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

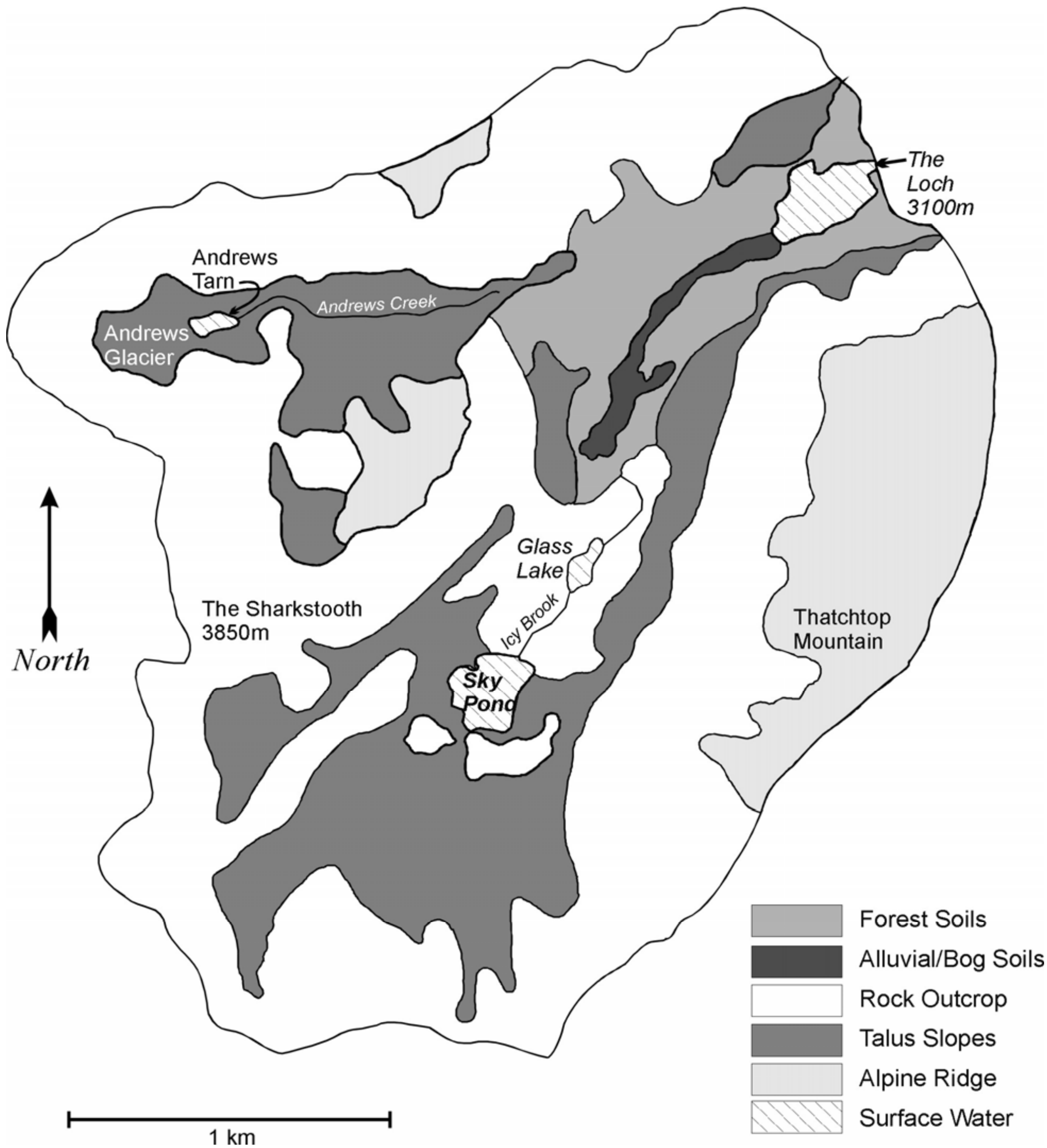


Figure III-1. Soils map of the Loch Vale watershed (from Walthall 1985).

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

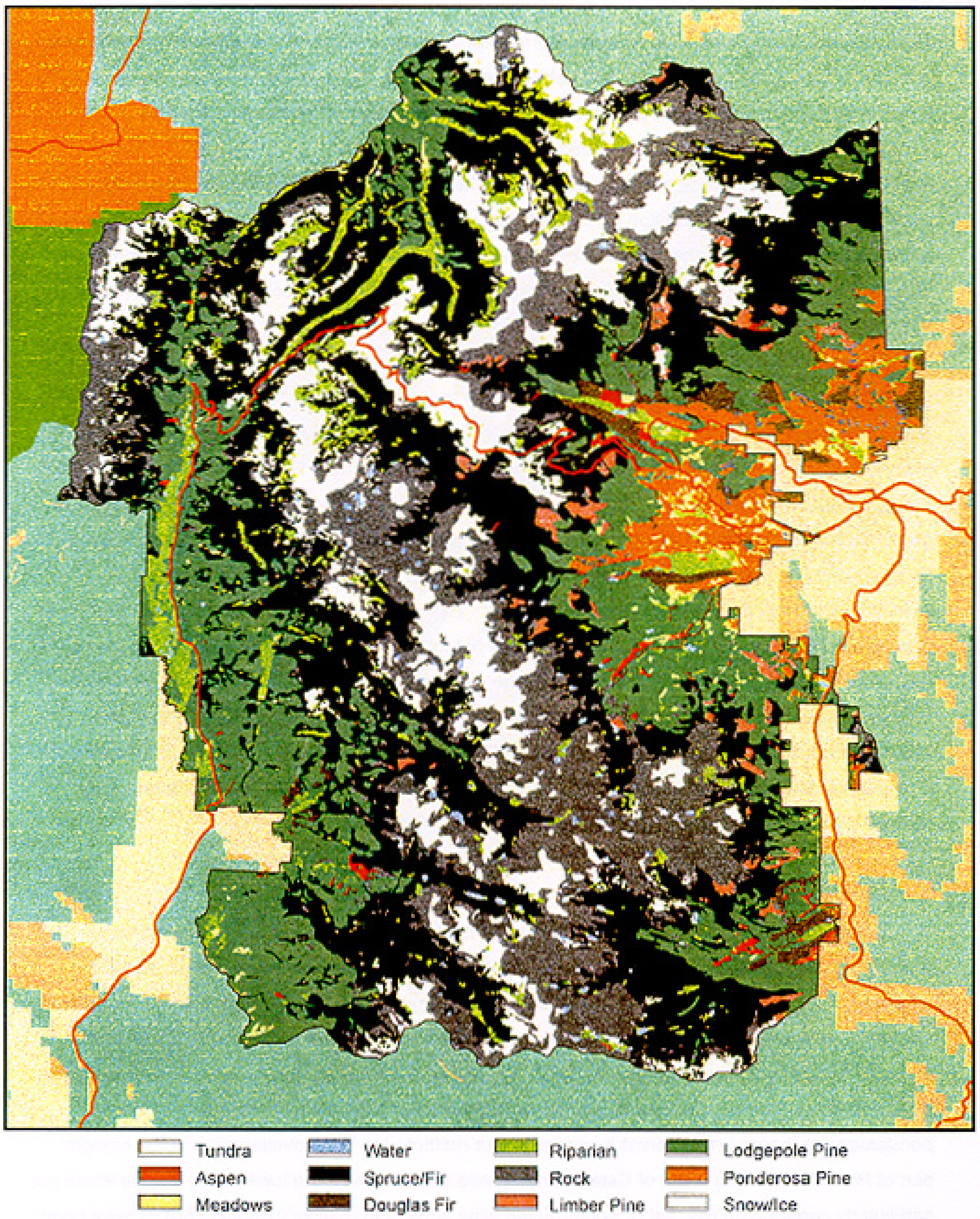


Figure III-2. Vegetation classification map of ROMO, from park database.

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. Wet deposition

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^{o}C) 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

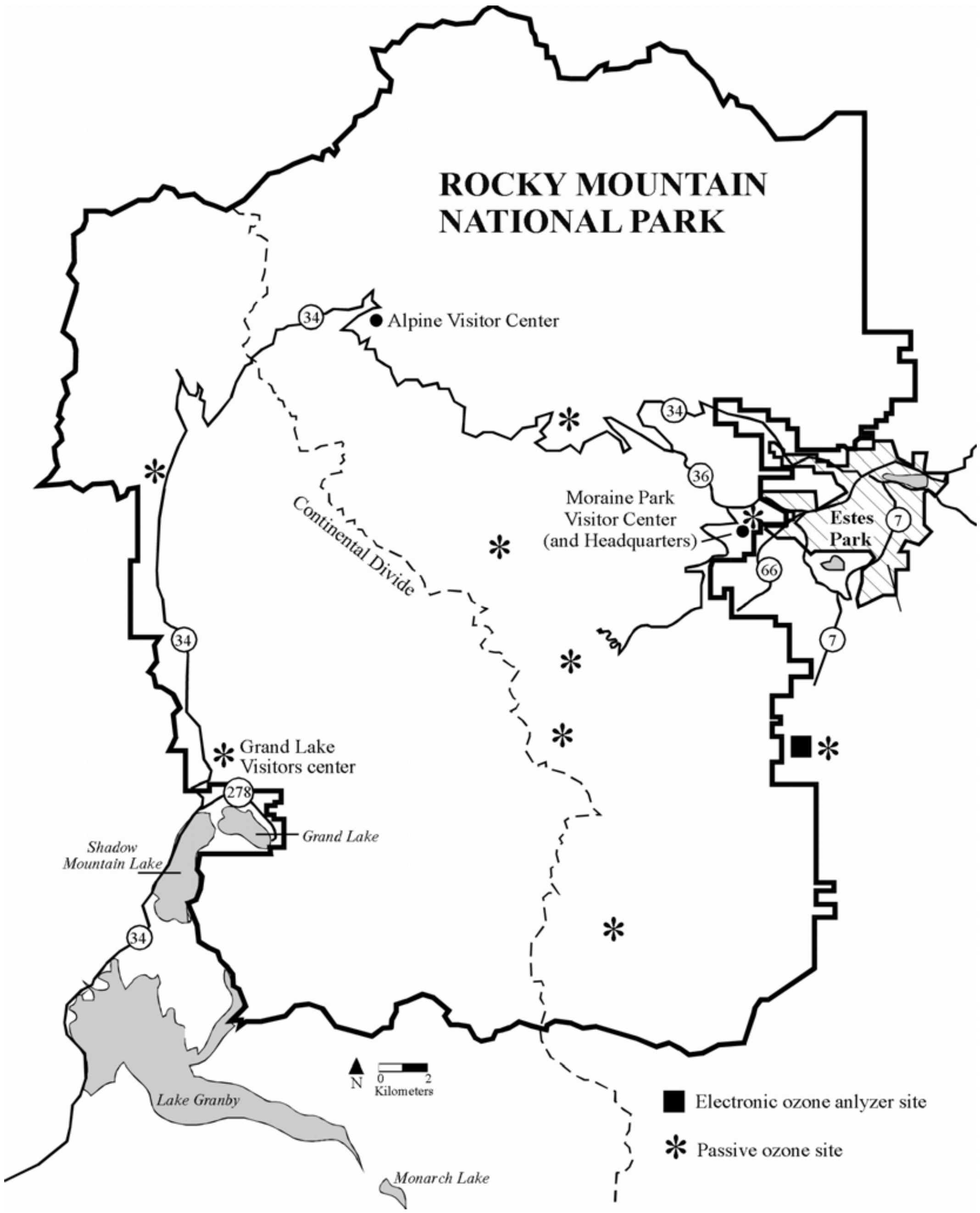


Figure III-3. Map of ROMO and air quality monitoring sites.

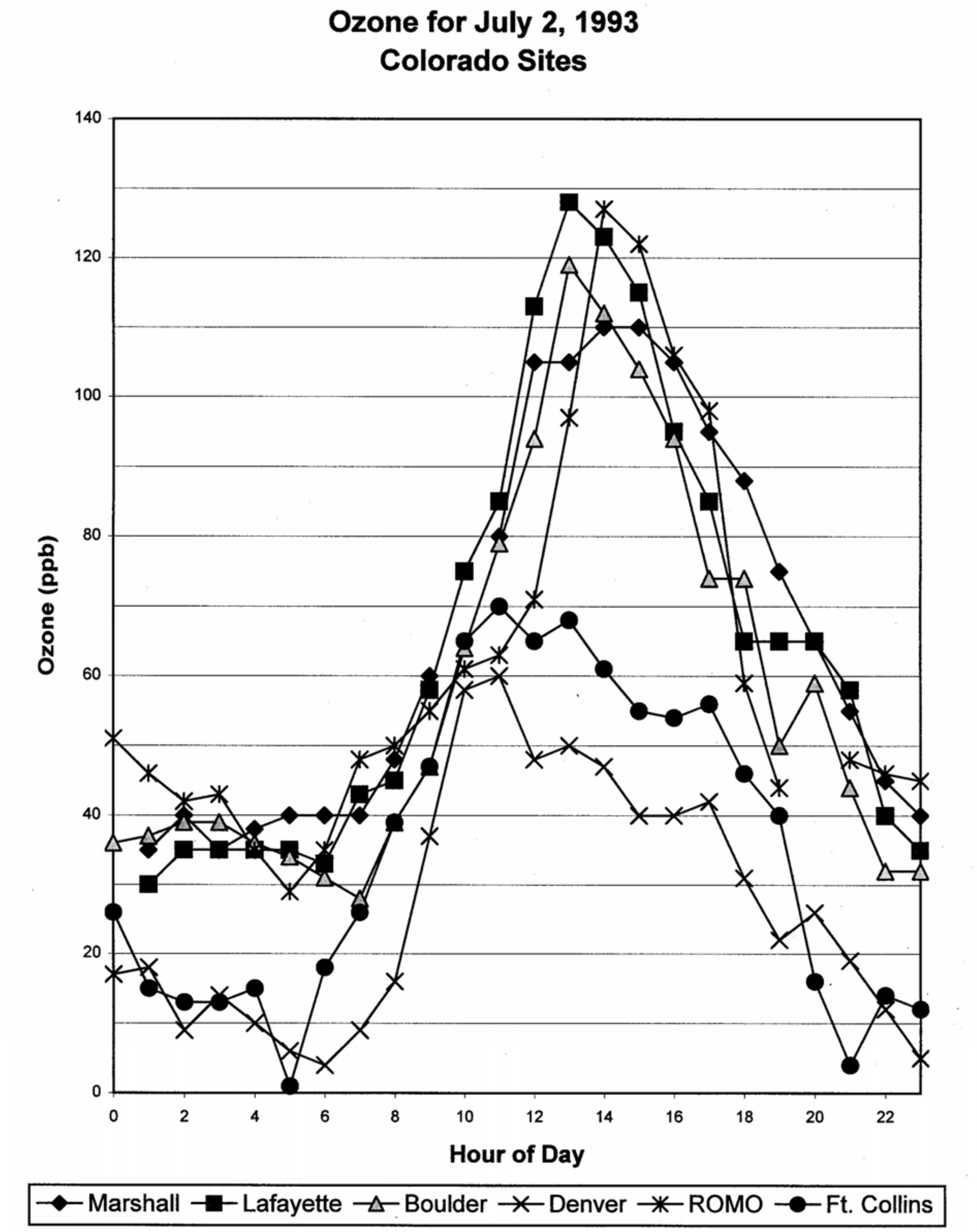


Figure III-4. Diurnal ozone concentrations at ROMO and other locations along the Front Range on July 2, 1993.

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

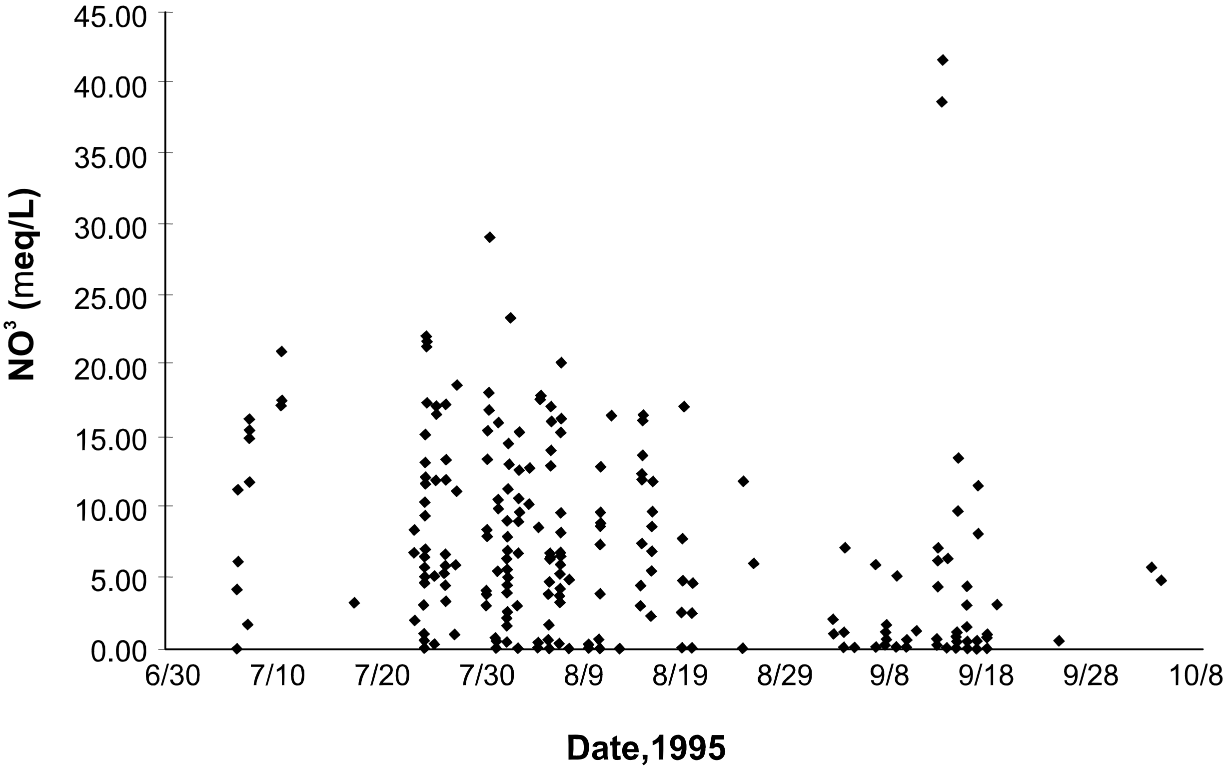
	1				3						
Forest Lakes (Lg. N)	4E2-055	2	98	3300	7.7	192	21	0.5	142	233	0.9
Panhandle Reservoir	4E3-005	20	4903	2592	7.8	377	30	0.0	240	420	3.2
Dowdy Lake	4E3-006	43	228	2481	8.0	952	67	1.5	696	1183	6.9
Parika Lake	4E3-009	1	70	3471	8.2	889	113	2.6	682	1050	1.2
No Name	4E3-016	2	21	2848	6.9	363	37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

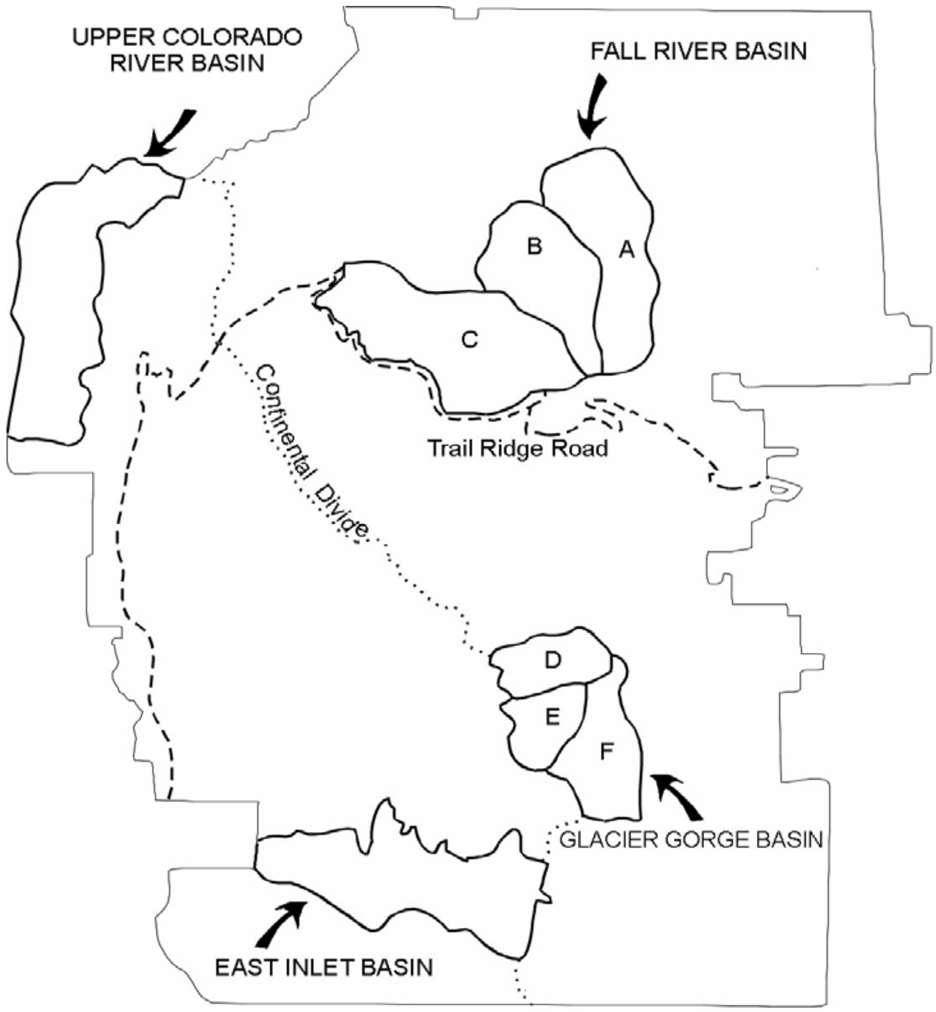


Figure III-5. Nitrate concentration measured in alpine lakes east of the Continental Divide throughout the length of Colorado and in southeastern Wyoming (Mussleman et al. 1996).

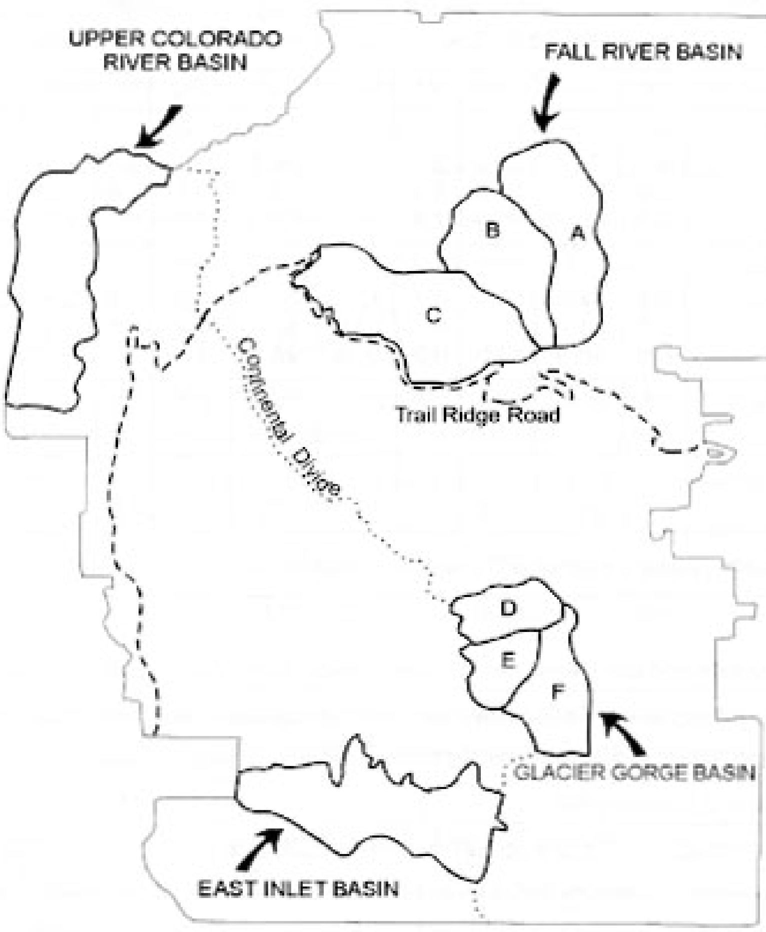


Figure III-6. Watersheds of ROMO studied by the U.S. Fish and Wildlife Service. Subbasins are (A) Roaring River, (B) Ypsilon Creek, (C) Upper Fall River, (D) Tyndall Gorge, (E) Loch Vale, (F) Glacier Creek (Gibson et al. 1983). (Note: park boundary shown does not reflect 1980 boundary change.)

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.

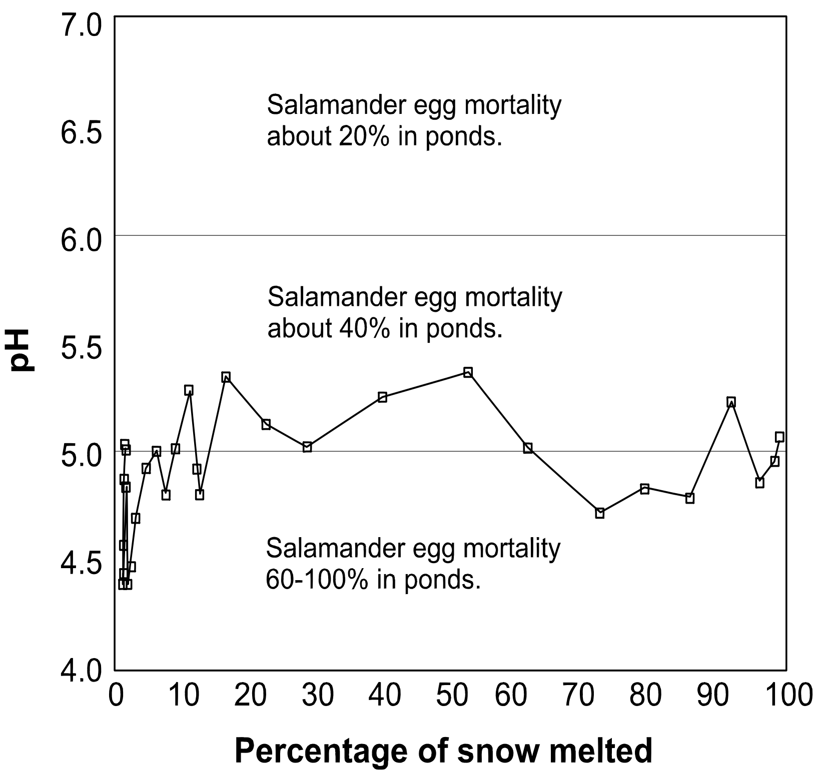


Figure III-7. Predicted acidity (expressed as pH) of snowmelt at Buffalo Pass, adjacent to the Mt. Zirkel Wilderness Area, compared with expected salamander egg mortality at various pH values, from Kiesecker (1991). (Source: Turk and Campbell 1997)

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

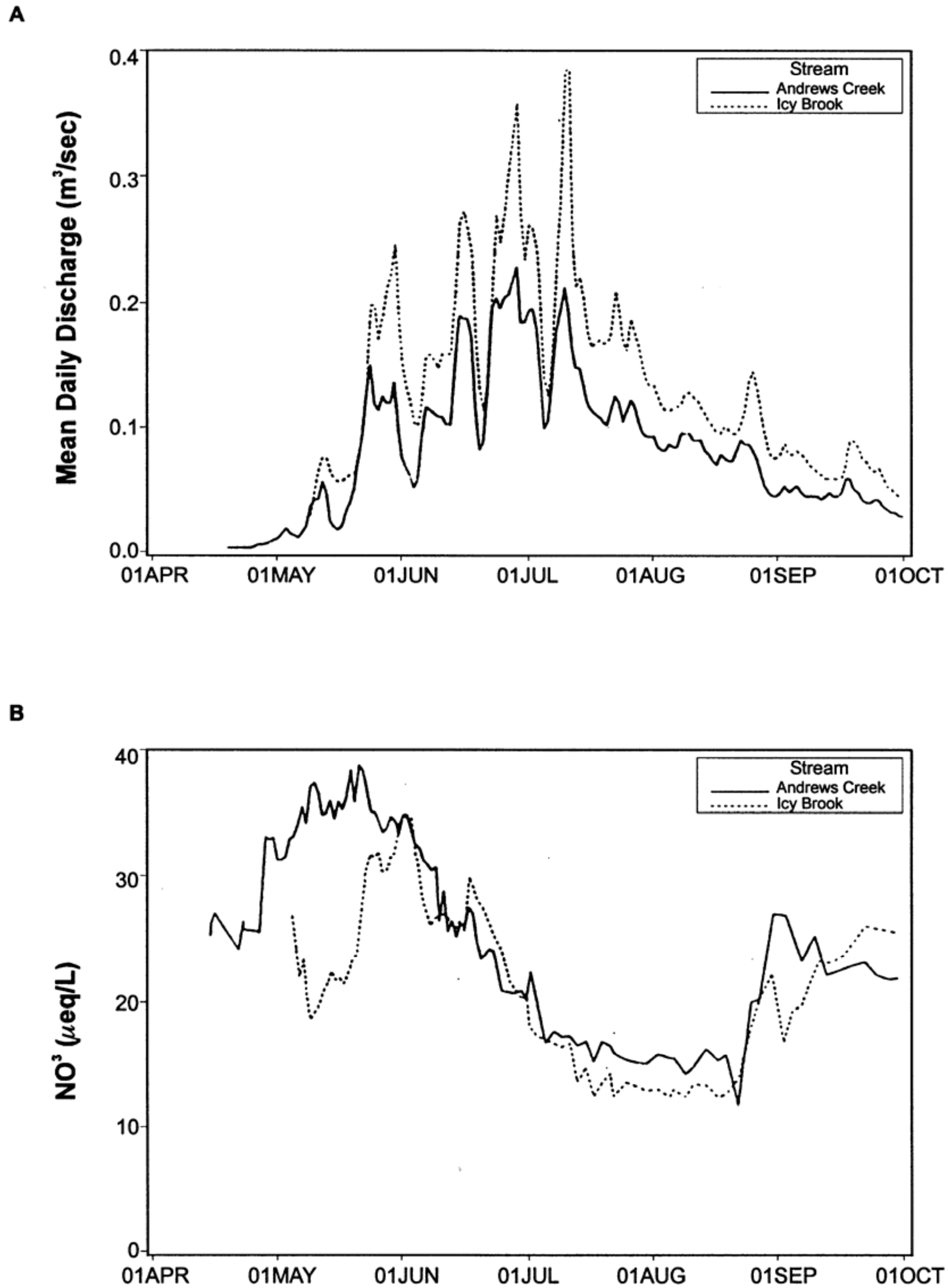


Figure III-8. Daily discharge (A) and nitrate concentration (B) in Icy Brook and Andrews Creek within the Loch Vale watershed in April-September 1992 (Campbell et al. 1995).

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

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

information and guidelines for addressing the use of lichens as sensitive receptors of air pollution is Stolte et al. (1993).

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.

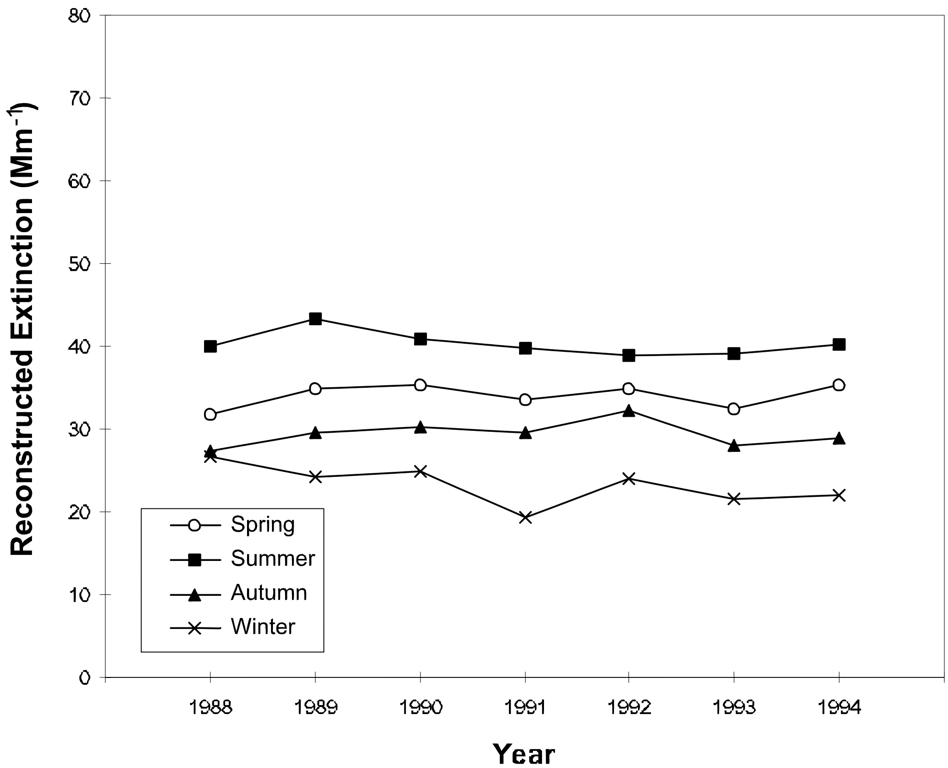


Figure III-9.Seasonal average reconstructed extinction (Mm^{-1}) for the period March 1988 through February 1995.

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.

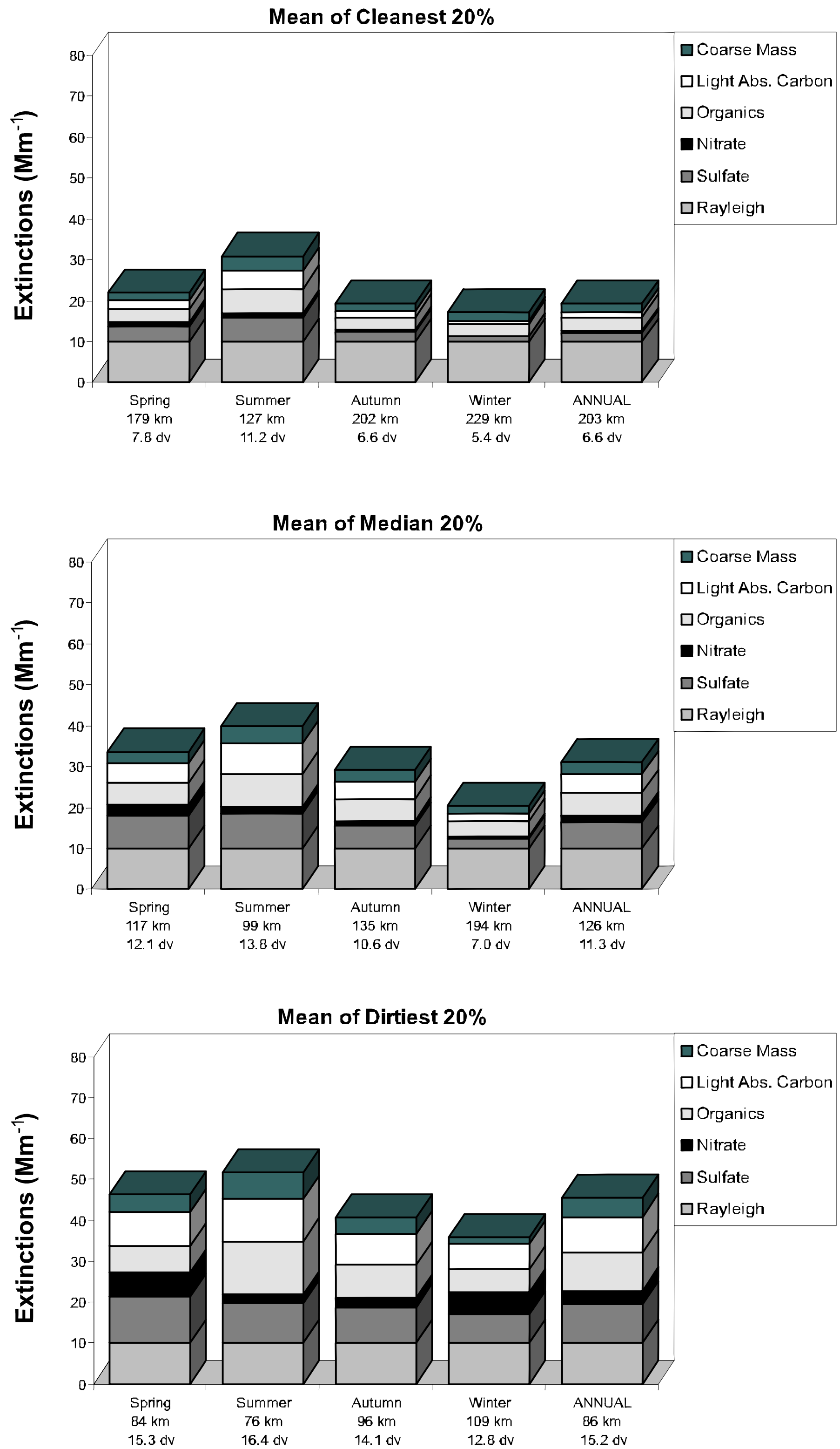


Figure III-10. Reconstructed extinction budgets for Rocky Mountain National Park, Colorado, March 1988 - February 1995.

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.

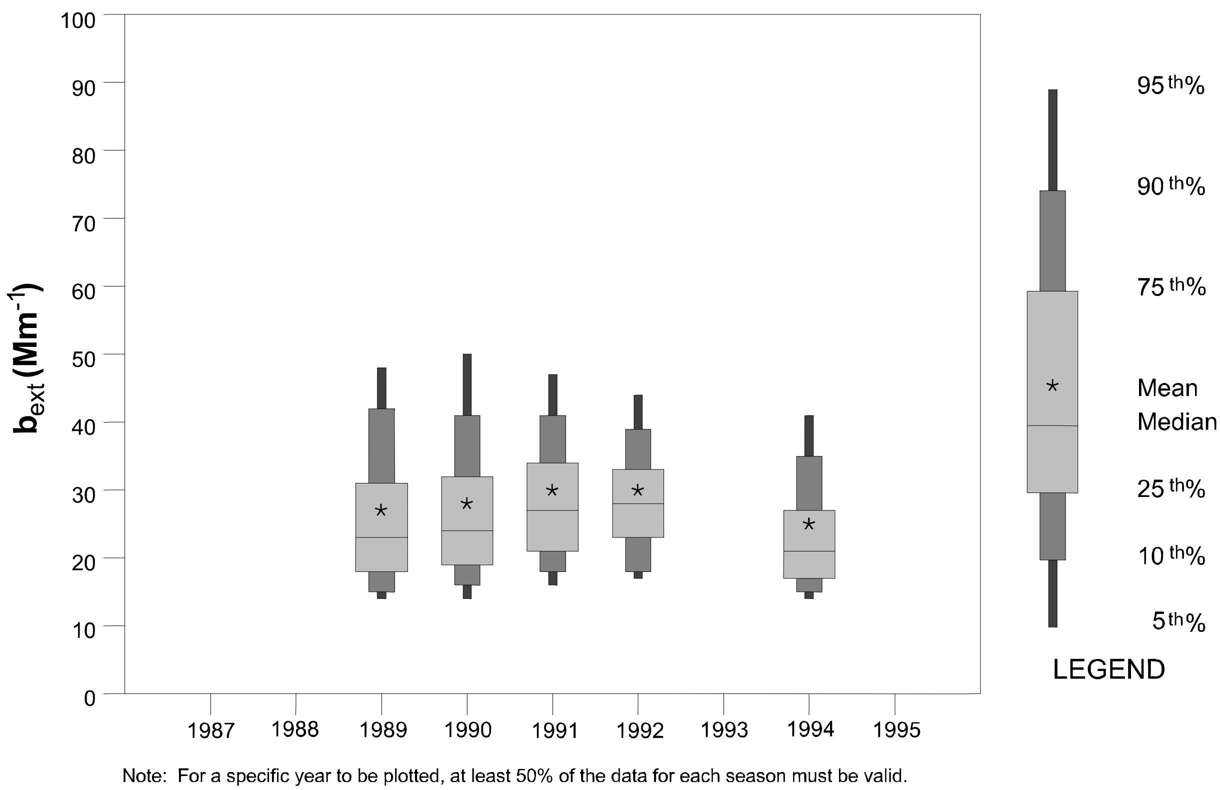


Figure III-11. Annual arithmetic mean and cumulative frequency statistics, Rocky Mountain National Park, Colorado, transmissometer data (filtered).

- <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.