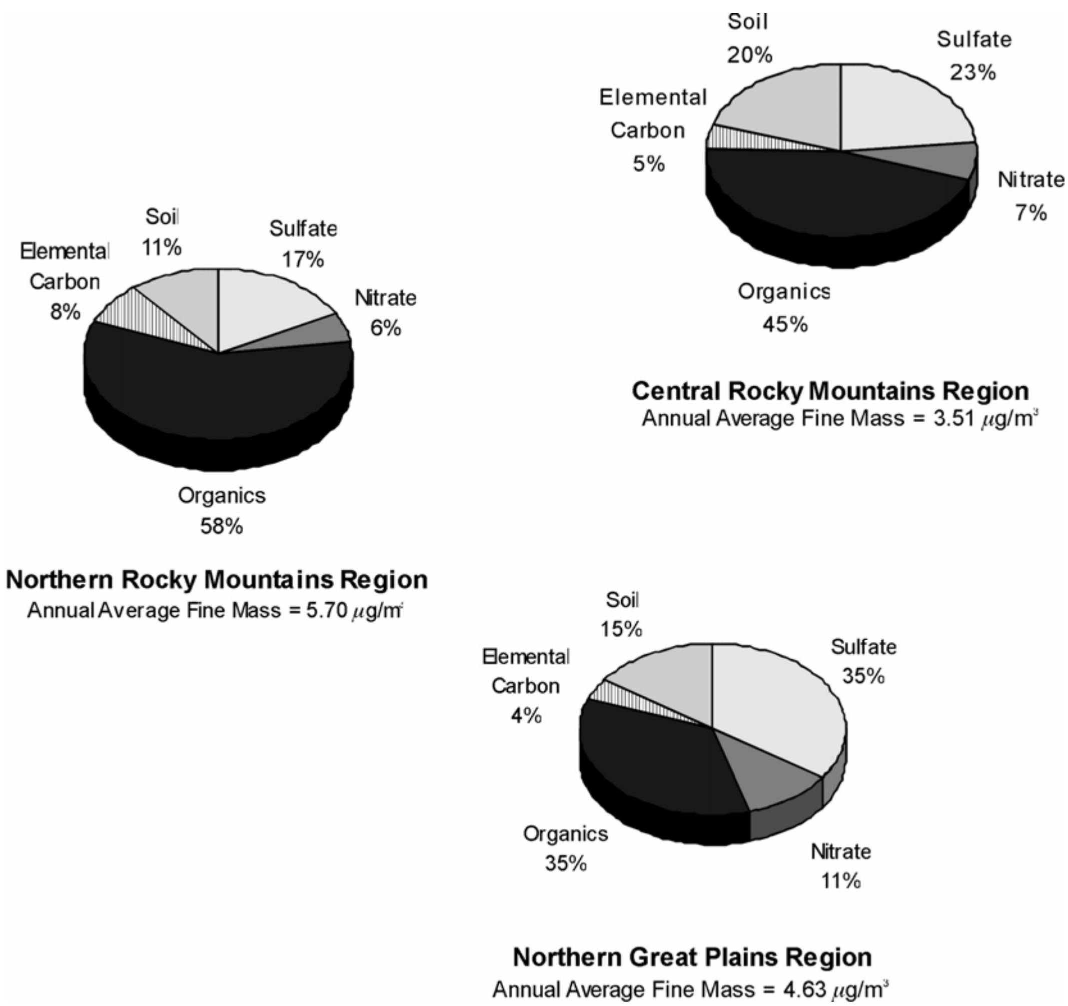
Table I-10 shows the mass concentrations ($\mu g/m^3$) of fine and coarse aerosol, and the chemical composition (mass budgets) of fine aerosol for the Central Rocky Mountains, Northern Rocky Mountains, and Northern Great Plains regions. It should be noted that these data are from one or two sites in each region and may not be representative of the entire regional area.

Figure I-4 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 1995 period.

Table I-10. Measured fine and coarse aerosol mass concentrations (in $\mu g/m^3$) for the central Rocky Mountains, northern Rocky Mountains, and northern Great Plains (March 1988 through February 1995).

Season	Fine Mass	Sulfate	Nitrate	Organics	Elemental Carbon	Soil	Coarse Mass
Central Rocky Mountains							
Spring	3.58	0.95	0.35	1.27	0.13	0.87	3.57
Summer	4.78	0.97	0.20	2.27	0.25	1.10	8.04
Autumn	3.36	0.77	0.21	1.60	0.18	0.60	4.39
Winter	2.31	0.58	0.27	1.13	0.13	0.21	2.86
Annual	3.51	0.82	0.26	1.57	0.17	0.70	4.83
Northern Rocky Mountains							
Spring	4.71	1.06	0.23	2.50	0.33	0.59	4.96
Summer	5.40	0.94	0.15	3.07	0.34	0.91	9.14
Autumn	7.30	1.97	0.32	4.67	0.63	0.71	7.27
Winter	5.50	1.03	0.60	3.07	0.53	0.26	3.26
Annual	5.70	1.00	0.32	3.31	0.45	0.62	6.21
Northern Great Plains							
Spring	4.99	1.94	0.60	1.47	0.17	0.81	5.50
Summer	5.21	1.80	0.16	2.18	0.19	0.87	7.55
Autumn	4.19	1.27	0.33	1.66	0.18	0.74	5.80
Winter	4.10	1.42	0.86	1.25	0.18	0.38	3.85
Annual	4.63	1.61	0.49	1.64	0.18	0.70	5.69

Figure I-4. Annual average measured fine aerosol mass budgets (in percent) for the central Rocky Mountains, northern Rocky Mountains, and northern Great Plains regions (March 1988 through February 1995).



e. <u>Visibility Special Studies for the Central Rocky Mountains Region</u>

In addition to the IMPROVE network, the Central Rocky Mountains region has been the focus of two studies that examined visibility, haze, and the sources of pollutants responsible for visibility impairment. These studies are:

- The Mount Zirkel Wilderness Area Reasonable Attribution Study of Visibility Impairment (1996 - 97), was initiated in response to the July 1993 U.S. Forest Service Certification of

Impairment in the Mount Zirkel Wilderness Area (west of ROMO). The primary goals of the study were to: 1) determine the extent of visibility impairment within the Mount Zirkel Wilderness Area, 2) build a better information base upon which the State could make a reasonable attribution decision regarding visibility impairment at Mount Zirkel Wilderness Area due to the Craig or Hayden power stations, and 3) determine the relative contribution of emissions from each source or group of sources to visibility impairment. Conclusions of the study stated: "Visibility (light extinction) in the Mount Zirkel Wilderness Area is among the best measured in U.S. Class I areas. The Craig and Hayden power plants in northwestern Colorado contribute to air pollution over the Mount Zirkel Wilderness, however, regional pollution sources (i.e., automobiles, coal and oil fired power plants, smelters, and wildfire emissions) appear to be the major contributor to the area's haze problem."

- The Northern Front Range Air Quality Study (1996 - 97, and previous Colorado Front Range studies, 1986-87, 1984, 1978, etc.), whose goals are, and have been, to: 1) determine the sources of existing air pollution in the Denver urban region, 2) estimate particle emissions, particulate precursors, and other substances, and 3) collect data necessary to support informed decision-making concerning the attainment and maintenance of federal National Ambient Air Quality Standards for airborne particles and state visibility goals. Data analysis for the Northern Front Range Air Quality Study will occur during 1997 and will be available in the future. No special visibility studies have been conducted in the Northern Rocky Mountains or Northern Great Plains.

f. Interpretation and Visibility Projections

Data from other IMPROVE visibility sites around the country have been presented graphically (Figures I-1 and I-2) so that visual air quality in the Central and Northern Rocky Mountains and Northern Great Plains regions can be understood in comparison with other U.S. regions. Long-term trends fall into three categories: increases, decreases, and variable. Using the visibility sites summarized for this report, the majority of data show no apparent temporal trend. Long term increases or decreases in visibility conditions may have occurred over the past 10, 20 or 50 years, but the data are sparse and, therefore, may not provide for an accurate trend analysis. Continued monitoring is necessary to identify local source impacts. Additional data and in-depth modeling and analysis are required to 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.

Other long-term future impacts may include: increased organics and elemental carbon concentrations due to increased gas and mineral developments in the Central and Northern Rocky Mountains regions, and higher concentrations of organics in the Central Rocky Mountains due to increased population growth along the Colorado Front Range.

II. REGIONAL CHARACTERISTICS

A. ENVIRONMENTAL SETTING

1. Central and Northern Rocky Mountains

The Rocky Mountain region from northern Colorado to northern Montana encompasses a wide variety of landscapes and ecosystems. Geology, soils, aquatic systems, vegetation, and fauna are highly variable at both large and small spatial scales due to the complex mountainous topography of this region. The Rocky Mountains are rugged glaciated mountains with many peaks up to 4,500 m in elevation. Mountainous topography is generally highly dissected with intervening valleys and plateaus. Geology is spectacularly varied with a great diversity of igneous, metamorphic, and sedimentary bedrock of various ages. Glacial till is found in many locations as a result of various glacial advances during the Pleistocene. The presence of glaciers in many high mountain valleys and cirques attests to the geomorphically dynamic landscapes of the Rockies.

Soils in the Rocky Mountains are diverse with respect to topography, parent material, vegetation, climate, and time of formation. Many different soil orders are found, with inceptisols, entisols, alfisols, spodosols, aridisols and mollisols being most common. Because of the influence of gravity on steep slopes, colluvium is the most common surficial component of soils at most locations in mountains. Alluvium is also common in river valleys. It is difficult to generalize about the nutrient status and biogeochemical cycling properties of soils in the Rockies. These factors, in conjunction with analyses of potential impacts of air pollutants, should generally be assessed on a watershed basis.

The climate of the southern and central Rockies is considered to be a semiarid steppe regime in which there is considerable variation in precipitation with altitude. Total precipitation is moderate but greater than in the plains regions to the west and east. Foothill regions annually receive only 25 to 50 cm of rainfall, while higher elevations may receive as much as 100 cm. In the higher mountains, a major portion of annual precipitation is snow. Climate is strongly affected by prevailing winds, resulting in generally wetter western slopes and drier eastern slopes. Average annual temperatures range from 2 to 7ºC, with higher temperatures in lower valleys (Bailey 1980).

The climate of the northern Rockies is considerably colder; the average temperature of the coldest month is often lower than 0ºC and the warmest month is lower than 22ºC. Annual precipitation ranges from 50 to 100 cm, with much of it falling as snow. Summers generally are dry because prevailing westerly winds during this season transport relatively dry air masses from the Pacific Northwest (Bailey 1980).

The vegetation of the Rocky Mountains is dominated by coniferous forest but overall is quite diverse: each of the national parks in the Rocky Mountains contains over 1,000 vascular plant species and a wide variety of lichens, bryophytes, and fungi. The abundance and distribution of plant species vary at different spatial scales. The most obvious pattern of variation is associated with elevation, as mediated by differences in precipitation, temperature, and soils.

Some tree species are found throughout the Rocky Mountains. For example, ponderosa pine (*Pinus ponderosa*) is common on drier, low-elevation sites from ROMO to GLAC. Douglas-fir (*Pseudotsuga menziesii*) is common at lower elevations and is often mixed with other coniferous species. Subalpine fir (*Abies lasiocarpa*) and Engelmann spruce (*Picea engelmannii*) are common species in subalpine forest ecosystems. Lodgepole pine (*Pinus contorta*) is often associated with fire disturbance. Quaking aspen (*Populus tremuloides*) is also associated with disturbance (fire, rockslides, and avalanches) as well as meadow margins, while other *Populus* species typically occur in riparian zones. A wide variety of grasses, forbs, and shrubs are found above treeline and in lower-elevation meadows. There are also many areas of bedrock and talus where little or no vegetation occurs.

With the exception of high elevation areas and most of the national park lands, most of the forests in the Rocky Mountains have been subjected to logging over the past 100 years. The majority of low elevation forests, particularly ponderosa pine forests, had been logged by the 1920s. Nearly all grasslands, including lower-elevation prairies and higher-elevation meadows have been grazed by domestic livestock; this includes many national park lands prior to their establishment as parks. The effect of these land use practices can still be seen throughout Rocky Mountain landscapes and provides a context for considering other potential environmental changes in the future.

Fire is an important disturbance at various spatial and temporal scales in all Rocky Mountain ecosystems. The impacts of fire are especially prominent at YELL, where fires of various intensity burned 500,000 hectares in 1988. However, the natural fire frequency is generally lower in some habitats than it was during the past century, primarily due to fire exclusion and altered ignition sources. As a result, some forest stands may have higher stem densities than they had a century ago. Similarly, some grasslands may have increased densities of shrubs and different combinations of grass and forb species (also affected by grazing). This may or may not reflect an “unnatural” condition in some situations but should be considered when assessing the vigor and productivity of ecosystems, particularly with respect to other stressors.

2. Northern Great Plains

The Great Plains region of western North Dakota and South Dakota is characterized by rolling plains and tablelands of moderate relief. They fall within a broad region that slopes gradually eastward down from the foothills of the Rocky Mountains. The highest elevations are found in the Black Hills of western South Dakota, with Harney Peak being the highest point (2,200 m).

This region is geomorphically diverse with relatively old geological formations free of glacial influences. Sedimentary rocks of a wide variety of ages dominate, although metamorphic and igneous rocks are also found in abundance. The Black Hills (where WICA is located) contain all three rock types, and the geologic map of this area contains 21 distinct mapping units (Darton 1951). Badlands of North Dakota in the vicinity of THRO and of South Dakota in the vicinity of BADL are comprised of a range of sedimentary materials including surficial clays that are eroded by active stream channels and ephemeral streams.

Soils of the northern Great Plains are diverse in terms of depth and amount of development. Inceptisols, entisols, and alfisols are found in the Black Hills, while mollisols are common in areas dominated by grasses. Entisols are also common in areas dominated by alluvium. A significant amount of badland landscape consists of areas generally not classified as soil because of lack of profile development and vegetation. Soils in grassland and alluvial areas generally have high base cation content. Calcification and salinization occur in some soils.

The climate is semiarid continental in which maximum precipitation occurs in summer, although total moisture supply is low and evaporation generally exceeds precipitation. Average annual temperature is 8ºC in much of the region. Winters are cold and dry, and summers are warm to hot. Annual precipitation is 35 to 45 cm at lower elevation plains locations and approximately 75 cm in the Black Hills (Froiland and Weedon 1990). With prevailing winds from the west, air masses reaching the northern Great Plains tend to have little moisture remaining after passing over the Rocky Mountains. The Black Hills further reduce precipitation to the east.

One of the most common tree species in the southern part of this region is ponderosa pine. This is the dominant tree in much of the Black Hills. It is also the most ozone-sensitive conifer in western North America. White spruce (*Picea glauca*) is also widely distributed at higher elevations and in cooler drainages of the Black Hills. Rocky Mountain juniper (*Juniperus scopulorum*) is the most widespread conifer at lower elevations in the Black Hills and elsewhere in the western Dakotas. Aspen is found on disturbed sites associated with coniferous forest and in riparian habitats. A variety of hardwoods are found in riparian areas and drainages, including plains cottonwood (*Populus deltoides*), bur oak (*Quercus macrocarpa*), American elm (*Ulmus americana*), green ash (*Fraxinus pennsylvanica*), box elder (*Acer negundo*), and willows (*Salix* spp.).

Much of the lower elevation of the northern Great Plains is covered by mixed-grass prairie or shortgrass prairie. These grasslands are comprised of a diversity of grass, shrub, and forb species, with tallgrass and bunchgrass species generally more common on microsites with higher soil moisture. The dominant grassland species include wheatgrass (*Agropyron* spp.), needlegrass (*Stipa* spp.), grama (*Bouteloua* spp.), prairie junegrass (*Koeleria pyramidata*), and buffalo grass (*Buchloe dactyloides*). Various species of sagebrush (*Artemisia* spp.) and currant (*Ribes* spp.) are the most common shrub species mixed with different grassland types.

Nearly all of the northern Great Plains has been greatly disturbed by human activities. Forested areas - primarily the Black Hills - were extensively logged during the past century. Grasslands have been extensively grazed, including on national park lands until they were given protected status. Fire is an important source of ecological disturbance throughout the northern Great Plains, although fire frequency is now lower in some habitats than in the past few centuries due to fire exclusion and altered ignition sources. In the absence of fire, some forest stands have relatively high stem densities, particularly in the understory. Some grasslands may have previously burned every few years due to human- and lightning-caused ignitions. Fire exclusion and reduced human ignitions have probably encouraged a reduction in fire-adapted bunchgrass species and an increase in shrub species at some sites. In savanna areas (e.g., at WICA), ponderosa pine can encroach on prairie in the absence of fire. The past history of land use and disturbance must be considered when assessing the potential impacts of air pollutants and other stressors.

B. REGIONAL AIR QUALITY

The Rocky Mountain and northern Great Plains states are sparsely populated compared to Eastern states. Wyoming is the least populated state in the nation, and by the year 2000 the population is expected to reach only 522,000 inhabitants (Table II-1). The Dakotas and Montana each have fewer than a million people and expect moderate increases in population over the next 30 years. There are few large urban areas in Wyoming, Montana, and the Dakotas, and none qualify as metropolitan areas (larger than one million people). Colorado is the most populated of the Rocky Mountain states with a metropolitan area (Denver) and several smaller urban areas, Boulder, Colorado Springs and Fort Collins. The Front Range area of Colorado is currently experiencing a growth boom with urban and suburban development expanding at a rapid pace.

Utah lies to the west of the Rocky Mountains and is experiencing rapid growth in the Salt Lake City/Ogden region. Urban development is the largest source of VOC and NO_x emissions and these pollutants can be transported several hundred kilometers. Consequently air quality in southwestern Wyoming and in the vicinity of GRTE and YELL are potentially impacted by emissions from Salt Lake City; continued population growth could exacerbate these impacts.

Population growth in Idaho is expected to remain moderate (Table II-1). Air quality in Idaho is threatened more by growing industrial emissions than from urban development. Power plants, oil refineries and chemical plants are major sources of pollutant emissions in Idaho.

Most of the population growth in the Rocky Mountain and northern Great Plains regions is occurring near urban centers. While most of the national parks included in this report are remote from urban areas (ROMO is an exception; Figure II-1), regional transport of pollutants from urban areas to wildland areas may pose a threat to the air quality of the parks.

Figure II-1. National parks and major cities of the Rocky Mountain and northern Great Plains regions.

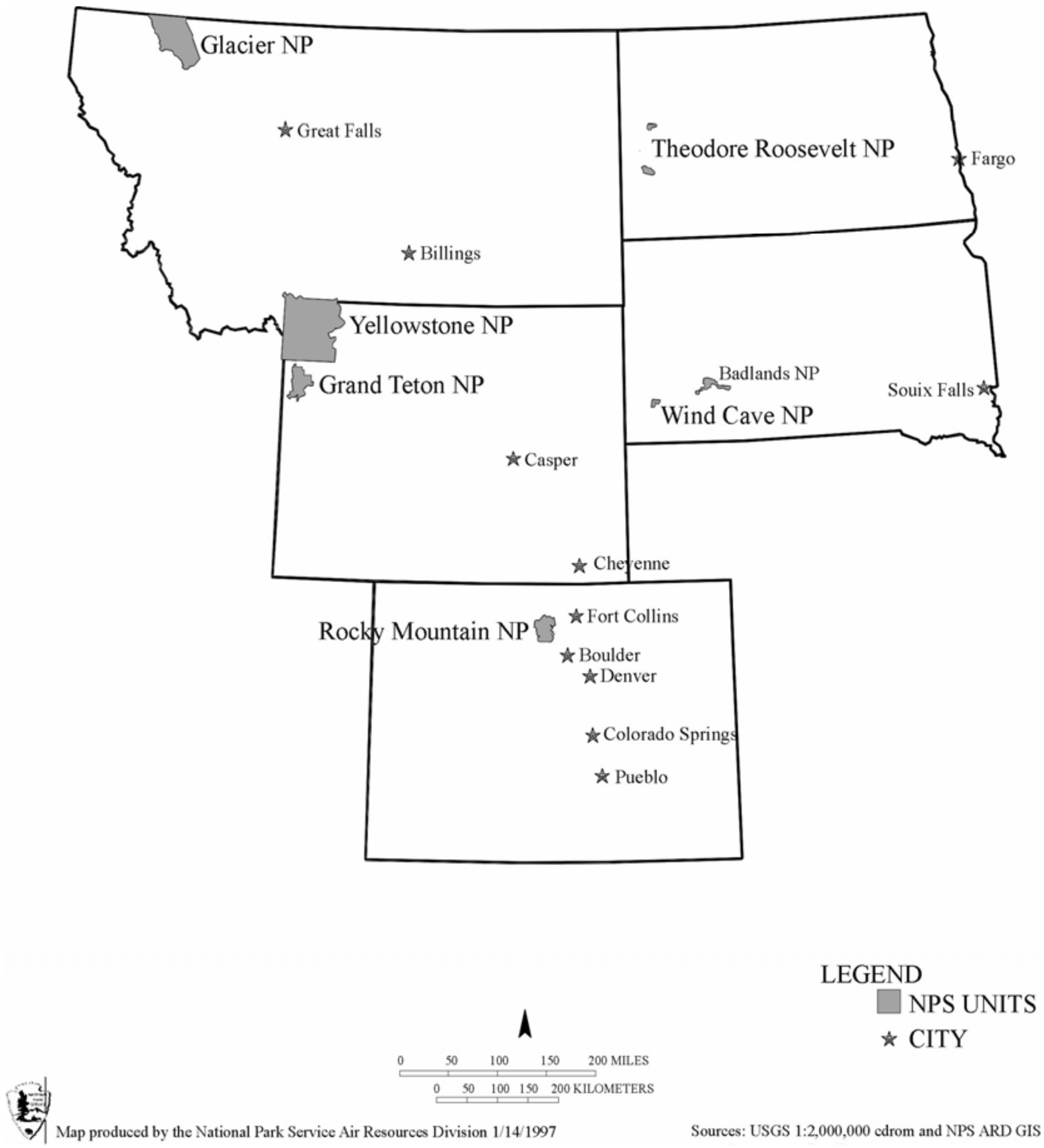


Table II-1. Projected population growth in Rocky Mountain and Northern Great Plains states. (Source: U.S. Department of Commerce 1990)			
	1990	2000	2020
Colorado	3,294,394	4,059,000	4,871,000
Montana	799,065	920,000	1,071,000
N. Dakota	638,800	643,000	719,000
S. Dakota	696,000	770,000	863,000
Idaho	1,012,000	1,056,00	1,097,00
Utah	1,722,850	2,148,00	2,749,00
Wyoming	453,588	522,000	658,000

Air quality data are summarized in this section on a state-by-state basis, because state agencies are responsible for administering air quality programs. A regional assessment of deposition and air quality can be found at the end of this section and park-specific summaries can be found within each park's section.

Gaseous pollutant monitoring and meteorological sites for each state in the Rocky Mountains and northern Great Plains are illustrated in US EPA's AIRS (Atmospheric Information Retrieval Systems) maps (Figures II-2 to II-6). Monitoring stations for SO_2, NO_2, and ozone are mostly located in or near urban areas, and consequently there is limited information on pollutant distribution to wilderness and rural areas. The NPS and USDA Forest Service maintain and operate monitoring sites in some wildland areas. Meteorological sites are scattered across most states, however they are generally placed near communities or towns. Wyoming has most of its meteorological sites in the eastern portion of the state where the majority of the population resides.

Categories of SO_2, NO_x and VOC emissions for the Rocky Mountain and Northern Great Plains states are listed in Tables II-2, II-3 and II-4. Annual emissions for Utah and Idaho are included because air quality in western portions of Wyoming and Montana are probably influenced by emissions from these states. Colorado and Utah have the highest total NO_x emission levels, mainly from fossil fuel combustion by electric power utilities and on-road vehicles. Colorado has the highest VOC levels, mainly from vehicles and industrial solvents. Areas in Colorado downwind of NO_x and VOC sources may be at risk for ozone pollution. Colorado, Wyoming and North Dakota all have annual SO_2 emissions exceeding 100,000 tons/year. In these states, electric utilities are the major sources of SO_2, followed by industrial fuel combustion (including oil and gas refining) and mining operations.

Figure II-2. Air quality monitoring and meteorological sites for Colorado. Data from U.S. EPA's Atmospheric Information Retrieval Systems (AIRS).

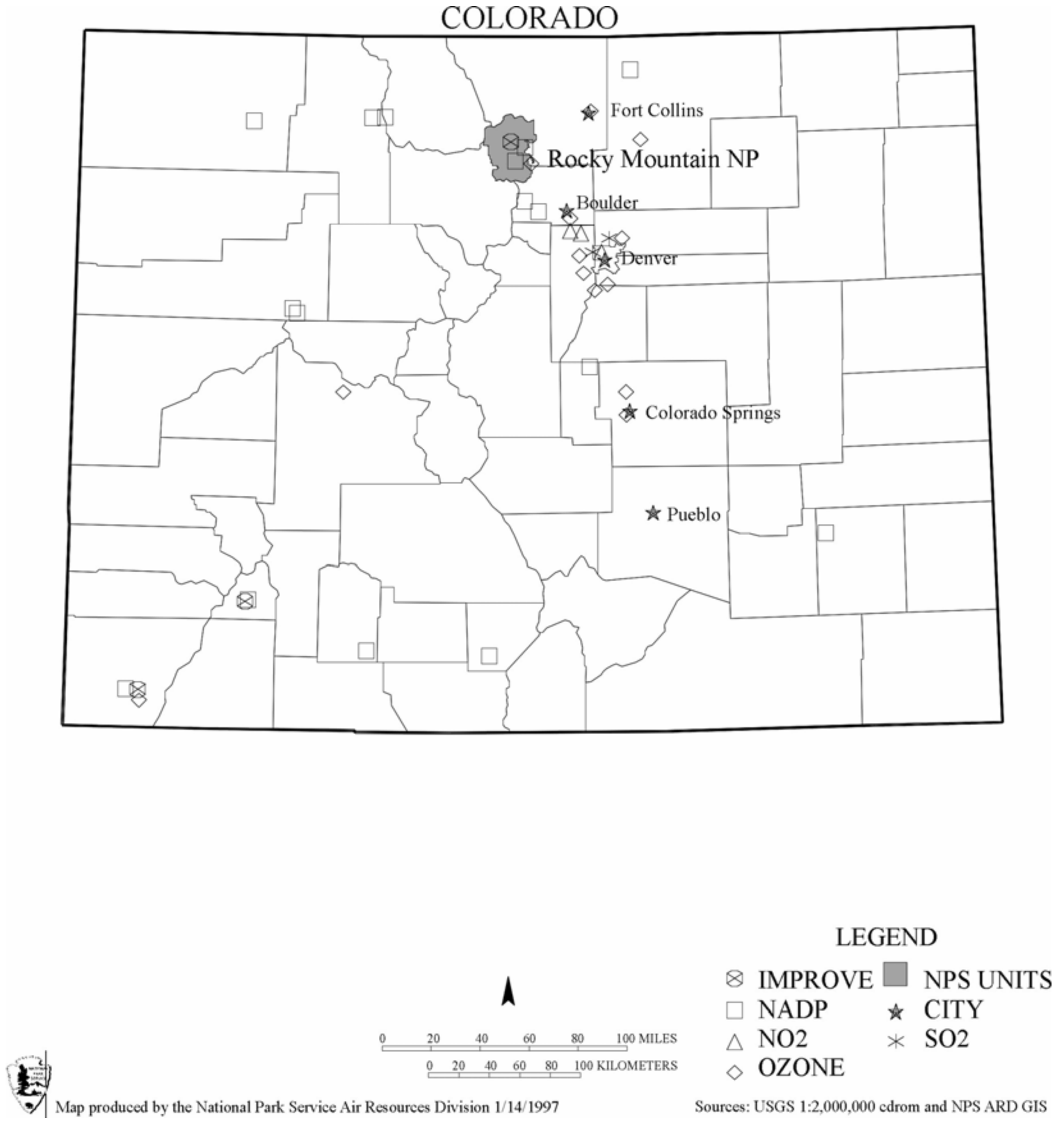


Figure II-3. Air quality monitoring and meteorological sites for Wyoming. Data from U.S. EPA's Atmospheric Information Retrieval Systems (AIRS).

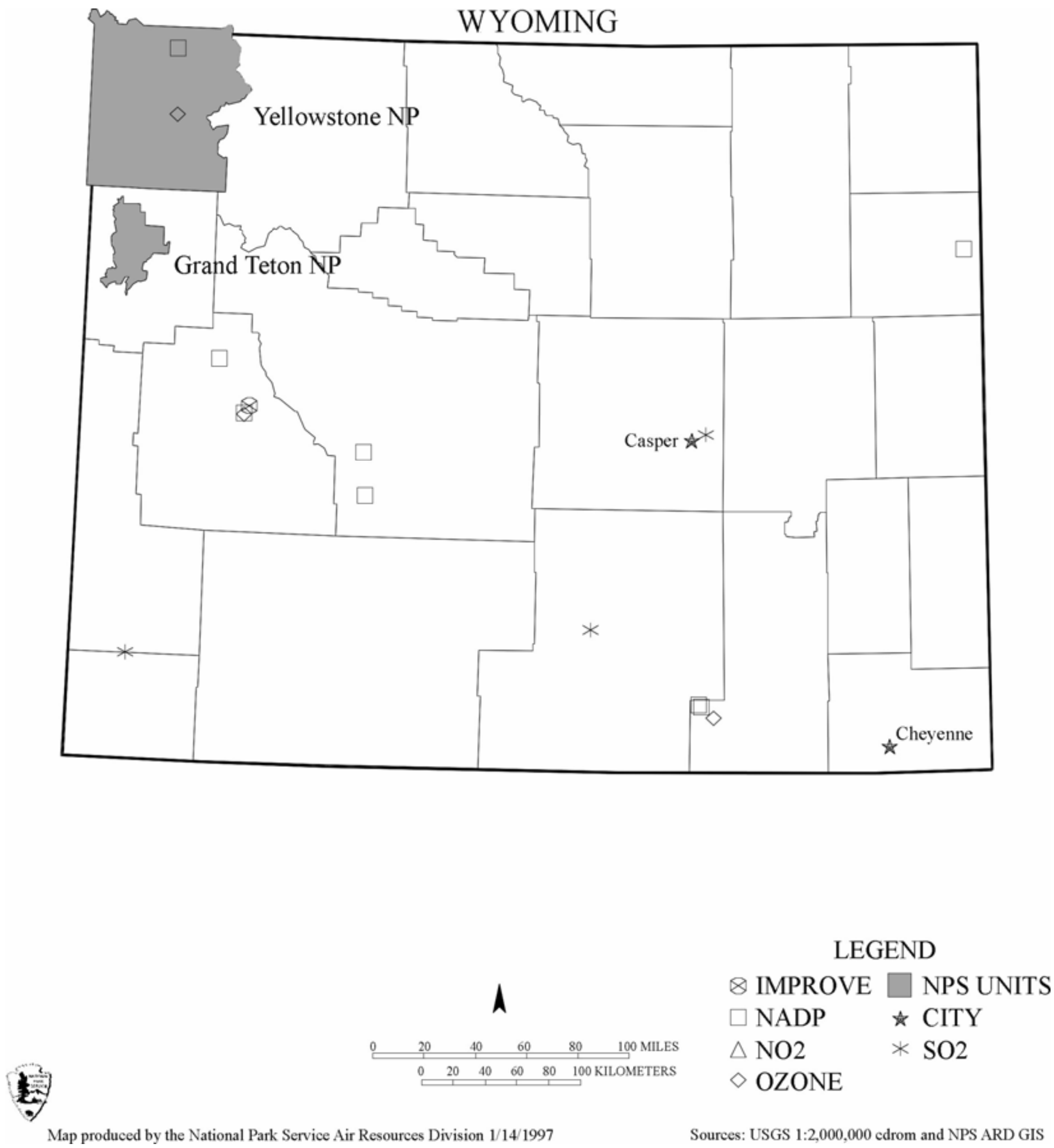


Figure II-4. Air quality monitoring and meteorological sites for Montana. Data from U.S. EPA's Atmospheric Information Retrieval Systems (AIRS).

Figure II-5. Air quality monitoring and meteorological sites for North Dakota. Data from U.S. EPA's Atmospheric Information Retrieval Systems (AIRS).

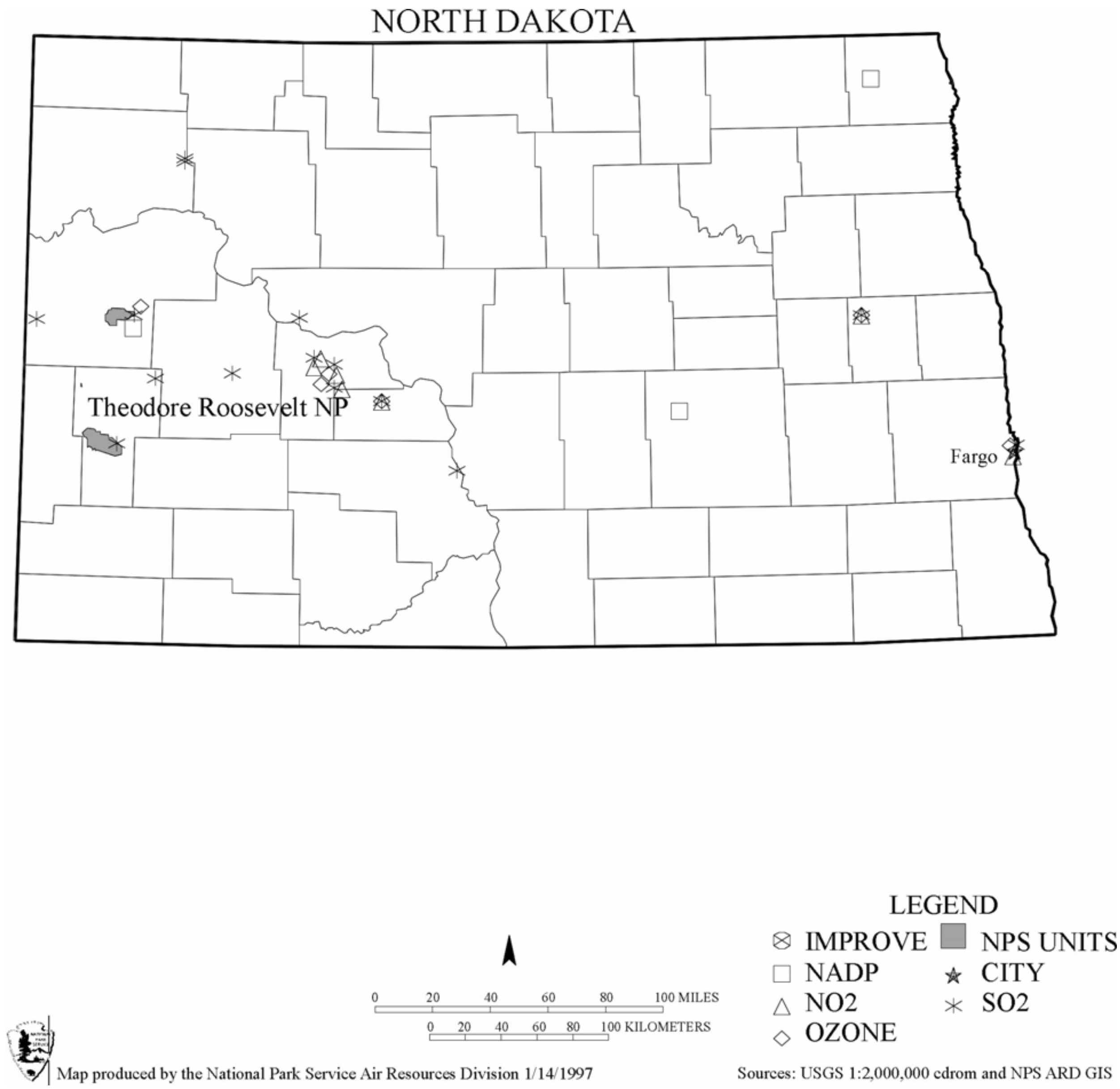


Figure II-6. Air quality monitoring and meteorological sites for South Dakota. Data from U.S. EPA's Atmospheric Information Retrieval Systems (AIRS).

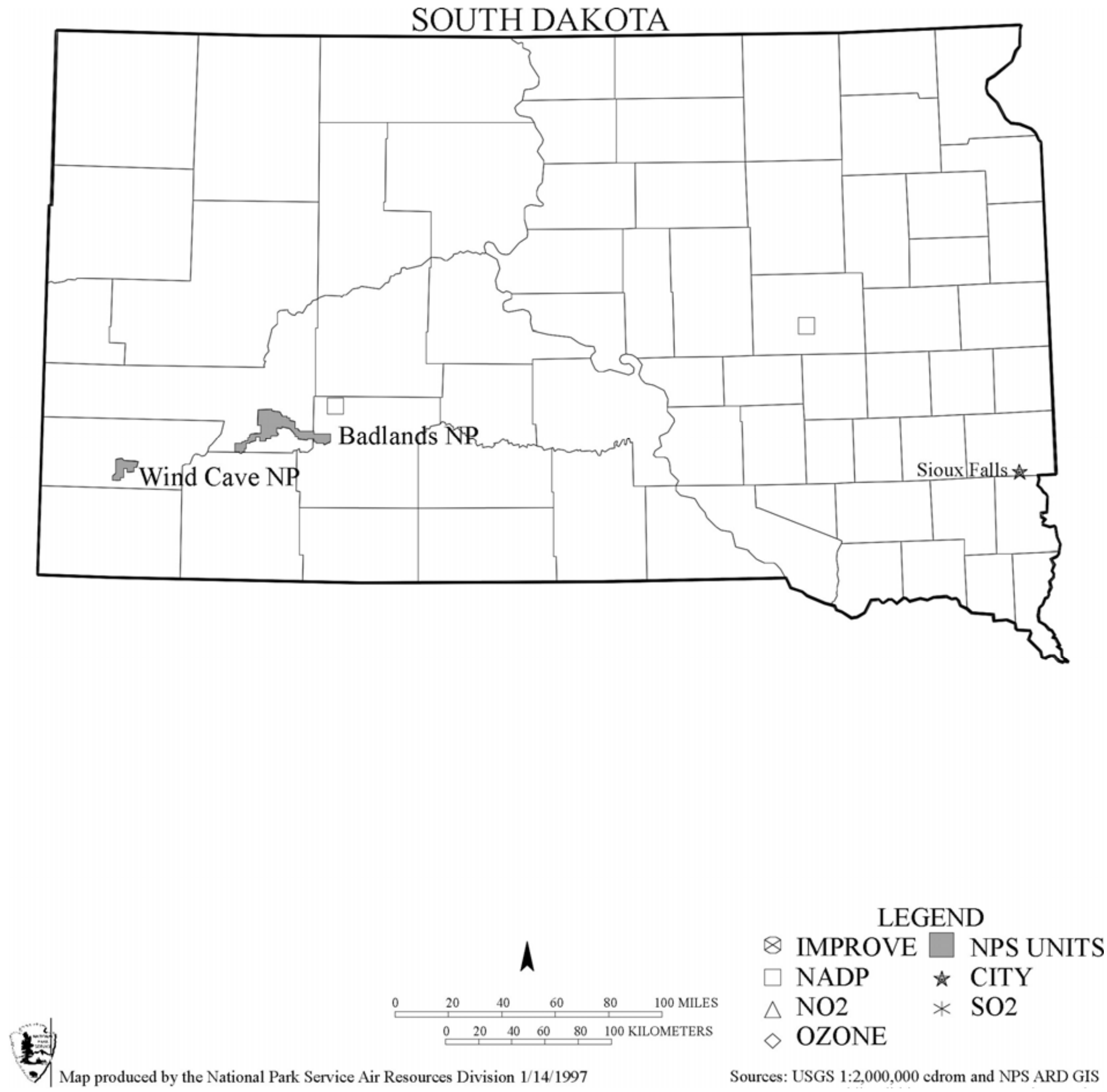


Table II-2. Annual emissions of SO_2 for Rocky Mountain and northern Great Plains states in 1994 (1000 short					
	Fuel Combustion Electric Utility	Fuel Combustion Industrial	Metals Processing	Fuel Combustion Other	Chemical Alliec
Colorado	88	6	0	3	0
Idaho	0	8	0	3	17
Montana	22	17	28	1	3
North Dakota	144	56	0	3	8
South Dakota	33	3	0	1	0
Utah	34	22	12	6	0
Wyoming	78	32	2	2	9
Total	399	144	42	19	37

[a] waste disposal, recycling, on-road vehicles, non-road sources and miscellaneous sources

Table II-3. Annual emissions of NO_x for Rocky Mountain and northern Great Plains states in 1994 (1000					
	Fuel Combustion Electric Utility	On-Road Vehicles	Fuel Combustion Industrial	Non-Road Sources	Fuel Comb Othe
Colorado	131	108	36	54	14
Idaho	0	43	12	15	2
Montana	66	34	13	29	2
North Dakota	124	24	20	14	2
South Dakota	22	30	3	6	2
Utah	105	58	30	28	6
Wyoming	173	27	67	24	2
Total	621	324	181	170	30

[a] includes industrial processes, waste disposal, recycling and miscellaneous sources

Table II-4. Annual emissions of VOCs for Rocky Mountain and northern Great Plains states in 1994 (100
1995)

	Solvent Utilization	On-Road Vehicles	Waste Recycling	Non-Road Sources	Storage and Transport
Colorado	73	94	5	35	21
Idaho	23	32	2	11	11
Montana	20	24	2	9	11
North Dakota	33	18	3	6	11
South Dakota	31	22	2	8	11
Utah	36	55	4	17	15
Wyoming	9	19	1	5	8
Total	225	264	19	91	88

[a] includes solid waste disposal, wildfires, miscellaneous sources

Emissions of SO_2 from point sources for 1996 are shown in Figure II-7. Montana, Idaho and Utah have numerous point sources with high emissions of SO_2 , but most are located greater than 200 km from a national park. Colorado, North Dakota, and South Dakota have numerous SO_2 point sources (although many are too small to be regulated) near park boundaries, posing a potential threat to park resources. SO_2 emissions may affect resources in ROMO due to the proximity of numerous sources in the Denver area and Yampa Valley west of the park. Despite its low population, Wyoming has a large number of SO_2 point sources scattered throughout the state, with several located 100 km east of YELL. Prevailing winds from the southwest may protect the park from the impact of these neighboring emissions sources. In general, YELL and GRTE are remote from upwind urban and industrial development and therefore experience excellent air quality. However, eastern Wyoming has several point sources within 100 km of the Black Hills, posing a potential threat to air quality at WICA due to prevailing wind patterns.

NO_x emissions from point sources are presented in Figure II-8. The distribution of sources is similar to that of SO_2 sources. Wyoming, Colorado, North Dakota and South Dakota have numerous sources of NO_x within 100 km of a park boundary. In Montana, Idaho and Utah, most point sources are not in proximity to a park.

1. Emission Trends and Monitoring Data

Air quality monitoring in Rocky Mountain and northern Great Plains National Parks is summarized in Table II-5. The time series of data for various air quality components varies greatly within and among parks. However, even short-term measurement periods provide a basis for estimating pollutant deposition within the parks. Interpretation of these measurements can be supplemented in some cases by data from adjacent sites (e.g., NADP data) and by monitoring data previously collected in the parks (e.g., passive ozone sampler data).

a. <u>Montana</u>

Agriculture, wood-burning stoves, coal mines, industrial activity and unpaved roads are the major sources of particulates in Montana (Martin et al. 1995). Several areas have been in non-attainment for particulate matter (PM) following the Clean Air Act amendments (1977). Efforts to bring these areas into compliance with State Implementation Plans (SIP) have been marginally successful.

SO_2 is of concern in areas near point sources. Coal-fired power plants are located in Billings and Colstrip. Petroleum refineries are located in both Billings and Great Falls. Various other SO_2 -emitting facilities are also located in Billings. Colstrip, Columbia Falls, Missoula, Great Falls and Billings are sites of major VOC point sources (>100 tons/yr). The state's largest point sources for NO_x are located in Missoula, Billings and Great Falls. Emissions from Missoula and Columbia Falls

Figure II-7. 1996 SO_2 point sources for the Rocky Mountain and northern Great Plains regions (provided by NPS-ARD).

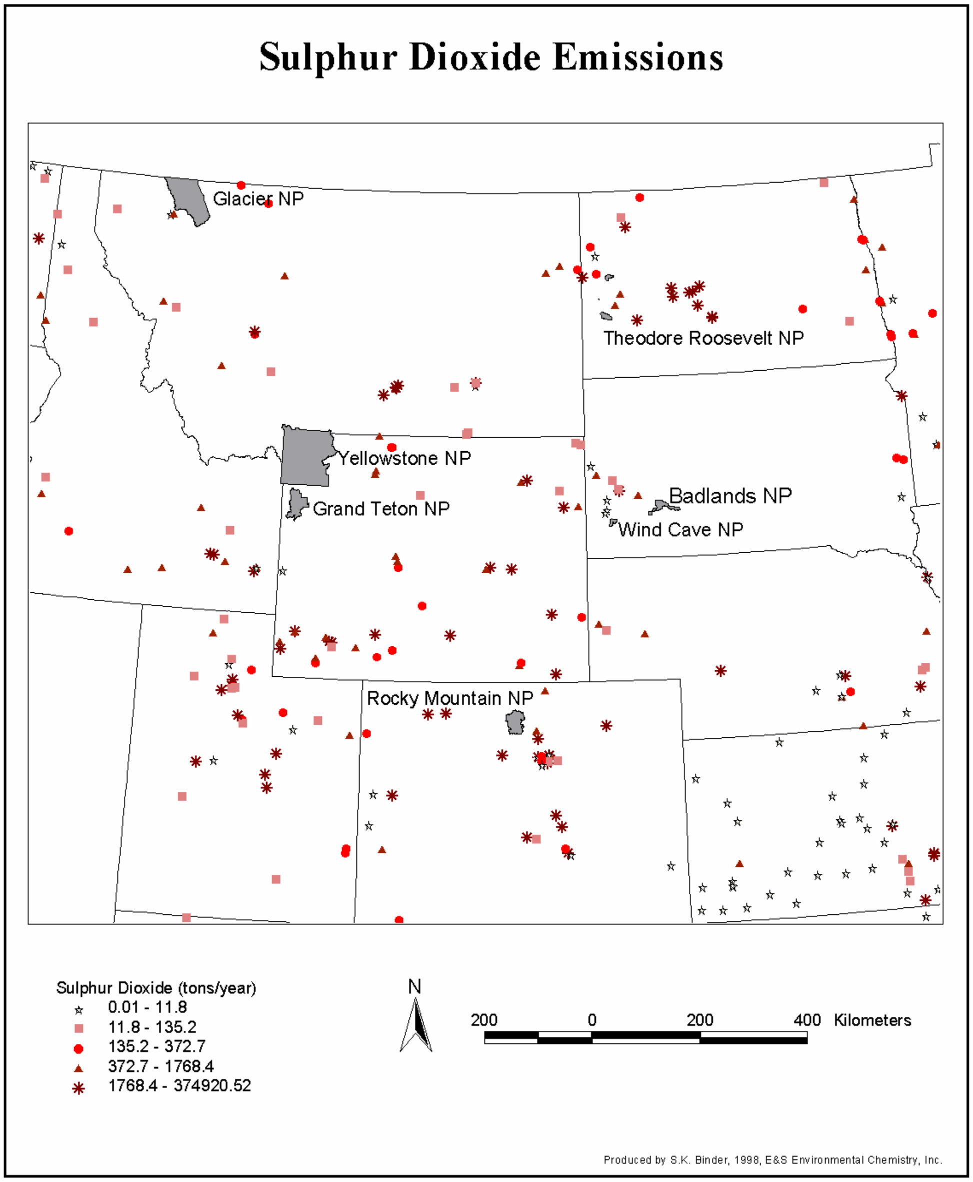


Figure II-8. 1996 NO_x point sources for the Rocky Mountain and northern Great Plains regions (provided by NPS-ARD).

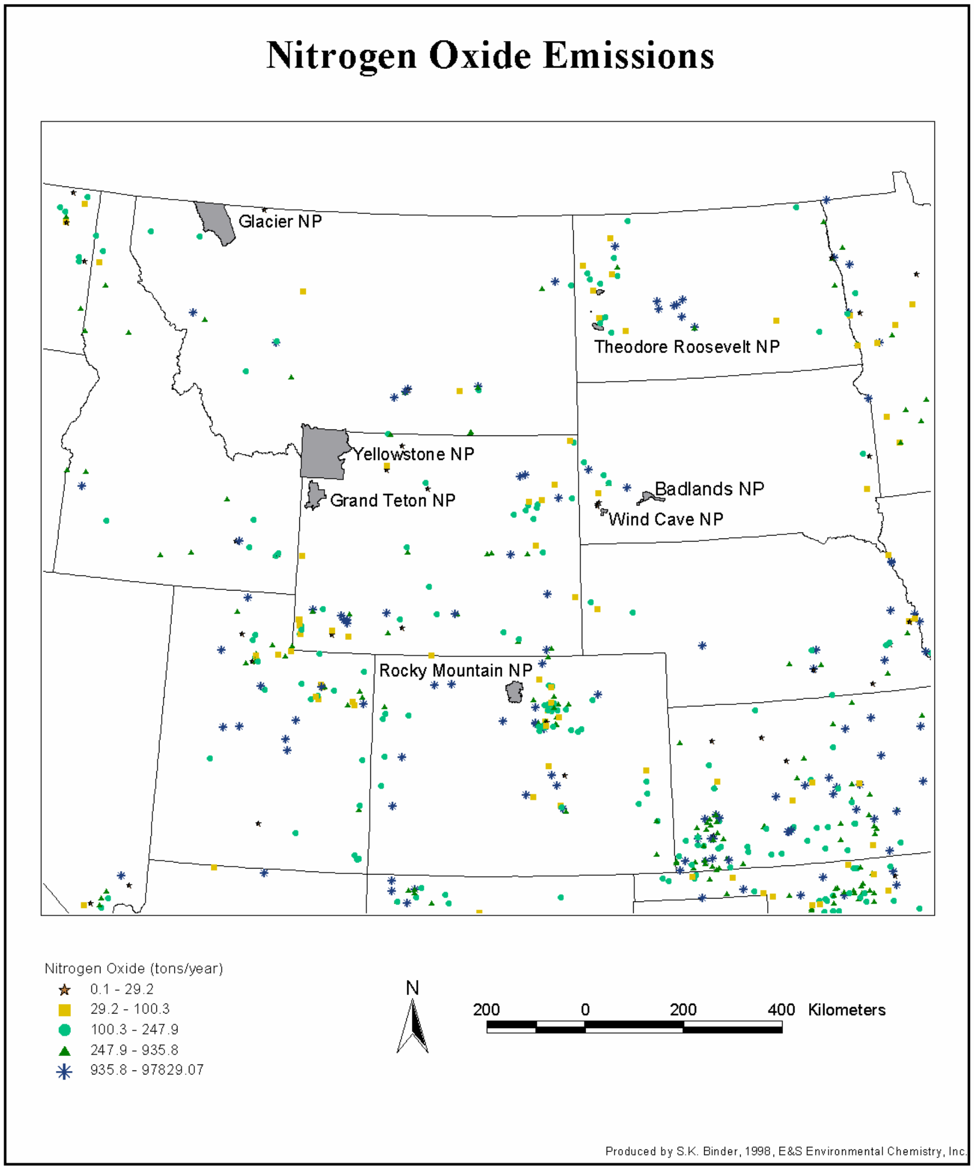


Table II-5. Current air quality monitoring in Rocky Mountain and northern Great Plains National Parks.

Park	Ozone	IMPROVE	SO_2	NADP	NDDN	Fluoride	H_2S
BADL	**	X		X[a]			
GLAC	X	X	X	X	X	X	
GRTE	**						
ROMO	X **	X	X	X	X		
THRO	X		X	X			X
WICA	**						
YELL	X	X	X	X	X		

[a] located approximately 20 km northeast of BADL
** indicates data have been collected with passive ozone samplers

may also have some impact on air quality in GLAC during periods of southerly winds (prevailing winds are from the west/southwest).

Lead pollution is a concern in East Helena, where the primary source of pollution is the ASARCO lead smelter. East Helena has been in non-attainment for lead since 1978, and the current SIP was expected to bring the area into compliance by 1997 (Martin et al. 1995). Fluoride, a byproduct of aluminum manufacturing, has been a local pollution problem for the Columbia Falls area (near GLAC) since 1957 when the Anaconda Aluminum Company began operating. Maximum emissions of fluoride reached 1,500 tons/year by 1969 but have been reduced to 180 tons/year since 1981.

b. Colorado

NO_x, VOC, SO_2, and particulates are emissions of concern in Colorado. Major sources of NO_x are fuel combustion associated with electric utilities and motor vehicles (Table II-3). VOC sources include vehicles, industry, and solvent utilization. Sulfur dioxide emissions are mainly from fuel combustion associated with electric utilities. Although emissions of NO_x, VOC, and SO_2 have declined between 1975 and 1994 and there have been no NAAQS exceedences of NO_2 in Colorado, projected population growth in the state could result in future degradation of air quality.

Ozone is a pollutant of concern in the Denver area to the east of the Rocky Mountains. The mountains restrict airflow, resulting in frequent occurrences of haze and stagnant air due to inversions caused by persistent high pressure systems in the region. During such inversions, ozone and ozone precursors build up and can result in high ozone levels at ROMO. In 1993, ambient ozone levels in ROMO, 100 km northwest of Denver, exceeded the hourly maximum NAAQS (pre-

1997 standard) on one occasion.

c. Wyoming

NO_x and VOC emissions and particulate matter from oil and gas facilities, mining, power generation, and other industrial sources are pollutants of concern in Wyoming. Electric utility and fossil fuel burning associated with industry are the dominant sources of NO_x and SO_2 (Tables II-2 and II-3) in the state. There is also some oil and gas development on Bureau of Land Management (BLM) lands in southwestern Wyoming and in the Shoshone and Bridger-Teton National Forests. Proposed new industrial sources in addition to thousands of small sources (< 250 tons/year) may result in increased NO_x and VOCs in the future.

The Wyoming Department of Environmental Quality monitored SO_2 in several locations throughout the state between 1972 and 1985, but levels were not high enough to warrant further monitoring. Nitrogen dioxide was monitored between 1975 and 1985, but also discontinued. Lead has been monitored since 1977 at three sites: Casper, Cheyenne and Rock Springs. Lead is monitored every 5 to 6 years, and the most recent data are from 1989. Lead levels in Wyoming have not exceeded the NAAQS since monitoring began. The state does not monitor ozone, and the only continuous ozone analyzer in Wyoming is operated and maintained by the NPS in YELL.

d. South Dakota

Coal burning power plants, wood stoves, unpaved roads, and grass fires are the major sources of PM-10 in South Dakota. The Department of Environment and Natural Resources currently monitors PM-10 at eight sites in three counties. The state does not currently monitor SO_2, NO_2, NO_x, or ozone, although an inventory of emissions by point sources has been compiled in an AIRS data base. South Dakota point sources of emissions of NO_x, VOC, and SO_2 are summarized in Tables II-2, II-3 and II-4. Most point sources for these pollutants are located in the eastern portion of the state.

e. North Dakota

Monitoring by the North Dakota State Department of Health indicates that air quality is generally very good in North Dakota. The population is sparse and there are no large metropolitan areas. There have been no exceedences of state standards for ozone or NO_2 since at least 1985. The major sources of pollution are associated with oil and gas production as well as coal-fired electrical generation. These sources are located in the western portion of the state, as are the Class I areas. The primary pollutants associated with these activities are SO_2 and H_2S. PSD Class I increments for SO_2 at THRO are being exceeded, although no effect on resources has been detected. Because oil and gas activities have the potential to increase in North Dakota, these increment exceedences will also likely increase unless they are mitigated. Most of the sources in the vicinity of THRO have emissions too low to be regulated, but the total of these many small sources could have significant impacts on resources that are sensitive to elevated S inputs.

f. Utah

Although Salt Lake City lies over 200 km to the southwest of Jackson Hole and the Teton Mountains, it may still affect the air quality in GRTE and YELL. Annual emissions of SO_2 and NO_x from Salt Lake City are relatively high. In 1994, emissions of SO_2 totaled 45,200 tons per year and NO_x emissions totaled 71,000 tons per year. Winds from the south and southwest may deliver these pollutants to the vicinity of the parks. However, there have been no studies on regional transport of pollutants from Salt Lake City to confirm this. The relatively high emissions from Salt Lake City are the largest component of total state emissions, although power plants at other locations (e.g., Bonanza) may produce emissions that affect GRTE and YELL. The largest component of NO_x and SO_2 emissions is from electric utilities.

g. Idaho

Idaho has no urban areas and there are currently no large point sources of NO_x, SO_2, or VOC within 200 km of GLAC, YELL or GRTE. Coal-fired power plants at the Idaho National Engineering Lab and a phosphate plant at Idaho Falls contribute some emissions to the overall state total. Industrial growth in the region is expected to expand and may result in increasing emissions that could be transported to the vicinity of the parks.

2. Regional Emission Patterns and Air Quality Issues

There is no single regional emission problem that strongly affects all national parks in the Rocky Mountains and northern Great Plains. Some parks are subject to deposition of pollutants from urban areas, some are affected by long-distance transport of pollutants from industrial facilities and electric utilities, and some are affected by local sources. Therefore, the quantity of emissions received and the potential threat to natural resources must be analyzed individually for each park. General patterns of emissions and air quality concerns are summarized briefly here and in more detail in sections on the individual parks.

There are serious concerns about air quality at ROMO because of elevated concentrations of ozone during the summer, a potential threat to vegetation, and increased levels of N deposition, a potential threat to terrestrial and aquatic systems. In addition sulfates and nitrates contribute to visibility impairment. Most of the emissions are from urban areas in the Front Range of the Rocky Mountains between Colorado Springs and Fort Collins and especially the Denver metropolitan area. Rapid population growth in this area poses an ongoing threat to air quality in the Front Range because of high VOC emissions which contribute to ozone formation, and because of high NO_x emissions and ammonium (NH_4^+) deposition which contribute to ozone formation and N deposition to terrestrial and aquatic systems. High emissions of NO_x from the Salt Lake City area of Utah may also contribute to the regional air quality that affects ROMO, although there are no data that clearly demonstrate this. Although pollution control measures are becoming increasingly strict in the Denver area, summertime ozone exposure and total N deposition will continue to be concerns at ROMO for the foreseeable future.

Emission threats to YELL and GRTE are relatively minor. Most deposition in this area is due to long-distance transport from sources to the west. Industrial and electric utility facilities in Idaho produce NO_x and VOCs that can contribute to ozone formation. The Salt Lake City, Utah, region to the southwest may also contribute ozone precursors to the YELL/GRTE region. Emissions data for Wyoming indicate that the state produces large quantities of NO_x and SO_2. Most of these emissions are from industrial and power-generation facilities to the east of YELL and GRTE; only relatively uncommon easterly winds would transport these pollutants into the parks. However, oil and gas developments on Bureau of Land Management (BLM) land in southwestern Wyoming have been proposed.

Industrial and electric-utility facilities in Idaho produce NO_x and VOCs that can contribute to ozone formation in GLAC. The Seattle-Tacoma-Vancouver area (and possibly Spokane) to the west may also contribute ozone precursors to the GLAC region, although the nature of this long-distance transport is unknown. Sources of NO_x and VOCs in Montana - most of which are to the east and south of GLAC - may contribute to degradation of the park's air quality only during periods of nonprevailing easterly and southerly winds. Emissions of fluoride from an industrial facility in Columbia Falls, Montana, have been a source of concern for many years, with potential acute and cumulative impacts on vegetation, but emissions have declined in recent years.

The northern Great Plains parks are also remote from cities and large emission sources, although there are some emissions that could potentially threaten park resources. At THRO, hundreds of small oil wells adjacent to the park emit both SO_2 and H_2S. While the emissions per well may not be large, the combined emissions of all the wells are substantial. Furthermore, some wells are very close to the park boundary where there could be acute impacts to vegetation on a local basis. Additional SO_2, NO_x, and VOCs are potentially transported from industrial and electric-utility facilities in Montana. The cumulative and acute effects of the S point sources are the greatest concern at THRO. The development of additional coal-fired power plants in eastern Wyoming would increase emissions transported to THRO.

Small quantities of emissions from the Rapid City, South Dakota, area may reach WICA, but of greater importance are regional-scale sources from eastern Montana and eastern Wyoming. Prevailing westerly winds transport NO_x, SO_2, and VOCs eastward over the Black Hills region and WICA, providing sufficient precursors for ozone formation during the summer when it is sunny and warm. The development of additional coal-fired power plants in eastern Wyoming would increase emissions transported to WICA.

Regional-scale emissions of NO_x and SO_2 from industrial and electric-utility facilities in eastern Wyoming and western South Dakota are the greatest concern at BADL. Annual emissions of NO_x in Wyoming are particularly high, and although VOCs are relatively low, they may influence the airshed of BADL sufficiently to provide precursors for ozone formation during warm, dry summer weather. The development of additional coal-fired power plants or other point sources in eastern Wyoming would be expected to increase emissions transported to BADL.

III. ROCKY MOUNTAIN NATIONAL PARK

A. GENERAL DESCRIPTION

Rocky Mountain National Park (ROMO) straddles the Continental Divide in the northern Front Range of the Colorado Rocky Mountains, and was created in 1915 to protect the scenic beauty, natural resources, historic value, and recreational opportunities of this region (ROMO 1992). ROMO is 108,200 ha in area, with approximately 93% of the park in existing or proposed wilderness. Annual park visitation has averaged 3.0 million since 1993 (ROMO 1996).

The park has an extensive boundary of 235 km, of which 61% is contiguous with national forest and 39% with private lands. Metropolitan and agricultural areas along the eastern edge of the Colorado Front Range are important source areas for atmospheric pollutants that may impact the park. The largest city is Denver, 60 km to the southeast, but other potential urban source areas include Boulder, Longmont, Loveland and Fort Collins. Additional sources include the Yampa Valley west of the park and cattle feed lots to the east in Greeley.

1. Geology and Soils

Glacial till and colluvium are common parent materials for ROMO soils. The soils are moist, loamy, or silty with low clay content. The thickness of the glacial and alluvial overburden is roughly proportional (indirectly) to the steepness of the stream gradient in the valleys (Gibson et al. 1983). Soils and overburden are relatively deep in the center of many stream valleys (i.e., 0-7 m glacial till; 3-3.7 m alluvial materials; 0-1.9 m soil) with decreasing thickness along the side slopes (Gibson et al. 1983). The watersheds having greatest sensitivity to acidic deposition are often those with little accumulated glacial till, and a high proportion of exposed bedrock.

Limited local investigations of soil type and properties have been conducted in ROMO (Herzog 1982, Walthall 1985). Walthall (1985) measured soil properties related to acid deposition in the Loch Vale watershed and other drainage systems in the park, and created a general soils map of Loch Vale (Figure III-1). The kinds of soils maps and data developed through the Natural Resources Conservation Service county soil reports are not available for the park, however. An effort is underway by the Rocky Mountain Nature Association, National Park Service, and Natural Resources Conservation Service to develop a soils basemap and database (Bachand and Petersen 1993). The final map and database will be completed in 1999.

2. Climate

The climate of ROMO is temperate montane. Mean annual precipitation is about 33 cm at lower elevations, and greater than 100 cm near the Continental Divide, most of which falls as snow. Snowpack accumulation is highly variable, with little accumulation on exposed ridges and south

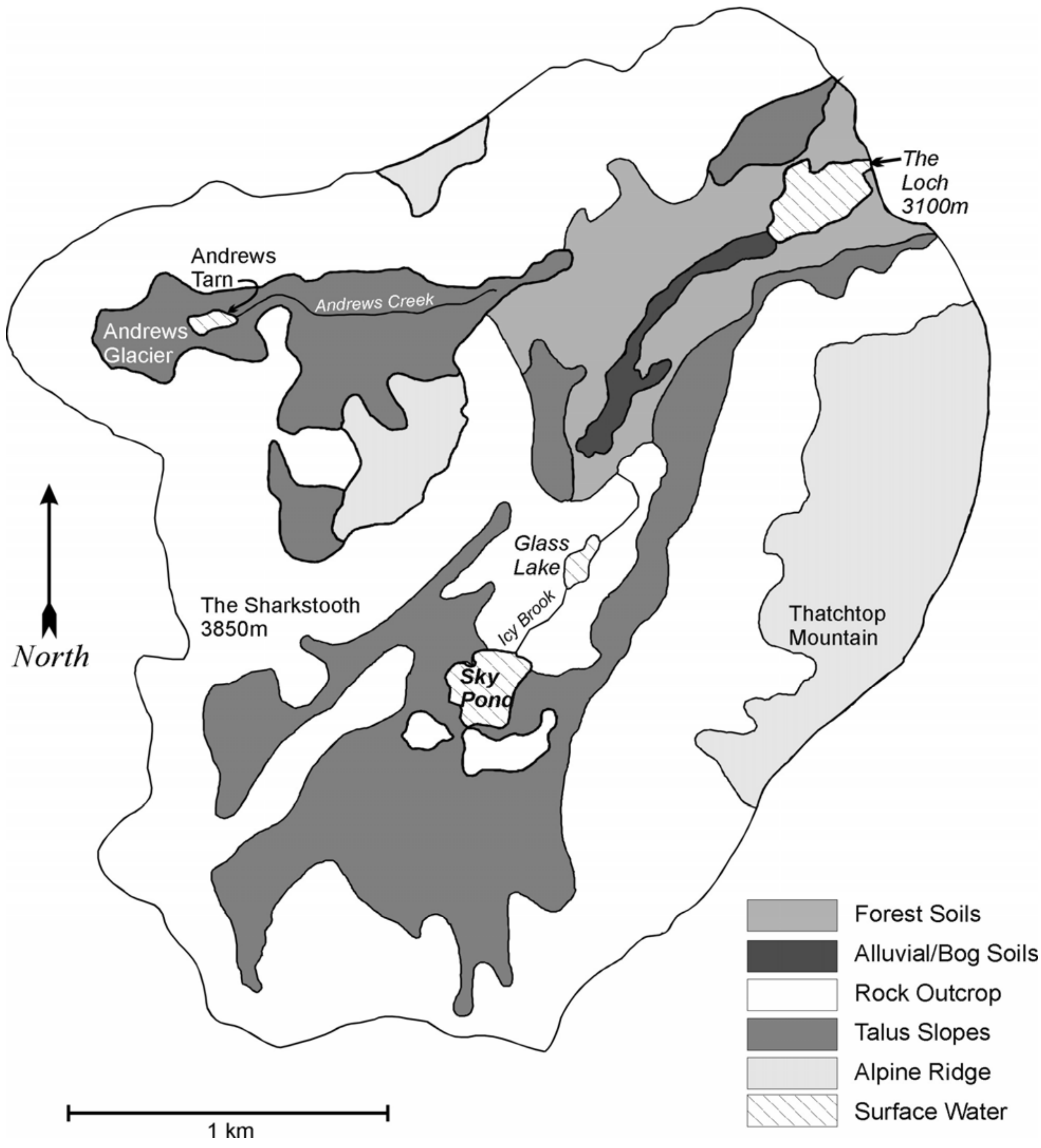


Figure III-1. Soils map of the Loch Vale watershed (from Walthall 1985).

facing slopes. Accumulation is heaviest during April and May. Depths of one to several meters at maximum accumulation are common in sheltered locations. Temperature ranges from about -5°C in winter to 25°C in summer at the lower elevations. Sub-freezing temperatures can be encountered at the higher elevations during any month.

The hydrologic cycle of high-elevation watersheds in ROMO is characterized by a lengthy period of snowpack accumulation during autumn, winter, and early spring, followed by a snowmelt period during late spring and early summer. In late summer and early fall, runoff is predominantly baseflow, with some snowmelt continuing and some stormflow from precipitation events (Campbell et al. 1995).

The predominant direction of air mass movement over the Front Range is from west to east (Barry 1973), with periodic upslope movement from the east and southeast (Kelley and Stedman 1980). Wind speeds in the park have been recorded at 320 kph. High winds generally can be found during the winter to early spring months. Wind rose data from ROMO during the period 1989 through 1995 showed a distinct pattern of predominant air movement from the northwest; there is greater variation in ROMO than suggested by data from the meteorological tower due to topographic variation. However, a second frequent wind direction was from the south and southeast, from the general direction of the Denver metropolitan area. This is important because air masses that move directly from the Denver area to ROMO have the potential to transport high levels of nitrogen, sulfur, and ozone-forming compounds to the park. The easterly upslope storm track also carries air masses across agricultural (livestock and fertilized cropland) and industrial/metropolitan areas of Colorado before reaching the vicinity of ROMO (Bowman 1992). Higher atmospheric concentration of ammonia, NO_x gases, and nitric acid particulates have been measured near the park during upslope events (Parrish et al. 1986, Langford and Fehsenfeld 1992).

3. Biota

ROMO is located in the physiographic region normally referred to as the southern Rocky Mountains. The park contains a diversity of plant communities associated with steep elevation gradients, topographic variation, and differences in soil moisture. ROMO vegetation, as classified within the park geographic information system, includes nine general vegetation types based primarily on dominant species: ponderosa pine, lodgepole pine, Douglas-fir, spruce/fir, limber pine, aspen, riparian, tundra, and meadows (Figure III-2; Emerick 1995). Ponderosa pine (*Pinus ponderosa* var. *scopulorum*) forest is common, but restricted to low elevation sites in the eastern part of ROMO, west and north of Estes Park. These forests tend to be low-density stands which are particularly common on dry, flat or south-facing sites subject to relatively frequent fire. These open stands have a variety of shrub, grass, and herbaceous plants in the understory, which provide food

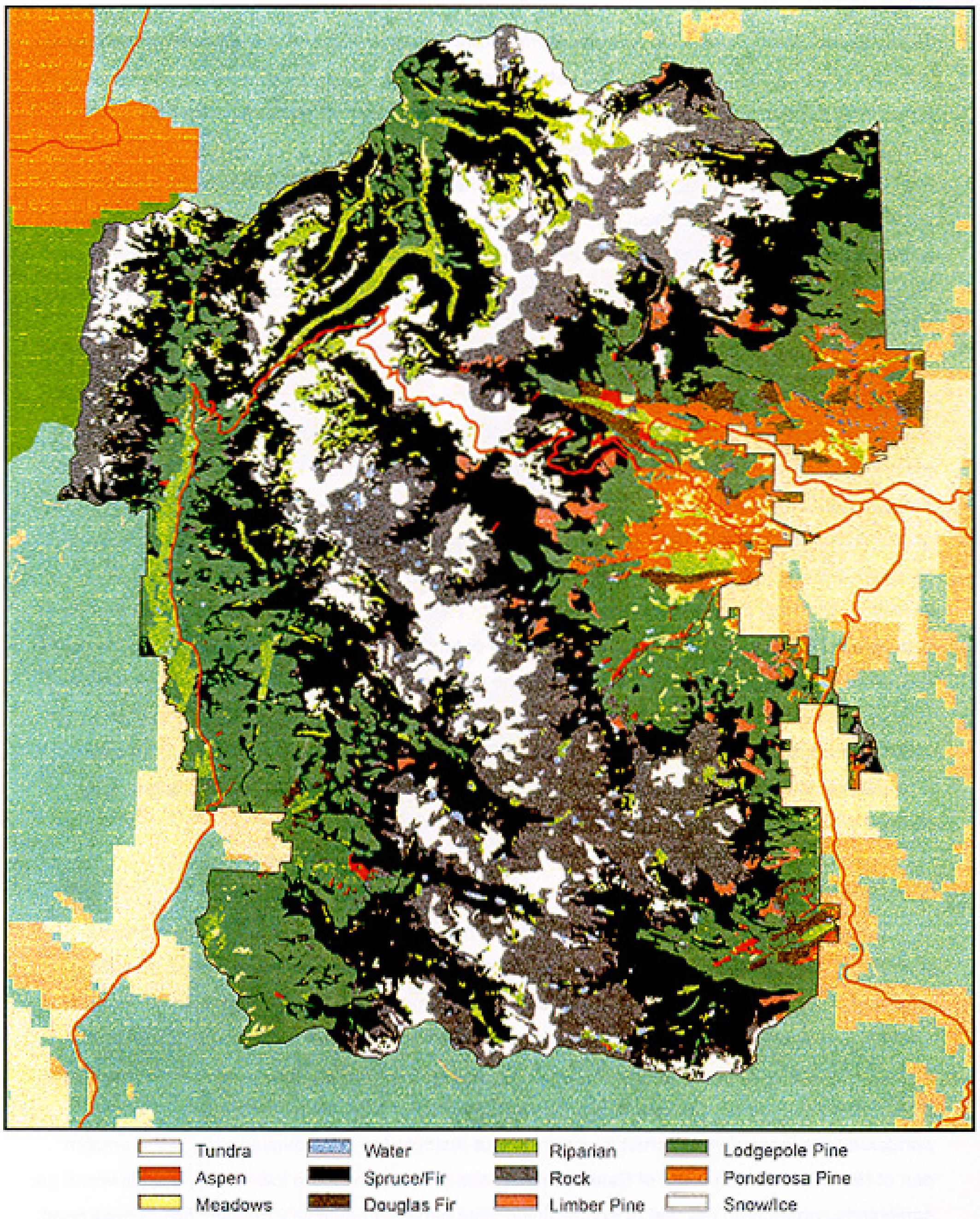


Figure III-2. Vegetation classification map of ROMO, from park database.

Figure III-2. Vegetation classification map of ROMO, from park database.

and habitat for many wildlife species. Ponderosa pine is highly sensitive to atmospheric ozone (Miller and Millecan 1971, Peterson and Arbaugh 1989), which makes it an appropriate bioindicator for this air pollutant (Peterson et al. 1993). The occurrence of ponderosa pine on the east side of the park coincides with the region of highest ozone concentrations within the park. Mountain meadows and shrublands occur in association with ponderosa pine. Big sagebrush (*Artemisia tridentata*), antelope bitterbrush (*Purshia tridentata*) and shrubby cinquefoil (*Pentaphylloides floribunda*) are particularly common shrubs mixed with a diversity of grasses and herbaceous species.

Lodgepole pine (*Pinus contorta*) is found at mid-elevations on both the east and west sides. Extensive stands of various ages grow on both sides of Kawuneeche Valley, along the upper half of the Bear Lake Road, and along the road to Wild Basin ranger station. Younger stands may be nearly exclusively lodgepole pine. The dominance of lodgepole pine indicates that there are large areas of cold, dry habitat suitable for this species, as well as the fact that stand-replacing fire has historically been a common disturbance in this region. The understory is often sparse, with species such as broom huckleberry (*Vaccinium scoparium*), common juniper (*Juniperus communis*), and kinnikinnik (*Arctostaphylos UVa-ursi*). Douglas-fir (*Pseudotsuga menziesii*) forests are found on mid-elevation, north-facing slopes up to about 3,000 m elevation. Although it may be the dominant species at a site, Douglas-fir is usually found mixed with ponderosa pine, lodgepole pine, Engelmann spruce (*Picea engelmannii*), and subalpine fir (*Abies lasiocarpa*). Stands are normally quite dense with a sparse understory. Spruce-fir forests are composed mainly of Engelmann spruce and subalpine fir in a relatively continuous band from about 2,800 m to treeline. They tend to be dense forests of tall trees often with a substantial understory of immature trees. Limber pine (*Pinus flexilis*) is common at treeline, but is also found at other locations in the park mixed with other conifer species, especially on rocky, exposed sites. Limber pine tends to occur in low density stands and is of very low stature near treeline.

Quaking aspen (*Populus tremuloides*) is locally abundant in the park below 3,000 m and is occasionally found above this elevation. Aspen is typically found near the margins of meadows and stream valleys, but is also common on moist sites disturbed by fire or avalanches. These deciduous forests normally have a more lush understory than coniferous forests. Aspen is sensitive to ozone and sulfur dioxide. Narrowleaf cottonwood (*P. angustifolia),* another *Populus* species found in ROMO, is also sensitive to ozone.

Riparian vegetation types associated with streams, ponds, and lakes comprise about 4% of ROMO vegetation. Most of these systems are dominated by shrubs such as dwarf birch (*Betula glandulosa*) and willow species (*Salix* spp.) at higher elevations and willows, alder (*Alnus* spp.), river birch (*Betula fontinalis*) and aspen at lower elevations. Narrowleaf cottonwood and Colorado blue spruce (*Picea pungens*) can also be found in a few locations in the park. Riparian systems are common throughout the park but tend to be narrowly distributed in areas of highest soil moisture.

Alpine tundra encompasses a variety of plant communities found on different soils and under different snowpack conditions. These communities include: (1) meadows and turfs, which are relatively snowfree during winter and consist of dense grasses, sedges, and herbaceous species; (2) wet meadows, which are found in depressions where meltwater accumulates, and consist of several herbaceous species and sedges; (3) fellfields, which are found in rocky, exposed sites and consist of cushion plants, succulents, and plants with mat-like morphology; (4) shrub communities, which exist where snow is deep enough to cover and protect shrubs (primarily willows); and (5) snowbank communities, which are found under late-lying snowbanks and consist of sedges, rushes, some herbaceous species, and lichens (Emerick 1995). Very little is known about the sensitivity of alpine plant species to air pollution despite the fact that alpine tundra comprises about one-third of ROMO vegetation.

The diverse fauna of ROMO includes 260 species of birds, 66 species of mammals, 6 species of amphibians, and 1 reptile species. Wildlife management is the topic of 87 project statements in the resource management plan. Wildlife management is centered on elk (*Cervus elaphus*) and bighorn sheep (*Ovis canadensis*) populations, concerns about high densities of small mammals at viewpoints due to visitor feeding, river otter (*Lutra canadensis*) populations, amphibian populations, and restoration of greenback cutthroat trout (*Oncorhynchus clarki stomais*) and peregrine falcon (*Falco peregrinus*) (ROMO 1992). Encroachment of human populations at park boundaries is a particular concern with respect to loss of winter habitat and alteration of migration routes for elk. Recent increases in elk populations in ROMO have produced high rates of herbivory with potential changes in distribution and biomass of terrestrial vegetation.

Research projects currently underway on vegetation in ROMO include restoration, rehabilitation, effects of global climate change on species composition and control of non-native species. Current monitoring and research projects focus primarily on the impacts of N deposition on soils and biogeochemical cycling in ROMO, especially in the Loch Vale watershed.

B. EMISSIONS

ROMO lies in the Front Range of the Colorado portion of the Rocky Mountains 60 km northwest of the Denver-Boulder urban areas and 30 km west of Fort Collins. The proximity to large urban areas makes the park vulnerable to pollution from both point and mobile sources (including more dispersed sources such as livestock feedlots). Total point source emissions of SO_2, NO_x, and VOCs within 140 km of ROMO are summarized by county in Table III-1. Over half of the total SO_2 emissions for the state are generated within this 140 km range. Coal-burning power plants are the major emission sources of SO_2 and NO_x in this region.

Table III-1. 1994 emissions (tons/year) within 140 km of ROMO. (Source: Electric Power Research Institute 1995)

County Name	SO_2	NO_x	VOCs
Adams	16,464	16,777	2,322
Albany	1,319	2,531	0
Arapahoe	0	711	155
Boulder	4,125	4,214	995
Denver	4,567	6,436	7,486
Jefferson	2,834	2,180	2,159
Larimer	1,488	3,112	0
Laramie	1,521	2,630	3,418
Moffat	11,475	18,093	150
Routt	11,277	9,894	0
Weld	320	7,149	618
Total	55,390	73,900	17,303

The Denver metropolitan area is the largest urban center in Colorado with over 2 million inhabitants. Approximately 75% of the population live in the suburban counties surrounding Denver. In 1994, 87% of workers in the Denver-Boulder area commuted in privately owned vehicles: 75% drove alone and 12% carpooled (U.S. Department of Transportation 1994). The high proportion of automobile commuters and large number of suburban residents contribute to NO_x and VOC production in the region. Agricultural activities likely contribute to the regional emissions of NH_3.

NO_x, NH_3, and VOC emissions from adjacent counties that are upwind during parts of the year pose a potential threat to ROMO and surrounding wildland areas. NO_x and VOCs are major precursors of tropospheric ozone, and winds from the southeast may transport ozone and ozone precursors to the foothills and higher elevations within the park. In wildland areas such as ROMO, local sources of ozone-degrading NO_x are minimal and ozone levels can remain elevated for several days (Logan 1985, Sandroni et al. 1994).

C. MONITORING AND RESEARCH ACTIVITIES

1. Air Quality

a. <u>Wet deposition</u>

Atmospheric deposition of N and S in snow and rain along the Front Range in northern Colorado is among the highest of any area in the Rocky Mountains (Turk et al. 1992). Annual inorganic N loading in wet deposition in the Colorado Front Range is about twice that of the Pacific states and is similar to some states in the Northeast (Williams et al. 1996a). The volume-weighted mean annual concentrations of N in precipitation at Loch Vale, in the southeastern portion of the park, were 11 μeq/L NO_3^- and 7 μeq/L NH_4^+ during 1992 (Campbell et al. 1995).

Nitrate concentrations in the snowpack at maximum snowpack accumulation in the northern Colorado Front Range were among the highest measured in Colorado (Turk et al. 1992). Concentrations of NO_3^- and SO_4^{2-} in the snowpack along the Continental Divide in northern Colorado were found to be twice the regional background level found throughout the Rocky Mountains in 1991 and 1992 (Turk et al. 1992). Synoptic snow survey data were collected at three sites within ROMO during March and April, 1995. Concentrations of NO_3^- in snowpack were in the range of 10 to 13 μeq/L at all three sites (Table III-2). Snowpack concentrations of SO_4^{2-} and NH_4^+ were also slightly elevated above the expected background concentrations, in the range of 7 to 10 μeq/L for SO_4^{2-} and 4 to 6 μeq/L for NH_4^+ (G. Ingersoll, unpublished data). Such snowpack monitoring data are useful for examining spatial and temporal variation in deposition at areas other than where NADP sites are located. At many sensitive high-elevation sites, snowpack chemistry data are the only kind of deposition data available. Even though snowpack data do not provide deposition information for the entire year, they can help to identify hot spots for further research and to better quantify broad regional patterns in deposition in a cost-effective fashion.

Table III-2 Synoptic snow survey data at ROMO sites in March and April, 1995. All units are μeq/L, except for pH. (Source: G. Ingersoll, USGS, unpublished data)

Site Name	Date	Snowpack Concentration of Major Ions									
		pH	H^+	NH_4^+	Ca^{2+}	Mg^{2+}	Na^+	K^+	Cl^-	SO_4^{2-}	NO_3^-
Loch Vale	4/12/95	4.99	10.2	6.2	8.6	2.0	2.0	2.6	1.4	9.8	12.9
Lake Irene	3/29/95	4.97	10.7	3.7	3.1	0.7	1.4	<0.3	0.9	7.1	9.6
Phantom Valley	3/28/95	4.89	12.9	4.6	3.8	1.2	1.7	0.5	1.1	8.3	11.4

ROMO has a high-elevation NADP/NTN site located in the Loch Vale watershed at an elevation of 3,159 m and a lower elevation site at Beaver Meadows (2,490 m). The Beaver Meadows site has been in operation since 1980 and the Loch Vale site since 1983. Both sites receive precipitation with elevated levels of S and N, compared with western parks in general. Annual average concentrations of major ions in wetfall are presented in Table III-3 and Table III-4.

Wet S deposition at Loch Vale decreased from 3.1 kg S/ha/yr in 1985 to values around 2 kg S/ha/yr during the period 1987 though 1995 (Table III-5). The pattern of sulfur loss in discharge (which can provide some information about inputs) was similar to the yearly water discharge pattern during the past decade, with most sulfur losses occurring during snowmelt. Total sulfur losses from the Loch Vale basin were considerably higher than wet inputs, and ranged from 3.3 to 4.2 kg S/ha/yr

Table III-3. Wetfall chemistry at the NADP/NTN site at Beaver Meadows. Units are in μeq/L, except precipitation (cm).

Year	Precip	H^+	SO_4^{2-}	NH_4^+	NO_3^-	Ca^{2+}	Mg^{2+}	Na^+	K^+	Cl^-
1995	53.7	11.1	12.8	12.6	16.5	7.1	1.3	2.1	0.5	2.0
1994	35.2	12.9	16.1	15.9	20.9	9.1	1.7	2.0	0.8	2.7
1993	41.2	4.5	11.5	12.6	14.1	6.8	1.3	2.3	1.1	2.1
1992	36.6	5.5	14.8	15.3	17.1	8.9	1.7	2.7	3.3	4.1
1991	38.4	6.9	14.5	12.8	16.3	9.9	1.8	2.5	1.2	2.6
1990	46.6	5.8	13.7	14.5	15.8	15.9	2.3	2.2	0.5	2.3
1989	33.2	5.5	16.4	14.0	17.8	13.2	2.3	3.8	1.9	4.4
1988	25.4	6.5	15.9	5.9	15.2	12.8	2.4	4.4	0.7	2.7
1987	16.4	6.4	8.0	5.8	9.8	3.4	0.8	2.7	0.3	1.7
1986	44.0	8.5	16.1	13.0	15.1	8.3	1.7	2.6	2.7	3.4
1985	41.8	10.2	19.0	9.7	15.3	10.5	3.3	2.7	1.0	3.6
1984	43.1	7.3	20.3	13.8	18.2	13.3	4.4	4.9	2.1	3.3
1983	45.3	8.1	16.8	8.6	13.8	10.7	2.8	4.4	0.9	2.7
1982	36.8	10.8	19.0	7.5	15.0	10.6	2.9	2.4	1.2	2.6
1981	33.4	10.1	33.0	19.1	23.5	19.1	6.7	5.1	1.2	4.3
Average	38.07	8.01	16.53	12.07	16.29	10.64	2.49	3.12	1.29	2.97

Table III-4. Wetfall chemistry at the NADP/NTN site at Loch Vale. Units are in μeq/L, except precipitation (cm).

Year	Precip	H^+	SO_4^{2-}	NH_4^+	NO_3^-	Ca^{2+}	Mg^{2+}	Na^+	K^+	Cl^-
1995	142.8	9.5	9.1	6.1	9.3	5.2	1.1	2.0	0.4	1.4
1994	117.0	12.4	12.3	9.5	14.3	5.8	1.2	1.8	0.5	1.9
1993	121.1	5.2	12.2	7.7	11.4	6.6	1.5	5.8	1.0	5.1
1992	93.9	6.5	11.0	7.0	11.9	5.8	1.2	2.0	0.5	1.8
1991	100.3	6.2	10.8	6.0	11.4	6.6	1.3	2.7	0.4	2.4
1990	112.6	6.3	11.3	8.1	13.0	9.8	2.2	2.4	0.4	2.5
1989	90.5	4.9	12.5	7.7	11.4	8.7	1.9	3.8	0.4	2.2
1988	78.0	7.5	11.4	3.3	8.9	6.4	1.2	2.8	0.3	1.8
1987	96.4	7.1	10.3	5.5	9.7	4.9	1.3	3.3	0.4	2.0
1986	106.3	6.5	14.4	7.3	11.5	9.3	2.2	3.0	0.5	2.0
1985	109.8	9.0	17.5	6.2	13.1	12.2	3.5	3.5	1.0	3.4
1984	111.1	7.2	14.8	6.9	11.0	8.7	3.1	3.1	0.5	2.6
Average	106.65	7.36	12.3	6.78	11.41	7.5	1.81	3.02	0.53	2.43

Table III-5. Wet deposition (kg/ha/yr) of S and N at the NADP/NTN site at Loch Vale.

Year	Sulfur	NO_3-N	NH_4-N	Total Inorganic N
1995	2.1	1.9	1.2	3.1
1994	2.3	2.3	1.6	3.9
1993	2.4	1.9	1.3	3.3
1992	1.6	1.6	0.9	2.5
1991	1.7	1.6	0.9	2.5
1990	2.0	2.0	1.3	3.3
1989	1.8	1.4	1.0	2.4
1988	1.4	1.0	0.4	1.3
1987	1.6	1.3	0.7	2.0
1986	2.5	1.7	1.1	2.8
1985	3.1	2.0	1.0	3.0
1984	2.6	1.7	1.1	2.8
Average	2.09	1.7	1.04	2.74

(Baron et al. 1995). This information, coupled with discovery of small pyrite deposits within the basin, suggests a significant mineral source of S in the Loch Vale basin (Mast et al. 1990). Interpretation of potential ecosystem effects of decreased sulfur emissions and deposition since 1984 is obscured by the apparent internal watershed sources of sulfur (Baron et al. 1995). Wet N deposition at Loch Vale during the period 1983 through 1995 has generally been in the range of 2 to 3 kg N/ha/yr, with a maximum of 3.9 kg N/ha/yr in 1994 (Table III-5). There has been no trend in seasonal or annual inputs (Baron et al. 1995). Greater wet inputs of N occurred during years with higher precipitation, particularly those years with higher precipitation during winter. N deposition was statistically correlated with patterns of precipitation using a Pearson product-moment correlation ($p > 0.01$). Wet deposition at Beaver Meadows is considerably lower than at Loch Vale for both S and N (Table III-6).

Annual loading of inorganic N in wet deposition to the Colorado Front Range is about 3 kg N/ha/yr at Loch Vale and near 5 kg N/ha/yr at Niwot Ridge (Table III-7), which is quite high for the western United States, although still relatively modest by comparison with many areas of the eastern United States and Europe. Annual NO_3^--N loading at Niwot Ridge, an alpine research area south of

Table III-6. Wet deposition (kg/ha/yr) of S and N at the NADP/NTN site at Beaver Meadows.

Year	Sulfur	NO_3-N	NH_4-N	Total Inorganic N
1995	1.1	1.2	1.0	2.2
1994	0.9	1.0	0.8	1.8
1993	0.8	0.8	0.7	1.5
1992	0.9	0.9	0.8	1.7
1991	0.9	0.9	0.7	1.6
1990	1.0	1.0	1.0	2.0
1989	0.9	0.8	0.7	1.5
1988	0.6	0.5	0.2	0.8
1987	0.2	0.2	0.1	0.4
1986	1.1	0.9	0.8	1.7
1985	1.3	0.9	0.6	1.5
1984	1.4	1.1	0.8	1.9
1983	1.2	0.9	0.5	1.4
1982	1.1	0.8	0.4	1.2
1981	1.8	1.1	0.9	2.0
Average	0.99	0.84	0.65	1.51

Table III-7. Comparison of annual volume-weighted mean concentrations for NH_4^+, NO_3^-, and total annual loading of inorganic N from selected NADP sites, 1991-1994. All means are arithmetic means of annual values. (Source: Williams et al. 1996a)

Site	NH_4^+ (μeq/L)	NO_3^- (μeq/L)	Inorganic N (kg/ha/yr)
Loch Vale, CO	7.2	11.6	2.73
Niwot Ridge, CO	5.0	15.6	4.71
GLEES, WY	6.7	11.5	3.09
Acadia NP, ME	5.6	13.2	3.60
Hubbard Brook, NH	8.3	21.9	4.87
Yosemite NP, CA	8.3	8.4	2.53
Olympic NP, WA	1.1	1.6	1.04

ROMO, has approximately doubled over the last decade, from 2 to 4 kg NO_3^- N/ha/yr, based on NADP data. Niwot Ridge is located at 3,500 m elevation about 60 km northwest of the Denver Metropolitan area and is exposed to the same general airshed as ROMO. An increase in precipitation amount during that period of time explained about half of the observed variation in annual wet NO_3^- deposition (Williams et al. 1996a). At the GLEES site, to the north of ROMO in southeastern Wyoming, annual average NO_3^- wet deposition has also increased since measurements began in 1986 from about 1 to 2 kg NO_3^- N/ha/yr.

b. Occult/Dry Deposition

Whereas dry deposition in the Rocky Mountains contributes less than 25% of total deposition of most chemical species (with the exception of Ca^{2+}) in winter, based on measurements from the maximum snowpack accumulation (Table III-8), the contribution of dry deposition in summer is uncertain. A comparison of the chemical composition of lakes and wetfall suggested no significant dry deposition of SO_4^{2-} across the Rocky Mountain region in general (Turk and Spahr 1991). However, such an analysis is inconsistent with the fact that the area around ROMO (northwest of Denver) is the major portion of the Front Range expected to be impacted by dry deposition from upslope air movement from the Denver metropolitan area.

Bulk precipitation and throughfall chemistry were measured in an old growth Engelmann spruce/subalpine fir forest in the lower section of the Loch Vale watershed by Arthur and Fahey (1993). They calculated that dry deposition represented 56% of the atmospheric input of NO_3^- to the forest during the period May through October, 1986 and 1987. Values for dry deposition as a fraction of total deposition for cations ranged from 57% for NH_4^+ to 66% for Mg^{2+} and 83% for K^+. In contrast, no dry deposition of SO_4^{2-} was calculated. Arthur and Fahey (1993) reported SO_4^{2-} wet

Table III-8. Water budget and volume-weighted mean (VWM) chemical concentrations in the snowpack (seasonal), precipitation (seasonal and annual), and stream water (annual) in the Loch Vale watershed. (Source: Campbell et al. 1995)

	Precipitation				Streams	
	Snowpack: Maximum Accumulation April 1992	NADP: October-March 1992	NADP: April-September 1992	NADP: WY1992	Andrews Creek WY1992	Icy Brook WY1992
Water, cm	44	47	43	90	85	64
Calcium, μeq/L	7	2	9	6	57	66
Magnesium, μeq/L	2	1	2	1	14	16
Sodium, μeq/L	2	2	3	2	18	18
Potassium, μeq/L	1	<1	1	<1	4	4
Hydrogen ion, μeq/L	5	6	7	6	<1	<1
Ammonium, μeq/L	6	4	10	7	1	1
Nitrate, μeq/L	10	8	14	11	23	21
Chloride, μeq/L	2	2	2	2	3	4
Sulfate, μeq/L	9	7	16	11	31	37
ANC, μeq/L	<1	<1	<1	<1	30	33
Silica, μmol/L	<2	<2	<2	<2	35	27

WY = water year

deposition at this spruce-fir site equal to 15.7 mg/m^2/week (0.157 kg/ha/wk) from May to October in 1986 and 1987. This value is similar to results from other studies at high-elevation sites in Colorado. For example, Grant and Lewis (1982) measured SO_4^{2-} input of 16.3 mg/m^2/week (0.163 kg/ha/wk) at Como Creek. Lewis et al. (1984) found that SO_4^{2-} deposition at 19 high-elevation (> 2,000 m) sites ranged from 5.0 to 35.8 mg/m^2/week (0.05 to 0.358 kg/ha/wk), with an average value of 14.3 mg/m^2/week (0.143 kg/ha/wk). Sulfate concentrations in streamwater throughout the Loch Vale watershed are about three times those in precipitation.

Atmospheric concentrations of S and N species have been measured in ROMO since 1995 as part of the National Dry Deposition Network (NDDN). Dry deposition flux calculations are not yet available, but are expected to be available in the near future. The scientific consensus is that dry deposition of sulfur at ROMO is not a major component of total deposition, and that the observed high concentrations of SO_4^{2-} in streamwater at Loch Vale are due largely to the occurrence of sulfide-bearing minerals in the watershed (Campbell et al. 1995; J. Baron, pers. comm.). Dry deposition to exposed bedrock surfaces appears to be important, however, at least during the snow-free season. Volume-weighted concentrations of NO_3^- and SO_4^{2-} in runoff from a bedrock catchment at Loch Vale were two to four times higher than in precipitation (Clow and Mast 1995). About 15% of the solute increase could be accounted for by evaporation from the rock surface. However, it is unclear to what extent runoff NO_3^- concentrations were increased by N-fixation of lichens or the extent of sulfur contribution from mineral deposits in the bedrock. Thus, the data of Clow and Mast (1995) cannot be used to quantify dry deposition fluxes to this watershed.

Cress et al. (1995) measured dry deposition of N to a snowpack in 1993 at Niwot Ridge. Changes in the concentration and mass of NO_3^- and NH_4^+ were measured in buckets filled with excavated snow and installed level with the snow surface. Exposure times ranged from 3 hrs up to 48 hrs. Ambient samples of HNO_3, particulate NO_3^- and particulate NH_4^+ were collected using filter packs with a Teflon pre-filter (for particulates) and a nylon filter for HNO_3. Ambient concentrations were determined by dividing the measured mass of N by the volume of air sampled. A large increase was observed of all three N species after April 16. This increase in ambient N correlated in general with the seasonal change in late spring from westerly winds to upslope winds, and the arrival of convective air masses from the Denver urban corridor (Baron and Denning 1993, Cress et al. 1995).

The concentration of N in snow samples increased by 100% between calendar days 146 and 155, an increase attributed by Cress et al. (1995) to increased dry deposition to the snowpack as a result of the extremely high ambient atmospheric concentrations of HNO_3 and particulate NO_3^- that were measured on day 153. The meteorological data showed a northwesterly flow directly from the Denver area on day 153. These data suggest that contamination of the atmosphere of alpine areas in the vicinity of Niwot Ridge (and presumably also ROMO) can result in the deposition of a significant mass of nitrogen to the snowpack in a very short period of time (Cress et al. 1995). Thus, it appears that deposition to ROMO can be strongly influenced by patterns of air movement within the region.

Total N deposition was estimated by Sievering et al. (1992) for two alpine sites at Niwot Ridge during the period 1987 to 1989. Comparisons were made of wet plus dry inputs for the four-month growing season and the remaining eight-month dormant period. During the growing season, atmospheric N input was estimated to be greater than 1 mg N/m^2/day directly from the atmosphere with a similar amount contributed from snowmelt as a result of deposition to the previous winter snowpack.

At a nearby subalpine forested site, wet plus dry deposition of nitrogen to a lodgepole pine canopy was found to vary from < 1 mg N/m^2/day to 2 mg N/m^2/day (Sievering et al. 1989). These fluxes correspond to total annual N loading estimates of about 3 to 7 kg N/ha/yr.

Concentrations of atmospheric particulate NO_3^- (pNO_3^-) appear to have increased during the last decade at some sites along the Front Range. For example, Rusch and Sievering (1995) compared concentrations of pNO_3^- measured during the month of July in several research efforts at the C-1 research site at Niwot Ridge (e.g., Fahey et al. 1986, Parrish et al. 1986, Marquez 1994). The data suggested an approximate doubling of ambient pNO_3^- in the last decade. Wet N deposition also doubled during that period of time.

Studies conducted by Langford and Fehsenfeld (1992) at Niwot Ridge and also 25 km to the east, near the eastern edge of the forest at Boulder, illustrated that the forest canopy will act as both a source and a sink for atmospheric NH_3. During periods of westerly flow (low in NH_3), the forest acted as a source of NH_3 with mean NH_3 emission rates of about 1.2 $ng/m^2/sec$. Periods of easterly (upslope) flow induced by insulation of the mountain surfaces often occur between mid-morning and late afternoon during the summer. During these periods, the forest (especially the eastern edge) is exposed to NH_3-enriched air masses from the agricultural plains to the east. During upslope conditions, the forest became a net sink for NH_3, with a mean uptake rate of about 10 $ng/m^2/sec$ (20^oC) near Boulder and decreasing from east to west as NH_3 was depleted from the air masses.

Ratios of NO_3^- to SO_4^{2-} in wetfall (0.8) and bulk precipitation (1.1) were high at Loch Vale compared to other mountainous sites in the region (Arthur and Fahey 1993). This may be due to the interception by Loch Vale and surrounding areas of southeasterly and easterly winds from the Denver area and agricultural areas east of the park, which are enriched in nitrogen compounds.

The N loading to alpine and subalpine systems in ROMO may be functionally much higher than is reflected by the total annual deposition measured or estimated for the watersheds. It may therefore be misleading to compare total N loading rates of 3 to 7 kg N/ha/yr, for example, of some alpine systems in the Front Range with the higher loading rates found in parts of eastern United States and northern Europe. There are several reasons for this. First, the actual N loading to both soils and drainage waters at high-elevation sites during summer comprises 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 (under the snowpack) mineralization of the primary production of the previous summer (Brooks et al. 1995a,b). This N load from internal ecosystem cycling will likely 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). Therefore, N released by internal cycling can be coupled to external (depositional) inputs. Thus, the functional N loading to terrestrial and aquatic runoff receptors in alpine and subalpine areas during the summer growing season is much higher (perhaps more than double) than is represented by 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.

c. Gaseous Monitoring

Ozone and SO_2 are the only gaseous pollutants currently monitored in ROMO. A continuous ozone analyzer has been in operation year round on the Leiffer Ranch (elevation 2,700 m) east of the Long's Peak campground (12 km from park Headquarters) since 1986 (Figure III-3). The average annual maximum 1-hour ozone concentration for ROMO between 1989 and 1994 was 99 ppbv. Annual maximum 1-hour concentrations are the highest of all monitored parks in the Rocky Mountain region and exceeded the NAAQS in 1993 (127 ppbv), indicating that upwind regional ozone precursor sources have a substantial impact on air quality in the park (Table III-9). The mean daytime 7-hour ozone concentration during the growing season ranged between 40 and 55 ppbv during this time period. The SUM60 exposure index is another indicator that can be important in assessing ozone exposures of plant species. This index is the sum of all hourly ozone concentrations equaling or exceeding 60 ppbv. The SUM60 exposure index at ROMO ranged between 1,837 and 34,586 in this time period. For comparison, national parks in highly polluted areas (e.g., southern California) can have SUM60 exposure indexes exceeding 100,00 ppbv-hour (Joseph and Flores 1993).

Table III-9. Summary of ROMO ozone concentrations (ppbv) from NPS monitoring sites (Joseph and Flores 1993).

	1987	1988	1989	1990	1991	1992	1993	1994[a]
1-hour maximum	98	119	98	70	95	98	127	109
Average daily mean	46	45	35	34	41	-	-	44
Growing season 7-hour mean	55	53	45	40	49	-	-	54
SUM60 exposure index (ppbv-hour)	19,361	32,828	11,796	1,837	15,270	-	-	34,586

[a] NPS Monitoring data
- indicates data not available

An analysis of diurnal concentration data indicates that maximum ozone concentrations at ROMO lag slightly behind those of lower elevation urban areas (Figure III-4; Colorado Dept. of Public Health 1994). Minimum values also tend to remain at or above 40 ppbv during much of the summer because of the lack of local sources of NO_x to scavenge ozone molecules.

Passive ozone samplers were used in ROMO for a three-week long monitoring study in 1995 to measure ozone concentrations across an elevation gradient. Samplers were used to measure ozone on the east and west slopes of the Continental Divide ranging from 2,300 to 3,400 m in elevation. Results of this study indicate that on the east side sites, weekly average ozone

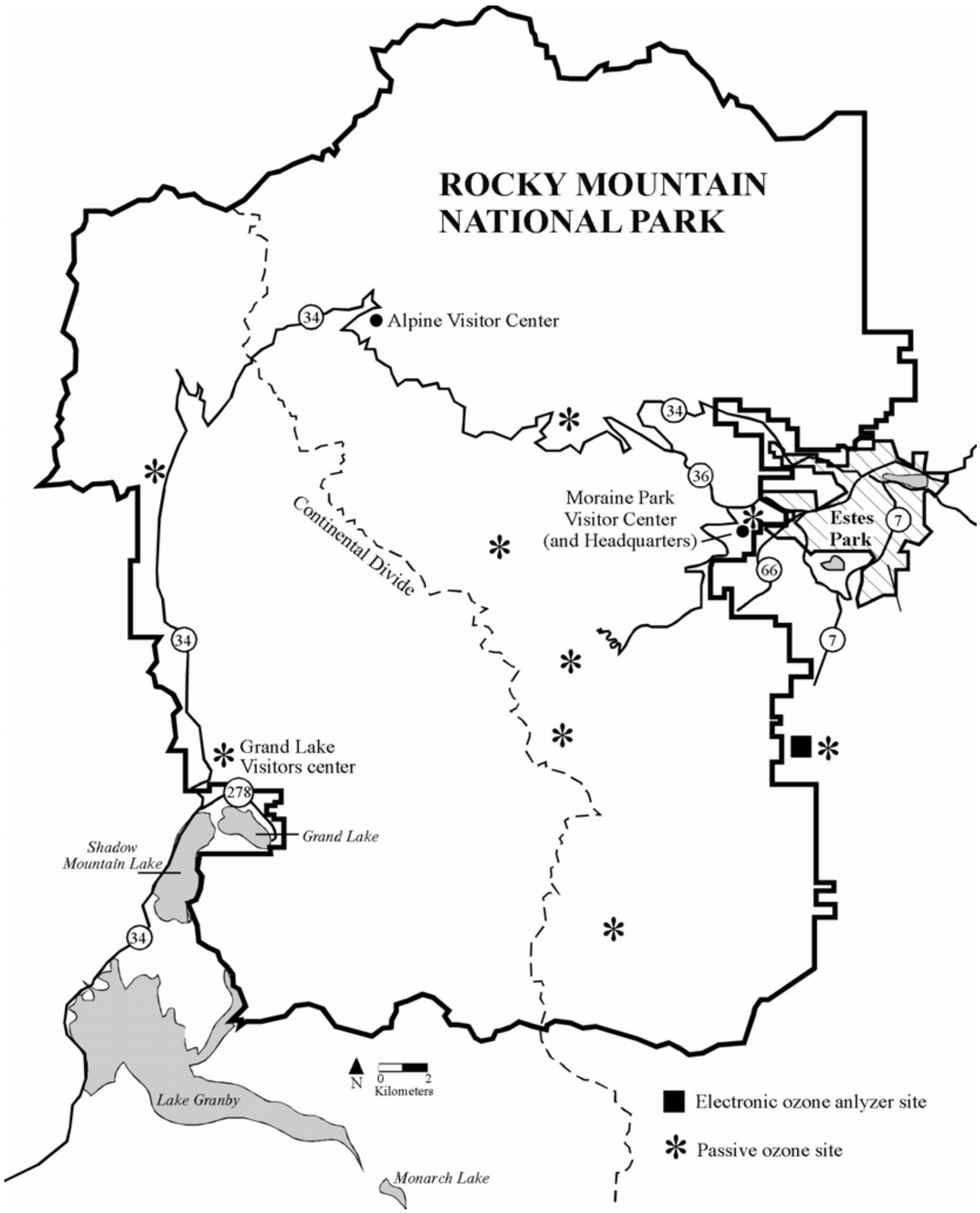


Figure III-3. Map of ROMO and air quality monitoring sites.

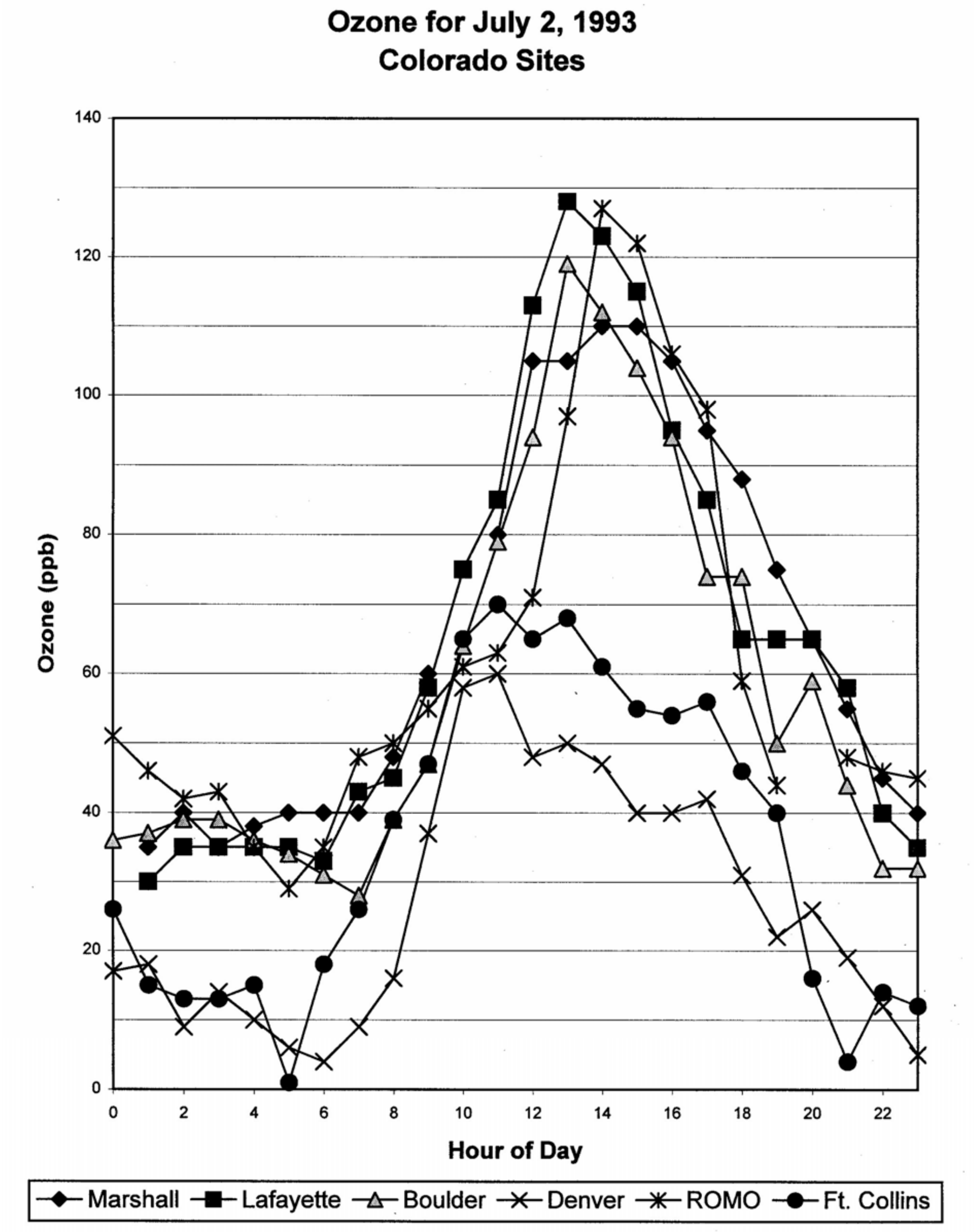


Figure III-4. Diurnal ozone concentrations at ROMO and other locations along the Front Range on July 2, 1993.

concentrations increase with elevation to approximately 3,000 m (Table III-10). Above this elevation concentrations drop about 10 ppbv per 500 m, suggesting that the average height of the mixing layer is at about 3,000 m. On the west side, ozone concentrations are lower compared to east side sites. The west-side low-elevation site may be protected from the regional ozone plume moving from the east by the mountain ridges along the Continental Divide. Up-valley winds may deliver ozone precursors and ozone to higher elevations where they are circulated and distributed (Thyer and Buettner 1961, Broder and Gygax 1985). Other recent reports based on passive ozone samples confirm this pattern of higher ozone concentrations at higher elevations (Ray 1993, Brace and Peterson 1996,1998).

Table III-10. Weekly average ozone concentrations (ppbv) at passive sampling sites in ROMO. Details of sampling and analysis are described in Ray (1993).

Sample Location	Elev (m)	Wk 1	Wk 2	Wk 3
East side				
Cub Lake Trail	2,360	46.4	45.5	49.2
Headquarters	2,660	47.8	41.8	48.3
Monitoring site (Estes Park)	2,740	49.8	51.2	49.5
Bear Lake	2,900	49.1	45.8	44.6
Wild Basin	2,926	58.9	58.7	59.0
Hidden Valley Slope	3,170	52.0	37.8	48.9
Loch Vale	3,340	50.2	49.4	49.6
West side				
Grand Lake Visitor Center	2,600	40.7	39.5	32.7
Never Summer Ranch	2,731	33.1	31.3	32.8

Sulfur dioxide has been measured in ROMO since 1991, and annual average 24-hr concentrations range from 0.04 to 0.09 ppbv. In 1993, the maximum 24-hour SO_2 concentration in the park was 0.42 ppbv. These values are much lower than the concentration that is considered potentially damaging to some vegetation (Treshow and Anderson 1989). Maximum 24-hr SO_2 concentrations measured at ROMO in 1995 (0.13 ppbv) are only one-third of the 1993 level (0.42 ppbv; Table III-11). However, it is important to remember that a maximum value may be an anomaly. Mean values are a better representation of typical conditions.

In addition to the air quality monitoring efforts underway in ROMO, the U.S. Forest Service and other national parks and monuments in the Rocky Mountain region are also involved in air quality monitoring (Table III-12). Most monitoring efforts are focused on acid deposition (lake chemistry) and visibility.

Table III-11. Maximum and mean SO_2 24-hour integrated sample. The clean-air reference is estimated to be 0.19 ppbv (Urone 1976). (Source: J. Ray, NPS Air Resources Division)

	SO_2 concentration (ppbv)				
	1991	1992	1993	1994	1995
Maximum	0.28	0.08	0.42	0.37	0.13
Mean	0.07	0.04	0.09	0.08	0.05

Table III-12. Additional air quality monitoring on federal lands in the central Rocky Mountains.

Location	Type of monitoring
Arapaho and Roosevelt National Forests	Lake chemistry
Black Canyon of the Gunnison National Monument	Visibility, passive ozone
Bridger-Teton National Forest	NADP, lake chemistry, visibility, ozone, NDDN
Colorado National Monument	Visibility, ozone (1984-1992)
Dinosaur National Monument	Visibility
Grand Mesa, Uncompaghre, Gunnison National Forests	Visibility, lake chemistry
Grand Teton National Park	Lake chemistry, passive ozone, snow chemistry
Great Sand Dunes National Monument	Visibility, ozone 1988-1991, NADP (in Alamosa), SO_2 1988-1992
Medicine Bow National Forest	Lake chemistry, NADP, visibility, air chemistry
Mesa Verde National Park	Visibility, NADP, ozone, SO_2
Pike and San Isabel National Forests	Visibility, lake chemistry
Rio Grande and San Juan National Forests	Lake chemistry, visibility, NADP
Routt National Forest	Lake chemistry, NADP, air chemistry, visibility
Shoshone National Forest	NADP, lake chemistry
Yellowstone National Park	NADP, ozone, SO_2, lake chemistry, visibility, snow chemistry, NDDN
White River National Forest	NADP, lake chemistry, visibility

2. Aquatic Resources

a. Water Quality

Aquatic resources in ROMO include a wealth of lakes and streams of exceptional quality. The natural lakes and stream valleys were formed by glaciation. The majority of the surface waters in the park are found in alpine and subalpine settings, most of which are accessible only on foot or horseback. Many high-elevation surface waters are fed by small glaciers. Because of the proximity of so many ROMO surface waters to the Continental Divide, human impacts on the water quality are minimized. With the exception of anthropogenic atmospheric contributions of pollutants, human impacts on most lakes and streams in the park, especially those in remote locations, are restricted to a few dams and irrigation channels, as well as the impacts of hiking, camping, and horseback-riding activities. Atmospheric deposition of air pollutants therefore represents one of the most important potential threats to aquatic resources in this park.

Lakes and streams in ROMO tend to be clear-water, low ionic strength, oligotrophic systems. Concentrations of virtually all dissolved constituents except oxygen (e.g., nutrients, organic material, major ions, weathering products) tend to be very low. ROMO surface waters can be categorized as clear, cold, dilute systems that are highly sensitive to degradation by human activities.

The air quality related values associated with aquatic resources in the park include water quality and aquatic biota. Both can be adversely impacted by atmospheric deposition of nitrogen or sulfur. Sulfur deposition can cause chronic and/or episodic acidification of surface waters. Nitrogen deposition can cause acidification, eutrophication, and excessive algal productivity. Common water quality measurements to determine the status of water quality AQRVs include pH, ANC, and concentrations of SO_4^{2-}, and NO_3^-. Measurements of the response of biota can include algal species composition and abundance and the presence or absence of acid-sensitive fish, amphibian and invertebrate species.

Although chronic acidification of surface waters is not currently a problem in the Rocky Mountain region (Turk and Spahr 1991), episodic acidification during snowmelt may be occurring at some sites and is an important concern. In addition, because many lakes and streams in the region have low ANC, there is concern about potential chronic acidification if levels of atmospheric deposition of N or S increase in the future.

Analyses of 1985 fall samples from lakes in ROMO (n=22) and in adjacent wilderness areas (n=14) from the Western Lake Survey (Eilers et al. 1988) showed that the median ANC for these lakes was 80 μeq/L, with 20% of the lakes having ANC < 41 μeq/L (Table III-13). The minimum ANC value measured in this subpopulation was 19 μeq/L (Table III-14). These ANC values are similar to ANC values for other sensitive areas of the West. Minimum pH and base cation values were 6.48 and 47 μeq/L, respectively. Sulfate concentrations ranged from 10 to 113 μeq/L. This illustrates the importance of watershed sources of S to many lakes in the area, because

Table III-13. Population statistics[a] for ANC[b], C_B[c], SO_4^{2-}, DOC, and SO_4^{2-}-C_B for wilderness lakes within selected geomorphic units of the West compared with two major park areas in the East and the Midwest. (Source: Eilers et al. 1988)

Wilderness Subpopulation	Lakes Sampled	Estimated Population Size	ANC (µeq/L)			C_B (µeq/L)			SO_4^{2-} (µeq/L)			DOC (mg/L)			SO_4^{2-}/C_B		
			Q1	M	Q4	Q1	M	Q4	Q1	M	Q4	Q1	M	Q4	Q1	M	Q4
Western Lake Survey[d]																	
Sierra Nevada, CA	71	1,787	29	53	104	42	67	115	4	7	14	0.4	0.7	1.5	0.01	0.09	0.17
Oregon Cascades	21	217	29	86	169	38	93	184	<1.5	<1.5	3	1.2	1.9	2.2	<0.01	<0.01	0.05
N. Washington Cascades	52	537	47	106	193	67	134	239	8	17	42	0.3	0.6	1.2	0.06	0.11	0.20
Bitterroot Mtns., ID/MT	44	394	42	79	176	52	99	205	5	10	15	0.8	1.2	1.9	0.04	0.07	0.10
Wind River Range, WY	44	830	73	104	165	113	146	237	21	24	32	0.6	1.1	3.0	0.04	0.15	0.18
Front Range, CO	36	144	41	80	330	77	130	413	18	24	90	0.7	1.0	1.2	0.08	0.16	0.22
Eastern Lake Survey[e]																	
Adirondack Park, NY[f]	127	1,091	8	81	206	132	234	387	104	118	134	2.6	4.1	5.8	0.33	0.45	0.57
Northeastern Minnesota[g]	147	1,457	98	185	403	218	314	550	47	62	84	5.5	9.2	12.7	0.10	0.18	0.25

[a] Q1=20th percentile, M=median, Q4=80th percentile

[b] Based on original Gran titration results computed for the two contract laboratories in the WLS and the four laboratories in the Eastern Lake Survey. Alternative ANC values are available that were computed using a consistent algorithm for both surveys

[c] C_B (base cations)=$Ca^{2+} + Mg^{2+} + Na^{+} + K^{+}$

[d] Includes lakes between 1 and 2,000 ha in surface area.

[e] Includes lakes between 4 and 2,000 ha in surface area.

[f] Includes lakes within the Adirondack Park boundary.

[g] Includes lakes within the Boundary Waters Canoe Area, the Voyagers National Park, and portions of the Superior National Forest.

Table III-14. Continued

Lakes in the Colorado Front Range Outside ROMO

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)
Blue Lake	4E1-040	9	259	3446	6.9	25	21	5.8	41	57	0.7
Crater Lake	4E1-041	10	275	3141	8.5	68	14	0.3	64	78	0.8
Green Lakes (NE)	4E1-043	4	47	3434	7.7	288	146	6.3	353	439	1.0
Caribou Lake	4E1-045	2	57	3400	7.5	307	34	4.7	218	390	0.9
No Name	4E1-046	4	49	3610	7.0	27	10	8.7	30	47	0.3
Upper Diamond Lake	4E1-047	2	49	3580	7.6	110	13	8.2	79	133	0.5
Jasper Lake	4E1-048	7	254	3298	7.6	148	33	1.4	133	193	1.3
King Lake	4E1-049	4	18	3486	6.9	56	21	1.4	52	88	0.6
Woodland Lake	4E1-050	2	171	3346	7.5	177	39	2.4	157	242	1.2
No Name	4E1-055	9	212	3422	6.8	33	19	7.6	43	65	0.6
Bob Lake	4E1-056	3	31	3532	7.3	60	29	7.7	64	97	0.7
Red Deer Lake	4E1-059	6	85	3163	7.4	79	30	0.7	80	118	1.8
Stapp Lakes (Lg. NE)	4E2-015	2	21	2861	7.2	111	38	2.2	83	195	11.3
Pumphouse Lake	4E2-021	2	28	3457	7.4	154	20	0.1	103	197	1.1
Crater Lakes (NW)	4E2-022	3	54	3361	7.4	129	17	5.8	100	168	0.4
Clayton Lake	4E2-023	2	166	3349	7.4	169	30	6.3	132	230	1.8
Murray Lake	4E2-024	4	142	3690	7.6	302	37	1.0	198	354	1.2
Duck Lake	4E2-025	18	570	3392	7.3	274	55	0.8	207	368	2.1
Chicago Lakes (Lg. North)	4E2-026	9	342	3483	7.2	226	73	9.9	241	335	1.8
No Name	4E2-05	1	10	3510	9.	692	31	0.3	553	808	9.9

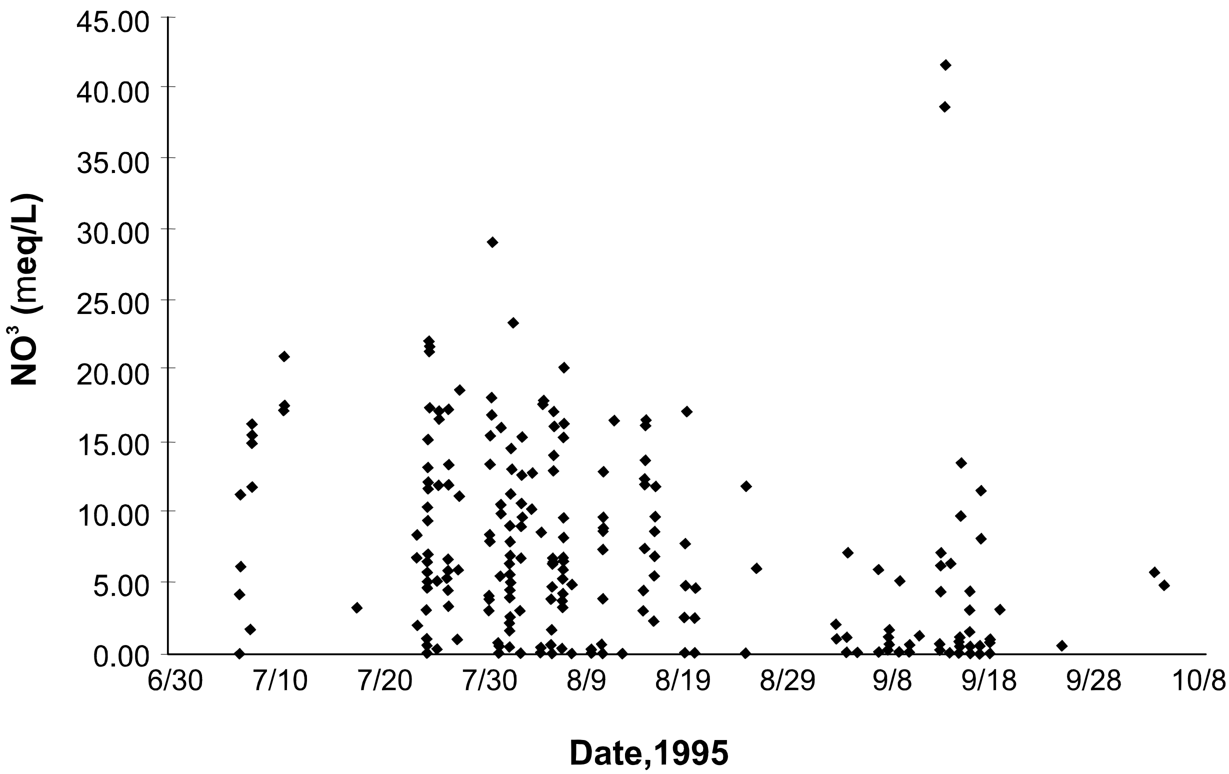
	1				3						
Forest Lakes (Lg. N)	4E2-055	2	98	3300	7.7	192	21	0.5	142	233	0.9
Panhandle Reservoir	4E3-005	20	4903	2592	7.8	377	30	0.0	240	420	3.2
Dowdy Lake	4E3-006	43	228	2481	8.0	952	67	1.5	696	1183	6.9
Parika Lake	4E3-009	1	70	3471	8.2	889	113	2.6	682	1050	1.2
No Name	4E3-016	2	21	2848	6.9	363	37	0.1	245	451	3.6
North Catamount Reservoir	4E3-035	85	1585	2848	7.6	418	139	0.4	414	633	1.5

concentrations would be more similar (smaller range of values) if atmospheric deposition was the primary source (e.g., Turk and Spahr 1991). Nitrate concentrations ranged from 0 to 16 μeq/L in these Front Range lakes, with a population-weighted mean of 4 μeq/L. However, the lakes were sampled by WLS in the fall when NO_3^- concentrations are expected to be low relative to concentrations measured during spring snowmelt.

Several studies have addressed the question of how much chronic acidification has occurred to date in Rocky Mountain lakes. Lewis (1982) concluded, based on recent and historic data comparisons, that acidification of some lakes in the Front Range of Colorado was probable because of their proximity to emission sources in Denver. Paleolimnological analyses of four lakes in the Rocky Mountains indicated no recent declines in pH (Charles and Norton 1986). An analysis of precipitation and lake chemistry for lakes in the Mt. Zirkel Wilderness Area located west of the Front Range suggested that the maximum amount of acidification from deposition acidity associated with strong acid anions probably did not exceed 9 μeq/L (Turk and Campbell 1987). It is our judgement that, if chronic acidification of any acid-sensitive lakes in ROMO has occurred to date, such acidification has been small in magnitude.

Musselman et al. (1996) conducted a synoptic survey of surface water chemistry in the mountainous areas east of the Continental Divide throughout the length of Colorado and in southeastern Wyoming that are exposed to increased atmospheric emissions and deposition of N and S. A total of 267 high-elevation lakes in catchments with a high percentage of exposed bedrock or glaciated landscape were selected for sampling. More than 10% of the lakes had ANC < 50 μeq/L. None were acidic (ANC < 0), although several had ANC < 10 μeq/L. The lowest pH was 5.4 at the GLEES research site in Medicine Bow National Forest in southeastern Wyoming. Most lakes had pH > 6.0. Many of the lakes had high NO_3^- concentrations, especially those sampled during the first half of the field study (July and early August). Two lakes also showed high NO_3^- concentrations in September (Figure III-5).

A lake and stream sampling program was conducted by the U.S. Fish and Wildlife Service in four large watersheds of ROMO (Figure III-6; Gibson et al. 1983). The study watersheds included the East Inlet and Upper Colorado River basins on the west side of the Continental Divide, and the Glacier Gorge and Fall River basins on the east side of the Divide. Water samples were collected under base flow conditions, i.e., sampling did not occur within 24 hours after rainstorms. Lake



samples were collected at each inlet, outlet, and lake center location. Stream samples were collected 25 m below each confluence and at 150 m elevation intervals. The lakes and streams were generally low in ionic strength.

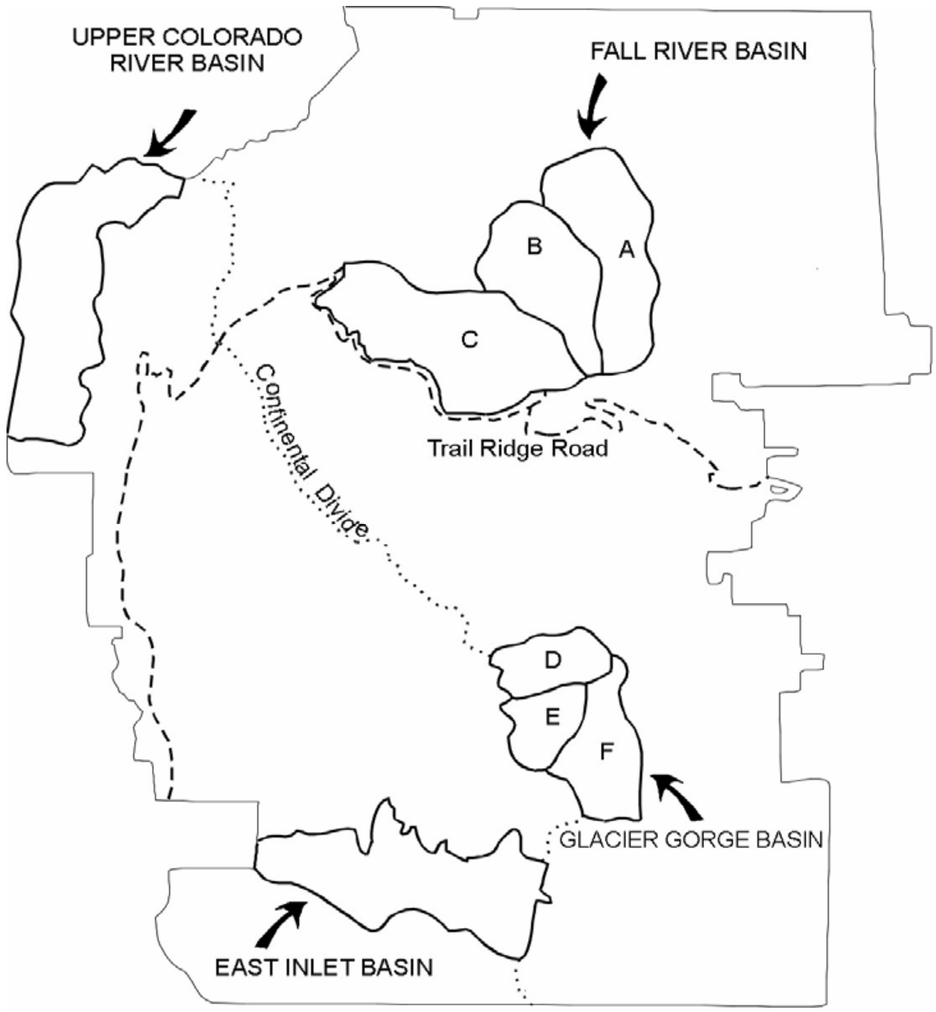


Figure III-5. Nitrate concentration measured in alpine lakes east of the Continental Divide throughout the length of Colorado and in southeastern Wyoming (Mussleman et al. 1996).

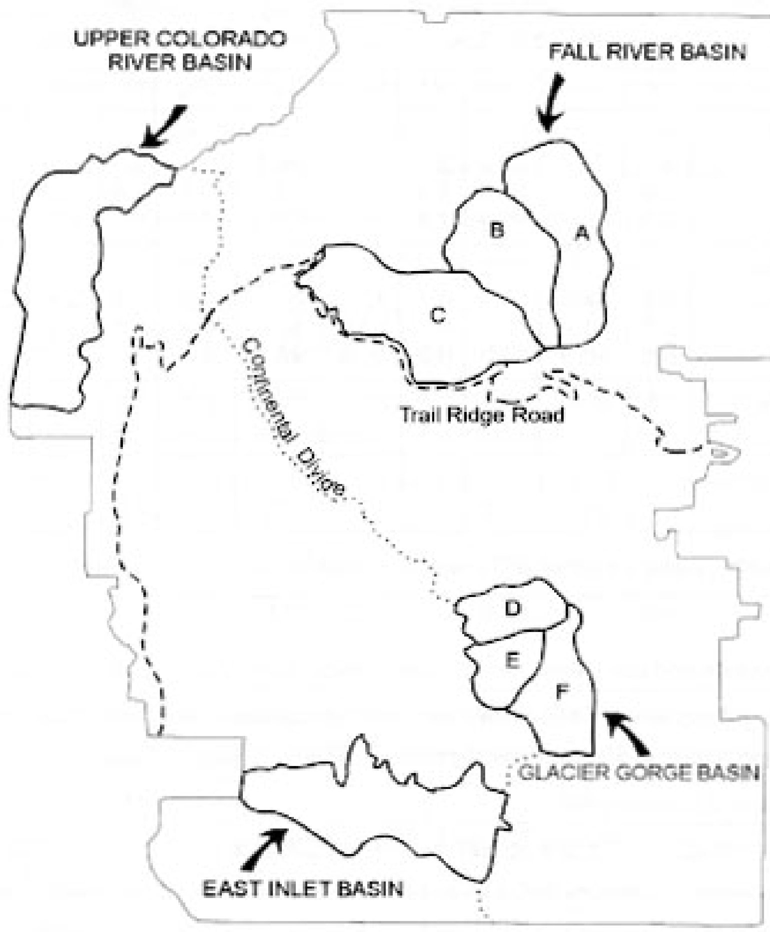


Figure III-6. Watersheds of ROMO studied by the U.S. Fish and Wildlife Service. Subbasins are (A) Roaring River, (B) Ypsilon Creek, (C) Upper Fall River, (D) Tyndall Gorge, (E) Loch Vale, (F) Glacier Creek (Gibson et al. 1983). (Note: park boundary shown does not reflect 1980 boundary change.)

Average pH and ionic concentrations found within the eight study subbasins are reported in Table III-15. Four of the subbasins had average alkalinity values less than 50 μeq/L and two had average alkalinity between 50 and 100 μeq/L, suggesting widespread sensitivity to acidic deposition effects. Each had average pH values in the range of 6.0 to 6.9. Two subbasins (Upper Fall River and Upper Colorado River) had relatively high alkalinity (180 and 332 μeq/L, respectively) and pH

(7.1 and 7.5) and we consider them to be insensitive to acidification effects. Alkalinity, base cation concentrations, and silica were all found to be inversely related to elevation in the subbasins with

Table III-15. Mean ionic concentrations of sampled streams and rivers in ROMO watersheds[a]. (Source: Gibson et al. 1983). See Figure III-6 for watershed locations.

Watershed	pH	Alk	Na^+	K^+	Mg^{2+}	Ca^{2+}	NH_4^+	Cl^-	NO_3^-	SO_4^{2-}	SiO_4
Fall River Basin											
Roaring River	6.9	74.4	29.0	3.6	21.7	66.5	0.1	6.0	8.3	34.7	68.8
Ypsilon Creek	6.6	48.0	22.9	4.1	15.2	43.6	0.1	3.4	9.8	30.2	61.3
Upper Fall River	7.1	180.5	40.2	7.5	67.3	106.9	0.1	7.9	4.9	46.3	110.8
Glacier Gorge Basin											
Andrews Creek	6.5	38.8	16.1	3.7	13.1	55.7	0.0	3.7	12.5	32.3	36.5
Loch Vale	6.0	41.2	15.0	3.1	13.9	52.9	0.4	4.5	17.0	28.2	33.3
Glacier Creek	6.6	40.3	14.2	3.0	10.0	46.3	0.1	3.1	11.3	13.3	32.8
Upper Colorado River Basin	7.5	331.8	34.6	7.6	80.0	234.3	0.3	7.9	6.3	64.3	81.5
East Inlet Basin											
East Inlet	6.8	85.5	26.5	2.5	16.0	90.0	0.1	4.2	5.6	35.7	67.9

[a] All concentrations are in μeq/L, except SiO_4, which is in μM/L

homogeneous mineralogy and low alkalinities (Glacier Creek, Loch Vale, Ypsilon Creek, Roaring River, and East Inlet; Gibson et al. 1983). However, using probability-sampling results from the Western Lakes Survey, Eilers et al. (1988) found little or no relationship between ANC and lake elevation.

The acid-base chemistry of lake and stream waters in ROMO is primarily a function of the interactions among several key parameters and associated processes: atmospheric deposition, bedrock geology, the depth and composition of surficial deposits and associated hydrologic flowpaths, and the occurrence of soils, tundra, and forest vegetation. High concentrations of base cations, alkalinity, and silica occur in the upper Colorado River basin, an area underlain by highly weatherable ash flow tuff and andesite. In contrast, the alkalinity and base cation concentrations are much lower in Glacier Creek, a watershed underlain by Silver Plume granite (Gibson et al. 1983).

In the Roaring River Subbasin, results were reported by Gibson et al. (1983) for 17 samples, which ranged in pH from 6.03 to 7.05 and in alkalinity from 26 to 96 μeq/L. Twenty samples were collected from Ypsilon Creek watershed, with a pH range of 5.63 to 7.00 and an alkalinity range of 16 to 66 μeq/L. Only six samples were collected within Tyndall Gorge watershed; all had pH between 5.61 and 5.81 and alkalinity below 62 μeq/L. Five of the six samples had alkalinity of 24 to 39 μeq/L. All 15 samples from Loch Vale had alkalinity less than 50 μeq/L and pH of 5.9 to 7.0. pH values in Glacier Creek (n=18) ranged from 5.88 to 6.90; alkalinity ranged from 10 to 65 μeq/L. East

Inlet had somewhat higher pH and alkalinity values; most samples ranged in alkalinity from 60 to 100 μeq/L (Gibson et al. 1983).

The study basins and subbasins were ranked in terms of their presumed sensitivity to acidification on the basis of cation concentrations and pH of stream and lake water samples collected in the study areas (Table III-16). The three subbasins that comprise the Glacier Gorge basin (Loch Vale, Glacier Creek, and Tyndall Gorge) and one of the subbasins (Ypsilon Lake Subbasin) within the Fall River basin were consistently ranked by Gibson et al. (1983) as most sensitive to potential effects of acidic deposition. Surface waters in all four of these subbasins had pH between 6.4 and 6.5, calcium concentrations less than 55 μeq/L, and magnesium concentrations less than 13 μeq/L. These were the four subbasins with surface waters lowest in pH and base cation concentrations of the subbasins studied. Three of them (Tyndall Gorge, Loch Vale, and Ypsilon Lake) received a large percentage of their drainage water from snowmelt during summer. Using alkalinity as a criterion of acidification sensitivity, based on the study by Gibson et al. (1983), the Glacier Gorge basin and the two subbasins (Ypsilon Creek and Roaring River) of the Fall River basin are the most sensitive areas of ROMO to potential acid deposition impacts.

Table III-16. Ranking of basins and subbasins studied by Johnson and Herzog (1982) by cation concentrations and pH of water samples.

Calcium (μeq/L)		**Magnesium (μeq/L)**	
Ypsilon Lake	35	Upper Colorado	9
Loch Vale	40	Loch Vale	12
Glacier Creek	52	Glacier Creek	12
Tyndall Gorge	54	Ypsilon Lake	12
Roaring River	67	Tyndall Gorge	13
East Inlet	85	East Inlet	17
Fall River	106	Roaring River	21
Upper Colorado	267	Fall River	66
Box Canyon	303	Box Canyon	109
Sodium (μeq/L)		**pH**	
Loch Vale	13	Loch Vale	6.44
Tyndall Gorge	17	Glacier Creek	6.49
Ypsilon Lake	18	Tyndall Gorge	6.51
Glacier Creek	18	Ypsilon Lake	6.52
East Inlet	27	Roaring River	6.83
Roaring River	27	East Inlet	6.86
Fall River	39	Fall River	7.06
Upper Colorado	39	Upper Colorado	7.56
Box Canyon	81	Box Canyon	7.69

Baron and Bricker (1987) documented episodic pH and ANC depressions during snowmelt in Loch Vale during three successive years, but surface waters did not become acidic (ANC $\leq$ 0). Similarly, Denning et al. (1991) showed a dramatic decline in the ANC of Loch Vale between mid-April and mid- to late-May in 1987 and 1988, to ANC values as low as 28 μeq/L (and pH around 6.2). This is in spite of the fact that meltwater ANC drops to between 0 and -10 μeq/L for extended periods during snowmelt (Denning et al. 1991). If a large component of the snowmelt was transported directly to surface waters, the latter would become acidic during snowmelt. Because surface water does not become acidic or exhibit the low pH of meltwater (often 4.8 to 5.0), direct pathways from the snowpack to the streams are not dominant (Denning et al. 1991) or are offset by more alkaline drainage from watershed soils.

Peak concentrations of nutrients and DOC in surface waters occurred at the beginning of the snowmelt. This indicates that soil solution is flushed into surface water at that time. After the initial flushing, the ionic strength of surface water decreases throughout the melting period due to the dilution of soilwater with the large contribution of meltwater (Denning et al. 1991). The decline in ANC in surface waters is caused by several things, including dilution of base cation concentrations by meltwater, increase in organic acid anions, and increase in NO_3^- concentrations.

Dissolved organic carbon (DOC) concentrations, indicative of organic acidity, are extremely low in most acid-sensitive waters in ROMO. McKnight et al. (in press), in a study of Loch Vale watershed, found lowest DOC (0.4 mg/L) in Sky Pond, an alpine lake that drains a talus slope watershed. Andrews Creek and Icy Brook gained some DOC as they flowed through wet sedge meadows. The subalpine lake, the Loch, receives additional organic material from surrounding forest soils and had the highest DOC (0.7 mg/L).

During snowmelt, peak concentrations of DOC are attained. For example, the DOC in Sky Pond increased to about 1.6 mg/L, likely due to snowmelt flushing through organic-rich soil (Baron et al. 1991). Nevertheless, DOC and organic acid anion concentrations are always low in these sensitive aquatic systems. With respect to our evaluation of potential atmospheric impacts, natural organic acidity is of less importance than in many acid-sensitive regions elsewhere.

Cooper (1990) conducted detailed studies of the hydrology, water chemistry, soils, and vegetation of the Big Meadows wetland complex, located at an elevation of 2,865 m in the southwestern corner of ROMO. The Tonahutu Creek Valley was formed by glaciers eroding bedrock and depositing till. The relatively flat valley floor was created by the filling in of a lake and the deposition of about 45 m of alluvial sediments in the valley center. Peat bodies in Big Meadows are sloping fens that are weakly minerotrophic in early summer when a flush of dilute snowmelt flows through the system. The major inlet to Big Meadows is Tonahutu Creek, which is fed largely by snowmelt runoff and aquifers. The creek water is relatively low in Ca^{2+} concentration (mean of 59 μeq/L in June of 1988 and 70 μeq/L in November of 1988) but ANC is sufficiently high (HCO_3^-

concentrations ranging from 86 to 144 μeq/L during the summer of 1988) that acidification to biologically-important levels is unlikely to occur at any reasonable level of anticipated future deposition.

A great deal of research has been conducted on the interactions between atmospheric pollutants and water quality at the Loch Vale watershed. Biogeochemical and hydrological processes have been studied intensively at this site since 1983 (e.g., Baron 1992, Denning et al. 1991, Campbell et al. 1995, Baron and Campbell 1997). A general description of the watershed is as follows. Loch Vale watershed is a 7-km^2 basin situated along the Continental Divide in the southeastern portion of ROMO. Fifty-five percent of the surface area is exposed bedrock. Twenty-six percent is talus, where large boulders are interspersed with tundra underlain by thin, minimally-developed Entisol soils (Walthall 1985). Alpine tundra covers 11% of the watershed, and the remainder is glaciers and lakes (2%), well-developed subalpine forest soils (5%), and alluvial and bog soils located in saturated areas and adjacent to streams (1%) (Walthall 1985, Baron and Campbell 1997).

Loch Vale is located 80 km northwest of Denver, and ranges in elevation from 3,100 to 4,000 m. The five-year average (1984-1988) total precipitation is 113 cm/yr (Baron 1992). A spruce-fir forest and small subalpine meadows dominate the landscape at the lower elevations on Cryoboralf soils (Arthur and Fahey 1992, Walthall 1985).

A number of factors predispose watersheds in ROMO such as Loch Vale to potential adverse effects of nitrogen deposition. These include:

- Steep watershed gradient
- Short hydrologic residence time of lakewaters
- Large input of N to lakes and streams during the early phases of snowmelt
- High percentage of watershed covered by exposed bedrock and talus; small percentage of watershed covered by forest
- Phosphorus limitation of aquatic ecosystem primary production in some surface waters.

Thus, it is not surprising that the Loch Vale watershed leaches relatively high amounts of NO_3^- under only moderate levels of N deposition. In order to understand the response of this watershed (and other similar watersheds in the park) to atmospheric N deposition, it is important to consider a variety of hydrologic and biogeochemical processes that occur in different parts of the basin. These are described in general terms below.

Campbell et al. (1995) studied the water chemistry of the two major tributaries to the Loch, Andrews Creek and Icy Brook. The catchments for the two streams are entirely alpine, consisting of rock outcrops, talus slopes, and some tundra. Only 5 to 15% of the catchments are covered by well-developed soil. Total storage of soil water was estimated to be less than 5% of the total outflow at the Loch (Baron and Denning 1992). Volume-weighted mean annual concentrations of NO_3^- in the streams were 21 and 23 μeq/L, respectively. Total N export was approximately equivalent to atmospheric inputs, assuming about 25% evapotranspiration. Nitrate concentrations in individual samples ranged from 12 μeq/L in late summer to 39 μeq/L during snowmelt.

The Loch Vale watershed can, for all practical purposes, be considered nitrogen-saturated (e.g., Aber et al. 1989, Stoddard 1994). It is not clear to what extent the terrestrial and aquatic systems are receiving N inputs in excess of the assimilative capacities of watershed biota, however. The apparent N-saturation may be entirely hydrologically-mediated. In other words, hydrologic flowpaths and brief soil water residence times may limit the opportunity for biological uptake to the extent that the ecosystems may be N-limited but still be unable to utilize atmospheric inputs of N (Campbell et al. 1995). Nevertheless, the implications of this apparent N-saturation are important with respect to the estimation of critical loads of N deposition (Williams et al. 1996a). For example, critical loads for N deposition have been estimated to be 10 kg N/ha/yr for northern Europe, based on empirical results that showed little or no N leaching to surface waters below this level (Dise and Wright 1995). Clearly, leaching of NO_3^- to surface waters occurs at much lower levels of N deposition at ROMO and probably at other areas of the Front Range.

Whereas the median lakewater NO_3^- concentration measured in the Western Lake Survey (Landers et al. 1987) was less than 1 μeq/L, the annual mean NO_3^- concentration at the outlet to Loch Vale is 16 μeq/L, and ranges from less than 1 μeq/L in winter to about 31 μeq/L during the peak snowmelt period (Baron 1992). High NO_3^- concentrations have been observed in all lakes and streams in the Loch Vale watershed. Lakewater NO_3^- concentrations of nearby lakes are similar to those in Loch Vale. Baron (1992) reported median NO_3^- concentration for the period 1983-1988 for the Loch, Sky Pond, and Glass Lake of 16, 15, and 13 μeq/L, respectively.

Surface water NO_3^- concentrations are also high in similar terrain outside the park. Nitrate concentrations were measured by Toetz and Windell (1993) in a 13-ha subalpine lake, Lake Albion, located at 3,300 m elevation in the Green Lakes Valley, Colorado Front Range in late June and early July, 1984. Lakewater NO_3^- concentrations were generally near 3 μeq/L, although concentrations in the lake inlet were much higher: 62, 8, and 6 μeq/L, respectively for the 28 June, 5 July, and 12 July sampling dates. The mean pH of Lake Albion was 6.5 (standard deviation 0.28) during the period 1982 through 1987 (Caine and Thurman 1990).

During the last 10 years, the annual minimum concentrations of NO_3^- in surface waters during the growing season have increased from below detection limits to about 10 μeq/L in high-elevation catchments at Niwot Ridge and in GLEES in southeastern Wyoming (Williams et al. 1996b). Wet NO_3^- deposition to adjacent NADP collectors has more or less doubled during that time period at both sites.

Williams et al. (1996b) sampled 53 ephemeral streams during snowmelt runoff in the Green Lakes Valley in 1994 and also sampled an additional 76 sites from the central Colorado Rocky Mountains to the Wyoming border in 1995. Nitrate concentrations in streamwater during snowmelt increased to 44 μeq/L in the Green Lakes Valley and during the growing season increased to 23 μeq/L in the regional sampling conducted in 1995. Landscape type had a significant effect on NO_3^- concentrations ($p < 0.01$) in drainage waters throughout the Colorado Rocky Mountains. Tundra areas had significantly lower NO_3^- concentrations than talus and bedrock areas, suggesting that tundra ecosystems are still N-limited and that nitrification combined with limited plant uptake account for the high concentrations of NO_3^- observed in waters draining talus and bedrock areas (Williams et al. 1996b).

In response to an hypothesized overall pattern of climate change, it has been suggested that high-elevation environments may be expected to experience cooler temperatures and increased precipitation. Such a trend has been observed during the past 45 years at Niwot Ridge (Brooks et al. 1995a, Williams et al. 1996c). Deeper snowpack accumulation and longer period of snowpack cover would be expected to result in warmer soil temperatures and higher rates of subnivian mineralization. This hypothesis was tested by Brooks et al. (1995a) who constructed a 2.6 x 60 m snow-fence at the Niwot Ridge Long-Term Ecological Research site. The fence resulted in a snowpack that was significantly deeper than reference areas and also deeper than at the same area during the previous winter. The average period of continuous snow cover in the main snow drift increased by 115 days compared to reference sites. The deeper and earlier snowpack insulated soils from the extreme ambient air temperatures, resulting in a 9°C increase in minimum soil surface temperature and a 12°C increase in minimum soil temperature at 15 cm depth. Warmer soils contributed to greater microbial activity, measured as CO_2 flux through the snowpack, which continued through much of the winter. CO_2 production was 55% greater than production before construction of the snow fence (Brooks et al. 1995a). Such effects of snowpack are important with respect to N mineralization in alpine and subalpine environments. Soil heterotrophic respiration under seasonal snowpack has been shown to mineralize 20 to 50% of yearly above-ground primary production at alpine and subalpine sites in Wyoming (Sommerfeld et al. 1993). Brooks et al. (1995a) concluded that the timing of snowpack development is the most important factor controlling microbial activity in alpine soils during winter.

Inputs to the soil inorganic N pool at Niwot Ridge due to mineralization and nitrification under deep snowpack (17-20 kg N/ha) were an order of magnitude higher than inputs directly from snowmelt (< 1.5 kg N/ha, Brooks et al. 1995b). Nitrogen mineralization seemed to be a function of the severity of the freeze and the length of time the soils were insulated by snowpack. Mineralization was often higher under deeper, earlier-accumulating snowpacks. Under shallower, late-accumulating snowpacks, N mineralization was lower and also more variable (5 to 15 kg N/ha, Brooks et al. 1995b). The severity with which the soils freeze may also determine the amount of N mineralization. Brooks et al. (1995b) found the highest mineralization inputs under a shallow snowpack that experienced a severe freeze, followed by an extended period of snow cover. Such a result may be attributable to the release of labile carbon and nitrogen compounds from cell membranes that were ruptured by the freeze/thaw process followed by an extended period of mineralization under snowpack (Schimel et al. 1995).

Much of the water that flows into lakes and streams in ROMO first passes through a portion of the watershed and makes contact with soils, talus or exposed bedrock. Interactions between runoff water and these surfaces modifies the runoff water chemistry. Soil solution data from Loch Vale illustrate the differences in N uptake and mobility with landscape type. In forest soil solutions, median concentrations of both NO_3^- and NH_4^+ were < 1 μeq/L and concentrations reached as high as 10 μeq/L when a pulse of N-rich snowmelt water passed through the soils. In contrast, groundwater springs discharging from areas of talus had median NO_3^- concentrations of 40 μeq/L and the highest values approached 200 μeq/L (Campbell et al. 1996). In view of such high concentrations of NO_3^- in drainage from talus fields, Campbell et al. (1996) concluded that the source of much of the inorganic N in surface waters of Loch Vale is likely shallow groundwater that flows through talus. This high-N source mixes with water that has lower concentrations of N, resulting in streamwater with peak NO_3^- concentrations of about 40 μeq/L and that remain above 10 μeq/L throughout the growing season (Campbell et al. 1996). Thus, the sensitivity of alpine and subalpine lakes and streams in ROMO is strongly influenced by the upslope topography.

Baron et al. (1994) simulated nitrogen cycling and key processes in alpine tundra and subalpine forest at Loch Vale for a range of N deposition, using the CENTURY model (Parton et al. 1988,1993; Sanford et al. 1991). The simulated response of the forest system to increased N deposition was more pronounced than the simulated tundra response. Due to the high percentage of tundra compared to forest vegetation in the watershed, however, simulated stream discharge was dominated by leachate from tundra. At deposition below 2 kg N/ha/yr, N losses from tundra were simulated to remain steady at about 1 kg N/ha/yr. Simulated N losses increased as deposition increased above 2 kg N/ha/yr.

Baron and Campbell (1997) developed an annual nitrogen budget for Loch Vale watershed, based on measured, calculated, and model-simulated values for nitrogen inputs, outputs, and internal cycling. They used nine-year average wet deposition values of NO_3^--N (1.6 kg/ha) and NH_4^+-N (1.0 kg/ha) and an assumed ratio of nitrogen dry to wet deposition equal to 0.5 to estimate total average N deposition equal to 3.9 kg/ha. Estimates of nitrogen imports and exports from alpine tundra and subalpine forest were generated using the CENTURY model, a general model of the nutrient dynamics of plant-soil ecosystems (Sanford et al. 1991, Parton et al. 1993). Results of these model simulations were previously published by Baron et al. (1994). An estimated 49% of the

N input was immobilized. Tundra and aquatic algae were the largest reservoirs for incoming N, at 19% and 15% of the total input, respectively. Rocky areas stored an estimated 11% and forests 5% (Baron and Campbell 1997). An estimated 1.7 kg N/ha/yr was lost from the Loch Vale watershed via streamflow. This represents 44% of the estimated total N deposition and 25% of the wet N inputs to the watershed. Baron and Campbell (1997) concluded that the budget calculations suggested that N storage within bedrock areas was significant, accounting for about 10% of total annual N inputs, and that algal N uptake is important to the overall watershed N budget, despite large N fluxes during spring and summer growing seasons.

It is our judgement that the greatest threat from air pollution to aquatic resources in ROMO is nitrogen deposition and consequent lake and stream acidification. Both chronic and especially episodic acidification (loss of ANC) have probably already occurred in some acid-sensitive park waters. However, the magnitude of acidification likely has been small and it has probably not had a significant impact on aquatic biota. There is no evidence that any surface waters in the park have become chronically acidic as a consequence of nitrogen deposition. However, the aquatic resources in portions of ROMO are considered to be at great risk to adverse impacts of atmospheric nitrogen. Continued systematic monitoring of deposition and water quality should be considered high priority activities.

Because of the documented poor N retention capacity of many alpine watersheds in ROMO, we expect that any increase in the atmospheric N load will result in increased concentrations of NO_3^- in alpine and subalpine lakes and streams. If such changes are sufficiently large, surface water acidification, particularly episodic acidification, of aquatic ecosystems will likely occur.

b. Aquatic Biota

Due to the high elevation of much of the park and barriers to fish migration, many of the lakes and streams in the park were historically fishless (Rosenlund and Stevens 1988). Park-wide stocking of both native and non-native fish species continued throughout this century until the 1960's. By 1969, stocking of non-native fish species was abandoned. Lakes not capable of maintaining fish reproduction were allowed to revert to their fishless status, and management emphasis shifted to the restoration of native fish species. Of the 147 lakes within ROMO, about 59 lakes had fish populations maintained by natural reproduction or stocking in 1969. By 1987, that number had dropped to 49 lakes, of which 18 were populated with pure strains of native fish (e.g., the threatened greenback cutthroat trout [*Onchorhynchus clarkii stomias*]) in their native drainages (Rosenlund and Stevens 1988).

The only trout native to the park were the greenback cutthroat and the Colorado River cutthroat (*O. c. pleuriticus)*. Yellowstone cutthroat (*O.c. lewisi*), brown (*Salmo trutta*), brook (*Salvelinus fontinalis*), and rainbow trout (*O. mykiss)* have all been stocked to some degree in park waters.

Since 1975, the National Park Service has been removing non-native fish from some park waters. Catch-and-release fishing only is permitted for the native greenback cutthroat.

Cold water and lack of suitable spawning habitat limit fish reproduction in many of the high-elevation lakes. Many lakes potentially susceptible to adverse impacts from atmospheric inputs of N or S either have no fish at present or were historically fishless. The potential adverse impacts of air pollutants on fisheries must be evaluated within the context of probable native fish distributions.

Several of the drainages in ROMO thought to be highly sensitive, based on lakewater ANC and pH (Gibson et al. 1983), to potential adverse impacts of acidic deposition do support viable fish populations. Ypsilon Lake supports a virtually pure population of Colorado River cutthroat trout, although it is located in the South Platte drainage, rather than the Colorado River drainage (they were planted there). Loch Vale and Glass Lake support hybrid greenback cutthroat trout). Icy Brook and Sky Pond support brook trout (*Salvelinus fontinalis)*. Glacier Creek supports brook trout in the upper reaches and cutthroat and rainbow trout in the middle and lower reaches (Rosenlund and Stevens 1988).

The boreal toad (*Bufo boreas*) has experienced recent widespread decline throughout the southern Rocky Mountains since about 1975 (Corn et al. 1989, Carey 1993). Leopard frogs (*Rana pipiens*) have also declined in Colorado and Wyoming (Corn and Fogleman 1984, Corn et al. 1989). Harte and Hoffman (1989) hypothesized that episodic acidification was the principal cause of the decline of at least one amphibian species in Colorado, the tiger salamander (*Ambystoma tigrinum*). If episodic acidification is an important factor with respect to amphibian decline in the Rocky Mountains, then two conditions must be met: episodic acidification to toxic levels (of pH, Al, etc.) must occur, and sensitive life stages of the amphibian species must also be present at the time of the episodic acidification. Contrary to the situation in eastern North America, where spring and summer rainstorms are the dominant hydrological event that influences the chemistry of amphibian breeding habitats (Freda et al. 1991), episodic acidification events in ROMO occur primarily during early snowmelt in the spring. The life history strategies of most amphibian species make them unlikely to be exposed to acidification during snowmelt (Corn and Vertucci 1992, Vertucci and Corn 1996). For example, direct mortality of *Bufo boreas* embryos from exposure to low pH is unlikely in ROMO because most snowmelt and associated pH depression occurs prior to egg deposition (Vertucci and Corn 1996). In addition, *Rana pipiens* generally occupies lakes at lower elevations that tend to be insensitive to episodic acidification (Corn and Vertucci 1992). Vertucci and Corn (1996) concluded that there is no evidence that episodic acidification has led to acidic conditions in the Rocky Mountains or that amphibian embryos are present during the initial phases of snowmelt when episodic acidification might occur.

Quite different conclusions were reached by Kiesecker (1996) and Turk and Campbell (1997). In the area around Dumat Lake, just south of the Mt. Zirkel Wilderness Area, 60% to 70% of tiger salamander eggs were dead or unviable in ponds at about pH 5.0 or less, about 40% in ponds at pH between 5.0 and 6.0, and about 20% at about pH 6.0 or greater (Kiesecker 1991). Turk and Campbell (1997) used bulk snowpack acidity data from Buffalo Pass, adjacent to the Mt. Zirkel Wilderness Area, and measured amplication factors from an acid pulse during snowmelt at Loch Vale to predict the acidity of meltwater at Buffalo Pass. Their estimates of snowmelt acidity were high (e.g., pH less than 5.0) for a large portion of the snowmelt (Figure III-7). It is important to note, however, that the greatest concentration of acidity in snowpack within the Rocky Mountains appears to occur in and near the Mt. Zirkel Wilderness Area (Turk and Campbell 1997), where this study was conducted.

The extent to which acidic snowmelt influences the pH of amphibian breeding habitat in ROMO remains uncertain. We would not expect such high levels of acidity pulses as were estimated by Turk and Campbell (1997) for Buffalo Pass simply because the bulk snowpack at ROMO seems to be higher in pH. In the absence of more detailed surveys of the chemistry of known or suspected amphibian breeding habitat within the park, we are not able to draw any firm conclusions at this time regarding potential effects on amphibians.

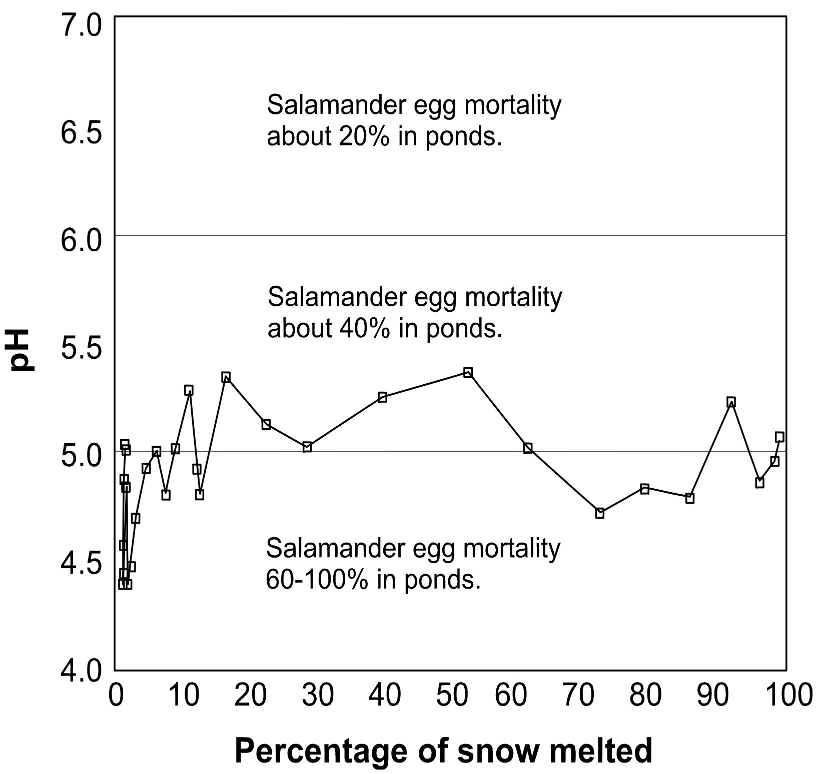


Figure III-7. Predicted acidity (expressed as pH) of snowmelt at Buffalo Pass, adjacent to the Mt. Zirkel Wilderness Area, compared with expected salamander egg mortality at various pH values, from Kiesecker (1991). (Source: Turk and Campbell 1997)

Toetz and Windell (1993) evaluated the status of phytoplankton with respect to lake acidification in Lake Albion in the Green Lakes Valley. Plankton samples were collected on six occasions during summer, 1984. Diatoms were identified and assigned to pH tolerance categories, based on literature values. Toetz and Windell (1993) concluded that the diatom flora was comprised primarily of alkaliphilic and pH-indifferent species. Only 8% of the species with known pH tolerance were considered acidiphilic. This suggests that the phytoplankton community in Lake Albion has not experienced a significant impact from acidic deposition.

Uptake of nutrients by phytoplankton in Sky Pond and Glass Lake, two mainstem lakes on Icy Brook in the Loch Vale watershed, caused NO_3^- concentrations to be lower in Icy Brook than in Andrews Creek during May, despite similar discharge levels (Figure III-8). Phytoplankton blooms beneath ice cover have been documented in the Loch (Spaulding et al. 1992), and diatoms were found to be present in large numbers in Sky Pond at the beginning of snowmelt (McKnight et al. 1986). It is therefore possible that primary production in these lakes has been increased to some extent by atmospheric N deposition.

3. Vegetation

The greatest threat to vegetation in ROMO is ozone pollution from urban areas southeast of the park and from valley and foothill areas where ozone is synthesized in transit from local sources of NO_x and VOCs. There has been one exceedence of the NAAQS in the park (in 1993), and with expected regional population growth and suburban development, ozone levels could increase. Vegetation is the resource which is most sensitive to ozone (based on current knowledge of ozone-sensitive organisms), and several tree species have been identified as potential bioindicators (see below).

One of the most ozone-sensitive western tree species is ponderosa pine (especially var. *ponderosa*), for which extensive data are available on field (Miller and Millecan 1971, Pronos and Vogler 1981, Peterson and Arbaugh 1988) and experimental (Temple et al. 1992) exposures. The evidence for ozone impacts on ponderosa pine is based on observable symptoms of foliar chlorosis and reduced growth (Peterson et al. 1991, Peterson and Arbaugh 1992) as well as physiological (Darrall 1989, Bytnerowicz and Grulke 1992) data. The cause-and-effect relationship, especially for trees growing in forests of southern California and the southern Sierra Nevada, is clear and quantifiable.

The Rocky Mountain variety of ponderosa pine (var. *scopulorum*) is known to be somewhat more tolerant to ozone and has a higher threshold for symptoms of injury under experimental exposures than var. *ponderosa* (Aitken et al. 1984). In 1980, the Forest Service conducted a survey of ponderosa pine in the Front Range west of Denver in order to determine if any trees had

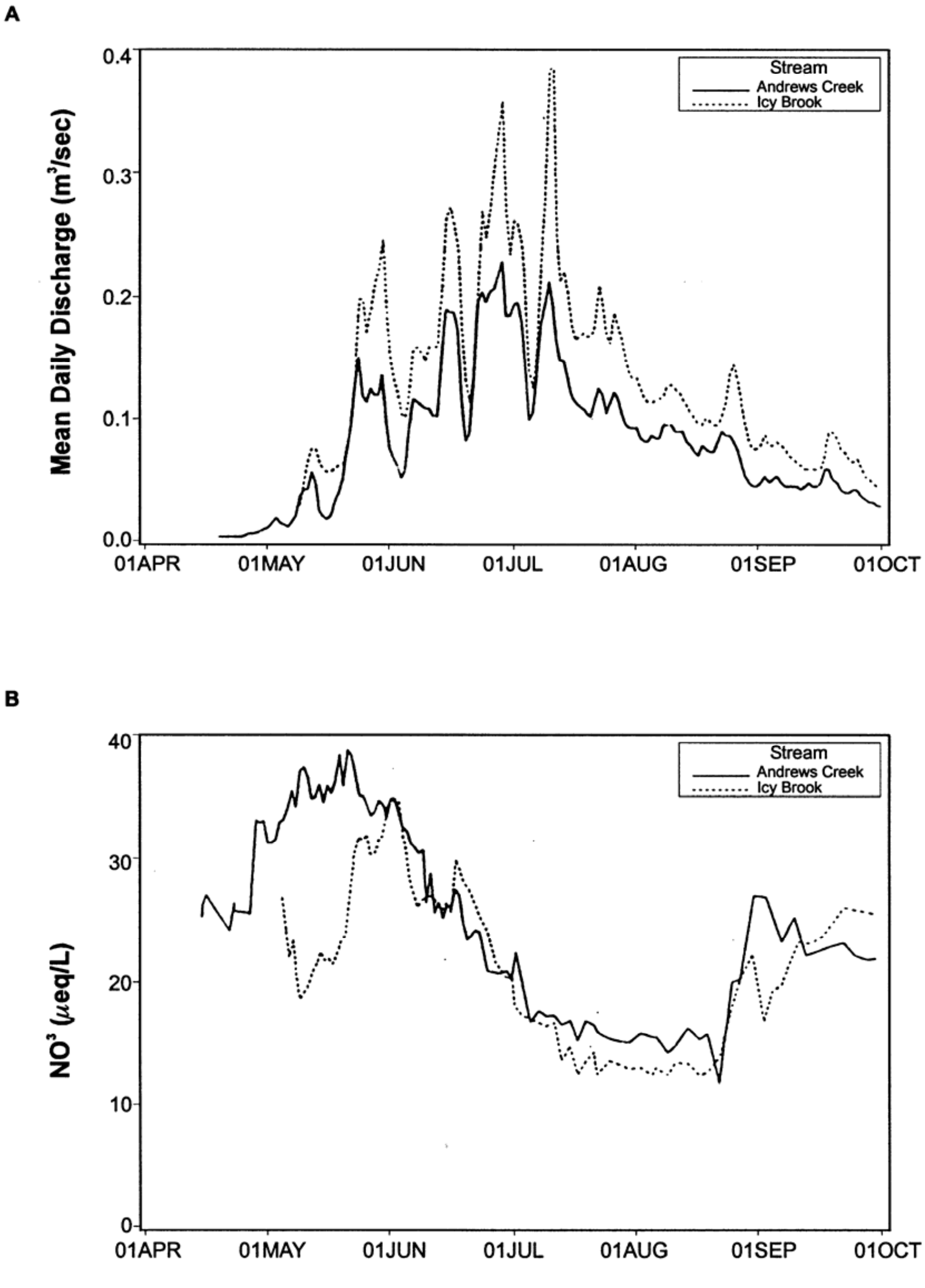


Figure III-8. Daily discharge (A) and nitrate concentration (B) in Icy Brook and Andrews Creek within the Loch Vale watershed in April-September 1992 (Campbell et al. 1995).

evidence of ozone injury. No symptoms were found at that time (James and Staley 1980). In 1987, the National Park Service conducted an extensive survey of ponderosa pine pathological condition in ROMO, with data collected at plots throughout the range of the species in the park (Stolte 1987). No symptoms of ozone injury were noted in any trees in this survey. This was surprising because ozone exposures equivalent to those measured at ROMO would have been expected to result in at least minor symptoms in ponderosa pine in California. Similarly, a study of ponderosa pine at 30 stands throughout the Front Range (20 stands east side, 10 west of the Rampart Range with presumed lower ozone) determined that there were no visible symptoms of ozone injury at any locations and that long-term growth was unaffected by recent elevated ozone levels (Graybill et al. 1993, Peterson et al. 1993). Needle retention was slightly less in trees directly west of Denver where ozone concentration was presumed to be highest.

It is unclear whether the surprising lack of symptoms found in ponderosa pine in the Front Range (compared to California) is due to insufficiently phytotoxic levels of ozone or simply due to the greater tolerance of var. *scopulorum*. Nevertheless, the well-documented and quantifiable symptomatology of ponderosa pine makes it a good sensitive receptor for ozone, even if var. *scopulorum* has lower sensitivity. Furthermore, this species is locally common on the east side of ROMO where ozone concentrations are highest.

Quaking aspen, an ozone-sensitive hardwood species, grows at various locations in riparian ecosystems and in fire- or avalanche-disturbed areas in ROMO. Numerous studies have documented the sensitivity of this species to ozone under field and experimental 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 aspen includes chlorosis, stippling, necrotic spotting, and leaf margin burn. Symptoms generally vary seasonally, with stippling being most prominent in the spring and black, bifacial (both leaf surfaces) necrosis appearing in late summer (J.P. Bennett, pers. comm.). Great care must be taken in distinguishing ozone symptoms from various pathogens and insect herbivores commonly found on this species. Black cottonwood (*Populus trichocarpa*) is another potential bioindicator for ozone (Woo 1996) which has symptoms similar to those of aspen. However, it is generally regarded as less sensitive to ozone than aspen (Table III-17). Neither of these hardwood species has the clarity of ozone symptomatology found in ponderosa pine.

Aspen is also considered to be sensitive to SO_2 (Table III-17) and may be the best bioindicator for this gaseous pollutant. Injury is similar to that normally found for ozone (stippling, followed by bifacial necrosis), although SO_2-induced injury rapidly bleaches to a light tan color (ozone injury remains dark) (Karnosky 1976). There could be some confusion of ozone injury and SO_2 injury, although given current concentrations of these pollutants at ROMO, it is more likely that ozone injury would be detected.

Table III-17. Vascular plant species of ROMO with known sensitivities to sulfur dioxide, ozone and nitrogen oxides. (L = low, M = medium, H = high, blank = unknown). (Sources: Esserlieu and Olson 1986; Bunin 1990; USDA Forest Service 1993; Eilers et al. 1994; Electric Power Research Institute 1995; Binkley et al. 1996; Brace and Peterson 1996,1998)

Species Name	SO_2 Sensitivity	O_3 Sensitivity	NO_x Sensitivity
Abies lasiocarpa	L	L	
Acer glabrum	H		
Agoseris glauca	M		
Alnus tenuifolia	H		
Amaranthus retroflexus	M		
Amelanchier alnifolia	H	M	
Angelica pinnata		L	
Arctostaphylos UVa-ursi	L	L	
Artemisia ludoviciana	M		
Artemisia tridentata	M	L	
Betula occidentalis	M		
Bouteloua gracilis	L		
Bromus tectorum		M	
Ceanothus velutinus	L		
Cercocarpus montanus	M		
Chenopodium fremontii		L	
Cirsium arvense		L	
Cirsium undulatum	M		
Clematis ligusticifolia	M		
Collomia linearis		L	
Conium maculatum		L	
Convolvulus arvensis	H		
Crataegus chrysocarpa	L		
Epilobium angustifolium		L	
Erigeron peregrinus		L	
Erodium cicutarium	L	M	
Fragaria virginiana		H	
Gentiana amarella		M	
Geranium richardsonii	M	M	
Hackelia floribunda	L		
Helianthus anuus	H	L	
Juniperus communis	L		
Juniperus scopulorum	L		
Lemna minor	L		
Lolium perenne		M	
Mahonia repens	L	L	
Mimulus guttatus		L	
Oryzopsis hymenoides	M		
Physocarpus monogyna		H	
Picea engelmannii	M	L	
Picea pungens	M	L	
Pinus contorta	M	M	H
Pinus flexilis	L		

Pinus ponderosa	M	H	H
Poa annua	H	L	
Poa pratensis		L	

Table III-17. Continued.

Species name	SO_2 sensitivity	O_3 sensitivity	NO_x sensitivity
Polemonium foliosissimum		L	
Polygonum douglasii		L	
Populus angustifolia	M		
Populus balsamifera	M	H	
Populus trichocarpa		M	
Populus tremuloides	H	H	
Potentilla fruticosa		L	
Prunus virginiana	M	H	
Pseudotsuga menziesii	M	L	H
Rosa woodsii	M	L	
Rubus idaeus	H		
Rumex crispus		L	
Salix scouleriana		M	
Shepherdia canadensis	L		
Sorbus scopulina	M		
Taraxacum officinale		L	
Thalictrum fendleri		L	
Tragopogon dubius	M		
Trifolium pratense	L		
Trifolium repens		H	
Trisetum spicatum	M		
Urtica gracilis		L	
Viola adunca		L	

An inventory of vascular plants found in ROMO was compiled in 1988 and is available in the NPFlora database. Table III-17 summarizes vascular plant species of ROMO with known sensitivity to ozone, SO_2, and NO_x. This table is based on a variety of sources from the published literature and other information. It should be noted that the various studies used a wide range of field and experimental approaches to determine pollutant pathology, and that sensitivity ratings are general estimates based on published information and our expert opinion. While it will not be possible for park staff to collect data on all the species indicated in Table III-17, the list can be used by park managers as a preliminary list of potential bioindicator species. Of the many plant species in ROMO, it is likely that there are many other species which have high sensitivity to air pollution, but we currently have no information about them.

An inventory of lichen species found in ROMO was compiled by Wetmore and Bennett (unpublished data). Table III-18 summarizes lichen species of ROMO with known sensitivity to ozone and SO_2. As in Table III-17, this table is based on a variety of sources from the published literature and other information. It should be noted that diagnostic symptoms of air pollutant injury to lichens are difficult to identify, and that some species have reduced productivity or even mortality without exhibiting visible symptoms (Nash and Wirth 1988). One of the best sources of background information and guidelines for addressing the use of lichens as sensitive receptors of air pollution is Stolte et al. (1993).

Table III-18 Lichen species of ROMO with known sensitivities to ozone and SO_2.(L = low, M = medium, H = high, blank = unknown). (Sources: USDA Forest Service 1993, National Park Service 1994, Electric Power Research Institute 1995, Binkley et al. 1996)

Species Name	Ozone Sensitivity	SO_2 Sensitivity
Acarospora chlorophana		H
Biatoria vernalis		M
Buellia punctata		L-M
Caloplaca holocarpa		M
Candelariella vitellina		M
Candelariella xanthostigma		M
Evernia divaricata		
Parmelia sulcata	M-H	L-H
Peltigera didactyla	H	
Phaeophyscia ciliata	M	
Physcia adscendens		M
Xanthoria fallax		M-H
Xanthoria polycarpa	L	M

The potential impacts of N deposition on terrestrial resources within ROMO is also an important concern, but unfortunately one about which we can conclusively say little at this time. The major issues are, in our view, the following: 1) "terrestrial eutrophication", whereby excess fertilization leads to increased ecosystem productivity, increased spread of exotic plant species, and decreased native plant species diversity (e.g., Huston 1994); 2) nitrogen saturation, whereby N supply exceeds the vegetative uptake capacity and NO_3^- leaches out of the soil in high concentrations, and 3) soil acidification, which can cause high concentrations of dissolved inorganic Al in soil solution, which can be toxic to plant roots, and also contribute to base cation depletion from the soils. The first two issues may be important at ROMO, as discussed below, but we have insufficient information upon which to base any robust conclusions. The third issue, soil acidification, is unlikely unless deposition of N and/or S was to increase dramatically.

There is some concern that increased N loading might contribute to shifts in plant species composition and species diversity, particularly of alpine and subalpine ecosystems (which receive the highest N deposition) within the park. Because N is the primary nutrient limiting terrestrial productivity, N addition is believed to cause such changes in terrestrial ecosystems (Tilman 1988, DeAngelis 1992). We are not aware of any research, however, that has conclusively documented such changes either in high-elevation ecosystems specifically or in any ecosystems receiving N deposition comparable to ROMO.

Because of inherent differences among plant species, the relative growth of each depends on the relative abundance of different critical resources, particularly of those resources most likely to be limiting. A growth rate advantage for a particular species in response to critical resource abundance may in some cases result in long-term dominance by that species. This is the non-equilibrium interpretation of the observed patterns of species composition that have been attributed to an equilibrium balance of inversely related competitive abilities for two or more resources which are hypothesized to produce succession as a consequence of sequential competitive equilibria (Tilman 1982,1985; Huston 1994). Each species has its own requirements and optima for the factors that determine how well it will perform under any set of environmental conditions. There is rarely a uniform acceleration of growth by different plant species in response to resource enrichment because fast-growing species generally show greater response to growth stimulus, such as fertilization or increased moisture, than do slow-growing species (Chapin et al. 1986). A disproportionate increase in the growth rate of the fastest growing species results in relatively greater dominance by those species (Huston 1994).

Wedin and Tilman (1996) presented results of 12 years of experimental N addition to 162 grassland plots in Minnesota. N loading dramatically changed plant species composition, decreased species diversity, and increased aboveground productivity in their plots. Species richness declined by more than 50% across the N-deposition gradient, with the greatest losses at 10 to 50 kg N/ha/yr.

This loss of species diversity was accompanied by large changes in plant species composition with C_4 grasses declining and the weedy Eurasian C_3 grass *Agropyron repens* becoming dominant at high N addition rates (Wedin and Tilman 1996). The authors concluded that N loading is a major threat to grassland ecosystems and causes loss of diversity, increased abundance of non-native species, and the disruption of ecosystem functioning. It is entirely feasible that such impacts might also apply to alpine and subalpine ecosystems, such as found in ROMO, as well. A major uncertainty, however, is the rate of N loading at which such changes may be manifested. N loading to ROMO ecosystems is about an order of magnitude lower than the loading rates used in the experimental approach of Wedin and Tilman (1996). It is our judgement that N loading at ROMO may indeed be of concern with respect to plant species diversity, especially if N loading rates were to increase dramatically in the future. We find no basis, however, for concluding that current N deposition rates are causing such effects.

It is clear that some high-elevation watersheds in ROMO are N-saturated in a functional sense. In other words, a relatively high proportion of incoming N is not taken up by terrestrial biota, but rather leaches to surface waters (e.g., Stoddard 1994). However, it is likely that this apparent N-saturation in ROMO is hydrologically, rather than biologically mediated, and that soils in these watersheds still have the capacity to retain some additional incoming N. Because such a high percentage of these high elevation watersheds are covered by exposed bedrock and talus, areas which are generally lacking in soils, the watersheds may leach much of the N inputs even though the soils within the watersheds retain most N that is deposited to the soil surface. We have no data that suggest that soils per se in ROMO are currently N-saturated.

4. Visibility

As part of the IMPROVE network, visual air quality in ROMO, has been monitored using an aerosol sampler, transmissometer, and camera. The aerosol sampler began operation in March 1988 and continues to operate in its second, now permanent, location west of Highway 7 approximately 1 mile north of the Longs Peak trailhead. The transmissometer has operated from November 1987 through the present at the Many Parks Curve parking area, approximately 5 miles west of Estes Park. The 35mm camera operated from October 1985 through March 1995 at the same Many Parks Curve parking location. Data from this IMPROVE site have been summarized to characterize the full range of visibility conditions for the March 1988 through February 1995 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 1994 includes March 1994 through February 1995). 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

IMPROVE aerosol samplers consist of four separate particle sampling modules that collect 24-hour filter samples of the particles suspended in the air. The filters are then analyzed in the laboratory to determine the mass concentration and chemical composition of the sampled particles. Particle data can be used to provide a basis for inferring the probable sources of visibility impairment. Practical considerations limit the data collection to two 24-hour samples per week. (Wednesday and Saturday from midnight to midnight). Detailed descriptions of the aerosol sampler, laboratory analysis, and data reduction procedures used can be found in the draft 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. The extinction coefficient represents the ability of the atmosphere to scatter and absorb light. Higher extinction coefficients signify lower visibility. A tabular and graphic summary of average reconstructed extinction values by season and year for the March 1988 through February 1995 period are provided in Table III-19 and Figure III-9, respectively.

Table III-19. Seasonal and annual average reconstructed extinction (Mm^{-1}) and standard visual range (km), ROMO, Colorado, March 1988 through February 1995.

YEAR	Spring (Mar, Apr, May)		Summer (Jun, Jul, Aug)		Autumn (Sep, Oct, Nov)		Winter (Dec, Jan, Feb)		Annual (Mar - Feb)[a]	
	b_{ext} (Mm^{-1})	SVR (km)	b_{ext} (Mm^{-1})	SVR (km)	b_{ext} (Mm^{-1})	SVR (km)	b_{ext} (Mm^{-1})	SVR (km)	b_{ext} (Mm^{-1})	SVR (km)
1988	31.7	123	40.1	98	27.4	143	26.7	147	31.2	125
1989	34.9	112	43.4	90	29.5	133	24.3	161	32.9	119
1990	35.4	111	41.0	95	30.3	129	24.8	158	32.6	120
1991	33.6	116	39.7	99	29.5	133	19.3	203	30.3	129
1992	34.8	112	38.8	101	32.3	121	23.9	164	32.3	121
1993	32.4	121	39.1	100	28.0	140	21.5	182	30.0	129
1994	35.3	111	40.3	97	28.9	135	21.9	179	31.6	124
Mean[b]	34.0	115	40.3	97	29.4	133	23.2	169	31.7[c]	123[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 1995 period.
[c] Combined annual period data represent the mean of all combined season means.

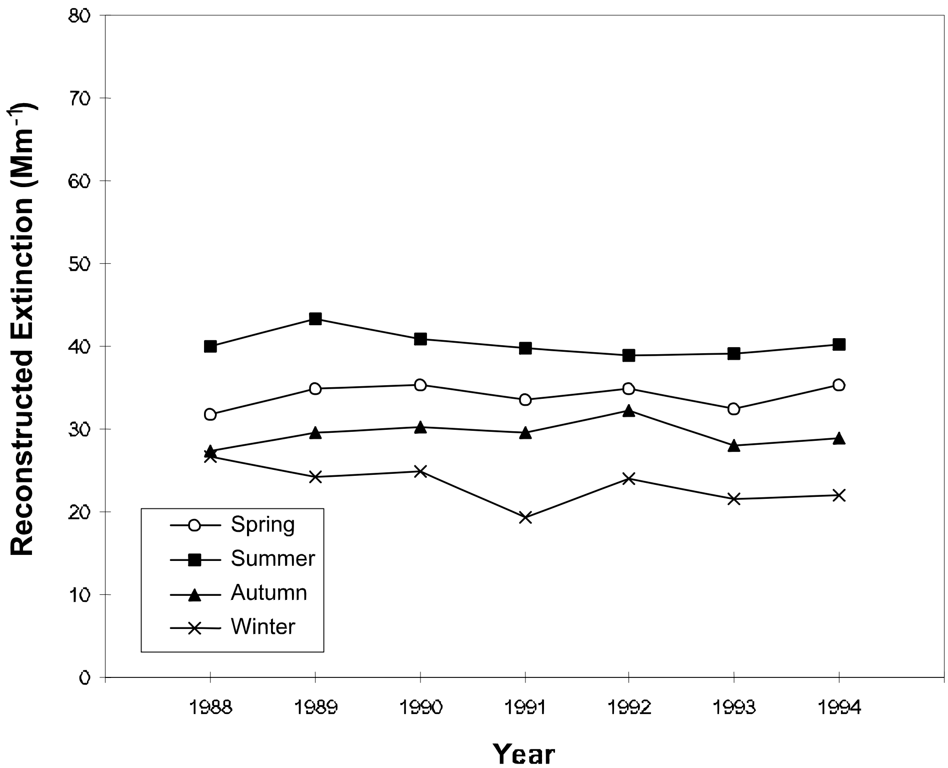


Figure III-9.Seasonal average reconstructed extinction (Mm^{-1}) for the period March 1988 through February 1995.

Reconstructed extinction budgets generated from aerosol sampler data apportion the extinction at ROMO to specific aerosol species (Figure III-10). 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% of days, mean of the median 20% of days, and mean of the dirtiest 20% of days. 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, visual range (in kilometers), and deciview (dv). Standard Visual Range (SVR) can be expressed as:

$$SVR\ (km) = 3{,}912 / (b_{ext} - b_{Ray} + 10)$$

where b_{ext} is the extinction coefficient expressed in inverse megameters (Mm^{-1}), b_{Ray} is the site specific Rayleigh values (elevation dependent), 10 is the Rayleigh coefficient used to normalize visual range, and 3,912 is the constant derived from assuming a 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.

Deciview is defined as:

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

where b_{ext} is the extinction coefficient expressed in inverse megameters (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 Rayleigh atmosphere and increases as visibility is degraded. The segment at the bottom of each stacked bar 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.

The 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. 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.

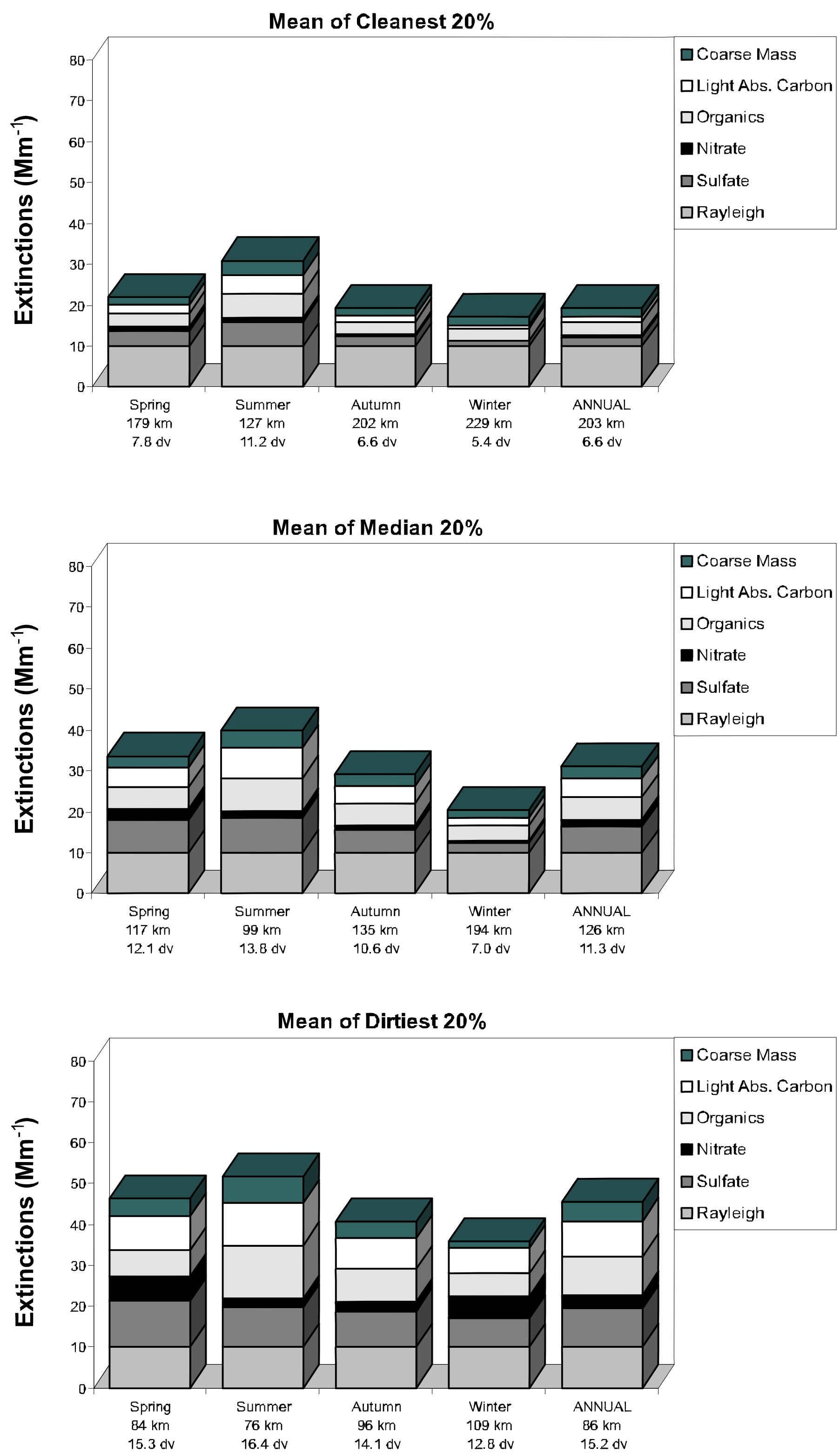


Figure III-10. Reconstructed extinction budgets for Rocky Mountain National Park, Colorado, March 1988 - February 1995.

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. 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, 1993 and 1994).

Table III-20 provides a tabular summary of the “filtered” seasonal and combined period arithmetic mean extinction values. Table III-21 provides a tabular summary of the "filtered" seasonal and combined period 10% (clean) cumulative frequency values. 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 February 1995 period.
- Combined annual period data represent the unweighted mean of all combined seasonal b_{ext} values.

Figure III-11 provides a graphic representation of the "filtered" annual mean, median, and cumulative frequency values (5th, 10th, 25th, 75th, 90th, and 95th percentiles). No data are reported for annual periods with one or more invalid seasons.

When comparing reconstructed (aerosol) extinction, Table III-19, with measured (transmissometer) extinction, Table III-20, 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.

Table III-20. Seasonal and Annual Arithmetic Means, ROMO, Colorado Transmissometer Data (Filtered), March 1988 through February 1995.

Year	Spring (Mar, Apr, May)			Summer (Jun, Jul, Aug)			Autumn (Sep, Oct, Nov)			Winter (Dec., Jan, Feb.)			Annual (Mar – Feb)[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	130	29	10.6	--	--	--	--	--	--	216	17	5.3	***	***	***
1989	144	26	9.6	105	36	12.8	150	25	9.2	185	20	6.9	150	27	9.8
1990	139	27	9.9	139	27	9.9	134	29	10.6	--	--	--	***	***	***
1991	150	25	9.2	111	34	12.2	118	32	11.6	177	21	7.4	143	28	10.3
1992	122	31	11.3	111	34	12.2	130	29	10.6	--	--	--	***	***	***
1993	--	--	--	130	29	10.7	150	25	9.2	194	19	6.4	***	***	***
1994	139	27	9.9	144	26	9.6	162	23	8.3	194	19	6.4	169	24	8.6
Mean[b]	137	28	10.1	122	31	11.3	138	27	10.0	192	19	6.5	153	26[c]	9.6

--No data are reported for seasons with <50% valid data.

*** No annual data are reported for periods with one or more invalid seasons.

Table III-21. Seasonal and Annual 10% (Clean) Cumulative Frequency Statistics, ROMO Transmissometer Data (Filtered), March 1988 through February 1995.

Year	Spring (Mar, Apr, May)			Summer (Jun, Jul, Aug)			Autumn (Sep, Oct, Nov)			Winter (Dec., Jan, Feb.)			Annual (Mar – Feb)[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	185	20	6.9	--	--	--	--	--	--	298	12	1.8	***	***	***
1989	216	17	5.3	162	23	8.3	243	15	4.1	259	14	3.4	236	17	5.5
1990	243	15	4.1	228	16	4.7	194	19	6.4	--	--	--	***	***	***
1991	259	14	3.4	169	22	7.9	162	23	8.3	228	16	4.7	216	19	6.3
1992	162	23	8.3	139	27	9.9	177	21	7.4	--	--	--	***	***	***
1993	--	--	--	177	21	7.4	216	17	5.3	228	16	4.7	***	***	***
1994	185	20	6.9	259	14	3.4	243	15	4.1	259	14	3.4	259	16	4.5
Mean[b]	203	18	6.0	181	21	7.2	201	18	6.1	252	14	3.6	227	18[c]	5.8

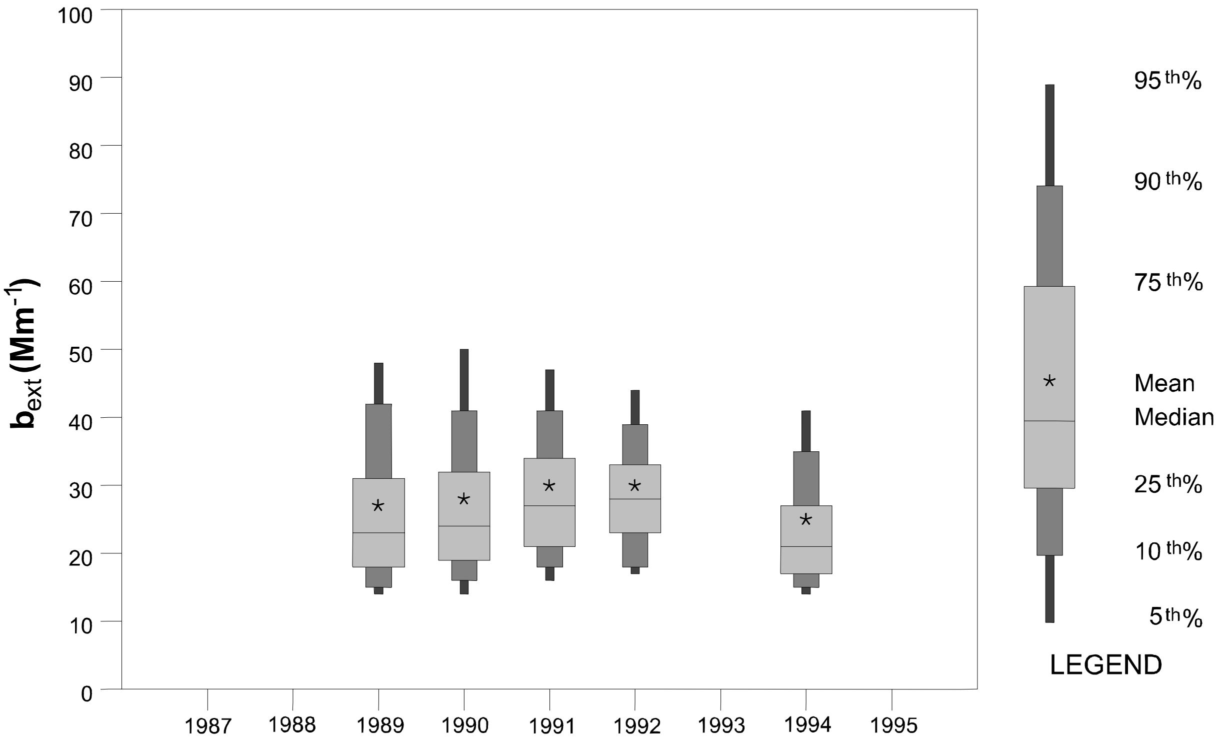
--No data are reported for seasons with <50% valid data.

*** No annual data are reported for periods with one or more invalid seasons.

[a] Annual data represent the mean of all valid seasonal b_{ext} values for each March through February annual period.

[b] Combined season data represent the mean of all valid seasonal b_{ext} values for each season of the March 1988 through February 1995 period.

[c] Combined annual period data represent the mean of all combined seasonal b_{ext} values.



Note: For a specific year to be plotted, at least 50% of the data for each season must be valid.

Figure III-11. Annual arithmetic mean and cumulative frequency statistics, Rocky Mountain National Park, Colorado, transmissometer data (filtered).

· <u>Relative Humidity (RH) Cutoff</u> - Daily average reconstructed measurements are flagged as invalid when the daily average RH is greater than 98%. Hourly average transmissometer measurements are flagged invalid when the hourly average RH is greater than 90%. These flagging methods often result in data sets that do not reflect the same period of time, or misinterpret short-term meteorological conditions.

Note: The weather algorithm only flags 10%-20% of the data for a majority of the sites west of the Mississippi River. RH cutoffs have little effect on final mean extinctions in the western United States.

Reconstructed extinction is typically 70%-80% of the measured extinction. With a ratio of 82%, this relationship shows good agreement for ROMO, Colorado.

c. Camera Data - View Monitoring

An automatic 35mm camera system operated at ROMO from October 1985 through March 1995. Color 35mm slide photographs of Elk Ridge were taken three times per day until Spring 1988. The camera alignment changed at that time to photograph Longs Peak until March 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 Elk Ridge vista photographs presented in Figure III-12 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. Visibility Summary

Data from other IMPROVE visibility sites around the country have been presented graphically (Figures I-1 and I-2) so that visual air quality in the Rocky Mountains and Northern Great Plains regions can be understood in perspective. Figure III-9 and Table III-20 have been provided to summarize ROMO visual air quality during the March 1988 through February 1995 period. Long-term trends fall into three categories: increases, decreases, and variable. Using the visibility sites summarized for this report, the majority of data show little change or trends.

Non-Rayleigh atmospheric light extinction at ROMO, unlike many rural western areas, can have a large nitrate component during the winter and spring when the poorest visibility occurs. However, at other times, like in most areas, atmospheric light extinction is typically associated with sulfate, organics, and soil. 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 pre-settlement visibility below current levels during some summer months.

The IMPROVE aerosol monitoring network, established in March 1988, consists of sites instrumented with aerosol sampling modules A through D. Many of the IMPROVE sites are successors to sites where aerosol monitoring with stacked filter units (SFU) was carried out as early as 1979. SFU data were collected at ROMO from September 1979 through December 1987. Although no long-term trends are apparent in the 1988 through 1994 site-specific or regional data presented in this report, Sisler et al. (1996) provides a review of SFU data combined with IMPROVE sampler data for evidence of temporal trends in aerosol concentrations. Below are excerpts of their findings for the ROMO site.

Rocky Mountain National Park
on a "clear" day

Representative Conditions:
Visual Range: 300-360 km
b_{ext}: 13-11 Mm^{-1}
Haziness: 3-1 dv

Rocky Mountain National Park
on an "average" day

Representative Conditions:
Visual Range: 110-150 km
b_{ext}: 36-26 Mm^{-1}
Haziness: 13-10 dv

Rocky Mountain National Park
on a "dirty" day

Representative Conditions:
Visual Range: 75-95 km
b_{ext}: 52-41 Mm^{-1}
Haziness: 17-14 dv

Figure III-12. Photographs illustrating visibility conditions at Rocky Mountain National Park.

Figure III-12. Photographs illustrating visibility conditions at Rocky Mountain National Park.

A hallmark of sites impacted by sulfate pollution is a distinct seasonal trend of sulfate manifested by high concentrations during the summer and low concentrations during the winter. Sulfate seasonality can be attributed to many factors with seasonal changes in meteorology and photochemistry being the most influential. Sites that demonstrate the most sulfate seasonality are in the East and southern California, while sites in the Intermountain West have little or no seasonality. Absorption (elemental carbon concentrations) also demonstrates a seasonal trend at many sites and tends to be highest during the summer and early autumn. The seasonality of absorption is attributed primarily to seasonal changes in emissions. In the West, where the absorption seasonality is strongest, controlled burning and wildfires have a strong influence, while in the East the seasonality is less pronounced.

ROMO sulfur concentrations have a fairly constant median level with a mixed pattern of variability. The median sulfur concentration rarely moved in the same direction for more than two seasons. However, a clear downward trend in absorption (elemental carbon) was unique to ROMO. For ROMO, the median absorption in the winter of 1982-1983 (0.4 $\mu g/m^3$) dropped to 0.125 $\mu g/m^3$ by the winter of 1994-1995. Since 1991-1992, the median absorption has never exceeded 0.2 $\mu g/m^3$. The trend of the 75th percentile is even more impressive, the maximum occurs in the winter 1982-1983 at about 0.8 $\mu g/m^3$ and drops to less than 0.2 $\mu g/m^3$ in recent years. (Sisler et al., 1996).

D. RESEARCH AND MONITORING NEEDS

1. Deposition

Atmospheric deposition of S and N appear to be relatively well-quantified for ROMO. The location of a deposition monitoring station at Loch Vale is very appropriate for two reasons. First, Loch Vale is situated in the southeastern portion of the park, which probably receives the highest levels of atmospheric N deposition and is most likely to be further impacted by future changes in emissions from the growing Front Range urban corridor. Second, Loch Vale is one of the most sensitive watersheds in the park to potential adverse ecological impacts of atmospheric deposition. Monitoring data collected at Loch Vale provide important information to support ongoing long-term ecological research at this site. Thus, it is critically important to continue monitoring atmospheric deposition, particularly of N, at Loch Vale. We also recommend continued deposition monitoring at Beaver Meadows. Although these data are not as useful for surface water effects as are the data from Loch Vale because the latter is located at higher elevation, we believe that potential impacts on aquatic and terrestrial resources in ROMO are of sufficiently great concern to warrant monitoring stations in two different sections of the park. There does not seem to be a need for additional efforts aimed at further categorizing deposition to ROMO at this time. If regional deposition of N or S increases substantially in the future, then a deposition monitoring station on the west side of the park may also be warranted.

2. Gases

Ozone pollution is a potential threat to ROMO, and air quality monitoring efforts should be directed at determining where ozone levels are highest in the park. In addition to maintaining the continuous ozone analyzer near park headquarters, a second analyzer should be installed for 3-5 years during summer at a high elevation site (e.g., near the Alpine Visitors Center) to compare with diurnal ozone concentrations at the lower elevation site. Studies of ozone profiles at ROMO (Ray, unpublished data) and other mountainous areas including the Swiss Alps (Sandroni et al. 1994), Sierra Nevada (Miller et al. 1986) and Cascade Mountains (Brace and Peterson 1996,1998) have shown that topography and elevation can influence diurnal exposure. Low-elevation sites have diurnal profiles characterized by low ozone levels during the nighttime and early morning hours, maximum levels during the mid or late afternoon, and low levels again in the evening. High-elevation sites typically have lower maximum ozone concentrations than low elevation sites, but ozone remains elevated during the morning and nighttime hours. Plant species at higher elevations may be at risk from exposure to elevated levels of ambient ozone during the morning hours when they are physiologically active.

A network of passive ozone samplers should be established to compare ozone measurements on the east-and west-facing portions of the park, with passive samplers colocated with continuous monitors. The east-facing side of the park likely receives a greater portion of the regional ozone and ozone precursors when winds are southeasterly, with lower ozone concentrations along the west side due to the barrier effect of ridgetops along the Continental Divide. The network of passive ozone samplers should include one or two transects across an elevation gradient on the east and west slopes of the Continental Divide. Sampling sites used in the 1995 passive sampling study (Figure III-3, and Table III-10) can be used plus additional sites on the west slopes. There is good access for sampling along the Trail Ridge Road and Fall River Road. Weekly samples for two months each summer along east and west transects would provide basic information on spatial variation of ozone distribution in the complex terrain of ROMO.

Ozone monitoring efforts should continue to extend beyond the boundaries of the park to identify regional sources of ozone and ozone precursors. Collaboration between ROMO, the Colorado Department of Health and other federal and state agencies in monitoring NO_x, VOC, and ozone levels in and downwind of urban areas will yield important information on formation, transport, distribution, and persistence of ozone in the park and the mountainous areas adjacent to the Boulder-Denver and Fort Collins areas.

3. Aquatic Systems

Effects of atmospheric deposition on water chemistry in ROMO are reasonably well understood. We are convinced that N deposition is causing elevated concentrations of NO_3^- in surface waters in many watersheds within and outside the park on the east side of the Continental Divide. These elevated NO_3^- concentrations have most likely resulted in decreased alkalinity of lakes and streamwaters, although it does not appear that any surface waters in the park are acidic (ANC$\leq$ 0) as a consequence. Because some watersheds are already leaching relatively high levels of NO_3^- under current deposition, we believe that further increases in atmospheric N loading will cause increased leaching of NO_3^- from exposed bedrock areas and talus slopes (where vegetative and microbial uptake is limited) in high-elevation watersheds and perhaps increased leaching from tundra and subalpine forest soils. It has not been demonstrated that soils in the park have become N-saturated, however. For the reasons outlined above, it is critical to continue monitoring surface water chemistry within sensitive areas of the park. Monitoring should continue in the Loch Vale watershed. The National Park Service may also want to consider monitoring an adjacent high-elevation site that does not appear to have watershed sources of sulfur. This might be very useful in the future, in the event that S deposition increases substantially above current levels. Justification for this recommendation is two-fold. First, the aquatic resources in ROMO are clearly very sensitive to acidic deposition of any kind. Second, it is exceedingly difficult to quantify acidification (loss of ANC) unless long-term monitoring is initiated early in the acidification process. Based on available data (Gibson et al. 1983, Landers et al. 1987), it appears that there are several good candidate sites. These include Glacier Creek (Gibson et al. 1983) and several lakes sampled during the Western Lakes Survey (Table III-14). For example, Spectacle Lake, Arrowhead Lake, and Black Lake all had ANC less than about 30 µeq/L, pH around 6.5, and SO_4^{2-} concentration less than 20 µeq/L. We propose additional water chemistry sampling at some of these sites to verify appropriate chemistry for routine monitoring. Because available data suggest both lower acidic deposition and lower surface water sensitivity to acidic deposition on the western side of the Continental Divide in ROMO, we do not see a pressing need to monitor the acid-base status of water chemistry on the west side of the park at this time.

There is a need for additional episodic monitoring of surface water chemistry in sensitive aquatic resources in ROMO. Such monitoring would entail collection of lakewater and streamwater chemistry soon after ice out on high-elevation lakes. In many cases, safety considerations prevent sampling during the early phases of snowmelt, but collection of monitoring data in late June or early July would be very useful. These data would help to 1) clarify the extent to which episodic acidification occurs under current deposition, 2) quantify the relative roles of S and N in episodic acidification of aquatic resources in the park, and 3) establish a baseline for episodic acidification for comparison with future years when deposition may be higher. We recommend a sampling program of about 10 lakes and streams, distributed across the known (and presumed) sensitive portions of the park, to be sampled three times per year at approximately monthly intervals from the earliest practical sampling date for each watershed. This monitoring should be continued for at least two years.

Although we are convinced that atmospheric deposition of N has had some effects on water chemistry in some areas of ROMO, it is unclear the degree to which biological resources in aquatic ecosystems of the park have been affected or are at risk. Additional studies of the distribution of acid-sensitive biota within low-alkalinity waters (< 50 μeq/L) seem warranted.

Loch Vale watershed is more vulnerable to atmospheric inputs of N than to inputs of S (Baron et al. 1995). Due to the probable internal watershed sources of S and the observed lack of response of streamwater SO_4^{2-} concentration to recent changes in sulfur deposition, Baron et al. (1995) concluded that the Loch Vale watershed is unresponsive to the levels of S deposition observed within the Rocky Mountains over the last 10 years. In contrast, concentrations of NO_3^- in surface waters in Loch Vale are relatively high, are likely controlled largely by atmospheric inputs of nitrogen, and will increase if deposition increases in the future. We regard nitrogen as the primary air pollutant of concern with respect to aquatic resources throughout the park.

Future increases in atmospheric nitrogen deposition may impact terrestrial, as well as aquatic ecosystems in ROMO. However, we do not believe that substantial additional terrestrial research regarding N-driven acidification is needed at this time with respect to nitrogen effects. We do, however, think that it would be advantageous to periodically monitor the concentration of NO_3^- in soil solution of terrestrial systems of Loch Vale to document the extent to which N-saturation of terrestrial systems develops. Continued deposition and aquatic monitoring at Loch Vale will provide much important information regarding N input/output budgets, which reflect terrestrial processes within the watershed. If N deposition increases substantially in the future (~ doubles), then the issue of terrestrial research regarding nitrogen should be revisited.

Based on the observed high degree of sensitivity of several lakes in ROMO to future increases in acidic deposition, we recommend modeling of one or more watersheds to better quantify the loading rates for S and N above which adverse impacts would be expected, based on current scientific understanding. Such a modeling effort is currently underway (contact: K. Tonnessen, ARD, Denver).

4. Terrestrial Systems

It is difficult to recommend a monitoring system for detecting air pollutant effects on plants in ROMO because there are (1) no known visible symptoms of air pollutant effects on plants in the field in the Rocky Mountain region, and (2) few data on pollution effects in plant species found in ROMO. We must therefore rely on data published on other species and from experimental studies. Ozone injury is the most likely potential damage that would be observed in ROMO. Therefore monitoring of ozone-sensitive species is recommended in areas where ozone concentrations are expected to be highest. Specific species and locations recommended for monitoring are listed below. Additional details, as well as quality assurance/quality control protocols, can be found in Appendix A.

We recommend placing vegetation plots at two locations in the eastern portion of ROMO: (1)

ponderosa pine in meadows west of ROMO headquarters, and (2) subalpine fir at higher elevation. The plots surveyed in Stolte (1987) should be relocated, and a subset of the plots should be included for monitoring. Three additional plots could be located along the trail leading to Loch Vale. This trail and valley are located along the east slope where maximum impacts of ozone and other pollutants might be expected. All sites are within a one-day hike along existing trails. Locations of these plots can be changed to other sites in ROMO, such as previously-surveyed plots measured in Stolte (1987), if ambient air quality data indicate that other areas have a higher risk of pollutant effects. If additional plots become necessary, they could be located along transects that evaluate other conifer species, such as Engelmann spruce, lodgepole pine, Douglas-fir or whitebark pine. These plots should be located on the east side of ROMO where ozone exposure is the highest. If monitoring of herbaceous plants becomes desirable, candidate species in ROMO include strawberry (*Fragaria virginiana*), ninebark (*Physocarpus monogyna*), and red clover (*Trifolium repens*). Additional information on herbaceous monitoring is found in Appendix A.

Given the concern about the possibility of nitrogen deposition causing changes in plant species composition and diversity in high elevation plant communities in ROMO, it may be useful to establish a baseline of plant species distribution and abundance. We therefore recommend establishment of herbaceous plots in at least two locations, situated in alpine tundra and subalpine meadow habitat types. The Loch Vale watershed might be an appropriate location.

Recent reports suggest that exotic species are becoming more dominant in some areas of ROMO (Stohlgren et al., unpublished manuscript), and that this trend may be correlated with increased levels of soil N. Therefore, if N loading in terrestrial ecosystems increases, then there may be subsequent effects on the distribution and abundance of plant species. Data from vegetation monitoring plots in ROMO should be analyzed to determine long-term changes in vegetation composition particularly as it is related to N deposition patterns in the park.

5. Visibility

IMPROVE aerosol and optical monitoring should continue at ROMO. Ongoing and future monitoring is necessary to identify local source impacts. Additional data and in-depth modeling and analysis are required to further evaluate historical trends and future projections of impact from existing and future sources. For example, back trajectory analysis and spatial/temporal pattern analysis of episodes are recommended to determine the source region contributions to elevated aerosol concentrations. Future research is also recommended to minimize the uncertainty in estimates of how various aerosol species affect visibility.

To enhance monitoring efforts at ROMO, particle and optical monitoring could be established in the west area of the park. Dramatic differences in elevation can produce significantly different visibility conditions. Visibility monitoring sites at different elevations would therefore be desirable. This additional monitoring would help to better assess visibility impairment at the park.

IV. GRAND TETON NATIONAL PARK

A. GENERAL DESCRIPTION

Grand Teton National Park (GRTE) consists of 126,530 ha located in northwestern Wyoming. GRTE is surrounded by Bridger-Teton and Targhee National Forests, and lies 10 km south of Yellowstone National Park (YELL). The park was established to protect the scenic and geological values of the Teton Range and Jackson Hole, and to perpetuate the park's indigenous plant and animal life. Natural resources of the park are managed under ecosystem concepts aimed at perpetuating natural systems rather than individual species or features (GRTE 1995). Approximately 3.7 million people visit GRTE each year.

1. Geology and Soils

The Teton Mountains, a 67-km long range, stretch along a north-south line and reach a height of 4,230 m. The Teton Mountain Range slopes steeply down to Jackson Hole, an intermountain valley about 75 km long and 10 to 20 km wide. The Snake River flows south through the valley, which varies from about 1,800 to 2,100 m elevation. To the east of Jackson Hole are the Absaroka Mountains and to the southeast the Gros Ventre Mountain Range. The lower elevation relief is characterized by several terrace levels and glacial moraines, especially on the west side of the valley. Glacial ice has carved numerous U-shaped valleys and cirques. Erosion has formed deep V-shaped valleys within the mountain range. The park drains into the Snake River, a tributary of the Columbia River.

Late in the Tertiary age, faulting uplifted the mountains on the east side of Jackson Hole. Volcanic activity to the north filled portions of the basin with volcanic conglomerate and tuff. Starley soils developed in these areas. Rhyolite flows from the Yellowstone region covered the northern section and provided the parent material for Hechtman soils. One of the last events of diastrophism was the faulting and uplift of the Teton Mountains and the down-dropping of the Jackson Hole valley floor. This large vertical displacement exposed granite, gneiss and schist, from which Teewinot soils have developed. The sedimentary rocks have all been removed by erosion in the central portion of the range. Sedimentary rocks, and associated Starman and Tongue River soils, persist at the northern and southern ends of the range (Figure IV-1). They are mainly limestone, sandstone, and clay shale. The granite, gneiss, and schist of the high mountain areas of the park are expected to be fairly resistant to weathering and would not be expected to contribute significant amounts of base cations to drainage waters.

Within the past million years, Jackson Hole has experienced numerous periods of glaciation. Most of the ice entered the valley from the high country of the Yellowstone area rather than from the smaller Teton Range. The most recent, the Pinedale glaciation, flowed south through the valley and

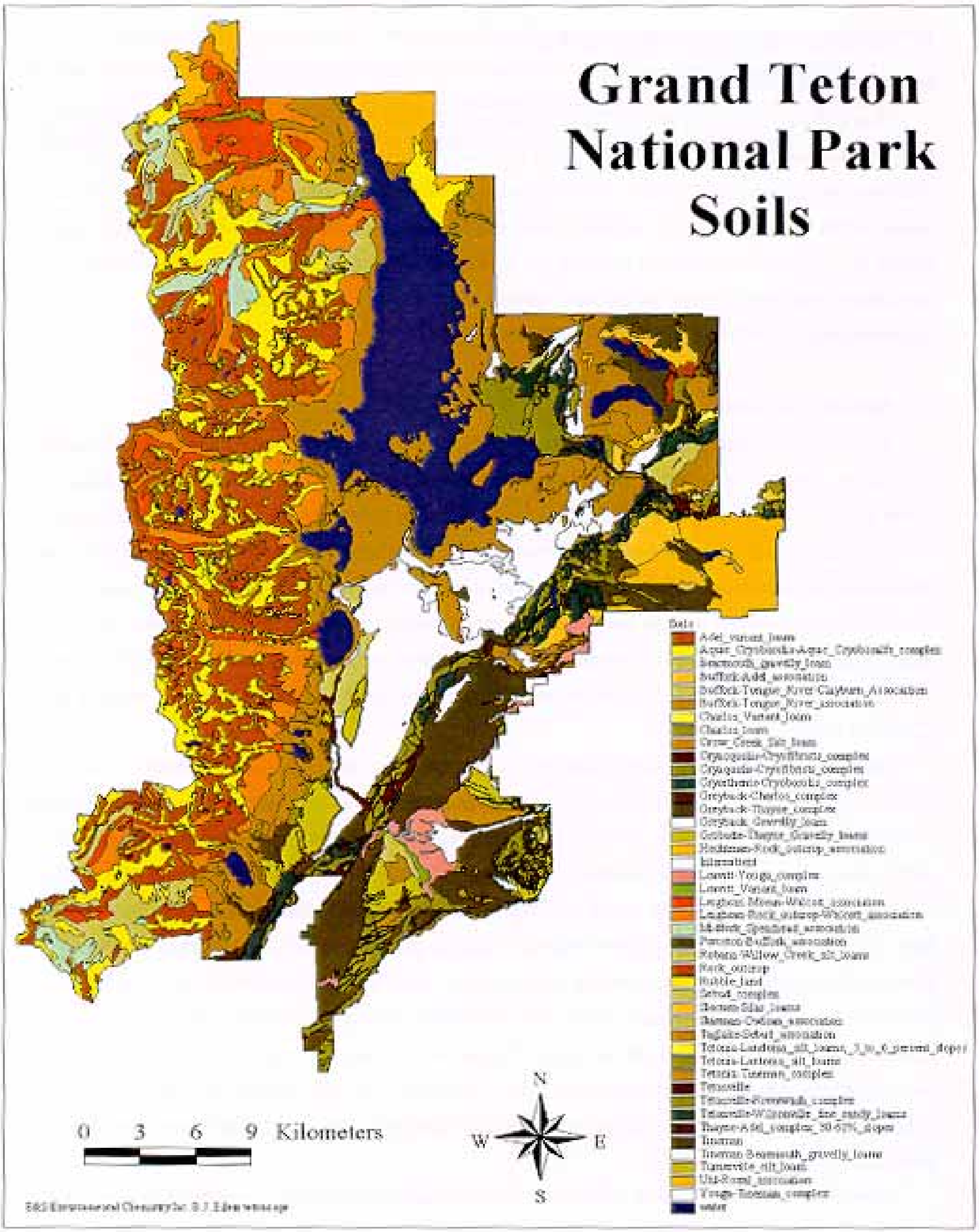


Figure IV-1. Soil classifications of GRTE.

Figure IV-1. Soil classifications of GRTE. (Source: GRTE database) created the series of river terraces visible on both sides of the valley (Olson and Bywater 1991). The retreat of the most recent large-scale glaciers occurred about 12,000 to 15,000 years ago. The dozen smaller glaciers now present in the Teton Range are relatively new, having been formed in the last few thousand years. Most surface waters within drainages fed by these glaciers likely receive sufficient contributions of base cations from glacial scouring to render them insensitive to the adverse effects of acidic deposition.

2. Climate

The climate of GRTE is classified as cold-snowy forest with humid winters. Temperature fluctuations are large between summer and winter and between daily maxima and minima due to high elevation and dry air which permits rapid incoming and outgoing radiation. Temperatures can be extremely cold in winter, and below freezing temperatures can be encountered during any month. Meteorological data have been collected in the area of GRTE since 1889 (Dirks and Martner 1982). There is a significant north-south gradient in annual precipitation values in the park region, with highest amounts to the north and lowest amounts to the south in the rain shadow of the high peaks of the Teton Mountains. Winter and spring precipitation tend to be highest (Dirks 1975). Average annual precipitation varies from about 41 cm at Jackson to about 154 cm near the summit of the Teton Mountains. Average annual snowfall varies from about 2 m at Jackson to over 7.7 m at high elevation. Snowmelt generally peaks in May and June. Thunderstorms are frequent during summer. The average annual temperature in Jackson is about 3°C with extremes of -43°C and 38°C. Surface winds display a wide range of prevailing directions and mean speeds depending on the topography and elevation of the site (Dirks and Martner 1982). At the higher locations, the prevailing winds are consistently from the southwest.

3. Biota

Over 1,000 species of vascular plants and over 200 species of fungi occur in GRTE and nearby Teton County, WY. These include 117 exotic species of vascular plants. Ongoing efforts to characterize and classify vegetation at GRTE have culminated in the currently used classification of cover types (Figure IV-2). Loope and Gruell (1973) emphasized the critical influence of fire on vegetation distribution and abundance in the park. Approximately 58% of GRTE is nonforested, consisting of alpine tundra, rock, meadows, grassland, and shrublands. Of the forested portion of the park (Figure IV-3), 28% is lodgepole pine (*Pinus contorta*) forest, 7% Engelmann spruce (*Picea engelmannii*) and subalpine fir (*Abies lasiocarpa*), 4% Douglas-fir (*Pseudotsuga menziesii*), 2% whitebark pine (*Pinus albicaulis*), and 1% aspen (*Populus tremuloides*) (Greater Yellowstone Coordinating Committee 1987).

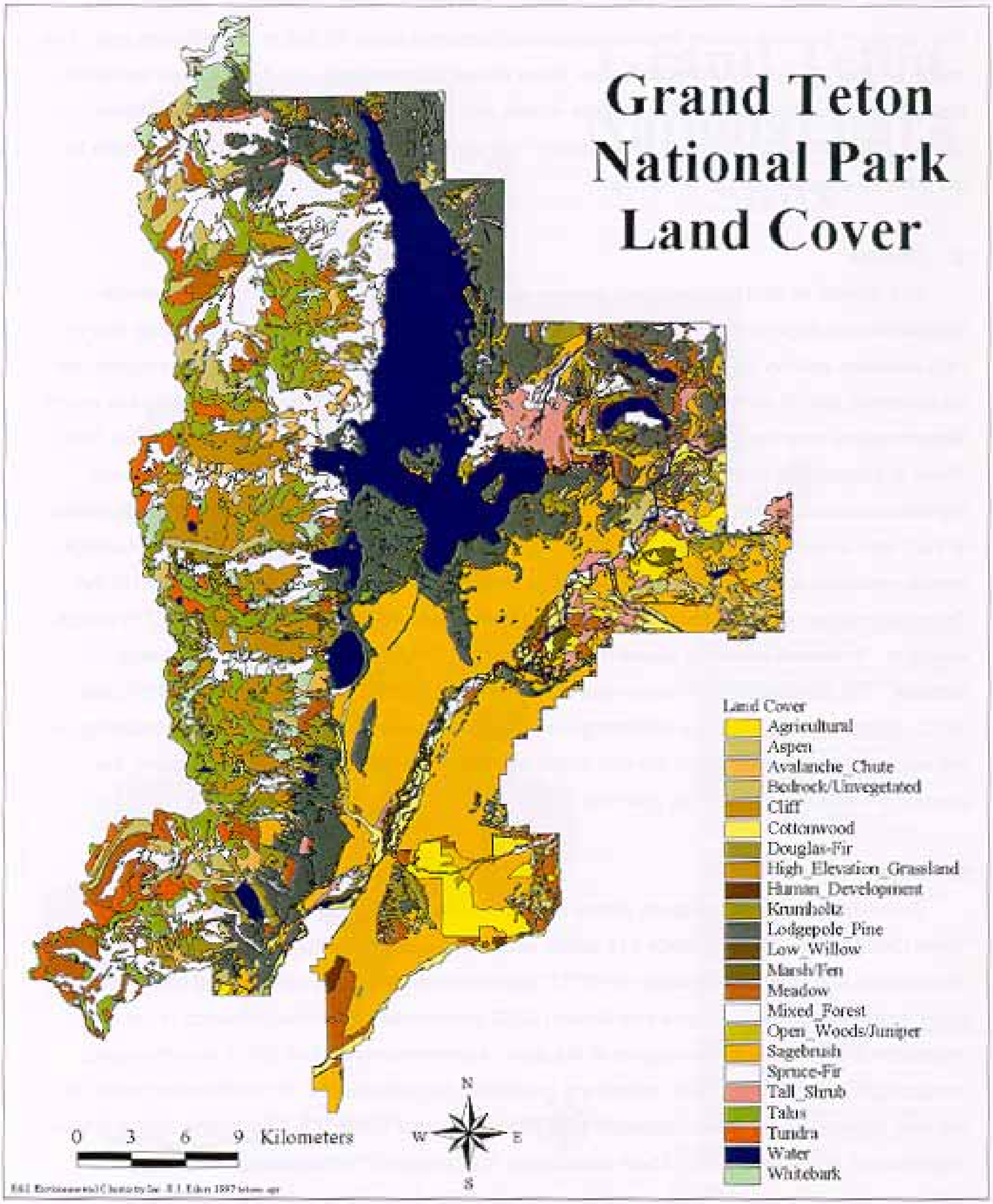


Figure IV-2. Land cover of GRTE.

Figure IV-2. Land cover of GRTE. (Source: GRTE database)

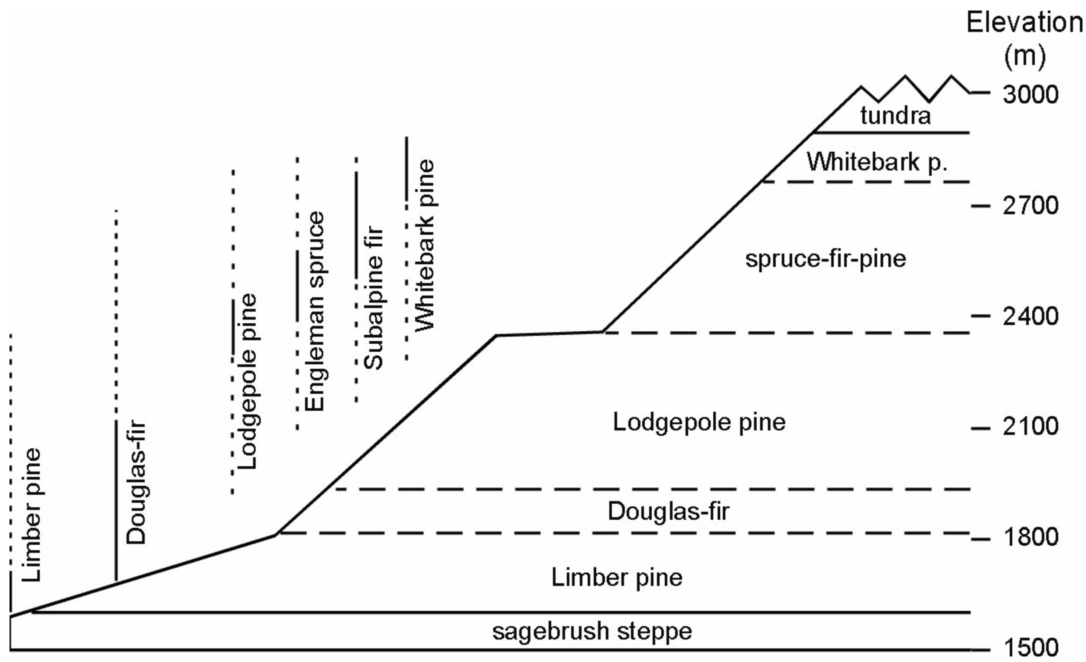


Figure IV-3. The elevational distribution of major forest types near GRTE. The solid vertical lines show the altitudinal ranges over which each tree species is important as a forest dominant; the dotted lines indicate the range over which the species can be found. Adapted from Baker (1976) and Whitlock (1993).

Sagebrush (*Artemisia* spp.) dominates the floor of Jackson Hole, except on glacial moraines where lodgepole pine and spruce-fir are common (Knight 1994), and grows in association with a wide range of other shrub and grass species. Riparian woodlands, which include black cottonwood (*Populus balsamifera* var. *trichocarpa*) and a variety of other hardwoods and conifers, are found adjacent to rivers and streams, often in association with willow (*Salix* sp.) shrublands and other shrub species. Aspen groves are found in areas of higher soil moisture, often in a mosaic with Douglas-fir and sagebrush. Plant communities can be further subdivided into various habitat types according to Steele et al. (1983). Fire and mountain pine beetles (*Dendroctonus ponderosae*) are the primary disturbance agents in the GRTE landscape, and largely help to retain the successional species aspen and lodgepole pine as significant components of the vegetation (Loope and Gruell 1973, Steele et al. 1983). Fire has influenced both the forest and sagebrush communities, although fire frequency is now probably less than it has been in the past due to fire exclusion (Knight 1994). The impact of a beetle epidemic during the 1970s is still visible in places.

There are 54 species of mammals in GRTE, including black bear (*Ursus americana*), elk (*Cervus elaphus*), pronghorn (*Antilocapra americana*), bison (*Bison bison*), moose (*Alces alces*), mule deer (*Odocoileus hemionus*), and several species of smaller mammals. There are over 300 bird species, including white pelican (*Pelecanus erythrorhyncos*), trumpeter swan (*Cygnus buccinator*), sandhill crane (*Grus canadensis*), Canada goose (*Branta canadensis*), bald eagle (*Haliaeetus leucocephalus*), and golden eagle (*Aquila chrysaetos*). There are nine reptile and amphibian species and a cold-water fish fauna of 16 species, including native cutthroat trout (*Oncorhynchus clarki*) and several introduced species of trout.

Gulley and Parker (1985) collected 27 genera of benthos from 46 alpine lakes in the park. Chironomids, limnephilids, and dytiscids were the most widely distributed families found. Other families, including Lestidae and Elmidae, were restricted in their distribution and found only in one basin. The authors tested the hypothesis that the number of benthic genera present in each lake was correlated with the water chemistry and physical conditions of the lakes. Stepwise multiple regression was employed with elevation, log lake surface area, log TMDS (total measured dissolved solids), and pH as independent variables. The only variable that was significantly correlated with the number of benthic genera was elevation ($p < 0.001$), which explained 26% of the variation in number of benthic genera.

Twenty-eight species of zooplankton were identified from the lakes surveyed by Gulley and Parker (1985). Each species and the basins in which it was found is listed in Table IV-1. The zooplankton species found in these lakes were considered to be fairly common high-elevation species. Temperature was the variable most significantly correlated with zooplankton abundance. Zooplankton biomass was positively correlated with temperature and lake surface area. As elevation decreased and lake size and temperature increased, the number of zooplankton species increased. These regressions only explained a small portion (20% to 31%) of the variance in the dependent variables (Gulley and Parker 1985).

Concern has increased in recent years regarding the decline in the populations of amphibians in many areas of the western United States (Wake 1991). Amphibian declines in relatively undisturbed areas such as national parks are of particular concern. Peterson et al. (1992) surveyed amphibians at eight sites in the Greater Yellowstone area during the spring and summer of 1991. Only one site was located in GRTE. Lower Moose Pond is a large inactive beaver pond at 2070 m elevation southwest of Jenny Lake. Western toads (*Bufo boreas)* and spotted frogs (*Rana pretiosa*) had been observed in this area in 1951 (C.C. Carpenter, pers. comm., Peterson et al. 1992). Lower Moose Pond had pH of 6.8 and ANC of 175 μeq/L on the May 25th sampling date. It would be considered insensitive to acidification from acidic deposition. Peterson et al. (1992) found adult, egg, and larval stages of spotted frogs at this site and heard calls (but no sightings) of western chorus frogs (*Pseudacris triseriata*). No other species of amphibians were observed.

Table IV-1. Species of zooplankton collected during the summers of 1982 and 1983 from 70 small lakes in Grand Teton National Park. (Source: Gulley and Parker 1985)

Genus	Species	Drainages[a]
Alona	*guttata*	NS, PB, GR, VA
Bosmina	*longirostris*	GR, AV
Brachionus	*spp*	VA
Camptocercus	*rectirostris*	VA
Ceriodaphnia	*quadrangula*	BP, VA
Ceriodaphnia	*reticulata*	VA
Chaoborus	*spp*	BP, VA
Conochilus	*spp*	VA
Cyclops	*bicuspidatus thomasi*	GR, AV, PB
Cyclops	*spp*	WB, GR, BP, VA
Daphnia	*pulex*	WB, NS, MO, PB, GR, AV, BP, VA
Daphnia	*rosea*	PB, GR, AV
Diaphanosoma	*brachyurum*	VA
Diaptomus	*leptopus*	VA
Diaptomus	*lintoni*	VA
Diaptomus	*shoshone*	WB, NS, MO, GR, AV, VA
Diaptomus	*spp*	WB, NS, MO, LE, PB, GR, AV, SD
Diaptomus	*tyrrelli*	SS, VA
Eucyclops	*agilis*	VA
Eurycercus	*lamellatus*	VA
Holopedium	*gibberum*	NS, SS, PB, VA
Kellicottia	*spp*	NS
Keratella	*spp*	GR, VA
Leptodora	*kindtii*	GR
Macrocyclops	*albidus*	VA
Subclass Ostracoda		VA
Pleuroxus	*procurvatus*	VA
Simocephalus	*vetulus*	VA

[a] AV = Avalanche; LE = Leigh; ND = North Death; SD = South Death; BP = Bearpaw; MP = Moose Ponds; NS = North Snowshoe; SS = South Snowshoe; GG - Glacier Gulch; MO = Moran; PB = Paintbrush; VA = Valley; GR = Garnet; NC = North Cascade; SC = South Cascade; WB = Webb

4. Aquatic Resources

There are about 90 alpine and subalpine lakes and ponds in the park. They are located above about 2700 m elevation. The majority are in remote areas that are difficult to access (Gulley and Parker 1985). Most are less than 10 ha in area. Larger lakes are found at lower elevation. Many lakes in the park were formed behind the terminal moraines of glaciers. Jackson Lake, situated on the Snake River, is the largest natural lake in the area, with a depth of over 120 m and a length of 32 km. Jackson Dam, situated at the natural outlet of Jackson Lake, provides a major storage reservoir on the Snake River. Originally constructed in 1907, it was reconstructed to its current storage capacity of 847,000 acre-feet in 1917. Other large lakes in the park include Leigh Lake, Jenny Lake, and Phelps Lake. All are situated along the border between the valley, Jackson Hole, and the Teton Mountains. The multitude of small lakes and streams are distributed throughout the mountainous areas of the park, especially in the central and southern portions of the range.

The small size, shallow depth, and circular shape, which are typical of the average lake located at high elevation in the Tetons, are indicative of glacially carved lake basins (Hutchinson 1957). The mean maximum depth of alpine lakes in the park is 6.8 m, and the average mean depth is 3.6 m, based on bathymetric measurements of 46 alpine lakes (Gulley and Parker 1985). An important physical-chemical relationship is that between mean lake depth and the amount of dissolved oxygen under the ice in winter. Generally, there is less chance for winter fish kill due to oxygen depletion with increasing mean depth (Wetzel 1975). However, alpine lakes in the Tetons, although mostly shallow, probably do not become anoxic under ice cover because oligotrophic lakes deplete oxygen under the ice very slowly (Mathias and Barcia 1980, Gulley and Parker 1985). Most likely, those lakes shallow enough to winter-kill will freeze solid in winter. This likely occurs in the park in lakes with depths around 1 m (e.g., Nelder and Pennak 1955, Gulley and Parker 1985).

B. EMISSIONS

There is little industrial activity and low population in northwestern Wyoming, resulting in good regional air quality. Most of the industrial activity in Wyoming occurs in the eastern counties near the cities of Gillette and Casper, and in the southwestern counties around Rock Springs. Oil and gas processing, electric utility power plants and industrial fossil-fuel combustion in southwestern Wyoming and southeastern Idaho are the major sources of gaseous pollutants and deposition to the GRTE area. There may also be some long-range transport of pollutants from the Salt Lake City area. Annual emissions of gaseous SO_2, NO_x and VOC in Wyoming are mainly from fossil fuel burning by industrial sources (Tables II-2, II-3 and II-4), and levels are moderate relative to other Western states.

Point sources of SO_2, NO_x, and VOC located within 150 km of GRTE (with emissions exceeding 100 tons/yr) are listed in Table IV-2. Adjacent counties in Idaho and Montana are included. SO_2 emissions in Wyoming are mainly from oil and gas refineries, the largest of which is Amoco

Production Co. (emitting over 1,200 tons/yr) located in Elk Basin, 125 km northeast of GRTE. The largest regional sources of SO_2 within 150 km of GRTE are the Monsanto Company and Nu-West Industries, both mining operations located approximately 100 km south of GRTE in Caribou County, southeastern Idaho. The annual emission of SO_2 from these companies is approximately 9,000 tons/yr combined.

Table IV-2. Point sources of SO_2, NO_x, and VOC in tons per year (annual emissions exceeding 100 tons per year of at least one pollutant) within 150 km of GRTE. (Source: Wyoming Department of Environmental Quality 1995, Martin 1996, Idaho Department of Health and Welfare, unpublished data)

	SO_2	NO_x	VOC
Wyoming			
Amoco Co.	1,218	603	208
Marathon Oil	21	7	869
Oregon Basin Gas	455	25	49
Questar Pipeline		100	10
Williams Field Services (3 sites)		1,381	740
Williston Basin IPC		162	69
Idaho			
Ash Grove Cement	889	802	31
Basic American Foods	498	174	4
Basic American Foods	193	37	54
Chevron Pipeline			437
Idaho National Engineering Labs	91	1,229	35
Idaho Pacific Corp.	131	156	1
Idaho Supreme Potatoes	136	151	1
J.R. Simplot - siding facility	2,554	646	131
Monsanto Co.	4,703	2,021	
Nu-West Industries, Inc.	4,369		
Pillsbury Co.	1	156	7
Montana			
Holnam Inc.	32	1,330	2
Luzenac America	25	242	25
Montana Power Co.		100	25

Sources of NO_x in Wyoming include electric utilities and industrial fossil-fuel combustion (Table II-3). NO_2 was monitored at several sites in the state between 1975 and 1985, and because there were no exceedences of the NAAQS during that time, monitoring was discontinued. Highest levels during the monitoring period were recorded in Rock Springs, approximately 200 km south of GRTE. The largest regional source of NO_x is Monsanto Corporation located in Caribou County, southeastern Idaho. Major stationary sources of VOC emissions, important ozone precursors, are relatively low in Wyoming. However, there are thousands of small VOC and NO_x sources that cumulatively may add up to much higher emission totals (T. Blett, pers. comm.).

Degradation of visibility from particulates is the most important air quality concern in the park. Forest fires and unpaved roads are important sources of particulates. For most visitors, scenic vistas are an important reason for their visit to the park. Seasonal increases in CO and particulates associated with woodburning stoves in Jackson (40 km south of GRTE) and high snowmobile use in the park (Snook 1996) are growing concerns for GRTE.

Potential future impacts on GRTE's natural resources could be caused by the following sources of pollution: (1) increasing residential and business development in Jackson Hole south of the park, including woodburning stoves and fireplaces, automobiles, and air traffic; (2) increasing use of prescribed burning in and around Jackson Hole; (3) proposed oil and gas development and associated activities south, east, and west of the park (including on BLM land); (4) agricultural practices in Idaho west of the park; and (5) metropolitan and industrial development along the western slope of the Wasatch Mountains in the Salt Lake City, Utah area.

C. MONITORING AND RESEARCH ACTIVITIES

1. Air Quality

a. Wet Deposition

There is no deposition monitoring station in GRTE for S and N. However, there is a NADP monitoring station in YELL to the north. Both parks are exposed to the same general air masses, and both experience prevailing winds mostly from the southwest. There are no large point sources of N or S adjacent to either park that might cause major differences in local deposition. We therefore rely on deposition data from YELL to evaluate deposition issues for GRTE.

Precipitation volume and chemistry have been monitored at the NADP site at Tower Junction in YELL since 1980. Annual precipitation amounts are generally in the range of 30 to 45 cm per year at this site. The concentrations of SO_4^{2-}, NO_3^-, and NH_4^+ in precipitation are low, with each generally below 10 μeq/L (Table IV-3). The combined low amount of precipitation and low concentrations of acid-forming precursors in wetfall results in very low levels of S and N deposition. Sulfur deposition is generally well below 1 kg/ha/yr, and N deposition is seldom above this amount (Table IV-4).

Snowpack samples were collected in late March or April from two sites (Rendevous Mountain, Garnet Canyon) in GRTE in 1993 through 1998. Data are currently available for the period through

1997. Sulfate concentrations in snow were similar at the two sites and ranged between 3 and 9 μeq/L (mean, 5 μeq/L at both sites). Nitrate ranges and means were very similar to those for SO_4^{2-} (mean, 5 μeq/L at both sites) and NH_4^+ concentrations were somewhat lower (mean 3 and 4 μeq/L at Rendevous Mountain and Garnet Canyon, respectively). There were no apparent trends from year to year at either of the sites for any of the variables (G.P. Ingersoll, pers. comm.).

Table IV-3. Wetfall chemistry at the NADP/NTN site at Tower Junction, YELL. Note there is no deposition monitoring station in GRTE, although deposition in Yellowstone and GRTE is expected to be very similar. Units are in μeq/L, except precipitation (cm).

Year	Precip	H^+	SO_4^{2-}	NH_4^+	NO_3^-	Ca^{2+}	Mg^{2+}	Na^+	K^+	Cl^-
1995	38.7	5.3	5.8	7.7	7.7	4.8	0.9	1.8	0.5	1.8
1994	36.3	4.6	9.7	9.8	10.3	12.5	2.3	3.1	0.9	2.5
1993	39.6	2.6	8.6	8.3	8.1	6.6	1.4	2.6	0.4	2.1
1992	45.0	2.7	8.1	8.3	8.5	12.1	2.0	2.0	0.6	2.4
1991	45.5	2.6	11.0	6.8	9.7	14.8	2.7	4.1	0.6	2.5
1990	36.6	3.4	12.0	9.4	11.6	18.4	3.2	3.5	1.2	3.8
1989	42.9	2.1	8.7	8.8	9.4	12.2	2.3	3.0	0.6	3.3
1988	27.3	1.9	7.1	3.5	4.6	9.9	1.7	3.6	0.7	2.6
1987	8.7	2.6	4.5	3.2	4.6	5.7	1.1	2.5	0.3	1.9
1986	38.8	2.8	8.1	5.7	6.7	9.0	1.9	2.5	0.7	2.7
1985	36.1	3.0	6.6	2.9	5.4	6.4	2.0	2.0	0.6	2.5
1984	37.5	5.6	12.2	7.2	9.0	14.4	3.9	4.0	1.4	4.0
1983	34.5	3.6	11.0	5.0	5.8	10.9	2.8	3.8	1.1	3.2
1982	57.1	6.0	12.7	5.0	7.5	11.8	4.1	4.1	1.2	4.0
1981	34.1	4.4	32.6	9.6	11.7	21.9	8.2	26.1	1.6	13.4
1980	24.7	7.4	22.2	11.7	14.6	14.4	3.1	6.8	1.3	5.9
Average	36.5	3.8	11.3	7.1	8.5	11.6	2.7	4.7	0.9	3.7

Table IV-4. Wet deposition (kg/ha/yr) of sulfur and nitrogen at the NADP/NTN site at Tower Junction, YELL. Note there is no deposition monitoring station in GRTE, although deposition in Yellowstone and GRTE is expected to be very similar.

Date	Sulfur	NO_3-N	NH_4-N	Total Inorganic N
1995	0.4	0.4	0.4	0.8
1994	0.6	0.5	0.5	1.0
1993	0.5	0.4	0.5	0.9
1992	0.6	0.5	0.5	1.1
1991	0.8	0.6	0.4	1.1
1990	0.7	0.6	0.5	1.1
1989	0.6	0.6	0.5	1.1
1988	0.3	0.2	0.1	0.3
1987	0.1	0.1	0.0	0.1
1986	0.5	0.4	0.3	0.7
1985	0.4	0.3	0.1	0.4
1984	0.7	0.5	0.4	0.9
1983	0.6	0.3	0.2	0.5
1982	1.2	0.6	0.4	1.0
1981	1.8	0.6	0.5	1.0
1980	0.9	0.5	0.4	0.9
Average	0.7	0.4	0.4	0.8

Table IV-5. Monthly average ozone levels (ppbv) in GRTE for 1995 determined with passive ozone samplers. (Source: Ray 1995, unpublished data)

Month	Monthly ozone average (ppbv)
May	35.6
June	35.8
July	34.9
August	39.1
September	36.6

b. Occult/Dry Deposition

There are no data available on dry or occult deposition of S or N to sensitive resources within GRTE. However, we expect that the contributions of both dry and occult deposition of S and N are low relative to the wet deposition amounts summarized in Table IV-4. This is because there are no significant emission sources in close proximity to the park.

c. Gaseous Monitoring

Gaseous pollutants are not regularly monitored in GRTE, although ozone and SO_2 are monitored in YELL to the north. During the summer of 1995, ozone was monitored in GRTE near park headquarters at Moose Junction using passive ozone samplers. Monthly average ozone levels were between 35 and 39 ppbv (Table IV-5). Passive sampler data were also collected in 1996 and 1997. Because the passive ozone data reflect a mean concentration and do not indicate diurnal variation in ozone levels, it is unknown what the maximum hourly ozone concentrations or the diurnal pattern of exposure were at this site. It is reasonable to assume that concentrations and diurnal patterns are similar to those at YELL, although local sources of NO_x from the Jackson area could facilitate ozone breakdown at night.

2. Water Quality

Alpine lakes in GRTE exhibit a range of characteristics that contribute to their sensitivity to potential acidic deposition impacts (e.g., Marcus et al. 1983): bedrock resistant to weathering, shallow soil, steep slope, low watershed to lake surface area ratio, high lake flushing rate, high precipitation, high snow accumulation, and short growing season.

Surface water alkalinity values tend to be high throughout most of the low elevation areas of the park. Lakes and streams with alkalinity less than 400 μeq/L are generally restricted to the high mountain areas near the western border of the park.

Miller and Bellini (1996) evaluated the trophic status of 17 lakes in GRTE. Phosphorus and chlorophyll *a* concentrations were measured in an effort to detect aspects of lake water quality suggestive of eutrophication. A review of the literature did not yield earlier data on lake trophic status in the park, and the data collected in this study will therefore constitute the baseline for future evaluations of eutrophication in the park. Six of the study lakes were located in the mountains. Samples were collected in July and August and analyzed for specific conductance, pH, and total phosphorus concentrations. Specific conductance was below 20 μS/cm in all of the mountain lakes, suggesting low concentrations of dissolved ions. Two lakes (Amphitheater and Surprise) had very low specific conductance (< 10 μS/cm) and had pH in the range 6.0 to 6.5 (Table IV-6).

Table IV-6. Water quality data collected in 1995 by Miller and Bellini (1996) in mountain lakes of GRTE.

Lake	Approximate Elevation (m)	Lake Area (ha)	Specific Conductance (μS/cm)		pH Measurements	
			July	August	July	August
Amphitheater	3,000	2.4	9.7	8.9	6.0	-
Lake of the Craggs	3,000	4.0	15.3	11.8	8.2	7.7
Delta	2,750	3.2	11.5	9.8	7.9	-
Holly	2,960	3.2	17.4	15.1	8.0	8.1
Solitude	2,800	12.1	7.1	14.8	7.8	8.4
Surprise	2,900	1.2	8.7	8.3	6.5	-

Amphitheater and Surprise Lakes, the smallest of the lakes studied, are located in close proximity to each other, approximately at treeline, on a ridge between Glacier Gulch and Garnet Canyon to the northwest of Bradley Lake. Neither is fed by a glacier. Neither has a significant inlet or outlet stream, based on examination of 7.5-minute maps. The absence of inlet streams, and particularly the absence of glacial meltwater contributions, would be expected to predispose these lakes to acidic deposition effects. Based on the available information, these lakes would be expected to be moderately to highly sensitive to acidification from acidic deposition. Key elements of acid-base chemistry were not measured, however, including alkalinity and the concentrations of base cations, sulfate and nitrate.

Water quality was also measured by Miller and Bellini (1996) for seven moraine lakes, all of which had pH >8.0 and specific conductance greater than about 17 μS/cm. The moraine lakes tend to be larger than the mountain lakes; the smallest was Bradley Lake at 28 ha and the largest, Jenny Lake at 486 ha. These lakes are likely insensitive to acidification from acidic deposition. Similarly, all of the four valley lakes sampled had high specific conductance (> 100 μS/cm) and pH > 8.5 and would not be sensitive to acidic deposition.

Based on the phosphorus and chlorophyll *a* measurements, mountain lakes were oligotrophic to mesotrophic, as were moraine lakes. Trophic status of the valley lakes was more variable, with some in the eutrophic range.

Two lakes within GRTE, and three lakes in proximity to the park, were sampled as part of EPA's Western Lake Survey. None were particularly sensitive to acidic deposition. The lowest measured ANC value was 154 μeq/L, in a lake with pH of 7.3 (Table IV-7). Gulley and Parker (1985) surveyed 70 lakes and ponds in Grand Teton National Park during the months of June, July, and August of 1982 and 1983. Forty-six alpine lakes and ponds and 24 lower-elevation lakes were sampled. The majority of the lakes were relatively dilute, with specific conductance less than about 30 μS/cm. Twenty-two of the lakes were highly dilute, with specific conductance $\leq$10 μS/cm. They were all located at high elevation; all were above 2500 m and most were above 2900 m elevation. All except one (montane) lake were located in alpine settings. Dilute lakes tended to be small; most were less than 2 ha in area. The largest was 6 ha. Watershed areas were also small in most cases (< 100 ha). pH values were generally in the range of 7.0 to 8.0. Only four lakes had pH below 7.0 and two below 6.0. Calcium concentrations were below 30 μeq/L in six of the dilute lakes (Table IV-8).

Lakes and ponds were sampled by Gulley and Parker (1985) in 13 alpine drainage basins within the park (Figure IV-4). An analysis of variance (ANOVA) test was performed to test for differences in major ion chemistry among drainage basins. No significant difference was found among alpine basins for the majority of chemical parameters tested. The only ion to show a significant difference ($p < 0.05$) was for Mg^{2+}, and that difference was driven by the very high Mg^{2+} concentration of one lake (Schoolroom). Gulley and Parker (1985) concluded that the alpine basins of the park exhibited remarkably homogeneous water chemistry. This occurred because of the similar physical, geochemical, and vegetative characteristics of the alpine basins. All basins where lakes were sampled, except Schoolroom Lake and Avalanche Canyon, had bedrock geology of Precambrian gneiss, schist, and granite.

Table IV-7. Results of lakewater chemistry analyses by the Western Lake Survey for selected variables in GRTE and adjacent areas. (Source: Eilers et al. 1987)

Lake name	Lake ID	Lake area (ha)	Watershed area (ha)	Elevation (m)	pH (μeq/L)	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 GRTE											
Grassy L.	4D3-02	117.9	883	2198	7.3	153	18.9	0.3	93	188	1.5
Trapper L.	4D3-07	1.4	367	2107	7.6	435	38.7	2.1	298	491	1.1
Lakes Outside GRTE											
Hidden L.	4D3-02	7.2	129	2214	7.3	241	22.9	0.1	139	285	1.3
Loon L.	4D3-06	9.9	160	1970	7.0	486	21.0	0.3	320	549	4.2
Upper	4D3-06	47.0	3012	2022	8.8	2197	55.7	0.8	1481	2399	1.4

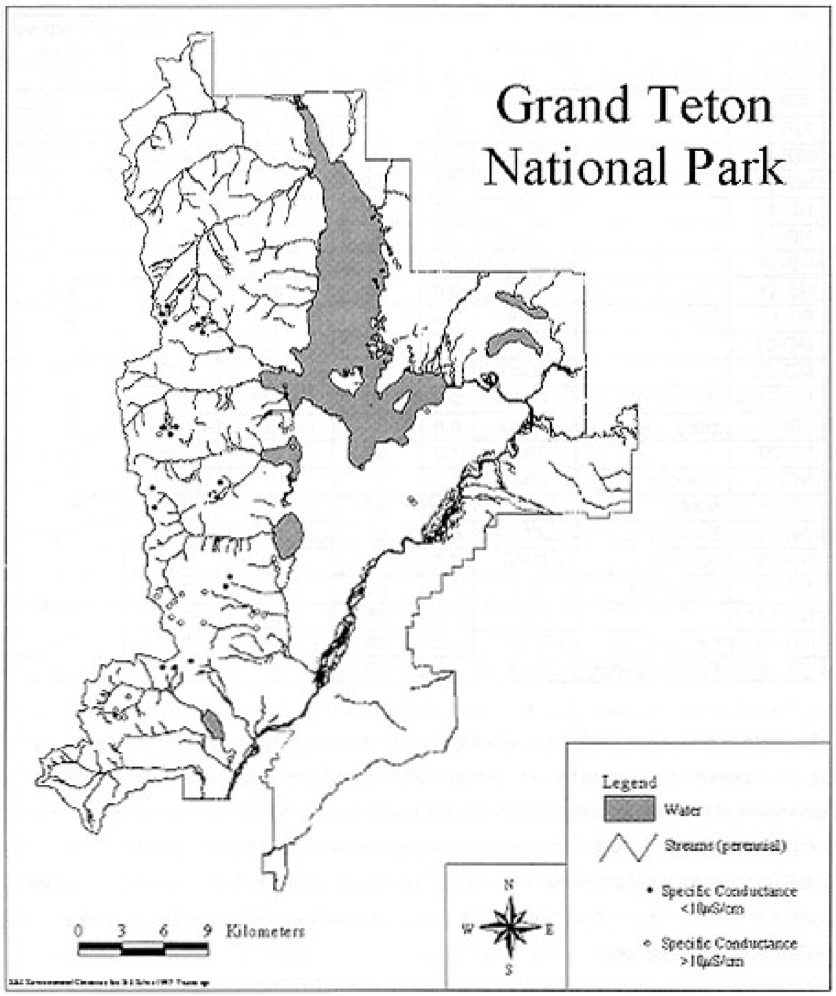


Figure IV-4. Location of lakes sampled by Gulley and Parker in GRTE. Dilute lakes (specific conductance <10 µS/cm) are shaded; other sampled lakes are indicated by open circles. From Gulley and Parker 1985.

Figure IV-4. Location of lakes sampled by Gulley and Parker in GRTE. Dilute lakes (specific conductance <10 μS/cm) are shaded; other sampled lakes are indicated by open circles. From Gulley and Parker 1985.

Table IV-8. Selected characteristics of dilute lakes and ponds (specific conductance $\leq$10 μS/cm) in GRTE surveyed by Gulley and Parker (1985).							
Lake Number	Lake Name	Elevation (m)	pH	Ca^{2+} (μeq/L)	Specific Conduct. (μS/cm)	Surface Area (ha)	Watershed Area (ha)
WB-02		3048	7.5	50	10	1.31	27.0
WB-11		2966	6.8	80	10	6.26	38.0
WB-20		2757	6.4	70	9	5.89	100.0
WB-21		2957	7.8	55	10	0.32	0.7
NS-10		2804	7.7	240	9	2.26	120.0
NS-13		3048	7.7	50	9	0.28	3.2
NS-14		3048	7.6	30	9	0.33	13.0
NS-21		3048	8.0	65	10	1.39	6.0
SS-10	Dudley	2500	NA	50	9	3.80	89.0
MO-01		2900	8.1	10	3	0.19	2.0
MO-23		3040	5.3	10	7	0.19	13.0
MO-24		3040	5.8	70	8	1.24	27.1
PB-10	Holly	2926	6.6	55	10	2.99	25.0
PB-20		2804	7.0	50	7	1.24	15.0
NC-10	Solitude	2804	7.2	40	10	14.77	180.0
NC-20	Mica	2926	7.5	55	8	3.92	67.0
GG-10	Delta	2791	8.0	80	10	2.69	140.0
GR-13	Surprise	2920	7.7	50	8	0.88	4.2
ND-10	Timberline	3131	7.6	25	6	1.92	27.0
ND-11		3100	8.0	55	9	1.12	86.0
ND-12		3131	8.2	30	8	0.11	5.0
SD-12	Forget-Me-Not	2900	7.1	10	4	0.11	3.0

The data presented in Table IV-8 suggest that there are a large number of alpine lakes and ponds in the park that are sensitive to potential acidification from acidic deposition, as reflected in the low values of specific conductance and calcium concentration. pH values were circumneutral, with two lakes having pH < 6.0, suggesting that chronic acidification probably had not occurred to any significant extent as of the sampling dates in 1982 and 1983. Sensitive lakes are located throughout most high elevation portions of the park, but especially in the northcentral portion of the Grand Teton Mountain Range.

Only a few alpine lakes sampled by Gulley and Parker (1985) had specific conductance >30 μS/cm (Table IV-9). For example, Schoolroom Lake, located at the base of Schoolroom Glacier, had a conductivity of 57 μS/cm, three times higher than other alpine lakes in the park, and total hardness equal to 31 mg/L as $CaCO_3$. Glacial silt from sedimentary rocks in the catchment at Schoolroom Lake affects the water chemistry of the lake by making its concentration of base cations higher than other alpine lakes in the region. In contrast, Delta Lake is situated directly below Teton

Table IV-9. Descriptive statistics for chemical data from 46 alpine lakes in GRTE (surface waters only). The symbol * means below detection limit. (Source: Gulley and Parker 1985)

Chemical Variable	Minimum	Maximum	Mean	Standard Deviation	Median
pH	5.3	8.6	7.3	0.1	7.4
Conductivity (μS/cm)	3.0	57.0	15.6	1.8	11.8
Total Hardness (as mg/L $CaCO_3$)	0.4	59.7	6.8	1.5	4.0
Ca^{2+} (μeq/L)	*	199.6	104.8	20.0	64.9
Mg^{2+} (μeq/L)	*	156.4	24.7	4.9	16.5
Alkalinity (μeq/L, converted from mg/L $CaCO_3$)	8	760	110	20	74
Cl^- (μeq/L)	28.2	375.1	132.5	11.3	124.1
Total Dissolved Solids (mg/L)	2.7	70.8	13.8	1.8	10.8

Glacier, but is much more dilute (total hardness equal to 4.0 mg/L as $CaCO_3$). Like Schoolroom Lake, Delta Lake receives large contributions of glacial silt. Teton Glacier resides on granite, gneiss, and schist, whereas Schoolroom Glacier is on limestone and dolomite (Gulley and Parker 1985).

Williams and Tonnessen (in review) sampled seventeen high-elevation headwater lakes throughout and adjacent to GRTE in August, 1996. Although none were acidic, about half had ANC values near 50 μeq/L, and almost all had ANC below 200 μeq/L. About one-third had pH in the range of 5.8 to 6.0. These data do not suggest that chronic lakewater acidification has occurred but do suggest sensitivity, especially to episodic acidification. Of particular importance was the observed concentrations of NO_3^-, which were relatively high in many of the lakes sampled. Six had NO_3^- concentrations in the range of 5 to 10 μeq/L and three had NO_3^- concentrations greater than 10 μeq/L (to a maximum of 13 μeq/L). Given that the lakes were sampled at the end of the growing season, these data suggest that the capacity for biological uptake of N has been reached due to current levels of N inputs. It is likely that future increase in N deposition to these watersheds would contribute to higher lakewater NO_3^- concentrations and possible chronic acidification. Furthermore, the observed high NO_3^- concentrations suggest that additional water quality data should be collected during early summer, during the snowmelt period.

Of the lakes sampled more recently in GRTE by Williams and Tonnessen (in review), those having the lowest ANC were Surprise and Amphitheater (ANC=50 and 63 μeq/L, respectively). These were also the two lakes having the lowest specific conductance (5 and 6 μS/cm, respectively). Both had very low concentration of SO_4^{2-} (6 μeq/L) and no measurable NO_3^-. There were also, however, some relatively low-ANC (< 100 μeq/L) lakes that had NO_3^- > 5 μeq/L (Delta, Holly, and LOTC). Any of these five lakes might be good candidates for inclusion in future monitoring efforts.

3. Terrestrial

Management priorities for terrestrial resources in GRTE include wildlife and habitat protection, control of the 117 exotic plant species, and management of approximately 3.7 million visitors each year. Development and the impact of visitors on habitat quality are major concerns of resource managers. Project statements in the Resource Management Plan address elk, moose, bighorn sheep, raptors, small mammal, bird and fish populations. At present there are no research or monitoring projects underway focusing on air pollution impacts on vegetation and other terrestrial resources in GRTE.

D. AIR QUALITY RELATED VALUES

1. Aquatic Biota

Sensitive fish species in the park include both native and non-native salmonids. Recent studies have shown that native western trout are sensitive to short-term increases in acidity. For example, Woodward et al. (1989) exposed native western cutthroat trout to pH depressions (pH 4.5 to 6.5) in the laboratory. Freshly-fertilized egg, eyed embryo, alevin, and swim-up larval stages of development were exposed to low pH for a period of seven days. Fish life stages were monitored for mortality, growth, and development to 40 days posthatch. The test fish were taken from the Snake River in Wyoming. Reductions in pH from 6.5 to 6.0 in low-calcium water (70 μeq/L) did not affect survival, but did reduce growth of swim-up larvae. Eggs, alevins, and swim-up larvae showed significantly higher mortality at pH 4.5 as compared to pH 6.5. Mortality was also somewhat higher at pH 5.0, but only statistically higher for eggs.

It is unlikely that aquatic biota in GRTE have experienced adverse impacts to date from acidic deposition. This is because deposition of S and N are apparently low, and the available lake water chemistry data are not indicative of chronic acidification. However, in view of the high sensitivity of lakewater chemistry to adverse effects of possible future increases in acidic deposition, aquatic biota constitute important AQRVs within the park.

2. Terrestrial Biota

Vegetation is the resource which is most sensitive to ozone and SO_2, and a few tree species found in GRTE have been identified as potential bioindicators (see below). Ozone and SO_2 levels have probably not exceeded the NAAQS in GRTE. Baseline data on the condition of sensitive species would be helpful for future comparisons if pollutant levels increase. Monitoring sensitive receptors (those species with known sensitivity to one or more pollutants) by using detailed descriptions and classifications of leaf or plant injury would be useful for long-term evaluation of ecosystem health.

One of the most ozone-sensitive western tree species is ponderosa pine (*Pinus ponderosa*, especially var. *ponderosa*), for which extensive data are available on field (Miller and Millecan 1971, Pronos and Vogler 1981, Peterson and Arbaugh 1988) and experimental (Temple et al. 1992) exposures. The evidence for ozone impacts on ponderosa pine is based on observable symptoms and reduced growth (Peterson et al. 1991, Peterson and Arbaugh 1992) as well as physiological (Darrall 1989, Bytnerowicz and Grulke 1992) data. The cause-and-effect relationship, especially for trees growing in forests of southern California and the southern Sierra Nevada, is clear and quantifiable.

The well-documented symptomatology of pines makes lodgepole pine an appropriate bioindicator for ozone, even if it is less sensitive than ponderosa pine (which is not found at GRTE). Lodgepole pine is widespread in GRTE, and several areas in the park are suitable for establishing long-term monitoring plots.

Of the hardwood species present at GRTE, quaking aspen is the most sensitive to ozone. Aspen grows at various locations in riparian ecosystems and in fire- or avalanche-disturbed areas in the park. Numerous studies have documented the sensitivity of this species to ozone under field and experimental 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 aspen includes chlorosis, stippling, necrotic spotting, and leaf margin burn. Symptoms generally vary seasonally, with stippling being most prominent in the spring and black, bifacial (both leaf surfaces) necrosis appearing in late summer (J.P. Bennett, pers. comm.). Great care must be taken in distinguishing ozone symptoms from the effects of various pathogens and insect herbivores commonly found on this species.

Aspen is also considered to be sensitive to SO_2 and may be the best sensitive receptor for this gaseous pollutant. Injury is similar to that normally found for ozone (stippling, followed by bifacial necrosis), although SO_2 -induced injury rapidly bleaches to a light tan color (ozone injury remains dark) (Karnosky 1976). Diagnosis of SO_2 injury must be carefully differentiated from ozone injury.

Black cottonwood is another potential sensitive receptor for ozone (Woo 1996) which has symptoms similar to those of aspen. However, it is generally regarded as less sensitive to ozone than aspen. Neither of these hardwood species has the clarity of ozone symptomatology found in ponderosa pine.

A species list of native plants is available in the NPFlora database. Table IV-10 summarizes vascular plant species of GRTE with known sensitivity to ozone, SO_2 and NO_x. This table is based on a variety of sources from the published literature and other information. It should be noted that the various sources used a wide range of field and experimental approaches to determine pollutant pathology, and that sensitivity ratings are general estimates based on published information and our expert opinion. While it will not be possible for Park staff to collect data on all the species indicated

Table IV-10. Plant species of GRTE with known sensitivities to SO_2, ozone, and NO_x. (H=high, M=medium, L=low, blank=unknown). (Sources: Esserlieu and Olson 1986, Bunin 1990, Peterson et al. 1993, National Park Service 1994, Electric Power Research Institute 1995, Binkley et al. 1996, Brace 1996)

Species Name	SO_2 Sensitivity	O_3 Sensitivity	NO_x Sensitivity
Abies lasiocarpa	L	L	
Acer glabrum	H		
Achillea millefolium		L	
Agastache urticifolia		M	
Agoseris glauca	M		
Amelanchier alnifolia	H	M	
Angelica pinnata		L	
Arctostaphylos UVa-ursi	L	L	
Artemisia tridentata	L	L	
Betula occidentalis	M		
Bromus carinatus		L	
Bromus tectorum		M	
Calochortus nuttllii		L	
Ceanothus velutinus	L		
Cerocarpus ledifolius	M		
Cichorium intybus		L	
Cirsium arvense		L	
Clematis ligusticifolia	M		
Collomia linearis		L	
Conium maculatum		L	
Convolvulus arvensis	H		
Cornus stolonifera	M	L	
Crataegus douglasii	L		
Descurainia californica		L	
Descurainia pinnata		L	
Epilobium angustifolium		L	
Erigeron peregrinus		L	
Erodium cicutarium	L	M	
Festuca idahoensis	H		
Fragaria virginiana		H	
Galium bifolium		L	
Geranium richardsonii	M	M	
Hackelia floribunda	L		
Hedysarum boreale		M	
Helianthus anuus	H	L	
Juniperus communis	L		
Juniperus scopulorum	L		
Lemna minor	L		
Lonicera involucrata	L	H	
Medicago sativa		M	

Mimulus guttatus		L	
Oryzopsis hymenoides	M		

Table IV-10. Continued.

Species Name	SO_2 Sensitivity	O_3 Sensitivity	NO_x Sensitivity
Osmorhiza occidentalis		L	
Phacelia heterophylla		L	
Phyllodoce empetriformis		L	
Picea engelmannii	M	L	
Picea pungens	M	L	
Pinus contorta	M	M	H
Pinus flexilis	L		
Poa annua	H	L	
Poa pratensis		L	
Polygonum douglasii		L	
Populus angustifolia	M		
Populus balsamifera	M	H	
Populus balsamifera subsp. *trichocarpa*	M	H	
Populus tremuloides	H	H	
Potentilla flabellifolia		L	
Potentilla fruticosa		L	
Prunus virginiana	M	H	
Pseudotsuga menziesii	M	L	H
Rhus trilobata	L	H	
Ribes viscosissimum	M		
Rosa woodsii	M	L	
Rubus idaeus	H		
Rubus parviflorus		M	
Rumex crispus		L	
Salix scouleriana		M	
Sambucus melanocarpa		M	
Sambucus racemosa		L	
Senecio serra		H	
Shepherdia canadensis	L		
Sorbus scopulina	M		
Symphoricarpos oreophilus	M		
Taraxacum officinale		L	
Thalictrum fendleri		L	
Toxicodendron radicans	L	L	
Tragopogon dubius	M		
Trifolium pratense	L		
Trifolium repens		H	
Trisetum spicatum	M		
Urtica gracilis		L	
Vaccinium membranaceum		M	

Veronica anagallis-aquatica		L	
Vicia americana		L	
Viola adunca		L	

in Table IV-10, the list can be used by Park managers as a guide for species that could have visible symptoms. Of the many plant species in GRTE, it is likely that there are many other species which have high sensitivity to air pollution, but we currently have no information about them.

An inventory of lichen species with known sensitivity to ozone and SO_2 is summarized in Table IV-11. As in Table IV-10, this table is based on a variety of sources from the published literature and other information. It should be noted that diagnostic symptoms of air pollutant injury to lichens are difficult to identify, and that some species show reduced productivity or even mortality without exhibiting visible symptoms (Nash and Wirth 1988). One of the best sources of background information and guidelines for addressing the use of lichens as sensitive receptors of air pollution is Stolte et al. (1993). Inventories of lichen species distribution and abundance in GRTE would provide a better baseline for further assessment of potential impacts of air pollution on lichens in the park.

Table IV-11. Lichen species of GRTE with known sensitivity to SO_2 and ozone. (H=high, M=medium, L=low, blank=unknown). (Sources: Peterson et al. 1993, National Park Service 1994, Electric Power Research Institute 1995, Binkley et al. 1996)

Species Name	SO_2 sensitivity	Ozone sensitivity
Bryoria fuscescens	M	
Buellia punctata	L-M	
Caloplaca cerina	M-H	
Caloplaca flavorubescens	H	
Caloplaca holocarpa	M	
Candelaria concolor	M-H	
Candelariella vitellina	M	
Candelariella xanthostigma	M	
Cetraria pinastri	M	
Cladonia chlorophaea	M	
Cladonia coniocraea	M	
Cladonia fimbriata	M-H	
Cladonia gracilis	L-M	
Collema tenax	M	
Evernia mesomorpha	M	
Hyperphyscia adglutinata	M	
Hypogymnia physodes	M	
Hypogymnia tubulosa	H	
Lecanora chlarotera	M	

Lecanora dispersa	L	
Lecanora hageni	L-M	
Lecanora muralis	M	
Lecanora saligna	M	
Lecidea atrobrunnea	L	
Parmelia caperata	M	
Parmelia subargentifera	M	
Table IV-11. Continued		
Species Name	SO_2 sensitivity	Ozone sensitivity
Parmelia subaurifera	H	
Parmelia subolivacea		L
Parmelia sulcata	L-H	M-H
Peltigera canina	L	H
Physcia adscendens	M	
Physcia aipolia	M	
Physcia caesia	M	
Physcia dubia	M	
Physcia millegrana	M	
Physcia stellaris	M	
Physconia detersa	M-H	L
Rhizoplaca chrysoleuca	H	
Rhizoplaca melanophthalma	H	
Usnea hirta	M	
Usnea subfloridana	M-H	
Xanthoria elegans	M	
Xanthoria fallax	M-H	
Xanthoria polycarpa	M-H	L

E. RESEARCH AND MONITORING NEEDS

1. Deposition and Gaseous Pollutants

GRTE and YELL are within the same airshed, and although data from YELL monitoring sites can be used to infer GRTE air quality, there also may be local sources of CO, SO_2, NO_x, VOC, and particulates which influence air quality within GRTE. Also, transported pollutants from southeastern Idaho and Utah may be deposited at higher concentrations in GRTE than YELL due to topography and airflow-circulation patterns. Monitoring of SO_2, NO_x, and ozone at GRTE would provide a more accurate picture of air quality in the park. Even a short period of monitoring would be helpful because the data could be calibrated with YELL data, giving YELL data predictive power for GRTE in the future. Given the observed greater sensitivity of aquatic resources in GRTE, as compared to YELL, to potential future increases in sulfur or nitrogen deposition, it would be desirable to monitor deposition within GRTE. Although we do not view this as a critical need at this time, installation of a wet deposition monitoring station in GRTE would be useful. If this is done, we recommend installation at the highest elevation that is practical in the mid-section of the park (in the vicinity of Jenny Lake).

Little is known of the present threats of ozone, SO_2 and NO_2 on native vegetation in GRTE. Ozone data are currently collected at YELL using a continuous analyzer. A second continuous analyzer installed seasonally (during the summer months) at a high elevation site in YELL or preferably in GRTE for three consecutive years would provide valuable information on the diurnal pattern of ozone exposure within the park. Studies of ozone profiles done in other mountainous areas including the Swiss Alps (Sandroni et al. 1994), Sierra Nevada (Miller et al. 1989) and Cascade Mountains (Brace 1996, 1998) have shown that topography and elevation can influence the diurnal exposure. Low-elevation sites have diurnal profiles characterized by low ozone levels during the nighttime and early morning hours, maximum levels during the mid or late afternoon, and low levels again in the evening. High elevation sites typically have lower maximum ozone concentrations than low elevation sites, but ozone remains elevated during the morning and nighttime hours. Plant species at higher elevations may be at particular risk from exposure to elevated levels of ambient ozone during the morning hours when they are physiologically active.

It would be useful to determine the spatial variability in ozone concentration across an elevational gradient. A network of passive ozone samplers could be established to compare ozone measurements from different locations in GRTE. During the summer months, a network of passive ozone samplers installed for three consecutive years would provide the data to describe spatial patterns and a reference point in time. The network should include one transect of four sites placed along an elevational gradient, ranging from Moose to above treeline. An additional transect of three sites along a north-south transect would provide data on the spatial variability of weekly ozone exposures in different areas of the park.

SO_2 monitoring is important considering the proximity of regional point sources of SO_2. An SO_2 analyzer in operation for two years every five years would provide important data on long-term impacts of regional SO_2 emissions on air quality in GRTE. SO_2 emissions from industrial sources upwind of GRTE may be expected to grow as demands on power generation, mining operations and oil and gas production increase. By installing an analyzer now, it will be possible to assess both current air quality and future impacts.

2. Aquatic Systems

Gulley and Parker (1985) recommended continued monitoring of eight alpine lakes in the park, selected on the basis of human use (high and low) and lake type defined on the basis of cluster analysis. The lakes recommended for monitoring ranged from those that we expect to be highly sensitive to potential acidification (Surprise Lake, Solitude Lake, Kit Lake), to those we expect to be moderately sensitive (Holly Lake, Grizzly Bear Lake), to one expected to be insensitive (Snowdrift Lake), based on calcium concentrations, specific conductance, and alkalinity. We suggest a monitoring strategy that focuses only on the highly-sensitive systems.

We believe that there is insufficient information on aquatic resources relative to air pollution effects for GRTE, particularly in the high mountain areas which are sensitive to acidification effects. Based on the limited data available, it appears that at least some and perhaps many of the high-elevation lakes in the park have very low specific conductance (< 10 μS/cm) and low ANC and are presumably sensitive to acidic deposition effects. We recommend a modest monitoring project to ascertain the major ion chemistry and biology of some high-elevation lakes in the park that are presumed sensitive to acidification. Based on the results of that monitoring effort, we recommend formulation of a viable long-term monitoring strategy for a small group (perhaps two or three) of the high-elevation lakes.

There is also a need for additional episodic monitoring of surface water chemistry in sensitive surface waters in GRTE. Such monitoring would entail collection of lakewater and streamwater chemistry soon after ice out on high-elevation lakes. In many cases, safety considerations prevent sampling during the early phases of snowmelt, but collection of monitoring data in late June or early July would be very useful. These data would help to 1) clarify the extent to which episodic acidification occurs under current deposition, 2) quantify the relative roles of S and N in episodic acidification of aquatic resources in the park, and 3) establish a baseline for episodic acidification for comparison with future years when deposition may be higher. We recommend a sampling program of about 10 lakes and streams, distributed across the known (and presumed) sensitive portions of the park, to be sampled three times per year at approximately monthly intervals for three years from the earliest practical sampling date for each watershed.

Based on the observed high degree of sensitivity of several lakes in GRTE to future increase in acidic deposition, we recommend modeling of one or more watersheds to better quantify the loading rates for S and N above which adverse impacts would be expected, based on current scientific understanding. Such a modeling effort is currently underway (contact: K. Tonnessen, ARD, Denver).

3. Terrestrial Systems

Vegetative monitoring for pollutant effects is not recommended for GRTE at this time. Past and current gaseous pollutant monitoring efforts have not revealed a serious threat to terrestrial resources. If SO_2 or NO_X emissions outside the park increase substantially in the future, vegetation monitoring should be initiated. In this event, three levels of monitoring, with increasing amounts of effort and expense, are presented in Appendix A. These monitoring activities are based on methods and protocols developed by the USDA Forest Service and National Park Service. Species and locations recommended for monitoring are listed below.

Lodgepole pine and quaking aspen plots could be established in the general vicinity of Moose. These plots should be located away from human-use areas and preferably in areas with good air flow. Lichen surveys (perhaps every five years) could be conducted to determine the total lichen flora at GRTE. Lichen monitoring methodology is described in Appendix A. A large-scale monitoring effort for lichens is not justified at this point, although protocols and guidelines in Stolte et al. (1993)

can be consulted for information on assessing injury.

If necessary, additional tree plots should be located: (1) near Moran, and (2) approximately half way between Moose and Moran. These areas and the Moose area are all located along the Snake River drainage where maximum air flow can be expected. Locations of these plots can be changed to other sites in GRTE if ambient air quality data indicate that other areas have a higher risk of pollutant effects. Additional plots that evaluate other conifer species, such as Engelmann spruce, Douglas-fir, and whitebark pine could also be established. If monitoring of herbaceous plants is desirable in the future, candidate species in GRTE include strawberry (*Fragaria virginiana),* skunkbush (*Rhus trilobata*), and red clover (*Trifolium repens*). Herbaceous species monitoring methodology is described in Appendix A.

4. Visibility

Visibility monitoring is not currently conducted at GRTE. Therefore, establishment of monitoring (particle and optical) would help assess visibility conditions at the park. Due to elevational differences between the YELL visibility monitoring site and GRTE, the extent to which the monitoring data collected at YELL represent conditions at GRTE remains uncertain.

V. YELLOWSTONE NATIONAL PARK

A. GENERAL DESCRIPTION

Yellowstone National Park (YELL) was established as the world's first national park in 1872. YELL, comprised of 1.1 million ha, is at the center of approximately 6 million hectares commonly referred to as the Greater Yellowstone Ecosystem. These lands are managed by the National Park Service, USDA Forest Service, U.S. Fish and Wildlife Service, Bureau of Reclamation, Bureau of Land Management, three states (Wyoming, Montana, and Idaho), and many private landholders. Efforts are underway to manage the natural resources of this region at large spatial scales, with coordination among agencies and stakeholders, sharing of data and information, and cooperative management activities. Within YELL, natural resource management focuses on preserving the components and processes of naturally evolving ecosystems (YELL 1995). YELL ranges in elevation from 1,600 to 3,500 m, and contains several broad volcanic plateaus, and parts of three mountain ranges: the Absaroka Mountains in the eastern, Gallatin Mountains in the northwestern, and Red Mountains in the southern portions of the park.

The spectacular geological features of YELL, geysers, hot springs, mud pots, and fumaroles, provided the initial motivation for creation of the park. Forested ecosystems dominated by lodgepole pine (*Pinus cortorta*) occupy about 80% of the park, with various other grassland, alpine, and riparian vegetation interspersed with forest. This mosaic of ecosystems provides habitat for a wide range of megafauna including bison (*Bison bison*), elk (*Cervus elaphus*), moose (*Alces alces*), grizzly bear (*Ursus arctos* subsp. *horribilis*), black bear (*U. americanus*), mule deer (*Odocoileus hemionus*), white-tailed deer (*O. virginianus*), bighorn sheep (*Ovis canadensis*), pronghorn (*Antilocapra americana*), mountain lion (*Felis concolor*), and gray wolf (*Canis lupus*). Wildlife are found in densities rarely observed in other areas of North America. YELL’s high quality aquatic habitats, including Yellowstone Lake and many trout-bearing streams, are also important features of the park's natural resources. Finally, the extensive backcountry of the park provides opportunities for hiking and solitude in a wilderness setting. A special feature of the current YELL landscape is the extensive area (400,000 hectares) that was burned in fires that occurred in 1988. The effects of this extreme fire event are a prominent ecological and visual component of park ecosystems and offer unique opportunities for observing and interpreting post-fire biological phenomena.

1. Soils and Geology

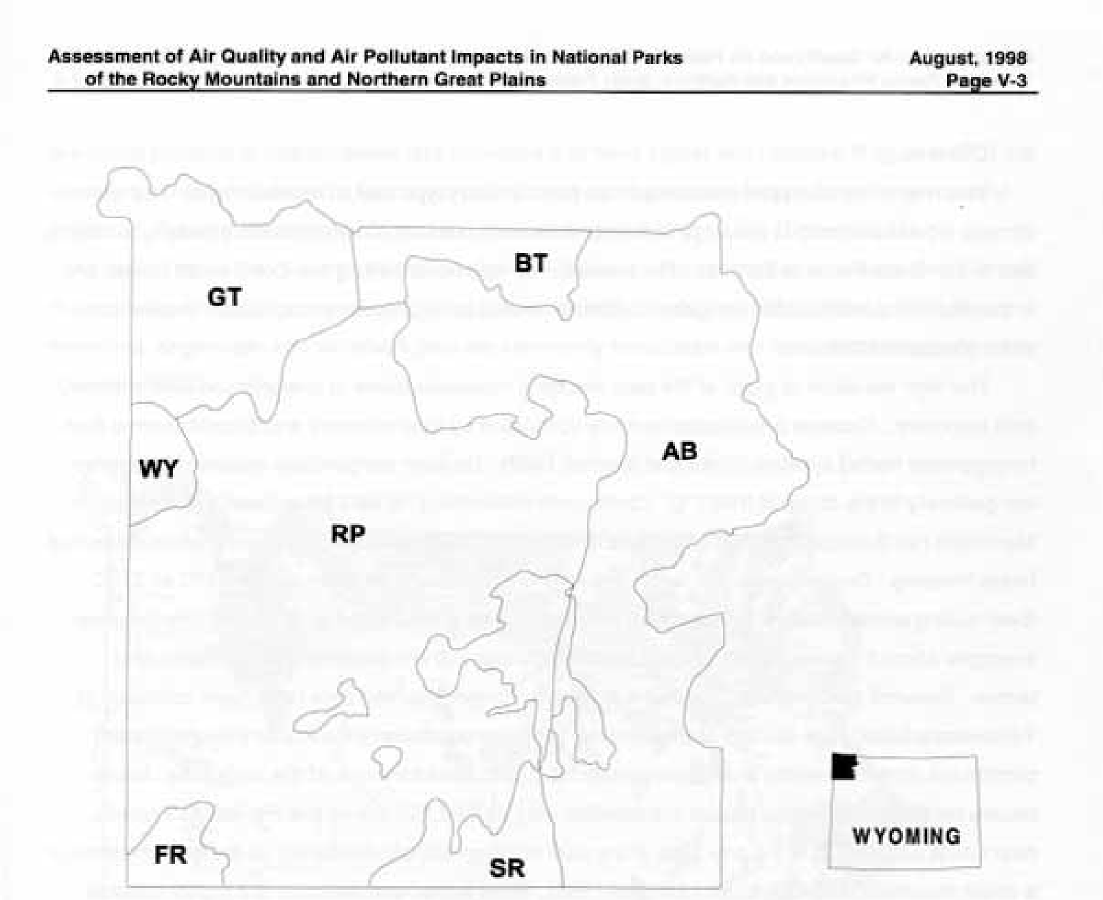
YELL is situated in an area comprised primarily of uplifted mountains of sedimentary rocks and the aftermath of two volcanic events (Romme 1982, Fournier et al. 1989). During the Eocene, volcanic activity created mountains in the north and along the eastern border of the current park. The volcanism formed mostly andesitic rocks that weather to soils relatively high in nutrients. During Quaternary times, a large caldera exploded in the central part of the park, leaving behind the lower one-third of a huge volcanic mountain (Mt. Washburn). The ash flows and later lava flows were rhyolitic in composition and are nutrient-poor. They produced the high plateau of the southwestern and central portions of the park (Figure V-1) (Despain 1990).

Soils derived from rhyolitic materials generally are more sensitive to acidic deposition because base cation concentrations tend to be lower. For example, calcium is much more abundant in andesite than in rhyolite. This, together with the higher clay content, provides for higher cation exchange capacity in the andesitic soils (Despain 1990).

Rock materials were transported from their places of origin by the glaciers that carved much of the Yellowstone area during the various periods of glaciation. The valleys of the Lamar River and the lower Yellowstone River were filled by a large glacier that transported parent material long distances and left behind deep layers of till. Elsewhere in the park, areas were generally covered by ice caps rather than large glaciers and the tills were not transported very far (Despain 1990).

The Yellowstone Plateau has an average elevation of about 2,500 m. The highest elevation in YELL is 3,500 m at Eagle Peak near the southeast corner of the park. The lowest point in the park (1,600 m) is in the northwest corner near Gardiner, Montana. The park straddles the point of contact of three major physiographic provinces (Clements 1910). These include the Middle Rocky Mountain Province which includes most of the park, the Northern Rocky Mountain Province, and the

Basin and Range Province. The Middle Rocky Mountain Province is characterized by mountains of



uplifted blocks with large intervening basins. Granitic rocks occur at the higher elevations and sedimentary rocks on the flanks of the mountains. The northern Rocky Mountain Province is characterized by high steep mountains and narrow basins of sedimentary rocks. It extends into the northern section of the park. Bordering the park on the west is the Basin and Range Province, which includes numerous mountain blocks and intermountain basins. Most of these mountains were formed along tension faults (Despain 1990).

The geyser basins of YELL are probably the best examples in the world of high-temperature geothermal activity. Boiling hot springs and geysers discharge significant quantities of circumneutral to slightly alkaline water. Such hydrothermal activity is most pronounced at the intersection of faults, especially within and around the Yellowstone Caldera. The waters are mostly chloride-rich and deposit siliceous sinter that accumulates into thick mounds and terraces. Acid-sulfate boiling pools and mud pots also occur in some of the geyser basins. Thermal waters at Mammoth are somewhat different. They flow to the surface through a thick sequence of sedimentary rocks that includes limestone, dolomite, and gypsum-bearing shales, and are therefore rich in sulfate and bicarbonate (Fournier 1989).

AB	Absaroka region is predominantly andesitic lava flows and breccia, with basalt, and some occurrence of rhyolite, sandstone, and limestone
BT	Beartooth region is a mix of Precambrian granites, Paleozoic and Mesozoic sandstones and shales, and Tertiary/Quaternary volanics
FR	Falls River region is Quaternary rhyolite and basalt, frequently overlain by alluvial and glacial deposits
GT	Gallatin region is Precambrian granites, Paleozoic and Mesozoic limestones, sandstones, and shales, Tertiary/Quaternary volcanics
RP	Rhyolite plateau region is predominantly Tertiary and Quaternary rhyolite flows
SR	Snake River region is Paleozoic and Mesozoic limestones, sandstones, and shales, with some outcroppings of Tertiary rhyolite and andesite
WY	West Yellowstone region is rhyolite overlain by alluvial, glacial, and lacustrine deposits

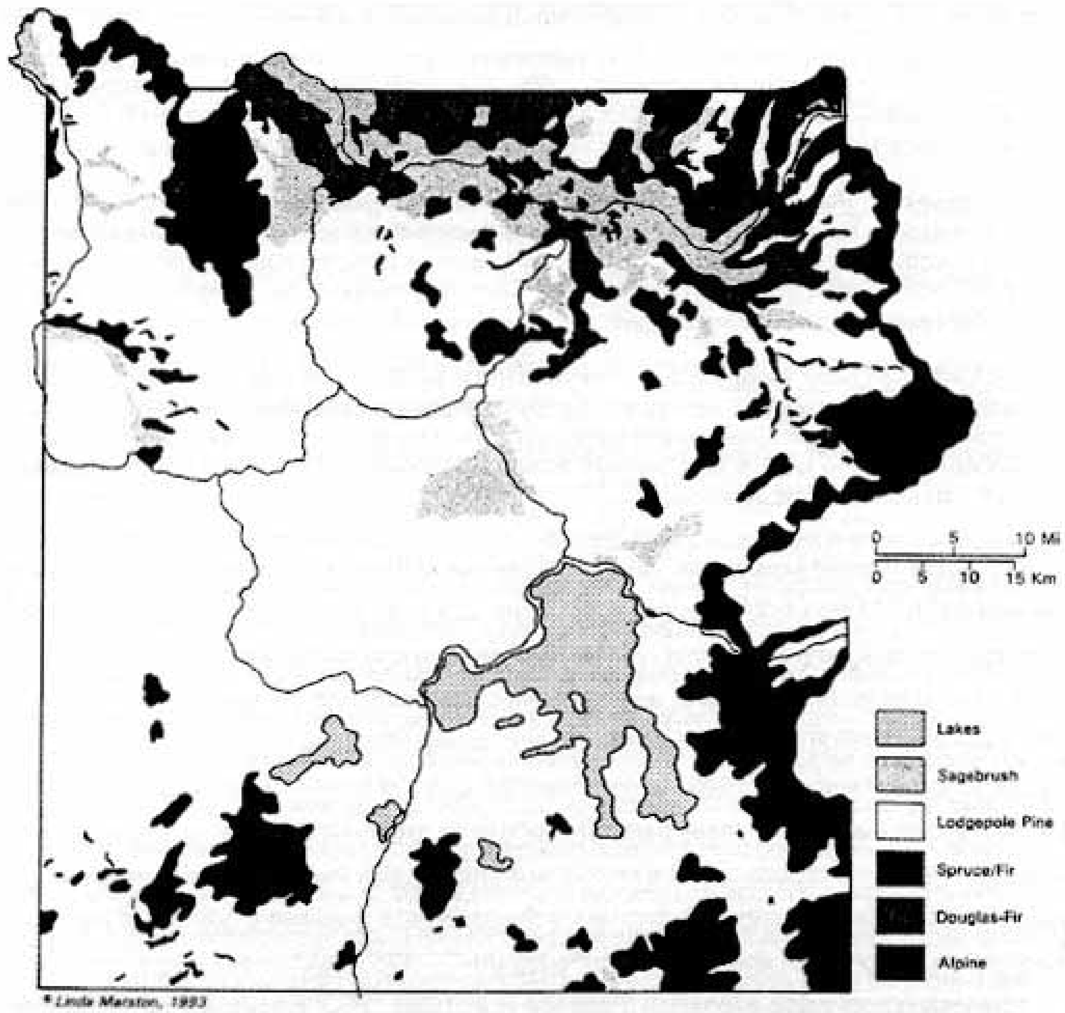


Figure V-2. Major vegetation types, roads, and large lakes in YELL. Based on classifications from a map prepared by Despain (1990).

Figure V-1. Regional-geological map of YELL (adapted from Cox 1973 by Gibson et al. 1983).

2. Climate

Two major climatic types are found in the park: a valley type and a mountain type. The valley climatic type is common to the large valleys and central plateaus. This climate is generally similar to that of the Great Plains to the east. The mountain climate occurs along the Continental Divide and in the Absaroka and Gallatin Ranges. It is characterized by high winter precipitation, mostly as snow (Despain 1990).

The high elevation of much of the park results in moderate levels of precipitation and relatively cool summers. Summer precipitation is more dominated by local showers and thunderstorms than by organized frontal systems (Dirks and Martner 1982). Daytime temperature maxima in summer are generally in the range of 21-27°C. Continuous meteorological data have been collected at Mammoth Hot Springs since 1887. Winters are cold with daily temperature maxima often remaining below freezing. Temperatures can reach extreme values in the park, from about -54°C to 37°C. Even during summer, below freezing temperatures can be encountered at any time. Precipitation averages about 64 cm annually, ranging between 30 and 100 cm depending on elevation and terrain. Snowfall reaches 5 to 10 m in the Absaroka Range. Rainfall data have been collected at Yellowstone Lake since the turn of the century. Winter precipitation (November through March) contributes most to the total annual precipitation and accounts for much of the variability. Snow course records suggest that annual precipitation may exceed 200 cm on the Pitchstone Plateau near Lewis Lake. This is the only area of the park not immediately downwind (in the rain shadow) of a major mountain range (Dirks and Martner 1982). Wind speed and direction are highly variable within the park, and are strongly dependent on local topography and elevation. Prevailing winds are from the southwest at the meteorological tower location.

3. Biota

The flora of YELL includes over 1,200 vascular plant species, including about 150 non-native species, and over 180 species of lichens (Eversman 1990). Most of these plants are documented in the park's herbarium. Vegetation and habitat maps and descriptions have been compiled for the park at a 1:125,000 scale. The vegetation has been altered drastically, at least with respect to age class and to some extent cover type, due to the extensive fires of 1988. Burned area and burn intensity maps have been produced at large scale and at 1:20,000.

Approximately 80% of the land area of YELL is forested. Lodgepole pine and some limber pine (*Pinus flexilis*) occupy about 700,000 ha of the total area of the park. Subalpine fir (*Abies lasiocarpa*) and Engelmann spruce (*Picea engelmannii*) are found at higher elevations and higher soil moisture sites at lower elevations; Douglas-fir (*Pseudotsuga menziesii*) is found at low elevations in the northern part of the park; whitebark pine (*Pinus monticola*) is found at higher elevations; and a small amount of aspen (*Populus tremuloides*) and cottonwood (*Populus *balsamifera* subsp. *trichocarpa*) are found primarily in riparian areas and drainages that have higher soil moisture (Figure V-2). Low-elevation nonforested areas of the park are dominated by sagebrush (*Artemisia* spp.) and several grass species such as wild rye (*Elymus* sp.) and needlegrass (*Stipa* sp.). High-elevation nonforested areas are dominated by distinctive low-stature plants adapted to an alpine environment. Subalpine fir and lodgepole pine tend to be associated with rhyolitic parent materials, while mesic meadows, sagebrush, and whitebark pine are commonly associated with andesitic parent materials (Despain 1990).

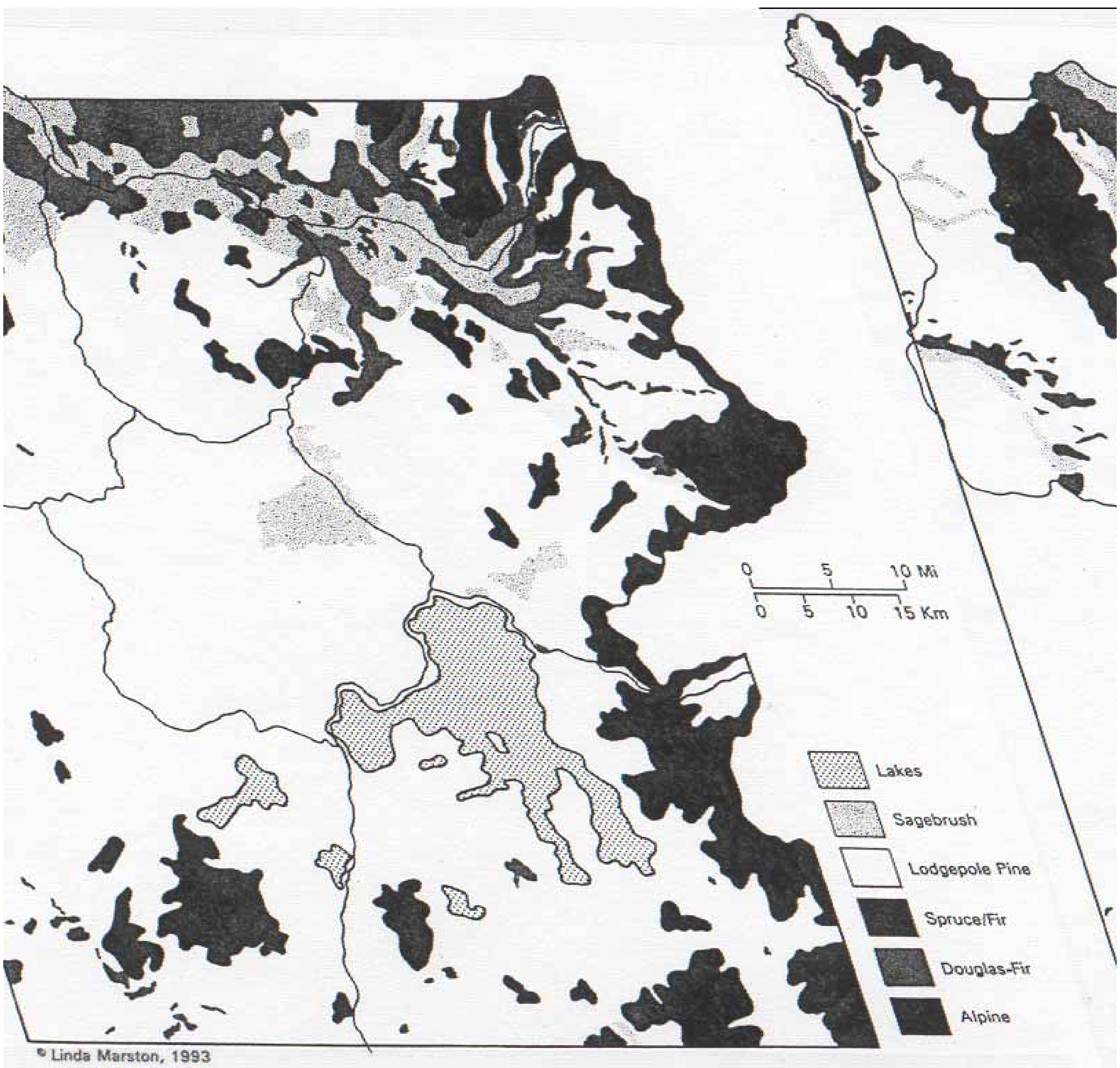


Figure V-2. Major vegetation types, roads, and large lakes in YELL. Based on classifications from a map prepared by Despain (1990).

Despain (1990) described five "geovegetation provinces" in the park:

- Gallatin Range -- Located in the northwest, this province occupies 7% of the park. It contains andesitic and sedimentary rocks, which weather to soils with high clay content. The province is 75% forested, with the dominant species subalpine fir, whitebark pine, Engelmann spruce, and Douglas-fir. Idaho fescue (*Festuca idahoensis*)and bearded wheatgrass (*Elymus trachycaulus*) dominate meadow vegetation.

- Absaroka Range -- Located along the eastern edge and southeast, this province occupies 32% of the park. Absaroka volcanics predominate. The province is 77% forested, with the dominant species lodgepole pine, whitebark pine, subalpine fir, and Engelmann spruce. Idaho fescue and bearded wheatgrass dominate meadow vegetation.

- Central Plateaus -- Located mostly in the central region, this province occupies 34% of the park and is underlain by Quaternary rhyolite. The mostly flat and undulating plateaus are 90% forested, with the dominant species lodgepole pine, with some subalpine fir and Engelmann spruce.

- Southwestern Plateaus -- Located in the southwest, this province occupies 18% of the park. This province is underlain by relatively recent rhyolitic flows with additional basalt flows. The flat to undulating plateaus are dominated by lodgepole pine, subalpine fir, and whitebark pine. Nonforested types are dominated by Idaho fescue and tufted hairgrass (*Deschampsia caespitosa*).

- Yellowstone-Lamar River Valleys -- Located in the north-central area, this province occupies 9% of the park area. The valleys are mostly filled with glacial drift comprised of andesitic and sedimentary materials. Nonforested vegetation dominates this province, with big sagebrush, Idaho fescue, and bearded wheatgrass the most common species. Douglas-fir is the dominant forest species.

The diversity of wildlife species in YELL is well-known to the scientific community as well as to the general public. There are 44 mammal species and 279 bird species recorded in YELL. Several of these species have been the subject of extensive research and monitoring. There are six species of reptiles, four species of amphibians, and 13 native species of fish in the park.

Peterson et al. (1992) sampled eight sites in and around YELL and GRTE on several occasions during the spring and summer of 1991 to determine the status of amphibian populations. The amphibian species that occur in these parks are apparently experiencing problems elsewhere within their ranges. Data were collected on the amphibians present, plus the physical and biological conditions at each site, including elevation, water chemistry, and presence of fish. All of the sampled ponds had pH $\geq$ 6.8 and sufficient buffering (ANC $\geq$ 175 μeq/L) so that they were not at risk of acidification from acidic deposition (Peterson et al. 1992). Four species of amphibians were found: western toad (*Bufo boreas*), spotted frog (*Rana pretiosa*), western chorus frog (*Pseudacris triseriata*), and tiger salamander (*Ambystoma tigrinum*).

4. Aquatic Resources

YELL encompasses near-pristine watershed areas and contributes to two of the nation's farthest reaching drainages: the Missouri and Columbia Rivers. Surface water resources in the park include about 600 streams and 175 lakes. There are about 4,400 km of free-flowing rivers and streams. Four large lakes (Yellowstone, Shoshone, Lewis, and Heart Lakes) account for about 94% of the park's lake surface. The largest lake in the park is Yellowstone Lake, which is 92 m deep and 386 km^2 in area. Major rivers include the Yellowstone, Snake, Lewis, Madison, Gibbon, Firehole, Gardiner, and Lamar Rivers. Water quality varies throughout the park, mostly as a function of geologic terrain and the influence of thermal features. Natural geothermal discharges, which are quite common in many portions of the park, affect the pH, alkalinity, temperature, salinity, sulfate concentrations, and base cation concentrations of drainage waters. Snowmelt is an important contributor to hydrologic budgets of watersheds in the park, and water quality therefore tends to vary seasonally.

Yellowstone Lake is noteworthy in a number of respects. The largest high-altitude lake in North America, it lies mostly within the Yellowstone Caldera, which has some of the highest measured geothermal heat fluxes in the world (Klump et al. 1988). Hydrothermal springs and hot gas fumeroles occur within the lake. Enhanced biological activity occurs around the geothermal vents which are characterized by high temperature, anoxia, and high concentrations of dissolved nutrients. Microbial communities are dense, as are the populations of oligochaete worms near fumeroles in the warm sediments.

Two subspecies of cutthroat trout (*Oncorhynchus clarki*) are native to the park: the Yellowstone cutthroat (*O. c. bouvieri*) and the west slope cutthroat (*O. c. lewisi*). In addition, non-native fish, including lake trout (*Salvelinus namaycush*), have been introduced. Fishing management is attempting to manage aquatic resources as functional components of the Greater Yellowstone Ecosystem and to preserve and restore native species and aquatic habitats.

The Yellowstone cutthroat was originally widely distributed in the intermountain region (Varley 1979), although its range is now greatly reduced. Four recognized strains occur in the park. The Yellowstone Lake strain, at one time the largest inland cutthroat trout population in the world, declined markedly in the 1950's and 1960's (Jones et al. 1986). A series of restrictive fishing regulations have been implemented. The McBride Lake strain is able to efficiently utilize available food resources in marginal oligotrophic environments. The Sedge Creek strain developed after Sedge Creek and Bear Creek were effectively isolated from Yellowstone Lake by Turbid Lake, a geothermal body of water, when the lake stage of Yellowstone Lake decreased. The toxicity of Turbid Lake prevents fish from migrating downstream. The Heart Lake cutthroat is the only piscivorus strain of Yellowstone cutthroat trout. The status of the West Slope cutthroat in the Upper

Missouri River has been greatly altered by the introduction of other salmonids. No verified pure populations of west slope cutthroat trout are known to exist in the park (Mary Hektner, YELL, pers. comm.).

B. EMISSIONS

YELL is located in the northwestern corner of Wyoming surrounded by Bridger-Teton, Shoshone, and Targhee National Forests and lies 10 km north of Grand Teton National Park. There is little industrial activity and low population in this portion of the state, resulting in good air quality. Most of the industrial activity in Wyoming is in the eastern counties near the cities of Gillette and Casper, and in the southwestern counties around Rock Springs. Oil and gas processing, electric utility power plants and industrial fossil-fuel combustion in southwestern Wyoming and southeastern Idaho are the major sources of gaseous pollutants in the YELL area. Annual emissions of gaseous SO_2, NO_x and VOC are primarily from fossil-fuel combustion by industrial sources (Tables II-2, II-3 and II-4), and levels are moderate relative to other western states.

Point sources of SO_2 , NO_x, and VOC located within 150 km of YELL (with emissions exceeding 100 tons/yr) are listed in Table V-1. Adjacent counties in Idaho and Montana are included. In Wyoming, SO_2 emissions are mainly from oil and gas refineries, the largest of which is Amoco Production Co. (emitting over 1,200 tons/yr) located in Elk Basin, 100 km east of YELL. The largest regional sources of SO_2 within 150 km of YELL are the Monsanto Company and Nu-West Industries, both mining operations located approximately 100 km south of YELL in Caribou County, southeastern Idaho. The annual emissions from these companies (approximately 9,000 tons/yr combined) may have an impact on resources in YELL.

Sources of NO_x in Wyoming are electric utilities and industrial fossil-fuel combustion (Table II-3). Current point sources of NO_x within 150 km of YELL are listed in Table V-1. The largest regional source of NO_x is Monsanto Corporation located in Caribou County, southeastern Idaho. Major stationary sources of VOC emissions, an important ozone precursor, are relatively low in Wyoming. However, there are thousands of small VOC and NO_x sources that cumulatively may add up to much higher emission totals (T. Blett, pers. comm.).

Seasonal increases in carbon monoxide (CO) are a concern in areas of high snowmachine use within YELL. Since the mid 1970's, snowmachine use has been an increasingly popular winter tourist activity. During the winter of 1993-94, over 87,000 tourists visited Old Faithful by snowmachine (Fussell 1997). Emissions from snowmachines are not regulated, and the vehicles are not required to utilize emission control equipment. Consequently, emissions of CO and unburned hydrocarbons from snowmachines can exceed those of automobiles (L.M. Snook pers. comm.). During winter 1994-95, one-hour samples taken near the west entrance station of YELL exceeded 35 ppm at two sites, and the 8-hour average CO concentration exceeded 8 ppm at one

Table V-1. Point sources (tons/yr) of SO_2, NO_x, and VOC (annual emissions exceeding 100 tons/yr of at least one pollutant) within 150 km of YELL. (Source: Idaho Department of Health and Welfare, unpublished data; Martin 1996; Wyoming Department of Environmental Quality 1995)

	SO_2	NO_x	VOC
Wyoming			
Amoco Co.	1,218	603	208
Marathon Oil	21	7	869
Oregon Basin Gas	455	25	49
Questar Pipeline		100	10
Williams Field Services (3 sites)		1,381	740
Williston Basin IPC		162	69
Idaho			
Ash Grove Cement	889	802	31
Basic American Foods	498	174	4
Basic American Foods	193	37	54
Chevron Pipeline			437
Idaho National Engineering Labs	91	1,229	35
Idaho Pacific Corp.	131	156	1
Idaho Supreme Potatoes	136	151	1
J.R. Simplot - siding facility	2,554	646	131
Monsanto Co.	4,703	2,021	
Nu-West Industries, Inc.	4,369		
Pillsbury Co.	1	156	7
Montana			
Cenex	2,865	892	892
Conoco	959	700	990
Exxon Co.	8,738	736	1,082
Holnam Inc.	32	1,330	2
Luzenac America	25	242	25
Montana Power Co.		100	25
Montana Power Co.	6,439	3,467	24
Montana Sulfur	3,422	11	
Western Sugar	486	360	17

site (NPS 1995). The health of park staff at entrance kiosks and of snowmachine users in groups may be at risk.

C. MONITORING AND RESEARCH ACTIVITIES

1. Air Quality

a. <u>Wet Deposition</u>

Precipitation volume and chemistry have been monitored at Tower Junction since 1980 by NADP/NTN (Figure V-3). Annual precipitation amounts are generally in the range of 30 to 45 cm per year at this site. The concentration of SO_4^{2-}, NO_3^-, and NH_4^+ are low, with each generally below 10 μeq/L (Table V-2). The combined low amount of precipitation and low concentrations of

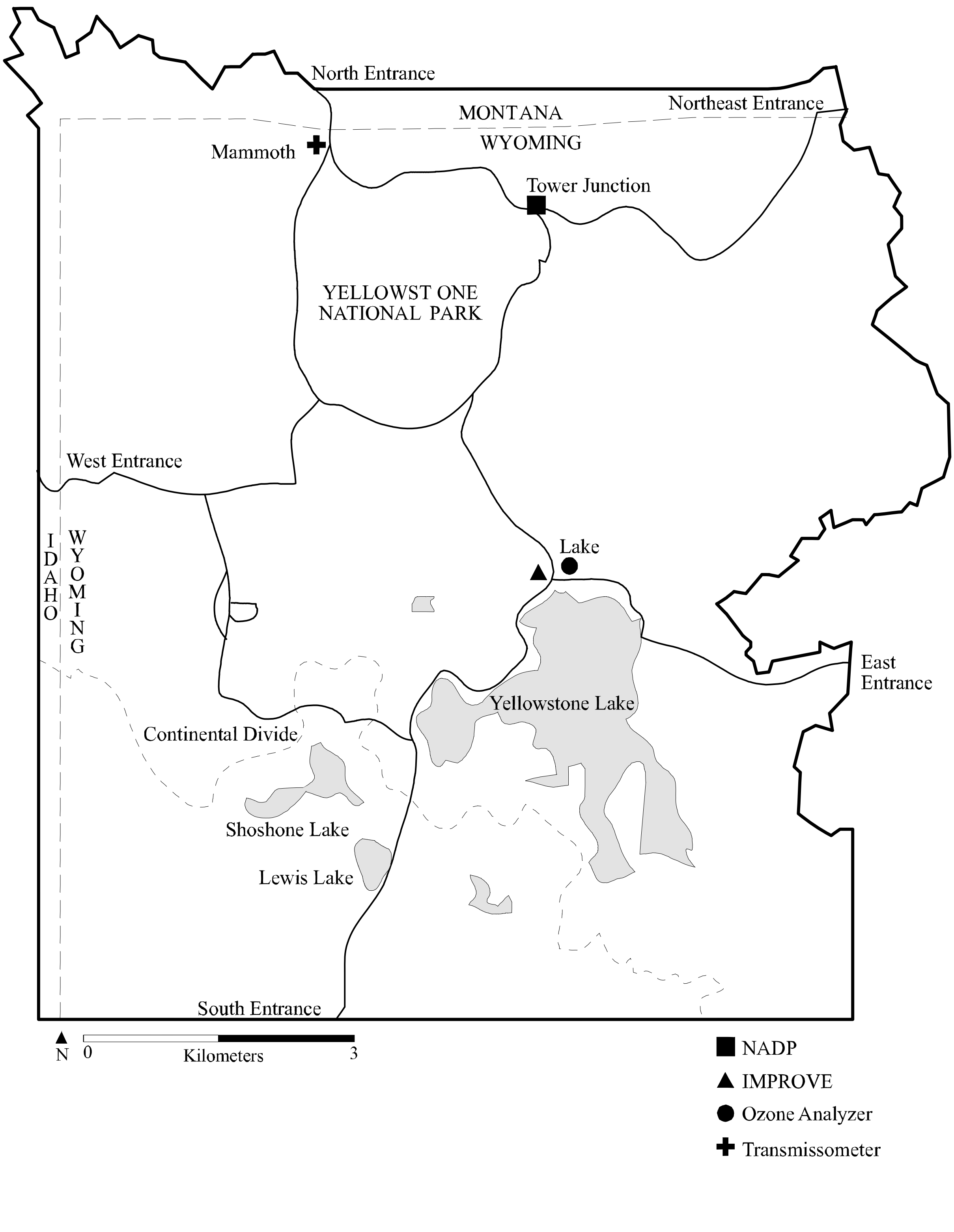


Figure V-3. Air quality monitoring locations at YELL. NOTE: ozone analyzer was moved in 1995.

Table V-2. Wetfall chemistry at the NADP/NTN site at Tower Junction, YELL. Units are in μeq/L, except precipitation (cm).										
Year	Precip	H^+	SO_4^{2-}	NH_4^+	NO_3^-	Ca^{2+}	Mg^{2+}	Na^+	K^+	Cl^-
1995	38.7	5.3	5.8	7.7	7.7	4.8	0.9	1.8	0.5	1.8
1994	36.3	4.6	9.7	9.8	10.3	12.5	2.3	3.1	0.9	2.5
1993	39.6	2.6	8.6	8.3	8.1	6.6	1.4	2.6	0.4	2.1
1992	45.0	2.7	8.1	8.3	8.5	12.1	2.0	2.0	0.6	2.4
1991	45.5	2.6	11.0	6.8	9.7	14.8	2.7	4.1	0.6	2.5
1990	36.6	3.4	12.0	9.4	11.6	18.4	3.2	3.5	1.2	3.8
1989	42.9	2.1	8.7	8.8	9.4	12.2	2.3	3.0	0.6	3.3
1988	27.3	1.9	7.1	3.5	4.6	9.9	1.7	3.6	0.7	2.6
1987	8.7	2.6	4.5	3.2	4.6	5.7	1.1	2.5	0.3	1.9
1986	38.8	2.8	8.1	5.7	6.7	9.0	1.9	2.5	0.7	2.7
1985	36.1	3.0	6.6	2.9	5.4	6.4	2.0	2.0	0.6	2.5
1984	37.5	5.6	12.2	7.2	9.0	14.4	3.9	4.0	1.4	4.0
1983	34.5	3.6	11.0	5.0	5.8	10.9	2.8	3.8	1.1	3.2
1982	57.1	6.0	12.7	5.0	7.5	11.8	4.1	4.1	1.2	4.0
1981	34.1	4.4	32.6	9.6	11.7	21.9	8.2	26.1	1.6	13.4
1980	24.7	7.4	22.2	11.7	14.6	14.4	3.1	6.8	1.3	5.9

acid-forming precursors results in very low levels of sulfur and nitrogen deposition. Sulfur deposition is generally well below 1 kg S/ha/yr and N deposition is seldom above this amount (Table V-3).

Snowpack samples were collected in March or April of 1993 through 1998 at five sites in YELL: West Yellowstone, Lewis Lake Divide, Sylvan Lake, Canyon, and Twenty-one Mile. Sulfate concentrations in snow were low at all sites, with five-year (1993-1997) mean concentrations ranging from 4.7 μeq/L at Sylvan lake to 8.0 μeq/L at West Yellowstone. Mean concentrations of NO_3^- in the snowpack ranged from 4.2 μeq/L at Canyon to 5.0 μeq/L at Lewis Lake Divide. Ammonium concentrations were similar to NO_3^- concentrations, with mean values ranging from 4.2 μeq/L at Canyon to 7.0 μeq/L at West Yellowstone (G.P. Ingersoll, pers. comm.). Although higher than comparable measurements made in GLAC, these snowpack ionic concentrations were similar to those in GRTE and are considered low.

b. Occult/Dry Deposition

There are no data currently available on dry or occult deposition fluxes of S or N to sensitive resources within YELL. However, we expect that the contributions of both dry and occult deposition of S and N are low relative to the wet deposition amounts summarized in Table V-3. This is

Table V-3. Wet deposition (kg/ha/yr) of sulfur and nitrogen at the NADP/NTN site at Tower Junction, YELL.

Year	Sulfur	NO_3-N	NH_4-N	Total Inorganic N
1995	0.4	0.4	0.4	0.8
1994	0.6	0.5	0.5	1.0
1993	0.5	0.4	0.5	0.9
1992	0.6	0.5	0.5	1.1
1991	0.8	0.6	0.4	1.1
1990	0.7	0.6	0.5	1.1
1989	0.6	0.6	0.5	1.1
1988	0.3	0.2	0.1	0.3
1987	0.1	0.1	0.0	0.1
1986	0.5	0.4	0.3	0.7
1985	0.4	0.3	0.1	0.4
1984	0.7	0.5	0.4	0.9
1983	0.6	0.3	0.2	0.5
1982	1.2	0.6	0.4	1.0
1981	1.8	0.6	0.5	1.0
1980	0.9	0.5	0.4	0.9

because there are no significant emission sources in close proximity to the park and because the amounts of wet deposition are low. Atmospheric concentrations of S and N species have been measured in YELL since 1996 as part of the National Dry Deposition Network (NDDN). Dry deposition flux calculations are not yet available, but are expected to be available in the near future.

c. Gaseous Monitoring

Ozone is not currently monitored by the state, and at present the only continuous ozone analyzer in Wyoming is operated and maintained by the NPS in YELL. Maximum hourly ozone concentrations at Tower Junction between 1987 and 1994 ranged between 61 and 98 ppbv (Table

V-4). These levels are below the NAAQS for ozone. The mean daytime 7-hour ozone concentration during the growing season ranged between 41 and 45 ppbv during this time period. The SUM60 exposure index is another indicator that can be important in assessing ozone exposures of plant species. This index is the sum of all hourly ozone concentrations equaling or exceeding 60 ppbv. The SUM60 exposure index at YELL ranged between 363 and 11,376 in this time period. For comparison, national parks in highly polluted areas (e.g., southern California) can have SUM60 exposure indexes exceeding 100,000 ppbv-hour (Joseph and Flores 1993). The continuous ozone analyzer at YELL was moved to the Lake area in 1995.

Table V-4. Summary of YELL ozone concentrations (ppbv) from NPS monitoring sites (Joseph and Flores 1993).

	1987[a]	1988	1989	1990	1991	1992	1993	1994[b]
1-hour maximum	88	98	71	61	64	75	62	72
Average daily mean	34	37	33	31	35	36	35	39
Growing season 7-hour mean	41	44	45	34	42	42	41	47
SUM60 exposure index (ppbv-hour)	2,378	11,376	6,658	483	1,169	6,315	363	6,015

[a] data collected from June through December
[b] NPS Monitoring data

Most of the monitoring data at YELL indicate that the park has excellent air quality. However, there are relatively few data from other agencies in the Greater Yellowstone Ecosystem area to support the limited database in the park. Concerns about visibility and the value of scenic vistas to the public were apparent during the large fires of 1988, which points out the social intolerance to degradation of the visual resource.

Sulfur dioxide has been measured in YELL since 1988, and annual average concentrations range from 0.02 to 0.08 ppbv (Table V-5). The highest 24-hour SO_2 concentration measured in the park was 0.73 ppbv. These values are much lower than the concentration that is considered potentially damaging to some vegetation (Treshow and Anderson 1989).

Levels of CO have been measured seasonally since 1992 in connection with increased snowmachine use in the Park. Because winter visitation of YELL is primarily via motorized snow vehicles and has increased nearly ten-fold during the past decade, the potential effects of vehicle emissions during winter have become an important concern for resource managers at YELL (Ingersoll et al. 1997). Air quality monitoring at the west entrance to the park during 1995 detected

Table V-5. Maximum and mean SO_2 24-hour integrated sample. The clean-air reference is estimated to be 0.19 ppbv (Urone 1976). (Source: J. Ray, NPS Air Resources Division)								
	SO_2 concentration (ppbv)							
	1989	1990	1991	1991	1992	1993	1994	1995
Maximum	0.18	0.30	0.73	0.20	0.20	0.03	0.14	0.07
Mean	0.02	0.05	0.08	0.03	0.05	0.04	0.04	0.02

CO levels exceeding Federal standards. At this entrance to the park, as many as 1,000 over-snow vehicles have entered on peak traffic days. Furthermore, analyses of snowpack chemistry within the YELL area during 1993-1995 suggested that the volume of snowmachine traffic might correlate with the concentration of NH_4^+, NO_3^-, or SO_4^{2-} in the snowpack. Therefore, Ingersoll et al. (1997) conducted a survey of snowpack chemistry at six sites within the park to determine if emissions from snowmachine traffic are detectable in the snowpack and also whether pollutant loads diminish rapidly with distance from the snowmachine thoroughfares. Three locations that represent variable levels of over-snow traffic were selected. At each of the three locations (West Yellowstone, Old Faithful, and Sylvan Lake) an off-road site was paired with a site in the snowmachine roadway (about 20 to 100 m apart), and snow core samples were collected and analyzed at each. Snowpack chemistry from the off-road and in-road sites at Sylvan Lake showed similar patterns (Table V-6), probably due to the low level of snowmachine traffic. However, the higher traffic sites showed greater deposition of all three ions. Also, comparisons of results for the in-road and off-road sites at West Yellowstone and Old Faithful showed concentrations of NH_4^+ and SO_4^{2-} (but not NO_3^-) higher in the snowpacked road, compared to the respective off-road site (Table V-6). Ingersoll et al. (1997) concluded that the source of NO_3^- was more likely regional, whereas the concentration of NH_4^+ and SO_4^{2-} seemed to be influenced by local air quality influences such as snowmachines. Although snowmachines may be causing increased local deposition of NH_4^+ and/or SO_4^{2-}, the concentrations in the snowpack are relatively low (< 10 µeq/L) and seem to decrease substantially at short distance from the roadways.

Table V-6. Snowmachine usage levels and chemical concentrations (µeq/L) at snow-sampling sites in YELL (data from Ingersoll et al. 1997).

Site Name	Level of Snowmachine Use	NH_4^+	NO_3^-	SO_4^{2-}
West Yellowstone (off-road)	high	5.1	7.9	4.2
West Yellowstone (in-road)	high	8.9	7.9	8.8
Old Faithful (off-road)	moderate-to-high	5.2	8.4	4.0
Old Faithful (in-road)	moderate-to-high	7.2	8.4	6.2
Sylvan Lake (off-road)	low	3.0	3.9	3.3
Sylvan Lake (in-road)	low	3.5	4.1	4.0

2. Water Quality

EPA's Storage and Retrieval (STORET) database contains about 29,000 observations from the Yellowstone area for 164 water quality parameters at 444 sites. Thirty-five sites within the park have long-term records consisting of multiple observations for several important water quality parameters. Water pH was measured 776 times at 228 sites from 1964 through 1992. Of those, only 44 measurements had pH less than 6.5, at about 20 sites. Total alkalinity by low-level Gran analysis was determined at eight sites, three of which had measured alkalinity less than 200 μeq/L (Grassy Lake Reservoir [0003], Fern Lake [0240], and No Name Lake [0127]). Fern Lake is known to have geothermal sources, based on sampling by the U.S. EPA's Western Lake Survey (WLS; Landers et al. 1987). Sulfate concentrations were measured 786 times at all sites from 1964 through 1992 and exceeded the drinking water criterion of 400 mg/L (8,300 μeq/L) at four sites near Mammoth Hot Springs.

The NPS maintains a database of water quality and flow data for YELL, based on retrievals from five of EPA's national databases, mostly from STORET. pH values are available in this database for one or more sampling occasions from 201 sites, 5 of which had pH < 5.5. However, none of the lakes or streams having pH < 5.5 exhibited other data that reflected sensitivity to acidification from acidic deposition (Table V-7). For example, sulfate concentrations ranged from 18 mg/L to 191 mg/L in the low-pH waters, more than an order of magnitude higher than would be attributable to atmospheric deposition due to air pollution. Chloride concentrations were also high in most of these surface waters, ranging from 1 mg/L to 121 mg/L. Four of the five lakes with pH < 5.5 had chloride concentration >57 mg/L. It is likely that all of these surface waters are impacted by geothermal discharge that cause the water to be low in pH.

The WLS sampled six lakes in YELL and nine lakes in surrounding areas (Landers et al. 1987; Table V-8). One of the lakes in the park was acidic (ANC = -24 μeq/L). This acidity was attributable to geothermal inputs as evidenced by the extremely high concentrations of SO_4^{2-} (818 μeq/L) and

Table V-7. Data on selected variables for the five lakes and streams in the YELL database that had measured pH less than 5.5.

Name	pH	Cl (μeq/L)	SO_4 (μeq/L)	Specific conductance (μS/cm)
YELL 0136	3.4	3,413	1,562	702
YELL 0192	4.3	2,003	3,979	692
Harlequin Lake	5.4	28	375	70
Beaver Lake	5.1	2,482	2,292	71
Nymph Lake	5.0	1,608	3,125	725

Table V-8. Results of lakewater chemistry analyses by the Western Lake Survey for selected variables in YELL and adjacent areas.										
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 YELL										
4D3-013	11.3	75	2006	9.4	1510	17	0.4	1092	1618	5.5
4D3-016	4.6	523	2287	5.7	1332	2909	3.5	599	6682	3.5
4D3-017	38.8	297	2514	4.8	-24	818	0.3	243	1330	6.2
4D3-019	20.8	168	2392	6.6	139	6	0.0	112	220	11.2
4D3-052	15.5	367	2198	8.3	705	30	0.3	311	980	4.8
4D3-073	3.4	119	2677	8.5	416	8	0.0	356	429	1.9
Lakes Outside YELL										
4D2-050	1.5	64	2920	7.0	59	8	0.3	35	75	1.6
4D2-003	4.2	49	2793	7.5	161	7	0.7	66	184	1.3
4D3-001	76.2	7127	2037	8.6	630	33	0.4	436	737	1.2
4D3-002	3.2	80	2915	6.8	57	28	5.0	54	88	0.6
4D3-004	4.9	38	2935	7.1	79	9	0.5	45	101.	1.5
4D3-006	3.0	75	2904	7.7	250	31	0.4	142	278	0.7
4D3-056	3.5	178	2935	6.9	45	11	0.3	32	66	1.8
4D3-028	2.1	31	2482	9.4	3795	109	1.6	1335	4284	16.7
4D3-024	20.7	481	3028	7.5	214	9	0.2	136.	236	0.7

sum of base cations (C_B, 1,330 µeq/L). One lake had an ANC of 139 µeq/L; its pH was also relatively low (6.6), mainly as a consequence of the high concentration of dissolved organic carbon (11 mg/L). Several of the lakes in surrounding areas surveyed by the WLS were more acid-sensitive than those in the park; four of nine had ANC less than 100 µeq/L and one was below 50 µeq/L. Most of these had low SO_4^{2-} concentrations that could reasonably be attributed to atmospheric inputs (~ 8 to 10 µeq/L) and low base cation concentrations.

Water quality data were collected from May through September, 1970 at about 100 sites in and adjacent to the park by US EPA (1972). Measured pH values ranged from 5.4 to 9.2, with most waters in the range of 7.3 to 7.6. Lowest and highest pH values were attributed to the presence of geothermal waters. Measured average alkalinity values were generally between 20 and 40 mg/L (400 to 800 µeq/L), although a value of 4 mg/L (80 µeq/L) was reported for Duck Lake in early summer. The lowest conductivity measurement found by EPA (1972) was 20 µS/cm, also in Duck Lake, which is located near West Thumb. Hardness as mg/L of $CaCO_3$ was 5.8 mg/L. Such values for specific conductance and hardness would suggest only moderate sensitivity to acidic deposition effects. Duck Lake is a closed basin hydrologically. The calcium concentration in Duck Lake, at 75 µeq/L, was the lowest concentration measured by EPA (1972) in the park.

Gibson et al. (1983) reported the chemistry of 106 lakes in YELL. Data from the major lakes were evaluated; lakes were included from every region of the park and from every major geological formation. The study relied on data already available, rather than conducting a survey as was done for ROMO (Gibson et al. 1983) . A spatial sensitivity map, using alkalinity as an index of vulnerability to acidification, was created for the park based on recent measurements of alkalinity (Figure V-4). Excluding lakes located in the midst of major geothermal areas, the lowest alkalinity measured was 40 μeq/L. Most of the lakes with alkalinity less than 200 μeq/L occurred within the large rhyolite flow which rose from the southwest and spread along the westcentral and central portions of the park (Figure V-1). Six of the 106 lakes evaluated by Gibson et al. (1983) had pH less

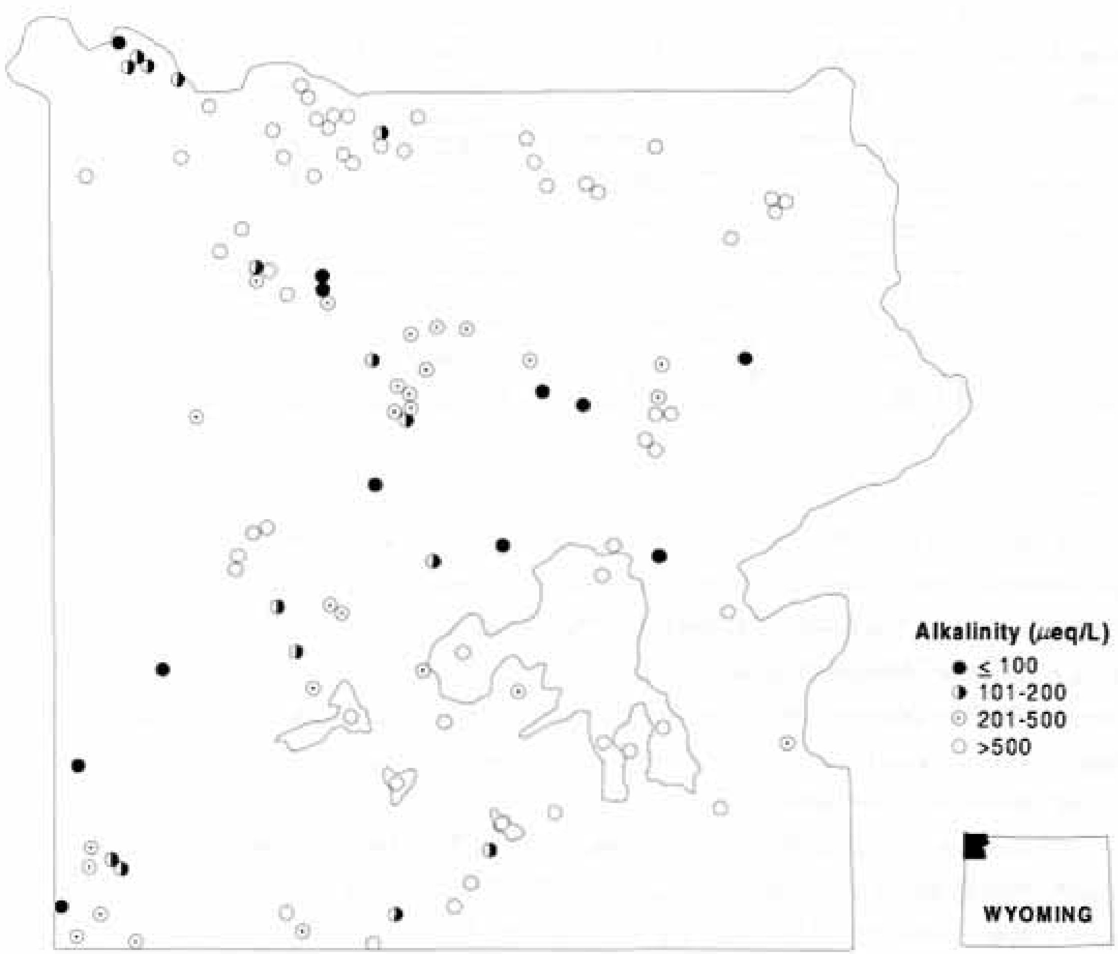


Figure V-4. Alkalinity map for lakes in YELL, prepared from existing data by Gibson et al. (1983).

than 6.5. The authors concluded that two of the six were unquestionably influenced by geothermal contributions and one probably so; the other three were dystrophic. The greatest proportion of high pH and high alkalinity lakes were found in the northern portions of the park. Most of the lakes with alkalinity less than 200 μeq/L were considered naturally barren of fish (Table V-9).

Table V-9. Total alkalinity and fish population status for lakes having ANC < 200 μeq/L in YELL. (Source: Gibson et al. 1983, Varley 1981)

Lake Name	Alkalinity (μeq/L)	Fish Status	
		Historical	Current
Wrangler	40	Fishless	Fishless
Summit	60	Fishless	Fishless
Shelf	80	Fishless	Fishless
Mt. Everts	160	Fishless	Fishless
Ice	200	Fishless	Fishless
Ranger	160	Fishless	(q)
Obsidian	80	Fishless	Brook trout
High	170	Fishless	Cutthroat
Forest	192	Fishless	Cutthroat (q)
Trilobite	200	Fishless	Brook trout
Robinson	100	Cutthroat	Brook trout (q)

(q) Status questionable

Engstrom et al. (1991) cored the sediments of eight small lakes (2-12 ha) in the northern portion of the park to examine the stratigraphic records of past changes in limnology attributable to ungulate grazing. The lakes are located at 1,700 to 2,100 m elevation in the northern winter range area of the park. Although detailed water chemistry was not determined, all eight lakes were reported to have specific conductance greater than 200 μS/cm. Such lakes therefore would have high concentrations of major ions, and would not be expected to be sensitive to potential acidification from acidic deposition.

The stream benthos and periphyton sampled in 1970 (EPA 1972) reflected water of excellent quality throughout the park, with the exception of Soda Butte Creek, which is impacted by leaching from upstream mine tailings outside the park. The flora and fauna of park streams remained relatively unchanged since the first intensive surveys in 1920 and reflected benthic communities typical of unpolluted streams (US EPA 1972).

3. Visibility

As part of the IMPROVE network, visual air quality in YELL has been monitored using an aerosol sampler, transmissometer, and camera. The aerosol sampler has operated from March 1988 through the present and is located at the Lake Village Ranger Station on the northwest shoreline of Yellowstone Lake. The transmissometer operated from July 1989 through July 1993, when monitoring had to be discontinued due to IMPROVE network funding limitations. The transmissometer was located at the southern edge of Mammoth Hot Springs, approximately 6 miles south of the Wyoming-Montana border. The 35mm camera operated from September 1986 through March 1995 and was located on the northern shore of Lake Yellowstone approximately one quarter mile east of the Lake Village Ranger Station. Data from this IMPROVE site have been summarized to characterize the full range of visibility conditions for the March 1988 through February 1995 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 1994 includes March 1994 through February 1995). Complete descriptions of visibility characterization, mechanisms of sources and visibility impacts, and IMPROVE monitoring techniques and rationale are provided in the Introduction of this report.

a. <u>Aerosol Sampler Data - Particle Monitoring</u>

IMPROVE aerosol samplers consist of four separate particle sampling modules that collect 24-hour filter samples of the particles suspended in the air. The filters are then analyzed in the laboratory to determine the mass concentration and chemical composition of the sampled particles. Particle data can be used to provide a basis for inferring the probable sources of visibility impairment. Practical considerations limit the data collection to two 24-hour samples per week. (Wednesday and Saturday from midnight to midnight). Detailed descriptions of the aerosol sampler, laboratory analysis, and data reduction procedures used can be found in the draft 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. The extinction coefficient represents the ability of the atmosphere to scatter and absorb light. Higher extinction coefficients signify lower visibility. A tabular and graphic summary of average reconstructed extinction values by season and year for the March 1988 through February 1995 period are provided in Table V-10 and Figure V-5, respectively.

Reconstructed extinction budgets generated from aerosol sampler data apportion the extinction at YELL to specific aerosol species (Figure V-6). The species shown are Rayleigh, sulfate, nitrate,

Table V-10. Seasonal and annual average reconstructed extinction (Mm^{-1}) and standard visual range (km), YELL, March 1988 through February 1995.

YEAR	Spring (Mar, Apr, May)		Summer (Jun, Jul, Aug)		Autumn (Sep, Oct, Nov)		Winter (Dec, Jan, Feb)		Annual (Mar - Feb)[a]	
	b_{ext} (Mm^{-1})	SVR (km)	b_{ext} (Mm^{-1})	SVR (km)	b_{ext} (Mm^{-1})	SVR (km)	b_{ext} (Mm^{-1})	SVR (km)	b_{ext} (Mm^{-1})	SVR (km)
1988	30.2	130	53.7	73	30.9	127	24.1	162	34.1	115
1989	30.7	127	42.1	93	33.7	116	24.7	158	33.0	119
1990	32.8	119	41.6	94	31.4	125	26.6	147	33.7	116
1991	29.6	132	39.5	99	31.0	126	26.9	145	32.1	122
1992	31.6	124	40.3	97	32.9	119	24.3	161	32.1	122
1993	27.4	143	29.8	131	33.7	116	21.9	179	28.2	139
1994	30.5	128	42.0	93	30.7	127	21.0	186	31.0	126
Mean[b]	30.4	129	41.3	95	32.0	122	24.2	162	32.0[c]	122[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 1995 period.
[c] Combined annual period data represent the mean of all combined season means.

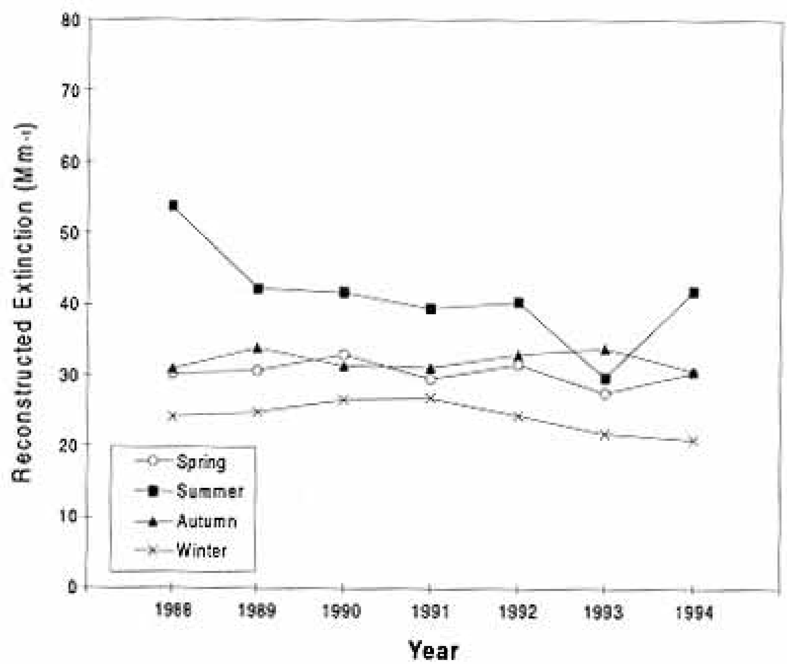


Figure V-5. Seasonal average reconstructed extinction (Mm^{-1}) YELL, March 1988 through February 1995.

Figure V-5. Seasonal average reconstructed extinction (Mm^{-1}) YELL, March 1988 through February 1995.

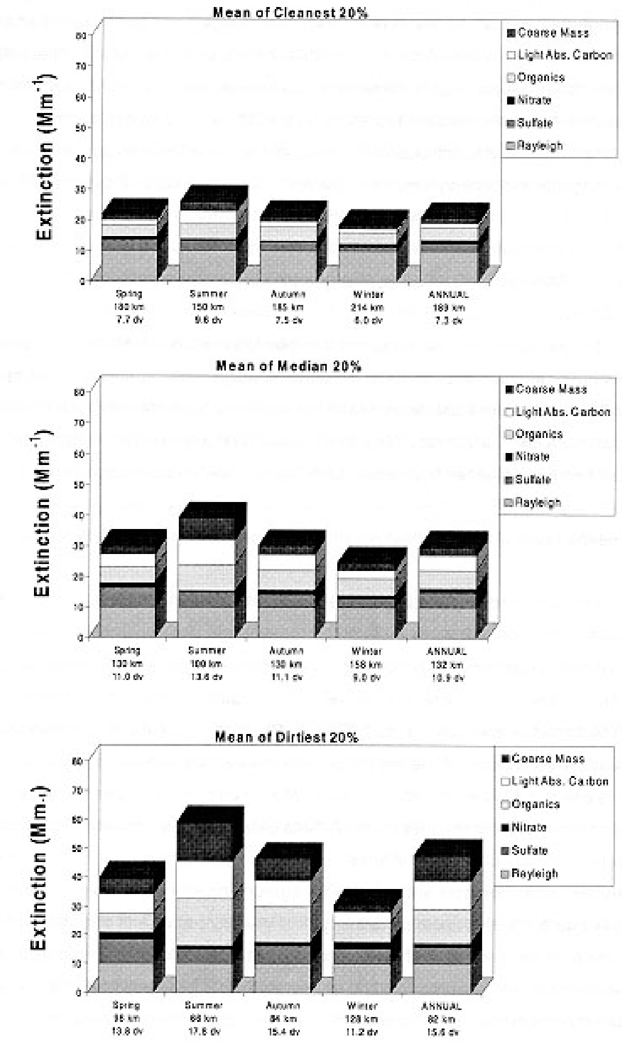


Figure V-6. Reconstructed extinction budgets for YELL, March 1988 - February 1995.

Figure V-6. Reconstructed extinction budgets for YELL, March 1988 - February 1995.

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% of days, mean of the median 20% of days, and mean of the dirtiest 20% of days. 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, visual range (in kilometers), and deciview (dv). Standard Visual Range (SVR) can be expressed as:

$$SVR\ (km) = 3{,}912\ /\ (b_{ext} - b_{Ray} + 10)$$

where b_{ext} is the extinction coefficient expressed in inverse megameters (Mm^{-1}), b_{Ray} is the site specific Rayleigh values (elevation dependent), 10 is the Rayleigh coefficient used to normalize visual range, and 3,912 is the constant derived from assuming a 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.

Deciview is defined as:

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

where b_{ext} is the extinction coefficient expressed in inverse megameters (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 Rayleigh atmosphere and increases as visibility is degraded. The segment at the bottom of each stacked bar 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.

The 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. 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.

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. 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, 1993 and 1994).

Table V-11 provides a tabular summary of the "filtered" seasonal and combined period arithmetic mean extinction values for the July 1989 through July 1993 period. Table V-12 provides a tabular summary of the "filtered" seasonal and combined period 10% (clean) cumulative frequency values. 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 or missing seasons.

- Combined season data represent the mean of all valid seasonal b_{ext} values for each season (spring, summer, autumn, winter) of the July 1989 through July 1993 period.

- Combined annual period data represent the unweighted mean of all combined seasonal b_{ext} values.

Extinction values in Tables V-11 and V-12 are also presented in units of standard visual range (in kilometers) and deciview (dv).

Figure V-7 provides a graphic representation of the "filtered" annual mean, median, and cumulative frequency values (5th, 10th, 25th, 75th, 90th, and 95th percentiles). No data are reported for annual periods with one or more invalid or missing seasons.

Table V-11. Seasonal and annual arithmetic means for YELL, transmissometer data (filtered) July 1989 thr July 1993.

Year	Spring (Mar, Apr, May)			Summer (Jun, Jul, Aug)			Autumn (Sep, Oct, Nov)			Winter (Dec, Jan, Feb)			Annual (March - Febr	
	SVR	b_{ext} (Mm^{-1})	dv	SVR	b_{ext} (Mm^{-1})	dv	SVR	b_{ext} (Mm^{-1})	dv	SVR	b_{ext} (Mm^{-1})	dv	SVR	b_{ext} (Mm^{-1})
1989				--	--	--	108	35	12.5	154	25	9.2	***	***
1990	138	28	10.3	121	32	11.6	115	33	11.9	133	29	10.6	131	31
1991	117	33	11.9	133	29	10.6	122	31	11.3	154	25	9.2	136	30
1992	133	29	10.6	114	34	12.2	122	31	11.3	161	24	8.8	136	30
1993	125	31	11.3	175	22	7.9							***	***
Mean[b]	128	30	11.1	129	29	10.7	116	33	11.8	150	26	9.5	136	29[c]

-- 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} values for each March through February annual period.
[b] Combined season data represent the mean of all valid seasonal b_{ext} values for each season of th 1989 through July 1993 period.
[c] Combined annual period data represent the mean of all combined seasonal b_{ext} values.

Table V-12. Seasonal and annual 10% (Clean) cumulative frequency statistics for YELL, transmissometer data (filtered) July 1989 through July 1993.

Year	Spring (Mar, Apr, May)			Summer (Jun, Jul, Aug)			Autumn (Sep, Oct, Nov)			Winter (Dec, Jan, Feb)			Annual (March - February)[a]		
	SVR	b_{ext} (Mm^{-1})	dv	SVR	b_{ext} (Mm^{-1})	dv	SVR	b_{ext} (Mm^{-1})	dv	SVR	b_{ext} (Mm^{-1})	dv	SVR	b_{ext} (Mm^{-1})	dv
1989				--	--	--	169	22	7.9	255	15	4.1	***	***	***
1990	202	19	6.4	183	21	7.4	150	25	9.2	192	20	6.9	190	21	7.5
1991	154	25	9.2	168	23	8.3	162	23	8.3	202	19	6.4	179	23	8.1
1992	183	21	7.4	168	23	8.3	156	24	8.8	239	16	4.7	192	21	7.4
1993	168	23	8.3	192	20	6.9							***	***	***
Mean[b]	175	22	7.9	171	22	7.8	159	24	8.5	219	18	5.6	190	21[3]	7.5

-- 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} values for each March through February annual period.
[b] Combined season data represent the mean of all valid seasonal b_{ext} values for each season of the July 1989 through July 1993 period.
[c] Combined annual period data represent the mean of all combined seasonal b_{ext} values.

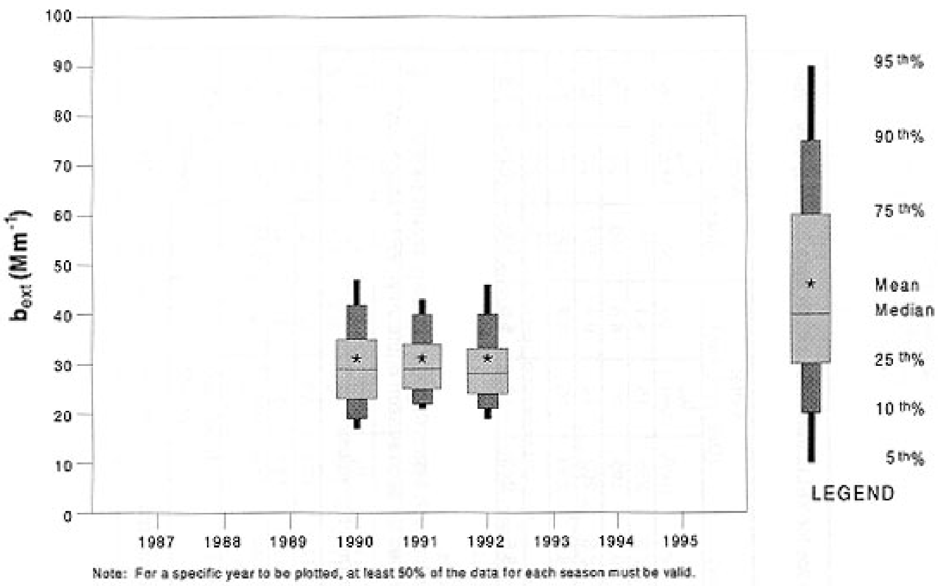


Figure V-7. Annual arithmetic mean and cumulative frequency statistics for YELL, transmissometer data (filtered).

Figure V-7. Annual arithmetic mean and cumulative frequency statistics for YELL, transmissometer data (filtered).

When comparing reconstructed (aerosol) extinction, Table V-10 with measured (transmissometer) extinction, Table V-11 the following differences/similarities should be considered:

<u>Data Collection</u> - 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.

<u>Point versus Path Measurements</u> - 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.

Note: Differences in daily averages may also be attributed to the 16 mile distance between the Yellowstone aerosol sampler and transmissometer. Optical and aerosol effects often vary with local emissions or meteorological conditions.

<u>Relative Humidity (RH) Cutoff</u> - Daily average reconstructed measurements are flagged as invalid when the daily average RH is greater than 98%. Hourly average transmissometer measurements are flagged invalid when the hourly average RH is greater than 90%. These flagging differences often result in data sets that do not reflect the same period of time, or misinterpret short-term meteorological conditions.

> Note: The weather algorithm only flags 10%-20% of the data for a majority of the sites west of the Mississippi River. RH cutoffs have little effect on final mean extinctions in the western United States.

Reconstructed extinction is typically 70%-80% of the measured extinction. With a ratio of 91%, this relationship shows fair agreement for YELL.

c. Camera Data - View Monitoring

An automatic 35mm camera system operated at YELL from September 1986 through March 1995. Color 35mm slide photographs of Overlook Mountain and Lake Yellowstone (to the south) were taken three times per day until the Autumn 1989. The camera alignment changed at that time to photograph Avalanche Peak and Lake Yellowstone (to the southeast) until March 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 Avalanche Peak photographs presented in Figure V-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.

d. Visibility Summary

Data from other IMPROVE visibility sites around the country have been presented graphically (Figures I-1 and I-2) so that visual air quality in the Rocky Mountains and northern Great Plains can be understood in perspective. Figure V-5 and Table V-11 have been provided to summarize YELL visual air quality during the March 1988 through February 1995, and July 1989 through July 1993 periods respectively. Long-term trends fall into three categories: increases, decreases, and variable. Given the visibility sites summarized for this report, the majority of data show little change or trends.

Non-Rayleigh atmospheric light extinction at YELL, like many rural western areas is largely due to sulfate, organics, and soil. 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 pre-settlement visibility below current levels during some summer months.

The IMPROVE aerosol monitoring network, established in March 1988, consists of sites instrumented with aerosol sampling modules A through D. Many of the IMPROVE sites are successors to sites where aerosol monitoring with stacked filter units (SFU) was carried out as early

Yellowstone National Park
on a "clear" day

Representative Conditions:

Visual Range:	260-300 km
b_{ext}:	15-13 Mm^{-1}
Haziness:	4-3 dv

Yellowstone National Park
on an "average" day

Representative Conditions:

Visual Range:	130-170 km
b_{ext}:	30-23 Mm^{-1}
Haziness:	11-8 dv

Yellowstone National Park
on a "dirty" day

Representative Conditions:

Visual Range:	75-95 km
b_{ext}:	52-41 Mm^{-1}
Haziness:	17-14 dv

Figure V-8. Photographs illustrating visibility conditions at Yellowstone National Park.

Figure V-8. Photographs illustrating visibility conditions at Yellowstone National Park.

as 1979. SFU data were collected at YELL from October 1979 through December 1987. No long-term trends are apparent in the 1979 through 1987 SFU data or the 1988 through 1994 site-specific or regional data presented in this report.

D. AIR QUALITY RELATED VALUES

1. Aquatic Biota

Sensitive fish species in the park include both native and non-native salmonids. Recent studies have shown that native western trout are sensitive to short-term increases in acidity. For example, Woodward et al. (1989) exposed native western cutthroat trout (*Oncorhynchus clarki*) to pH depressions (pH 4.5 to 6.5) in the laboratory. Freshly-fertilized egg, eyed embryo, alevin, and swim-up larval stages of development were exposed to low pH for a period of seven days. Fish life stages were monitored for mortality, growth, and development to 40 days posthatch. The test fish were taken from the Snake River in Wyoming. Reductions in pH from 6.5 to 6.0 in low-calcium water (70 μeq/L) did not affect survival, but did reduce growth of swim-up larvae. Eggs, alevins, and swim-up larvae showed significantly higher mortality at pH 4.5 as compared to pH 6.5. Mortality was also somewhat higher at pH 5.0, but only statistically higher for eggs. Survival of various life stages of Yellowstone cutthroat trout exposed to 7-day pH depressions in order to simulate episodic acidification was studied by Farag et al. (1993). They also evaluated the added toxicity associated with elevated concentrations of Al. However, we note procedural problems associated with their Al exposures (i.e., Al added in excess of the amount soluble at a given pH) and therefore only consider data for evaluating fish response to pH depressions. Of the four life stages studied in Yellowstone cutthroat trout, eggs were most sensitive to low pH. Eggs exposed to pH 5.0 experienced a statistically-significant reduction in survival when compared with eggs exposed for seven days to pH 6.5 water. Survival of alevin and swim-up larvae was reduced from near 100% at pH 6.5 to near zero percent at pH 4.5. Intermediate pH values (6.0, 5.5) in all cases showed reduced survival compared with the control (6.5) but not by a statistically significant amount. Eyed embryos were not sensitive to any of the exposures.

2. Terrestrial

Vegetation is the resource which is most sensitive to ozone and SO_2, and several tree species have been identified as potential bioindicators (see below). Additional studies would be needed to evaluate the impact of SO_2 and ozone on terrestrial ecosystems in YELL. While ozone and SO_2 levels have not exceeded the NAAQS in YELL, future levels could potentially damage sensitive plant species. Furthermore, baseline data on the condition of sensitive species in the absence of injurious pollutants would be helpful for comparison if pollutant levels increase. Monitoring sensitive receptors (those species with known sensitivity to one or more pollutants) by using detailed descriptions and classifications of sensitive indicators (characteristics of leaf or plant injury) will be necessary for long-term evaluation of ecosystem health.

One of the most ozone-sensitive Western tree species is ponderosa pine (*Pinus ponderosa*, especially var. *ponderosa*), for which extensive data are available on field (Miller and Millecan 1971, Pronos and Vogler 1981, Peterson and Arbaugh 1988) and experimental (Temple et al. 1992) exposures. The evidence for ozone impacts on ponderosa pine is based on observable symptoms and reduced growth (Peterson et al. 1991, Peterson and Arbaugh 1992) as well as physiological (Darrall 1989, Bytnerowicz and Grulke 1992) data. The cause-and-effect relationship, especially for trees growing in forests of southern California and the southern Sierra Nevada, is clear and quantifiable.

Lodgepole pine is more tolerant to ozone and does not exhibit symptoms of injury under low experimental exposures that would result in symptoms to ponderosa pine (Aitken et al. 1984). Nevertheless, the well-documented and quantifiable symptomatology of pines makes lodgepole pine an acceptable bioindicator for ozone, even if it has only moderate sensitivity. Lodgepole pine is very widespread in YELL, and several areas in the park are suitable for establishing long-term monitoring plots.

Of the hardwood species present in YELL, quaking aspen is the most sensitive to ozone and may be the most ozone-sensitive tree species in the park. Aspen grows at various locations in riparian ecosystems and in fire- or avalanche-disturbed areas in the park. Numerous studies have documented the sensitivity of this species to ozone under field and experimental 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 aspen includes chlorosis, stippling, necrotic spotting, and leaf margin burn. Symptoms generally vary seasonally, with stippling being most prominent in the spring and black, bifacial (both leaf surfaces) necrosis appearing in late summer (J.P. Bennett, pers. comm.). Great care must be taken in distinguishing ozone symptoms from the effects of various pathogens and insect herbivores commonly found on this species.

Aspen is also considered to be sensitive to SO_2 and may be the best sensitive receptor for this gaseous pollutant. Injury is similar to that normally found for ozone (stippling, followed by bifacial necrosis), although SO_2 -induced injury rapidly bleaches to a light tan color (ozone injury remains dark) (Karnosky 1976). There could be some confusion in differentiating ozone injury from SO_2 injury.

Black cottonwood is another potential sensitive receptor for ozone (Woo 1996), and it has symptoms similar to those of aspen. However, it is generally regarded as less sensitive to ozone than aspen. Neither of these hardwood species has the clarity of ozone symptomatology found in lodgepole pine.

A species list of native plants is available in the NPFlora database. Table V-13 summarizes vascular plant species of YELL with known sensitivity to ozone, SO_2 and NO_x. This table is based on a variety of sources from the published literature and other information. It should be noted that the various sources used a wide range of field and experimental approaches to determine pollutant pathology, and that sensitivity ratings are general estimates based on published information and our expert opinion. While it will not be possible for park staff to collect data on all the species indicated in Table V-13, the list can be used by park managers as a guide for identifying visible symptoms. Of the many plant species in YELL, it is likely that there are many other species which have high sensitivity to air pollution, but we currently have no information about them.

Table V-13. Plant species of YELL with known sensitivities to sulfur dioxide, ozone and nitrogen oxides. (H=high, M=medium, L=low, blank=unknown). (Sources: Esserlieu and Olson 1986, Bunin 1990, Peterson et al. 1993, Electric Power Research Institute 1995, Binkley et al. 1996, Brace and Peterson 1996)

Species Name	SO_2 Sensitivity	O_3 Sensitivity	NO_x Sensitivity
Abies lasiocarpa	L	L	
Acer glabrum	H		
Alnus tenufolia	H		
Betula occidentalis	M		
Bromus carinatus		L	
Bromus tectorum		M	
Ceanothus velutinus	L		
Chenopodium fremontii		L	
Cirsium arvense		L	
Cirsium undulatum	M		
Clematis ligusticifolia	M		
Collomia linearis		L	
Convolvulus arvensis	H		
Cornus stolonifera	M	L	
Descurainia pinnata		L	
Epilobium angustifolium		L	
Erigeron peregrinus		L	
Fragaria virginiana		H	
Galium bifolium		L	
Gaultheria shallon		L	
Gayophytum racemosum		L	
Gentiana amarella		M	
Geranium richardsonii	M	M	
Hackelia floribunda	L		
Hedysarum boreale		M	
Helianthus anuus	H	L	

Table V-13. Continued.			
Species Name	SO_2 Sensitivity	O_3 Sensitivity	NO_x Sensitivity
Juniperus communis	L		
Juniperus scopulorum	L		
Lemna minor	L		
Lolium perenne		M	
Lonicera involucrata	L	H	
Medicago sativa		M	
Mentzelia albicaulis		H	
Mimulus guttatus		L	
Oryzopsis hymenoides	M		
Phyllodoce empetriformis		L	
Picea engelmannii	M	L	
Picea glauca	M	L	
Pinus contorta	M	M	H
Pinus flexilis	L		
Poa annua	H	L	
Poa pratensis		L	
Polygonum douglasii		L	
Populus angustifolia	M		
Populus balsamifera subsp. *trichocarpa*	M	H	
Populus tremuloides	H	H	
Potentilla fruticosa		L	
Prunus virginiana	M	H	
Pseudotsuga menziesii	M	L	H
Rhus trilobata	L	H	
Ribes viscosissimum	M		
Rosa woodsii	M	L	
Rubus idaeus	H		
Rubus parviflorus		M	
Rumex crispus		L	
Salix scouleriana		M	
Senecio serra		H	
Shepherdia canadensis	L		
Sorbus scopulina	M		
Symphoricarpos oreophilus	M		
Taraxacum officinale		L	
Toxicodendron radicans	L	L	
Tragopogon dubius	M		
Trifolium pratense	L		
Trifolium repens		H	
Trisetum spicatum	M		

Vicia americana		L	
Viola adunca		L	

An inventory of lichen species with known sensitivity to ozone and SO_2 is summarized in Table V-14. As in Table V-13, this table is based on a variety of sources from the published literature and other information. It should be noted that diagnostic symptoms of air pollutant injury to lichens are difficult to identify, and that some species have reduced productivity or even mortality without exhibiting visible symptoms (Nash and Wirth 1988). One of the best sources of background information and guidelines for addressing the use of lichens as sensitive receptors of air pollution is Stolte et al. (1993). The inventory of lichen species performed by Eversman (1990) provides a good baseline for further assessment of potential impacts of air pollution.

Table V-14. Lichen species of YELL with known sensitivity to SO_2 and ozone. (H=high, M=medium, L=low, blank=unknown). (Sources: Peterson et al. 1993, National Park Service 1994, Electric Power Research Institute 1995, Binkley et al. 1996)

Species	SO_2 sensitivity	Ozone sensitivity
Acarospora chlorophana	H	
Aspicilia caesiocinerea	L	
Bryoria abbreviata		H
Bryoria fremontii		H
Bryoria fuscescens	M	
Candelaria concolor	M-H	
Candelariella vitellina	M	
Candelariella xanthostigma	M	
Cladonia chlorophaea	M	
Cladonia coniocraea	M	
Cladonia fimbriata	M-H	
Coeocaulon muricatum	L-M	
Collema nigrescens		M
Evernia mesomorpha	M	
Hypogymnia imshaugii	L-M?	L-M
Hypogymnia physodes	M	
Lecanora saligna	M	
Lecidea atrobrunnea	L	
Lepraria incana	L-M	
Letharia columbiana	L	L-M
Letharia vulpina	L	L-M

Melanelia exasperatula	M	
Melanelia subolivacea		L-M
Parmelia saxatilis	M	L
Parmeliopsis hyperopta	M	
Peltigera aphthosa	M	H

Table V-14. Continued.

Species	SO_2 sensitivity	Ozone sensitivity
Peltigera canina	L	H
Peltigera collina		H
Peltigera polydactyla	M	
Peltigera refuscens		M-H
Phaeophyscia orbicularis	M	H
Phaeophyscia sciastra		H
Physcia adscendens	M	
Physcia aipolia	M	
Physcia caesia	M	
Physcia dubia	M	
Physcia stellaris	M	
Physconia detersa	M-H	L
Pseudephebe pubescens		M
Rhizoplaca chrysoleuca	H	
Rhizoplaca melanophthalma	H	
Usnea hirta	M-H	
Xanthoparmelia cumberlandia	H	
Xanthoria candelaria	M-H	H
Xanthoria elegans	M	
Xanthoria fallax	M-H	L
Xanthoria polycarpa	M-H	L

E. RESEARCH AND MONITORING NEEDS

1. Deposition and Gases

Deposition of sulfur and nitrogen is monitored at Tower Junction. This site provides sufficient data for YELL, given that the aquatic resources within the park are generally insensitive to adverse effects of acidic deposition and that current deposition levels at Tower Junction are very low.

Ozone pollution in YELL is currently at a level below that which would be expected to adversely affect sensitive plant species. The continuous ozone analyzer at Lake should continue to be operated in order to document any future changes. In addition, it would be useful to establish a network of passive ozone samplers to compare ozone measurements from different locations in the park. Five samplers should be sufficient to spatially characterize the ozone distribution: (1)

colocated with the continuous ozone analyzer at Lake, (2) west of Mammoth Hot Springs, (3) near the northeast entrance, (4) near the Bechler Ranger Station, and (5) southeast portion of the park on the Yellowstone River. It would also be desirable to determine differences in ozone exposure across an elevation gradient within the park. Samplers should be situated where they are reasonably accessible but not within 50 m of a road or trail where they may be subject to excessive dust or vandalism. Weekly samples collected for two months during each summer for a period of three years should be sufficient to establish spatial patterns and a reference point in time.

Because of the current concern regarding CO and its potential impacts on human health, we recommend that YELL continue to monitor CO emissions in the park. Emphasis should be placed on high-traffic areas within the park, especially areas where park personnel and others are exposed to the gas for long periods of time. Monitoring will be particularly important during peak summer (passenger vehicle) and winter (snowmachine) traffic. We also recommend that the park consult with a human respiratory (or other appropriate) specialist to determine if current levels of CO exposure are sufficient to cause physiological impairment in park personnel and visitors.

2. Aquatic Systems

We do not recommend any additional monitoring or research on the sensitivity of aquatic ecosystems within YELL to air pollution degradation. There are several reasons for this. First, lakes and streams in the park are, for the most part, not sensitive to acidification impacts. We base this on examination of available water chemistry data from the park (e.g., Tables V-7, V-8, and V-9), including data from the Western Lakes Survey, STORET, and the National Park Service database for YELL. Base cation concentrations and ANC tend to be higher than commonly-accepted thresholds of sensitivity. Second, many of the surface waters in the park receive substantial contributions of mineral acid anions from geothermal sources; this would obfuscate future efforts to assess cause/effect relationships associated with air pollution impacts. Third, YELL is exposed to the same general airshed as GRTE which contains aquatic resources of greater sensitivity. Monitoring for air pollution effects on aquatic resources would be better conducted in GRTE.

3. Terrestrial Systems

Monitoring of terrestrial resources should be considered in order to establish a baseline of current conditions with respect to the potential impacts of ozone and SO_2. Three levels of monitoring associated with increasing amounts of effort and expense are detailed in Appendix A. These monitoring activities are based on methods and protocols developed by the USDA Forest Service and NPS. Species and locations recommended for monitoring are listed below.

If monitoring is implemented, we recommend placing plots at three locations in the vicinity of the Tower Lake monitoring station. These plots should be located away from human-use areas and preferably in areas with good air flow. Species recommended for monitoring are lodgepole pine and quaking aspen. If necessary, additional plots could be located near the south entrance. These areas are all located along river drainages where maximum air flow can be expected. Locations of these plots can be changed to other sites in YELL if ambient air quality data indicate that other areas have a higher risk of pollutant effects.

Additional tree plots that evaluate other conifer species, such as Engelmann spruce, and Douglas-fir could be located along elevational transects. These additional plots should be located adjacent to the location of plots discussed above. If herbaceous plants are included in the monitoring effort, candidate species that are known to exist in YELL include strawberry (*Fragaria virginiana),* skunkbush (*Rhus trilobata*), and red clover (*Trifolium repens*). A large-scale monitoring effort for lichens is not justified at this point, although protocols and guidelines in Stolte et al. (1993) can be consulted for information on assessing injury.

4. Visibility

IMPROVE aerosol monitoring continues today at YELL. Continued monitoring is necessary to identify potential future impacts. Additional data and in-depth modeling and analysis are required to further evaluate historical trends and future projections of impact from existing and future sources. For example, back trajectory analysis and spatial/temporal pattern analysis of episodes are recommended to determine the source region contributions to elevated aerosol concentrations. Future research is also recommended to minimize the uncertainty in estimates of how various aerosol species affect visibility.

VI. GLACIER NATIONAL PARK

A. GENERAL DESCRIPTION

Glacier National Park (GLAC) was created in 1910 and currently encompasses 410,000 ha in northwestern Montana. The unique resources and diversity of conifer forests, alpine tundra, and wildlife were formally recognized in 1976 when GLAC was designated a Biosphere Reserve under the Man and the Biosphere Program. GLAC includes rugged topography, active glaciers, clear streams and lakes, and spectacular scenic vistas.

GLAC is surrounded by lands with various levels of protection. The park shares a 66-km boundary with British Columbia and Waterton Lakes National Park in Alberta. In 1932, GLAC and Waterton were named the world's first international peace park. In addition, GLAC is adjacent to a large complex of wilderness areas (Great Bear, Bob Marshall, and Scapegoat) located on national forest lands. The Blackfeet Indian Reservation borders the park to the east and is primarily rangeland. The protection afforded to these surrounding lands minimizes outside disturbance to the natural resources of the park.

Principal management objectives for GLAC are to: (1) conserve and protect the integrity of the park's naturally functioning ecosystems, recognizing humans as part of these systems, and (2) conduct and encourage scientific research that contributes to the understanding and management of ecological and cultural systems. Annual visitation at the park is approximately 2 million, most of which occurs during summer.

1. Soils and Geology

GLAC has several unique geological features. The layers of the Precambrian Belt Supergroup are extremely well-delineated in the portion of the park above treeline, and the layered sedimentary structures are unusually well-preserved on the dry eastern slopes. The Lewis Overthrust Fault is also exceptionally visible in the park. Fifty small alpine glaciers of relatively recent post-Pleistocene age are found in the higher elevations of the park. Current and former glacial activity in GLAC has resulted in many hanging valleys, cirques, and arretes, as well as an extensive hydrological system of lakes and streams.

The landscape of GLAC was created, in part, by an overthrust fault of ancient sedimentary substrates. Glaciers and streams have eroded the sedimentary strata in a dendritic pattern, radiating outwards from the central axis of high ridges and mountain peaks. Glacial moraines are prominent, especially in the northwestern part of the park (Martinka 1992).

The park contains three distinct physiographic areas: the valleys of the North and Middle Forks of the Flathead River in the west, the central high mountains, and the plains in the eastern portion of the park. The central portion of the park is dominated by two mountain ranges that run northwest to southeast and contain many small glaciers and snowfields. The Livingston Range, in the west, extends from the Canadian border south to the Lake McDonald area. The Lewis Range in the east extends from the border south to Marias Pass. Extensive portions of both ranges lie above timberline (~2,000 m) and many of the peaks extend above 2,800 m. Elevation ranges from about 950 m along the western boundary to 3,190 m in the central mountains, and back down to about 1,370 m along the eastern boundary.

Limited soils data are available for the park. Dutton (1990) mapped soils in the Red Bench Fire area in the northwestern portion of the park. Most soils in the area (~70%) were classified as glacial till and drift soils. These tend to be deep sandy loams with 40 to 60% rocks, formed from glacial till deposits. Other soil types include alluvial soils, high in sand and rock fragments (17% of area) and wet soils with high organic content (10% of area). Soils were also mapped in the Lake McDonald drainage (Land and Water Consulting, Inc. 1995). At the higher elevations, soils were sparse and found in pockets of variable depth and rock contents. At the lower elevations, glacial ice mixed and deposited materials in a complex pattern. Volcanic eruptions of Mt. Mazama provided as much as 15 cm of volcanic ash to most local soils. As a result of ash influence, most McDonald drainage soils have loamy textures. Common soil mapping units included glacial till soils, quartzite and argillite bedrock colluvial soils, and limestone bedrock soils.

2. Climate

The western slopes in the park are influenced by maritime air masses that provide a moderate amount of precipitation. To the east, continental air masses modify the maritime influence, and create more variable conditions, including colder and drier winter months. Annual precipitation ranges from about 59 cm on the periphery to 250 cm or more in the central highlands (Martinka 1992).

January is the coldest month with mean temperature ranging from -7^{o} to -13^{o}C. July is the warmest month, with mean temperature of 8^{o} to 17^{o}C. Winds are generally from the west or southwest. Windrose data from East Glacier show a strong southwesterly component to the prevailing winds, which are often in excess of 10 mph.

3. Biota

Five major floristic provinces coincide in GLAC, including vegetative components of the dominant northern Rocky Mountain flora, Great Plains, Pacific slope, boreal region, and arctic and alpine provinces. GLAC contains about 1,000 species of vascular plants. The diversity of vegetation and plant communities is the result of the diversity of environmental conditions in the park, including wide variation in elevation, topography, precipitation, temperature, geological substrates, and soils. Fire is the dominant ecological disturbance throughout the park.

A variety of tree species dominate the forested ecosystems of GLAC. Mixed forests contain lodgepole pine (*Pinus contorta*), Engelmann spruce (Picea *engelmannii*), western larch (*Larix occidentalis*), western redcedar (*Thuja plicata*), western hemlock (*Tsuga heterophylla*), Douglas-fir (*Pseudotsuga menziesii*), and subalpine fir (*Abies lasiocarpa*) (Rockwell 1995). In some cases, these species are all found in close proximity to one another with species distribution differentiated by small differences in soil moisture and topography. Higher elevation subalpine forests are dominated by subalpine fir and Englemann spruce, limber pine (*Pinus flexilis*), and whitebark pine (*P. abicaulis*). At treeline, where tree growth is limited by cold temperatures and wind, these species often have a shrublike krummholz form. Various types of meadow communities can be found in the subalpine forest zone; these communities can generally be differentiated by differences in soil moisture and depth. Alpine areas above treeline contain a variety of shrubs, grasses, sedges, and cushion plants. On the drier east side of the park, relatively treeless prairies can be found, mixed in some cases with sagebrush (*Artemisia* spp.), aspen (*Populus tremuloides*) parklands, and ponderosa pine (*Pinus ponderosa)*. Riparian areas throughout the park are typically dominated by black cottonwood (*Populus balsamifera* subsp*. trichocarpa*) and white spruce (*Picea glauca*).

GLAC contains 60 mammal species and over 260 bird species. A variety of ungulates, predators, and other large mammals are found in the park, including elk (*Cervus elaphus*), mule deer (*Odocoileus hemionus*), whitetail deer (*O. virginianus*), moose (*Alces alces*), mountain goat (*Oreamnos americanus*), bighorn sheep (*Ovis canadensis*), mountain lion (*Felis concolor*), black bear (*Ursus americanus*), and grizzly bear (*Ursus arctos* subsp. *horribilis*). Because conservation of charismatic large fauna is a major management objective of the park, considerable data have been collected on the distribution, life history, and population dynamics of some of these species.

Between 1912 and 1950, tens of millions of nonnative salmonid eggs, fry, and fingerlings were introduced into GLAC waters. The principal species stocked was the Yellowstone cutthroat trout (*Oncorhynchus clarki bouvieri*); others included rainbow trout (*O. mykiss)*, brook trout (*Salvelinus fontinalis)*, lake trout (*S. namaycush*), lake whitefish (*Coregonus clupeaformis*) and kokanee (*O. nerka*) (Marnell et al. 1987). The west slope cutthroat (*O. c. lewisi*) is the only subspecies of cutthroat trout indigenous to the park. Other native salmonids within park waters include the bull trout (*Salvelinus confluentus*), mountain whitefish *(Prosopium williamsoni*), and pygmy whitefish (*Prosopium coulteri*). Several species of minnow, sucker, and sculpin are also native to park waters.

4. Aquatic Resources

GLAC contains 131 named lakes, 635 unnamed lakes, approximately 2,103 km of intermittent streams, and 2,506 km of perennial streams. The park is bordered on the west and south by the North and Middle Forks of the Flathead river, both of which are within the National Wild and Scenic

River system. Aquatic resources within the park are outstanding.

The headwaters of three major drainage systems are found in GLAC. Precipitation falling west of the Continental Divide flows into the North and Middle Forks of the Flathead River. Precipitation falling east of the Divide flows into the South Saskatchewan drainage and the headwaters of the Upper Missouri River.

Portions of the Missouri and Hudson Bay drainages within the park are headwater areas, and are therefore not influenced directly by land use in surrounding areas outside the park. The Flathead River Basin occupies the western half of the park. It is also primarily a headwater area within the park, but it does have some stream inputs from lands to the west of the park administered by the USDA Forest Service and lands to the north in Canada. These lands are impacted by a variety of land use activities, including logging and mining operations (Smillie and Flug 1984). Most waters in the park, however, are impacted only by within-park activities (e.g., recreation) and regional- to global-scale external stresses such as air pollution and climate change.

Many of the lakes and streams within the park have characteristics generally associated with acid sensitivity. They tend to be high in elevation, with little or no soils development in their watersheds, have steep slopes and flashy hydrology, and are hydrologically dominated by spring snowmelt. The majority of these surface waters are, however, actually not at all sensitive to acidification from acidic deposition due to the preponderance of glaciers within their watersheds and the occurrence of sedimentary bedrock. Both of these features contribute buffering in the form of base cations (Ca^{2+}, Mg^{2+}, Na^+, K^+) to drainage waters in sufficient quantity to neutralize any amount of SO_4^{2-} and NO_3^- that might be reasonably expected to be deposited from acidic deposition. There are some waters in the park that receive only modest contributions of base cations, however. These do not receive glacial meltwater contribution to any significant extent and are situated in watersheds with relatively inert bedrock. These sensitive waters are relatively rare within the park.

B. EMISSIONS

GLAC is located in the northwestern corner of Montana where emissions from urban and industrial sources are low. The Idaho-Montana border lies 125 km to the west, and currently there are no large emission sources in northern Idaho to cause concern for the park. The border between Canadian provinces Alberta and British Columbia lies at the northern end of the park. Emissions from British Columbia are low within 200 km of the park. Oil and gas processing plants in Waterton, Alberta are the largest point sources near the park, emitting approximately 30,000 tons/yr of SO_2 and 200 tons/yr of NO_x (Radian 1994). No major metropolitan areas are within 200 km of GLAC, and regional smog or haze typical of highly populated areas with high vehicle use is not present in the northwestern portion of the state (with the exception of valley locations in Missoula and other areas during winter). Particulates are Montana's main air quality concern, and major sources are agriculture, wood-burning stoves, coal mines, industrial activity and unpaved roads (Martin 1996). Forest fires, both natural and prescribed, in western Montana, Idaho and eastern Washington also contribute seasonally to Montana's particulates.

State-wide emission of SO_2, NO_x and VOC are summarized in Tables II-2, II-3 and II-4. One-half of the annual emissions of SO_2 are produced from mining processes and by electricity-generating power plants. SO_2 is a concern in four areas associated with industrial point sources: Billings, Helena, Colstrip and Great Falls. Oil refining, metals processing and electric power industries are the main sources in these areas. SO_2 sources within 200 km of GLAC are listed in Table VI-1 and are primarily timber- and metal-processing plants.

Major sources of nitrogen in Montana are point sources in Billings, Missoula, Colstrip and Great Falls (Martin 1996) and fossil fuel combustion associated with automobiles. The state has not monitored NO_2 in eight years, but there has been some company monitoring in Colstrip and Missoula as part of the PSD-permitting process. Local point sources within 200 km of GLAC generate relatively low annual emissions (Table VI-1). Colstrip, Columbia Falls, Missoula, Great Falls, and Billings all have major VOC point sources (greater than 100 tons/yr).

Table VI-1 Emissions of SO_2, NO_x, and VOC (tons/yr) from point sources emitting greater than 100 tons/yr. Montana counties are included that lie within 200 km of GLAC. (Source: Martin 1996)			
County	SO_2	NO_x	VOC
Flathead			
Plumb Creek		548	480
Columbia Falls Aluminum Co.	1,261		390
Lewis and Clark			
Asarco Incorporated	13,255		
Missoula			
Stone Container Co.	165	2,059	883
Louisiana-Pacific		180	162
Stimson Lumber			
Total	14,681	2,787	1,915

Winter inversions cause local increases of CO in Kalispell, 20 km south of West Glacier. Kalispell is the largest city in Flathead county, and most of the county's 70,000 residents live within a 25-km radius of the city. Emissions from wood-burning stoves, the Columbia Falls Aluminum Company, and automobiles combined with winter meteorological conditions cause the seasonal increases in CO (Montana Dept. Of Environmental Quality data).

Lead pollution is a concern in East Helena where the primary source of pollution is the ASARCO lead smelter. East Helena has been in non-attainment for lead since 1978, although the current state implementation plan was expected to bring the area into compliance by 1997 (Martin 1996). It is unlikely that a significant amount of lead pollution from ASARCO reaches GLAC from atmospheric transport.

Columbia Falls Aluminum Company (formerly Anaconda Aluminum Co.) is located 10 km south of GLAC in Columbia Falls and is a serious pollutant concern for the Flathead and GLAC region. The Columbia Falls aluminum plant has been in operation since 1955 and current annual emissions of SO_2 (1,261 tons/yr), CO (28,934 tons/yr) and fluoride from the company are the single largest source of pollution near the park (Martin 1996) (Figure VI-1). The maximum rate of fluoride emission, approximately 1,460 tons/yr, occurred in 1969; by 1971, the emission rate was reduced to 438 tons/yr as a result of changes in aluminum processing following a lawsuit against the Columbia Falls Aluminum Company. The current rate is approximately 150 tons/yr. Fluoride emissions can be trapped at lower elevations in the park during atmospheric inversions.

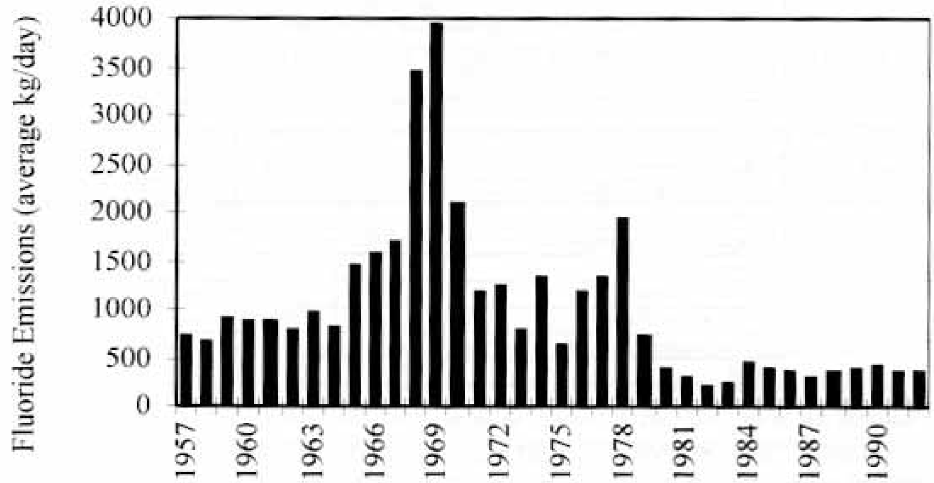


Figure VI-1. Annual average daily fluoride emissions from Columbia Falls Aluminum Co. plant from 1957 to 1992 (Colombia Falls Aluminum Co., Columbia Falls, MT).

Figure VI-1. Annual average daily fluoride emissions from Columbia Falls Aluminum Co. plant from 1957 to 1992 (Colombia Falls Aluminum Co., Columbia Falls, MT).

C. MONITORING AND RESEARCH ACTIVITIES

1. Air Quality

a. Wet Deposition

The concentration of major ions in precipitation at the NADP/NTN monitoring site in GLAC (at Apgar) is given in Table VI-2 for the years 1980 through 1995. Also provided are the precipitation amounts received during that time period. The concentration of SO_4^{2-} in precipitation has decreased since the early 1980s. Other parameters have remained relatively unchanged.

Wet deposition of both sulfur and nitrogen is very low at this site. In recent years, wet deposition of sulfur has fluctuated between 0.8 and 1.0 kg S/ha/yr. Total N wet deposition has been between 0.9 and 1.4 kg N/ha/yr (Table VI-3).

Snowpack samples were collected at Apgar Lookout in GLAC on March 18, 1996 and April 3, 1997. Concentrations of SO_4^{2-} and NH_4^+ in the snow samples were consistent between the two years, 5.2 and 4.9 μeq/L, respectively for SO_4^{2-}, and 1.9 and 2.3 μeq/L, respectively for NH_4^+. Nitrate concentrations differed by a factor of two, however, and were 2.5 μeq/L in 1992 and 5.2 μeq/L in 1993 (G.P. Ingersoll, pers. comm.). Concentrations of all three potentially-acidifying ions were generally somewhat lower in the samples collected at GLAC than in comparable snow samples collected at YELL and GRTE.

Table VI-2. Wetfall chemistry at the NADP/NTN site at Apgar, GLAC. Units are in μeq/L, except precipitation (cm).

Year	Precip	H^+	SO_4^{2-}	NH_4^+	NO_3^-	Ca^{2+}	Mg^{2+}	Na^+	K^+	Cl^-
1995	106.1	8.0	5.4	4.2	5.1	1.8	0.5	1.9	0.3	1.2
1994	67.3	8.5	7.6	5.8	7.4	3.3	0.9	1.9	0.7	1.3
1993	81.3	5.8	7.8	4.6	6.3	2.5	0.7	1.8	0.6	1.2
1992	71.0	4.7	8.1	4.8	6.7	4.9	1.2	2.4	0.8	1.4
1991	70.4	7.1	7.1	2.7	6.3	2.8	0.7	2.0	0.5	1.6
1990	112.2	5.5	7.2	5.1	5.8	2.9	0.8	2.0	0.5	2.1
1989	90.9	5.5	8.1	4.6	5.7	3.1	0.8	2.1	0.4	1.6
1988	72.9	5.5	7.9	3.1	5.4	4.2	1.0	2.4	0.4	1.7
1987	27.1	5.0	6.8	3.5	6.1	3.3	1.1	4.0	0.4	2.5
1986	75.7	7.1	7.7	2.9	5.4	3.1	1.0	2.3	0.5	1.7
1985	64.4	6.0	8.0	2.2	4.3	4.6	2.0	2.3	0.8	1.8
1984	80.0	5.6	10.4	4.0	6.7	4.9	2.2	2.5	1.0	2.8
1983	83.2	5.6	10.3	3.9	5.3	4.1	1.6	2.7	0.6	2.2
1982	91.9	5.6	8.9	3.9	6.0	4.3	2.0	2.3	0.5	1.8
1981	78.2	8.4	19.9	3.5	7.4	7.6	4.9	3.3	1.5	3.2
1980	56.3	12.4	16.0	3.4	6.5	4.1	1.1	2.4	0.6	4.5
Average	76.8	6.6	9.2	3.9	6.0	3.8	1.4	2.4	0.6	2.0

Table VI-3. Wet deposition (kg/ha/yr) of sulfur and nitrogen at the NADP/NTN site at Apgar, GLAC.

Date	Sulfur	NO_3-N	NH_4-N	Total Inorganic N
1995	0.9	0.8	0.6	1.4
1994	0.8	0.7	0.5	1.2
1993	1.0	0.7	0.5	1.3
1992	0.9	0.7	0.5	1.2
1991	0.8	0.6	0.3	0.9
1990	1.3	0.9	0.8	1.7
1989	1.2	0.7	0.6	1.3
1988	0.9	0.5	0.3	0.9
1987	0.3	0.2	0.1	0.4
1986	0.9	0.6	0.3	0.9
1985	0.8	0.4	0.2	0.6
1984	1.3	0.8	0.5	1.2
1983	1.4	0.6	0.5	1.1
1982	1.3	0.8	0.5	1.3
1981	2.5	0.8	0.4	1.2
1980	1.4	0.5	0.3	0.8
Average	1.1	0.6	0.4	1.1

b. Occult/Dry Deposition

Atmospheric concentrations of S and N species have been measured at GLAC since 1995 as part of the National Dry Deposition Network (NDDN). Dry deposition flux calculations are not yet available, but are expected to be available in the near future. Dry deposition of both N and S is expected to be low, however, probably less than 50% of wet deposition. Thus, the total deposition of both S and N is very low at GLAC, and shows no indication of increasing, given a lack of trends in the NADP data.

c. Gaseous Monitoring

Ozone, SO_2 and fluoride are gaseous pollutants monitored at GLAC (Table II-5). A continuous ozone analyzer has been in operation since 1991 at the air quality monitoring site located between Park Headquarters and Apgar (Figure VI-2). Average annual maximum one-hour ozone concentrations for 1991 to 1994 ranged between 61 and 77 ppbv (Table VI-4). These concentrations are below the NAAQS for ozone. The mean daytime 7-hour

Table VI-4. Summary of GLAC ozone concentrations (ppbv) from NPS monitoring sites (Joseph and Flores 1993)

	1991	1992	1993	1994*
1-hour maximum	62	77	58	61
Average daily mean	24	20	20	25
Growing season 7-hour mean	35	33	28	37
SUM60 exposure index (ppbv-hour)	670	961	0	61

* July through December data not collected

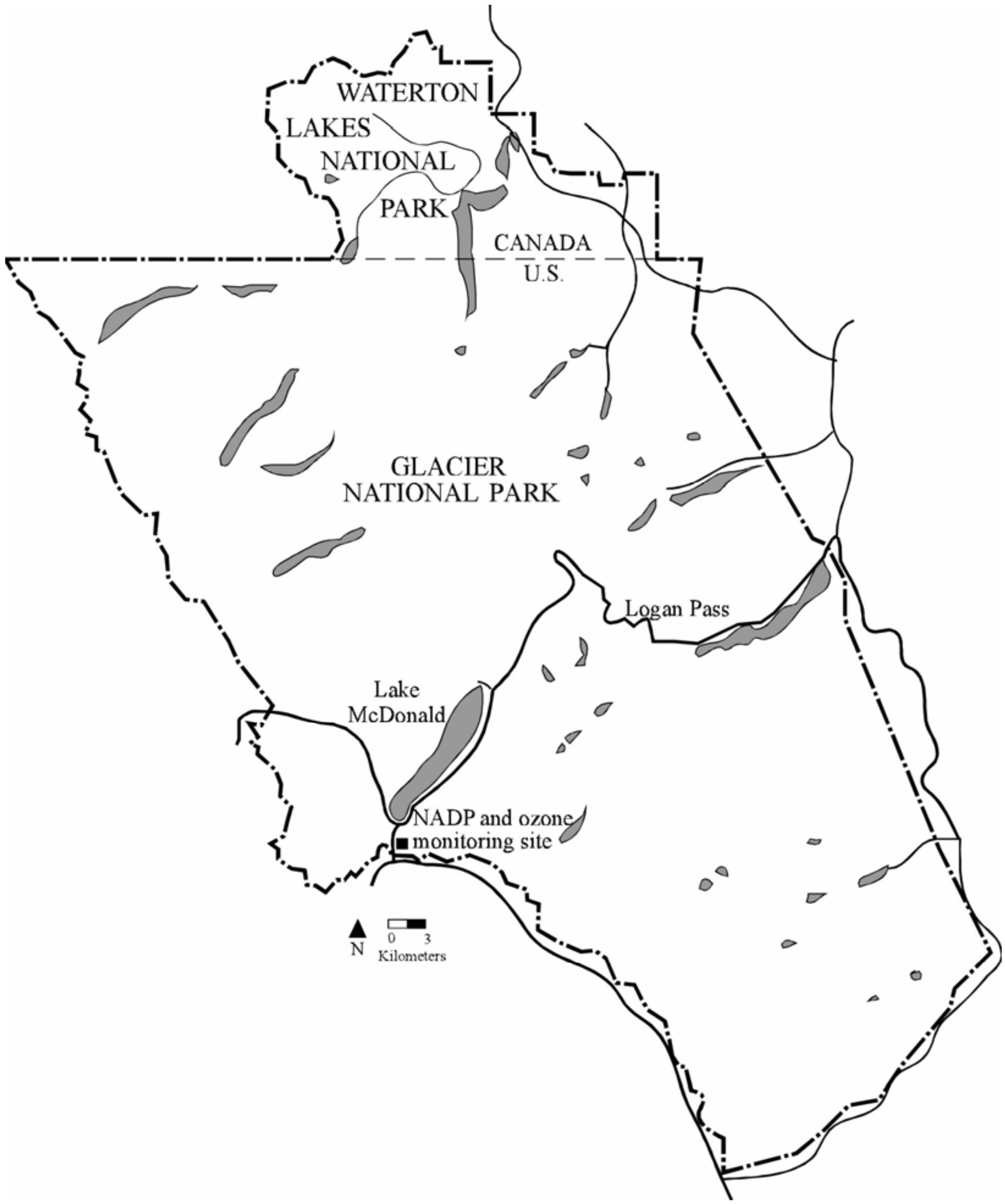


Figure VI-2. Location of the air quality monitoring site at GLAC.

Table VI-5. Yearly 24-hour average fluoride concentrations in GLAC. (Source: Columbia Falls Aluminum Co.

Year	Number of Days Sampled	HF (ppbv)
1977	275	123
1978	242	156
1979	342	103
1980	314	50

1981	344	44
1982	333	40
1983	360	43
1984	350	59
1985	345	81
1986	345	75
1987	361	65
1988	358	67
1989	285	82
1990	313	66
1991	316	79

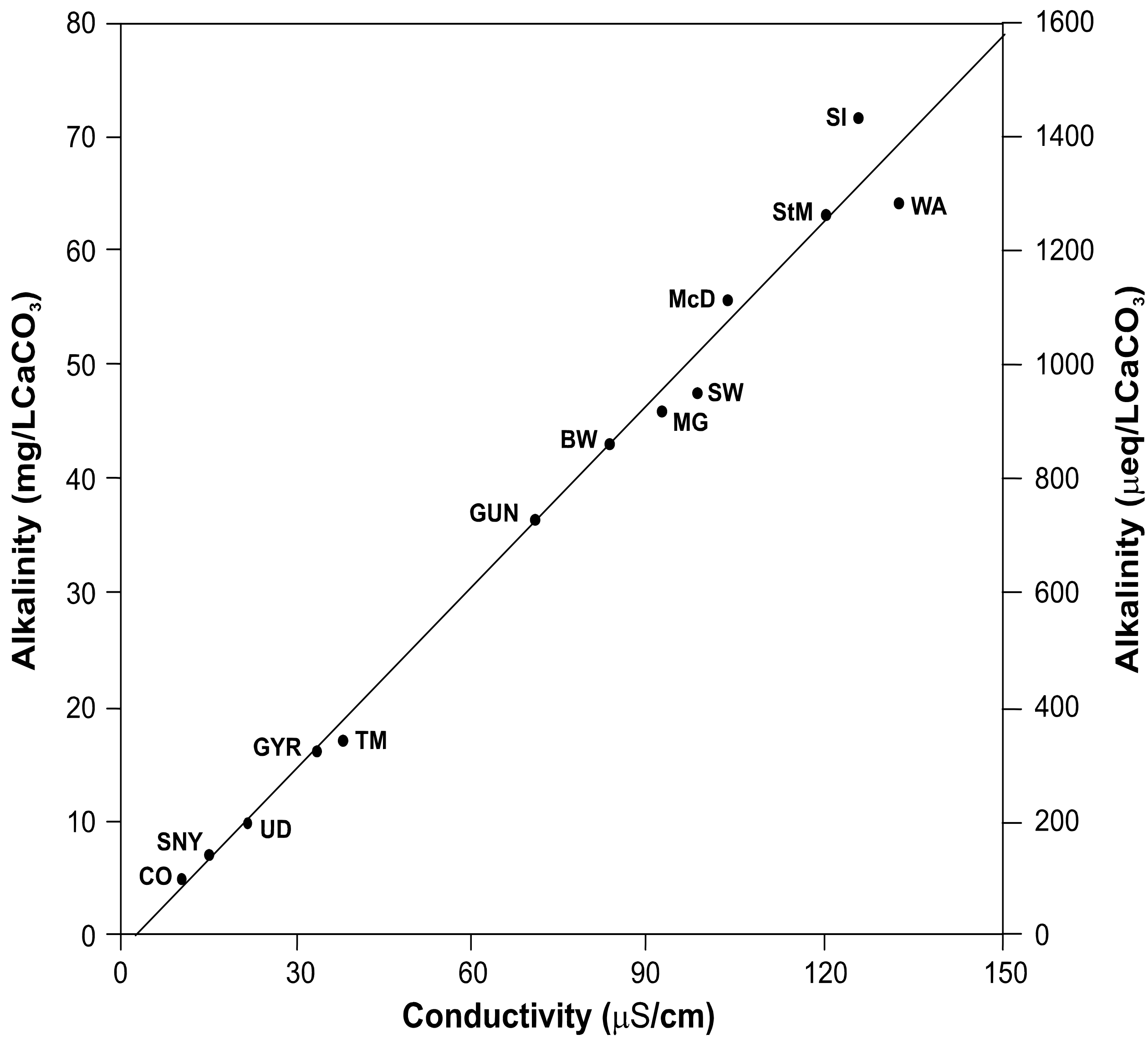
ozone concentration during the growing season ranged between 28 and 37 ppbv during this time period. The SUM60 exposure index is another indicator that can be important in assessing ozone exposures of plant species. This index is the sum of all hourly ozone concentrations equaling or exceeding 60 ppbv. The SUM60 exposure index at GLAC ranged between 61 and 961 in this time period. For comparison, National Parks in highly polluted areas (e.g., southern California) can have SUM60 exposure indexes exceeding 100,000 ppbv-hour (Joseph and Flores 1993).

The SO_2 measurements performed by NPS-ARD in 1991 and 1992 indicate that maximum annual concentrations in the park are between 0.15 and 0.30 ppbv. This is much less than the threshold dose (about 76 ppbv) that is considered potentially damaging to some crop species (Treshow and Anderson 1989). Mean values were 0.09 and 0.05 from 1991 and 1992, respectively. These values are below the estimated “clean air” reference of 0.19 ppbv (Urone 1976).

Columbia Falls Aluminum Company operates and maintains a sodium bicarbonate-tube fluoride monitor at GLAC. The fluoride monitor samples ambient air continuously for 24 hours and records daily cumulative concentrations of gaseous and particulate hydrogen fluoride. Yearly 24-hour averages of ambient hydrogen fluoride at this site between 1977 and 1992 ranged from 40 to 156 ppbv (Table VI-5). The decrease in fluoride emissions from the 1970s to the 1980s (Figure VI-1) corresponds with fluoride concentration data measured in the park.

2. Water Quality

Ellis et al. (1992) monitored water quality of eight small backcountry lakes and five large frontcountry lakes in GLAC. The objective was to establish a water quality baseline for the park. Data were collected during the period 1984 to 1990. The backcountry lakes were located in remote alpine headwater areas across the various regions and geologies of the park. Three of the lakes (Cobalt, Snyder, and Upper Dutch) had alkalinity less than about 200 μeq/L. Cobalt Lake had the lowest alkalinity (~100 μeq/L on average) and specific conductance (~10 μS/cm) of the study lakes (Figure VI-3) and would be expected to be sensitive to episodic effects of acidic deposition if sulfur or nitrogen deposition to the park increased substantially. If acidic deposition in the park increased dramatically



BW=Beaver Woman
CO=Cobalt
GUN=Gunsight
GYR=Gyrfalcon
McD=McDonald
MG=Medicine Grizzly
SNY=Snyder
SI=Stoney Indian
StM=St. Mary
SW=Swiftcurrent
TM=Two Medicine
UD=Upper Dutch
WA=Waterton

Figure VI-3. Mean alkalinity concentrations regressed against mean conductivity values for the frontcountry and backcountry lakes of GLAC, 1984-1990. R^2=0.989. (Source: Ellis et al. 1992) in the future, then perhaps Snyder Lake and/or Upper Dutch Lake, which also exhibited low alkalinity and specific conductance (Figure VI-3), would also be affected. The study lakes other than Cobalt, Snyder, and Upper Dutch would not be sensitive to acidification in response to any increase in deposition that would reasonably be expected to occur in the foreseeable future.

Measured pH values in Cobalt, Snyder, and Upper Dutch Lakes were generally in the range of about 6.0 to 7.0, although pH values as low as 5.5 were measured in all three lakes (Figure VI-4). Dissolved organic carbon concentration was low in all three lakes, in the range of 0.75 to 1.5 mg/L (Ellis et al. 1992). Thus, the amount of organic acidity in the lakes is low.

Cobalt Lake is situated in the southeastern portion of the park at an elevation of 2000 m. It lies immediately below a very steep alpine ridge, and therefore receives runoff that has limited contact with geological materials prior to entering the lake. Mean SO_4^{2-} concentration (n=9) was 9.6 μeq/L and mean sum of base cation concentration (n=9) was 85.2 μeq/L. These values suggest that watershed sources of sulfur are not significant and that there are not sufficient base cations to neutralize large amounts of acidic deposition.

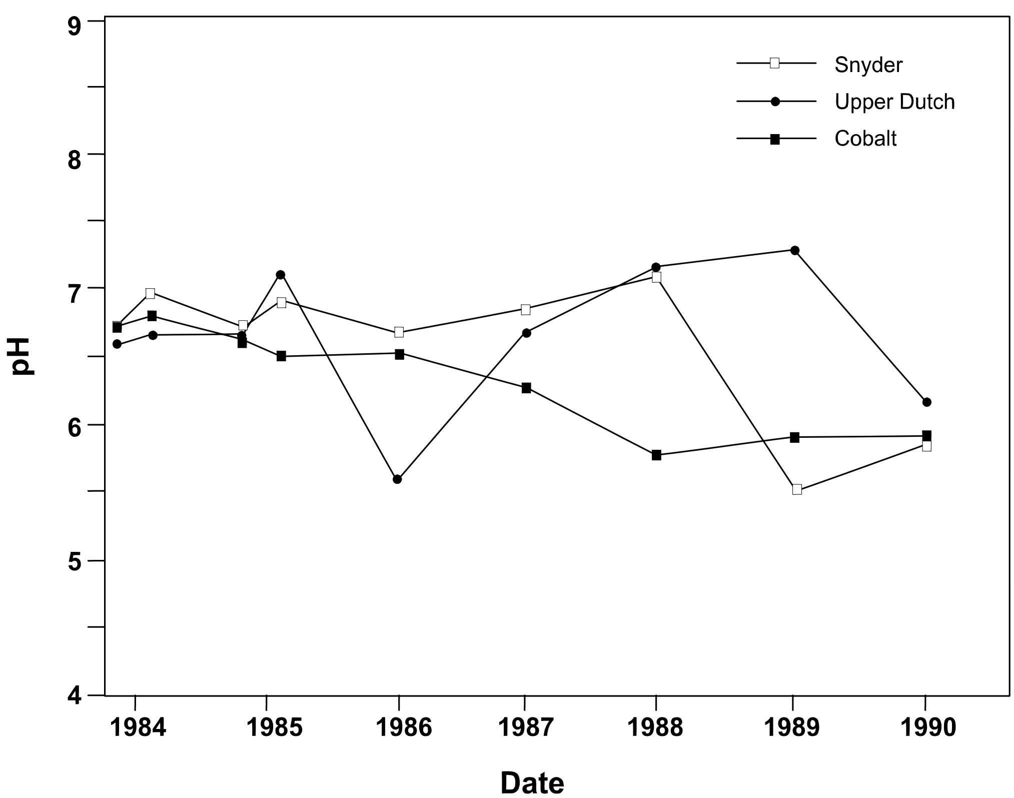


Figure VI-4. Time series plot of pH in Snyder, Upper Dutch and Cobalt Lakes, GLAC, 1984-1990. (Source: Ellis et al. 1992)

Soluble reactive phosphorus was consistently near the detection limit in all of the lakes studied by Ellis et al. (1992). The authors concluded that these lakes were oligotrophic to ultra-oligotrophic and were phosphorus-limited. Such lakes would not be expected to experience eutrophication in response to increased atmospheric nitrogen inputs. Rather, it would be expected that additional nitrogen contributed to the lakes would be reflected as increased lakewater NO_3^- concentration, which could contribute to acidification.

Water quality analyses were conducted by the U.S. Fish and Wildlife Service in 1978, 1979, and 1980 in 14 streams within the park: four North Fork Flathead River tributaries, seven Middle Fork Flathead River tributaries, and three tributaries of the Lake MacDonald watershed. Samples were collected during the months of June, July, August, and September for most of the streams included in the study. All samples analyzed had pH greater than 7.0, alkalinity greater than 600μeq/L, and specific conductance greater than 57 µS/cm (USFWS 1978, 1981). None of these streams would be at all sensitive to acidification from acidic deposition.

The EPA's Western Lakes Survey sampled five lakes in GLAC and ten other lakes in surrounding areas in the fall of 1986 (Landers et al. 1987). Measured values of selected important physical and chemical variables are listed in Table VI-6 for these fifteen lakes and their watersheds. The lowest pH value measured in the park was 7.1, although three of the lakes in surrounding areas had pH between 6.5 and 7.0. One of the lakes having lowest pH (6.6) contained significant natural organic acidity (DOC = 10 mg/L); the others were low in pH as a consequence of their dilute chemistry. Sulfate concentrations in lakewater were very low in lakes having low base cation concentrations. For example, the four lakes with total base cation concentrations less than 100 μeq/L all had SO_4^{2-} concentrations in the range of 5.7 to 10.1 μeq/L. Such concentrations of SO_4^{2-} are approximately what would be expected, based on SO_4^{2-} concentration in the precipitation (~6 to 8 μeq/L), negligible dry deposition of sulfur, and 30% to 50% evapotranspiration. These lakes are moderately to highly acid-sensitive, with ANC values of 21 to 84 μeq/L, although the two most sensitive were located outside the park boundaries. Many other lakes inside and outside the park had moderately elevated concentrations of SO_4^{2-}, in the range of 20 to 52 μeq/L. These relatively high concentrations of SO_4^{2-} are not attributable to atmospheric sulfur deposition. They reflect geological sources of sulfur in drainage waters, as also evidenced by the much higher concentrations (> 500 μeq/L) of base cations in all of the lakes that had SO_4^{2-} concentration greater than 20 μeq/L. Based on these data, it appears that GLAC and surrounding areas contain lakes that exhibit a mixture of acid sensitivities. Some lakes that have low concentrations of SO_4^{2-} (less than about 10 μeq/L) that can be reasonably attributed to atmospheric deposition inputs also have low concentrations of base cations. These lakes tend to be relatively acid-sensitive, and many have ANC values below 100 μeq/L. The lowest measured ANC in the park was 79 μeq/L. Other lakes are characterized by higher concentrations of base cations and SO_4^{2-} of geologic origin. These lakes are not acid-sensitive and have ANC values

Table VI-6. Results of lakewater chemistry analyses by the Western Lake Survey for selected variables in GLAC 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 GLAC											
No Name	4C3-00	2.8	88	1930	8.1	1210	28.6	11.6	929	1212	0.4
Feather Woman	4C3-01	3.7	44	2298	7.3	79	5.7	4.9	49	81	0.3
Harrison L.	4C3-01	162.0	5475	1126	8.0	543	32.1	4.5	375	613	0.5
Cobalt L.	4C3-01	4.4	62	2003	7.1	83	10.1	0.5	50	93	0.7
Glenns L.	4C3-06	104.8	4302	1482	8.1	1142	48.7	3.0	774	1210	0.7
Lakes outside GLAC											
	4C3-05	3.7	20	1979	6.6	21	8.0	0.1	29	57	1.5
	4C3-01	5.4	88	1932	8.1	1388	20.1	0.0	1096	1393	1.0
	4C3-01	6.2	173	1828	8.0	1209	32.3	0.7	832	1218	0.8
	4C3-02	1.8	108	2104	7.9	492	27.4	1.5	768	1092	0.3
	4C3-02	8.7	51	2050	7.5	360	10.0	2.1	295	386	0.8
	4C3-02	167.9	2188	1229	8.3	1409	52.4	0.1	965	1435	4.5
	4C3-05	2.3	7	1921	7.6	426	8.0	0.7	360	453	4.3
	4C3-06	12.7	77	1228	6.6	152	3.0	0.0	81	213	10.0
	4C3-03	6.4	54	2006	6.9	72	8.2	0.1	40	78	1.2
	4C3-05	1.8	31	2226	7.4	292	10.5	4.3	170	319	2.7

Table VI-7. pH measurements in lakes within GLAC, from NPS data base.		
Lake	Number of Observations	pH Range Measured
Beaver W.	9	7.6-8.0
Cobalt	10	5.4-6.8
Gunsight	9	6.6-7.9
Gyrfalcon	9	6.7-7.4
Lake 6182	1	7.3
McDonald	17	7.8-8.5
Medicine Gr.	9	6.4-8.3
Snyder	9	5.5-7.1
St. Mary	16	7.9-8.4
Stoney I	10	7.4-8.1
Swiftcurrent	11	7.8-8.5
Two Medicine	21	7.1-7.9
Upper Dutch	9	5.6-7.3
Waterton	15	7.9-8.5

greater than 500 μeq/L. Nitrate concentrations were generally below 5 μeq/L. One lake exhibited relatively high NO_3^- concentration (11.6 μeq/L) but this lake was not acid-sensitive.

The National Park Service database contains pH measurements from 14 lakes in the park (Table VI-7). Most lakes were sampled on multiple occasions. pH values generally ranged between 7.0 and 8.5. Several had measured pH values in the range that would suggest possible acid-sensitivity, however. These include Cobalt Lake (pH 5.4 to 6.8), Snyder Lake (pH 5.5 to 7.1), and Upper Dutch Lake (pH 5.6 to 7.3). Measurements of pH in Cobalt Lake during the period 1988 to 1990 were consistently under 6.0 (range of 5.4 to 5.9) (Ellis et al. 1991). Cobalt Lake had average Ca^{2+} + Mg^{2+} concentrations around 90 μeq/L and Snyder Lake had average Ca^{2+} + Mg^{2+} concentrations around 130 μeq/L. Each had SO_4^{2-} concentrations in the range of about 10 to 15 μeq/L. Neither would be considered highly sensitive to acid deposition impacts based on the observed concentrations of base cations relative to sulfate. However, measured pH values were often quite low. There are some problems with interpretation of these data however. The low reported pH values (< 6.0) are not consistent with the reported moderate concentrations of Ca^{2+} + Mg^{2+}. Clearwater lakes having pH less than 6 and SO_4^{2-} concentrations in the range of 10 to 15 μeq/L should have base cation sum (Ca^{2+} + Mg^{2+} + K^+ + Na^+) less than about 60 μeq/L and ANC less than about 50 μeq/L. In fact, Cobalt Lake (Lake ID 4C3-013) was sampled by the WLS and found to have pH of 7.1 and ANC of 84 μeq/L. Lakes such as Cobalt are expected to be sensitive to episodic acidification, but relatively insensitive to chronic acidification.

3. Terrestrial

In 1968-1970, observed foliar damage in ponderosa pine (*Pinus ponderosa*), lodgepole pine, western white pine (*P. monticola*) and Douglas-fir near the Columbia Falls Aluminum Co. plant led to concern of widespread fluoride impacts in downwind areas (US EPA 1973). These observations prompted the Forest Service and the National Air Pollution Control Administration (a predecessor of the EPA) in 1970 to develop a plan to analyze the extent of fluoride impacts on ecosystems in Flathead National Forest and Glacier National Park. The plan included evaluation of fluoride levels in foliage of conifers, shrubs and herbaceous species from sampling plots along transects radiating up to 25 km from the plant (Figure VI-5). From these sampling plots, the fluoride levels for all species were averaged and isolines were determined (Figure VI-6). Teakettle Mountain, 5 km northwest of Columbia Falls in Flathead National Forest, had fluoride concentrations in plants of 30 to 40 ppb. Within GLAC, fluoride concentrations in plants ranged from 10 to 60 ppb, with west-facing (windward) slopes of the Apgar Mountains and Belton Hills having higher levels than east-facing (leeward) slopes.

Seven permanent vegetation monitoring sites are located in the southwestern part of the park (Figure VI-7). Foliage samples (current and one-year old) of conifers, hardwoods, shrubs and mixed grass (deer and elk forage) collected in May, July and September are sent to the Environmental Studies Laboratory at the University of Montana for fluoride analysis. Summaries of analysis done on vegetation between 1968 and 1995 are listed in Table VI-8. In most species, there is variation in fluoride content throughout the season, but no regular or discernible pattern appears in the data, so only the maximum concentration is reported here.

Table VI-8. Maximum fluoride concentrations in foliage samples from GLAC, southwestern region downwind of Columbia Falls Aluminum Company (ppb by weight).

	1968[a]	1970[a]	1990[b]	1992[b]	1995[b]
Grasses		87	24	32	56
Mixed conifer	244	120			
Ponderosa pine	32	39			45
Douglas-fir					8
Lodgepole pine		15	16	19	6
Rocky Mountain Maple			11	16	12
Serviceberry			38	12	11
Pink dogbane			14	19	13
White pine			7	10	
Strawberry			26	43	18

[a] Carlson and Dewey (1971)
[b] Columbia Falls Aluminum Co. and GLAC (unpublished data)

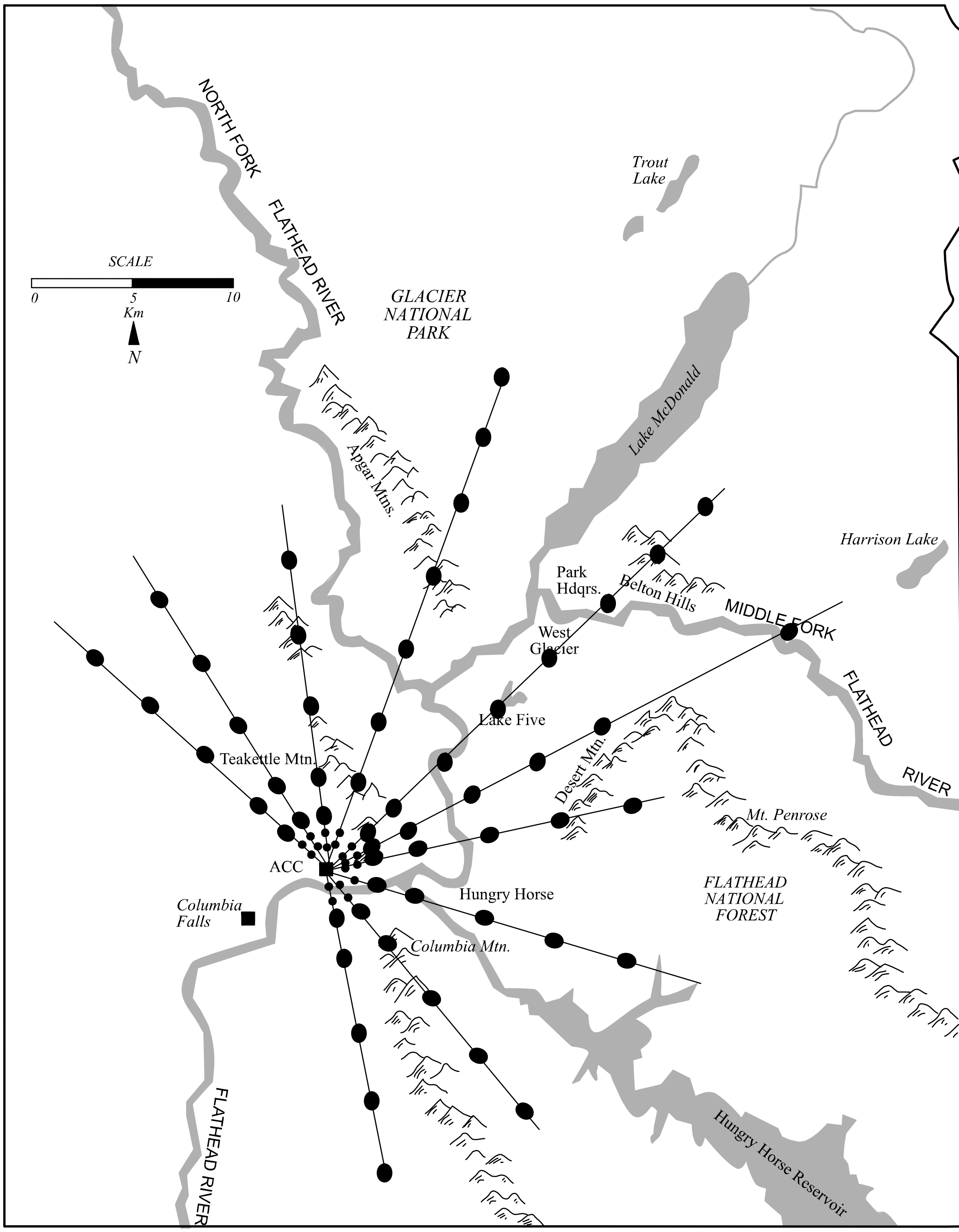


Figure VI-5. Transects of vegetation sampling for fluoride done in Flathead NF and GLAC in 1970 by USDA Forest Service and National Air Pollution Control Administration.

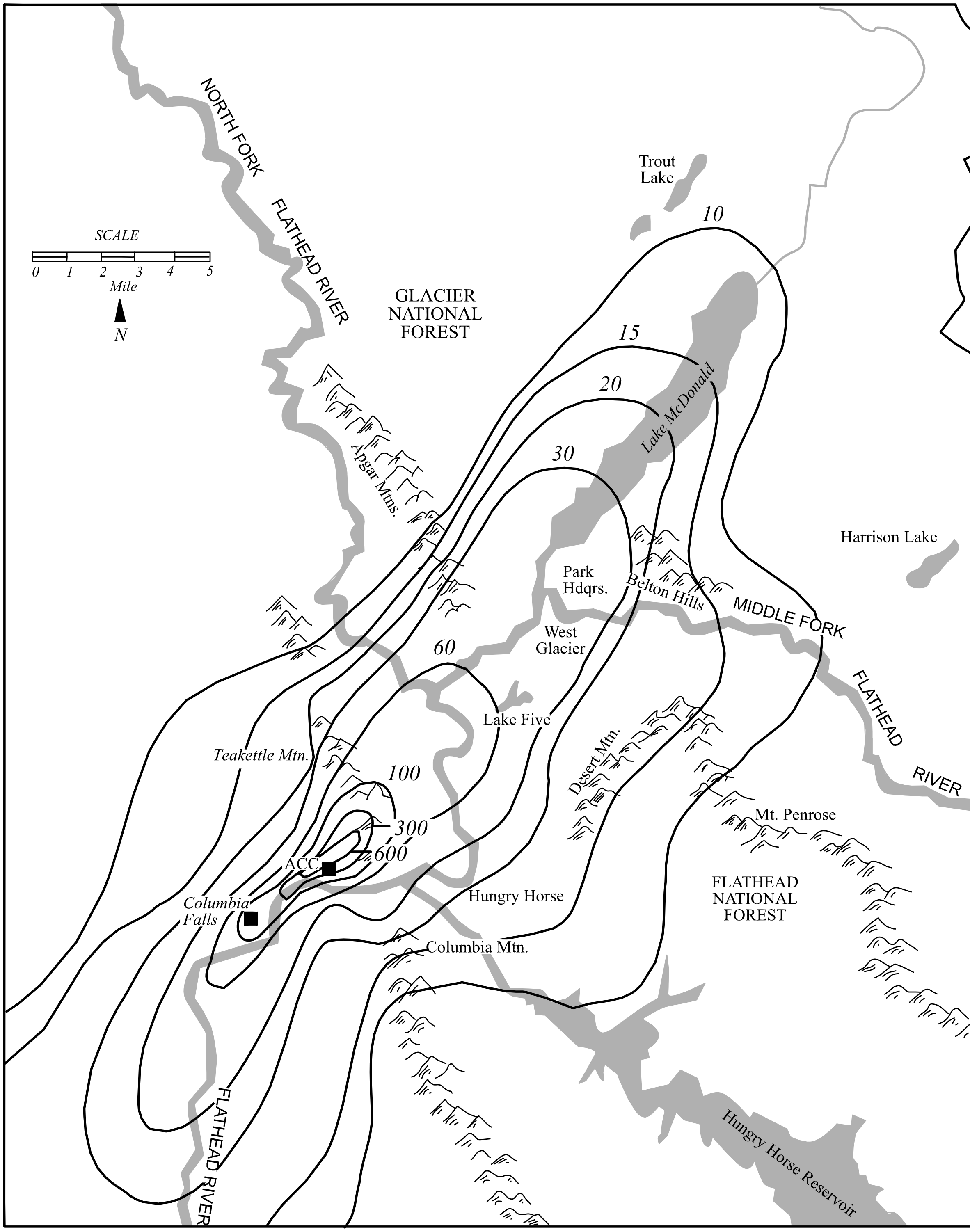


Figure VI-6. Isolines of fluoride (ppb) measured in foliage samples taken along transects from Columbia Falls Aluminum Company in 1970 (provided by GLAC).

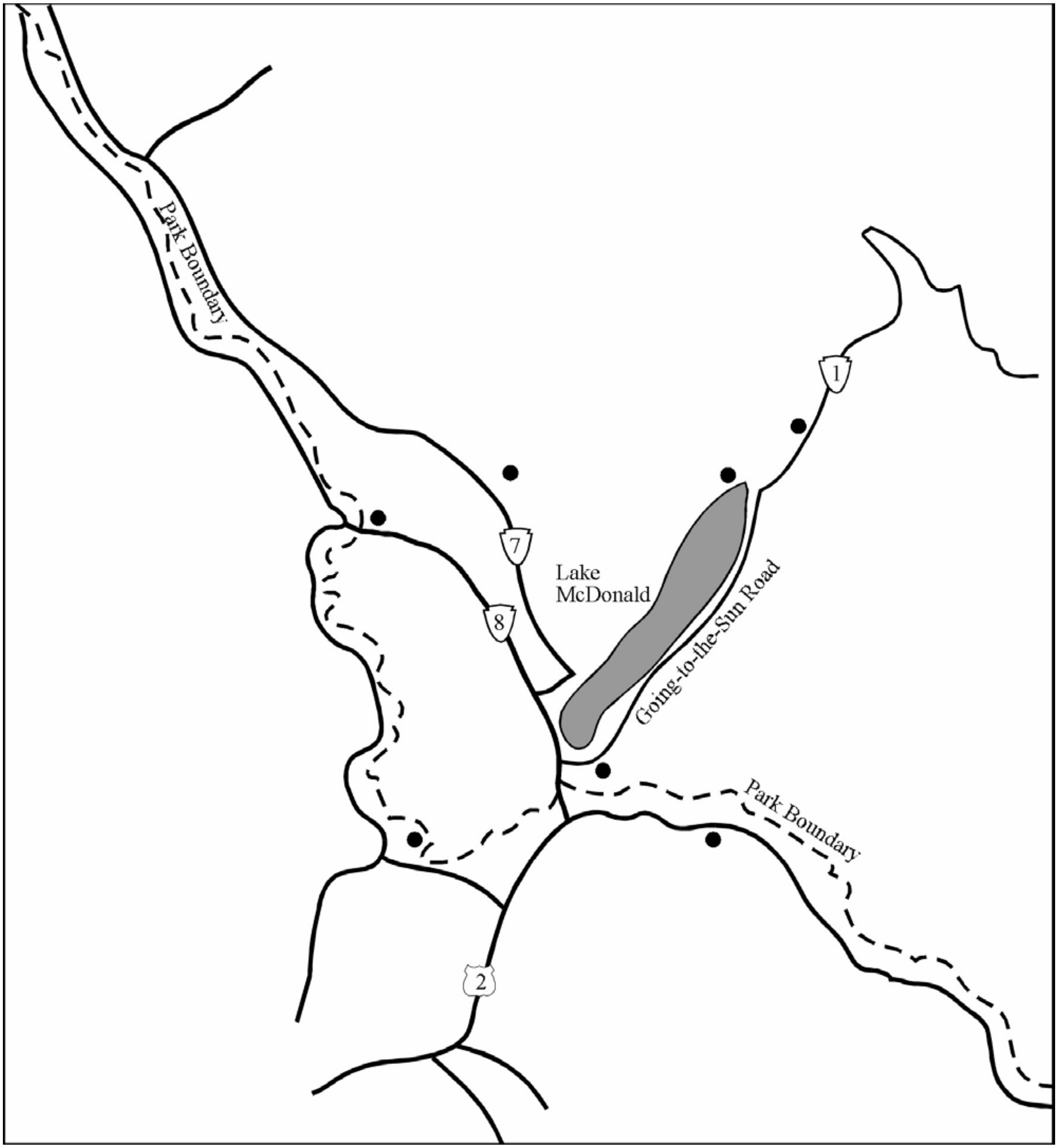


Figure VI-7. Permanent vegetation sampling locations in the southwestern portion of the park (provided by GLAC).

4. Visibility

As part of the IMPROVE network, visual air quality in GLAC, Montana, has been monitored using an aerosol sampler, transmissometer, and camera. The aerosol sampler has operated from March 1988 through the present in the vicinity of the Glacier horse stables, approximately 2 miles northwest of the GLAC west entrance. The transmissometer has been operating from March 1989 through the present and is located at the southern end of Lake McDonald, approximately 3 miles north of the park's west entrance. Two automatic 35mm camera systems were installed at the southern shore of Lake McDonald in 1985. The primary camera system was located at the Apgar pump house, viewing Lake McDonald and Garden Wall to the northeast from June 1985 through April 1995. A second camera system was located on the southeast shoreline of Lake McDonald, from 1985 through June 1991, viewing Teakettle Mountain and Lake McDonald to the southwest. Data from this IMPROVE site have been summarized to characterize the full range of visibility conditions for the March 1988 through February 1995 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 1994 includes March 1994 through February 1995). Complete descriptions of visibility characterization, mechanisms of sources and visibility impacts, and IMPROVE monitoring techniques and rationale are provided in the Introduction (Section I) of this document.

a. <u>Aerosol Sampler Data - Particle Monitoring</u>

IMPROVE aerosol samplers consist of four separate particle sampling modules that collect 24-hour filter samples of the particles suspended in the air. The filters are then analyzed in the laboratory to determine the mass concentration and chemical composition of the sampled particles. Particle data can be used to provide a basis for inferring the probable sources of visibility impairment. Practical considerations limit the data collection to two 24-hour samples per week. (Wednesday and Saturday from midnight to midnight). Detailed descriptions of the aerosol sampler, laboratory analysis, and data reduction procedures used can be found in the draft 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. The extinction coefficient represents the ability of the atmosphere to scatter and absorb light. Higher extinction coefficients signify lower visibility. A tabular and graphic summary of average reconstructed extinction values by season and year for the March 1988 through February 1995 period are provided in Table VI-9 and Figure VI-8, respectively.

Table VI-9. Seasonal and annual average reconstructed extinction (Mm^{-1}) and standard visual range (km) at GLAC, March 1988 through February 1995.

Year	Spring (Mar, Apr, May)		Summer (Jun, Jul, Aug)		Autumn (Sep, Oct, Nov)		Winter (Dec, Jan, Feb)		Annual (Mar-Feb)[a]	
	b_{ext} (Mm^{-1})	SVR (km)	b_{ext} (Mm^{-1})	SVR (km)	b_{ext} (Mm^{-1})	SVR (km)	b_{ext} (Mm^{-1})	SVR (km)	b_{ext} (Mm^{-1})	SVR (km)
1988	42.8	91	51.2	76	58.0	67	59.4	66	51.6	76
1989	48.9	80	49.3	79	59.7	66	71.2	55	56.5	69
1990	55.0	71	52.1	75	62.8	62	61.4	64	57.9	68
1991	49.4	79	51.0	77	74.5	53	73.4	53	61.3	64
1992	54.4	72	47.6	82	66.9	58	77.3	51	60.2	65
1993	43.6	90	43.4	90	72.9	54	67.2	58	56.1	70
1994	46.8	84	56.4	69	62.6	62	56.5	69	55.3	71
Mean[b]	48.7	80	50.1	78	65.3	60	66.6	59	57.7[c]	68[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 1995 period.

[c] Combined annual period data represent the mean of all combined season means.

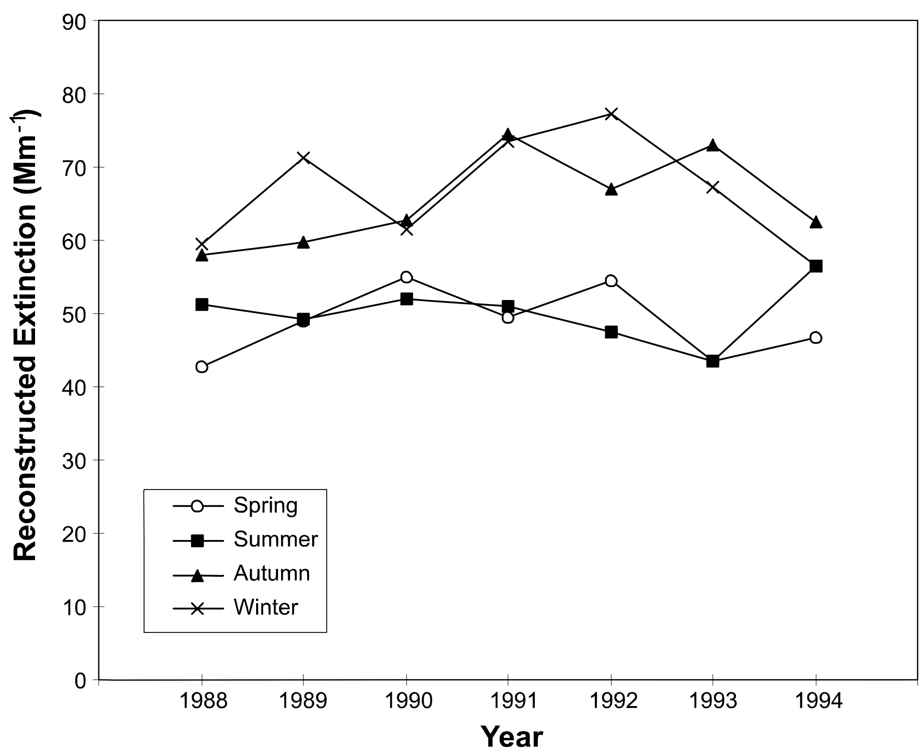


Figure VI-8. Seasonal average reconstructed extinction (Mm^{-1}) GLAC, March 1988 through February 1995.

Reconstructed extinction budgets generated from aerosol sampler data apportion the extinction at GLAC to specific aerosol species (Figure VI-9). 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% of days, mean of the median 20% of days, and mean of the dirtiest 20% of days. 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, visual range (in kilometers), and deciview (dv).

Standard Visual Range (SVR) can be expressed as:

$$\text{SVR (km)} = 3{,}912 / (b_{ext} - b_{Ray} + 10)$$

where b_{ext} is the extinction coefficient expressed in inverse megameters (Mm^{-1}), b_{Ray} is the site specific Rayleigh values (elevation dependent), 10 is the Rayleigh coefficient used to normalize visual range, and 3,912 is the constant derived from assuming a 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.

Deciview is defined as:

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

where b_{ext} is the extinction coefficient expressed in inverse megameters (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 Rayleigh atmosphere and increases as visibility is degraded. The segment at the bottom of each stacked bar 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.

The 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. 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.

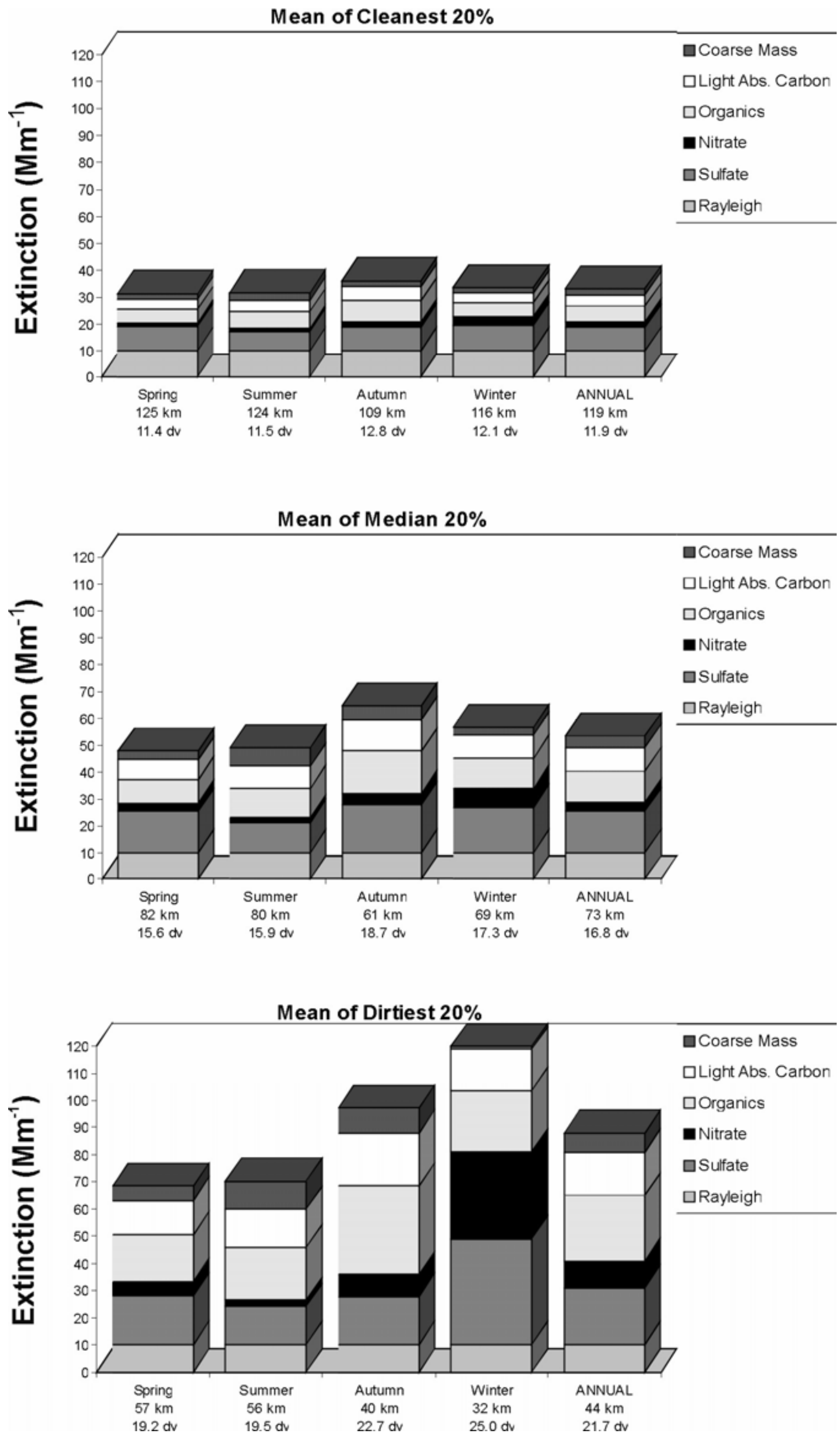


Figure VI-9. Reconstructed extinction budgets for GLAC, Montana, March 1988 - February 1995.

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. 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, 1993 and 1994).

Table VI-10 provides a tabular summary of the "filtered" seasonal and combined period arithmetic mean extinction values for the March 1989 through February 1995 period. Table VI-11 provides a tabular summary of the "filtered" seasonal and combined period 10% (clean) cumulative frequency values. 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 1989 through February 1995 period.
- Combined annual period data represent the unweighted mean of all combined seasonal b_{ext} values.

Figure VI-10 provides a graphic representation of the "filtered" annual mean, median, and cumulative frequency values (5th, 10th, 25th, 75th, 90th, and 95th percentiles). No data are reported for annual periods with one or more invalid seasons.

When comparing reconstructed (aerosol) extinction, Table VI-10 with measured (transmissometer) extinction, Table VI-11, 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.

Table VI-10. Seasonal and annual arithmetic means, GLAC, Montana transmissometer data (filtered) March 1989 through February 1995.

Year	Spring (Mar, Apr, May)			Summer (Jun, Jul, Aug)			Autumn (Sep, Oct, Nov)			Winter (Dec., Jan. Feb.)			Annual (Mar – Feb)[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
1989	94	42	14.4	94	42	14.4	81	49	15.9	97	41	14.1	91	44	14.7
1990	90	44	14.8	75	53	16.7	--	--	--	--	--	--	***	***	***
1991	102	39	13.6	90	44	14.8	84	47	15.5	104	38	13.4	95	42	14.4
1992	88	45	15.0	90	44	14.8	84	47	15.5	--	--	--	***	***	***
1993	104	38	13.4	94	42	14.4	97	41	14.1	117	34	12.2	103	39	13.5
1994	99	40	13.9	79	50	16.1	84	47	15.5	114	35	12.5	92	43	14.6
Mean[b]	92	41	14.2	83	46	15.2	83	46	15.3	103	37	13.1	93	43[3]	14.5

-- 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 March 1989 through February 1995 period.

[c] Combined annual period data represent the mean of all combined seasonal b_{ext} means.

Table VI-11. Seasonal and annual 10% (Clean) cumulative frequency statistics GLAC, transmissometer data (filtered) March 1989 through February 1995.

Year	Spring (Mar, Apr, May)			Summer (Jun, Jul, Aug)			Autumn (Sep, Oct, Nov)			Winter (Dec., Jan, Feb)			Annual (Mar – Feb)[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
1989	160	25	9.2	143	28	10.3	174	23	8.3	182	22	7.9	164	25	9.0
1990	143	28	10.3	133	30	11.0	--	--	--	--	--	--	***	***	***
1991	167	24	8.8	138	29	10.6	160	25	9.2	182	22	7.9	161	25	9.2
1992	138	29	10.6	143	28	10.3	138	29	10.6	--	--	--	***	***	***
1993	160	25	9.2	154	26	9.6	191	21	7.4	191	21	7.4	173	23	8.4
1994	167	24	8.8	138	29	10.6	160	25	9.2	148	27	9.9	153	26	9.7
Mean[b]	145	26	9.5	133	28	10.4	152	25	9.0	162	23	8.3	158	25[3]	9.3

-- 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 data represent the mean of all valid seasonal b_{ext} values for each March through February annual period.

[b] Combined season data represent the mean of all valid seasonal b_{ext} values for each season of the March 1989 through February 1995 period.

[c] Combined annual period data represent the mean of all combined seasonal b_{ext} values.

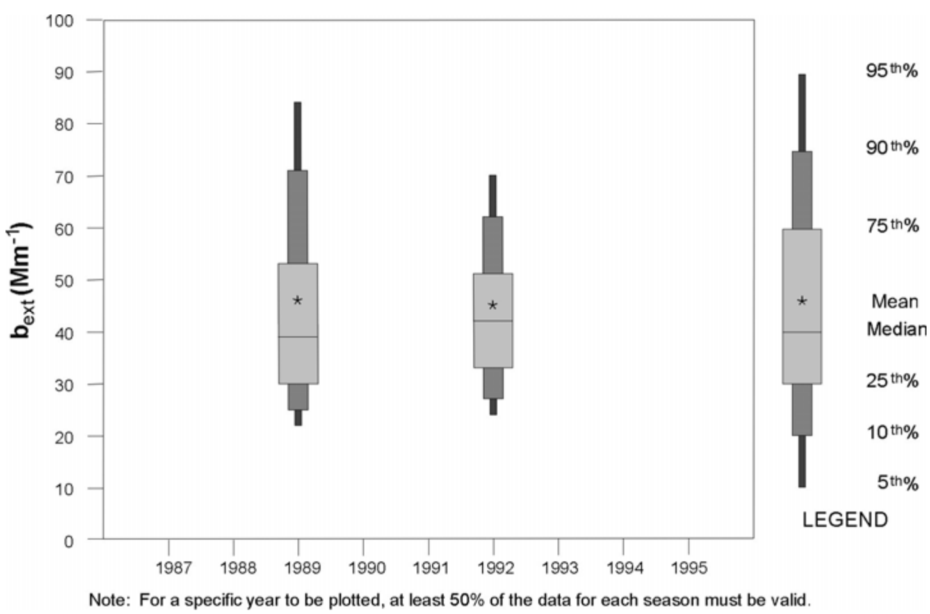


Figure VI-10. Annual arithmetic mean and cumulative frequency statistics, GLAC, Montana, transmissometer data (filtered).

- Relative Humidity (RH) Cutoff - Daily average reconstructed measurements are flagged as invalid when the daily average RH is greater than 98%. Hourly average transmissometer measurements are typically flagged invalid when the hourly average RH is greater than 90%. These flagging differences often result in data sets that do not reflect the same period of time, or misinterpret short-term meteorological conditions.

  Note: All March 1993 through February 1995 GLAC transmissometer data were processed using a site-specific relative humidity upper limit of 95%. The transmissometer site path at GLAC views across Lake McDonald and is relatively near the lake's surface. The site-specific RH limit was changed in the Spring of 1993 to reduce the number of hourly extinction values invalidated due to the higher relative humidity values associated with the site.

c. Camera Data - View Monitoring

Two automatic 35mm camera systems operated from the southern shoreline of Lake McDonald in GLAC. The primary system took photographs of Lake McDonald and Garden Wall three times per day from June 1985 through April 1995. A second camera system took photographs of Lake McDonald and Teakettle Mountain three times per day from June 1985 through June 1991.

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 Garden Wall vista photographs presented in Figure VI-11 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.

Glacier National Park
on a "clear" day

Representative Conditions:
Visual Range: 120-160 km
b_{ext}: 33-24 Mm^{-1}
Haziness: 12-9 dv

Glacier National Park
on an "average" day

Representative Conditions:
Visual Range: 70-90 km
b_{ext}: 56-43 Mm^{-1}
Haziness: 17-15 dv

Glacier National Park
on a "dirty" day

Representative Conditions:
Visual Range: 40-50 km
b_{ext}: 98-78 Mm^{-1}
Haziness: 23-21 dv

Figure VI-11 Photographs illustrating visibility conditions at Glacier National Park.

Figure VI-11 Photographs illustrating visibility conditions at Glacier National Park.

d. Visibility Summary

Data from other IMPROVE visibility sites around the country have been presented graphically (Figure I-1 and Figure I-2) so that visual air quality in the Rocky Mountains and Northern Great Plains regions can be understood in perspective. Figure VI-8 and Table VI-10 have been provided to summarize GLAC visual air quality during the March 1988 through February 1995 period. Seasonal extinction data should be reviewed with caution given the transmissometer's close proximity to Lake McDonald and higher than normal relative humidity values associated with the data. Long-term trends fall into three categories: increases, decreases, and variable. Given the visibility sites summarized for this report, the majority of data show little change or trends.

Non-Rayleigh atmospheric light extinction at GLAC, like many rural western areas, is largely due to sulfate, organics, and soil. 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 pre-settlement visibility below current levels during some summer months.

The IMPROVE aerosol monitoring network, established in March 1988, consists of sites instrumented with aerosol sampling modules A through D. Many of the IMPROVE sites are successors to sites where aerosol monitoring with stacked filter units (SFU) was carried out as early as 1979. SFU data were collected at GLAC from October 1982 through May 1986. Although some seasonal trends in absorption were observed for GLAC, no long-term trends were apparent in the 1982 through 1986 SFU data or the 1989 through 1994 site-specific or regional data presented in this report.

D. AIR QUALITY RELATED VALUES

1. Lakes and Streams

Sensitive fish species in the park include both native and non-native salmonids. Native western trout are sensitive to short-term increases in acidity. For example, eggs, alevins, and swim-up larvae of cutthroat trout were shown to have significantly higher mortality at pH 4.5 than at pH 6.5. Mortality was also higher at pH 5.0, but only statistically higher for eggs (Woodward et al. 1989).

It is unlikely that aquatic biota in GLAC have experienced adverse impacts to date from acidic deposition. This is because N and S deposition are low, the available lake water chemistry data do not suggest chronic acidification, and most sampled lakes appear to be relatively insensitive to acidification at any foreseeable levels of acidic deposition. However, in view of the apparent sensitivity of at least some waters in the park to future increases in acidic deposition and the outstanding quality of aquatic resources in GLAC, aquatic biota constitute important AQRVs within GLAC.

2. Terrestrial

Vegetation is the resource which is most sensitive to ozone and SO_2, and several tree species have been identified as potential bioindicators (see below). While ozone levels in the park have not exceeded the NAAQS, future increases in ozone might damage sensitive plant species. Potential impacts of fluoride are a source of concern due to emissions from the Columbia Falls Aluminum Company plant, because fluoride injury in conifer needles causes tip burn, reduced diameter growth and abscission of older foliage (Shaw et al. 1951). However, little is known about the sensitivity of native plant species to fluoride.

In addition to vegetation, insects, mammals, and other organisms may be impacted by fluoride. It is stored in plant tissues including leaves, stems, cambium and bark, and is passed on to numerous organisms which feed on these tissues. Studies in the GLAC region of fluoride content from foliage- and cambial-feeding, predaceous, and pollinating insects found the highest concentrations of fluorides (400 ppb dry weight) in bumblebees (Carlson and Dewey 1971). Predaceous species had fluoride levels ranging from 60 to 170 ppb dry weight and foliage feeders ranged from 21 to 48 ppb dry weight. High levels of fluoride have also been found in other organisms. Bone tissue from grouse, chipmunks, deer mice and deer in the GLAC region had between 4 and 6 times more fluoride than control animals (Gordon 1972, 1974). It is unknown what levels of fluoride will cause fluorosis, a toxic response in animals.

Ponderosa pine is not widespread in GLAC, although stands on the western edge of the park are suitable for establishing long-term monitoring plots for ozone. Of the hardwood species present at GLAC, quaking aspen is the most sensitive to ozone. Aspen grows at various locations in riparian ecosystems and in fire- or avalanche-disturbed areas in the park. Black cottonwood (*Populus trichocarpa* var. *balsamifera*), another potential bioindicator for ozone (Woo 1996), has symptoms similar to those of aspen. However, it is generally regarded as less sensitive to ozone than aspen. Neither of these hardwood species has the clarity of ozone symptomatology found in ponderosa pine.

Aspen is also considered to be sensitive to SO_2 and may be the best sensitive receptor for this gaseous pollutant. Injury is similar to that normally found for ozone (stippling, followed by bifacial necrosis), although SO_2 - induced injury rapidly bleaches to a light tan color, whereas ozone injury remains dark (Karnosky 1976). Caution should be used in distinguishing ozone injury from SO_2 injury.

Most fluoride sensitivity studies have focused on ponderosa pine (both varieties). Early accounts of fluoride injury in ponderosa pine were reported near Kaiser Aluminum Company in Mead, Washington, with symptoms of retarded stem-diameter growth and foliar necrosis (Shaw et al 1951, Lynch 1951, Adams et al 1956). A study conducted in the vicinity of Harvey Aluminum Company in The Dalles, Oregon, found foliar injury in ponderosa pine to be attributed to elevated fluoride and not fungal, insect or pathogen agents (Compton et al. 1961). Later studies done in Flathead National Forest and GLAC in 1974 indicated that western white pine appeared to be the most sensitive conifer to fluoride (Carlson and Dewey 1971); ponderosa pine, lodgepole pine, and Douglas-fir were moderately sensitive, and Engelmann spruce, western redcedar, and subalpine fir were most tolerant. Of the shrub species observed, Carlson and Dewey found that chokecherry (*Prunus virginiana*) and serviceberry (*Amelanchier alnifolia*) appeared more sensitive to fluoride than other shrubs. False-lily-of-the-valley (*Maianthemum dillatatum*) and disporum (*Disporum hookeri*) were the most sensitive of the herbaceous species studied. These are general classifications based on field data and observations, not on data from laboratory studies.

A species list of native plants is available in the NPFlora database. Table VI-12 summarizes vascular plant species of GLAC with known sensitivity to ozone, SO_2 and NO_x. This table is based on a variety of sources from the published literature and other information. It should be noted that the various sources used a wide range of field and experimental approaches to determine pollutant pathology, and that sensitivity ratings are general estimates based on published information and our expert opinion. While it will not be possible for park staff to collect data on all the species indicated in Table VI-12, the list can be used by park managers as a guide for identifying visible symptoms. Of the many plant species in GLAC, it is likely that there are many other species which have high sensitivity to air pollution, but we currently have no information about them.

Table VI-12. Plant species of GLAC with known sensitivities to sulfur dioxide, ozone and nitrogen oxides. (H=high, M=medium, L=low, blank=unknown). (Sources: Esserlieu and Olson 1986, Bunin 1990, Peterson et al. 1993, National Park Service 1994, Electric Power Research Institute 1995, Binkley et al. 1996, Brace and Peterson 1996)

Species Name	SO_2 Sensitivity	O_3 Sensitivity	NO_x Sensitivity
Abies lasiocarpa	L	L	
Acer glabrum	H		
Acer glabrum douglasii	M		
Achillea millefolium		L	
Agastache urticifolia		M	
Agoseris glauca	M		
Amaranthus retroflexus	M		
Ambrosia psilostachya		L	
Amelanchier alnifolia	H	M	
Arctostaphylos UVa-ursi	L	L	
Artemisia ludoviciana	M		
Artemisia tridentata	M	L	
Betula occidentalis	M		

Table VI-12. Continued.

Species Name	SO_2 Sensitivity	O_3 Sensitivity	NO_x Sensitivity
Betula papyrifera	H	H	
Betula papyrifera commutata	H		
Betula papyrifera subcordata	H		
Bromus carinatus		L	
Bromus tectorum		M	
Carex siccata		L	
Cassiope mertensiana		L	
Ceanothus sanguineus	M		
Ceanothus velutinus	L		
Ceanothus velutinus laevigatus	L		
Chenopodium fremontii		L	
Cirsium arvense		L	
Cirsium undulatum	M		
Clematis ligusticifolia	M		
Collomia linearis		L	
Convolvulus arvensis	H		
Cornus stolonifera	M	L	
Crataegus columbiana	M		
Crataegus douglasii	L		
Descurainia pinnata		L	
Erigeron peregrinus		L	
Fragaria virginiana		H	
Galium bifolium		L	
Gayophytum racemosum		L	
Gentiana amarella		M	
Geranium richardsonii	M	M	

Hackelia floribunda	L		
Hedysarum boreale		M	
Helianthus anuus	H	L	
Holodiscus discolor	H	H	
Hymenoxys richardsonii	L		
Juniperus communis	L		
Juniperus scopulorum	L		
Larix lyalii	H		
Larix occidentalis	H		
Lonicera involucrata	L	H	
Medicago sativa		M	
Mimulus guttatus		L	
Osmorhiza occidentalis		L	
Pachistima myrsinites		L	
Phacelia heterophylla		L	
Philadelphys lewisii	H		
Phyllodoce empetriformis		L	
Picea engelmannii	M	L	
Picea glauca	M	L	
Pinus contorta	M	M	H
Pinus contorta latifolia	H	M	
Pinus flexilis	L		
Pinus monticola	M	M	

Table VI-12. Continued.

Species Name	SO_2 Sensitivity	O_3 Sensitivity	NO_x Sensitivity
Pinus ponderosa	M	H	H
Poa annua	H	L	
Poa pratensis		L	
Polygonum douglasii		L	
Populus balsamifera trichocarpa	M	H	
Populus tremuloides	H	H	
Populus trichocarpa	M	H	
Potentilla fruticosa		L	
Prunus emarginata	H		
Prunus virginiana	M	H	
Pseudotsuga menziesii	M	L	H
Pseudotsuga menziesii glauca	M	L	
Ribes lacustre	M		
Ribes viscosissimum	M		
Rosa woodsii	M	L	
Rubus idaeus	H		
Rubus parviflorus	H	M	
Rumex crispus		L	
Salix scouleriana		M	
Sambucus racemosa		L	
Shepherdia canadensis	L		
Sorbus scopulina	M		
Sorbus sitchensis	H		
Symphoricarpos albus		H	

Taraxacum officinale		L	
Taxus brevifolia	L		
Thuja plicata	L		
Tragopogon dubius	M		
Trifolium pratense	L		
Trifolium repens		H	
Trisetum spicatum	M		
Tsuga heterophylla	M		
Tsuga mertensiana	H		
Urtica gracilis		L	
Vaccinium membranaceum		M	
Vicia americana		L	
Viola adunca		L	

An inventory of lichen species with known sensitivity to ozone and SO_2 is summarized in Table VI-13. As in Table VI-12, this table is based on a variety of sources from the published literature and other information. It should be noted that diagnostic symptoms of air pollutant injury to lichens are difficult to identify, and that some species have reduced productivity or even mortality without exhibiting visible symptoms (Nash and Wirth 1988). One of the best sources of background information and guidelines for addressing the use of lichens as sensitive receptors of air pollution is Stolte et al. (1993). The inventory of GLAC lichen species performed by DeBolt and McCune (1993) provides a good baseline for further assessment of potential impacts of air pollution.

Table VI-13. Lichen species of GLAC with known sensitivity to SO_2 and ozone. (H=high, M=medium, L=low, blank=unknown) (Sources: Peterson et al. 1993, Electric Power Research Institute 1995, Binkley et al. 1996).

Species	SO_2 sensitivity	Ozone sensitivity
Acarospora chlorophana	H	
Alectoria sarmentosa		H
Bryoria abbreviata		H
Bryoria capillaris	H	
Bryoria fremontii		H
Bryoria fuscescens	M	
Bryoria glabra	M	H
Bryoria implexa	H	
Bryoria oregana		H
Calicium viride	L-M	H
Caloplaca flavorubescens	H	
Caloplaca holocarpa	M	
Candelariella vitellina	M	
Cetraria canadensis		H
Cetraria cucullata	M	
Cetraria islandica	M	H
Cetraria nivalis	M	
Cetraria pinasrtri	M	

Chrysothrix candellaris	L	
Cladonia coniocraea	M	
Cladonia fimbriata	M-H	
Cladonia gracilis	L-M	
Coeocaulon muricatum	L-M	
Collema tenax	M	
Evernia prunastri	L-M	H
Hypocenomyces scalaris	M	
Hypogymnia imshaugii	L-M	L-M
Hypogymnia physodes	L-M	
Lecanora carpinea	M	
Lecanora dispersa	L	
Lecanora hageni	L-M	
Lecanora muralis	M	
Lecidea atrobrunnea	L	
Letharia columbiana	L	L-M
Letharia vulpina	L	L-M
Lobaria pulmonaria	H	L-M
Melanelia elegantula		L
Melanelia exasperatula	M	
Melanelia subaurifera		H
Melanelia subolivacea		L-M
Mycoblastus sanguinarius	L-M	
Parmelia saxatilis	M	L
Parmelia sulcata	L-M	M-H
Parmeliopsis ambigua	M	
Parmeliopsis hyperopta	M	

Table 13. Continued

Species	SO_2 sensitivity	Ozone sensitivity
Peltigera aphthosa	M	H
Peltigera canina	L	H
Peltigera collina		H
Peltigera didactyla		H
Peltigera refuscens		M-H
Phaeophyscia orbicularis	M	H
Phaeophyscia sciastra		H
Physcia adscendens	M	
Physcia aipolia	M	
Physcia caesia	M	
Physcia dubia	M	
Physcia stellaris	L-M	
Physcia tenella	M	L
Physconia detersa	M-H	L
Platismatia glauca	M	H
Pseudephebe minuscula		M
Pseudephebe pubescens		M
Ramalina obtusata	M	
Rhizocarpon geographicum	L	
Rhizoplaca chrysoleuca	H	
Rhizoplaca melanophthalma	H	

Umbilicaria polyphylla	L-M	
Usnea subfloridana	M-H	
Xanthoparmelia cumberlandia	H	
Xanthoria candelaria	M-H	H
Xanthoria elegans	M	
Xanthoria fallax	M-H	L
Xanthoria polycarpa	M	L

A fluoride injury index was developed by Carlson and Dewey (1971) from foliage collected in Flathead National Forest and GLAC (Table VI-14). Their classification was based on visual injury found in lodgepole pine, western white pine, whitebark pine, ponderosa pine and Douglas-fir between the 30 and 100 ppb dry weight isolines (Figure VI-6).

Table VI-14. Classification of visual injury of fluoride in conifers. (Source: Carlson and Dewey 1971)

Class	Fluoride Concentration (ppb)
Non-injured	< 60
Light	61-500
Moderate	501-999
Severe	> 1000

E. RESEARCH AND MONITORING NEEDS

1. Deposition and Gases

We recommend continued gas and deposition monitoring at the GLAC Fire Weather Station (Apgar). This will provide important long-term data for continued evaluation of potential resource effects. This is needed in view of the sensitivity of some resources and the outstanding quality of aquatic resources in the park, including the importance of the native cutthroat trout gene pool.

Little is known of the present threats of ozone, SO_2, and NO_2 on native vegetation in GLAC. Ozone data are currently collected at GLAC from a continuous electronic analyzer near park headquarters. An additional analyzer installed seasonally (for summer months) at a high elevation site (e.g., Logan Pass Visitors Center) for three consecutive years would provide valuable information on the diurnal pattern and elevational differences of ozone exposure within the park. Studies of ozone profiles done in other mountainous areas including the Swiss Alps (Sandroni et al. 1994), the Sierra Nevada and the Cascade Mountains (Brace and Peterson 1996, 1998) have shown that topography and elevation can influence the diurnal exposure. Low-elevation sites have diurnal profiles characterized by low ozone levels during the nighttime and early morning hours, maximum levels during the mid or late afternoon, and low levels again in the evening. High-elevation sites typically have lower maximum ozone concentrations than low elevation sites, but ozone remains elevated during the morning and nighttime hours. Plant species at higher elevations may be at risk from exposure to elevated levels of ambient ozone during the morning hours when they are physiologically active.

In addition to maintaining the permanent continuous ozone analyzer in the Apgar area, we believe that it would be useful to install a network of passive samplers to obtain data on spatial variability of weekly ozone exposures in different areas of the park. Two transects should suffice, placed along elevational gradients and sampled weekly for two months each summer for three years. Transects should be located in areas with maximum airflow and downwind from potential precursor sources (VOC and NO_x). An additional transect could be placed in an area considered least exposed to transported ozone and precursors.

Continued SO_2 monitoring is important, considering the proximity of Columbia Falls Aluminum Company, the largest point source of SO_2 in the area. In addition, continued fluoride monitoring will contribute valuable data on fluoride deposition in GLAC.

2. Aquatic Systems

Ellis et al. (1992) recommended continuous monitoring of three lakes in GLAC, using a mass balance approach to allow interpretation of the impact of incremental increases in acidic deposition on lake buffering capacity. This would include quantification of solute fluxes into and out of the alpine lakes, including measurements of discharge, snow pack surveys, and air temperature and humidity data (for evaporation estimation). They recommended depth profiling of key variables at least three times per year, once during winter ice cover, immediately following surface ice melt, and during late summer. Cobalt, Upper Dutch, and Medicine Grizzly were judged to be easily accessed and representative of the range of conditions found in the alpine lakes. These were the preferred candidate lakes for continuous monitoring. Ellis et al. (1992) also recommended monitoring other lakes on a five-year interval schedule.

It is our judgment that none of the lakes studied by Ellis et al. (1992) are sufficiently acid-sensitive as to warrant the intensive monitoring effort proposed. In our view, only Cobalt Lake is sufficiently dilute (sensitive) to be acidified by air pollutants to ANC approaching zero on an episodic basis in the foreseeable future. None of the lakes are at risk for chronic acidification to ANC < 0. Our judgement is based on the observations that 1) both ANC values and C_B minus (SO_4^{2-} + NO_3^-) values are consistently greater than about 70 to 80 µeq/L, and 2) the sum of (SO_4^{2-} + NO_3^-) concentrations in the lowest ANC lakes are less than 20 µeq/L and generally closer to about 10 µeq/L. Thus, the concentration of (SO_4^{2-} + NO_3^-) in these lakes would need to increase by more than a factor of four, perhaps by a factor of ten, before such lakes would become chronically acidic (ANC < 0) (cf., Husar et al. 1991, Sullivan and Eilers 1994). Cobalt Lake is a good candidate for continued chemical and ecological monitoring, however. We recommend water quality sampling twice per year, immediately after ice-out and during July or August. We do not advocate hydrologic or additional deposition monitoring for this lake at this time. Snyder Lake is also a good candidate for continued chemical and ecological monitoring because it is located on the west side of the Continental Divide in the central portion of the park. It is exposed to air masses that are transported to the park from the southwest. Even though it is located in a headwater area, the elevation is fairly low (1,597 m), which is likely a major factor responsible for the lake having fish present (Ellis et al. 1992). However, based on its measured lakewater chemistry, Snyder Lake is relatively insensitive to acidic deposition effects. The average sum of base cation concentrations measured by Ellis et al. (1992) was 152 μeq/L and average alkalinity at about the same level. Snyder Lake would not be expected to acidify due to acidic deposition in the future. Average SO_4^{2-} concentration was only 13 μeq/L, suggesting that most SO_4^{2-} in the lake is of atmospheric origin. Nevertheless, monitoring this lake will provide indication of changes in the contributions of SO_4^{2-} and NO_3^- to lakewater during snowmelt and on a chronic basis. It is relatively easily accessed and has the added benefit of a biological receptor in the form of native cutthroat trout. If, for example, lakewater concentrations of SO_4^{2-} or NO_3^- were to increase dramatically (perhaps three-fold) in the future, then additional biological monitoring would be appropriate.

Ellis et al. (1992) did not sample lakes in the northwestern portion of the park. The closest to that area of the park was Gyrfalcon Lake. Based on examination of topographic maps, Gyrfalcon Lake appears to receive substantial glacial meltwater input. This would account for its relatively high concentrations of base cations and alkalinity. Many lakes situated in areas expected to be highly sensitive to acidic deposition within this part of the park (i.e., cirque lakes below steep ridges) are, in fact, fed by glaciers. Due to the high degree of buffering generally found in glacial meltwater, none of these lakes would be expected to be highly sensitive to the effects of acidic deposition. There are a few appropriate candidate lakes for high acid-sensitivity, however, in the northwestern portion of the park. These include Long Bow Lake (and its unnamed upstream tributary lake) and Pocket Lake. Each is located on the west side of the Continental Divide in a headwater area and apparently without substantial glacial meltwater contributions. The logistics of sampling each of these lakes may be difficult. If feasible, we recommend sampling each lake once for water quality, and if one is found to be highly sensitive, adding it to the list of recommended long-term monitoring lakes.

3. Terrestrial Systems

Additional studies are needed to evaluate the impact of ambient fluoride on terrestrial ecosystems in GLAC. While adequate data exist on plant biomass content of fluoride in several plant species commonly found in GLAC, more detailed studies on physiological responses to ambient fluoride levels need to be developed. Plant injury symptoms and changes in growth and phenology need to be identified and documented through controlled chamber or greenhouse studies. If park managers want to accurately evaluate the impact of fluoride on vegetation, the following steps are recommended:

- Grasses, shrubs, and herbaceous species should be identified, analyzed, and reported separately instead of using the collective category "elk forage". This will provide more scientific and ecological value to the vegetation data.

- Controlled-fumigation studies should be included in the monitoring plan to elicit physiological response data on 4-5 tree, shrub and herbaceous species. Such studies should include measurements of changes in stem, root and leaf/needle growth, respiration, and foliar health (necrosis, chlorosis, tip burn) under a variety of exposure profiles. Exposures should include profiles of ambient, moderate, and high fluoride levels within a range of 20 - 1,000 ppbv, to reflect levels currently measured near Columbia Falls and the Flathead region.

- A complete photoguide of fluoride-induced injury in foliage can be compiled from symptoms obtained in the controlled fumigation studies. Consultation with a pathologist and/or entomologist is recommended to confirm that foliar injury is caused by fluoride, not by pathogens, nutrient deficiency, insects, fungal agents, or other factors.

- Controlled-fumigation studies should be considered for 4-5 common lichen species. Common foliose or species with large thallus surface area should be selected. Photographs of thallus injury can be compiled in a field-guide.

Currently, the threats to GLAC terrestrial resources from ozone and SO_2 are not sufficiently urgent to warrant monitoring of sensitive species. However, if pollution levels increase in the future, monitoring of terrestrial resources can be considered. Three levels of monitoring associated with increasing amounts of effort and expense are detailed in Appendix A. These monitoring activities are based on methods and protocols developed by the USDA Forest Service and NPS. Species and locations recommended for monitoring are listed below.

If monitoring for potential impacts of ozone and SO_2 is implemented, we recommend placing plots at three locations in the western and southwestern portion of GLAC, where airflow from Kalispell and Columbia Falls first reaches the park: (1) ponderosa pine in meadows west of headquarters and Apgar, (2) quaking aspen at lower to mid-elevation where reasonably continuous stands exist, and (3) subalpine fir at higher elevation.

Additional plots could be located along the Going-to-the-Sun Road from Apgar to Logan Pass. This valley is where maximum levels of ozone and other pollutants transported by prevailing winds might be expected. All sites should be accessible from the road. Locations of these plots can be changed to other sites in GLAC if ambient air quality data indicate that other areas have a higher risk of pollutant effects.

If lichens are included in a monitoring effort, they should be monitored on the south and west side of GLAC, as described in Appendix A. Three of the monitoring sites should be co-located with the tree monitoring plots in GLAC as described above. Additional lichen monitoring sites could be located in the alpine zone and along the elevational gradients described for tree monitoring above.

Additional tree plots could be located along elevational transects that evaluate other conifer species, such as Engelmann spruce, lodgepole pine, Douglas-fir, and whitebark pine. It is unknown how many of these species will be found along each transect, but it is recommended that at least two additional species be identified on each transect.

4. Visibility

IMPROVE aerosol and optical monitoring continues today at GLAC. Ongoing and future monitoring is necessary to identify local source impacts. Additional data and in-depth modeling and analysis are required to further evaluate historical trends and future projections of impact from existing and future sources. For example, back trajectory analysis and spatial/temporal pattern analysis of episodes are recommended to determine the source region contributions to elevated aerosol concentrations. Future research is also recommended to minimize the uncertainty in estimates of how various aerosol species affect visibility.

To enhance monitoring at GLAC, a monitoring site (particle and optical monitoring) could be established on the east side of the park. Dramatic differences in elevation can produce significantly different visibility conditions. Visibility monitoring sites at different elevations would therefore be desirable. This additional monitoring would help to better assess visibility impairment at the park.

VII. THEODORE ROOSEVELT NATIONAL PARK

...along the Little Missouri, it would be strange indeed, if any one found it otherwise than attractive in the bright, sharp fall weather...under the cloudless blue sky the air was fresh and cool...at night the stars shone with extraordinary brilliancy.

Theodore Roosevelt
January 20, 1915

A. DESCRIPTION

Theodore Roosevelt National Park (THRO) consists of three separate units (North, Elkhorn, and South) in western North Dakota and encompasses natural, scenic, and historical resources. The Little Missouri River winds through the North and South Units and forms the eastern boundary of the Elkhorn Unit.

Efforts to establish a park in the North Dakota badlands were initiated as early as 1917, although it was not until 1934 that the federal government began to acquire submarginal (for agriculture and grazing) lands for Roosevelt Regional Park; these lands were later designated as a Recreational Demonstration Area administrated by the National Park Service. Theodore Roosevelt Memorial NP was officially established in 1947 as a memorial to honor Theodore Roosevelt, and the park name was eventually changed to Theodore Roosevelt NP in 1978.

THRO is managed to protect and interpret the badlands ecosystems surrounding the Little Missouri River and the cultural resources resulting from human habitation of the area (THRO 1994). Maintenance and restoration of the natural environment, including physical and biological resources and ecosystem processes, is a critical management objective. Natural processes will be permitted to continue with a minimum of human disturbance. An additional objective is the protection and interpretation of human history, with emphasis on President Theodore Roosevelt. Maintaining natural areas and wilderness characteristics is potentially difficult due to development on and uses of adjacent private, state, and federal lands.

The three units of THRO comprise 26,578 ha. Although annual park visitation has been as high as 1 million, it is currently about 500,000, with 72% of visitation during June through August (THRO 1992). Approximately 42% of the park has been designated as wilderness.

1. Geology and Soils

The park falls within the Missouri Plateau and North Dakota Badlands sections of the Great Plains physiographic province. The badlands begin near the headwaters of the Little Missouri River in northeastern Wyoming and extend for 220 km, becoming deeply dissected eastward in the region of the park, and terminating where the river enters the Garrison Reservoir. The badlands consist of a complex of dissected canyons and coulees which have been eroded by the river and other streams over time. This has resulted in a variety of landforms, including buttes and ridges that support grassland vegetation. Rolling hills that extend eastward from the badlands also support grasslands, while drainages and riparian areas support a variety of shrubs, trees, and forbs.

A large portion of THRO consists of rugged badlands, comprised of Paleocene deposits that have been eroding since the Pliocene. Sandstone, siltstone, and clays are interspersed with beds of lignite in a complex array of stratigraphic patterns and colors that are an important visual resource for visitors. Some of the lignite strata have burned over time, baking the overlying clays into a bright pink to purple "scoria" that is a distinctive part of the landscape. Some areas of the badlands contain fossils created from Paleocene forests and swamps; fossils are revealed through continued erosion of surrounding geological strata.

Soils in the park developed from excessively drained, medium-textured, and calcareous parent material. Soil texture generally ranges from loams to clay loams. Saturated soils in this region tend to be highly erodible and can result in considerable slumping from the shoulder slope and backslope of existing landforms.

The Natural Resources Conservation Service completed a soil survey of THRO in 1994. Soils in the park are predominantly classified as torriorhents formed under prairie in a hot, dry climate. The major soils of the park are in the Badlands-Bainville Association. Some soils in THRO grade into haploborolls with deep soil profiles mainly confined to range sites on lower prairie slopes (e.g., the Morton soil series), but are generally quite localized.

The Havre soil series originates in alluvial bottoms, while the Patent soil series is derived from recently-deposited sediments on colluvial fans. Both of these series are fine-textured and often have a claypan and salt buildup. The more restricted Banks series occurs on the bottomlands along the Little Missouri River in recent alluvial deposits.

The Flasher series is comprised of coarse sandy soils on steep side slopes and crests of sandstone-capped ridges. The coarse gravel Parshall series is found on the high terrace remnants of the ancestral Little Missouri River such as the Petrified Forest Plateau. Both of these soils support prairie vegetation on gentle slopes. The widespread "badlands" component of the park is not classified as a soil by the Natural Resource Conservation Service although at least some vegetation is found on most slopes.

2. Climate

THRO has a continental climate characterized by cold winters and hot summers. Three major air masses influence climate in this area at various times of the year: 1) maritime pacific air from the west which has crossed the Rocky Mountains; 2) maritime tropical air from the south; and 3) continental polar air originating from Canada, Alaska and the arctic (ECOS Management Criteria, Inc. 1987). Each of the three air masses produces characteristic meteorological conditions. Maritime pacific air produces cool, stable, conditions in summer, and cold, dry conditions in winter. Maritime tropical air is warmer and more humid, is the major source of moisture, and is more prevalent during the summer months. Continental polar air is the cause of the extremely cold weather during the winter months; during the summer it results in cool, dry conditions.

Annual precipitation is 38 cm, most of which falls in the spring and summer, primarily during thunderstorms. Mean precipitation during June, the wettest month, is 8.8 cm. Average annual snowfall is 78 cm, with snow falling between October and May. During July and August, average maximum temperature is 30º C, and average minimum temperature is 16º C. Temperatures can reach as low as -18º C from October through April, with a record low of -44º C in 1950.

The prevailing wind direction in the vicinity of THRO is from the west to northwest, with a secondary maximum from the southeast (Bison Engineering/Research 1985). Wind rose data from the north and south units of THRO indicate that winds are frequently in excess of 15 mph. There are no well-defined air basins in this area. However, the complex valley-ridge terrain causes distinct local channeling of air flows, particularly during low wind speed conditions (U.S. Department of the Interior 1982). This effect is particularly pronounced during cold, stable atmospheric conditions characterized by temperature inversions.

3. Biota

Vegetation in THRO includes 574 species of vascular plants, most of which are adapted to a semiarid climate. At least 109 different species of bryophytes and 208 species of lichens have been identified in THRO as well. Grazing by domestic cattle and cultivation that occurred prior to creation of the park have altered the natural vegetation to some extent, and there are at least 57 species of exotic plants in the park.

Upland grasslands are found on deep, well-drained soils on moderate to shallow slopes dominated by wheat grasses (*Agropyron* spp.), needle grasses (*Stipa* spp.), blue grama (*Bouteloua gracilis*), smooth brome (*Bromus inermis*), little bluestem (*Andropogon scoparius*), fringed sage (*Artemisia frigida*), and sedges (*Carex* spp.). Dry breaks are found on areas of highly eroded silts or scoria surfaces and are dominated by sparse stands of little bluestem, blue grama, sideoats grama (*Bouteloua curtipendula*), red threeawn (*Aristida purpurea* var. *longiseta*), and scattered shrubs such as juniper (*Juniperus* spp.), saltbush (*Atriplex* spp.), and greasewood (*Sarcobatus vermiculatus*). Wooded draws are found in concavities in the landscape where soil moisture tends to be higher and where surface and subsurface water movement is greater; wooded draws are dominated by Rocky Mountain juniper (*Juniperus scopulorum*), green ash (*Fraxinus pennsylvanica*), and chokecherry (*Prunus virginiana*). The understory is dominated by snowberry (*Symphoricarpos* spp.), skunkbush sumac (*Rhus trilobata*) and a variety of graminoids, mosses, and lichens. Sagebrush and grassland bottoms are formed by alluvial deposits from the Little Missouri River and its larger tributaries, and comprise the higher floodplains and river terraces. They are dominated by silver sagebrush (*Artemisia cana*), western wheatgrass species, needle and thread (*Stipa comata*), and blue grama species, with fringed sage species, prairie rose species, and snowberry as additional woody components. Floodplain forests are found on the lowest terrace along perennial streams. They are dominated by plains cottonwood (*Populus deltoides*), with subdominants Rocky Mountain juniper, green ash, chokecherry, wildrye (*Elymus* spp.), wheat grasses, and sedges. Grasses and forbs may replace the woody understory in some locations. Riparian vegetation is associated with a narrow band between floodplain forest and a perennial stream. The dominant vegetation is normally various willow species (*Salix* spp.), with wildrye, prairie cordgrass (*Spartina pectinata*), and rushes in the understory. An alternative approach to classifying vegetation at THRO uses physiographic/ vegetation classes based on a combination of landform and gross structure of the vegetation (THRO 1994) (Table VII-1).

Over 250 species of vertebrate fauna are found in THRO. Large mammals include white-tailed deer (*Odocoileus virginianus*), mule deer (*O. hemionus*), bison (*Bison bison*), elk (*Cervus elaphus*), pronghorn (*Antilocapra americana*), and a few bighorn sheep (*Ovis canadensis*). Mammalian predators include coyote (*Canis latrans*), bobcat (*Lynx rufus*), red fox (*Vulpes vulpes*), and weasel (*Mustela* spp.). There are many species of small mammals including small rodents, striped skunk (*Mephitis mephitis*), beaver (*Castor canadensis*), porcupine (*Erethizon dorsatum*), and prairie dogs (*Cynomys* spp.).

Many raptors nest in the park including golden eagle (*Aquila chrysaetos*), prairie falcon (*Falco mexicanus*), and kestrel (*Falco sparverius*). Additional large nesting birds include turkey vulture (*Cathartes aura*), great horned owl (*Bubo virginianus*), screech owl (*Otus kennecotti*), burrowing owl (*Athene cunicularia*), red-tailed hawk (*Buteo jamaicensis*), rough-legged hawk (*Buteo lagopus*), northern harrier (*Circus cyaneus*), and wild turkey (*Meleagris galloparo*). Many species of passerine birds are also found in the park.

There is little information on the distribution and abundance of reptile and amphibian species in the park. Reptiles that are likely residents include western plains garter snake (*Thamnophis radix*), plains hog-nosed snake (*Hetecodon nasicus*), yellow-bellied racer (*Coluber constrictor*), bullsnake (*Pituophis melanoleucus sayi*), prairie rattlesnake (*Crotalus viridis*), and short-horned lizard (*Phrynosoma douglassii*). Amphibians that may be residents are the Great Plains toad (*Bufo cognatus*) and Rocky Mountain toad (*B. woodhousii*).

4. Aquatic Resources

The major surface water resource in THRO is the Little Missouri River. The river flows through 14 km of the South Unit and 21 km of the North Unit, and forms the eastern boundary of the Elkhorn

Table VII-1. Vegetation classes in THRO.	
Class	Description
Breaks	Areas mostly devoid of vegetation
Cottonwood forests	Forests along perennial streams dominated by plains cottonwood
Wooded draws	Dominated by either green ash or quaking aspen (*Populus tremuloides*)
Upland grasslands	Level to rolling grasslands found on the plains above the Little Missouri River valley
Old river terraces	Level grassland 50 to 150 m above the Little Missouri River
Grassland flats	Large, flat grassy alluvial deposits found 30 to 60 meters above the Little Missouri River
Bottom grasslands	Large, flat grassy alluvial deposits found on the higher floodplain of the Little Missouri River and its larger tributaries
Toe slopes	Gradually sloping lands formed by slumping and alluvial deposition; covered with grass, shrubs, and trees,
Rolling grasslands	Level to rolling grasslands in the North Unit found on the glaciated plains above the Little Missouri River
Achenbach hills	Hills found 200 meters above the Little Missouri River in the North Unit which are covered by grass and shrubs
Ridge and ravine	Lands highly dissected by drainages and covered by grasses, shrubs, and trees
Scoria hills	Lands influenced by scoria which have differential weathering, producing rugged and varied topography with a variety of grasses and shrubs
Sagebrush bottoms	Floodplains dominated by silver sagebrush along with substantial grass cover
Prairie dog towns	Lands formerly or currently influenced by prairie dogs (*Cynomys* spp.), with sparse vegetation near the center
Juniper woodlands	Cool, moist sites generally found on north-facing slopes dominated by Rocky Mountain juniper

Unit. The river is wild and free-flowing. Most other streams within the park are intermittent with little or no flow during the summer.

The Little Missouri River generally has high turbidity, high mineral content, wide fluctuations in flow rate, and a shifting bottom of various textures. It contains aquatic communities that are relatively species-poor, although it does support populations of several gamefish, including northern pike (*Esox lucius*), walleye (*Stizostedion vitreum*), sauger (*S. canadense*) and channel catfish

(*Ictalurus punctatus*). There are approximately 25 species of fish in the river that are found within the boundary of the park (THRO 1994). The silvery minnow (*Hybognathus nuchalis*) and the plains minnow (*Hybognathus placitus*) represent about 80% of the number of fish in the river as a whole. There is almost no information on invertebrate species in the river and its tributaries.

There are 10 developed springs and 15 flowing wells in the park. Limited data have been collected on flow rate and chemical characteristics of these hydrological resources. Many other springs and seeps are not inventoried or developed. There is little information on groundwater characteristics and variation in the water table. The U.S. Geological Survey is currently developing a profile of groundwater quantities for the park. There is concern that energy development adjacent to the park could affect water quality in the park through contamination of groundwater aquifers and streams by well blowouts, spills, or leakage of petroleum or saltwater.

B. EMISSIONS

Air quality within the park has generally been regarded as good, although there are concerns about current and future energy development adjacent to the park. The human population is sparse near THRO and there are no large metropolitan areas. The major sources of pollution are associated with oil and gas production as well as coal-fired electricity generation, which adds S and particulates to the atmosphere. There are approximately 20 major point sources of gaseous sulfur pollution in the vicinity of the park (Roger Andrascik, pers. comm.).

The greatest air quality impact to the park comes from oil and gas wells in the vicinity. Over 1,500 producing oil and gas wells have been drilled in Billings and McKenzie Counties, the two counties in which the park is located (Figures VII-1 and VII-2). There are 630 new wells proposed near the park in the next 10 years. Large areas around the park have been proposed to be managed for high mineral potential so it is likely that the levels of oil and gas production from this region will increase or at least remain at current levels.

An inventory of SO_2 emissions near THRO was conducted by the North Dakota State Department of Health (NDSDH) for the National Park Service. The inventory was conducted in 1981 and 1982 and included only point sources within approximately 50 km of any of the three units of the park. Of the total SO_2 emissions for the area (33,500 tons per year), 22,000 tons per year were attributed to oil and gas wells while the remaining 11,500 tons per year were emitted by larger point sources (North Dakota State Department of Health 1983).

In 1987, there were 16 sources within 250 km of THRO that emitted more than 250 tons per year of SO_2 (Weber and White 1990). The largest source is the Coal Creek power plant which emits 50,000 tons per year of SO_2. This power plant, along with the next four largest emitters, are located to the east of THRO.

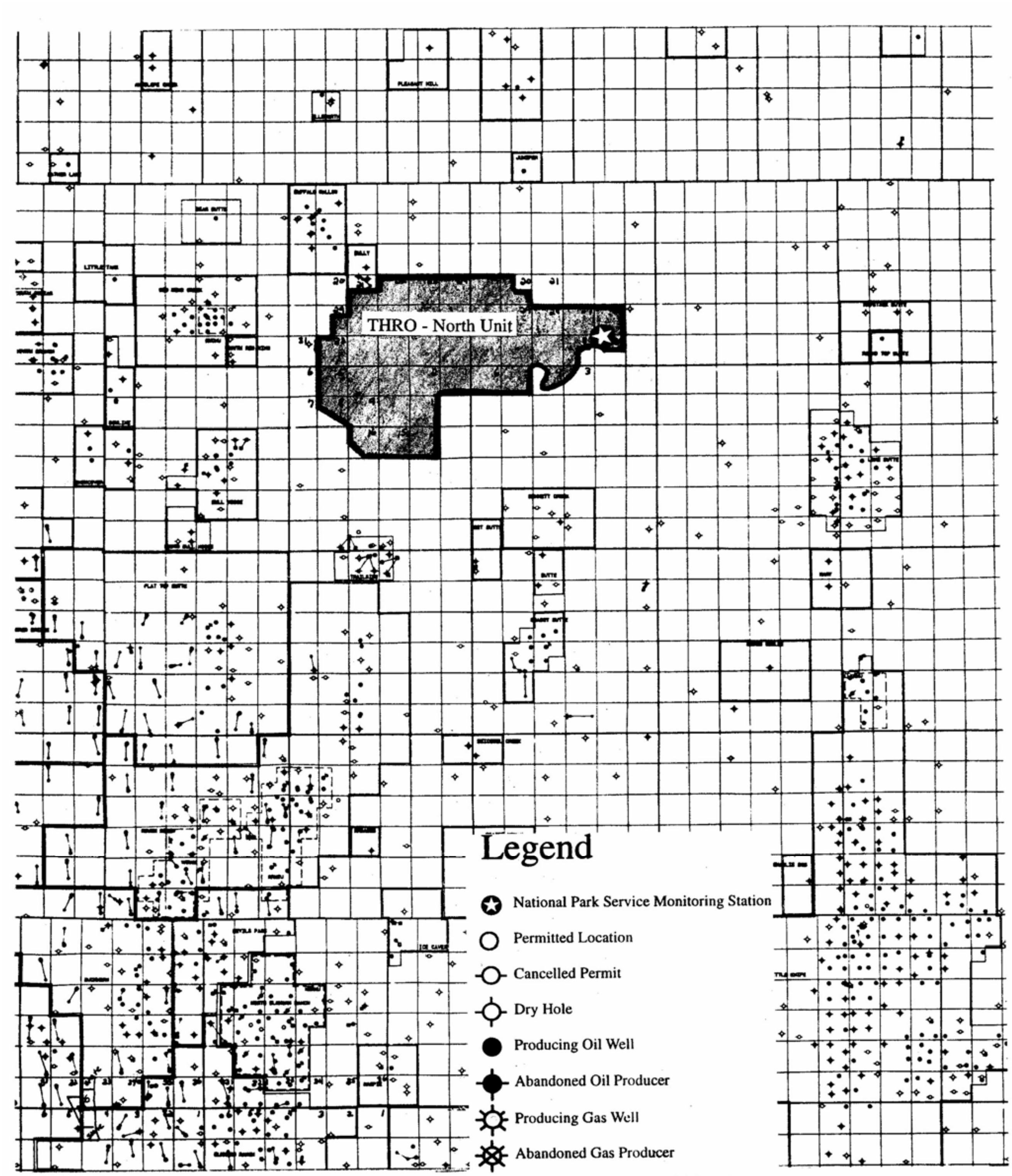
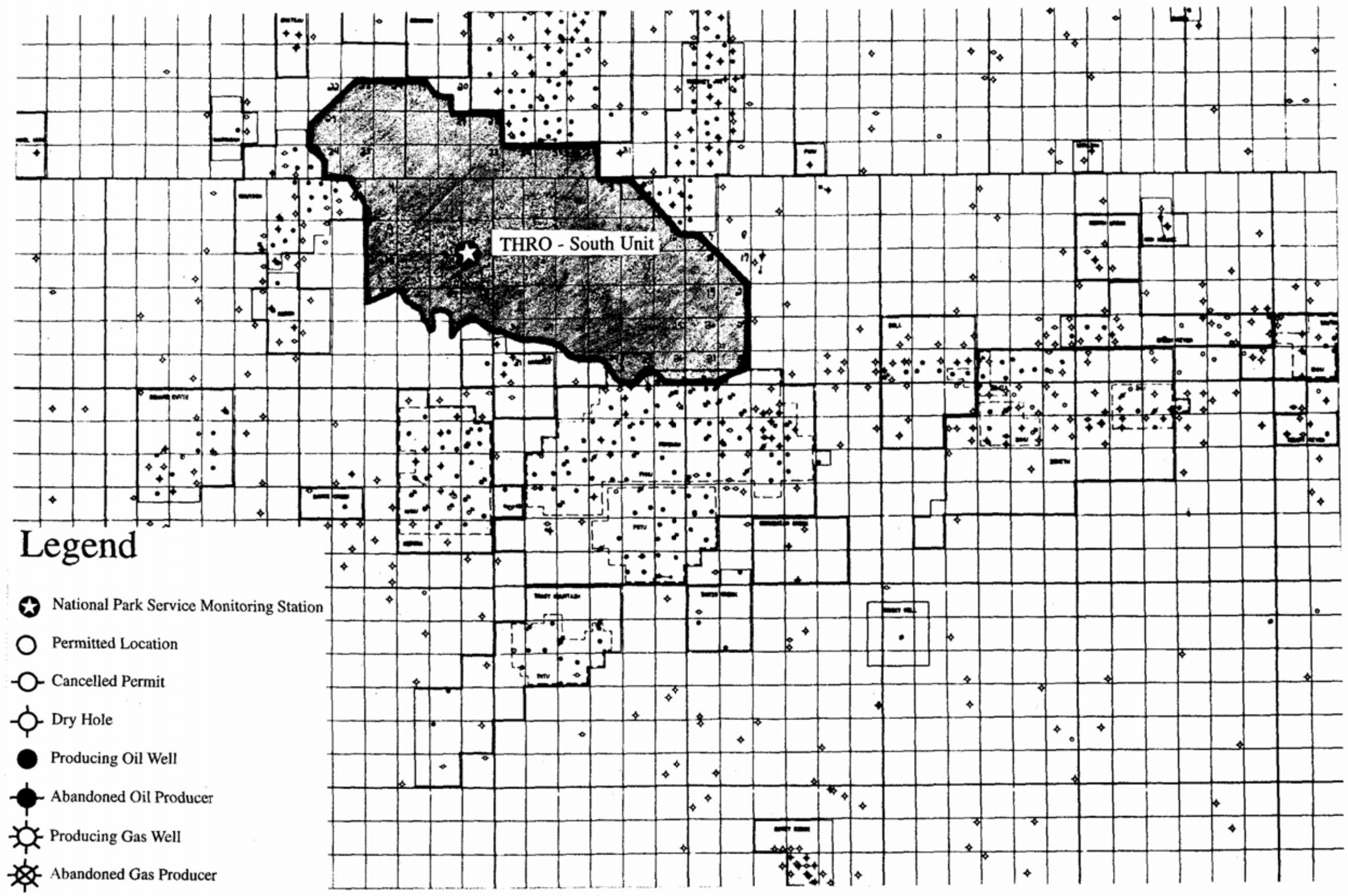


Figure VII-1. Oil and gas well locations in the vicinity of the North Unit. Modified from N.D.I.C. Oil and Gas Division. Scale: 1 square = 1 mile^2.


THRO - South Unit
Legend
National Park Service Monitoring Station
Permitted Location
Cancelled Permit
Dry Hole
Producing Oil Well
Abandoned Oil Producer
Producing Gas Well
Abandoned Gas Producer

THRO is unique among Class I air quality areas of the Rocky Mountains and the northern Great Plains because hydrogen sulfide (H_2S) emissions in its vicinity are a significant concern. H_2S is a highly toxic gas that is formed by the decomposition of organic matter containing S. It is found in mines, wells, sewers, cesspools, natural gas, volcanic gas and in some spring waters. It is a colorless, flammable gas with a density greater than that of air. Air pollution by H_2S is not a widespread problem but rather is typically localized near sources such as oil and gas wells, kraft paper mills, sewage treatment plants, petroleum refineries and coke ovens.

The oil fields of western North Dakota were developed primarily for the production of crude oil but many have also been developed with gas-gathering networks by which natural gas, which is produced in association with crude oil, can be transported to gas processing plants (Weber and White 1990). In areas not equipped with a gas-gathering pipeline system, the gas produced from a well is used to operate lease equipment and the excess is flared. The content of H_2S in raw gas can range from 0 to as high as 30% (NDSDH 1983).

The flaring of the gas converts the H_2S to SO_2. Reports of the efficiencies of conversion vary greatly, from 30% to 100%. H_2S is also emitted by fugitive releases of raw gas from leaky valves, pipe connections or tank hatches. Therefore, the amount of H_2S and SO_2 emitted from any particular well depends upon many factors including production rate, percentage of H_2S in the raw gas, the presence or absence of gas-gathering lines, flaring efficiency, and amount of raw gas fugitive releases. It should be noted that altering the flaring efficiency can vary the amount of either H_2S or SO_2 emitted but can never reduce the total amount of sulfur that is emitted.

From 1982 to 1984, the NPS reviewed seven PSD permit applications for energy conversion and natural gas sweetening facilities within a 200-km radius of the park. Emissions modeling predicted that some Class I air quality increments within the park would be exceeded, although NAAQS at the source areas would not be violated. However, because it could not be demonstrated that vegetation or visibility would be significantly affected by the sources, the Department of the Interior issued a certification of no adverse impact on the park, and the state granted construction permits for six of the plants. Proposals for coal-fired power plants in eastern Wyoming and a gas sweetening facility (Enron Gas Processing Company) in Rawson, ND, northwest of the park are a source of ongoing concern with respect to future air quality in the park (THRO 1992, 1994).

Installation of a new well is not subject to the PSD regulatory process. However, the additive effect of many of these small sources may be greater than that of a single, larger source. These smaller sources contribute to regional pollution problems and should not be considered insignificant because they are not categorized as “major”. In addition, a small source such as an oil well has the potential to cause high concentrations of pollutants in its immediate vicinity. Under certain meteorological conditions the plume of pollutants from a small source has the potential to impact sensitive plant species in the park. Therefore, it is important to consider small sources such as oil and gas wells as contributors to both regional and localized air quality problems.

In the last decade, a decrease in production has occurred in the oil and gas industry in North Dakota due to depressed market conditions. This decrease in production, along with the inclusion of gas gathering lines, has significantly decreased the emissions of SO_2 and NO_x. In addition, several large point sources of SO_2 have ceased operation. The decrease in emissions has presumably reduced the impact on the park as well as decreased North Dakota's contribution to regional haze. Future exploration and development during improved economic market conditions still may threaten air quality.

Historically, fires, and therefore smoke, have been a part of the Great Plains ecosystem. Fires are generally not considered to be a significant long term source of pollutants but they can result in episodes of degraded visibility and high particulate matter concentrations.

North Dakota has State Ambient Air Quality Standards that are more restrictive than NAAQS, including H_2S standards and a 1-hour standard for SO_2. The North Dakota state 1-hour SO_2 standard is 273 ppbv. The 24-hour SO_2 standard is 99 ppbv and the annual standard is 23 ppbv.

Hydrogen sulfide standards were initially set at 32 ppbv maximum half-hour concentration not to be exceeded more than twice in any five consecutive days and 54 ppbv maximum half-hour concentration not to be exceeded more than twice per year. In 1987, a single H_2S half-hour standard of 50 ppbv was promulgated. The standard is set primarily to control odors caused by H_2S and is not designed to protect human health or natural resources (Weber and White 1990).

North Dakota has adopted air quality rules that are specific to the oil and gas industry (Weber and White 1990). These rules are found in Chapter 33-15-20 "Control of Emissions from Oil and Gas Production Facilities" of the North Dakota Air Pollution Control Rules. The rules require the owner or operator of a well installed after July 1, 1987, or any well existing prior to this date that would emit 10 tons per year or more of any sulfur compound, to register the well with the NDSDH. Although not specifically a permitting process, the NDSDH can use the registration information to determine whether the oil production facility should comply with state air quality rules, including applicable ambient air quality standards and PSD increments. As of 1990, approximately 700 wells had been registered with the NDSDH. The rules also require owner/operators of oil and gas production facilities to flare waste gas that contains H_2S or treat it in an equally effective manner prior to releasing it to the atmosphere. If a flare system is used, the flare must be equipped with an automatic ignitor or a continuous burning pilot.

A local emission source at THRO is a vehicle pull-out near the Painted Canyon area, which is a popular rest stop for diesel trucks. Drivers may stop for several hours or overnight with their engines running. There is some concern from park resource managers regarding potential impacts on air quality in the Painted Canyon area. Generalized emission rates for idling diesel trucks are estimated to be 50 g/hour of NO_x, 92 g/hour of CO and 12 g/hour of VOCs. The negative effects of CO would be primarily on human health, particularly in the early morning hours when trucks have been running all night and atmospheric mixing is low. NO_x and VOCs are also a concern because they are the precursors of ozone and therefore would be of greatest concern during the summer months. Further monitoring would be necessary to determine the effect of trucks in this area.

C. MONITORING AND RESEARCH ACTIVITIES

1. Air Quality/Deposition

a. Wet Deposition

The concentrations of SO_4^{2-}, NO_3^-, and NH_4^+ in precipitation at THRO are moderate (Table VII-2). Sulfate and NH_4^+ concentrations are actually somewhat higher than at the two wet deposition monitoring stations in Rocky Mountain National Park, and the NO_3^- concentrations at THRO tend to be intermediate between the two Rocky Mountain NP sites. The volume of precipitation at THRO is low, however, generally in the range of 25 to 45 cm annually. As a consequence, wet deposition of S and N at the park is low. Wet S deposition is generally in the range of 1.0 to 1.4 kg/ha/yr, and wet N deposition is 1.0 to 2.0 kg/ha/yr, with somewhat more than half generally as NH_4^+ (Table VII-3).

Table VII-2. Wetfall chemistry at the NADP/NTN site at THRO. Units are in μeq/L, except precipitation (cm) and H^+.

Year	Precip	H^+	SO_4^{2-}	NH_4^+	NO_3^-	Ca^{2+}	Mg^{2+}	Na^+	K^+	Cl^-
1995	48.3	5.1	16.7	21.6	15	9.4	2.6	2.5	0.7	1.7
1994	44.2	8.4	20.1	18.9	16.4	11.1	3.1	2.3	0.7	1.8
1993	46.2	4.6	17.3	12.6	12.1	6.9	2.1	4.8	0.5	1.8
1992	31.2	6	18.4	15.6	13.3	9.1	2.5	3	0.4	1.9
1991	35.8	5.7	16.8	15.8	14.6	9.5	2.5	2.2	0.5	1.9
1990	30.8	5.3	23.9	20.7	14.2	11.5	3.8	2.7	0.7	2.5
1989	35.4	2.7	17.9	19.8	13.7	9.3	2.6	2.5	0.5	2.2
1988	22.8	6.3	21.1	12.9	13.4	11.6	3	5.3	0.7	2.6
1987	5	2.4	17	5.5	10.6	11.9	3.8	6.9	0.7	2.2
1986	53.2	3.9	18.3	11.5	9.5	9.8	3	2.2	0.5	1.5
1985	32.4	6.6	23.9	13.3	12.9	14.2	4.5	2.9	0.8	2.2
1984	22.7	5.7	37	14.9	17.2	29.9	12.1	8.3	2.2	5.2
1983	26.2	6.5	23.3	12.9	15.5	15.5	5	3.6	0.9	2.8
1982	54.3	6.1	20.6	9.9	12.2	14.6	5.6	2.7	1.6	2.2
1981	26.4	4.1	31.8	19.5	17.5	20.4	9.1	5.4	1	3.4
Average	34.3	5.3	21.6	15.0	13.9	13.0	4.4	3.8	0.8	2.4

Table VII-3. Wet deposition (kg/ha/yr) of sulfur and nitrogen at the NADP/NTN site at THRO.				
Date	Sulfur	NO_3-N	NH_4-N	Total Inorganic N
1995	1.3	1.0	1.5	2.5
1994	1.4	1.0	1.2	2.2
1993	1.3	0.8	0.8	1.6
1992	0.9	0.6	0.7	1.3
1991	1.0	0.7	0.8	1.5
1990	1.2	0.6	0.9	1.5
1989	1.0	0.7	1.0	1.7
1988	0.8	0.4	0.4	0.8
1987	0.1	0.1	0.0	0.1
1986	1.6	0.7	0.9	1.6
1985	1.2	0.6	0.6	1.2
1984	1.3	0.5	0.5	1.0
1983	1.0	0.6	0.5	1.0
1982	1.8	0.9	0.8	1.7
1981	1.3	0.6	0.7	1.4
Average	1.2	0.7	0.8	1.4

b. Occult/Dry Deposition

There are no dry deposition data available in the vicinity of THRO.

c. Gaseous Monitoring

Monitoring by the NDSDH indicates that air quality is generally good in the vicinity of THRO. Currently, gases are monitored only at the North Unit. Monitoring equipment at the South Unit was removed because ambient levels of SO_2 and H_2S rarely exceeded the minimum detection levels of the equipment. In general, the complex topography of the park makes it difficult to monitor gaseous pollutants; for example, with respect to sulfur, H_2S tends to settle while SO_2 tends to remain aloft. Currently, ozone, SO_2, and H_2S are the only gaseous pollutants monitored in THRO (Table II-5). Historical gaseous monitoring activities are summarized in Table VII-4.

Ozone has been monitored at THRO since 1983. Table VII-5 is a summary of maximum 1-hour ozone concentrations at THRO. The highest ozone concentration measured at the park was 80 ppbv (1983). Ozone concentrations are similar to those found at GLAC, BADL, and YELL and significantly lower than those at ROMO. Perhaps more importantly, with respect to damage to sensitive plant species, the mean daytime 7-hour ozone concentration during the growing season ranged between 37 and 51 ppbv during this time period. The SUM60 exposure index is another indicator that can be important in assessing ozone exposures of plant species. This index is the

Table VII-4. Gaseous monitoring data collected at the North (N) and South (S) Units of THRO. (Source: NDSDH 1985-1995)

	SO_2	Ozone	H_2S
1980	S,N		
1981	S,N		
1982	S,N		
1983	S,N	N	
1984	S,N	N	
1985	S,N	N	
1986	S,N	N	
1987	S,N	N	S,N
1988	S,N	N	S,N
1989	S,N	N	S,N
1990	S,N	N	S,N
1991	N	N	N
1992	N	N	N
1993	N	N	N
1994	N	N	N
1995	N	N	N
1996	N	N	N

sum of all hourly ozone concentrations equaling or exceeding 60 ppbv. The SUM60 index at THRO ranged between 121 and 12,488. For comparison, National Parks in highly polluted areas (e.g., southern California) can have SUM60 exposure indexes exceeding 100,000 ppbv-hour (Joseph and Flores 1993).

The primary air quality concern at THRO is sulfur exposure (in the forms of SO_2 and H_2S). Table VII-6 is a summary of these two pollutants as measured at the North and South units. The state 1-hour SO_2 standard is 273 ppbv. The highest concentration was about 18% of the standard.

“Pristine” air is expected to contain about 0.19 ppbv SO_2 (Urone 1976), and some grasses have been found to have biomass reductions when grown in atmospheres of 17 to 29 ppbv throughout the growing season (Crittendon and Read 1979).

The state H_2S standard is 50 ppbv (1-hour average). The highest concentration measured at the North Unit in 1995 was 66% of the standard. Although the H_2S standard has not been exceeded it should be noted that these standards were set in order to control odors. THRO vegetation may be potentially impacted at pollutant concentrations lower than the standards.

The current and previous monitoring stations for SO_2 and H_2S are located at the headquarters in each unit. These locations are relatively distant from most of the oil and gas fields (see Figures II-1 and VII-2). Several oil and gas wells are located close to the park border but relatively far from the monitoring station, thus the potential exists for high sulfur concentrations immediately within the

Table VII-5. Summary of THRO ozone concentrations (ppbv) from NPS monitoring sites (Joseph and Flores 1993, NPS unpublished monitoring data).

	1983[a]	1984	1985	1986[b]	1987[c]	1988[d]	1989[d]	1990[e]	1991[d]	1992	1993	1994
1-hour maximum	80	68	61	62	64	78	73	70	70	63	64	79
Average daily mean	28	29	31	32	32	42[1]	39	35	36	33	31	34
Growing season 7-hour mean	51	40	39	38	37	50	46	42	44	40	37	42
SUM60 exposure index (ppbv-hour)	4,496	2,766	121	365	186	12,488	10,983	3,099	3,038	791	685	1,123

[a] July, August, and September data not collected
[b] October, November, and December data not collected
[c] Data collected April through October only
[d] Data collected April through September only
[e] Data collected May through September only

Table VII-6. SO_2 and H_2S concentrations at THRO from 1987 to 1995. The state 1-hour SO_2 standard is 273 ppbv. The state H_2S standard is 50 ppbv (1-hour average) and one exceedence per year is allowed. (Source: North Dakota State Department of Health 1985-1995)

	Year	SO_2		H_2S	
		Maximum 1-hour value (ppbv)	Average value (ppbv)	Maximum 1-hour value (ppbv)	Average value (ppbv)
North Unit	1989	35.0	1.3	10.0	1.0
	1990	21.0	1.2	10.0	1.0
	1991	40.0	1.2	32.0	1.0
	1992	20.0	1.2	9.0	1.0
	1993	33.0	1.2	30.0	1.1
	1994	49.0	1.3	29.0	1.1
	1995	23.0	1.2	33.0	1.1
South Unit	1987			4.0	1.0
	1988			9.0	1.0
	1989			10.0	1.0
	1990			1.0	1.0

park boundary to remain undetected by the monitoring equipment. H_2S is not a regional pollutant like SO_2. High concentrations of H_2S tend to be found only in localized areas near the source while at distant monitoring sites no detectable levels are registered. A H_2S monitoring study of an oil field near the North Unit registered H_2S concentrations as high as 13,000 ppbv with equipment placed 0.4 km from the nearest well (Bicknell 1984). Clearly, H_2S emissions from these sources have the potential to impact sensitive plant species and perhaps wildlife at THRO. (The “rotten egg” smell of H_2S was detected during September 1996 adjacent to the park boundary suggesting that levels were relatively high [Peterson and Brace, personal observation]).

Concerns such as these prompted NPS to request that NDSDH monitor SO_2 and H_2S in the South Unit of the park close to the nearby Whiskey Joe oil field. This oil field appears as the large cluster of wells immediately adjacent and to the northeast of the South Unit in Figure VII-2. Because a suitable site was not found in the South Unit due to lack of a power source, poor winter accessibility and visibility concerns by the public, the site was established within the oil field (approximately 2 miles from the park). Monitoring began in July 1995 and was terminated in December 1997. Wind speed and direction were also measured.

Data from this special purpose monitoring site are presented in Table VII-7. The maximum SO_2 levels at this monitoring site were moderate and were sometimes higher and sometimes lower than those measured at the North Unit. For both years for which data are available, SO_2 levels were above the minimum detectable value of the instrument (2 ppbv) approximately 20% of the time (compared with about 10% of the time for the North Unit). This was not unexpected because SO_2 is

Table VII-7. SO_2 and H_2S concentrations (ppbv) from the special purpose monitoring site in the Whiskey Joe oil field near the South Unit of THRO.					
Sampling Period	Maximum 1-hour	2nd Highest 1-hour	Maximum 24-hour	2nd Highest 24-hour	Hourly Arithmetic Average
SO_2					
1995 Jul-Dec	28	26	9	7	1.7
1996 Jan-Dec	26	25	7	6	1.5
H_2S					
1995 Jul-Dec	192	187	45	35	5.2
1996 Jan-Dec	300	295	54	49	11.4

dispersed widely in the atmosphere. Hydrogen sulfide concentrations were consistently higher than at the North Unit and this was also not unexpected because the monitor was located inside of the oil field. In 1995, a recorded value of 192 at the Whiskey Joe site almost exceeded the state 1-hour standard of 200 ppbv and during 1996 it exceeded the standard 18 times. During both years, the Whiskey Joe site recorded the highest 24-hour concentrations and the highest hourly arithmetic average of all the NDSDH sites.

Background levels for H_2S for clean air are estimated to be around 0.2 ppbv (Smith 1990). Monitoring data for a number of pollutants, including H_2S, indicate that the air quality of the North Unit is generally more degraded than that of the South Unit (Bilderback 1990). However, this special purpose monitoring site near the South Unit clearly indicates that air quality can be heavily impacted in the immediate vicinity of oil and gas wells. This information, coupled with the fact that there are large numbers of oil and gas wells in the vicinity of all three units of the park indicate that these sources are having a measurable impact on air quality, especially near the boundaries of the park (Figures VII-1 and VII-2). Although it is unclear to what degree the resources of THRO are being impacted by current concentrations, air quality has been altered by local SO_2 and H_2S emissions.

Since June of 1980, park staff have collected data on incidents of smoke and H_2S odor, as observed from within park boundaries. These data include wind speed, wind direction, time of observation, observation point, and apparent source if known. Incidences of visibility impairment by smoke are also reported. The results are on file at park headquarters in Medora.

2. Water Quality

Water quality data are available from 76 locations in and around THRO (Figure VII-3). Sampling site locations were situated along the Little Missouri River and many of its tributaries (Figure VII-4). Measured pH values were around 8.0 at all monitoring sites except one, which had a very low value of 3.1 (Figure VII-5a). The low pH site, at the Frank Mine, also had an extremely high specific conductance (2560 μS/cm). This site is apparently impacted by acid mine drainage. None

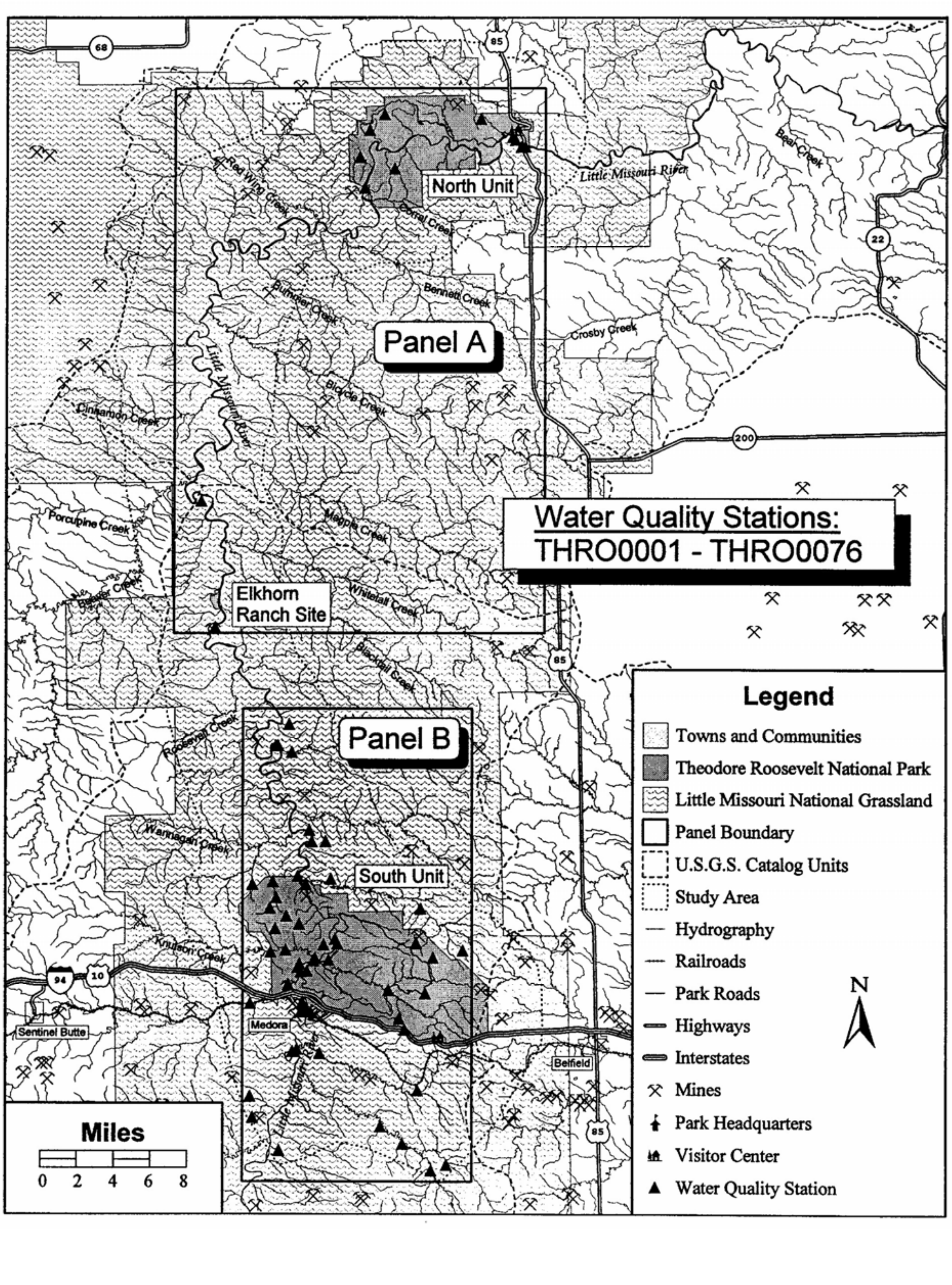


Figure VII-3. Water quality monitoring locations in THRO.

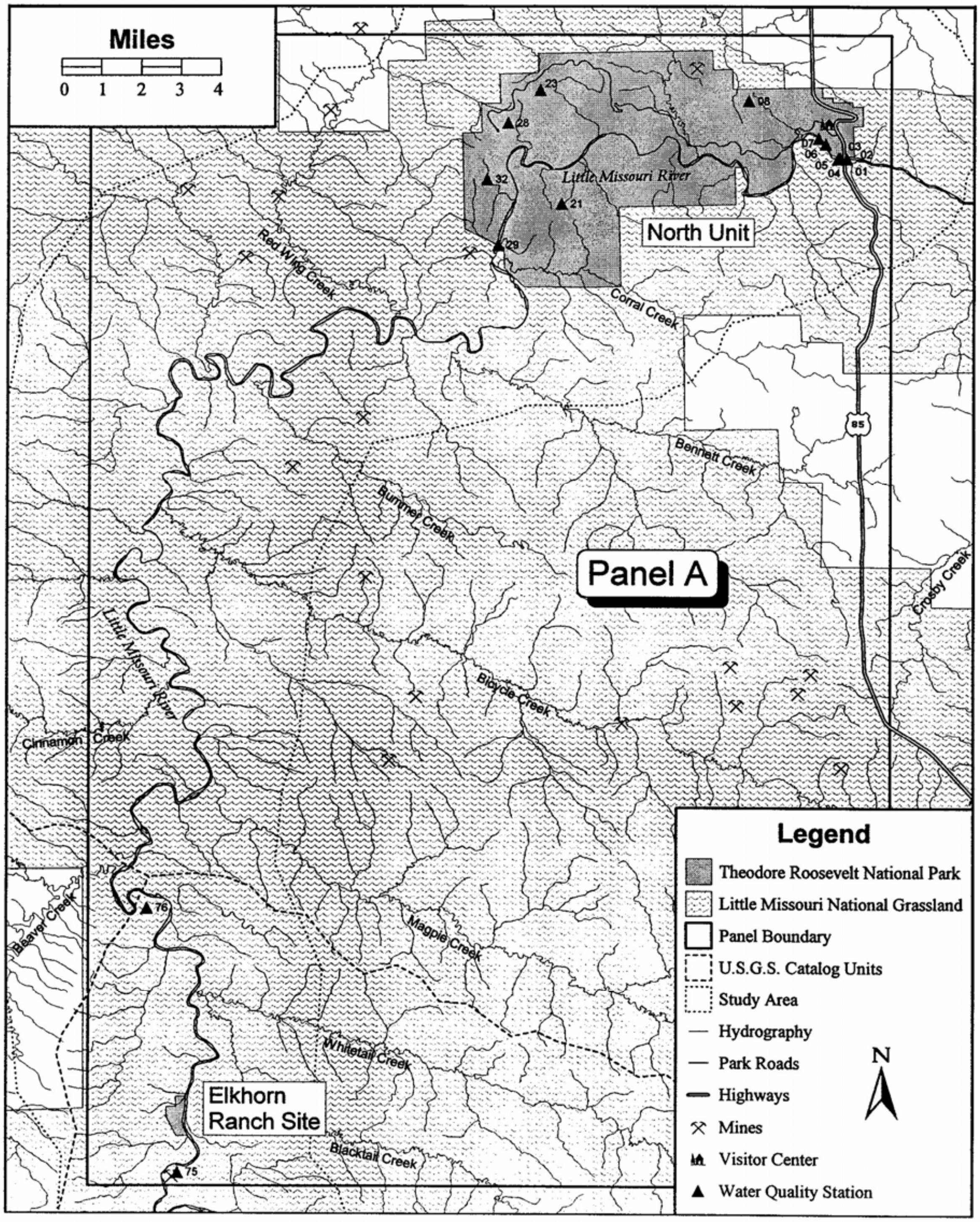


Figure VII-4. Water quality monitoring site identification numbers in the North Unit (Panel A) and the South Unit (Panel B).

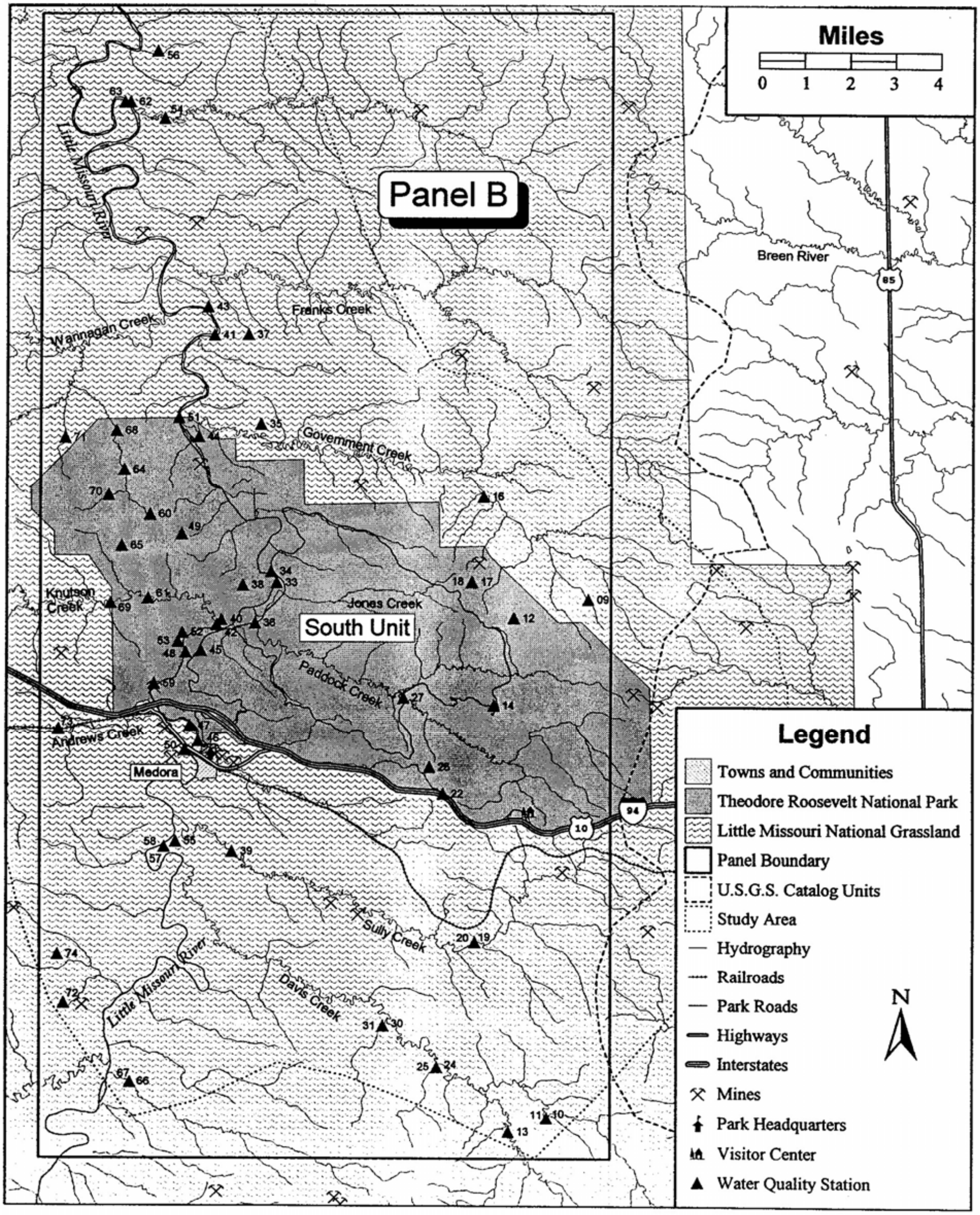


Figure VII-4. Continued.

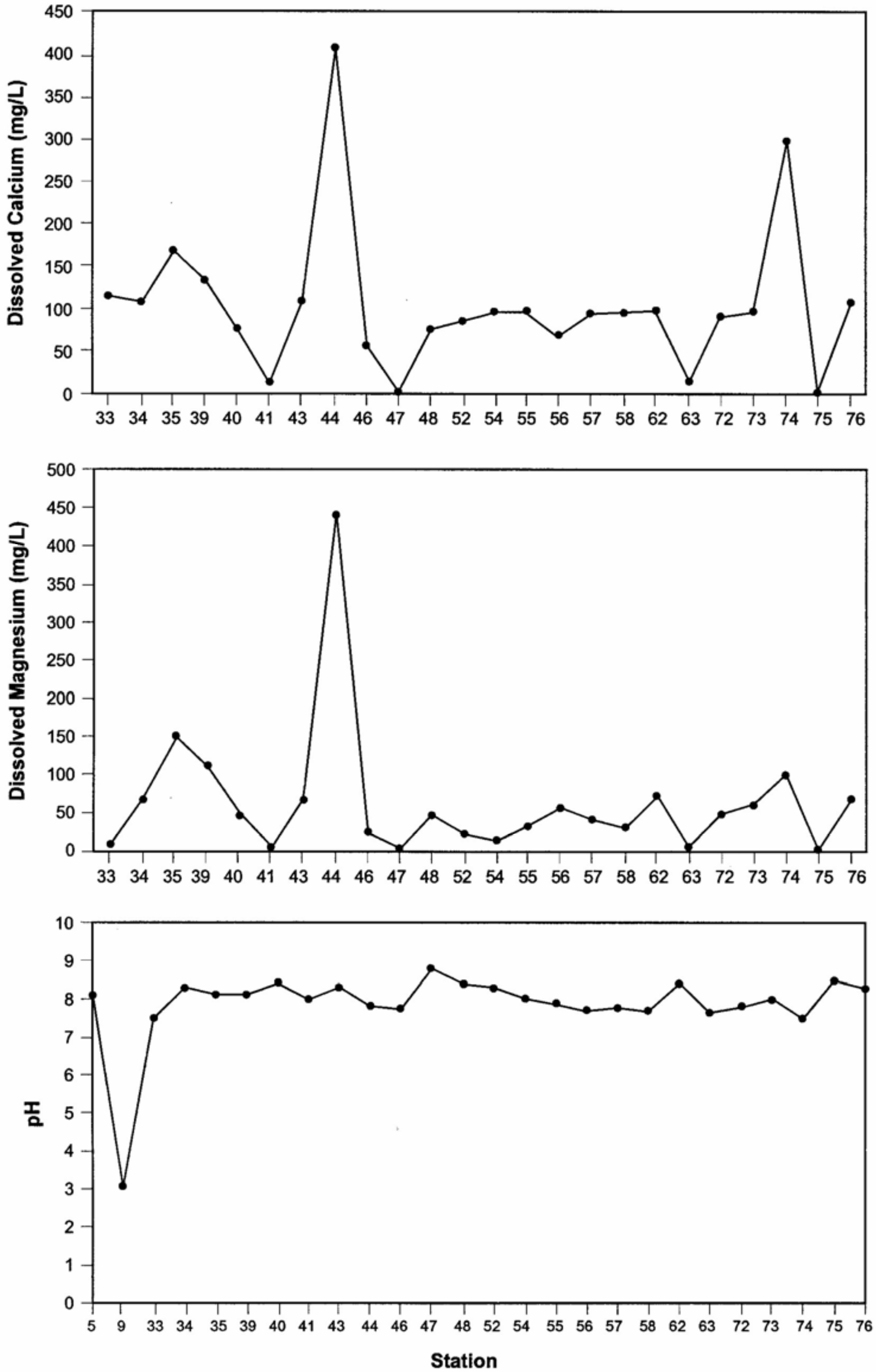


Figure VII-5. Measured Ca^{2+}, Mg^{2+}, and pH values of water quality monitoring sites for which data are available. Station numbers correspond to site locations shown on the preceding figure. For stations having more than one data point, mean values are shown.

of the data at any of the sites reflected sensitivity to acid deposition. There were two sites with moderate concentrations of base cations (Ca^{2+} + Mg^{2+} in the range of 250 to 300 μeq/L); all other sites had base cation concentrations at least twice as high as these (Figures VII-5b, VII-5c). Existing water quality data for THRO do not suggest sensitivity to potential acidification from acidic deposition. These data are in agreement with expectations based on the physical setting and geology of the park. Aquatic effects from acidic deposition are not currently an important issue in this park.

3. Terrestrial

There have been a number of studies done on the lichens and bryophytes of THRO in relation to air pollution effects (Bilderback 1987, Egan 1984, Gough et al. 1985, Wetmore 1983). There have also been a few studies on vascular vegetation (Gough et al. 1985, Bilderback 1990). None of the studies could conclusively demonstrate that the terrestrial resources of THRO were being significantly impacted by air pollution. However, they do provide valuable baseline information on the status of the terrestrial resources of THRO.

In 1983 a study by Wetmore (1983) established a lichen flora and provided information on distribution. An extensive collection found 208 species, including numerous rare boreal lichens and many significant range extensions. There was no indication that the air quality in THRO was affecting the lichens, because the floristic analysis shows a diverse flora without distribution void. In addition, chemical analyses were performed on seven lichen species: *Parmelia chlorochroa*, *P. flaventior*, *P. mexicana*, *P. soredica*, *P. sulcata*, *Rhizoplaca chrysoleuca*, and *R. melanophthalma*. The S levels in the lichens ranged from 480 to 1426 ppm (based on dry weight of thallus). Levels may be as low as 200-300 ppm in the arctic (clean area) and as high as 4,300-5,200 ppm in polluted areas. The levels found in THRO are considered to be close to background. Although different lichen species typically accumulate sulfur at different rates, there was no discernible difference between these seven species. There was also no discernible pattern of lichen S content distribution in the park. The multi-element analyses also showed no abnormal levels for the other elements that were analyzed, namely Ca, Mg, Na, K, P, Fe, Mn, Al, Cu, Zn, Cd, Cr, Ni, Pb, and B.

A report entitled “The ecological role of lichens in Theodore Roosevelt National Park, North Dakota” detailed the various roles that lichens play in the ecosystems found in THRO (Egan 1984). The report was prepared in response to the proposed construction of a gas processing plant approximately 25 kilometers from the North Unit. The report estimates the biomass of different classes of lichens and attempts to quantify their contribution to ecological processes such as the food chain, nitrogen fixation, soil formation and soil stabilization. With an assumption that either 5% or 10% of the sensitive lichens would die with the increased air pollution, the effects upon the ecological roles of lichens was judged to be insignificant. However, these findings were greatly limited by a lack of knowledge of the sensitivities of most lichen species to air pollution. There were

Table VII-8. Summary of mean sulfur concentra-tions the moss *Abietinella abietina* from the North and South its of THRO and the nearby Lone Butte oil field. ource: Bilderback 1987)	
Location	Total Sulfur (mg/g)
Lone Butte oil field	1078 ± 90
North Unit site 1	775 ± 34
North Unit site 2	806 ± 88
North Unit site 3	833 ± 61
South Unit site 1	706 ± 64
South Unit site 2	744 ± 97
South Unit site 3	668 ± 75
South Unit site 4	768 ± 65
South Unit site 5	767 ± 73

many other limitations on the study as well and more information is needed before a realistic assessment of the effect of increased air pollution upon lichen populations can be made.

Few data have been collected on trace metal deposition in the park. There was, however, a baseline elemental analysis study done on bryophytes (Bilderback 1987). Total S as well as 15 other elements were analyzed in the common moss *Abietinella abietina* from several locations within the park as well as from a nearby oil field. *Abietinella* was chosen because it was easily identifiable, abundant in THRO and exhibited visible damage in the Lone Butte oilfield. The S levels for the moss from the nearby oil field were significantly higher than those from the park. Levels of other elements were not found to be elevated and there was no discernible pattern in the S concentrations in moss samples taken within the park (Table VII-8). There was also no discernible pattern in the elemental analyses which included P, K, Ca, Mg, Al, Fe, Na, Mn, and other metals. These findings suggest that there is no significant accumulation of these elements in this species in the park that can be traced to a local source, however, it should be noted that little is known about the accumulation of sulfur and other metals by *A. abietina*. Different species vary widely in their accumulation efficiency, and *A. abietina* may or may not be an efficient accumulator.

Vascular plants in THRO have also been sampled and analyzed for sulfur. A study was completed by the USGS on the baseline elemental-composition of selected plants and soils in THRO (Gough et al. 1985). Samples of green ash leaves, western wheatgrass leaves and culms, big sagebrush stems and leaves, the lichen *Parmelia sulcata*, and soils were collected in the park to estimate the biogeochemical variability of the area. The study found no instances of phytotoxic or zootoxic conditions in the plant and soil materials sampled, except for possibly high levels of zinc in the lichen *Parmelia sulcata*. Concentrations of most elements were found to be uniform throughout both the North and South Units of the park. The study established baseline values for many elements and made recommendations for further study .

Bilderback (1990) sampled five plant species at numerous locations in THRO during 1987: big sagebrush, silver sagebrush, western wheat grass, Rocky Mountain juniper, and skunkbush sumac. The sampled vegetation was compared with that from two control sites (40 km south of the South Unit and 8.5 km west of the South Unit) which were chosen because of minimal oil and gas development in their vicinity. The sampled vegetation from the South Unit did not exhibit elevated levels of S. Mean S content of western wheatgrass in the South Unit ranged from 1.1 to 1.7 mg/g which was not significantly different ($p<0.05$) from the control sites. Visible symptoms of damage were observed on Rocky Mountain juniper and skunkbush sumac in an oil field near the North Unit in the same study. The stem tips of juniper were yellow or brown, and the leaves of sumac displayed interveinal yellowing (Bilderback 1990).

In the North Unit, some elevated S levels were found but it was not determined whether these levels were due to variation in soil chemistry or deposition of pollutants. Mean S content of western wheatgrass in the North Unit ranged from 1.2 to 2.2 mg/g, whereas control site values for western wheatgrass ranged from 1.1 to 1.4 mg/g. Total S content at four of the sites in the North Unit were not significantly higher than those at the control sites ($p>0.05$), although two of the sites were higher ($p<0.05$). Gough et al. (1985) also detected similar elevated levels of S in western wheatgrass for a similar location. Rocky mountain juniper and big sagebrush did not exhibit S contents that were higher than the control site ($p>0.05$). Silver sagebrush at two sites in the North Unit had S contents significantly higher than that at the South Unit ($p<0.05$).

D. AIR QUALITY RELATED VALUES

In 1977, the park prepared its first documentation of air quality related values, including historical air quality and important vistas. In 1980, the park prepared a list of integral vistas which included 24 separate sites.

1. Aquatic

There are no air pollution threats to aquatic resources in THRO. This is primarily a consequence of the high buffering capacity of soils in and around the park and consequent high concentrations of base cations and ANC in surface waters.

2. Terrestrial

The principal air pollution threat to terrestrial resources in THRO is S pollution in the form of SO_2 and H_2S. Although ozone pollution is presently not a major concern, the potential for synergistic effects from SO_2 and ozone on vegetation has been documented (Reinert et al. 1969, Tingey and Reinert 1975). Decreases in growth and yield of various crop species were somewhat greater than additive at concentrations of 50 ppbv of each pollutant throughout the growing season, but the differences were not always significant. Greater than additive effects were demonstrated for quaking aspen (*Populus tremuloides*) (Karnosky 1976). Exposure of eastern white pine (*Pinus strobus*) to either gas alone produced 3-4% chlorotic mottle and premature needle drop but was found to increase to 16% when both gases were present at the same doses (100 ppbv, 8 hours/day, 5 days/week for 4-8 weeks) (Treshow and Anderson 1989). Because ambient concentrations of both pollutants in THRO appear to be currently below the threshold doses for sensitive species (especially SO_2), there is no urgent need for concern about synergistic effects. However, if future monitoring indicates higher pollutant levels than are currently experienced in THRO, synergistic effects should be taken into consideration.

Bioindicators are those species for which pollutant sensitivity has been documented and for which extensive data exist on their dose-response to pollutants and on symptomatology. In some cases, bioindicators can be important indicators of 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. Much of this work has been conducted on species native to the northeastern and southwestern United States and very little work has been conducted on species of the Rocky Mountain and northern Great Plains regions.

Vascular plant species of THRO for which there is reference information include western wheatgrass, Rocky Mountain juniper, and skunkbush sumac. However, the use of western wheatgrass as a sensitive receptor may be limited due to its sensitivity to grazing pressures and its annual growth cycle. Rocky Mountain juniper has persistent vegetation, is not subject to grazing pressures, is common in western North Dakota and therefore is a good candidate for bioindicator. Skunkbush sumac is deciduous but warrants further investigation as a bioindicator as well.

Of the hardwood species present at THRO, quaking aspen is the most sensitive to both SO_2 and ozone. Aspen grows at various locations in riparian ecosystems in the park. Numerous studies have documented the sensitivity of this species to ozone under field and experimental 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 aspen includes chlorosis, stippling, necrotic spotting, and leaf margin burn. Symptoms generally vary seasonally, with stippling being most prominent in the spring and black, bifacial (both leaf surfaces) necrosis appearing in late summer (J.P. Bennett, pers. comm.). Great care must be taken in distinguishing ozone symptoms from various pathogens and insect herbivores commonly found on this species. Paper birch (*Betula papyrifera*) is also sensitive to both SO_2 and O_3, although its symtomatology is not as clear as that of aspen, and it is not as common in THRO. Black cottonwood *(P. balsamifera* subp. *trichocarpa*), which has symptoms similar to those of aspen, is another potential bioindicator for ozone (Woo 1996). However, it is generally regarded as less sensitive to ozone than aspen (Table VII-9). Neither of these hardwood species has the clarity of ozone symptomatology found in conifers such as ponderosa pine (*Pinus ponderosa*) which is too rare in THRO to make it a viable species for monitoring.

An inventory of vascular plants found in THRO was compiled in 1988 and is available in the NPFlora database. Table VII-9 summarizes vascular plant species of THRO with known sensitivity to ozone, SO_2, and NO_x. This table is based on a variety of sources from the published literature and other information. It should be noted that the various sources used a wide range of field and experimental approaches to determine pollutant pathology, and that sensitivity ratings are general estimates based on published information and our expert opinion. While it will not be possible for park staff to collect data on all the species indicated in Table VII-9, the list can be used by park managers to indicate potentially sensitive species. Of the many plant species in THRO, it is likely that there are many other species that those in the table which have high sensitivity to air pollution, but we currently have no information about them.

Table VII-9. Plant species of THRO with known sensitivities to SO_2, ozone, and NO_x. L = low, M = medium, H = high, none = unknown. (Sources: Esserlieu and Olson 1986, Bunin 1990, Blett et al. 1993, National Park Service 1994, Electric Power Research Institute 1995, Binkley et al. 1996, Brace and Peterson 1996)

Species	SO_2 Sensitivity	O_3 Sensitivity	NO_x Sensitivity
Acer negundo	M	M	
Achillea millefolium		L	
Agoseris glauca	M		
Agropyron smithii	M		
Amaranthus retroflexus	M		
Ambrosia psilostachya		L	
Amelanchier alnifolia	H	M	
Arctostaphylos UVa-ursi	L	L	
Artemisia cana	L		
Artemisia ludoviciana	M		
Artemisia tridentata	M	L	
Betula occidentalis	M		
Betula papyrifera	H	H	
Bouteloua gracilis	L		
Bromus tectorum		M	
Calochortus nuttllii		L	
Chenopodium fremontii		L	
Cirsium arvense		L	
Cirsium undulatum	M		
Clematis ligusticifolia	M		
Collomia linearis		L	
Convolvulus arvensis	H		
Cornus stolonifera	M	L	
Corylus cornuta	H	L	
Descurainia pinnata		L	
Festuca octoflora		L	
Fragaria virginiana		H	
Hackelia floribunda	L		
Hedysarum boreale		M	
Helianthus anuus	H	L	
Hymenoxys richardsonii	L		
Juniperus communis	L		

Table 9. Continued.

Species	SO_2 Sensitivity	O_3 Sensitivity	NO_x Sensitivity
Juniperus scopulorum	L		
Medicago sativa		M	

Oryzopsis hymenoides	M		
Pinus ponderosa	M	H	H
Poa pratensis		L	
Polygonum douglasii		L	
Populus tremuloides	H	H	
Potentilla fruticosa		L	
Prunus virginiana	M	H	
Rhus aromatica	H		
Rosa woodsii	M	L	
Rubus idaeus	H		
Rumex crispus		L	
Taraxacum officinale		L	
Toxicodendron radicans	L	L	
Tragopogon dubius	M		
Trifolium pratense	L		
Trifolium repens		H	
Vicia americana		L	
Viola adunca		L	

Table VII-10 summarizes lichen species of THRO with known sensitivity to ozone and SO_2. As in Table VII-9, this table is based on a variety of sources from the published literature and other information. It should be noted that diagnostic symptoms of air pollutant injury to lichens are difficult to identify, and that some species have reduced productivity or even mortality without exhibiting visible symptoms (Nash and Wirth 1988). One of the best sources of background information and guidelines for addressing the use of lichens as sensitive receptors of air pollution is Stolte et al. (1993).

Bryophytes are also quite sensitive to SO_2 exposure; for example, in western Europe most bryophytes have been eliminated from habitats exposed to 170 ppbv SO_2 during the growing season (Gilbert 1968, 1969).

Table VII-10. Lichen and bryophyte species of THRO with known sensitivities to SO_2 and ozone. L = low, M = medium, H = high, none = unknown. (Sources: Peterson et al. 1993, National Park Service 1994, Electric Power Research Institute 1995, Binkley et al. 1996)

Species	SO_2 Sensitivity	Ozone Sensitivity
Aspicilia caesiocinerea	L	
Barbula convoluta	M	
Brachythecium rutabulum	M	
Bryoria capillaris	H	
Bryoria fuscescens	M	
Bryum argenteum	L	
Buellia punctata	L-M	
Caloplaca cerina	M	
Caloplaca flavorubescens	H	
Caloplaca holocarpa	M	
Candelaria concolor	M-H	
Candelariella vitellina	M	
Candelariella xanthostigma	M	
Ceratodon purpureus	L	
Cetraria pinastri	M	
Cladonia chlorophaea	M	
Cladonia coniocraea	M	
Cladonia fimbriata	M-H	
Cladonia gracilis	L-M	
Collema tenax	M	
Encalypta procera	H	
Encalypta vulgaris	H	
Funaria hygrometrica	L	
Grimmia anodon	M	
Hyperphyscia adglutinata	M	
Hypnum cupressiforme	M	
Hypogymnia physodes	L-M	
Hypogymnia tubulosa	H	
Jaffeuliobryum raui	M	
Lecanora chlarotera	M	
Lecanora dispersa	L	
Lecanora hageni	L	
Lecanora muralis	M	
Lecanora saligna	M	
Lecidea atrobrunnea	L	
Orthotrichum pumilum	H	
Parmelia caperata	M	
Parmelia subaurifera	H	
Parmelia subolivacea		L
Parmelia sulcata	M-H	M-H
Phaeophyscia nigricans	L-M	
Physcia adscendens	M	
Physcia aipolia	M	

Physcia aipolia	M	
Physcia caesia	M	
Physcia millegrana	M	
Table 10. Continued		
Species	SO_2 Sensitivity	Ozone Sensitivity
Physcia stellaris	L-M	
Physconia detersa	M	
Pohlia cruda	L	
Rhizoplaca melanophthalma	H	
Schistidium strictum	M	
Tortula ruralis	M	
Usnea hirta	M	
Usnea subfloridana	M	
Xanthoria elegans	M	
Xanthoria fallax	M	
Xanthoria polycarpa	M	L

E. RESEARCH AND MONITORING NEEDS

Resource managers at THRO have insufficient data for predicting potential air pollution impacts on specific resources. For example, the sensitivity of native plant species to elevated ambient S compounds is poorly quantified. One can stand at certain locations on the park boundary and smell H_2S (D. Peterson, personal observation), which suggests that the gas is present in reasonably high concentrations; however, there is insufficient information on visible symptoms or physiological impairment of vegetation or effects on wildlife to determine if there are adverse effects. Numerous studies conducted on the effect of pollutants on park vegetation provide valuable baseline values for future comparison.

The park should establish strong public/private partnerships with the other federal and state agencies and other institutions to ensure that external threats are minimized and the interests of both the park and its neighbors are equitably met.

1. Deposition and Gases

Atmospheric concentrations of pollutants, especially S, need to be better quantified at THRO. A sulfation-plate study in 1987 found potentially elevated S deposition levels in the North Unit of the park, however the study was limited by an insufficient number of samples (Bilderback 1987). An extension of this study would be very useful in determining S distribution patterns. The possibility of utilizing portable ambient SO_2 and H_2S monitors in conjunction with the sulfation plates should be investigated. At the present time, the accuracy of such equipment is not sufficient to detect concentrations in the ppb range. However, if improvements in SO_2 and H_2S detection technology are made, it may become feasible to complete an intensive investigation of ambient levels in and around the park. The South Unit should also be included in this study. A qualified statistician should be consulted before sampling begins in order to determine the most effective sampling methodology. Statistical rigor is needed not only to lend credibility to the study results but also to facilitate spatial analysis.

Monitoring data collected at the North Unit provide important information to support ongoing long-term environmental monitoring. Thus, it is critical to continue monitoring atmospheric concentrations at the North Unit. The South Unit has historically experienced lower atmospheric concentrations and deposition rates than the North Unit, but potential economic development in the vicinity could degrade air quality, and the park needs to be prepared to reinitiate monitoring. Data from the special purpose monitoring site established near the South Unit attests to the importance of monitoring small emitters (oil and gas wells) that are very near the park and can heavily impact the air quality in a small area. Periodic monitoring adjacent to local sources should be considered to document the potential effect of non-PSD sources on air quality in the park.

A better spatial characterization of ozone distribution at THRO would be useful, although levels are currently below those believed to adversely affect sensitive plant species. The ozone analyzer at the North Unit should continue to be operated in order to document any future changes. In addition, a network of passive ozone samplers would be useful to compare ozone measurements from different locations in the park. Three samplers in the North Unit, three in the South Unit and one in the Elkhorn Unit should be sufficient to spatially characterize the ozone distribution, with weekly samples collected for two months during the summer. Two years of monitoring should be sufficient to establish spatial patterns and a reference point in time.

One of the samplers could be colocated with the existing ozone analyzer at the North Unit to facilitate comparisons and as a check on the accuracy of the samplers. The two other samplers in the North Unit could be situated along the Little Missouri River. Placement near the river assures good air flow around the sampler. For the same reason, samplers in the Elkhorn and South Units could also be placed near the river. Samplers should be situated where they are reasonably accessible but not within 50 meters of a road or trail where they may be subject to excessive dust or vandalism.

2. Aquatics

We do not recommend any additional research or monitoring efforts with respect to aquatic effects of air pollutants in this park. Atmospheric deposition of acidic compounds and nitrogen are low in the park and are not expected to change dramatically in the near future. More importantly, the aquatic resources in the park are not sensitive to air quality degradation.

3. Terrestrial

It is difficult to establish a monitoring system for detecting air pollutant effects on plants of THRO because there are (1) few known visible symptoms of air pollutant effects on plants in the field in the northern Great Plains region, and (2) few data on pollution effects in plant species found in THRO. We must therefore rely on data published on other species and from experimental studies.

Currently, the threats to THRO terrestrial resources from SO_2 and H_2S are sufficiently urgent to monitor sensitive species. Very little information on the sensitivity of THRO plant species to H_2S is available. Surveys of plants adjacent to oil wells can be conducted to determine any potential foliar impacts caused by H_2S. Plants with potential symptoms can then be compared to plants within park boundaries. Threats from ozone are less urgent and monitoring ozone-sensitive species is not recommended at this time. However, if ozone concentrations increase in the future, monitoring should be considered. Three levels of monitoring associated with increasing amounts of effort and expense are detailed in Appendix A. It is recommended that quaking aspen be used as a sensitive receptor for both ozone and SO_2 at THRO. We recommend placing plots at four locations in THRO: two plots in the North Unit and two at the South Unit. Colocation of these plots with the electronic ozone analyzer and passive ozone samplers should be done when possible. Monitoring should follow the methodology developed by Forest Service and National Park Service scientists for evaluating pollutant injury (Stolte and Miller 1991, Stolte et al. 1992).

If lichens are included in a future monitoring effort, they should be monitored at locations adjacent to the tree plots. The epiphytic lichen *Parmelia sulcata* is a species that is recommended for further monitoring. Not only is it one of the most common species but it was found to contain relatively high levels of a large number of elements which indicates that it may be more sensitive to atmospheric inputs than other lichen species (Gough et al. 1985). Another recommended species for monitoring is *Orthotrichum pumilum*, which grows on the trunks of green ash trees (Bilderback 1987). This species has been found to be sensitive to SO_2 and is common in the park. Species that could also be investigated further as indicators include *Grimnia anodon, Jaffeuliobryum raui, Schistidium strictum, Hypnum cupressiforme, Tortula ruralis, and Brachythecium rutabulum*. These species are found on the large sandstone boulders in the park and therefore are well exposed to atmospheric inputs and are easily photographed and documented. When these species were surveyed in 1986 and 1987, none of them exhibited decreased abundance or injury (Bilderback 1987).

Herbaceous plants may also be used in future monitoring efforts. Some of the S-sensitive vascular plant species of THRO for which there is baseline information include western wheatgrass, and skunkbush sumac. Extensive chemical analyses are already available providing a valuable baseline with which to compare future analyses. Resurveys of lichens done after significant new pollution sources begin operating in the area should compare their results with these to determine what effects the new sources may be having on the lichen communities of THRO. Again, monitoring should be done at locations adjacent to the tree plots wherever possible.

A study was completed by the USGS on the baseline elemental-composition of selected plants and soils in THRO (Gough et al. 1985). It has been over 10 years since the baseline study data was gathered. It is recommended that the park pursue funding to sample the same species again, particularly for S isotopes, to look for changes. A literature search should also be conducted to assess new knowledge that could be applied to this issue.

4. Visibility Recommendations

Energy development adjacent to the park may cause visibility impairment at THRO. Therefore, visibility monitoring (particle and optical) should be conducted at the park. The monitoring data could be used to assess potential impacts local energy development may have on visibility at the park.

VIII. WIND CAVE NATIONAL PARK

A. DESCRIPTION

Wind Cave National Park (WICA) is located in the Black Hills region in southwestern South Dakota. The park was established in 1903 to protect and manage a nationally significant cave resource. In 1912, Wind Cave National Game Preserve was superimposed on the surface of the park to provide permanent range for bison and other native animals. Management of the national park and game preserve was consolidated under the National Park Service in 1935.

WICA is 11,454 ha in area, of which 3.5% is developed. Lands south and east of the park are mainly used for dry-land ranching. Custer State Park comprises the northern boundary of the park while Black Hills National Forest lies to the west.

The remarkable cave systems located in WICA include the third longest limestone cave in the United States. The first recorded discovery of the cave was in 1881. The name “Wind Cave” refers to the strong winds which frequently blow through the entrance of the cave. The caves contain a diversity of geological formations which are valuable for both their rarity and beauty. The protection and interpretation of cave resources is the primary management objective of the park. In contrast, the surface resources include a diversity of flora and fauna that are mostly distinct from the caves. Aboveground ecosystems are managed primarily for maintenance of conditions that approximate those of pre-European settlement. The relatively small size of the park, boundary fences, and proximity to lands with contrasting management objectives provide a challenging situation for management of park resources. In 1991 annual park visitation totaled 100,000 for the cave and 1.1 million for the surface (WICA 1994).

1. Geology and Soils

The Black Hills, which constitute the easternmost extension of the Rocky Mountains, are an isolated and unglaciated group of mountains that rise above the surrounding Plains. WICA is located in the southeastern portion of the Black Hills and contains a diversity of geological materials. The oldest geological formation is a small portion of the core of the central Black Hills, consisting of Precambrian metamorphic (schists, slates, and quartzites) and igneous (granite and pegmatite) rocks (Darton 1951). All other materials are of sedimentary origin, including Mesozoic sandstones and shales of the Sundance and Spearfish formations; Paleozoic limestones, sandstones, and shales of the Minnekahta limestone/Opeche, Minnelusa sandstone, Pahasapa limestone/Englewood limestone, and Deadwood formations; and Cenozoic clays and gravels of the White River group.

WICA contains a diversity of soils associated with different topographic features and vegetation. Nearly all the soils are well-drained and many are skeletal with intermittent rock outcrops. Entisols dominate the shallowest soil profiles, Inceptisols and Alfisols dominate the forested sites, and Inceptisols and Mollisols dominate the mixed-grass prairies. The red hues of most of the well-drained soils in the region indicate that the soils of this region are highly weathered.

2. Climate

The park ranges from 1,110 to 1,528 m elevation. Climate is typical of the northern Plains region with generally moderate average temperatures, but with extreme summer and winter temperatures (>35°C and <-15°C) common in most years. Most precipitation occurs during late spring and during frequent summer thunderstorms. Winter precipitation commonly falls as snow but contributes only a small fraction of the annual precipitation of 40 cm.

Wind rose data from WICA during the period 1989 through 1992 show that the predominant wind direction is from the northwest. These winds tend to predominate between November and April (Weber 1982). A second frequent wind direction is from the east, which tends to occur between May and October.

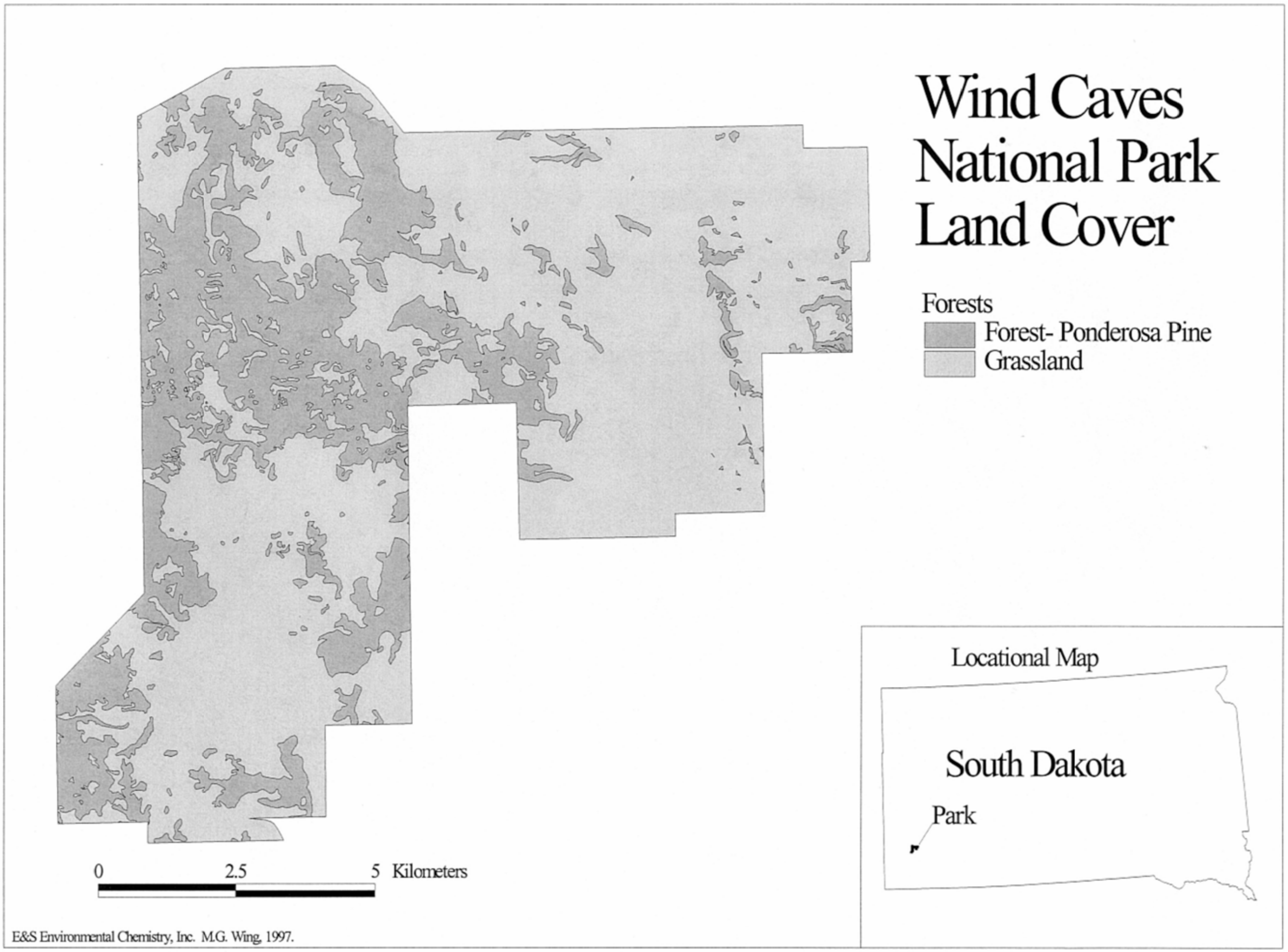
3. Biota

The vegetation of WICA can be roughly divided into three components: (1) mixed-grass prairies (approx. 75%), (2) ponderosa pine woodlands (approx. 20%), and (3) deciduous forest complex (including "woody draws") (approx. 5%) that dominates riparian areas. Forested and grassland areas are indicated in a general map of the park (Figure VIII-1).

A diversity of grass species, often mixed with forbs, dominate areas with low soil moisture, especially south-facing sites and thin, rocky, soils (Froiland and Weedon 1990). Some of the representative species include little bluestem (*Andropogon scoparius*), blue grama (*Bouteloua gracilis*), buffalo grass (*Buchloe dactyloides*), Japanese cress (*Bromus japonicus*), Western wheatgrass (*Agropyron smithii*), prairie junegrass *(Koelaria pyramidata*), pasture sage (*Artemisia frigida*), and prickly pear (*Opuntia* spp.). The presence of grasslands is facilitated by fire which kills encroaching woody vegetation. Grasslands in WICA provide critical forage for native ungulates and many other animals.

Ponderosa pine woodlands (*Pinus ponderosa*) contain a mix of subdominant shrubs, forbs, and grasses depending on local soil moisture and stand density. Ponderosa pine in WICA is generally found on north and east slopes, ridges and higher elevations, and sites with more soil moisture than those dominated by grasslands. Age, stem density, and basal area of the pines vary widely as a function of site history, including human disturbance and fire occurrence. Most of the pines are in younger age classes (<100 years) because of extensive cutting prior to creation of the park, although there are a few trees up to about 300 years old.

The deciduous woodland complex occurs primarily along and adjacent to riparian areas of the park (Magruder 1985). These areas maintain higher soil moisture in drainages which have surface



Wind Caves
National Park
Land Cover
Forests
Forest- Ponderosa Pine
Grassland
0
2.5
5 Kilometers
Locational Map
South Dakota
Park
E&S Environmental Chemistry, Inc. M.G. Wing, 1997.

water (often transient). Referred to locally as woody draws, they intersect drier adjacent sites dominated by grasses. Dominant species of deciduous woodland include plains cottonwood (*Populus deltoides*), bur oak (*Quercus marocarpa*), paper birch (*Betula papyrifera*), box elder (*Acer negundo*), American elm (*Ulmus americanus*), green ash (*Fraxinus pennsylvanica*), Bebb willow (*Salix bebbiana*) chokecherry (*Prunus virginiana*), western snowberry (*Symphoricarpos mollis*), and skunkbush sumac (*Rhus trilobata*).

The primary natural disturbance in WICA is fire, which can spread rapidly during the summer when fuel moistures are low, winds are high, and a lightning ignition source is available. Fire frequency was higher prior to the 20th century due to both natural and human ignitions in the area; grasslands may have burned every 6-7 years and forests every 15-25 years (Weaver 1967). While current policy is to suppress all unplanned ignitions, prescribed fire is being successfully used at WICA to reduce hazard fuels and stem densities in ponderosa pine stands.

By the time of the creation of WICA, both bison (*Bison bison*) and elk (*Cervus elaphus*) had been extirpated from the Black Hills region. Bison, elk, and pronghorn (*Antiolcapra americana*) were subsequently introduced from other locations. Bison are managed essentially as an island population because a fence prevents their movement from the park to adjacent lands. There are many smaller mammal species in the park , including black-tailed prairie dogs (*Cynomys ludovicianus*). Although the NPFauna database records the presence of many bird species and several reptile and amphibian species, there is little ecological information on them.

4. Aquatic Resources

Although there is little information on the hydrology of WICA, historical accounts indicate that surface water was more prevalent during the early part of the 20th century. Streams in and around the park have been impacted by water withdrawals and stream modification. Subsurface water movement in WICA is poorly understood. It is likely that the main parking lot and administrative developments in the park are having impacts on subsurface hydrology and potential water transport into the caves.

The magnificent geological formations contained in the caves of WICA have been created primarily through groundwater hydrological processes. In other words, they are affected by "bottom-up" water movement rather than "top-down" flow from the surface. As a result, air pollution sources such as acidic deposition have no real impact on cave resources. Despite their importance to overall park management, the caves are not relevant to this discussion of air quality.

B. EMISSIONS

The human population of South Dakota is small, and there are no major metropolitan areas. Ranching and farming are major industries (Weber 1982). South Dakota generally has lower emissions of SO_2, NO_x and VOCs than other Rocky Mountain and northern Great Plains states and emissions of criteria pollutants in the immediate vicinity of WICA are relatively low. The closest urban area to WICA is Rapid City, where there is light industry. The city lies in a valley where pollution can remain trapped for several days during inversions. Rapid City is the only urban area in South Dakota that does not meet EPA standards for particulates. Coal-fired power plants in southeastern Wyoming, upwind of WICA, may pose a potential threat to air quality in the Black Hills Area.

Local pollution comes from a variety of sources: (1) the sawmills in the Pringle and Custer areas, (2) a rock crushing mill and quarry at Hot Springs, (3) a feldspar mill in Custer, and (4) vehicles and home wood stoves in the Hot Springs area (WICA 1993). While small quantities of emissions from these sources and from Rapid City may reach WICA, of greater importance are regional-scale sources located to the west of WICA. Annual emissions of NO_x in Wyoming are particularly high, and although VOCs are relatively low, they may influence the airshed of WICA as well. Prevailing westerly winds transport NO_x, SO_2, and VOCs eastward over the Black Hills region and WICA, providing precursors for ozone formation during the summer when it is sunny and warm. The development of additional coal-fired power plants in eastern Wyoming would increase emissions transported to WICA.

A summary of statewide emissions of NO_x, SO_2, and VOC for 1994 is listed in Tables II-2, II-3 and II-4. Coal-burning power plants are major emission sources of SO_2 and NO_x in South Dakota. However, they are mostly located downwind and remote from WICA in the far eastern portion of the state. The one exception is Black Hills Power & Light located approximately 90 km north of WICA.

Historically, fires, and therefore smoke, have been a part of the Great Plains ecosystem. Fires are generally not considered to be a significant long term source of pollutants but they can result in episodic degraded visibility and high particulate matter concentrations.

C. MONITORING AND RESEARCH ACTIVITIES

1. Air Quality

a. <u>Wet Deposition</u>

WICA has no NADP site, so the NADP/NTN site at Cottonwood, SD (130 km northeast of WICA) is used to represent wet deposition at WICA. Wetfall chemistry data are available for this location since 1983. The data indicate no particular trends during this period of time (Table VIII-1). Hydrogen ion (acidic) inputs are relatively low, which suggests that the acidity of wet deposition is not a concern in this area. Ionic deposition of N and S are considerably higher than that of other elements.

Table VIII-1. Wetfall chemistry at the NADP/NTN site near WICA. Units are in μeq/L, except precipitation (cm) and H^+ (pH).										
Year	Precip	H^+	SO_4^{2-}	NH_4^+	NO_3^-	Ca^{2+}	Mg^{2+}	Na^+	K^+	Cl^-
1995	47.6	5.6	12.2	16.8	15.9	8.2	1.4	2.1	0.5	1.7
1994	38.2	6.3	16.9	23.6	18.7	11.1	2.0	2.1	0.7	2.0
1993	58.9	3.0	16.1	22.1	16.5	8.8	1.8	2.6	0.8	2.0
1992	53.3	6.3	18.2	20.6	15.1	6.6	1.1	1.7	0.5	1.6
1991	62.7	2.7	13.7	19.1	14.5	11.1	1.8	2.1	0.6	2.7
1990	29.1	4.4	16.0	22.5	18.8	8.7	1.8	2.8	0.5	3.0
1989	29.4	3.7	16.3	18.7	16.8	10.4	2.0	2.7	0.6	2.5
1988	36.1	4.6	12.9	5.5	11.1	8.6	1.4	2.5	0.5	1.4
1987	31.7	4.6	17.6	23.6	17.7	7.4	1.8	2.7	0.7	2.2
1986	56.4	5.9	17.9	12.6	15.7	12.5	2.2	2.3	0.9	1.9
1985	26.3	3.5	18.4	17.5	17.2	14.2	3.5	2.6	1.4	3.2
1984	41.7	1.9	20.5	22.6	17.7	14.4	4.2	3.7	1.0	3.0
Average	42.6	4.4	16.4	18.8	16.3	10.2	2.1	2.5	0.7	2.3

Sulfate deposition is moderately high due to geological sources of this ion. Despite these relatively high ionic concentrations of sulfate, wet deposition of S averages only 1.1 kg/ha/yr, which is quite low (Table VIII-2). This is because precipitation volumes are low. Total inorganic nitrogen deposition is only 2.0 kg/ha/yr, with approximately equal contributions from nitrate and ammonium. The values for S and N deposition, in combination with wetfall input of hydrogen and other ions, indicate that WICA is a relatively clean site and that there is no apparent threat from acidic deposition at the present time.

Table VIII-2. Wet deposition (kg/ha/yr) of sulfur and nitrogen at the NADP/NTN site near WICA.				
Date	Sulfur	NO_3-N	NH_4-N	Total Inorganic N
1995	0.9	1.1	1.1	2.2
1994	1.0	1.0	1.3	2.3
1993	1.5	1.4	1.8	3.2
1992	1.6	1.1	1.5	2.7
1991	1.4	1.3	1.7	3.0
1990	0.8	0.8	0.9	1.7
1989	0.8	0.7	0.8	1.5
1988	0.8	0.6	0.3	0.8
1987	0.9	0.8	1.1	1.8
1986	1.6	1.2	1.0	2.2
1985	0.8	0.6	0.6	1.3
1984	1.4	1.0	1.3	2.4
Average	1.1	1.0	1.1	2.1

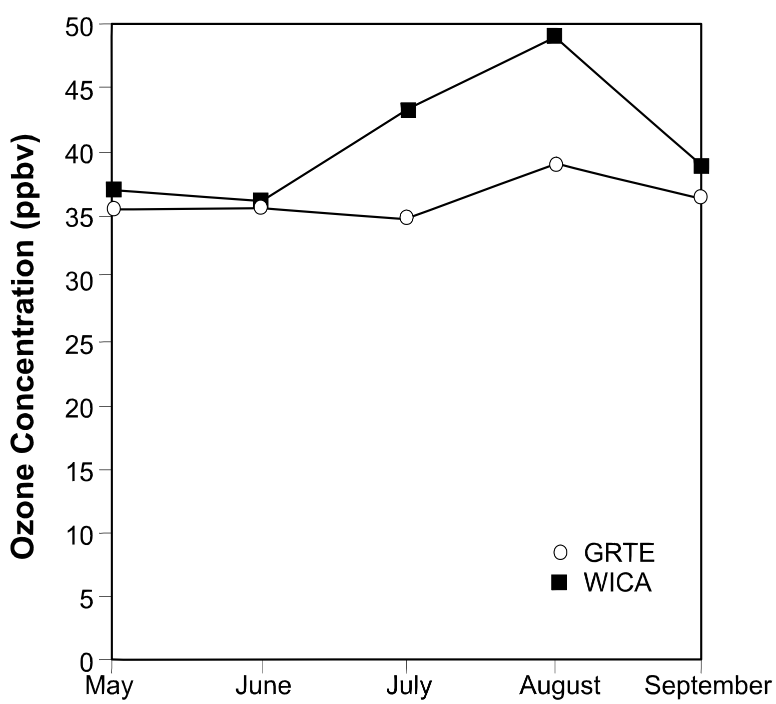
b. Occult/Dry Deposition

There are no dry deposition data available in the vicinity of WICA. Dry deposition is not expected to have a major impact on natural resources in the park.

c. Gaseous Monitoring

Air quality monitoring has been, and continues to be, limited in WICA (Table II-5). Most historical monitoring done in the park was short term or seasonal. The state of South Dakota also does not currently monitor gaseous pollutants. The South Dakota Department of Environment and Natural Resources currently monitors only PM-10. Since 1975, gaseous monitoring had been conducted using SO_2/NO_2 bubblers sampling every sixth day (because levels did not warrant continuous operation). Measured levels of SO_2 rarely exceeded the threshold limit of the instruments. The network was reduced in 1985, with all sites being eliminated as of December 1986. The data are on file with the South Dakota Air Monitoring Program.

Gaseous monitoring in the park has been limited to the use of passive ozone samplers. The samplers were deployed during the summer of 1995, 1996 and 1997 by NPS-ARD. Results of this study are summarized in Figure VIII-2. Weekly average values ranged between 36 and 50 ppbv.



Passive samplers were also deployed in GRTE at the same time, and concentrations in GRTE were

Figure VIII-2. Weekly average ozone concentration, based on passive ozone sampler data from WICA. Data from GRTE is provided for comparison.

found to be slightly lower or equal to those found in WICA during the summer of 1995. These relatively low values suggest that WICA will have minimal impacts on vegetation from ozone, although it is possible that the park could be subjected to regionally elevated ozone concentrations during warm weather and atmospheric inversions.

2. Water Quality

Surface waters are expected to be well-buffered against acid inputs. EPA's STORET data base contains 1,971 pH measurements for southwestern South Dakota, in an area that includes WICA. Only one measurement showed pH less than 6.1, having a reported pH of 1.5, and it was either due to an error or to direct acid discharge (e.g., acid mine drainage). The first pH percentile was 6.6. In other words, 99% of the measured values were higher than pH 6.6. Similarly, 99% of the calcium measurements (n=236) were higher than 10 mg/L (500 μeq/L). Surface water SO_4^{2-} concentrations tended to be very high due to geological sources of SO_4^{2-}. The median SO_4^{2-} concentration (n=77) in the data set was 88 mg/L (1,800 μeq/L). There is no evidence to suggest that surface waters in WICA would be responsive to acidic deposition impacts or that aquatic biota would be affected.

3. Terrestrial

No studies have been performed to date on the effects of air pollutants on the terrestrial resources of WICA. There have been no surveys for endangered, threatened or rare plants. There are also no vegetation status records or maps available for determining what changes may have occurred in species composition (WICA 1994). Much of the focus on resource management in the park relates to wildlife and cave resources. Park management acknowledges that there is a need for periodic monitoring and a need to remain apprised of any new industrial activities in the area.

D. AIR QUALITY RELATED VALUES

1. Vegetation

Current emissions of SO_2, NO_x and VOC may not pose an immediate threat to resources in WICA, but increased industrial activities will possibly bring changes in air quality. Vegetation is the resource which is most sensitive to ozone and SO_2 (based on current knowledge of ozone- and SO_2-sensitive organisms) and several tree species have been identified as potential bioindicators (see below). Additional studies would be needed to further evaluate the potential impact of SO_2 and ozone on terrestrial ecosystems in WICA. Furthermore, baseline data on the condition of sensitive species in the absence of injurious pollutants will be helpful for comparison if pollutant levels increase. Monitoring bioindicators by using detailed descriptions and classifications of sensitive indicators (characteristics of leaf or plant injury) will be necessary for long-term evaluation of ecosystem health.

Ponderosa pine is common in WICA. It is the dominant tree throughout the park and encroaches into grasslands and riparian areas. Ponderosa pine is one of the most ozone-sensitive Western tree species (especially var. *ponderosa*), for which extensive data are available on field (Miller and Millecan 1971, Pronos and Vogler 1981, Peterson and Arbaugh 1988) and experimental (Temple et al. 1992) exposures. The evidence for ozone impacts on ponderosa pine is based on observable symptoms of foliar chlorosis and reduced growth (Peterson et al. 1991, Peterson and Arbaugh 1992) as well as physiological (Darrall 1989, Bytnerowicz and Grulke 1992) data. The cause-and-effect relationship, especially for trees growing in forests of southern California and the southern Sierra Nevada, is clear and quantifiable. The Rocky Mountain variety of ponderosa pine (var. *scopulorum*) is known to be somewhat more tolerant to ozone and has a higher threshold for symptoms of injury under experimental exposures than var. *ponderosa* (Aitken et al. 1984).

Of the hardwood species present at WICA, quaking aspen (*Populus tremuloides*) is the most sensitive to ozone and is a potential secondary sensitive receptor after ponderosa pine. Aspen grows in isolated pockets in riparian areas and other high soil-moisture areas in the park. Numerous studies have documented the sensitivity of this species to ozone under field and experimental 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 aspen includes chlorosis, stippling, necrotic spotting, and leaf margin burn. Symptoms generally vary seasonally, with stippling being most prominent in the spring and black, bifacial (both leaf surfaces) necrosis appearing in late summer (J.P. Bennett, pers. comm.). Great care must be taken in distinguishing ozone symptoms from various pathogens and insect herbivores commonly found on this species.

Aspen is also considered to be sensitive to SO_2 and may be the best bioindicator for this gaseous pollutant. Injury is similar to that normally found for ozone (stippling, followed by bifacial necrosis), although SO_2 -induced injury rapidly bleaches to a light tan color (ozone injury remains dark) (Karnosky 1976). There could be some confusion of ozone injury and SO_2 injury.

Chokecherry, which is common in WICA and highly sensitive to ozone, is another potential bioindicator for ozone. Paper birch, which is common in WICA and highly sensitive to SO_2, is another potential bioindicator for SO_2.

An inventory of vascular plants found in WICA is available in the NPFlora database. Table VIII-3 summarizes vascular plant species of WICA with known sensitivity to ozone, SO_2, and NO_x. This table is based on a variety of sources from the published literature and other information. It should be noted that the various sources used a wide range of field and experimental approaches to determine pollutant pathology, and that sensitivity ratings are general estimates based on published

Table VIII-3. Plant species of WICA with known sensitivities to SO_2, ozone, and NO_x. L = low, M = medium, H = high, none = unknown. (Sources: Esserlieu and Olson 1986, Bunin 1990, Peterson et al. 1993, National Park Service 1994, Electric Power Research Institute 1995, Binkley et al. 1996)			
Species Name	SO_2 Sensitivity	O_3 Sensitivity	NO_x Sensitivity
Acer negundo	M	M	
Achillea millefolium		L	
Agoseris glauca	M		
Agropyron smithii	M		
Ambrosia psilostachya		L	
Amelanchier alnifolia	H	M	
Arctostaphylos uva-ursi	L	L	
Artemisia ludoviciana	M		
Betula papyrifera	H		
Bouteloua gracilis	L		
Bromus tectorum	L	M	
Cercocarpus montanus	M		
Cirsium arvense		L	
Cirsium undulatum	M		
Collomia linearis		L	
Convolvulus arvensis	H		
Crataegus chrysocarpa	L		
Fragaria virginiana		H	
Fraxinus pennsylvanica	M	H	
Geranium richardsonii	M	M	
Helianthus annuus	H	L	
Helianthus maximiliana	H		
Juniperus scopulorum	L		
Koeleria nitida	H		
Oryzopsis hymenoides	M		
Pinus ponderosa	M	H	H
Poa pratensis	H	M	
Populus angustifolia	M		
Populus deltoides	M	L	
Populus tremuloides	H	H	
Prunus virginiana	M	H	
Quercus macrocarpa		L	
Rosa woodsii	M	L	
Rubus idaeus	H		
Rumex crispus		L	
Shepherdia canadensis	L		
Solidago canadensis	H	L	
Solidago rigida	H		
Stipa comata	L		
Taraxacum officinale		L	
Tragopogon dubius	M		
Ulmus americana	M		
Urtica gracilis		L	
Vicia americana		L	
Yucca glauca	L		

Table VIII-4. Lichen species of WICA with known sensitivities to SO_2 and ozone. L = low, M = medium, H = high, none = unknown. (Sources: Peterson et al. 1993, Electric Power Research Institute 1995, Binkley et al. 1996)

Species	SO_2 sensitivity	Ozone sensitivity
Candelaria concolor	M-H	
Candelariella vitellina	M	
Cladonia chlorophaea	M	
Cladonia coniocraea	M	
Parmelia sulcata	L-H	M-H
Peltigera canina	L	H
Physcia aipolia	M	
Physcia caesia	M	
Physcia dubia	L	
Physcia stellaris	M	
Platismatia glauca	M	H
Usnea hirta	M-H	

information and our expert opinion. While it will not be possible for park staff to collect data on all the species indicated in Table VIII-3, the list can be used by park managers to indicate potentially sensitive species. Of the many plant species in WICA, it is likely that there are many species other than those in the table which have high sensitivity to air pollution, but we currently have no information about them.

Table VIII-4 summarizes lichen and bryophyte species of WICA with known sensitivity to ozone and SO_2. As in Table VIII-3, this table is based on a variety of sources from the published literature and other information. It should be noted that diagnostic symptoms of air pollutant injury to lichens are difficult to identify, and that some species have reduced productivity or even mortality without exhibiting visible symptoms (Nash and Wirth 1988). One of the best sources of background information and guidelines for addressing the use of lichens as sensitive receptors of air pollution is Stolte et al. (1993).

E. RESEARCH AND MONITORING NEEDS

Air quality at WICA generally has been considered to be excellent. Visibility has been a valued feature of park resources, with topographic features visible 60 to 90 km to the east. There is the potential for degradation of air quality if mineral and energy exploration increases and if coal-fired power plants are built to the west of the park in Wyoming.

1. Deposition and Gases

Deposition of pollutants could be better quantified at WICA. Although NADP data are available from the Cottonwood, SD site, 130 km to the northeast, it would be desirable to have better quantification of wet and dry deposition at WICA. Short-term collection of wet deposition data could be used to calibrate with the Cottonwood site and establish a reference for deposition of S and N. Short term collection of dry deposition data using NDDN-type collectors would further establish this reference for deposition, but is probably not warranted at the present time.

A better spatial characterization of ozone distribution at WICA would be useful, although levels are currently below those believed to adversely affect sensitive plant species. A continuous ozone analyzer at the Visitor Center should be operated for two summers in order to better establish a reference point in time. In addition, a small network of passive ozone samplers could be established to compare ozone measurements from different locations in the park. Three samplers should be sufficient to spatially characterize the ozone distribution, with collection of weekly samples for two months during the summer. Two years of monitoring would be sufficient to establish spatial patterns and a reference point in time. One of the samplers should be colocated with a continuous ozone analyzer (if possible) to facilitate comparisons and as a check on the accuracy of the samplers. Other samplers could be situated along the Rankin Ridge Trail and the Boland Ridge Trail. Samplers should be situated where they are reasonably accessible but not within 50 meters of a road or trail where they may be subject to excessive dust or vandalism.

Operation of an SO_2 analyzer at the Visitor Center would provide a better characterization of SO_2 deposition at WICA. Two years of monitoring should be sufficient to establish a reference point for SO_2. Because the State of South Dakota does not have an active SO_2 monitoring program, WICA should participate in any future efforts to documenting changes in air quality.

2. Terrestrial

A basic inventory of many natural resources is still in its infancy at WICA. Monitoring schemes, short and long term, are similarly lacking, making it impossible to compare data over time to predict or assess changes. There are no current air pollution threats to WICA vegetation, although ozone, SO_2 and possibly NO_2 could be sources of concern in the future if there are new point sources that significantly increase ambient pollutant levels.

If air pollutants increases in the future, monitoring of terrestrial resources should be considered. If monitoring is implemented, we recommend ponderosa pine and quaking aspen be used as bioindicators for ozone and quaking aspen for SO_2 at WICA. One plot of each species should be sufficient for initial monitoring. Three levels of monitoring associated with increasing amounts of effort and expense are detailed in Appendix A. These plots should be colocated with the passive ozone samplers wherever possible. Monitoring should follow the methodology developed by Forest Service and National Park Service scientists for evaluating pollutant injury (Stolte and Miller 1991, Stolte et al. 1992). If herbaceous species and lichens are included in a future monitoring effort, they should be monitored at locations adjacent to the tree plots if possible.

3. Visibility Recommendations

WICA is located approximately 100 km southwest of the visibility monitoring site at BADL. Due to the close proximity of the Badlands monitoring site and limited intervening terrain between WICA and BADL, visibility data collected at BADL can also be used to represent conditions at WICA.

IX. BADLANDS NATIONAL PARK

A. GENERAL DESCRIPTION

Badlands National Monument was authorized in 1929 in recognition of the remarkable geomorphology and abundant fossils in this region of South Dakota. In addition, a memorandum of agreement between the Oglala Sioux and the Secretary of the Interior added 58,000 ha of reservation land to the monument. The entire 100,000-ha protected area was officially designated as Badlands National Park (BADL) in 1978. Visitation (over 1.5 million in 1991) has increased substantially since that time (BADL 1994). For administrative purposes, the park contains units referred to as the North Unit (which contains Badlands Wilderness) and South Unit (on the Pine Ridge Reservation, including Palmer Creek). The southern portion of the park is managed by the National Park Service in consultation with the Oglala Sioux Tribe and within the context of a reservation, where some agreed upon consumptive uses are permitted.

The most prominent feature of BADL is the dramatic scenery that can be observed throughout the park. Erosional landscapes and a variety of geological substrates of different ages present a visual diversity of colors and patterns, and trace a fascinating geomorphological history in this region. The Badlands Wilderness, surrounded by cliffs and pinnacles, provides vistas of colorful buttes amidst grassland ecosystems. Bison (*Bison bison*) and other native ungulates can be seen throughout the park.

1. Geology

The badlands areas of southwestern South Dakota occur where the widespread peneplain is actively dissected by continual erosion. Erodible geological substrates and an arid climate which discourages rapid vegetation growth lead to a dynamic landscape with active erosion, gullying, and landslides. Geological materials of the White River Group consist of very fine clays which are poorly consolidated, with interspersed beds of sandstone and isolated concretions (Martin 1987, Froiland and Weedon 1990). In addition to clays, the layered badlands contain several different geological materials, including volcanic ash, chalcedony, quartz, and calcite. The distinctive geomorphological features of BADL are not only visually beautiful but contain one of the richest fossil-containing Tertiary deposits in North America.

2. Climate

BADL has hot summers and cold, dry winters. Temperatures reach as high as 40 ºC and as low as -40 ºC. Precipitation is highly variable but averages 40 cm per year, most of this falling as rain during spring and summer; average total snowfall depth is 60 cm. Rain often falls in intensive storms, producing torrents and ephemeral streams that provide the energy for further erosion of the badlands.

The predominant direction of air mass movement is from west to east over South Dakota. Wind rose data from BADL during the period 1989 through 1995 show that winds from the northwest predominate between November and April (Weber 1982). A second frequent wind direction, which tends to occur between May and October, is from the east.

3. Biota

BADL contains ecosystems that are still in the process of recovery from human use during settlement times and during the time park lands were privately owned. Much of this area was previously used for dryland farming, grazing by domestic livestock, market hunting of wildlife, and fossil collecting. A 1919 survey of the Badlands area revealed that there were few native wildlife or trees, although they were abundant just a few decades earlier. Starting in the 1930s, grasslands began to slowly recover with the inception of active resource management and the cessation of livestock grazing. About 97% of the park is now managed for natural zones, and various restoration efforts are underway to encourage the development of sustainable populations of native plants and animals. Inventory of park resources is incomplete and there is little ongoing monitoring. There is little or no information on bird, fish, reptile, amphibian, cave life, or aquatic systems. Knowledge of plant communities, species composition, or carrying capacities for the wildlife they support is likewise lacking. Efforts are underway, however, to improve inventory and monitoring of the condition of natural resources in the park (Plumb undated).

The dominant vegetation of BADL is often referred to as mixed-grass prairie, although this designation encompasses a variety of grassland communities (Cushman and Jones 1988, Froiland and Weedon 1990). The dominant grassland community of this region occurs as a moderately dense short to medium tallgrass prairie dominated by western wheatgrass (*Agropyron smithii*), green needlegrass (*Stipa viridula*), blue grama (*Bouteloua gracilis*), and needle-and-thread (*Stipa comata*). Associated species include fringed sage (*Artemisia frigida*), prairie junegrass (*Koeleria pyramidata*), little bluestem (*Andropogon scoparius*), silky wormwood (*Artemisia dracunculus*), purple coneflower (*Echinacea angustifolia*), and various other forbs. Blue grama and buffalo grass (*Buchloe dactyloides*) form a mosaic of patches in combination with western wheatgrass and green needlegrass. The mixed-grass prairie of BADL, although still recovering, has been recognized as the finest extant. That portion of the prairie preserved in the Badlands Wilderness is especially important for its thriving population of black-tailed prairie dogs (*Cynomys ludovicianus*) (BADL 1994).

Trees are relatively uncommon on a large scale but are locally common in areas of higher soil moisture such as drainages with ephemeral streams and ponds, or slumps near the shoulder slope of rocky ridges, which produce topographic depressions. Red cedar (*Juniperus virginiana*) and Rocky Mountain juniper (*J. scopulorum*) dominate the vegetation in these rocky slumps. In lower, flatter areas of the landscape, several deciduous species dominate, including plains cottonwood (*Populus deltoides*), peach-leaved willow (*Salix amygdaloides*) and other willows, box elder (*Acer negundo*), green ash (*Fraxinus pennsylvanica*), and American elm (*Ulmus americana*). Riparian woodland in this region is best developed along the White River. Much of BADL at lower elevations and along eroded backslopes is sparsely vegetated and, in the case of highly active erosional situations, devoid of vegetation. Species that are commonly found in these areas include curly-cup gumweed (*Grindelia squarrosa*), broom snakeweed (*Gutierrezia sarothrae*), wild buckwheat (*Polygonum convolvulus*), and a wide range of rare plant species.

Animal populations of the BADL region are probably still recovering from hunting and habitat alteration. The bison is perhaps the species most valued by park visitors, and considerable effort has been expended to restore populations of these native ungulates. In 1964, bighorn sheep (*Ovis canadensis canadensis*) were introduced into the park to replace the extinct Audubon bighorn sheep (*O. canadensis auduboni*) which were present until the 1920s. The herd is currently divided into three bands, two in the North Unit and one in the South Unit, for a total of about 110 animals. As in most other areas of the western United States, predators such as the grizzly bear (*Ursus arctos* subsp. *horribilis*) and gray wolf (*Canis lupus*) have been extirpated, with little likelihood of reintroduction in the forseeable future.

Other common mammals in the park include mule deer (*Odocoileus hemionus*), whitetail deer (*O. virginianus*), American pronghorn (*Antilocapra americana*), coyote (*Canis latrans*), black-tailed prairie dog, white-tailed jackrabbit (*Lepus townsendii*), cottontail (*Sylvilagus floridanus*), and thirteen-lined ground squirrel (*Spermophilus tridecemlineatus*). In addition, the park has an active program for restoring black-footed ferret (*Mustela nigripes*) populations. Common or highly valued bird species include golden eagle (*Aquila chrysaetos*), three hawk species (*Buteo* spp.), magpies (*Pica pica*), western meadowlark (*Sturnella neglecta*), cliff swallow (*Hirundo pyrrhonata*), turkey vulture (*Cathartes aura*), great-horned owl (*Bubo virginianus*), burrowing owl (*Athene cunicularia*), and killdeer (*Charadrius vociferus*).

Other important animals include prairie rattlesnake (*Crotalus viridis*), bullsnake (*Pituophis melanoleucus sayi*), western painted turtle (*Chrysemys picta belli*), and blotched tiger salamander (*Ambystoma tigrinum melanostictum*). A conspicuous scarcity of lizards has been noted at BADL with the only lizard found in the park being the rare eastern short-horned lizard (*Phrynosoma douglassi brevirostre*). Several species of reptiles considered rare by the state of South Dakota which would be expected to be found within the park include the six-lined racer (*Cnemidophorus sexlineatus*), lesser earless lizard (*Holbrookia maculata maculata*), many-lined skink (*Eumeces multivirgatus multivirgatus*), and western box turtle (*Terrapene ornata*).

4. Aquatic Resources

The White River is the major stream in the BADL region. Although the river actually cuts through only a small portion of the park (near the White River Visitor Center), many tributaries from the park enter the White River. It is responsible for many of the large-scale patterns of erosion found in the area. Sage Creek is an ephemeral tributary of the Cheyenne River in the western part of the North Unit. Bear Creek, Battle Creek, and Cedar Creek all have their headwaters in BADL and are tributaries of the Cheyenne River. A small portion of the park has intermittent upper tributaries that drain in the Bad River. The White, Cheyenne and Bad Rivers are all part of the Missouri River system. There are many small ephemeral streams and small ponds in drainages and convexities throughout the park. Both standing and running waters in the park nearly always have a high sediment content, and most standing water is cloudy in appearance because it contains fine clays in colloidal suspension. Water is clearly a limiting factor to vegetation and animals in this region. There are also a number of stock dams and wildlife water impoundments that provide a year-round source of water for domestic livestock in the South Unit and a variety of wildlife throughout the park.

B. EMISSIONS

Emissions of criteria pollutants in the immediate vicinity of BADL are relatively low. Ranching and farming are the primary industries on the Pine Ridge Reservation, and the area is sparsely populated with no major metropolitan areas (Weber 1982). South Dakota has lower emissions of SO_2, VOCs and NO_x than other Rocky Mountain and northern Great Plains states. Pollutant sources close to BADL include Rapid City and Black Hills Power & Light Company, 80 and 120 km to the northwest of BADL, respectively. Rapid City is the only urban area in South Dakota that does not meet EPA standards for particulates. Other large sources, including coal-fired power plants, are located in the easternmost portion of the state, remote and generally downwind from BADL. A summary of state-wide emissions of NO_x, SO_2 and VOC for 1994 is listed in Tables II-2, II-3, and
II-4.

While small quantities of emissions from Rapid City sources may reach BADL, of greater importance are regional-scale sources located to the west of BADL. Emissions of NO_x and SO_2 from industrial and electric-utility facilities in eastern Wyoming and western South Dakota are the greatest potential concern at BADL. Annual emissions of NO_x in Wyoming are particularly high, and a portion of these emissions may reach the BADL airshed. Westerly winds transport NO_x, SO_2, and VOCs eastward over the Black Hills region and BADL, providing sufficient precursors for ozone formation during the summer when it is sunny and warm. The development of additional coal-fired power plants in eastern Montana and eastern Wyoming would increase emissions transported to BADL.

C. MONITORING AND RESEARCH ACTIVITIES

1. Air Quality

Table II-5 is a summary of current air quality monitoring being conducted in BADL.

a. <u>Wet Deposition</u>

The NADP/NTN site at Cottonwood (approximately 20 km northeast of BADL) is used to represent wet deposition at BADL. Wetfall chemistry data are available for this location since 1983. The data indicate no particular trends during this period of time (Table IX-1). Hydrogen ion (acidic) inputs are relatively low, which suggests that the acidity of wet deposition is not a concern in this area. Ionic deposition of N and S are considerably higher than that of other elements. Sulfate deposition is moderately high due to geological sources of this ion. Despite these ionic concentrations of sulfate, wet deposition of S averages only 1.1 kg/ha/yr, which is quite low (Table IX-2). Average total inorganic nitrogen deposition is only 2.1 kg/ha/yr, with approximately equal contributions from nitrate and ammonium. The values for S and N deposition, in combination with wetfall input of hydrogen and other ions, indicate that BADL is a relatively clean site and that there is no apparent threat from acidic deposition at the present time.

b. <u>Occult/Dry Deposition</u>

There are no dry deposition data available in the vicinity of BADL. Dry deposition is not expected to have a major impact on natural resources in the park.

Table IX-1. Wetfall chemistry at the NADP/NTN site near BADL. Units are in μeq/L, except precipitation (cm).

Year	Precip	H^+	SO_4^{2-}	NH_4^+	NO_3	Ca^{2+}	Mg^{2+}	Na^+	K^+	Cl^-
1995	47.6	5.6	12.2	16.8	15.9	8.2	1.4	2.1	0.5	1.7
1994	38.2	6.3	16.9	23.6	18.7	11.1	2.0	2.1	0.7	2.0
1993	58.9	3.0	16.1	22.1	16.5	8.8	1.8	2.6	0.8	2.0
1992	53.3	6.3	18.2	20.6	15.1	6.6	1.1	1.7	0.5	1.6
1991	62.7	2.7	13.7	19.1	14.5	11.1	1.8	2.1	0.6	2.7
1990	29.1	4.4	16.0	22.5	18.8	8.7	1.8	2.8	0.5	3.0
1989	29.4	3.7	16.3	18.7	16.8	10.4	2.0	2.7	0.6	2.5
1988	36.1	4.6	12.9	5.5	11.1	8.6	1.4	2.5	0.5	1.4
1987	31.7	4.6	17.6	23.6	17.7	7.4	1.8	2.7	0.7	2.2
1986	56.4	5.9	17.9	12.6	15.7	12.5	2.2	2.3	0.9	1.9
1985	26.3	3.5	18.4	17.5	17.2	14.2	3.5	2.6	1.4	3.2
1984	41.7	1.9	20.5	22.6	17.7	14.4	4.2	3.7	1.0	3.0
Average	42.6	4.4	16.4	18.8	16.3	10.2	2.1	2.5	0.7	2.3

Table IX-2. Wet deposition (kg/ha/yr) of sulfur and nitrogen at the NADP/NTN site near BADL.

Date	Sulfur	NO_3-N	NH_4-N	Total Inorganic N
1995	0.9	1.1	1.1	2.2
1994	1.0	1.0	1.3	2.3
1993	1.5	1.4	1.8	3.2
1992	1.6	1.1	1.5	2.7
1991	1.4	1.3	1.7	3.0
1990	0.8	0.8	0.9	1.7
1989	0.8	0.7	0.8	1.5
1988	0.8	0.6	0.3	0.8
1987	0.9	0.8	1.1	1.8
1986	1.6	1.2	1.0	2.2
1985	0.8	0.6	0.6	1.3
1984	1.4	1.0	1.3	2.4
Average	1.1	1.0	1.1	2.1

c. Gaseous Monitoring

BADL joined the air quality monitoring network in 1987 with the installation of a continuous ozone analyzer. Continuous ozone monitoring has since been discontinued and there is no monitoring of gaseous pollutants currently being conducted at the park (Table II-5). Passive ozone monitoring was initiated at one location in 1997.

Ozone was monitored at BADL from 1988 to 1992. Table IX-3 is a summary of ozone concentrations at BADL. As Table IX-3 indicates, the highest 1-hour ozone concentration measured at the park was 72 ppbv (1988). Ozone concentrations are similar to those found at GLAC, THRO, and YELL and significantly lower than those at ROMO. During the ozone monitoring period, BADL had some of the lowest average ozone concentrations in the NPS monitoring network and was one of only 10 NPS sites that maintained second-highest 1-hour ozone averages at or below 75 ppbv during 1987-1991 (Joseph and Flores 1993). Perhaps more importantly, with respect to damage to sensitive plant species, the mean daytime 7-hour ozone concentration during the growing season ranged between 38 and 45 ppbv during this time period. These levels are below those found to damage sensitive plant species. The SUM60 exposure index is another indicator that can be important in assessing ozone exposures of plant species. This index is the sum of all hourly ozone concentrations equaling or exceeding 60 ppbv. The SUM60 index at BADL ranged between 793 and 5,251. For comparison, National Parks in highly polluted areas (e.g., southern California) can have SUM60 exposure indexes exceeding 100,000 ppbv-hour (Joseph and Flores 1993).

Table IX-3. Summary of BADL ozone concentrations (ppbv) from NPS monitoring sites (Joseph and Flores 1993).

	1988	1989	1990	1991
1-hour maximum	72	71	63	66
Average daily mean	34	31	29	31
Growing season 7-hour mean	45	43	38	40
SUM60 exposure index (ppbv-hour)	5,251	3,079	793	738

The state of South Dakota also does not currently monitor gaseous pollutants. Between 1975 and 1986, gaseous monitoring was conducted by the South Dakota Department of Environment and Natural Resources using SO_2/NO_2 bubblers sampling every sixth day (since levels did not warrant continuous daily operation) at various locations around the state. Measured levels of SO_2 rarely exceeded the threshold limit of the instruments (1 ppbv). The data are on file with the South Dakota Air Monitoring Program.

The low SO_2 values at BADL measured by NPS-ARD (four months per year, Table IX-4) are in agreement with the values measured by the state and with the values measured at other national parks in the region indicating that SO_2 is not an issue of immediate concern in BADL.

Table IX-4. Maximum and mean SO_2 24-hour integrated sample for BADL. The clean air baseline is estimated to be 0.19 ppbv (Urone 1976). (Source*:* J. Ray, NPS Air Resources Division)						
SO_2 concentration (ppbv)	1988	1989	1990	1991	1992	1993
Maximum	3.64	2.04	1.02	0.62	0.60	2.35
Mean	0.10	0.14	0.16	0.13	0.11	0.38

2. Water Quality

Surface waters are expected to be well-buffered against acid inputs. EPA's STORET data base contains 1,971 pH measurements for southwestern South Dakota, in an area that includes BADL. Only one measurement showed pH less than 6.1, having a reported pH of 1.5, and it was either due to an error or to direct acid discharge (e.g., acid mine drainage). The first pH percentile was 6.6. In other words, 99% of the measured values were higher than pH 6.6. Similarly, 99% of the calcium measurements (n=236) were higher than 10 mg/L (500 μeq/L). Surface water sulfate concentrations tended to be very high due to geological sources of SO_4^{2-}. The median SO_4^{2-} concentration (n=77) in the data set was 88 mg/L (1,800 μeq/L). Thus, none of the available water-quality data suggest acid-sensitivity. There is no evidence to suggest that surface waters in BADL would be responsive to acidic deposition impacts. Acidic deposition, especially the low levels currently experienced at BADL, should not be a threat to water quality due to the high base cation content and pH of the water and soils of the region.

3. Terrestrial

No studies have been performed on the effects of air pollutants on the terrestrial resources of BADL. Park management acknowledges that baseline pollutant monitoring has been carried out to some degree and that the next step is to identify air pollution sensitive species, document their current status, and establish a monitoring plan.

4. Visibility

As part of the Interagency Monitoring of Protected Visual Environments (IMPROVE) network, visual air quality in BADL has been monitored using an aerosol sampler, transmissometer, and camera. The aerosol sampler has operated from March 1988 through the present and is located at park headquarters slightly south of the Cedar Pass Visitor Center. The transmissometer has operated from January 1988 through the present and is located at the park's northeast entrance. The 35mm camera operated from August 1987 through March 1995. The camera system was located at the Pinnacles Overlook, approximately 14 miles south of Wall, South Dakota, and viewed Sheep Mountain to the southwest. Data from this IMPROVE site have been summarized to characterize the full range of visibility conditions for the March 1988 through February 1995 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 1994 includes March 1994 through February 1995). 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

IMPROVE aerosol samplers consist of four separate particle sampling modules that collect 24-hour filter samples of the particles suspended in the air. The filters are then analyzed in the laboratory to determine the mass concentration and chemical composition of the sampled particles. Particle data can be used to provide a basis for inferring the probable sources of visibility impairment. Practical considerations limit the data collection to two 24-hour samples per week. (Wednesday and Saturday from midnight to midnight). Detailed descriptions of the aerosol sampler, laboratory analysis, and data reduction procedures used can be found in the draft 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. The extinction coefficient represents the ability of the atmosphere to scatter and absorb light. Higher extinction coefficients signify lower visibility. A tabular and graphic summary of average reconstructed extinction values by season and year for the March 1988 through February 1995 period are provided in Table IX-5 and Figure IX-1, respectively.

Reconstructed extinction budgets generated from aerosol sampler data apportion the extinction at BADL to specific aerosol species (Figure IX-2). 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% of days, mean of the median 20% of days, and mean of the dirtiest 20% of days. 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, visual range (in kilometers), and deciview (dv). Standard Visual Range (SVR) can be expressed as:

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

Table IX-5. Seasonal and annual average reconstructed extinction (Mm^{-1}) for BADL, March 1988 through February 1995.

Year	Spring (Mar, Apr, May)		Summer (Jun, Jul, Aug)		Autumn (Sep, Oct, Nov)		Winter (Dec, Jan, Feb)		Annual (Mar - Feb)[a]	
	b_{ext} (Mm^{-1})	SVR (km)	b_{ext} (Mm^{-1})	SVR (km)	b_{ext} (Mm^{-1})	SVR (km)	b_{ext} (Mm^{-1})	SVR (km)	b_{ext} (Mm^{-1})	SVR (km)
1988	47.7	82	55.6	70	36.7	107	38.5	102	45.4	86
1989	53.8	73	51.5	76	39.5	99	42.0	93	46.4	84
1990	50.9	77	48.3	81	40.1	98	38.5	102	44.1	89
1991	50.5	77	52.2	75	39.5	99	42.2	93	46.1	85
1992	55.2	71	46.7	84	43.7	90	59.0	66	51.0	77
1993	49.1	80	37.1	105	36.7	107	51.5	76	43.2	91
1994	45.4	86	48.6	80	44.1	89	45.6	86	45.8	85
Mean[b]	50.4	78	48.6	80	40.0	98	45.3	86	46.1[c]	85[3]

[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 1995 period.
[c] Combined annual period data represent the mean of all combined season means.

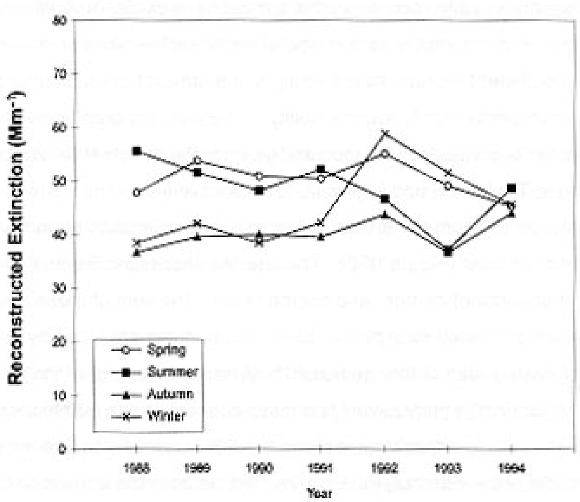


Figure IX-1. Seasonal average reconstructed extinction (Mm^{-1}) for BADL, March 1988 through February 1995.

Figure IX-1. Seasonal average reconstructed extinction (Mm^{-1}) for BADL, March 1988 through February 1995.

where b_{ext} is the extinction coefficient expressed in inverse megameters (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 a 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.

Deciview is defined as:

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

where b_{ext} is the extinction coefficient expressed in inverse megameters (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. The segment at the bottom of each stacked bar 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.

The 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. The measured extinction data are used to verify the calculated reconstructed extinction and can also be

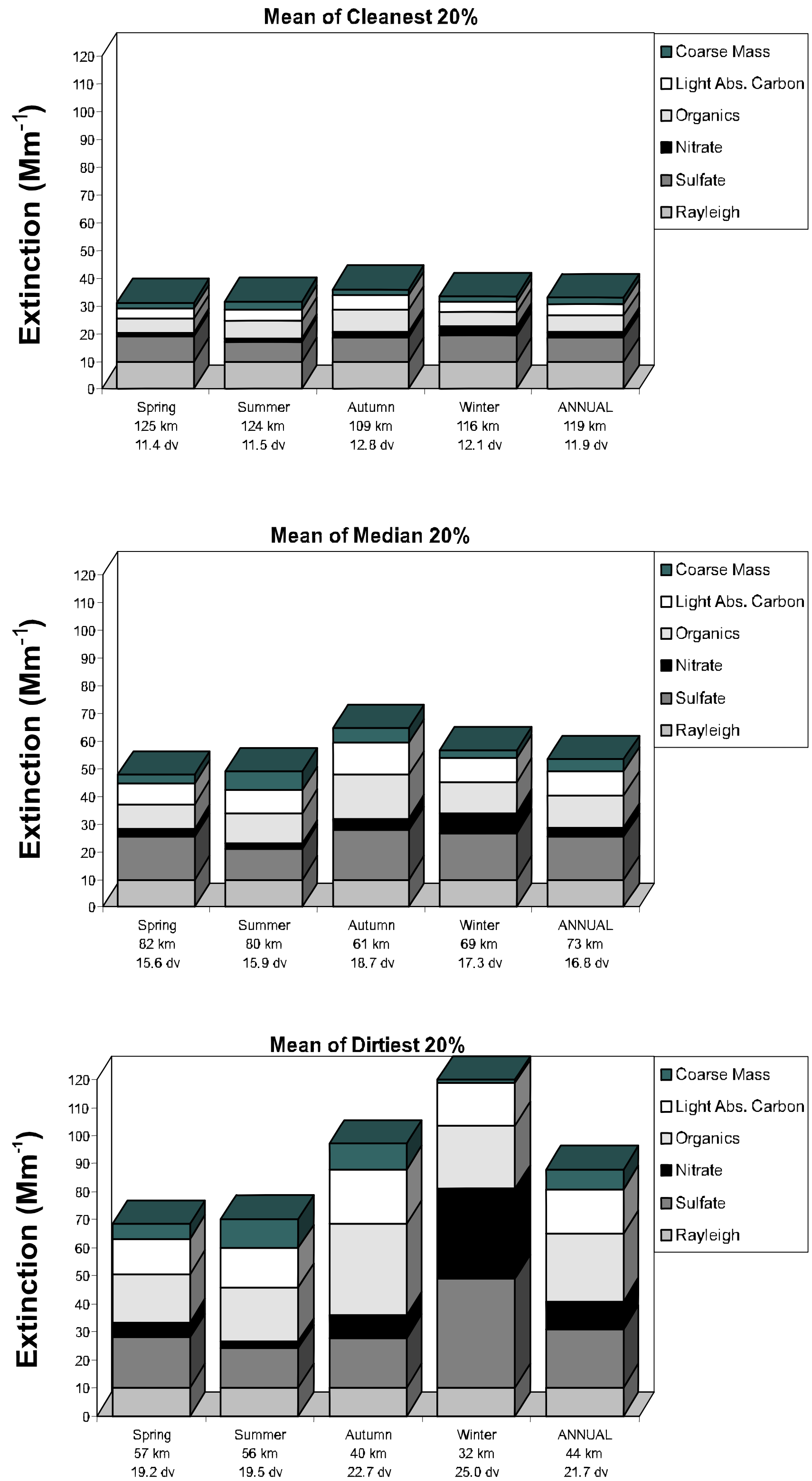


Figure IX-2. Reconstructed extinction budgets for BADL, March 1988 - February 1995.

used to run plume and regional haze models ant to analyze trends and conditions. Because of the larger spatial and temporal range of the aerosol data, reconstructed extinction data are preferred.

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. 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, 1993 and 1994).

Table IX-6 provides a tabular summary of the “filtered” seasonal and combined period arithmetic mean extinction values for the March 1988 through February 1995 period. Table IX-7 provides a tabular summary of the "filtered" seasonal and combined period 10% (clean) cumulative frequency values. 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 February 1995 period.
- Combined annual period data represent the unweighted mean of all combined seasonal b_{ext} values.

Figure IX-3 provides a graphic representation of the "filtered" annual mean, median, and cumulative frequency values (5th, 10th, 25th, 75th, 90th, and 95th percentiles). No data are reported for annual periods with one or more invalid seasons.

When comparing reconstructed (aerosol) extinction, Table IX-5 with measured (transmissometer) extinction, Table IX-6, 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.

Table IX-6. Seasonal and annual arithmetic means transmissometer data (filtered) for BADL, March 1988 through February 1995.

Year	Spring (Mar, Apr, May)			Summer (Jun, Jul, Aug)			Autumn (Sep, Oct, Nov)			Winter (Dec., Jan. Feb.)			Annual (Mar – Feb)[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
1989	92	43	14.6	–	–	–	129	31	11.3	114	35	12.5	***	***	***
1990	90	44	14.8	92	43	14.6	111	36	12.8	121	33	11.9	102	39	13.6
1991	105	38	13.4	78	51	16.3	108	37	13.1	125	32	11.6	101	40	13.7
1992	92	43	14.6	73	54	16.9	92	43	14.6	111	36	12.8	90	44	14.8
1993	105	38	13.4	129	31	11.3	129	31	11.3	125	32	11.6	121	33	11.9
1994	84	47	15.5	65	61	18.1	95	42	14.4	143	28	10.3	89	45	14.9
Mean[b]	94	42	14.4	83	48	15.7	109	37	13.0	122	33	11.8	120	33[c]	12.0

--No data are reported for seasons with <50% valid data.

*** No annual data are reported for periods with one or more invalid seasons.

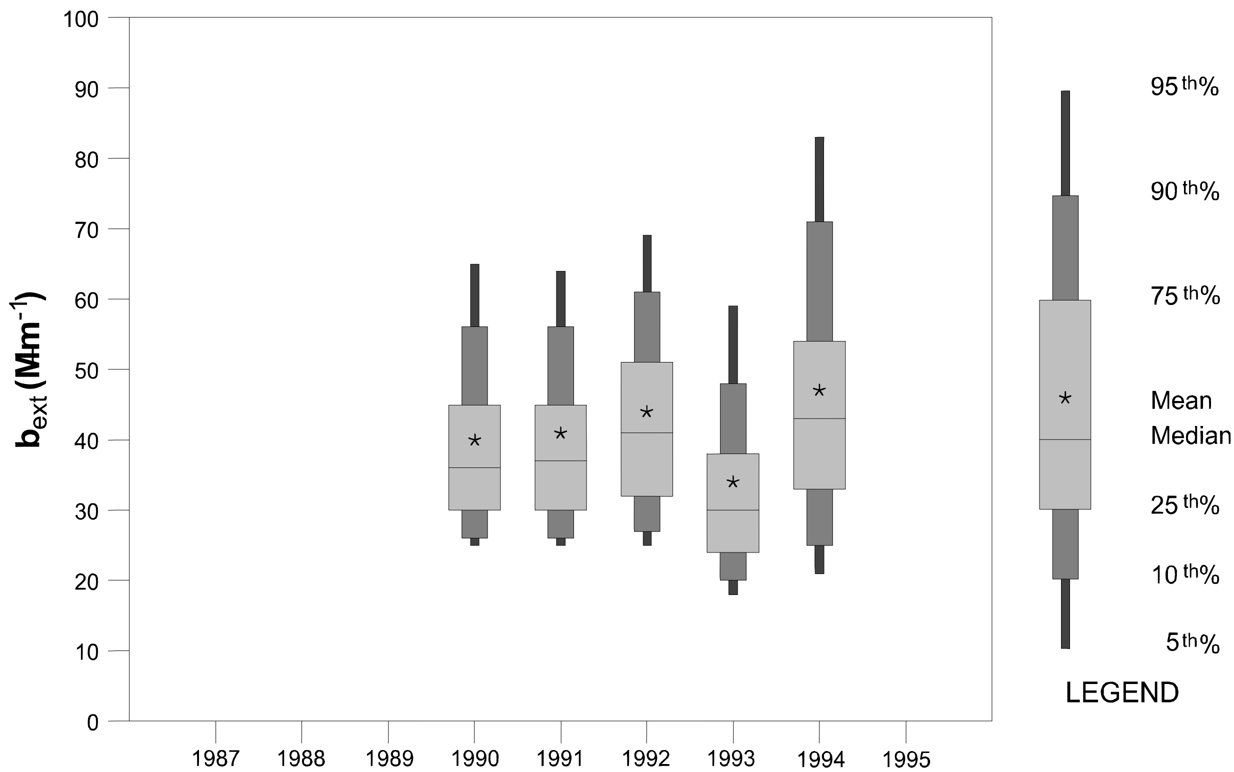
[a] Annual period data represent the mean of all valid seasonal b_{ext} means for each March through February annual period.

[b] Combined season data represent the mean of all valid seasonal b_{ext} means for each season of the March 1988 through February 1995 period.

[c] Combined annual period data represent the mean of all combined seasonal b_{ext} means.

Table IX-7 Seasonal and annual 10% (clean) cumulative frequency statistics transmissometer data (filtered) for BADL, March 1988 through February 1995.

Year	Spring (Mar, Apr, May)			Summer (Jun, Jul, Aug)			Autumn (Sep, Oct, Nov)			Winter (Dec., Jan. Feb.)			Annual (Mar – Feb)[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	–	–	–	–	–	–	255	16	4.7	239	17	5.3	***	***	***
1989	192	21	7.4				202	20	6.9	191	25	9.2	***	***	***
1990	143	28	10.3	143	28	10.3	154	26	9.6	161	25	9.2	150	27	9.8
1991	148	27	9.9	105	38	13.4	161	25	9.2	154	26	9.6	138	29	10.6
1992	133	30	11.0	97	41	14.1	143	28	10.3	192	22	7.9	132	30	11.1
1993	175	23	8.3	225	18	5.9	183	22	7.9	239	17	5.3	202	20	6.9
1994	125	32	11.6	97	41	14.1	154	26	9.6	225	18	5.9	132	29	10.7
Mean[b]	149	27	9.9	120	33	12.0	173	23	8.5	188	21	7.6	210	19[c]	6.6



Note: For a specific year to be plotted, at least 50% of the data for each season must be valid.

-- 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 data represent the mean of all valid seasonal b_{ext} values for each March through February annual period.

[b] Combined season data represent the mean of all valid seasonal b_{ext} values for each season of the March 1988 through February 1995 period.

[c] Combined annual period data represent the mean of all combined seasonal b_{ext} values.

Figure IX-3. Annual arithmetic mean and cumulative frequency statistics, BADL, South Dakota, 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.

- Relative Humidity (RH) Cutoff - Daily average reconstructed measurements are flagged as invalid when the daily average RH is greater than 98%. Hourly average transmissometer measurements are flagged invalid when the hourly average RH is greater than 90%. These flagging differences often result in data sets that do not reflect the same period of time, or misinterpret short-term meteorological conditions.

  Note: The weather algorithm only flags 10%-20% of the data for a majority of the sites west of the Mississippi River. RH cutoffs have little effect on final mean extinctions in the western United States.

Reconstructed extinction is typically 70%-80% of the measured extinction. With a ratio of 72%, this relationship shows good agreement for BADL.

c. Camera Data - View Monitoring

Color 35mm slide photographs of Sheep Mountain were taken three times per day. 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 Sheep Mountain photographs presented in Figure IX-4 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. Visibility Summary

Data from other IMPROVE visibility sites around the country have been presented graphically (Figures I-1 and I-2) so that visual air quality in the Rocky Mountains and northern Great Plains regions can be understood in perspective. Figure IX-6 and Table IX-6 have been provided to summarize BADL visual air quality during the March 1988 through February 1995 period. Seasonal variance in the mean of the dirtiest 20% fractions are driven primarily by sulfate and nitrate extinctions. Long-term trends fall into three categories: increases, decreases, and variable. Given the visibility sites summarized for this report, the majority of data show little change or trends.

Non-Rayleigh atmospheric light extinction at BADL is largely due to sulfate, organics, nitrates, and soil. 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 (Grand Canyon Visibility Transport Commission 1996). Smoke from frequent fires is suspected to have reduced pre-settlement visibility below current levels during some summer months.

D. AIR QUALITY RELATED VALUES

1. Surface Waters

Surface waters at BADL are mostly ephemeral in nature, and most of the small pools of water remaining after rainfall contain a large amount of clay in colloidal suspension. These small pools are bioindicators for the potential impacts of acidic deposition. Acid neutralizing capacity (ANC) is the best indicator of buffer capacity against acidic input. Although there are no data on ANC for the pools at BADL, the high clay content suggests that ANC would be very high, with no reasonable possibility of current levels of deposition reducing ANC measurably or of affecting aquatic life in the pools.

Badlands National Park
on a "clear" day

Representative Conditions:
Visual Range: 320-380 km
b_{ext}: 12-10 Mm^{-1}
Haziness: 2-0 dv

Badlands National Park
on an "average" day

Representative Conditions:
Visual Range: 100-130 km
b_{ext}: 39-30 Mm^{-1}
Haziness: 14-11 dv

Badlands National Park
on a "dirty" day

Representative Conditions:
Visual Range: 60-80 km
b_{ext}: 65-49 Mm^{-1}

Haziness: 19-16 dv

Figure IX-4. Photographs illustrating visibility conditions at Badlands National Park.

Figure IX-4. Photographs illustrating visibility conditions at Badlands National Park.

2. Vegetation

Vegetation is the resource which is most sensitive to ozone and SO_2, and several tree species have been identified as potential biodindicators (see below). Additional studies are needed to evaluate the impact of SO_2 and ozone on terrestrial ecosystems in BADL. While ozone and SO_2 levels at BADL have not exceeded the NAAQS, it is possible that there may be subtle effects under current conditions; increased concentrations in the future could potentially damage sensitive plant species. Furthermore, baseline data on the condition of sensitive species in the absence of injurious pollutants will be helpful for comparison with future conditions if pollutant levels increase. Monitoring bioindicators by using detailed descriptions and classifications of leaf or plant injury assists in the long-term evaluation of ecosystem health.

Bioindicators exist for which pollutant sensitivity has been documented and for which extensive data exist on their dose-response relationship to pollutants and on symptomatology. In some cases, these species can be important indicators of 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. Much of this work has been conducted on species native to the northeastern and southwestern United States and very little work on air pollutant effects has been conducted on species of the Rocky Mountains and northern Great Plains.

Although ponderosa pine is not common in BADL, it is present in the Sheep Mountain area on approximately 80 ha. Ponderosa pine is one of the most ozone-sensitive western tree species (especially var. *ponderosa*) for which extensive data are available on field (Miller and Millecan 1971, Pronos and Vogler 1981, Peterson and Arbaugh 1988) and experimental (Temple et al. 1992) exposures. The evidence for ozone impacts on ponderosa pine is based on observable symptoms of foliar chlorosis and reduced growth (Peterson et al. 1991, Peterson and Arbaugh 1992) as well as physiological (Darrall 1989, Bytnerowicz and Grulke 1992) data. The cause-and-effect relationship, especially for trees growing in forests of southern California and the southern Sierra Nevada, is clear and quantifiable. The Rocky Mountain variety of ponderosa pine (var. *scopulorum*) is known to be somewhat more tolerant to ozone and has a higher threshold for symptoms of injury under experimental exposures than var. *ponderosa* (Aitken et al. 1984).

Of the hardwood species present at BADL, green ash is the most sensitive to ozone. Green ash grows in riparian areas of BADL and is more common in the North Unit. Diagnostic ozone symptomatology for green ash includes dark pigmented stippling and bifacial interveinal necrosis (NAPAP, undated). Green ash does not have the clarity of ozone symptomatology found in ponderosa pine but can be used as a secondary bioindicator.

Green ash is also sensitive to SO_2 and may be the best bioindicator for this gaseous pollutant since SO_2 injury in conifer species is difficult to diagnose. There may be species present at BADL that are more sensitive to SO_2, but until such time as there is sufficient information on their dose-response relationship and symptomatology, green ash should be used as a bioindicator. Species with determinate terminal growth, such as green ash, are more sensitive to SO_2 early in the summer. There could be some confusion of ozone injury and SO_2 injury.

An inventory of vascular plants found in BADL was compiled in 1988 and is available in the NPFlora database. Table IX-8 summarizes vascular plant species of BADL with known sensitivity to ozone, SO_2, and NO_x. This table is based on a variety of sources from the published literature and other information. It should be noted that the various sources used a wide range of field and experimental approaches to determine pollutant pathology, and that sensitivity ratings are general estimates based on published information and our expert opinion. While park staff will not be able to collect data on all the species indicated in Table IX-8, the list can be used by park managers to indicate potentially sensitive species. Of the many plant species in BADL, it is likely that there are many other species which have high sensitivity to air pollution, but we currently have no information about them.

Table IX-8. Plant species of BADL with known sensitivities to SO_2, ozone, and NO_x. L = low, M = medium, H = high, none = unknown. (Sources: Esserlieu and Olson 1986, Bunin 1990, Peterson et al. 1993, National Park Service 1994, Electric Power Research Institute 1995, Binkley et al. 1996)

Species Name	SO_2 Sensitivity	O_3 Sensitivity	NO_x Sensitivity
Acer negundo	M	M	
Achillea millefolium		L	
Agoseris glauca	M		
Agropyron smithii	M		
Amaranthus retroflexus	M		
Ambrosia psilostachya		L	
Artemisia ludoviciana	M		
Atriplex canescens	L		
Atriplex confertifolia	L		
Bouteloua gracilis	L		
Bromus tectorum		M	
Chrysothamnus nauseosus	M		
Cirsium arvense		L	
Clematis ligusticifolia	M		
Collomia linearis		L	
Convolvulus arvensis	H		
Crataegus succulenta	L		
Descurainia pinnata		L	
Festuca octoflora		L	
Fraxinus pennsylvanica	M	H	
Gutierrezia sarothrae	M		
Hackelia floribunda	L		
Helianthus anuus	H	L	
Juniperus scopulorum	L		
Medicago sativa		M	
Oryzopsis hymenoides	M		

Table IX-8. Continued.

Species Name	SO_2 Sensitivity	O_3 Sensitivity	NO_x Sensitivity
Phlox hoodii	L		
Pinus ponderosa	M	H	H
Poa pratensis	M	M	
Populus deltoides	M	L	
Potentilla fruticosa		L	
Prunus virginiana	M	H	
Rhus trilobata	L		
Ribes americanus	M		
Rosa woodsii	M	L	
Spartina pectinata	M		
Symphoricarpos albus		H	
Taraxacum officinale		L	
Toxicodendron radicans	L	L	
Tragopogon dubius	M		
Trifolium pratense	L		
Ulmus americana	M		
Vicia americana		L	
Yucca glauca	L		

Table IX-9 summarizes lichen species of BADL with known sensitivity to ozone and SO_2. As in Table IX-8, this table is based on a variety of sources from the published literature and other information. It should be noted that diagnostic symptoms of air pollutant injury to lichens are difficult to identify, and that some species have reduced productivity or even mortality without exhibiting visible symptoms (Nash and Wirth 1988). One of the best sources of background information and guidelines for addressing the use of lichens as bioindicators of air pollution is Stolte et al. (1993).

Table IX-9. Lichen species of BADL with known sensitivities to SO_2 and ozone. L = low, M = medium, H = high, none = unknown. (Sources: Peterson et al. 1993, Electric Power Research Institute 1995, Binkley et al. 1996, Will-Wolf 1997)

Species	Ozone sensitivity	SO_2 sensitivity
Acarospora chlorophana		H
Buellia punctata		L-M
Caloplaca cerina		M-H
Caloplaca flavorubescens		H
Caloplaca holocarpa		M
Candelaria concolor		M-H
Candelariella vitellina		M
Cladonia chlorophaea		M
Cladonia fimbriata		M
Collema tenax		M
Hyperphyscia adglutinata		M
Lecanora chlarotera		M

Table IX-9. Continued		
Species	Ozone sensitivity	SO_2 sensitivity
Lecanora dispersa		L
Lecanora hageni		L
Lecanora muralis		M
Melanelia exasperatula		M
Melanelia subaurifera	H	
Ochrolechia androgyna		H
Parmelia sulcata	M-H	L-H
Peltigera canina	H	L
Peltigera didactyla	H	
Phaeophyscia ciliata	M	
Phaeophyscia nigricans		L-M
Phaeophyscia orbicularis		M
Physcia adscendens		M
Physcia aipolia		M
Physcia millegrana		M
Usnea hirta		M-H
Xanthoria elegans		M
Xanthoria fallax		M-H
Xanthoria polycarpa	L	M

E. RESEARCH AND MONITORING NEEDS

1. Deposition and Gaseous Monitoring

The NADP data available from the Cottonwood site 20 km from BADL should be adequate to characterize wet deposition at BADL. If park managers want better quantification of deposition at BADL, short-term collection of wet deposition data could be used to calibrate with the Cottonwood site and establish a reference for deposition of S and N. Short-term collection of dry deposition data using NDDN-type collectors would further establish this reference for deposition, but is probably not warranted at the present time. If additional sources of pollutants (e.g., large sources such as power plants, small sources such as oil wells), are developed in the vicinity of the park, long-term deposition data for the park would help identify potential trends in emissions associated with those sources.

A better spatial characterization of ozone distribution at BADL would be useful, although levels are currently below those believed to adversely affect sensitive plant species. A network of passive ozone samplers could be established to compare ozone measurements from different locations in the park. Three samplers in the North Unit and three in the South Unit should be sufficient to spatially characterize the ozone distribution, with weekly samples for two months during the summer. Another sampler in the North Unit could be situated in the Badlands Wilderness with the third located equidistant between the first two. Samplers in the South Unit could be situated in (1) Palmer Creek Unit, (2) western Stronghold Unit, and (3) eastern Stronghold Unit. Samplers should be situated where they are reasonably accessible but not within 50 m of a road or trail where they may be subject to excessive dust or vandalism. Two years of monitoring should be sufficient to establish spatial patterns and a reference point in time.

Operation of an SO_2 analyzer at the Park Headquarters would provide a better characterization of SO_2 deposition at BADL. Two years of monitoring should be sufficient to establish a reference point for SO_2. Because the state of South Dakota does not have an active SO_2 monitoring program, BADL should document any future changes in air quality.

2. Terrestrial Systems

Although the park has passed its fiftieth anniversary and is the primary representative of the prairie biogeographic region in the National Park System, a basic inventory of many natural resources is still in its infancy (BADL 1994). Monitoring schemes, short and long term, are similarly lacking, making it impossible to compare data over time to predict or assess changes. If the concentrations of ozone, SO_2, or NO_2 increase in the future, it will be important to document the condition of terrestrial resources of BADL so that a reference point in time is established.

If pollutant levels increase in the future, monitoring of terrestrial resources should be considered. If monitoring is implemented, we recommend that ponderosa pine and green ash be used as bioindicators for ozone and green ash for SO_2 at BADL. One plot of each species should be sufficient for initial monitoring. Three levels of monitoring associated with increasing amounts of effort and expense are detailed in Appendix A. Co-location of these plots with the passive ozone samplers should be done wherever possible. Monitoring should follow the methodology developed by U.S. Forest Service and National Park Service scientists for evaluating pollutant injury (Stolte and Miller 1991, Stolte et al. 1992).

If herbaceous species and lichens are included in a future monitoring effort, plots should be established adjacent to the tree plots if possible. Monitoring methodologies can be found in Appendix A. Inventories of lichen species distribution and abundance in BADL would provide a better baseline for further assessment of potential impacts of air pollution on lichens in the Park.

3. Visibility

IMPROVE aerosol and optical monitoring should continue at BADL. Ongoing and future monitoring is necessary to identify local source impacts. Additional data and in-depth modeling and analysis are required to further evaluate historical trends and projections of impact from existing and future sources. For example, back trajectory analysis and spatial/temporal pattern analysis of episodes are recommended to determine the source region contributions to elevated aerosol concentrations. Future research is also recommended to minimize the uncertainty in estimates of how various aerosol species affect visibility.

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XI. APPENDIX A -- TERRESTRIAL MONITORING METHODOLOGIES

Insufficient information exists on the ozone sensitivity of plant species native to national parks, making it difficult to estimate the current and long-term impact of regional ozone on terrestrial resources. Similarly, little is known of the impact of NO_2 and SO_2 on native vegetation. Studies can be conducted to identify sensitive receptors from each national park that may be at risk from exposure to ambient pollutant levels. Physiological and dose-response (controlled exposure) studies are expensive and time consuming but provide reasonable quantitative results. Candidate species for exposure studies should be those which are common and widespread. They should be species which are not prone to chronic fungal or other pathogen-caused foliar diseases that make field identification of pollutant injury difficult. From controlled exposure studies, detailed descriptions and photographs of foliar injury can be compiled into a guide for resource managers to use in the field.

In the absence of park-specific information on sensitive receptors, the current literature must be relied upon to provide candidate species. For each park, appropriate sensitive receptor species have been identified, based on the literature. Specific species and recommended locations for monitoring are presented in the individual park chapters in the “Research and Monitoring” sections. We propose that these species be monitored at each park according to the following methodologies. We propose three levels of monitoring, with increasing amounts of effort and expense. These monitoring activities are based on methods and protocols developed by the USDA Forest Service and NPS.

Level 1

Monitoring at this level consists of basic measurements and observations used to evaluate (1) ozone symptoms in trees, and (2) potential changes in lichen populations. Potential changes to herbaceous plants are also important, but there is far more information currently available for measuring pollutant impacts on conifers and lichens.

Level 1 monitoring of trees consists of repeated, standardized measurements of trees in permanent plots. It follows the methodology developed by Forest Service and National Park Service scientists for evaluating pollutant injury (Stolte and Miller 1991, Stolte et al. 1992). Monitoring of conifers involves identification of fifteen trees within a 20x100 m plot, at sufficient distance from roads, trails, or other potential impacts. Each tree should be at least 10 cm diameter at breast height (dbh) and be free of significant crown or stem defects. There should be a prunable live crown within 10 m of the ground.

Variables that are directly or indirectly affected by air pollutants are measured at the foliage, branch, and tree level. Foliage and branch level variables are measured from five pruned branches from the lower crown of trees as follows (Stolte et al. 1992) : (1) percent of foliar surface area with chlorotic mottle caused by pollutants, (2) percent surface area with necrosis, (3) percent surface area with other injury (biotic and abiotic), (4) number of internodes with live needles per branch (number of years of needles retained), (5) percent of needles remaining per stem internode with needles, (6) modal needle length per stem internode with needles. Tree level variables are (Stolte et al. 1992): (1) percent live crown, (2) crown density (upper and lower crown), (3) dbh.

Measurements should be conducted at the same time each year, preferably in late August when there is maximum opportunity for symptoms to have developed. Surveys should preferably be conducted every year. Observers should be trained by someone with expertise in air pollutant pathology in order to be able to identify and quantify pollutant effects (National Acid Precipitation Assessment Program, undated).

Variables that are measured can be evaluated individually or combined to calculate an index of the physiological status of a tree (Stolte and Miller 1991). We recommend assessing each variable individually and analyzing these data for individual trees. Plot means and other statistics can be calculated if desired. The following factors are most important with respect to primary effects of pollutants (primarily ozone) on conifers (Stolte et al. 1992): (1) chlorotic mottle -- the appearance of these symptoms reduces photosynthesis and production, (2) needle retention -- increased senescence of needles and reduced numbers of years of needles retained reduces the amount of carbon fixation, (3) percent live crown -- live crown ratio is reduced as lower branches die first in affected trees, (4) needle length -- reduced length of emergent needles may indicate carbon reserves are becoming limiting. These factors should be emphasized with respect to physiological status of trees regardless of how measurements are analyzed.

Lichens are regarded by many as potential bioindicators of SO_2 and ozone injury (Nash and Wirth 1988), although it is difficult to diagnose and measure the relationship between pollutant exposure and lichen injury in the field. It is recommended that periodic lichen surveys (perhaps every five years) be conducted to determine the total lichen flora at any given park. Site locations should be recorded for particularly rare species. The flora can then be compared to older records of lichen species, including local herbaria records. If the modern flora has fewer species, this may suggest the loss of sensitive species due to air-pollution stress (especially ozone) (Sigal and Nash 1983). While not diagnostic of pollution stress, this approach may identify species which may be at risk. A large-scale monitoring effort for lichens is not justified at this point, although protocols and guidelines in Stolte et al. (1993) can be consulted for information on assessing injury.

Lichen monitoring consists of two separate efforts. First, ten 2 m^2 plots are established along a transect at each site. All lichen species present are recorded annually. In addition, 10 individuals of a lichen species sensitive to ozone should be marked at each site. The dimensions and morphological characteristics of each individual should be recorded annually. It is recommended that caespitose (shrublike) fruticose lichens and large, loosely attached and suberect foliose lichens be used for monitoring, because it is easier to measure growth changes in these species over time. It is strongly recommended that lichen monitoring be established after consulting protocols and guidelines in "Lichens as Bioindicators of Air Quality" (Stolte et al. 1993) in cooperation with someone trained in lichen taxonomy and ecology.

Level 2

Level 2 monitoring consists of the two permanent tree plots discussed in Level 1, as well as three additional plots. A lichen survey should be conducted as in Level 1.

Level 3

Level 3 monitoring consists of the plots established in Level 2, as well as additional plots along transects that evaluate other species. A lichen survey should be conducted as in Level 1.

Monitoring of herbaceous plants is recommended at this level. In the absence of information on the sensitivity of species to pollutants, we recommend that monitored species have the following characteristics: perennial, relatively common, distributed over a range of ecosystem types (wide elevation range), and relatively large leaves. Herbaceous monitoring plots should include at least 20 plants of a given species within a 10x50 m plot. Plant characteristics measured should include (1) percent of foliar surface area with chlorotic mottle or stippling caused by pollutants, (2) percent surface area with necrosis, (3) percent surface area with other injury (biotic and abiotic). Only leaves on the upper part of plants should be measured. These measurements should be made annually for at least five years, preferably in early August; measurement should always be conducted prior to the onset of seasonal leaf senescence that would obscure other foliar characteristics.

ACKNOWLEDGMENTS

This report was funded through a cooperative agreement between the National Park Service and the University of Washington. The project coordinator was Tonnie Maniero, National Park Service Air Resources Division. The authors thank Ms. Maniero and staff with the ARD in Denver, Colorado, for providing data and pertinent information on the parks and constructive comments at various stages in the project. Resource managers and librarians in the parks were very helpful in reviewing park activities related to air pollution studies and in providing existing data and comments on the draft report. The principal park contacts were Jeff Connor and Craig Axtell (ROMO); Mason Reid and Bob Schiller (GRTE); Mary Hektner and Stu Coleman (YELL); Bill Michels, Lindy Key, and Jack Potter (GLAC); Roger Andrascik (THRO); Ross Rice (WICA); and Bruce Bessken (BADL). Dr. Kathy Tonnessen was instrumental in providing data, facilitating contacts, and providing comments on drafts of the report. Dee Morse coordinated preparation and revisions of the visibility sections of the report.

We are indebted to a large number of scientists and agency officials throughout the region for sharing databases and information on work in progress and for providing comments on draft materials. In particular, we wish to thank J. Baron, T. Blett, D. Campbell, E. Compaso, T. Dotts, S. Fullerton, D. Harmon, J. Ray, J. Sueker, R. Thomas, J. Turk, and M. Williams.

The following E&S employees provided support for the project: Jayne Charles and Jennifer Davis, word processing; Susan Binder, preparation of figures; and Kellie Vaché, Joseph Bischoff, and Joseph Bernert, data analysis.

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
BADL	Badlands National Park
b_{abs}	Light absorption coefficient
b_{ext}	Light extinction coefficient
BLM	Bureau of Land Management
b_{Ray}	Rayleigh extinction coefficient
Ca	Calcium
C_B	Base cations
Cl	Chloride
CO	Carbon monoxide
DOC	Dissolved organic carbon
EMAP	Environmental Monitoring and Assessment Program
EPA	Environmental Protection Agency
GIS	Geographic information system
GLAC	Glacier National Park
GRTE	Grand Teton National Park
H	Hydrogen
HCO_3	Bicarbonate
HF	Hydrogen fluoride
H_2S	Hydrogen sulfide
IAP	Integrated Assessment Report
IMPROVE	Interagency Monitoring of Protected Visual Environments
K	Potassium
LAC	Limit of acceptable change
MAGIC	Model of Acidification of Groundwater in Catchments
MERLIN	Model of Ecosystem Retention and Leaching of Inorganic Nitrogen
Mg	Magnesium
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
NDSDH	North Dakota State Department of Health
NH_3	Ammonia
NH_4-N	Ammonium
NO	Nitric oxide
NO_2	Nitrogen dioxide
NO_3-N	Nitrate
NO_X	Nitrogen oxides
NPS	National Park Service
O_3	Ozone
P	Phosphorus
PM-10	Particulate matter less than 10 μm diameter
PM-2.5	Particulate matter less than 2.5 μm diameter

PSD	Prevention of significant deterioration
RH	Relative humidity
ROMO	Rocky Mountain National Park
S	Sulfur
SFU	Stacked filter unit
SIP	State implementation plan
SO_2	Sulfur dioxide
SO_4	Sulfate
STORET	Storage and Retrieval Database (EPA)
SVR	Standard visual range
THRO	Theodore Roosevelt National Park
USFS	U.S. Forest Service
USFWS	U.S. Fish and Wildlife Service
VMT	Vehicular miles traveled
VOC	Volatile organic compound
VR	Visual range
VWM	Volume-weighted mean
WICA	Wind Cave National Park
YELL	Yellowstone National Park

<u>Units</u>

cfs	cubic feet per second
cm	centimeter
dv	deciview
ha	hectare
kg	kilograms
km	kilometer
L	liter
m	meter
meq	milliequivalent
mg	milligram
Mm^{-1}	inverse megameters
ppbv	parts per billion by volume
ppmv	parts per million by volume
yr	year
μeq	microequivalents
μg	micrograms
μm	micrometers
μS	microsiemens

LIST OF FIGURES

LIST OF TABLES

As the nation's principal conservation agency, the Department of the Interior has responsibility for most of our nationally owned public lands and natural and cultural resources. This includes fostering wise use of our land and water resources, protecting our fish and wildlife, preserving the environmental and cultural values of our national parks and historical places, and providing for enjoyment of life through outdoor recreation. The Department assesses our energy and mineral resources and works to ensure that their development is in the best interests of all our people. The Department also promotes the goals of the Take Pride in America campaign by encouraging stewardship and citizen responsibility for the public lands and promoting citizen participation in their care. The Department also has a major responsibility for American Indian reservation communities and for people who live in island territories under U.S. administration.

NPS D-657 August 1998