



Air Quality Related Values (AQRVs) for Rocky Mountain Network (ROMN) Parks

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

Natural Resource Report NPS/ROMN/NRR—2016/1183



ON THE COVER

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

Photographs courtesy of the National Park Service

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Summary

This report describes the Air Quality Related Values (AQRVs) of the Rocky Mountain Network (ROMN). AQRVs are those resources sensitive to air quality and include streams, lakes, soils, vegetation, fish and wildlife, and visibility. The ROMN parks that are included in the NPS Inventory and Monitoring (I&M) Program, and discussed in this report, are Florissant Fossil Beds National Monument (FLFO), Glacier National Park (GLAC), Grant-Kohrs Ranch National Historic Site (GRKO), Great Sand Dunes National Park and Preserve (GRSA), Little Bighorn Battlefield National Monument (LIBI), and Rocky Mountain National Park (ROMO). GLAC, GRSA, and ROMO are designated as Class I, giving them a heightened level of protection against harm caused by poor air quality under the Clean Air Act (CAA).

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

Although the ROMN parks are in relatively remote areas, they experience air pollutants from regional and local sources, including large power plants, urban areas, agriculture, oil and gas development, and fires. Rocky Mountain NP and FLFO are located in close proximity to several relatively large population centers along the eastern edge of the Colorado Front Range, including Denver, Colorado Springs, and Fort Collins. Glacier NP, GRSA) and the other small parks in the ROMN (LIBI and GRKO) are more remote.

Atmospheric N and S pollutants can cause acidification of streams, lakes, and soils. Levels of S deposition are relatively low in the network region; N deposition is somewhat higher, especially along the Colorado Front Range, including at ROMO and FLFO. High elevation lakes, soils, and streams are common in ROMN parks. They are particularly vulnerable to acidification as they often have limited buffering capacity. However, because S and N deposition levels in the ROMN are relatively low, chronic acidification has not been documented in the network parks. Episodic acidification is likely more prevalent.

In addition to contributing to the potential for acidification, N is a nutrient and therefore N deposition can also cause undesirable enrichment of natural ecosystems, leading to changes in plant species diversity and soil nutrient cycling. ROMO receives the highest levels of N deposition of the network parks and ecosystem impacts due to N have been documented in this park, including:

- increased concentration of nitrate (NO_3^-) in surface water

- change in the species composition and abundance of diatoms in alpine lakes
- changes in the chemistry of soil and tree foliage, consistent with the onset of N-saturation
- changes in the species composition and abundance of alpine plants

Ozone pollution can harm human health, reduce plant growth, and cause visible injury to foliage. Both short-term concentrations, important to visitor and park staff health, and long-term cumulative exposures, important to plant health, are elevated at ROMO, and are estimated to be somewhat elevated at GRSA and FLFO. Portions of ROMO are located in counties designated as nonattainment for the national O₃ standard, and periodically experience unhealthy air.

Particulate pollution can cause haze, reducing visibility. This is an important concern in the ROMN because the parks are known for their magnificent vistas. Haze has reduced the clarity of these views to some extent in all the parks in the network. Sulfates from large power plant emissions; N emissions from vehicles, power plants, industry, and agriculture; and organics from fires and other sources all contribute to haze.

Airborne contaminants, including mercury (Hg) and other heavy metals, can accumulate in food webs, reaching toxic levels in top predators. Contaminants deposited from the atmosphere onto remote national park watersheds in the western United States can accumulate in fish to levels that are of concern regarding human and wildlife health. The Western Airborne Contaminant Assessment Project (WACAP) study analyzed fish, vegetation, lake sediments, and other ecosystem components in GLAC and ROMO and found relatively high concentrations of toxics in both parks, including:

- Polycyclic aromatic hydrocarbon (PAH) concentrations in snow, sediments, and vegetation were found to be significantly elevated in GLAC, and attributed partly to emissions from a local aluminum smelter (Usenko et al. 2010). Fire is also an important source of PAH in the ROMN.
- Concentrations of banned pesticides were high in fish from GLAC and ROMO, with some fish exceeding health thresholds for humans and wildlife.
- Evidence of endocrine disruption was found in some fish from GLAC and ROMO.

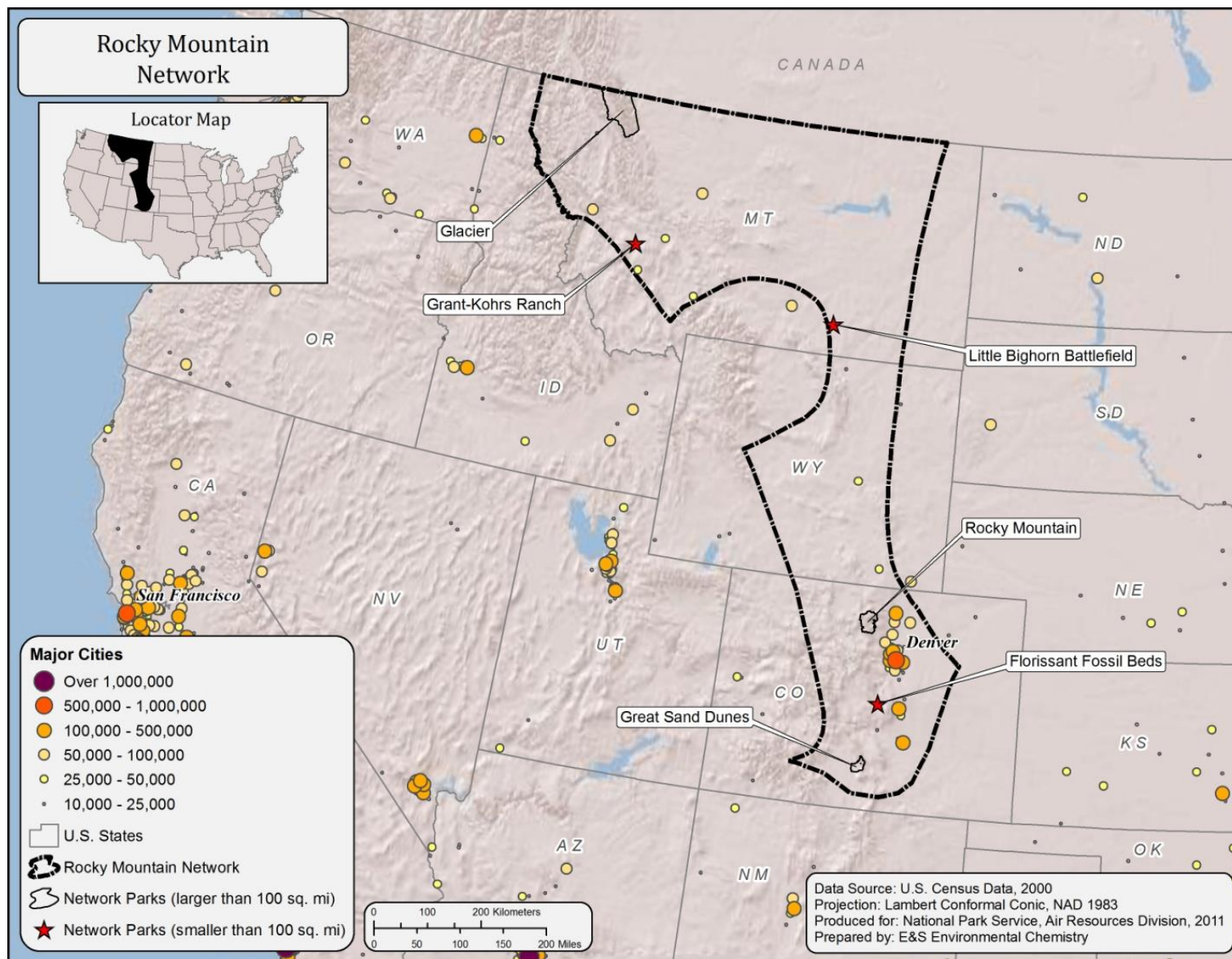
Background

There are three parks in the Rocky Mountain Network (ROMN) that are larger than 100 square miles: Glacier National Park (GLAC), Great Sand Dunes National Park and Preserve (GRSA), and Rocky Mountain National Park (ROMO). In addition, there are three smaller parks: Florissant Fossil Beds National Monument (FLFO), Grant-Kohrs Ranch National Historic Site (GRKO), and Little Bighorn Battlefield National Monument (LIBI). Larger parks generally have more available data with which to evaluate air pollution sensitivities and effects. In addition, the larger parks generally contain more extensive resources in need of protection against the adverse impacts of air pollution. These parks exhibit varying levels of air pollutant exposure and sensitivity to effects. Air Quality Related Values (AQRVs) have previously been assessed in the Rocky Mountain region by Peterson et al. (1998). This report represents an update to that compilation.

Human population centers around ROMN parks are sparse, except along the eastern edge of the Colorado Front Range. ROMO is located within about 40 km of the Colorado Front Range urban corridor, which has the highest human population density in the Rocky Mountain region (Dennehy et al. 1993). FLFO is also situated close to the Front Range urban corridor, in proximity to Colorado Springs. This corridor experienced an increase in human population of about 53% from 1980 to 2000 (Porter and Johnson 2007). Map 1 shows the network boundary along with locations of each park and population centers of more than 10,000. GLAC, GRSA, and other small parks in this network are more distant from human population centers.

Atmospheric deposition of air pollutants represents one of the most important potential threats to aquatic and terrestrial resources in the ROMN, especially in the higher elevation parks including ROMO, GLAC, and GRSA. Aquatic resources in ROMO and GLAC 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 these parks 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 remoteness of so many surface waters in ROMO GRSA, and GLAC, human impacts on the water quality are minimized. With the exception of anthropogenic atmospheric contributions of pollutants and climate change, direct human impacts on most lakes and streams in these parks are limited. Especially for surface waters in remote locations, potential impacts are restricted mainly to a few dams and irrigation channels, as well as the impacts of hiking, camping, and horseback-riding activities.

More than one-fourth of ROMO is above treeline, which is located at an elevation of about 3,500 m in this park. There are more than 60 peaks in the park that are over 3,600 m elevation. Precipitation generally increases with elevation, and the higher peaks receive most of their precipitation as snow (Doesken et al. 2003).



Map 1. Network boundary and locations of parks and population centers greater than 10,000 people near the ROMN.

Atmospheric Emissions and Deposition

Emissions

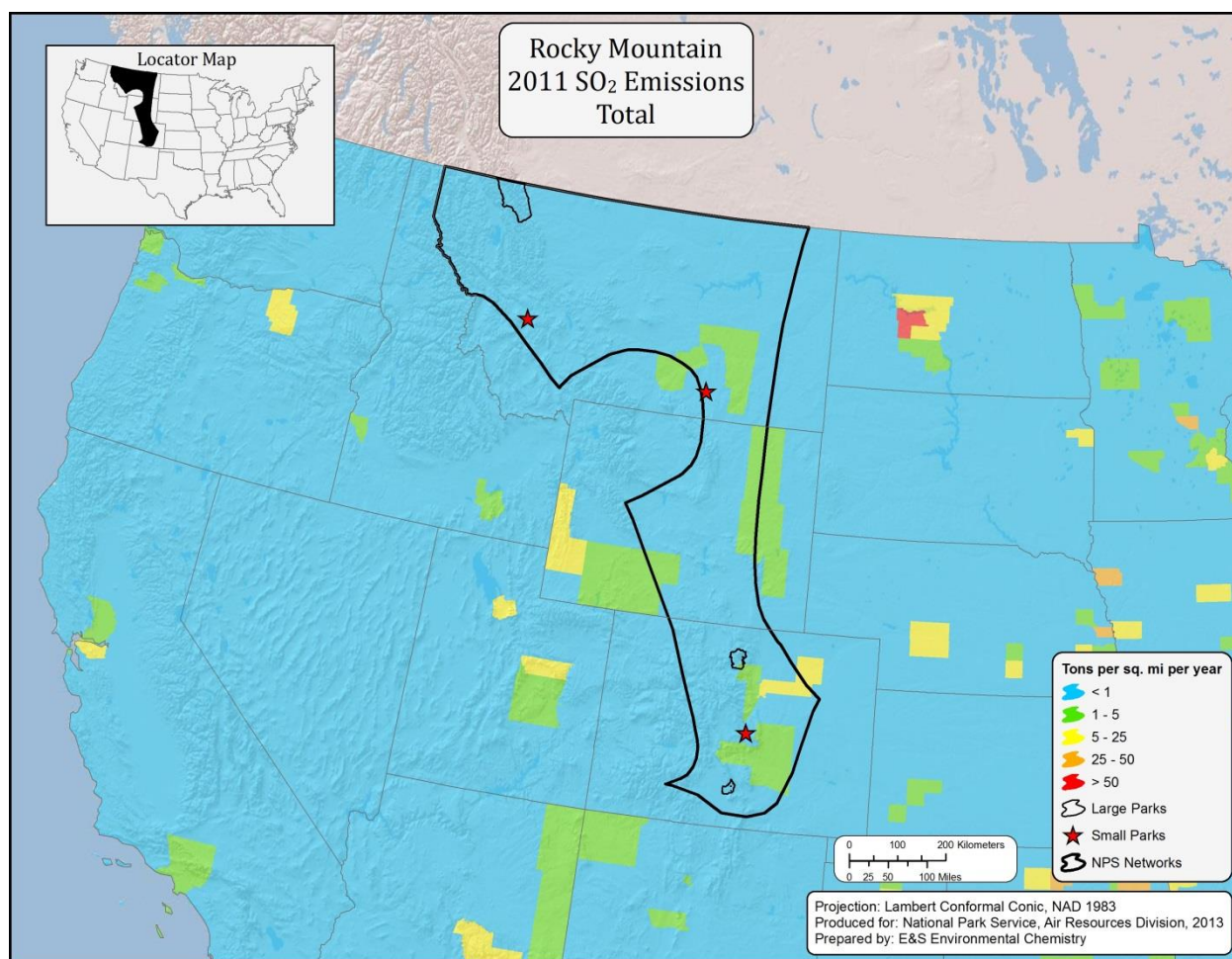
In general, air quality in the Rocky Mountains is considerably better than in most other areas of the conterminous United States. National parks in the Rocky Mountains receive generally low levels of most atmospheric pollutants (Sisterson et al. 1990, Smith 1990). Nevertheless, sensitive aquatic and terrestrial ecosystems, especially those at high elevations, are degraded by existing pollution. Elevated emissions of nitrogen oxides (NO_x) and elevated concentrations of ozone (O_3) have been measured in some areas (Sisterson et al. 1990). Agricultural activities, including feedlots, are significant sources of reduced nitrogen (NH_3) in some areas. The parks also have impaired visibility at times (Savig and Morse 1998). Parks within the ROMN (GLAC and ROMO) had two of the three highest average snowpack pesticide concentrations of all western parks included in a recent U.S. Environmental Protection Agency (EPA) survey of air toxics (Landers et al. 2008). Emissions of NO_x from the western states that lie between ROMO and the west coast decreased about 13% between 1990 and 2001. Emissions in Colorado decreased 8.7% during that time period. Further decreases are ongoing, but may be counteracted to some extent by the relatively recent and pronounced oil and gas (O&G) development in the region. Federal mobile source standards should reduce mobile source NO_x emissions in the Denver metropolitan area by 71% by the year 2022. Although stationary and area source emissions are projected to increase, total NO_x emissions in the Denver metropolitan area are predicted to decrease by 28% between 2001 and 2022 (Silverstein and Taipale 2006).

County-level emissions near the ROMN, based on data from the EPA's National Emissions Inventory (NEI) during a recent time period (2011), are depicted in Maps 2 through 4 for sulfur dioxide (SO_2), NO_x , and NH_3 , respectively. Most counties near the ROMN parks had SO_2 emissions in the range of 1 to 5 tons/ mi^2/yr (Map 2). Patterns in NO_x emissions were generally similar, with highest values in the range of 5 to 25 tons/ mi^2/yr (Map 3). Emissions of NH_3 near ROMN parks were somewhat lower, with most counties showing emissions levels below 2 tons/ mi^2/yr (Map 4).

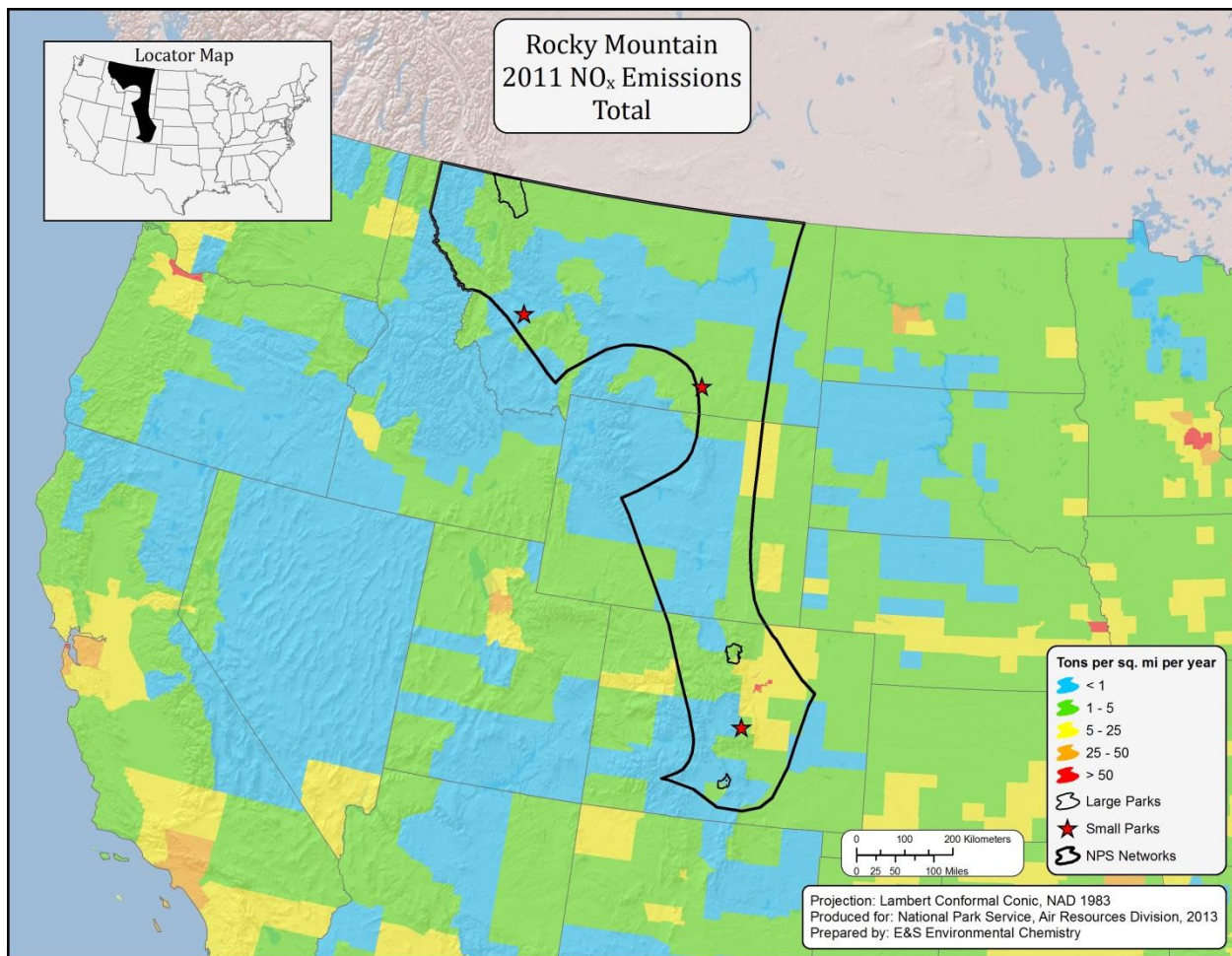
The predominant direction of air mass movement over the Front Range is from west to east (Barry 1973), with periodic upslope movement from the east and southeast (Kelley and Stedman 1980). Wind rose data from ROMO during the period 1989 through 1995 showed a distinct pattern of predominant air movement from the northwest. However, a second frequent wind direction was from the south and southeast, from the general direction of the Denver metropolitan area (Peterson et al. 1998). This is important because air masses that move directly from the Denver area to ROMO have the potential to transport high levels of N, S, and O_3 -forming compounds to the park. The easterly upslope storm track also carries air masses across agricultural (livestock and fertilized cropland) and industrial and metropolitan areas of Colorado before reaching the vicinity of ROMO (Bowman 1992). Higher atmospheric concentrations of NH_3 , NO_x gases, and nitric acid particulates have been measured near the park during upslope events (Langford and Fehsenfeld 1992, Parrish et al. 1986).

Air mass back trajectory analysis has been used to identify major source regions of atmospheric ammonium sulfate ($(\text{NH}_4)_2\text{SO}_4$) aerosol emissions that impact Class I national parks in the western

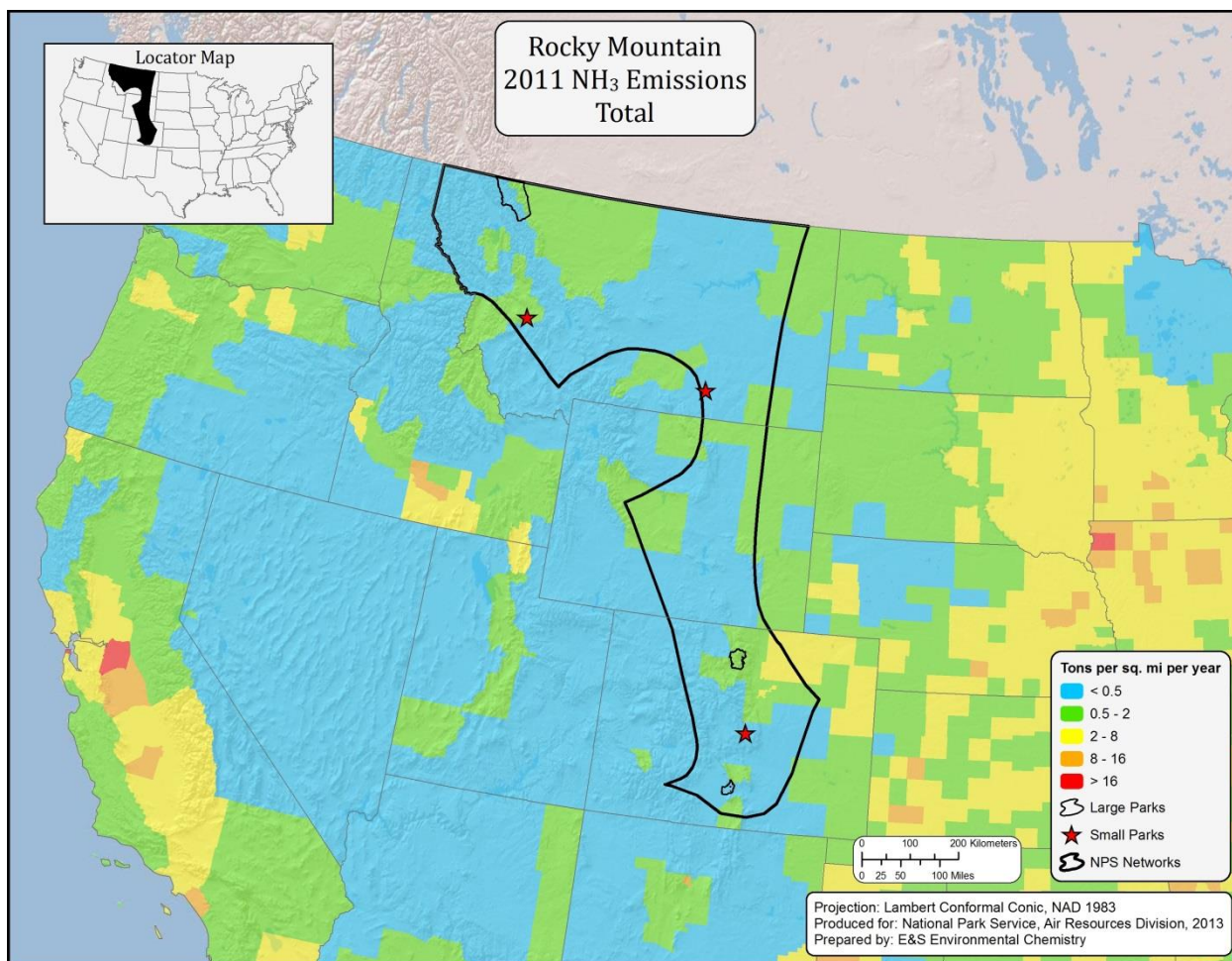
United States (Xu et al. 2006). This approach estimates the amount of time that an air mass spent over each of a group of pre-specified source regions. An implicit assumption is that the amount of time that an air mass spends over a region is linearly related to that region's contribution to the pollutant concentration in the air measured at the receptor site location. Results of this modeling study, conducted using 2000-2002 data and the NOAA HYSPLIT v4.6 model, suggested that ocean shipping and other port emissions along the Pacific coast contributed substantially to atmospheric aerosol concentrations over large portions of the western United States, including the Rocky Mountain region. The largest source areas for atmospheric pollutants in the Rocky Mountain region, however, appeared to be the northwestern United States, northwestern Colorado, southeastern Colorado, and Arizona (Xu et al. 2006).



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



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



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

Production of O&G across the western United States contributes substantial N, S, and volatile organic compound (VOC) emissions. Oil and gas production has been accelerating. The Western Regional Air Partnership (WRAP) has compiled emissions data from these sources, which are individually minor, but collectively substantial. WRAP inventories identified over 100,000 tons per year of NO_x emissions in the WRAP region which had not previously been quantified (Gribovicz 2011).

ROMO is located just to the west of large areas of livestock, crop, and pasture lands. Pesticide use in this region is more than 1.3 million kg/yr, mainly applied to corn and hay (Kimbrough and Litke 1998). The heaviest agricultural pesticide use around GLAC is to the west of the park in central Washington and southern Idaho, and also to the northeast in Alberta, Canada (Mast et al. 2006). Pesticides applied to agricultural lands in proximity to these national parks have the potential to volatilize, be transported in the atmosphere, and then deposit on sensitive ecosystems.

The highest atmospheric concentrations of reduced N in the vicinity of ROMO are found on the Colorado eastern plains. The highest concentrations of atmospheric oxidized N occur along the Front

Range urban corridor (Benedict et al. 2013b). Both of these source regions are located to the east of ROMO. Benedict et al. (2013b) further indicated that the atmospheric concentrations of reactive N species east of the Continental Divide were, on average, more than 50% higher than those found west of ROMO. This supports the conclusion that emissions sources to the east of the park are especially important to the pollutant deposition at ROMO.

Measurements of gas phase and particulate N (reduced and oxidized forms) and other constituents were made at ground-level at nine sites located across Colorado (Benedict et al. 2013b). The highest concentrations of reduced N were found in the eastern plains, attributable largely to agricultural sources. Oxidized N concentrations were highest in the Front Range urban corridor. Both of these source areas lie to the east of ROMO. Upslope easterly winds transport these pollutants into the park (Benedict et al. 2013b).

Benedict et al. (2013a) constructed a complete reactive N deposition budget for a site in the eastern portion of ROMO. The budget included all dry and wet forms of N, both inorganic and organic. It was based on data collected over a one-year period (2008-2009). The total reactive N deposition was estimated to be 3.65 kg N/ha/yr. The largest contributors (54% combined) were wet deposition of nitrate (NO_3^-) and ammonium (NH_4^+). The next highest contributors were wet deposition of organic N and dry deposition of NH_3 (37% combined). These latter two species are often not measured in deposition estimates.

Deposition

Sulfur and Nitrogen

Atmospheric N deposition in the Colorado Front Range has increased in recent decades due to a greater prevalence of large industrial animal feedlots, and increased urbanization, human population, and distance driven (Fenn et al. 2003b). Total inorganic wet N deposition at the high elevation NADP site at Loch Vale in ROMO was estimated to be about 3.1 kg N/ha/yr, based on a five-year average centered on 2001 (Blett and Morris 2004) and about 2.9 kg N/ha/yr during the period 2008-2012 (Morris et al. 2014). Wet inorganic N deposition at Loch Vale increased from 1985 to 2000 by about 2% per year, largely due to increased wet NH_4^+ deposition (Burns 2003, Clow et al. 2003). Dry inorganic N deposition is less certain, with an estimate based on the Clean Air Status and Trends Network (CASTNET) equal to 0.8 kg N/ha/yr (Blett and Morris 2004). This CASTNET estimate is probably biased low (Porter and Johnson 2007) because the CASTNET monitoring site is located at lower elevation than the NADP/NTN site that measures wet deposition and because CASTNET does not include measurement of all N species.

Total estimated S deposition in 2002 was generally less than 2 kg S/ha/yr throughout the ROMN region, with scattered small areas estimated to receive 2 to 5 kg S/ha/yr (Sullivan et al. 2011b). Wet S and N deposition hotspots in northern Colorado were probably due in large part to emissions of SO_2 and NO_x from coal-fired power plants in northwestern Colorado and southwestern Wyoming (Mast et al. 2001, Nanus et al. 2003, Turk and Campbell 1997). Total N deposition within the network region ranged from less than 2 kg N/ha/yr to as high as 5 to 10 kg N/ha/yr, especially along the Front Range

and at higher elevations. Throughout most of the network region, estimated total N deposition was in the range of 2 to 5 kg N/ha/yr (Sullivan et al. 2011b).

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

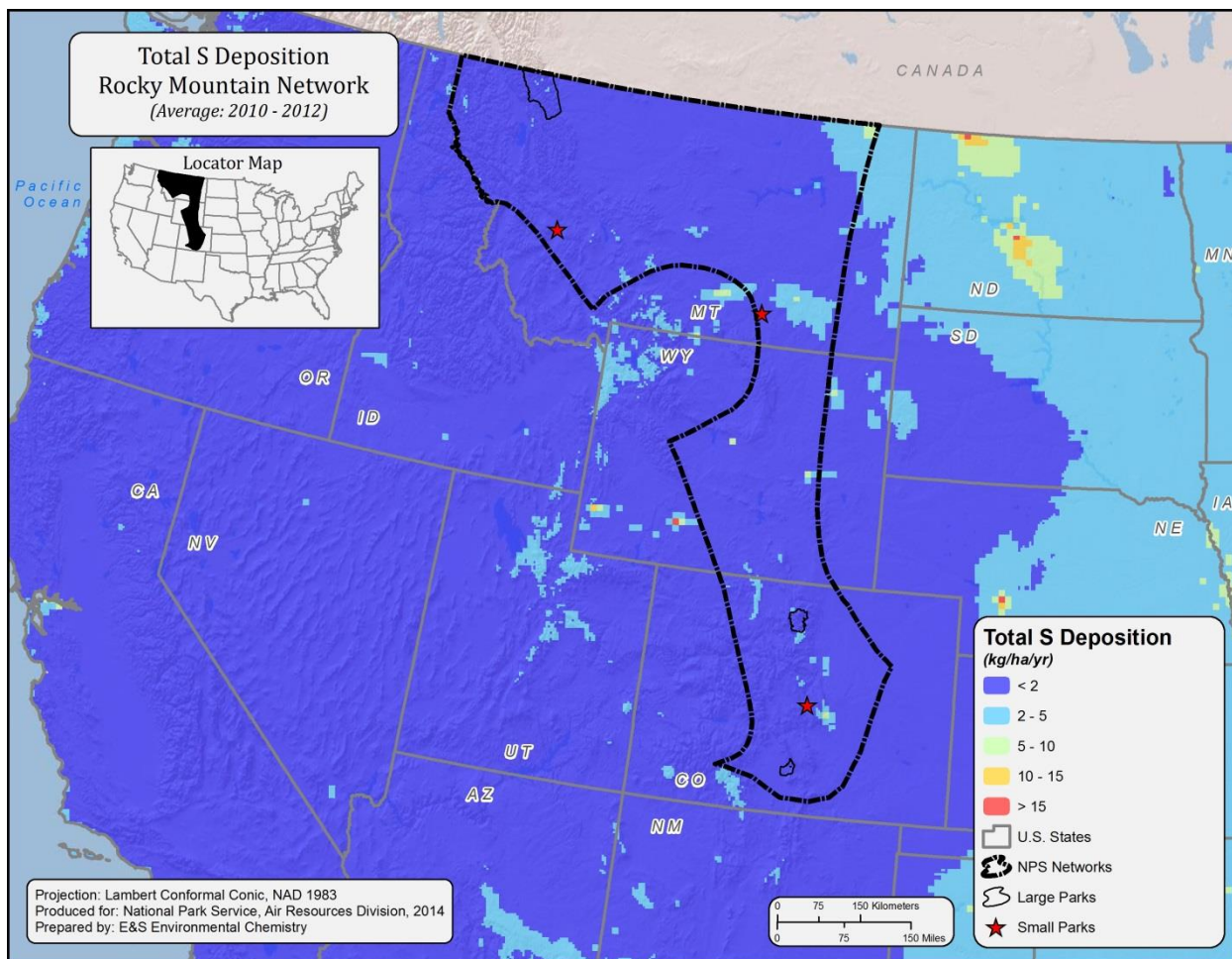
Atmospheric S and N deposition levels have decreased at some parks in ROMN and increased at others since 2001, based on TDEP estimates (Table 1). Decreases in total S deposition over the previous decade, where they occurred, were less than 20% for the parks in this network. Oxidized and reduced N showed opposite patterns, with oxidized N decreasing and reduced N increasing at all of the parks in the network since the monitoring period 2000-2002.

Total S deposition in and around the ROMN for the period 2010-2012 was generally less than 2 kg S/ha/yr at park locations within the network area (Map 5). Oxidized inorganic N deposition for the period 2010-2012 was less than 5 kg N/ha/yr throughout the park lands within the ROMN (Map 6). Most areas received less than 5 kg N/ha/yr of reduced inorganic N from atmospheric deposition during this same period (Map 7); parts of ROMO received higher amounts. Estimated total N deposition was higher than 5 kg N/ha/yr at ROMO, but generally lower elsewhere within the ROMN (Map 8).

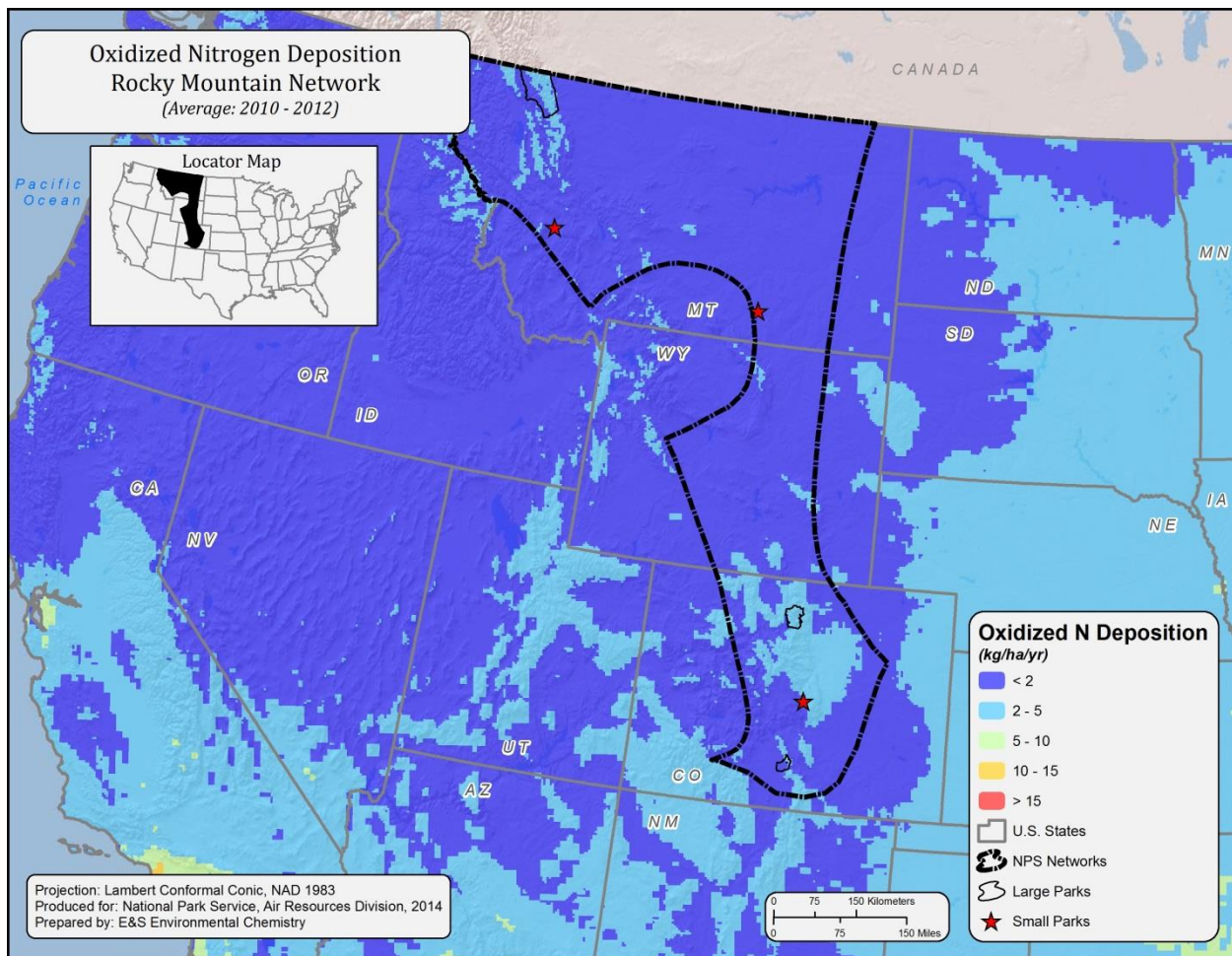
In the Rocky Mountains, deposition chemistry is influenced by a complex collection of emissions sources. Some parks are subject to deposition of pollutants from urban areas, some are more heavily affected by long-distance transport of pollutants from industrial facilities and electric utilities, and some are affected by local sources. Therefore, the quantity of emissions received and the potential threat to natural resources differ from park to park within the Rocky Mountain region (Peterson et al. 1998). For example, in the Mt. Zirkel Wilderness of northwestern Colorado, SO_4^{2-} and NO_3^- in the snow appeared to originate largely from emissions sources in the Yampa Valley, about 75 km to the west (Turk et al. 1992), whereas ROMO is more affected by emissions from the Front Range to the southeast of the park. A comparison of winter snowpack and NADP/National Trends Network (NTN) precipitation chemistry in Colorado (Heuer et al. 2000) suggested that high-elevation

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

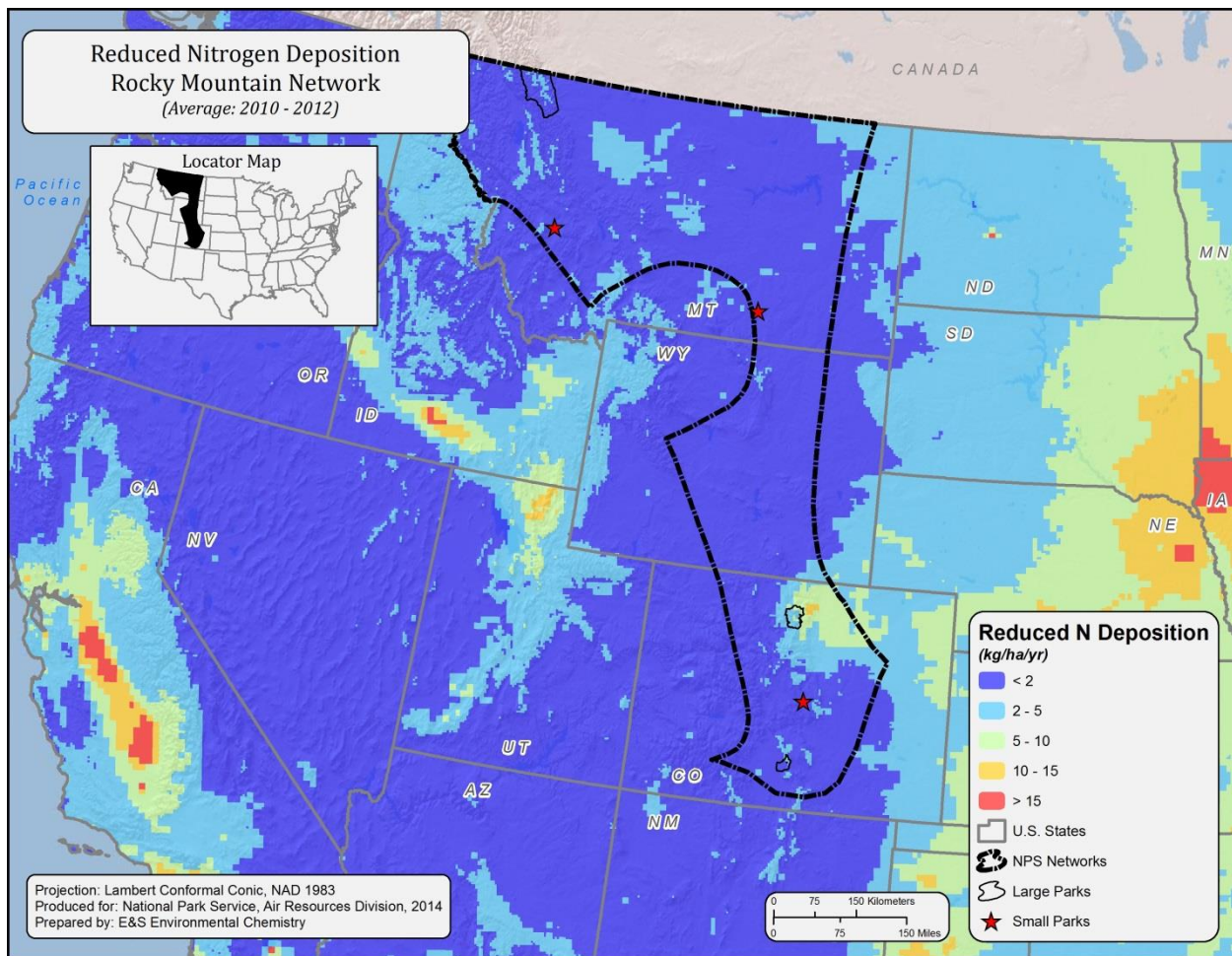
Park Code	Park Name	Parameter	2001 Average (kg/ha/yr)	2011 Average (kg/ha/yr)	Absolute Change (kg/ha/yr)	Percent Change	2011 Minimum (kg/ha/yr)	2011 Maximum (kg/ha/yr)	2011 Range (kg/ha/yr)
FLFO	Florissant Fossil Beds	Total S	1.29	1.12	-0.17	-13.1%	1.04	1.18	0.13
		Total N	3.51	3.26	-0.25	-7.2%	3.00	3.42	0.42
		Oxidized N	2.26	1.82	-0.44	-19.6%	1.67	1.90	0.23
		Reduced N	1.25	1.44	0.19	15.4%	1.34	1.52	0.18
GLAC	Glacier	Total S	1.75	1.42	-0.33	-18.4%	0.69	2.24	1.54
		Total N	3.59	3.70	0.11	3.8%	1.84	5.47	3.63
		Oxidized N	2.28	2.03	-0.26	-10.9%	1.04	2.92	1.88
		Reduced N	1.31	1.68	0.37	30.7%	0.80	2.59	1.79
GRKO	Grant-Kohrs Ranch	Total S	0.46	0.49	0.04	8.8%	0.46	0.53	0.07
		Total N	1.64	1.73	0.08	5.1%	1.66	1.80	0.14
		Oxidized N	0.98	0.80	-0.18	-17.8%	0.77	0.85	0.08
		Reduced N	0.66	0.92	0.26	40.4%	0.83	1.03	0.20
GRSA	Great Sand Dunes	Total S	0.93	0.89	-0.04	-6.3%	0.48	1.88	1.39
		Total N	2.74	2.83	0.09	0.4%	1.75	5.61	3.86
		Oxidized N	1.77	1.48	-0.29	-18.6%	0.93	2.68	1.75
		Reduced N	0.98	1.35	0.38	36.4%	0.81	3.03	2.22
LIBI	Little Bighorn Battlefield	Total S	0.90	1.34	0.45	49.7%	1.33	1.35	0.01
		Total N	2.60	2.78	0.18	6.9%	2.69	2.80	0.12
		Oxidized N	1.37	1.26	-0.11	-7.8%	1.25	1.31	0.06
		Reduced N	1.23	1.51	0.29	23.3%	1.37	1.55	0.18
ROMO	Rocky Mountain	Total S	1.79	1.62	-0.17	-7.3%	1.08	2.20	1.12
		Total N	5.53	7.27	1.74	34.5%	4.19	11.98	7.79
		Oxidized N	3.37	2.78	-0.60	-16.8%	1.91	3.54	1.63
		Reduced N	2.16	4.49	2.33	115.0%	2.21	8.97	6.76



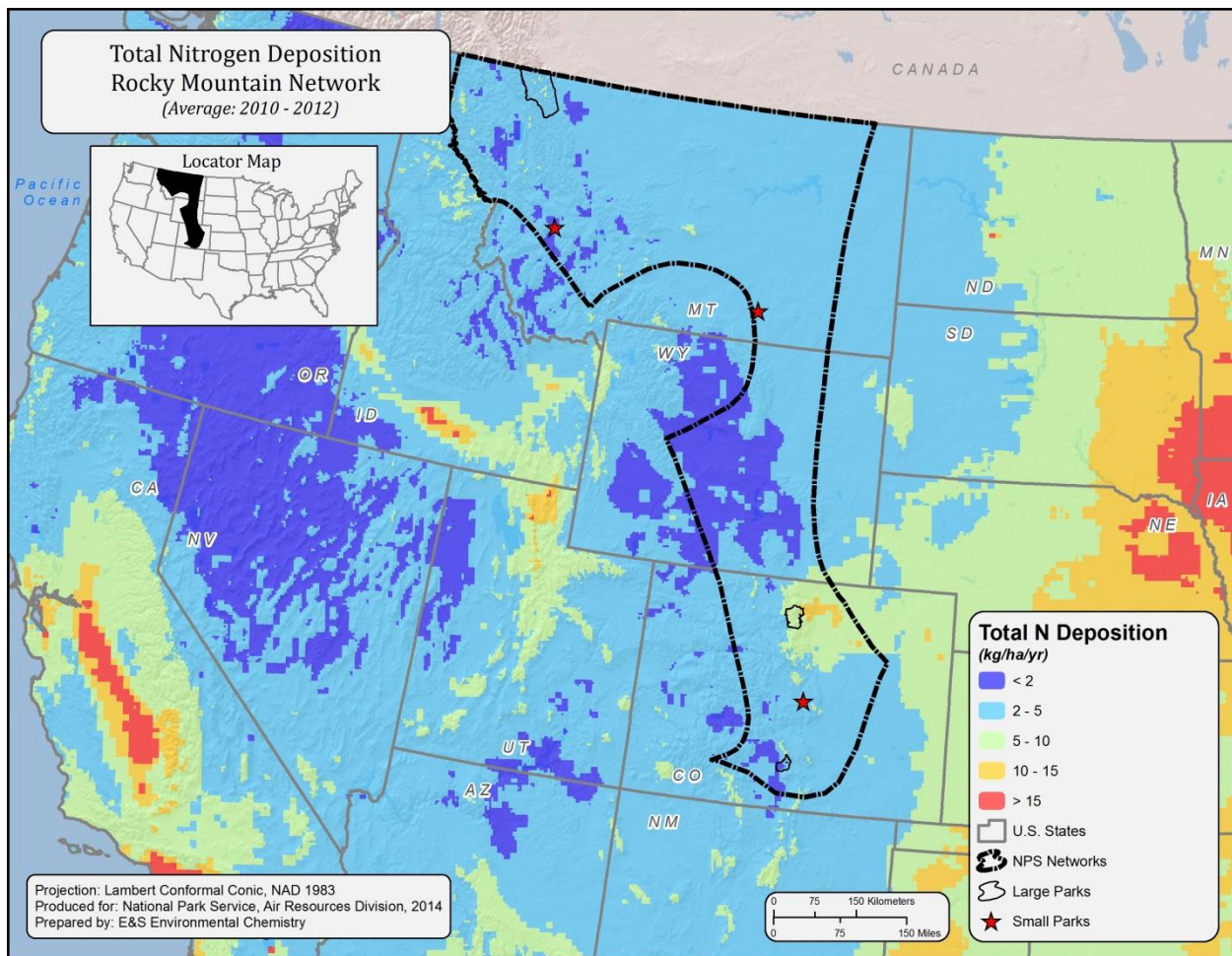
Map 5. Total S deposition for the three-year period centered on 2011 in and around ROMN. (Source: Schwede and Lear 2014)



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



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



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

watersheds in ROMO are influenced by air pollution sources located on both sides of the Continental Divide. Concentrations of NO_3^- and SO_4^{2-} in the snowpack were higher on the eastern slope during winter, but there were not significant differences between east and west in snowpack NH_4^+ concentrations. During summer, NADP/NTN precipitation chemistry showed that concentrations of NO_3^- and NH_4^+ were higher on the eastern slope, with no significant difference for SO_4^{2-} . For all three solutes, summer concentrations in precipitation were about twice as high as winter concentrations.

Therefore, annual patterns in wet deposition are more strongly driven by summer conditions (Heuer et al. 2000). Winter precipitation in Colorado originates mainly from the northwest (Hansen et al. 1978, Parrish et al. 1990). During spring and summer, there are often upslope movements of air masses from the Denver-Boulder-Fort Collins urban corridor to the vicinity of ROMO (Parrish et al. 1990).

The Rocky Mountain Atmospheric Nitrogen and Sulfur (RoMANS) Study was conducted to improve understanding of S and N deposition to ROMO and their sources. The study involved two five-week sampling periods in 2006, during spring and summer. These seasons typically have relatively high levels of S and N deposition at ROMO. During spring, large upslope events transport pollutants to the park from the east. During summer, frequent (almost daily) afternoon precipitation due to convection activity brings increased wet deposition.

In RoMANS, a weight of evidence approach was used to apportion deposited S and N to source regions. It included both simple qualitative spatial data analysis and quantitative hybrid receptor modeling. Results indicated that the species concentrations generally increased in ROMO when air transport was from the east. The final inorganic N deposition apportionment budget indicated that the source areas responsible for most N deposition in ROMO during spring were northeastern Colorado (~ 40%) and the Denver/Front Range region (~ 25%). In contrast, during summer, N source regions were more varied, with about 25% from southwesterly transport, 20% from local source areas, 13% from western Colorado, and 8% from the greater Denver area (Malm et al. 2009).

A detailed N deposition budget was calculated by Benedict et al. (2013a) at Loch Vale over a one-year period from November, 2008 to November, 2009. The estimated total reactive N deposition was 3.65 kg N/ha/yr, with highest deposition in July and lowest deposition in January. The two largest contributors were wet deposition of NO_3^- and NH_4^+ , which together accounted for 54% of the total. The next two largest contributors were wet deposition of organic N and dry deposition of NH_3 (combined 37% of total). These latter two deposition pathways are rarely included in deposition budgets, including those used to assess critical loads and their exceedances (Benedict et al. 2013a).

Wet deposition is the primary process for depositing N in ROMO during both spring and summer. The RoMANS study found that atmospheric N contributions to ROMO were more than twice as high during summer as compared to spring. The major contributors of reactive N during summer were wet NH_4^+ deposition (34%), followed by wet NO_3^- (28%), dry NH_3 (16%), and wet organic N (12%; Beem et al. 2010).

Nartz et al. (2011) used ice core samples from Upper Fremont Glacier, Wyoming, as proxy records for the chemical composition of upwind emissions and atmospheric deposition. Ice core NO_3^- and SO_4^{2-} content reflected the sharp increase in emissions that occurred between about 1950 and the mid-1970s. The isotope $\delta^{15}\text{N}$ values in ice cores all fell within the expected range from motor vehicle emissions, suggesting that motor vehicles account for much of the N deposited from the atmosphere to receptors in this area.

Snowfall that accumulates in the seasonal snowpack generally accounts for the majority of the annual precipitation in headwater areas of the Rocky Mountains (Ingersoll et al. 2005, Western Regional Climate Center 2004). Atmospherically deposited contaminants accumulate in these snowpacks and are released to soils and surface waters during the snowmelt. The U.S. Geological Survey (USGS) monitors the chemistry of the Rocky Mountain snowpack at multiple locations from New Mexico to Montana. Concentrations of NH_4^+ in snow are highest in the vicinity of Yellowstone (YELL) and

Grand Teton (GRTE) national parks. Nitrate and SO_4^{2-} concentrations in snow tend to be highest in northern Colorado and southern Wyoming (Ingersoll et al. 2005, 2009, 2010, 2011).

Snow accounts for about 50 to 80% of total precipitation at high-elevation snowpack telemetry (SNOTEL) sites in Colorado. However, NO_3^- concentrations in snow at Loch Vale in ROMO are only about half of the concentrations in rain (Fenn et al. 2009). Thus, both snow and rainfall are important contributors to wet NO_3^- deposition in ROMO.

The concentration of N in lichen tissue can be used as an index of N deposition. For example, McMurray et al. (2013) used epiphytic lichens to monitor N deposition near natural gas drilling operations in the Wind River Range, Wyoming. The concentration of N in *Usnea lapponica* was strongly correlated ($r = 0.96$) with N deposition measured in forest throughfall.

Based on assumed pre-industrial background atmospheric N deposition of 0.5 kg N/ha/yr in 1900 and two decades of measured wet N deposition at the Loch Vale NADP/NTN site, Baron (2006) reconstructed the N deposition history of ROMO, based on correlation with NO_x emissions data for Colorado and 10 other western states. The estimated mean annual wet N deposition for the period 1950-1964 was about 1.5 kg N/ha/yr. This time period corresponded with the alteration of lake diatom assemblages in ROMO that were attributed to N deposition (Baron 2006).

Retention of S in alpine watersheds of ROMO, such as Loch Vale, is limited (Michel et al. 2000). As a consequence, changes in S deposition should be detected in the chemistry of surface water within a short time period. Total S losses in drainage water from the Loch Vale basin were considerably higher than wet S inputs, and ranged from 3.3 to 4.2 kg S/ha/yr (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 S emissions and deposition since 1984 is obscured by these internal watershed sources of S (Baron et al. 1995).

Grenon and Story (2009) reported increasing trends in wet deposition of NH_4^+ within at least one season of the year since the 1980s for all monitoring sites in the northern Rocky Mountains region. This result agreed with analyses reported by Ingersoll et al. (2008) and Lehmann et al. (2005). This trend of increasing NH_4^+ wet deposition has occurred across much of the western United States, likely due in part to increased agricultural emissions of NH_x . The concentrations of NO_3^- in precipitation at NADP/NTN monitoring sites in the Rocky Mountains have also generally increased over time, especially near urban areas (Fenn et al. 2003b, Lehmann et al. 2005). In contrast, wet S deposition decreased between 1985 and 2002 at NADP/NTN sites throughout the western United States (Lehmann et al. 2005), likely due to reduced power plant emissions.

In 2004, the Colorado Department of Public Health and Environment (CDPHE), the NPS, and the EPA signed a Memorandum of Understanding (MOU), known as the Rocky Mountain Park Initiative, to address atmospheric N deposition and related air quality issues in ROMO. Subsequently, a Nitrogen Deposition Reduction Plan (NDRP) was issued in 2007. The MOU agencies endorsed a wet deposition level of 1.5 kg N/ha/yr as a threshold for identifying adverse

ecosystem effects from N deposition in ROMO (<http://www.colorado.gov/cdphe/rmnpinitiative>). To achieve this threshold, a glidepath approach was selected to accomplish gradual improvement over time. In 2012, the calculated five-year average (2008-2012) wet N deposition was 2.9 kg N/ha/yr, which was slightly (0.2 kg N/ha/yr) higher than the 2012 interim milestone (Morris et al. 2014).

Toxics

Various studies have identified organochlorine compounds in remote arctic or alpine ecosystems (cf., Blais 2005, Heit et al. 1984, Simonich and Hites 1995, Wania and Mackay 1996). Compounds detected commonly include dichlorodiphenyltrichloroethane (DDT), hexachlorocyclohexane (HCH), hexachlorobenzene (HCB), and polychlorinated biphenyls (PCBs). Such compounds tend to concentrate in aquatic biota in response to biomagnification processes and have been associated with disruption of endocrine systems in aquatic species. Based on results of earlier studies in remote mountainous areas of Europe (Grimalt et al. 2001) and Canada (Blais et al. 1998), Mast et al. (2006) suggested that some remote areas of the Rocky, Cascade, and Sierra Nevada mountains might be susceptible to organochlorine compound accumulation due to low temperatures, high precipitation, and proximity to agricultural and urban source areas. Because relatively little is known about atmospheric deposition of organochlorines to these sensitive ecosystems, USGS in cooperation with NPS, conducted a baseline study in ROMO and GLAC (Mast et al. 2006).

Mast et al. (2006) measured organochlorine compounds and current-use pesticides in snow and lake sediment samples collected from GLAC and ROMO. Measurements of snow chemistry can provide important information regarding atmospheric deposition. The pesticides most frequently found in snow samples were endosulfan, dacthal, and chlorothalonil. In sediment samples, low concentrations of DDE and DDD were commonly found. These are degradation products of DDT that has been banned for use in the United States since 1972. Sediment core measurements for Mills Lake in ROMO indicated that atmospheric deposition of DDT to high-elevation ecosystems in ROMO has been decreasing since the 1970s. Dacthal and endosulfan were detected in nearly all snow samples analyzed, confirming that these current-use pesticides are being deposited in high-elevation ecosystems and accumulating in lake sediments (Mast et al. 2006).

The Western Airborne Contaminant Assessment Project (WACAP) studied airborne contaminants in a number of national parks in the western United States, including three parks in the ROMN (ROMO, GLAC, and GRSA) from 2002 to 2007. In ROMO, the Mills Lake watershed on the east side of the Continental Divide had higher semivolatile organic compounds (SOC) and mercury (Hg) fluxes in snow and lake sediment, as compared with the Lone Pine Lake watershed on the west side. This pattern suggested that the Continental Divide might serve as a geographic barrier protecting the west side of ROMO from pollutants travelling with airflows from agricultural sources that occur to the east. However, air monitors on either side of the Continental Divide did not detect obvious differences in SOC concentrations between the east and the west. Dominant SOC in the ROMN were polycyclic aromatic hydrocarbons (PAHs), dacthal, endosulfans, hexachlorobenzene, hexachlorocyclohexane- α , and hexachlorocyclohexane- γ (HCH). In the air at GRSA, there were moderate concentrations of dacthal, endosulfans, hexachlorobenzene, hexachlorocyclohexane- α , and

chlordanes and high concentrations of hexachlorocyclohexane- γ and PAHs, as compared with other WACAP parks (Landers et al. 2008).

Atmospheric input of toxic materials to high-elevation ecosystems has been evaluated through study of snowpack chemistry. Hageman et al. (2010) analyzed snowpack samples collected from 56 remote alpine and arctic locations in eight western national parks, including ROMO and GLAC, during the period 2003 to 2005. Four current use pesticides (CUPs) were commonly measured at all sites and years: dacthal, chlorpyrifos, endosulfans, and HCH- γ . The relative pesticide concentration profiles were consistent from year to year, but unique for individual parks, indicating the importance of regional sources. The historic use pesticides (HUPs) were strongly correlated with regional cropland presence. The amount of CUPs used in the regions located one day upwind, based on mass back trajectory analysis, helped to explain the distribution of CUPs among the study parks.

Acidification

The network rankings developed by Sullivan et al. (2011b) in a coarse screening analysis for acid Pollutant Exposure, Ecosystem Sensitivity to acidification, and Park Protection yielded an overall network acidification Summary Risk ranking for the ROMN that was relatively high among networks. This was despite the low levels of S and N emissions and deposition at ROMN parks. The overall level of concern for acidification effects on I&M parks within this network was judged by Sullivan et al. (2011b) to be High. This judgment was based largely on high presumed watershed sensitivity due in part to high elevations and steep slopes. While rankings are an indication of risk, park-specific data, particularly data on ecosystem sensitivity, are needed to fully evaluate risk from acidification.

All parks in this network were ranked by Sullivan et al. (2011b) in the lowest quintile among all I&M national park lands (including GRSA) to the middle quintile (including ROMO) for acid Pollutant Exposure (Table 2). The three large parks (GLAC, GRSA, and ROMO) in the network were all ranked in the top quintile for Ecosystem Sensitivity to acidification (Table 2). FLFO was ranked in the second highest quintile for Ecosystem Sensitivity to acidification.

Ecosystem sensitivity to acidification rankings take into account elevation and land slope, which influence the degree of acid neutralization provided by soils and bedrock within the watershed. Watershed slope within and among the parks in the ROMN varies, from HUCs having average slope less than 10° to as high as 40° to 50° in substantial portions of GLAC and ROMO.

There are many high-elevation lakes and first-order streams in GLAC and ROMO and also some in GRSA. Nearly all streams in GLAC, and especially in ROMO, are first through third order and occur on steep terrain. The vast majority are first-order streams, with relatively small drainage areas. Such streams tend to be more likely to be sensitive to acidification than the larger, higher-order streams found at lower elevation.

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

Park Name ²	Park Code	Estimated Acid Pollutant Exposure	Estimated Ecosystem Sensitivity to Acidification
Florissant Fossil Beds	FLFO	Moderate	High
Glacier	GLAC	Low	Very High
Grant-Kohrs Ranch	GRKO	Very Low	Very Low
Great Sand Dunes	GRSA	Very Low	Very High
Little Bighorn Battlefield	LIBI	Low	Very Low
Rocky Mountain	ROMO	Moderate	Very High

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

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

Within the ROMN, episodic acidification attributable to atmospheric S and N deposition may be of concern primarily in ROMO. Moderate levels of N deposition, coupled with high watershed sensitivity to acidification, may have contributed to some episodic acidification of surface waters in this park at existing levels of N deposition. To date, however, neither episodic nor chronic acidification has been documented in the park (Sullivan et al. 2005).

The National Acid Precipitation Assessment Program (NAPAP) State of Science and Technology Reports and the Integrated Assessment (NAPAP, 1991) concluded that high-elevation areas of the West contained many of the watersheds most sensitive to the potential effects of acidic deposition in the United States. These highly sensitive surface waters are especially numerous in ROMO and elsewhere within the Colorado Front Range. The acid-base chemistry of lake and stream waters in the ROMN appears to be 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 (Sullivan 2000). High-elevation areas in the ROMN can contain large areas of exposed bedrock and alpine meadows, with little soil or vegetative cover to neutralize acidic inputs. Sensitivity to adverse effects from acidic deposition in the Rocky Mountains varies widely since the individual ranges that comprise the Rocky Mountains are discontinuous, with highly variable geological composition. Lakes having lowest acid neutralizing capacity (ANC) occur in clusters, such as in the Bitterroot Mountains in Montana and the Wind River Range in Wyoming (Schoettle et al. 1999). For that reason, assessments of the sensitivity of Rocky Mountain aquatic resources to acidification should be specific to individual ranges (Sullivan and Eilers 1994, Turk and Spahr 1991). For example, high concentrations of base cations, alkalinity, and silica occur in the upper Colorado River basin, a portion of ROMO 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).

Nanus et al. (2009) described the sensitivity of high mountain lakes to acidification in GLAC, ROMO, and GRSA within the ROMN and of YELL and GRTE within the Greater Yellowstone Network (GRYN). They developed statistical relations between lake ANC and basin characteristics for 151 surveyed lakes. Results were confirmed through lake sampling in 2004 at 58 lakes. Lakes having ANC below 100 $\mu\text{eq/L}$ were generally located in watersheds above 3000 m elevation with < 30% of the watershed area having NE aspect, and with >80% of the watershed area having bedrock with low buffering capacity. The most acid-sensitive lakes were generally located in ROMO and GRTE (Nanus et al. 2009).

Data from over 12,500 streams and lakes were used by the Critical Loads of Atmospheric Deposition Science Committee (CLAD) of the NADP (<http://nadp.sws.uiuc.edu/committees/clad/>) to develop steady-state critical loads for acidity of surface waters based on multiple approaches for estimating base cation weathering: modified F-factor, regional regression model, and MAGIC model. Water quality data were obtained from a variety of sources including EPA Long Term Monitoring (LTM) sites, lake surveys, EMAP Assessments, and National Stream Surveys; USGS; NPS Vital Signs program; and USFS air program. The average water quality measurements from the most recent five

years of data were used for sites with long-term water monitoring records. The CLAD database included 3 sites in GRSA, 9 sites in GLAC, and 26 sites in ROMO. The CL estimates were as low as 0 eq S/ha/yr at ROMO, but were above 377 eq S/ha/yr at all modeled sites in the other two parks.

Studies Conducted in ROMO

A lake and stream sampling program was conducted in the early 1980s by the U.S. Fish and Wildlife Service in four large watersheds of ROMO (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 baseflow conditions. The 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 basins. 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, and had surface waters with the lowest pH and base cation concentrations of the subbasins studied. 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. Three of them (Tyndall Gorge, Loch Vale, and Ypsilon Lake) received a large percentage of their drainage water from snowmelt. 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. The 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).

Table 3. Population estimates of water chemistry percentiles for selected lake populations in the Rocky Mountains¹. The 1st and 5th percentiles (P₁, P₅) are presented for pH, ANC (µeq/L), and sum of base cation concentrations (SBC; µeq/L) and the 95th and 99th (P₉₅, P₉₉) percentiles are shown for SO₄²⁻ (µeq/L) and NO₃⁻ (µeq/L). The median (P₅₀) and 90th percentiles are shown for dissolved organic carbon (DOC; mg/L).

Population	n ²	N ³	pH		ANC		SBC		SO ₄ ²⁻		NO ₃ ⁻		DOC	
			P ₁	P ₅	P ₁	P ₅	P ₁	P ₅	P ₉₅	P ₉₉	P ₉₅	P ₉₉	P ₉₅	P ₉₉
Idaho Batholith	88	937	6.34	6.42	21	33	30	45	30	43	3	4	1.2	2.4
NW Wyoming	38	648	6.56	6.56	38	38	64	66	41	2,909	13	32	1.0	4.8
Colorado Rockies	121	1,173	6.02	6.65	25	42	58	80	915	2,212	10	13	1.3	5.7

¹ Data from (Landers et al., 1987); excluding Fern Lake (4D3-017) in NW Wyoming which was naturally acidic

² Number of lakes sampled

³ Number of lakes in the population represented by the sampling

Table 4. Population estimates of the percentage of lakes in selected subregions of the Rocky Mountains with ANC and NO₃⁻ within defined ranges. (Source: Data from Landers et al. 1987)

	ANC (µeq/L)			NO ₃ ⁻ (µeq/L)	
	< 0	< 25	< 50	> 5	> 10
Idaho Batholith	0	2.0	23.6	4.6	3.9
NW Wyoming ¹	0	2.3	12.8	22.8	8.9
Colorado Rockies	0	0.9	5.5	9.8	1.8

¹ Excluding Fern Lake (4D3-017) which is a naturally acidic lake

The isotopic study of Kester et al. (2003) identified precipitation and the watershed as being the two major sources for SO₄²⁻ in Loch Vale. The relative proportion of the SO₄²⁻ sources varied over time in relation to the onset and duration of snowmelt. Based on the potential presence of pyrite deposits, geochemical processes may be important to the S budget in this watershed.

In the EPA's Western Lakes Survey (WLS), mid-lake NO₃⁻ concentrations in most lakes sampled in the ROMN were near 0 during fall (Landers et al. 1987). However, fall NO₃⁻ concentrations were high in some areas in the Rocky Mountains (Table 3). For example, nearly one fourth of the lakes in northwestern Wyoming had NO₃⁻ > 5 µeq/L and almost 10% had NO₃⁻ > 10 µeq/L (Sullivan and Eilers 1994; Table 4). In the Colorado Rocky Mountain subregion, about 10% of the lakes had fall NO₃⁻ concentrations above 5 µeq/L (Table 4). Analyses of 1985 WLS fall samples from 22 lakes in ROMO and 14 lakes in adjacent wilderness areas (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. The minimum ANC value measured in this subpopulation was 19 µeq/L. These ANC values are similar to ANC values for other highly 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.

The wide range in SO₄²⁻ concentrations indicates that some lakes in the area may be receiving S inputs from geologic sources, rather than purely atmospheric inputs. (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. The WLS sampled lakes during the fall, when NO₃⁻ concentrations would be expected to be low compared to expected concentrations during spring snowmelt.

Musselman et al. (1996) conducted a synoptic survey of surface water chemistry in the mountainous areas along the eastern edge of the Continental Divide in Colorado and southeastern Wyoming that are exposed to relatively high (by Rocky Mountain standards) deposition of N. A total of 267 high-elevation lakes situated in watersheds having high percentage of exposed bedrock or glaciated landscape were selected for sampling. None of the lakes were chronically acidic (ANC < 0), although several had ANC < 10 µeq/L, and more than 10% of the lakes had ANC < 50 µeq/L.

In 1999, Clow et al. (2003) resampled 69 of the lakes located within western national parks that had been surveyed earlier in the WLS. In general, lake SO₄²⁻ concentrations decreased between 1985 and 1999, in response to regional decreases in SO₂ emissions and S deposition. In addition, lake NO₃⁻

concentrations were slightly lower in 1999, probably at least in part because rain prior to the 1985 survey may have caused elevated NO_3^- concentration in some lakes (Clow et al. 2003). This finding reinforced the idea that changes in precipitation should be included in an assessment of water chemistry trends (cf., Bayley et al. 1992, Webster and Brezonik 1995). Lake ANC was relatively consistent between the two surveys in most parks. However, ANC increased in lakes sampled in ROMO because concentrations of strong acid anions (SO_4^{2-} and NO_3^-) decreased while base cations remained stable.

A great deal of research has been conducted on the interactions between atmospheric pollutants and water quality at an integrated watershed study site at the Loch Vale in ROMO. Biogeochemical and hydrological processes have been studied intensively at Loch Vale since 1983 (e.g., Baron 1992, Baron and Campbell 1997, Baron et al. 2009, Campbell et al. 1995a, Denning et al. 1991). Loch Vale watershed is a 7-km² basin situated along the Continental Divide in the southeastern portion of ROMO, located 80 km northwest of Denver. The watershed ranges in elevation from about 3,100 m to 4,000 m.

Given these characteristics, 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 ROMO and elsewhere in the Front Range) to atmospheric N deposition, it is important to consider a variety of hydrologic and biogeochemical processes that occur in different parts of the basin. Campbell et al. (1995a) 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 $\mu\text{eq/L}$, respectively. Total N export was approximately equivalent to atmospheric inputs, assuming about 25% evapotranspiration. Nitrate concentrations in individual samples of stream water ranged from 12 $\mu\text{eq/L}$ in late summer to 39 $\mu\text{eq/L}$ during snowmelt.

Baron et al. (2009) explored the relationships between recent temperature warming and stream NO_3^- concentrations at Loch Vale. The mean annual NO_3^- concentration since 2000 was about 50% higher than during the period 1991-1999. Precipitation was below average between 2000 and 2006, and both summer and fall air temperature increased steadily since 1991. Baron et al. (2009) concluded that recent increases in stream NO_3^- concentrations at Loch Vale were caused largely by warmer summer and fall temperatures that are melting glaciers in the watershed.

The Loch Vale watershed can, for all practical purposes, be considered N-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 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 (Baron et al. 1994, Campbell et al.

1995a). 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).

A number of factors predispose watersheds in ROMO, including Loch Vale, and elsewhere throughout the Front Range to potential acidification in response to N deposition. These include:

- Steep watershed gradient
- Short hydrologic residence time of lake waters
- 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.

Baron et al. (1986) investigated metal and diatom stratigraphy, and inferred pH profiles of four subalpine lakes in ROMO. They found no evidence of historical influence on pH attributable to atmospheric deposition at that time. Other paleolimnological studies of Rocky Mountain lakes reported similar results: stratigraphy of metals (primarily lead) exhibited temporal dynamics related to the increase and decline of precious metal mining in the region, but these were asynchronous with other metal or biological indicators of acidification (Wolfe et al. 2003). Both the study by Wolfe et al. (2003) and a study by Saros et al. (2003) showed no paleolimnological evidence of acidification of high-elevation Rocky Mountain lake waters over time, but did show evidence of eutrophication from atmospheric N deposition.

DayCent-Chem, a model that simulates the daily dynamics of plant production, soil organic matter, cation exchange, mineral weathering, elution, stream discharge, and stream solute concentrations, was able to simulate daily stream chemistry dynamics over 13 years in an alpine watershed in the Colorado Front Range (Hartman et al. 2007). Hindcasts of stream chemical dynamics back to 1900 suggested historical changes in pH coincident with maximum SO₂ emissions in the late 1960s and early 1970s. Model simulations suggested annual mean pH values decreased to the range of 5.6 to 5.8 during the years of maximum regional SO₂ emissions, and have since recovered to circumneutral values. Simulated ANC responded to both SO₂ and NO_x emissions, decreasing to annual values of 20 to 25 µeq/L during years of highest SO₂ or NO_x emissions, compared with current mean annual ANC values near 50 µeq/L (Hartman et al. 2005a).

The DayCent-Chem model was also used to project a timeline to acidification for an alpine watershed in ROMO (Hartman et al. 2005b). At ambient levels of N deposition of 4 to 6 kg N/ha/yr, acidification was not projected to occur over 48 years of simulation. Higher assumed N deposition contributed to simulated episodic acidification over time at a future deposition level of 7.0 to 7.5 kg N/ha/yr (Hartman et al. 2005b). MAGIC model simulation results suggested that a sustained N deposition load of 12.2 kg N/ha/yr would be required over a period of 50 years to cause chronic acidification of the Andrews Creek watershed in ROMO to ANC = 0 µeq/L (Sullivan et al. 2005).

Studies Conducted in GLAC

The central portion of GLAC is dominated by two mountain ranges that run northwest to southeast and contain many small glaciers and snowfields. Extensive portions of both ranges lie above timberline (~2,000 m) and many of the peaks extend above 2,800 m. Elevation ranges from about 950 m along the western boundary to 3,190 m in the central mountains, and back down to about 1,370 m along the eastern boundary. Glaciers provide substantial amounts of base cations to many drainage waters in this park.

Many of the lakes and streams within GLAC have characteristics generally associated with acid sensitivity. They tend to be high in elevation, with little or no soil development in their watersheds, have steep slopes and flashy hydrology, and are hydrologically dominated by spring snowmelt. The majority of these surface waters, however, are actually not at all sensitive to acidification from acidic deposition, due in part to the preponderance of glaciers within their watersheds. Glaciers contribute buffering in the form of base cations (calcium [Ca^{2+}], magnesium [Mg^{2+}], sodium [Na^+], potassium [K^+]) to drainage waters in GLAC in sufficient quantity to neutralize any amount of SO_4^{2-} and NO_3^- that might be reasonably expected to be deposited from acidic deposition (Peterson et al. 1998). There are some waters in the park that receive only modest contributions of base cations, however. These do not receive glacial meltwater contribution to any significant extent and are situated in watersheds with relatively inert bedrock. These sensitive waters are relatively rare within GLAC.

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

The EPA's WLS sampled 5 lakes in GLAC and 10 other lakes in surrounding areas in the fall of 1986 (Landers et al. 1987). Measured values of selected important physical and chemical variables are listed in Table 5 for these fifteen lakes and their watersheds. The lowest pH value measured in the park was 7.1, although three of the lakes in surrounding areas had pH between 6.5 and 7.0. One of the lakes having lowest pH (6.6) contained significant natural organic acidity (dissolved organic carbon [DOC] = 10 mg/L); the others were low in pH as a consequence of their dilute chemistry. Sulfate concentrations in lake water were very low in lakes having low base cation concentrations. For example, the four lakes with total base cation concentrations less than 100 $\mu\text{eq/L}$ all had SO_4^{2-} concentrations in the range of 5.7 to 10.1 $\mu\text{eq/L}$. Such concentrations of SO_4^{2-} are approximately what would be expected, based on SO_4^{2-} concentration in the precipitation (~6 to 8 $\mu\text{eq/L}$), negligible dry deposition of S, and 30% to 50% evapotranspiration. These lakes are moderately to highly acid-sensitive, with ANC values of 21 to 84 $\mu\text{eq/L}$, although the two most sensitive were located outside the park boundaries. Many other lakes inside and outside the park had moderately elevated concentrations of SO_4^{2-} , in the range of 20 to 52 $\mu\text{eq/L}$. These relatively high concentrations of SO_4^{2-} are not attributable to atmospheric S deposition. They reflect geological sources of S in drainage

waters, as also evidenced by the much higher concentrations ($> 500 \mu\text{eq/L}$) of base cations in all of the lakes that had SO_4^{2-} concentration greater than $20 \mu\text{eq/L}$. Based on these data, it appears that GLAC and surrounding areas contain lakes that exhibit a mixture of acid sensitivities. Some lakes that have low concentrations of SO_4^{2-} (less than about $10 \mu\text{eq/L}$) that can be reasonably attributed to atmospheric deposition inputs also have low concentrations of base cations. These lakes tend to be relatively acid-sensitive, and many have ANC values below $100 \mu\text{eq/L}$. The lowest measured ANC in the park was $79 \mu\text{eq/L}$. Other lakes are characterized by higher concentrations of base cations and SO_4^{2-} of geologic origin. These lakes are not acid-sensitive and have ANC values greater than $500 \mu\text{eq/L}$. Nitrate concentrations were generally below $5 \mu\text{eq/L}$ in WLS lakes sampled in GLAC. One lake exhibited relatively high NO_3^- concentration ($11.6 \mu\text{eq/L}$) but this lake was not acid-sensitive (Peterson et al. 1998).

Table 5. Results of lakewater chemistry analyses by the Western Lake Survey for selected variables in GLAC and adjacent areas. (Source: Peterson et al. 1998)

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)	SBC (µeq/L)	DOC (mg/L)
<u>Lakes within GLAC</u>											
No Name	4C3-004	2.8	88	1930	8.1	1210	28.6	11.6	929	1212	0.4
Feather Woman L.	4C3-010	3.7	44	2298	7.3	79	5.7	4.9	49	81	0.3
Harrison L.	4C3-011	162.0	5475	1126	8.0	543	32.1	4.5	375	613	0.5
Cobalt L.	4C3-013	4.4	62	2003	7.1	83	10.1	0.5	50	93	0.7
Glenns L.	4C3-062	104.8	4302	1482	8.1	1142	48.7	3.0	774	1210	0.7
<u>Lakes Outside GLAC</u>											
	4C3-053	3.7	20	1979	6.6	21	8.0	0.1	29	57	1.5
	4C3-016	5.4	88	1932	8.1	1388	20.1	0.0	1096	1393	1.0
	4C3-017	6.2	173	1828	8.0	1209	32.3	0.7	832	1218	0.8
	4C3-021	1.8	108	2104	7.9	492	27.4	1.5	768	1092	0.3
	4C3-022	8.7	51	2050	7.5	360	10.0	2.1	295	386	0.8
	4C3-026	167.9	2188	1229	8.3	1409	52.4	0.1	965	1435	4.5
	4C3-055	2.3	7	1921	7.6	426	8.0	0.7	360	453	4.3
	4C3-060	12.7	77	1228	6.6	152	3.0	0.0	81	213	10.0
	4C3-031	6.4	54	2006	6.9	72	8.2	0.1	40	78	1.2
	4C3-059	1.8	31	2226	7.4	292	10.5	4.3	170	319	2.7

Ellis et al. (1992) monitored water quality of eight small backcountry lakes and five large front country lakes in GLAC. The objective was to establish a water quality baseline for the park. Data were collected during the period 1984 to 1990. The backcountry lakes were located in remote alpine headwater areas across the various regions and geologies of the park. Three of the lakes (Cobalt, Snyder, and Upper Dutch) had alkalinity less than about 200 $\mu\text{eq/L}$. Cobalt Lake had the lowest alkalinity (~ 100 $\mu\text{eq/L}$ on average) and specific conductance (~ 10 $\mu\text{S/cm}$) of the study lakes and would be expected to be sensitive to episodic effects of acidic deposition if S or N deposition increased substantially in the future. The study lakes other than Cobalt, Snyder, and Upper Dutch would not be sensitive to acidification in the foreseeable future (Peterson et al. 1998). Measured pH values in Cobalt, Snyder, and Upper Dutch Lakes were generally in the range of about 6.0 to 7.0, although pH values as low as 5.5 were measured in all three lakes. Dissolved organic carbon concentration was low in all three lakes, in the range of 0.75 to 1.5 mg/L (Ellis et al. 1992). Thus, the amount of organic acidity in the lakes was low.

Cobalt Lake is situated in the southeastern portion of GLAC at an elevation of 2,000 m. It lies immediately below a very steep alpine ridge, and therefore receives runoff that has limited contact with geological materials prior to entering the lake. Based on analysis of nine samples, mean SO_4^{2-} concentration was 9.6 $\mu\text{eq/L}$ and mean sum of base cation concentration was 85.2 $\mu\text{eq/L}$. These values suggest that watershed sources of S were not significant and that there were insufficient base cations to neutralize substantial amounts of acidic deposition (Peterson et al. 1998).

Studies of Potential Episodic Acidification of Aquatic Ecosystems

Temporal variability in surface water and soil solution chemistry and patterns in nutrient uptake by terrestrial and aquatic biota influence acidification processes and pathways. Chemical conditions are constantly changing in response to episodic, seasonal, and inter-annual cycles and processes. In particular, climatic fluctuations that govern the amount and timing of precipitation inputs, snowmelt, vegetative growth, depth to groundwater tables, and evapoconcentration of solutes influence soil and surface water chemistry and the interactions between pollution stress and sensitive aquatic and terrestrial biological receptors in the ROMN.

During hydrologic episodes, which are driven by rainstorms and/or snowmelt events, both discharge and water chemistry change, sometimes substantially. These processes have been well studied in ROMO. The most important factor governing watershed sensitivity to episodic acidification is the pathway followed by snowmelt water and stormflow water through the watershed. These pathways determine the extent of acid neutralization provided by soils and bedrock. Watersheds having high-elevation, steep topography, extensive areas of exposed bedrock, deep snowpack accumulation, and shallow, base-poor soils tend to be most sensitive to episodic acidification. These watershed conditions are common within ROMO and GLAC and lead to an increase in the proportion of flow derived from water that has moved laterally through the surface soil without infiltration to deeper soil horizons (Wigington et al. 1990). The chemistry of this drainage water typically reflects the lower pH, higher organic content, and lower ANC of upper soil horizons (Sullivan 2000).

Available data from intensive study sites in the Rocky Mountain region (e.g., Loch Vale in ROMO and the Glacier Lakes Watershed, WY) suggested that episodic depression of stream pH may be

more pronounced than for lakes. However, there are no available systematic regional stream chemistry data with which to assess regional sensitivity of streams to acidic deposition in the ROMN. Spatial variability in water chemistry can be considerable in lakes, and this complicates efforts to quantify the magnitude of episodic effects (Gubala et al. 1991). Moreover, synoptic lake surveys are typically conducted during the summer or autumn "index period," during which time lakewater chemistry exhibits relatively low temporal variability. Acid anion concentrations in most Rocky Mountain lakes are low during fall, but can be higher during snowmelt (Williams and Melack 1991). Although summer or autumn is an ideal time for surveying lakewater chemistry in terms of minimizing variability, samples collected under low-flow conditions provide little relevant data on episodic processes, and in particular on the dynamics or importance of N as an agent of acidification. Nitrate concentrations in lakewater are elevated during the autumn season only in lakes having watersheds that exhibit fairly advanced symptoms of N saturation (Stoddard 1994).

Episodic acidification due to atmospheric deposition is most commonly associated with N deposition, and effects tend to be most pronounced during snowmelt. However, snowmelt can flush into surface waters N that was deposited from the atmosphere to the snowpack and also N that was mineralized within the soil under the snowpack during winter (Campbell et al. 1995b). A substantial component of the NO_3^- flux may have been derived from mineralization of organic N (Ley et al. 2004). Much of the N released from the snowpack during the melting period is retained in underlying soils and only a component of that is flushed to surface waters. Where soils are sparse, as in alpine regions of the ROMN, much of the snowpack N is flushed to surface waters. Even though there is evidence through use of isotopic tracers that much of the N in stream water was cycled microbially, leaching of N from the snowpack can cause water acidification of alpine streams (Campbell et al. 2002, Williams and Tonnessen 2000).

In ROMO, episodic acidification is a concern for surface waters throughout high-elevation areas. In the Green Lakes Valley, located below Niwot Ridge near ROMO, atmospheric N deposition has been shown to contribute to episodic stream water acidification (Williams and Tonnessen 2000). Episodic effects probably occur to a limited extent in ROMO.

Baron et al. (2011b) estimated empirical CLs to protect against increased NO_3^- concentration in lake waters in the Rocky Mountains and Sierra Nevada in the range of 1.0 to 3.0 kg N/ha/yr. This was taken as the range of CL to protect against nutrient enrichment in western mountain lakes.

Based on measurements of microbial biomass, CO_2 flux through the snowpack, and soil N pools, Williams et al. (1996b) concluded that N cycling under the snowpack in Colorado during the winter and spring was sufficient to supply the NO_3^- measured in stream waters. Brooks et al. (1996) investigated soil N dynamics throughout the snow-covered season on Niwot Ridge, CO. Sites with consistent snow cover had a 3 to 8 cm layer of thawed soil under the snowpack for several months before snowmelt began. Nitrogen mineralization in this thawed layer contributed to reactive N pools that were significantly larger than the pool of N stored in the snowpack. As snowmelt began, soil inorganic N pools decreased sharply, concurrent with a large increase in microbial biomass N. As snowmelt continued, both microbial N and soil inorganic N decreased, presumably due to increased demand by growing vegetation (Brooks et al. 1996).

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 ($\text{ANC} \leq 0$). Similarly, Denning et al. (1991) showed a dramatic decline in the ANC of The Loch between mid-April and mid- to late-May in 1987 and 1988, to ANC values as low as 28 $\mu\text{eq/L}$ (and pH around 6.2). This was in spite of the fact that meltwater ANC dropped to between 0 and -10 $\mu\text{eq/L}$ for extended periods during snowmelt (Denning et al. 1991). If a large component of the snowmelt had been transported directly to surface waters, the latter would have 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.

Williams et al. (1996b) sampled 53 ephemeral streams in the Green Lakes Valley of Colorado during snowmelt in 1994. They also sampled an additional 76 sites from the central Colorado Rocky Mountains to the Wyoming border in 1995. Nitrate concentrations in stream water during snowmelt ranged to 44 $\mu\text{eq/L}$ in the Green Lakes Valley and during the growing season ranged to 23 $\mu\text{eq/L}$ in the regional sampling conducted in 1995. Tundra areas had significantly lower NO_3^- concentrations than talus and bedrock areas, suggesting that tundra ecosystems were still N-limited and that nitrification combined with limited plant uptake accounted for the high concentrations of NO_3^- observed in waters draining talus and bedrock areas (Williams et al. 1996b).

Brooks et al. (1995) reported inputs to the soil inorganic N pool at Niwot Ridge due to mineralization and nitrification (17 to 20 kg N/ha) under deep snowpack that were an order of magnitude higher than inputs directly from snowmelt (~ 1.5 kg N/ha). Mineralization varied with 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, mineralization was lower and more variable (5 to 15 kg N/ha, Brooks et al. 1995). The severity with which the soils freeze may be an important determining factor of the amount of N mineralization. Highest mineralization inputs were found under a shallow snowpack that experienced a severe freeze, followed by an extended period of snow cover.

Time-intensive discharge and chemical data for two alpine streams in Loch Vale watershed identified strong seasonal control on streamwater NO_3^- concentrations (Campbell et al. 1995a). In spite of the paucity of soil cover, the chemical composition of streams was regulated much as in typical forested watersheds. Soils and other shallow groundwater matrices such as boulder fields appeared more important in controlling surface water chemistry than their abundance would indicate (Campbell et al. 1995b). Spring streamwater NO_3^- concentrations ranged to 40 $\mu\text{eq/L}$, compared with summer minimum values near 10 $\mu\text{eq/L}$. Elution of acidic waters from snowpack along with dilution of base cations originating in shallow groundwater caused episodes of decreased ANC in alpine streams (Campbell et al. 1995a).

It does not appear that chronic acidification has occurred in the ROMN, although episodic acidification has been reported for lakes throughout the Colorado Front Range (Williams and Tonnessen 2000). The data that would be needed for determining the extent and magnitude of

episodic acidification have not been collected to a sufficient degree in acid-sensitive areas of the ROMN to support regional assessment of episodic acidification (Sullivan 2000).

Effects of Acidification on Biota

In acid-sensitive lakes in the western United States, one focus of studies on aquatic effects of acidification has been on native cutthroat trout (*Oncorhynchus clarkii*). Native trout are sensitive to short-term increases in acidity (Woodward et al. 1989). It is important to note, however, that many high-elevation western lakes and streams were historically fishless. The top predators in such aquatic ecosystems were often amphibians or crustaceans. Thus, even though cutthroat trout might be considered native to the region, this species is not necessarily native to a particular lake or stream.

Episodic acidification can be the limiting condition for aquatic organisms in streams or lakes that exhibit chronic chemistry that is suitable for aquatic biota, but nevertheless experience occasional episodic acidification (cf., Wigington et al. 1993). The EPA's Episodic Response Project (ERP) suggested that the chemical response to acidification that has the greatest effect on biota is usually increased inorganic aluminum (Al_i) concentration. There is no evidence at present to indicate that increased Al_i concentration in response to episodic acidification has harmed fish or other aquatic biota in the ROMO region, but such effects are possible.

It is unlikely that aquatic biota in GLAC have experienced adverse impacts from acidic deposition. This is because N and S deposition are low, the available lake water chemistry data do not suggest chronic acidification, and most sampled lakes appear to be relatively insensitive to both chronic and episodic acidification at any foreseeable future levels of acidic deposition.

Nutrient Nitrogen Enrichment

The dominant sources of N to watersheds vary across the continental United States (USGS 1999). In most areas, animal manure and fertilizer application account for the vast majority of N sources to large watersheds. In a few of the USGS sampling sites that are included within the National Water Quality Assessment (NAWQA) Program, however, atmospheric sources account for an estimated one-fourth or more of nonpoint N inputs. These include the South Platte River in Colorado, within the ROMN region (USGS 1999).

The network rankings for nutrient N Pollutant Exposure, Ecosystem Sensitivity to nutrient N enrichment, and Park Protection developed in a coarse assessment by Sullivan et al (2011a) yielded an overall network nutrient N enrichment Summary Risk ranking for the ROMN that was in the highest quintile among all networks. This was despite the relatively low levels of N emissions and

Table 6. Estimated park rankings¹ in ROMN according to risk of nutrient enrichment impacts on sensitive receptors. (Source: Sullivan et al. 2011a)

Park Name ²	Park Code	Estimated Nutrient N Pollutant Exposure	Estimated Ecosystem Sensitivity to Nutrient N Enrichment
Florissant Fossil Beds	FLFO	Moderate	Moderate
Glacier	GLAC	Low	High
Grant-Kohrs Ranch	GRKO	Very Low	High
Great Sand Dunes	GRSA	Very Low	Very High
Little Bighorn Battlefield	LIBI	Very Low	Very High
Rocky Mountain	ROMO	Moderate	Very High

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

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

deposition in this network. The overall level of concern for nutrient N enrichment effects on I&M parks within this network was judged by Sullivan et al. (2011a) to be High. Although rankings provide an indication of risk, park-specific data, particularly regarding nutrient enrichment sensitivity, are needed to fully evaluate risk from nutrient N addition.

All parks in this network were ranked in the lowest quintile (including GRSA) to the middle quintile (including ROMO) for nutrient N Pollutant Exposure (Table 6). In contrast, all parks in the network except FLFO were ranked either in the highest quintile (including GRSA and ROMO) or second highest quintile (including GLAC) for Ecosystem Sensitivity to nutrient N enrichment (Table 6). The predominant vegetation types thought to be highly sensitive to nutrient N enrichment that occur within the parks in the ROMN are alpine, wetland, and grassland and meadow; arid and semi-arid vegetation types are also fairly common. Alpine lands are widely distributed throughout GLAC and ROMO. Wetlands, and to a lesser extent grassland and meadow, are common in GLAC. Grassland

and meadow and arid and semi-arid lands are both common in ROMO. There are many alpine areas and high-elevation lakes within GLAC, ROMO and GRSA which might be more prone than ecosystem receptors at lower elevation to N-limitation and therefore potentially more susceptible to eutrophication in response to atmospheric N input.

Aquatic Nitrogen Enrichment

Some freshwater aquatic ecosystems in the United States are sensitive to nutrient enrichment effects from atmospheric N deposition. In order to be sensitive to such effects, the lake or stream must be N-limited. Conventional wisdom previously held that most lakes and streams in the United States are phosphorous (P)-, rather than N-limited. Relatively new research suggests that this may not always be the case. Surveys of the literature of fertilization experiments and lake studies found that oligotrophic waters are commonly N-limited, especially undisturbed northern temperate or boreal lakes that receive low levels of atmospheric N deposition (Bergström et al. 2005, Elser et al. 2009a, Elser et al. 2009b). There is increasing evidence from surveys and paleolimnological research

Table 7. Proportion of lakes in three nutrient limitation classes (N-limited, P limited, and N and P co-limited) based on ratios of dissolved inorganic nitrogen (DIN; $\text{NO}_3^- + \text{NH}_4^+$) to total phosphorus (TP). Data are from the Western Lake Survey (Landers et al. 1987), conducted in the fall of 1985. (Source: Baron et al. 2011a)

Region	Number of Lakes ¹	Number of N-Limited Lakes (%) ²	Number of P- Limited Lakes (%) ³	Number of Lakes with Co-limited N:P (%) ⁴
Rockies	6,666	2,998 (45%)	2,259 (34%)	1,409 (21%)

¹ The Western Lake Survey was a stratified random sample of lakes; estimates of the number of lakes in each region are based on the target population size for the survey.

² Lakes with DIN:TP ratios (by weight) less than four were characterized as N-limited, based on the work of Morris and Lewis (1988).

³ Lakes with DIN:TP ratios (by weight) greater than 12 were characterized as P limited, based on the work of Morris and Lewis (1988).

⁴ Lakes with DIN:TP ratios (by weight) between 4 and 12 could not be assigned to nutrient limitation class, and are characterized as either co-limited or limited by something other than N or P.

suggesting N limitation is common, or was common, in many lakes prior to human settlement. Bergström and Jansson (2005) found a consistent pattern showing N limitation for watersheds that receive N deposition below approximately 2.5 kg N/ha/yr, co-limitation of N and P for deposition between ~2.5 and 5.0 kg N/ha/yr, and P limitation in areas with N deposition greater than 5.0 kg N/ha/yr. An examination of WLS data (Eilers et al. 1987) found enhanced N concentrations in high elevation lakes adjacent to and downwind of urban centers (Fenn et al. 2003a). Therefore, eutrophication effects on freshwater ecosystems from atmospheric deposition of N are of greatest concern in lakes and streams that have very low productivity and nutrient levels and that are located in remote areas. Baron et al. (2011a) estimated that 45% of 6,666 lakes represented in the Rocky Mountain region of the WLS were likely N-limited, based on having dissolved inorganic N (DIN):total P (TP) ratio less than 4 (Table 7).

Although all of the large parks in the ROMN have characteristics that make them sensitive to nutrient N enrichment, research on ecosystem responses to N fertilization within the ROMN has mainly been conducted in the Front Range of Colorado, including in ROMO. Lakes and streams in ROMO, in particular, tend to be clear-water, low ionic strength, oligotrophic aquatic 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.

Lakes in the Colorado Front Range tend to have higher NO_3^- concentrations than do lakes elsewhere in Colorado (Baron et al. 2000, Elser et al. 2009b, Musselman et al. 2004). In addition to the presence of N sources along the Front Range in Colorado, it has been hypothesized that N retention in soils of the Front Range may be constrained by soil freezing, as compared with the Sierra Nevada, for example (Sickman et al. 2002).

Nitrate concentrations in lake water in ROMO are higher on the east side of the park (Baron et al. 2000). Data from a survey of 44 lakes east and west of the Continental Divide in Colorado indicated that lakes on the western side of the Continental Divide averaged 6.6 $\mu\text{eq/L}$ of NO_3^- , whereas lakes on the eastern side of the Continental Divide averaged 10.5 $\mu\text{eq/L}$ of NO_3^- concentration. Nitrogen deposition appears to have stimulated productivity and altered algal species assemblages at low deposition rates of 1.5 to 2.2 kg N/ha/yr (Baron 2006). In the Colorado Front Range, NO_3^- concentrations in lakes above 15 $\mu\text{eq/L}$ have commonly been measured, suggesting some degree of N-saturation (Baron 1992).

In situ N addition field experiments showed that some low-nutrient lakes in ROMO are N-limited (cf., Lafrancois et al. 2004). Collectively, these studies suggest that atmospheric N deposition is changing the structure and function of sensitive high-elevation aquatic and terrestrial ecosystems on the east side of the park (Porter and Johnson 2007).

Additions of N can stimulate algal growth in N-limited lakes. Studies have shown an increase in lake phytoplankton biomass with increasing N deposition in the Snowy Range in Wyoming (Lafrancois et al. 2003), the Sierra Nevada Mountains in California (Sickman et al. 2003), Sweden (Bergström et al. 2005), and across Europe (Bergström and Jansson 2006). However, not all species of diatoms or other algae are equally responsive to N supply. Differences in resource requirements allow some species to gain a competitive edge over others upon nutrient addition, and as a consequence, shifts in algal assemblages have been observed (Lafrancois et al. 2004, Saros et al. 2005, Wolfe et al. 2001, Wolfe et al. 2003). Interlandi and Kilham (2001) demonstrated that species diversity declines with increasing availability of N; maximum species diversity was maintained when N levels were extremely low ($< 3 \mu\text{M NO}_3^-$) in lakes in the Yellowstone National Park (Wyoming, Montana) region. Sediment cores from lakes in the Colorado Front Range showed increasing representation of mesotrophic diatom taxa in recent times, as compared with pre-development conditions (Wolfe et al. 2001). Community shifts in phytoplankton were observed in the Snowy Range, with chrysophytes favored in lakes having lower N and cyanophytes and chlorophytes favored in lakes having higher N (Lafrancois et al. 2003). These results corroborate earlier work on resource requirements for these algal species (Tilman 1981).

Chlorophytes generally prefer high N concentrations and are able to rapidly dominate the flora when N concentrations increase (Findlay et al. 1999). This occurs in both circumneutral and acidified waters (Findlay et al. 1999, Wilcox and Decosta 1982). Two species of diatom, *Asterionella formosa* and *Fragilaria crotonensis*, now dominate the flora of some alpine and montane Rocky Mountain lakes (Baron et al. 2000, Interlandi and Kilham 1998, Saros et al. 2003, Saros et al. 2005, Wolfe et al. 2001, Wolfe et al. 2003). *A. formosa* and *F. crotonensis* have extremely low resource requirements for P and moderate requirements for N, allowing for rapid response to increased N availability (Michel et al. 2006). They were among the first diatoms to increase in abundance following watershed settlement and agricultural development in European lake watersheds in the 12th and 13th centuries (Anderson et al. 1995, Lotter 1998), and North American settlements in the 18th and 19th centuries (Christie and Smol 1993, Hall et al. 1999). In these studies, as well as in a Swedish lake influenced by acidic deposition, these two diatom species expanded following initial disturbance, and were later replaced by other species more tolerant of either acidification or eutrophication (Hall et al. 1999, Renberg et al. 1993). Moreover, the growth of *A. formosa* has been stimulated with N amendments during *in situ* incubations, using bioassays and mesocosms (76 µM N/L, Lafrancois et al. 2004, 6.4 to 1616 µM N/L, McKnight et al. 1990, 18 µM N/L, Saros et al. 2005). Mesocosm enrichments in Wyoming lakes found positive responses of *A. formosa* and *F. crotonensis* to N, but not to P or Si enrichment (Saros et al. 2005). Furthermore, studies of diatom remains in lake sediments (Wolfe et al. 2003) have shown declines in the oligotrophic diatom species *Aulacoseria perglabra*, *Cyclotella stelligera*, and *Achnanthes* spp. coincident with increases in abundance of *A. formosa* and *F. crotonensis*.

Increased N deposition can cause changes in the species composition of algal communities in sensitive oligotrophic lakes. For example, Baron (2006) found that diatom reconstructions from lake sediment cores in ROMO during the period 1850 to 1964 suggested changes in algal abundance associated with wet N deposition of only about 1.5 kg N/ha/yr. Similar results were found by Saros et al. (2003) in the Beartooth Mountains in Wyoming. The freshwater algae thought to be most sensitive to effects of increased N deposition included *A. formosa* and *F. crotonensis*. These species have been shown to increase in abundance following N inputs and in some cases dominate the flora of affected alpine and montane lakes (cf., Saros et al. 2005, Wolfe et al. 2003). These are considered to be opportunistic algae that can respond rapidly to slight nutrient enrichment.

Lake sediment records, including diatom stratigraphies, suggest that changes attributable to atmospheric N inputs began in Rocky Mountain lakes during the period 1950 to 1960 (Das et al. 2005, Enders et al. 2008, Wolfe et al. 2001, 2003). Changes in diatom abundance included decreases in oligotrophic species such as *Aulacoseria perglabra*, *Cyclotella stelligera*, and *Achnanthes* spp. and increases in the more mesotrophic species *Asterionella formosa* and *Fragilaria crotonensis* (Wolfe et al. 2001, 2003).

Documented and potential future impacts from atmospheric N deposition on sensitive aquatic and terrestrial resources in ROMO led to the formation of the Rocky Mountain National Park Initiative, a collaborative process involving NPS, EPA, and the Colorado Department of Public Health and Environment. Through this initiative process, a critical load target of N deposition has been

developed in order to protect aquatic resources in the park against eutrophication (Porter and Johnson 2007). The NPS established a wet N deposition critical load of 1.5 kg N/ha/yr for protecting high-elevation lakes in ROMO against biological effects associated with nutrient enrichment. NPS also entered into a MOU with the U.S. EPA and Colorado Department of Public Health and Environment to address harmful impacts of N deposition in this park (Cheatham 2011). The MOU was intended to facilitate interagency cooperation in reversing N CL exceedance in ROMO. The Nitrogen Deposition Reduction Plan was endorsed by the three participating agencies and the Colorado Air Quality Control Commission (<http://www.colorado.gov/cdphe/rmnpinitiative>). The NPS adopted and the MUO agencies endorsed a CL of wet N deposition of 1.5 kg N/ha/yr at the Loch Vale deposition monitoring site. In order to achieve this resource management goal, a glidepath approach was selected, as described by Morris et al. (2014). The baseline wet N deposition for the period 2002-2006 was 3.1 kg N/ha/yr. Although some progress has been made, the target five-year rolling average value for 2012 of 2.7 kg N/ha/yr was not met; the five-year rolling average in 2012 was 2.9 kg N/ha/yr (Morris et al. 2014).

Paleolimnological studies of mountain lakes that have experienced little disturbance other than by atmospheric deposition and climate change have reported changes in diatom species assemblages, increases in cell numbers, and pigment-inferred increases in whole lake primary production. These inferred changes have been coincident with regional surrogates for increased N deposition. Such changes have included increases in human population, industrial animal production, and fossil fuel combustion emissions (Das et al. 2005, Saros et al. 2003, Wolfe et al. 2001, Wolfe et al. 2003). In most, but not all, of these studies, the observed changes in ecology were inconsistent with changes in climate and more concordant with effects from increased atmospheric N deposition.

Findings regarding nutrient enrichment of high-elevation lakes in ROMO are not necessarily transferable to other locations in the ROMN. For example, soluble reactive P was consistently near the detection limit in all of the lakes studied by Ellis et al. (1992) in GLAC. The authors concluded that these lakes were oligotrophic to ultra-oligotrophic and were P-limited. Such lakes might be co-limited by N and P.

Nanus et al. (2012) calculated spatially explicit estimates of nutrient N deposition for the Rocky Mountains using a geostatistical approach. To do this, they established the response of sensitive diatoms to variation in surface water NO_3^- concentration and identified a threshold NO_3^- concentration above which ecological effects were observed. Response of the diatom *A. formosa* to changes in NO_3^- concentration during nutrient addition experiments was determined. Nanus et al. (2012) found that surface water NO_3^- concentration was positively correlated with north-facing aspect, elevation, slope, and N deposition. Growth experiments using *A. formosa* showed maximum algal growth at 0.5 μM NO_3^- , and this level was specified as the threshold for algal growth, which was used to determine nutrient critical loads. The lowest CL levels (< 1.5 kg N/ha/yr) occurred at high elevation locations having steep slope, sparse vegetation, abundant exposed rock, and talus. Such areas were commonly in exceedance of the CL by more than 1.5 kg N/ha/yr. Atmospheric N deposition exceeded the CL in $21 \pm 8\%$ of the Rocky Mountain study area; this estimate was sensitive to selection of the NO_3^- threshold of ecological effects (Nanus et al. 2012).

The critical load for lake acidification in the Green Lakes Valley in Colorado was estimated empirically to be about 4 kg N/ha/yr (Williams and Tonnessen 2000). Baron et al. (2011b) developed an empirical CL of 4.0 kg N/ha/yr for western mountain lakes to protect against episodic N pulses in lakes having low ANC. The authors were not able to determine the empirical CL for N-driven acidification.

Changes to aquatic food webs have not been as thoroughly explored as changes to algal assemblages. However, a few studies have suggested declines in zooplankton biomass (Lafrancois et al. 2004, Paul et al. 1995) in response to N-related shifts in phytoplankton biomass toward less palatable taxa with higher C:P ratios (Elser et al. 2001). Thus, nutrient N input can potentially disrupt food webs in ways that scientists are only beginning to understand.

Terrestrial Nitrogen Enrichment

Alpine vegetation in the southern Rocky Mountains responds to increased N supply by increasing plant productivity for some species, and this increase in productivity is accompanied by changes in species composition and abundance (Bowman et al. 1993, Bowman et al. 2012). Many of the dominant plant species do not respond to additional N supply with increased production. Rather, many subdominant species, primarily grasses and some forbs, increase in abundance when the N supply is increased (Fenn et al. 2003a). Factors that govern the sensitivity of alpine tundra to N deposition include low rates of primary production, short growing season, low temperature, and wide variation in moisture availability in the alpine environment (Bowman and Fisk 2001). Alpine plant communities have also developed under conditions of low nutrient supply, in part because soil-forming processes are poorly developed, and this also contributes to their N-sensitivity.

Baron et al. (1994) estimated N uptake by plants and soil in Colorado using the CENTURY model. Simulated N export increased in alpine ecosystems at relatively low levels of N deposition (3.4 kg N/ha/yr). The adjacent subalpine forests had more substantial forest N retention.

Nitrogen cycling in Rocky Mountain alpine environments is strongly tied to variations in moisture regime (Bowman et al. 1993, Bowman 1994, Fisk et al. 1998). Blowing snow is transported across alpine landscapes by wind and tends to accumulate in certain depression areas. These areas receive much higher levels of moisture and winter season N deposition than other more wind-swept portions of the alpine environment (Bowman 1992). Fenn et al. (2003a) suggested that as much as 10 kg N/ha/yr may leach through the snow during the initial phases of snowmelt in some of the alpine areas in Colorado that accumulate substantial snowpack. It is these moist meadow areas that may be most affected by N deposition, and they are also the areas most likely to show changes in plant species composition and impacts on N cycling (Bowman and Steltzer 1998).

Baron et al. (2000) showed that small differences in N deposition between the east (3 to 5 kg N/ha/yr) and the west (1 to 2 kg N/ha/yr) side of the Continental Divide in Colorado were associated with substantial declines in foliar Mg^{2+} levels and increased foliar N:Mg²⁺ and N:Ca²⁺ ratios in old-growth stands of Engelmann spruce (*Picea engelmannii*). It is not known whether such differences in foliar nutrient ratios affected the health or growth of these forests. Nevertheless, analyses by Baron et

al. (2000) suggested that the eastern slope of the Colorado Front Range may be at the beginning of a trajectory of N saturation change.

Most major ecosystem types, including temperate forest, grassland, and tropical forest, tend to be dominated by a single physiognomic type of vegetation. In contrast, arctic and alpine tundra tend to be dominated by multiple types of plant communities (Chapin et al. 1980). Across a relatively small area of tundra, there may be a wide variety of plant community types in which graminoids, forbs, mosses, lichens, deciduous shrubs, or evergreen shrubs dominate (Bliss et al. 1973). There can be important differences among these plant growth forms in their use of, and response to, addition of nutrients (Schlesinger and Chabot 1977, Thomas and Grigal 1976).

Alpine ecosystems are adapted to cold temperature, short growing season, high soil moisture, and periodically low soil O₂ level. Plants respond to reduced N availability by changing the allocation of biomass to favor root growth (Bloom et al. 1985) or changing the efficiency with which N is used or stored (Chapin 1980). Increased abundance of nitrophilous (nitrogen-loving) plant species has been demonstrated in alpine plant communities at Niwot Ridge, about 10 km south of ROMO (Korb and Ranker 2001). The Niwot Ridge experimental site receives slightly higher atmospheric N deposition than does Loch Vale (Burns 2003). Results of fertilization experiments suggested that the lowest amount of atmospheric N deposition expected to alter alpine plant communities at Niwot Ridge and Loch Vale is about 4 kg N/ha/yr (Bowman et al. 2006, 2012). This deposition level is similar to ambient deposition measured at Loch Vale in ROMO.

Changes in alpine plant species composition precede detectable changes in soil chemistry in response to increased N deposition (Sverdrup et al. 2012). Changes in species composition of dry meadows (among the most sensitive alpine plant community types) are probably ongoing along the Front Range (Bowman et al. 2006, 2012), in response to ambient N deposition, which varies from about 3 to 6 kg N/ha/yr. Results of a modeling study by Baron et al. (1994) suggested that subalpine forest soils in the Front Range would exhibit increased N leaching at N deposition above about 4 kg N/ha/yr.

Monitoring of alpine vegetation is conducted through the Global Observation Research Initiative in Alpine Environments (GLORIA) network. It involves monitoring of alpine sites worldwide, including at GLAC, GRSA, and ROMO within ROMN. The standard GLORIA protocol includes monitoring on a five-year interval at four locations in each study region. Monitoring data collected in this project may be useful in the future for evaluating potential changes in alpine vegetation communities in ROMN in response to changes in climate and atmospheric N deposition.

The potential impacts of nutrient N deposition on terrestrial resources within ROMO is an important concern. The major issues appear to be 1) "terrestrial eutrophication", whereby excess fertilization leads to increased ecosystem productivity, increased spread of exotic plant species, and decreased native plant species diversity (cf., Huston 1994); and 2) N saturation, whereby N supply exceeds the vegetative uptake capacity and NO₃⁻ leaches out of the soil in high concentrations. McMurray et al. (2013) found that N deposition in throughfall at a level of 4.1 kg N/ha/yr coincided with clear damage to lichen thalli in the Wind River Range of Wyoming.

Rueth and Baron (2002) compared N dynamics of Engelmann spruce (*Picea engelmannii*) forest stands east and west of the Continental Divide in Colorado. Nitrogen deposition, arising mainly from agricultural and urban areas of the South Platte River Basin, was moderate (3 to 5 kg N/ha/yr) on the east slope, but only 1 to 2 kg N/ha/yr on the west slope. East slope sites showed lower soil organic horizon C:N, lower foliar C:N, higher potential net mineralization, and higher percent N, N:Mg, and N:P ratios in foliage. These results suggested that even moderate levels of N deposition input can cause measurable changes in spruce forest biogeochemistry. It is unclear, however, to what extent such biogeochemical changes affect forest growth or health.

Research on experimental N enrichment effects on alpine and subalpine ecosystems has been conducted at the Loch Vale Watershed in ROMO and the Niwot Ridge Long Term Ecological Research (LTER) site, both located east of the Continental Divide in Colorado (see review by Burns 2004). Biomass production responses of alpine communities to increased N deposition are dependent on moisture regimes (Fisk et al. 1998) and are driven by shifts in species composition. Addition of 25 kg N/ha during summer caused a community shift towards greater dominance of hairgrass (*Deschampsia* sp.) in wet alpine meadows, but the increase in plant biomass (+67%) and plant N content (+107%) following N fertilization was higher in graminoid-dominated dry meadows than in forb-dominated wet meadows (+53% plant biomass, +64% standing N crop; Bowman et al. 1995, Burns 2004).

In a study at Niwot Ridge, additions of 20, 40, and 60 kg N/ha/yr (on top of ambient N deposition near 5 kg N/ha/yr) over an eight-year period to a dry alpine meadow led to a change in plant species composition, an increase in species diversity and plant biomass, and an increase in tissue N concentration at all treatment levels within 3 years of application. Much of the response was due to increased cover and total biomass of sedges (*Carex* spp.). There was a significant decrease in *Kobresia* sp. biomass with increasing N input. Vegetation composition appeared to respond at lower N input levels than those that caused measurable changes in soil inorganic N content.

Effects of N deposition to terrestrial alpine ecosystems in the ROMN are thought to include community-level changes in plants, lichens, and mycorrhizae. Subtle effects have been shown to occur at what would be considered relatively low levels of N deposition in the eastern United States (about 4 kg N/ha/year; Bowman et al. 2006). Bowman et al. (2006, 2012) concluded that alpine plants may be more sensitive indicators of the effects of increased N inputs than soils. Changes in plant species composition occurred at all treatment levels within three years. Changes in an individual species (*Carex rupestris*) were estimated to occur at deposition levels near 4 kg N/ha/yr. Changes in the plant community, based on the first axis of a detrended correspondence analysis, were estimated to occur at deposition levels near 10 kg N/ha/yr. In contrast, increases in NO₃⁻ leaching, soil solution NO₃⁻ concentration, and net nitrification occurred at levels above 20 kg N/ha/yr. The authors concluded that changes in vegetation composition preceded detectable changes in soil indicators of ecosystem response to N deposition.

Nitrogen deposition to the alpine tundra of Niwot Ridge altered N cycling and provided the potential for replacement of some plant species by more competitive, faster-growing species (Baron et al. 2000, Bowman and Steltzer 1998, Bowman 2000). Many plants that grow in alpine tundra, as is true

of plants growing in other low resource environments, tend to have some similar characteristics, including slow growth rate, low photosynthetic rate, low capacity for nutrient uptake, and low soil microbial activity (Bowman and Steltzer 1998, Bowman 2000). Such plants generally continue to grow slowly when provided with an optimal supply and balance of resources (Chapin 1991, Pearcy et al. 1987). In addition, plants adapted to cold, moist environments grow more leaves than roots as the relative availability of N increases. These patterns of vegetative development and their response to added N affect plant capacity to respond to variation in available resources and to environmental stresses such as frost, high winds, and drought.

Changes in alpine plant species composition on Niwot Ridge have included increased cover of the plant species that tend to be most responsive to N fertilization in some of the long-term monitoring plots (Fenn et al. 2003a, Korb and Ranker 2001). These changes have probably developed in response to changes in N deposition. However, the influences of climatic change, particularly changes in precipitation (Williams et al. 1996a), and pocket gopher disturbance (Sherrod and Seastedt 2001) could not be ruled out as contributors to vegetation change (Fenn et al. 2003a). Other environmental factors also affect the species make-up of alpine ecosystems, but long-term experimental fertilization plots demonstrate a clear response of alpine flora to N, including shifts toward graminoid plants that shade smaller flowering species, and accompanying changes in soil N cycling (Bowman et al. 2006).

Changes in plant species in response to N deposition to the alpine zone can result in increased leaching of NO_3^- from the soils because the plant species favored by higher N supply are often associated with greater rates of N mineralization and nitrification than the pre-existing species (Bowman et al. 1993, Bowman et al. 2006, Steltzer and Bowman 1998, Suding et al. 2006). Total organic N pools in the soils of dry alpine meadows are large compared to pools of NH_4^+ and NO_3^- (Fisk and Schmidt 1996). However, positive response to inorganic N fertilization has been demonstrated, and thus some plant species appear to be restricted in their ability to take up organic N from the soil and are growth-limited by the availability of inorganic N (Bowman et al. 1993, 1995, Theodose and Bowman 1997).

Bowman et al. (2012) added NH_4NO_3 at rates of 5, 10, and 30 kg N/ha/yr to an alpine dry meadow plant community in ROMO. Three years after fertilization, they measured aboveground biomass, plant N concentration, and soil and soil solution chemistry. Plant species composition was measured annually. Plant species richness and diversity did not change in response to the N addition, but one species of sedge, *Carex rupestris*, increased in cover from 34% to 125%. Based on the rate of change in *C. rupestris* in the N treatment plots and the ambient control plots (receiving 4 kg N/ha/yr), and assuming that the change in sedge cover was attributable solely to N addition, Bowman et al. (2012) estimated the N critical load of nutrient N loading to protect plant community composition at 3 kg N/ha/yr. Inorganic N in soil solution increased above ambient levels at input rates (experimental addition plus ambient loading) between 9 (resin bag measurements) and 14 (lysimeter measurements) kg N/ha/yr, suggesting N saturation at these levels of N loading. There was no indication of change after three years in soil pH or extractable base cations, indicating that soil acidification had not occurred.

McDonnell et al. (2014) evaluated potential long-term impacts of N deposition and climate change on a subalpine plant community at Loch Vale, using the ForSAFE-Veg model. ForSAFE-Veg is a coupled biogeochemical, vegetation niche and plant competition model. The model had earlier been applied in a generalized fashion to the Rocky Mountain region (Porter et al. 2012, Sverdrup et al. 2012). Simulated changes in N deposition, temperature, and precipitation over the previous century caused pronounced changes in model projections of plant species cover. The model estimate of the critical load of N deposition required to protect against a change in plant cover of 10% was between 1.9 and 3.5 kg N/ha/yr. Ambient N deposition is slightly higher than that, suggesting that the CL for protection against nutrient N enrichment has been exceeded at Loch Vale.

Pardo et al. (2011b) compiled data on empirical CL for protecting sensitive resources in Level I ecoregions across the conterminous United States against nutrient enrichment effects caused by atmospheric N deposition. Data compiled by Pardo et al. (2011b) suggest that ambient N deposition may exceed the lower limit of the expected CL to protect against nutrient enrichment effects in some of the parks in ROMN, mainly GLAC and ROMO. These potential exceedances were reported for the protection of mycorrhizal fungi (GLAC only), lichens, herbaceous plants, and forest vegetation and to limit NO₃- leaching in drainage waters (Table ROMN 8).

Ellis et al. (2013) estimated the CL for nutrient-N deposition to protect the most sensitive ecosystem receptors in 45 national parks. The lowest terrestrial CL of N is generally estimated for protection of lichens (Geiser et al. 2010). Changes to lichen communities may signal the beginning of other changes to the ecosystem that might affect structure and function (Pardo et al. 2011a). Ellis et al. (2013) estimated the N CL for GLAC, GRSA, and ROMO in the range of 2.5-7.1 kg N/ha/yr for protection of lichens.

Table 8. Empirical critical loads for nitrogen in ROMN, by ecoregion and receptor from Pardo et al. (2011b). Ambient N deposition reported by Pardo et al. (2011b) is compared to the lowest critical load for a receptor to identify potential exceedance, indicated by graying. A critical load exceedance suggests that the receptor is at increased risk for harmful effects.

NPS Unit	Ecoregion	N Deposition (kg N/ha/yr)	Critical Load (kg N/ha/yr)				
			Mycorrhizal Fungi	Lichen	Herbaceous Plant	Forest	Nitrate Leaching
Florissant Fossil Beds NM	Northwestern Forested Mountains	3.4	5 - 10	2.5 - 7.1	4 - 10	4 - 17	4 - 17
Glacier NP	Northwestern Forested Mountains	5.7	5 - 10	2.5 - 7.1	4 - 10	4 - 17	4 - 17
Grant-Kohrs Ranch NHS	Northwestern Forested Mountains	1.9	5 - 10	2.5 - 7.1	4 - 10	4 - 17	4 - 17
Great Sand Dunes NP & Pres	North American Deserts	2.2	NA	3	3 - 8.4	NA	NA
Great Sand Dunes NP & Pres	Northwestern Forested Mountains	2.2	5 - 10	2.5 - 7.1	4 - 10	4 - 17	4 - 17
Little Bighorn NM	Great Plains	2.4	12 - 12	NA	5 - 25	NA	10 - 25
Rocky Mountain NP	Northwestern Forested Mountains	4.6	5 - 10	2.5 - 7.1	4 - 10	4 - 17	4 - 17

Ozone Injury to Vegetation

The O₃-sensitive plant species that are known or thought to occur within the I&M parks in the ROMN are listed in Table 9. Those considered to be bioindicators because they exhibit distinctive symptoms when injured by O₃ (e.g., dark stipple), are designated by an asterisk. Each park in this network contained at least three O₃-sensitive and/or bioindicator species. GLAC contained eight sensitive species, four of which are recognized as bioindicators. GRSA contained six sensitive species, four of which are recognized as bioindicators. ROMO contained seven sensitive species, six of which are recognized as bioindicators.

Table 9. Ozone-sensitive and bioindicator plant species known or thought to occur in the I&M parks of the ROMN. (Data Source: E. Porter, National Park Service, pers. comm., August 30, 2012); lists are periodically updated at <https://irma.nps.gov/NPSpecies/Report>.

Species	Common Name	Park ¹					
		FLFO	GLAC	GRKO	GRSA	LIBI	ROMO
<i>Amelanchier alnifolia</i>	Saskatoon serviceberry		x	x	x		x
<i>Apocynum androsaemifolium</i> spp. <i>Androsaemifolium</i> *	Spreading dogbane		x				
<i>Apocynum androsaemifolium</i> *	Spreading dogbane	x			x	x	x
<i>Apocynum cannabinum</i>	Dogbane, Indian hemp		x				
<i>Artemisia ludoviciana</i> *	Silver wormwood	x		x		x	x
<i>Asclepias syriaca</i> *	Common milkweed					x	
<i>Fraxinus pennsylvanica</i>	Green ash			x		x	
<i>Physocarpus malvaceus</i> *	Pacific ninebark		x				
<i>Pinus ponderosa</i> var. <i>scopulorum</i> *	Ponderosa pine				x		x
<i>Populus tremuloides</i> *	Quaking aspen	x	x	x	x		x
<i>Prunus virginiana</i>	Choke cherry			x		x	
<i>Prunus virginiana</i> var. <i>melanocarpa</i>	Choke cherry				x		x
<i>Rhus trilobata</i> *	Skunkbush				x	x	x
<i>Robinia pseudoacacia</i>	Black locust					x	
<i>Rubus parviflorus</i>	Thimbleberry		x				
<i>Rudbeckia laciniata</i> ²	Cutleaf coneflower						x
<i>Salix scouleriana</i> *	Scouler's willow		x				x
<i>Vaccinium membranaceum</i> *	Huckleberry		x				

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

² Kohut et al. (2012)

* Bioindicator species

Ozone Exposure Indices and Levels

The W126 (a measure of cumulative O₃ exposure that preferentially weights higher concentrations) and SUM06 (a measure of cumulative exposure that includes only hourly concentrations over 60 parts per billion [ppb] O₃) exposure indices calculated by NPS staff are given in Table 10, along with Kohut's (2007) O₃ risk ranking. The NPS and Kohut ranking systems differ. The NPS ranking

system (NPS 2010) is a quick assessment of O₃ condition that ranks O₃ exposure levels according to injury thresholds from the literature (Heck and Cowling 1997), using a 5-year average of either the W126 or SUM06 index. Both metrics are calculated over a 3-month period. The W126 was classified as Moderate exposure at values between 7 and 13 ppm-hr, as defined by NPS (2010). Values higher than 13 ppm-hr were classified as High exposure, and values lower than 7 ppm-hr were classified as Low exposure. The SUM06 was classified as Moderate at values between 8 and 15 ppm-hr. Higher and lower values were classified as High and Low, respectively, as defined by NPS (2010). Using these criteria, O₃ levels at the ROMN parks are rated Low to High (Table 10).

Table 10. Ozone assessment results for I&M parks in the ROMN based on estimated average 3-month W126 and SUM06 ozone exposure indices for the period 2005-2009 and Kohut's (2007) ozone risk ranking for the period 1995-1999¹.

Park Name ²	Park Code	W126		Sum06		Kohut O ₃ Risk Ranking
		Value (ppm-hr)	Ranking	Value (ppm-hr)	Ranking	
Florissant Fossil Beds	FLFO	17.49	High	24.73	High	Low
<i>Glacier</i>	GLAC	2.30	Low	1.67	Low	Low
Grant-Kohrs Ranch	GRKO	6.35	Low	6.71	Low	Low
<i>Great Sand Dunes</i>	GRSA	15.49	High	21.34	High	Low
Little Bighorn Battlefield	LIBI	7.18	Moderate	7.75	Low	Low
<i>Rocky Mountain</i>	ROMO	17.35	High	23.75	High	Low

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

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

Kohut's approach constitutes a more rigorous assessment of potential risk to plants in that it considers both O₃ exposure and environmental conditions (soil moisture). Kohut also used injury thresholds from the literature, but evaluated a different O₃ metric (after Lefohn et al. 1997), the W126 over a 5-month period in conjunction with the N100 (number of hours over 100 ppb O₃). The rationale for the N100 statistic is that higher O₃ concentrations are most likely to cause plant injury. Kohut examined five individual years of O₃ exposure and soil moisture data and considered the effects of low soil moisture on O₃ uptake each year when assigning risk. Soil moisture is important because dry conditions induce stomatal closure in plants, which has the effect of limiting O₃ uptake and injury. In areas where low soil moisture levels correspond with high O₃ exposure, uptake and injury are limited by stomatal closure even when exposures are relatively high. Kohut's (2007) ranking was Low across all parks in this network.

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

Ozone Formation

Among the parks in the ROMN, O₃ appears to be of greatest concern in ROMO, GRSA, and FLFO, given the relatively high values of W126 and SUM06 in these parks. Values in GLAC, GRKO, and LIBI are much lower (Table 10). The ROMO area is designated nonattainment for O₃. One of the greatest threats to vegetation in ROMO is O₃ pollution from urban areas southeast of the park and from valley and foothill areas where O₃ is synthesized in transit from local sources of NO_x and VOCs.

NPS (2010) reported long-term trends in annual fourth highest 8-hour daily maximum O₃ concentration for 31 monitoring sites in 27 national parks having more than 10 years of data through 2008. Statistically significant increases were reported for only four parks, including ROMO. This park is the recipient of relatively O₃-rich air masses originating from the Denver-to-Fort Collins area, especially during the summer (Peterson et al. 1998).

Exposure of plants to O₃ is relatively high in ROMO and two other parks in the ROMN (FLFO and GRSA) based on both the W126 and SUM06 statistics (Table 10). Sources of NO_x that contribute to these high exposures include motor vehicles, power plants, and other human sources. At ROMO, the largest number of exceedances of the 8-hr 0.075 ppm O₃ standard occurred in 2002 and 2003 (six and seven days in exceedance, respectively). These were also two of the top three years in terms of greatest summer burned area (BA) by forest fire in the surrounding region. Jaffe et al. (2008) concluded that fire plays an important role in elevating the background O₃ level and increasing the likelihood of an exceedance of the 8-hr O₃ standard.

Recent research by Kohut et al. (2012) discovered foliar O₃ symptoms on cutleaf coneflower (*Rudbeckia laciniata*), but not on spreading dogbane (*Apocynum androsaemifolium*) or quaking aspen (*Populus tremuloides*), in ROMO. During the 1990s, there was concern that O₃ might be affecting ponderosa pine (*Pinus ponderosa* var. *scopulorum*), but further research suggested that the Rocky Mountain variety of this species was not particularly O₃-sensitive.

The study by Kohut et al. (2012) was the first documentation of O₃ symptoms on vegetation in the Rocky Mountain region. Kohut et al. (2012) also reported SUM06 and W126-3 mo levels in ROMO that were higher than the common thresholds for observing foliar symptoms. The increase in O&G drilling in Wyoming in recent years may have caused or contributed to relatively high concentrations of O₃ in some remote areas that previously had low background O₃ levels (Kohut et al. 2012, Wyoming Department of Environmental Quality 2010). As a consequence, Kohut et al. (2012) advocated initiation of a comprehensive program to assess foliar O₃ symptoms on O₃-sensitive plant species in riparian and moist plant communities in the Rocky Mountain region.

Forest fires emit considerable quantities of NO_x and hydrocarbons, and therefore can contribute to O₃ formation in nonurban settings such as throughout the ROMN. The concentration of O₃ in the atmosphere in the nonurban western United States has increased since the late 1980s by about 5 ppb (Jaffe and Ray 2007). Jaffe et al. (2008) investigated the role of forest fire in this trend. The summer BA was significantly correlated with O₃ concentration at CASTNET and NPS atmospheric chemistry monitoring sites within six national parks, including ROMO and GLAC. For mean and maximum

fire years, the concentration of O₃ in the atmosphere appeared to be increased by fire by an average of 3.5 and 8.8 ppb, respectively. The estimated amount of biomass consumed (BC) was a slightly better predictor of O₃ concentration than BA. This relationship between atmospheric O₃ concentration and BA or BC is especially important because the frequency of extreme fire years in the western United States appears to be increasing (Jaffe et al. 2004) due to increased spring and summer temperature, earlier snowmelt, and dryer forest conditions (Cook et al. 2004, Westerling et al. 2006). Jaffe et al. (2008) concluded that the increase in fires in the western United States has largely been responsible for the observed increase in summer O₃ concentrations, above the levels caused by vehicular, power plant, and industrial emissions reported by Jaffe and Ray (2007). Increasing temperature in the future will likely further influence the effects of fire on O₃ formation.

Ozone Exposure Effects

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 evidence of O₃ injury. No symptoms were found (James and Staley 1980). In 1987, the NPS 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 O₃ injury were noted in any trees in that survey. 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 O₃) determined that there were no visible symptoms of O₃ injury at any locations and that long-term growth was unaffected by elevated O₃ levels (Graybill et al. 1993, Peterson et al. 1993). The Rocky Mountain variety of ponderosa pine is known to be somewhat more tolerant to O₃ and has a higher threshold for symptoms of injury under experimental exposures than var. *ponderosa* (Aitken et al. 1984), which is found further to the west.

Quaking aspen, an O₃-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 O₃ under field and experimental conditions (Coleman et al. 1996, Karnosky et al. 1992, Wang et al. 1986) although there is considerable variability in sensitivity among different genotypes (Berrang et al. 1986). Of the tree species present at GLAC, quaking aspen is probably the most sensitive to O₃. Aspen grows at various locations in riparian ecosystems and in fire- or avalanche-disturbed areas in the park.

Visibility Degradation

Natural Background and Ambient Visibility Conditions

Three of the parks in the ROMN are classified as Class I under the Clean Air Act (CAA): GLAC, GRSA, and ROMO. The other ROMN parks are classified as Class II for the purposes of air quality management. The CAA set a specific goal for visibility protection in Class I areas: “the prevention of any future, and the remedying of any existing, impairment of visibility in mandatory Class I federal areas which impairment results from manmade air pollution” (42 U.S.C. 7491). In 1999, EPA passed the Regional Haze Rule (RHR), which requires each state to develop a plan to improve visibility in Class I areas, with the goal of returning visibility to natural conditions in 2064. Natural background visibility, or natural haze, assumes no human-caused pollution, but varies with natural processes such as windblown dust, fire, volcanic activity and biogenic emissions. Visibility is monitored by the Interagency Monitoring of Protected Visual Environments Network (IMPROVE) and typically reported using the haze index deciview¹ (dv). Some of the best visibility in the conterminous United States occurs in the Rocky Mountains (Savig and Morse 1998).

Haze is monitored by IMPROVE for GLAC, GRSA, and ROMO in the ROMN. Data are also available from an IMPROVE sampler in the Lolo National Forest (MONT1) that are considered to be representative of visibility conditions in GRKO. A monitoring site is considered by IMPROVE to be representative of an area if it is within 60 mi (100 km) and 425 ft (130 m) in elevation of that area.

Ambient visibility estimates reflect pollution levels and were used to rank conditions at parks in order to provide park managers with information on spatial differences in visibility and air pollution. Rankings range from very low haze (very good visibility) to very high haze (very poor visibility). Only parks with on-site or representative IMPROVE monitors were used in generating the baseline visibility ranking. Table 11 gives the relative park haze rankings on the 20% clearest, 20% haziest, and average days.

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

Table 11. Estimated natural haze and measured ambient haze in I&M parks averaged over the period 2004 through 2008¹ in national parks in the ROMN. Data from the Lolo National Forest in Montana (MONT1) are considered representative of GRKO.

Park Name ²	Park Code	Site ID	Estimated Natural Haze (dv)					
			20% Clearest Days	20% Haziest Days	Average Days			
Florissant Fossil Beds	FLFO	No Site						
<i>Glacier</i>	GLAC	GLAC1	2.42	9.18	5.08			
Grant-Kohrs Ranch ³	GRKO	MONT1	1.48	7.73	3.98			
<i>Great Sand Dunes</i>	GRSA	GRSA1	1.23	6.66	3.47			
Little Bighorn Battlefield	LIBI	No Site						
<i>Rocky Mountain</i>	ROMO	ROMO1	0.28	7.15	3.28			
Park Name	Park Code	Site ID	Measured Ambient Haze (For Years 2004 through 2008)					
			20% Clearest Days		20% Haziest Days		Average Days	
			dv	Ranking	dv	Ranking	dv	Ranking
Florissant Fossil Beds	FLFO	No Site						
<i>Glacier</i>	GLAC	GLAC1	7.02	Moderate	18.90	Moderate	12.47	Moderate
Grant-Kohrs Ranch ³	GRKO	MONT1	3.28	Low	15.60	Low	8.22	Low
<i>Great Sand Dunes</i>	GRSA	GRSA1	3.65	Low	11.52	Very Low	7.48	Very Low
Little Bighorn Battlefield	LIBI	No Site						
<i>Rocky Mountain</i>	ROMO	ROMO1	2.06	Very Low	12.66	Very Low	7.26	Very Low

¹ Parks are classified into one of five haze ranks (Very Low, Low, Moderate, High, or Very High haze).

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

³ Data are borrowed from a nearby IMPROVE site. A monitoring site is considered by IMPROVE to be representative of an area if it is within 60 mi (100 km) and 425 ft (130 m) in elevation of that area.

Haze measurements for the period 2004 through 2008 were higher than the estimated natural condition for all parks in the ROMN. Measured ambient haze in GLAC was considered Moderate for all groups (20% clearest, average, and 20% haziest days); haze in GRKO was considered Low for all groups; and haze in ROMO was ranked Very Low for all groups. GRSA was ranked Low for the 20% clearest days and Very Low for the 20% haziest days and average days.

Representative photos of selected vistas under three different visibility conditions are shown in Figure 1 for GLAC, GRSA, and ROMO. Photos were selected to correspond with the clearest 20% of visibility conditions, haziest 20% of visibility conditions, and annual average visibility conditions at each location. This series of photos provides a graphic illustration of the visual effect of these differences in haze level on a representative vista in each of these parks.

IMPROVE data allow estimation of visual range (VR). Data indicate that in GLAC, pollution has reduced average VR from 140 to 45 miles (225 to 72 km). On the haziest days, VR has been reduced from 95 to 20 miles (153 to 32 km). Severe haze episodes occasionally reduce visibility to 6 miles (10 km). At GRSA, pollution has reduced average VR from 170 to 100 miles (274 to 161 km). On the haziest days, VR has been reduced from 120 to 65 miles (193 to 105 km). Severe haze episodes

occasionally reduce visibility to 22 miles (35 km). At ROMO, pollution has reduced average VR from 170 to 100 miles (274 to 161 km). On the haziest days, VR has been reduced from 120 to 60 miles (193 to 97 km). Severe haze episodes occasionally reduce visibility to 18 miles (29 km). At the MONT1 site, representative of GRKO, pollution has reduced average VR from 160 to 95 miles (258 to 153 km). On the haziest days, VR has been reduced from 110 to 50 miles (177 to 81 km). Severe haze episodes occasionally reduce visibility to 4 miles (6 km).

Composition of Haze

Various pollutants make up the haze that causes visibility degradation. IMPROVE measures these pollutants and reports them as ammonium sulfate, ammonium nitrate, elemental carbon, coarse mass, organic mass, sea salt, and soil. Sulfates form in the atmosphere from SO₂ emissions from coal-burning power plants, smelters, and other industrial facilities. Nitrates form in the atmosphere from NO_x emissions from combustion sources including vehicles, power plants, industry, and fires. Organic compounds are emitted from a variety of both natural (biogenic) and anthropogenic sources, including agriculture, industry, and fires. Soil can enter the atmosphere through both natural processes and human disturbance.

At IMPROVE sites throughout the interior Columbia River basin, from North Cascades (NOCA) and Glacier (GLAC) national parks in the north to Lassen Volcanic (LAVO) and Yellowstone (YELL) national parks in the south, carbon in various forms, including fine particulate organics and soot, dominates the light extinction budget (Schoettle et al. 1999). The second most important contributor is SO₄²⁻.

Figure 2 shows estimated natural (pre-industrial), baseline (2000-2004), and current (2006-2010) levels of haze and its composition for the monitored parks in the ROMN. The RHR requires at least three years of valid data out of a five-year period to calculate the five-year average for the baseline period (2000-2004). Sites having fewer than three years of valid data required use of the RHR_VS algorithm substitution to calculate the baseline. This process entailed using substituted valid data for incomplete years. GLAC has no valid measured data for the years 2002, 2003, and 2009; RHR2_VS substituted data were used for 2002 and 2003. For more information on this process, see http://vista.cira.colostate.edu/improve/Data/IMPROVE/summary_data.htm.

In GLAC, on the 20% haziest days and average days, organics were the largest contributors to the light extinction coefficient (b_{ext}); on the 20% clearest days the largest contributor was SO₄²⁻. Organics account for about one third (20% clearest days) to one-half (20% haziest days) of the b_{ext} in GLAC. Elemental C, coarse mass, and NO₃⁻ were also significant contributors to haze in this park. In GRKO, on the 20% haziest and average days, organics were the largest contributors to b_{ext} (65.3% and 52.2%, respectively). On the 20% clearest days, SO₄²⁻ was the largest contributor in this park. In ROMO and GRSA, SO₄²⁻ was the largest contributor to haze on the 20% clearest days and annual average days. For the 20% haziest days, in GRSA SO₄²⁻ remained the largest contributor to haze, but in ROMO the contribution of organics exceeded that of SO₄²⁻. In GRSA, coarse mass contributed substantially to the overall b_{ext} , especially on the days having haziest visibility (Figure 2).

Glacier

20% Clearest Days

Taken: 3:00 PM

Haze = 8 dv

$B_{\text{ext}} = 23 \text{ Mm}^{-1}$

VR = 170 km



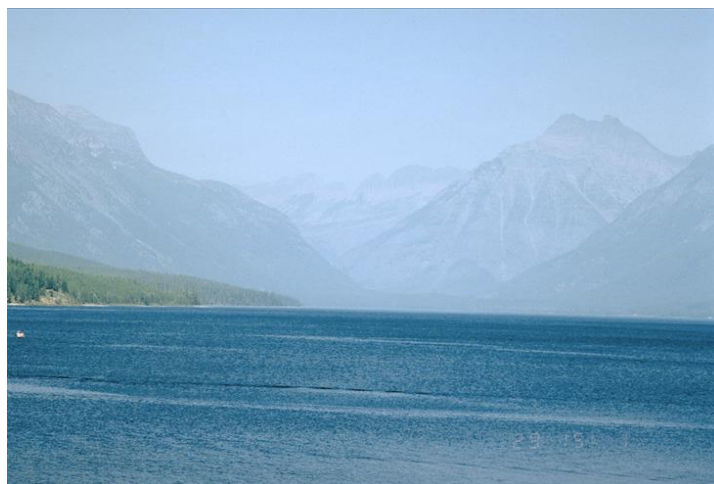
20% Haziest Days

Taken = 3:00 PM

Haze = 19 dv

$B_{\text{ext}} = 65 \text{ Mm}^{-1}$

VR = 60 km



Average Days

Taken = 3:00 PM

Haze = 13 dv

$B_{\text{ext}} = 36 \text{ Mm}^{-1}$

VR = 110 km

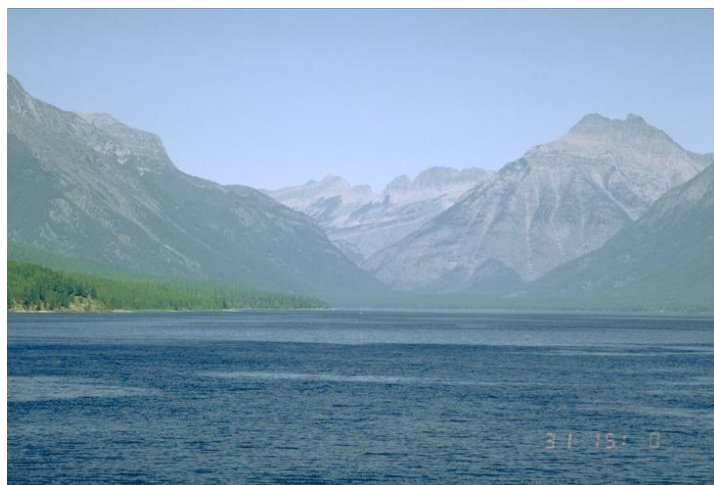


Figure 1a. Three representative photos of the same view in each of GLAC illustrating the 20% clearest visibility, the 20% haziest visibility, and the annual average visibility. B_{ext} is total particulate light extinction; VR is visual range.

Great Sand Dunes

20% Clearest Days

Taken: 9:00 AM

Haze = 4 dv

$B_{\text{ext}} = 14 \text{ Mm}^{-1}$

VR = 270 km



20% Haziest Days

Taken = 9:00 AM

Haze = 12 dv

$B_{\text{ext}} = 33 \text{ Mm}^{-1}$

VR = 120 km



Average Days

Taken = 9:00 AM

Haze = 8 dv

$B_{\text{ext}} = 22 \text{ Mm}^{-1}$

VR = 180 km



Figure 1b. Three representative photos of the same view in each of GRSA illustrating the 20% clearest visibility, the 20% haziest visibility, and the annual average visibility. Bext is total particulate light extinction; VR is visual range.

Rocky Mountain

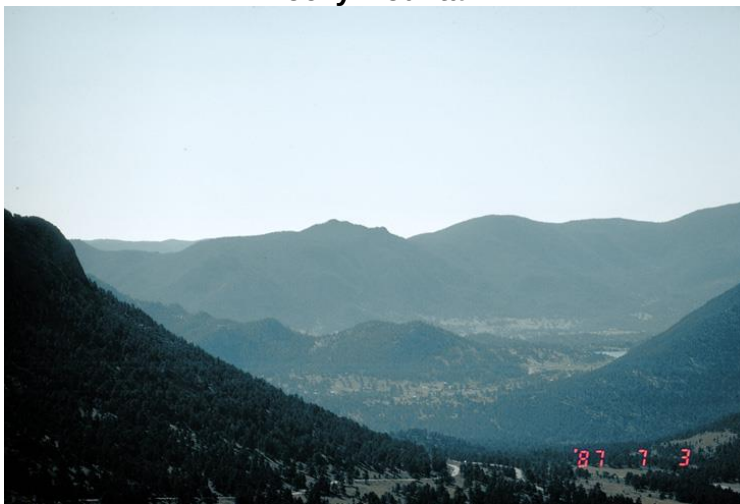
20% Clearest Days

Taken: 9:00 AM

Haze = 3 dv

$B_{\text{ext}} = 13 \text{ Mm}^{-1}$

VR = 300 km



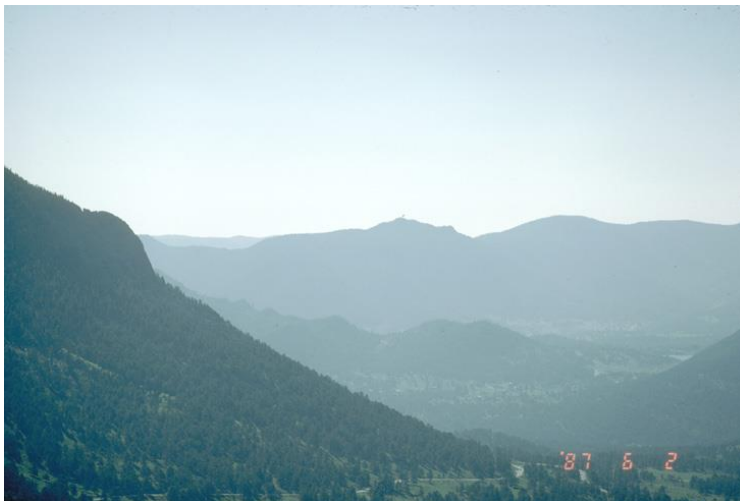
20% Haziest Days

Taken = 9:00 AM

Haze = 14 dv

$B_{\text{ext}} = 41 \text{ Mm}^{-1}$

VR = 95 km



Average Days

Taken = 9:00 AM

Haze = 7 dv

$B_{\text{ext}} = 20 \text{ Mm}^{-1}$

VR = 200 km



Figure 1c. Three representative photos of the same view in each of ROMO illustrating the 20% clearest visibility, the 20% haziest visibility, and the annual average visibility. B_{ext} is total particulate light extinction; VR is visual range.

Aerosol and optical measurements reported by Savig and Morse (1998) for the Central Rocky Mountains were made at two locations in the mountainous Class I areas of Colorado and Wyoming, at ROMO and YELL. The annual average total reconstructed extinction for the March 1988 through February 1995 period was 31.7 Mm^{-1} , of which, 68% was due to aerosol extinction. Seasonal variation was pronounced, with a maximum total extinction of 40.4 Mm^{-1} in summer and a minimum of 23.6 Mm^{-1} during winter. The seasonal variance was driven primarily by differences in organic extinction and absorption (Savig and Morse 1998). Ammonium sulfate light scattering in the northwestern United States, including the northern section of the ROMN, is unique in that it does not peak during summer (Grenon and Story 2009). For example, at Boise, Idaho and Missoula, Montana, $(\text{NH}_4)\text{SO}_4$ light scattering typically peaks during winter (Debell et al. 2006).

Light extinction in the Northern Rocky Mountains region was represented by Savig and Morse (1998) using one site in GLAC. Aerosol and optical monitoring were conducted at the site. The annual average total reconstructed light extinction for the March 1988 through February 1995 period was 57.1 Mm^{-1} , of which 82% was due to aerosol extinction. A modest seasonality in total extinction occurred, ranging from 66.9 Mm^{-1} in winter to 48.7 Mm^{-1} during spring. The seasonal variance was driven primarily by differences in SO_4^{2-} and NO_3^- extinction (Savig and Morse 1998).

Non-Rayleigh (mainly human-caused) atmospheric light extinction at ROMO, unlike many rural western areas, can have a large NO_3^- component during the winter and spring when the poorest visibility occurs. However, at other times, like in most areas, atmospheric light extinction in ROMN is typically associated with SO_4^{2-} , organics, and soil (Savig and Morse 1998). Nitrate accounts for a greater proportion of b_{ext} at ROMO than at other monitored parks in the ROMN. Historically, visibility varied mostly with patterns in weather, wind (and the effects of wind on coarse particles) and smoke from fires.

A substantial component of the regional haze observed in GLAC is caused by prescribed and wildland fires. Smoke plumes can contribute to visibility impairment downwind from the fire locations (Malm 1999). Fires have become more severe than they were historically, and this has been attributed to more extreme weather and buildup of fuel from fire suppression (Agee 1997, Allen et al. 2002, Covington 2000, Flannigan et al. 1998, McKenzie et al. 2006). This pattern may continue into the future in response to climate change. Both empirical (McKenzie et al. 2004) and process (Lenihan et al. 1998) models suggest that the area burned by wildfires is likely to increase in the western United States in response to a warming climate.

GLAC

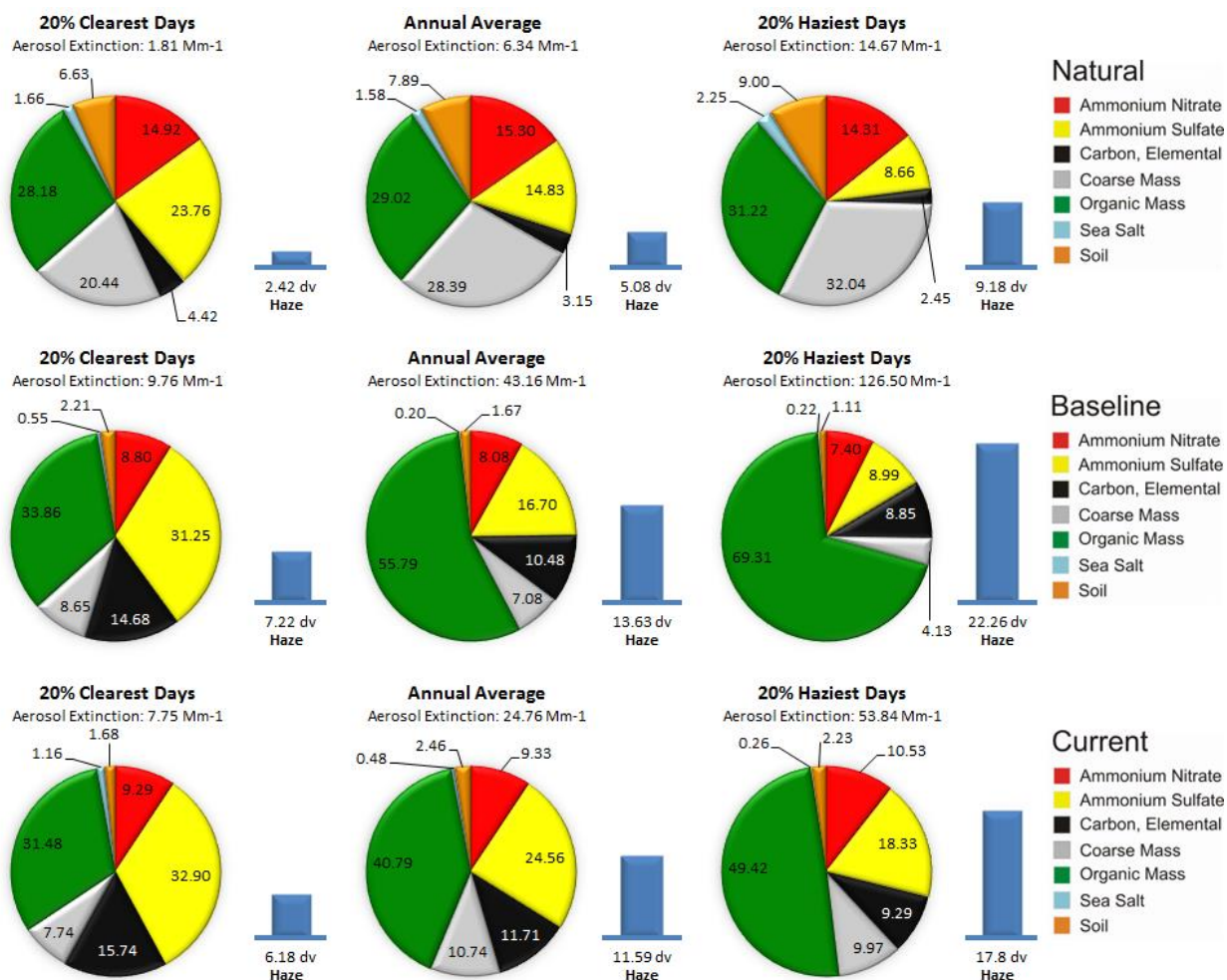


Figure 2a. Estimated natural (pre-industrial), baseline (2000-2004), and current (2006-2010) levels of haze (blue columns) and its composition (pie charts) on the 20% clearest, annual average, and 20% haziest visibility days for GLAC. There were no data available for ROMO for natural haze levels. Data for GRKO were taken from a nearby site. GLAC has no valid, measured data for the years 2002, 2003, and 2009, but has RHR2_VS substituted data for the years 2002 and 2003. (Data Source: NPS-ARD)

MONT1 (representing GRKO)

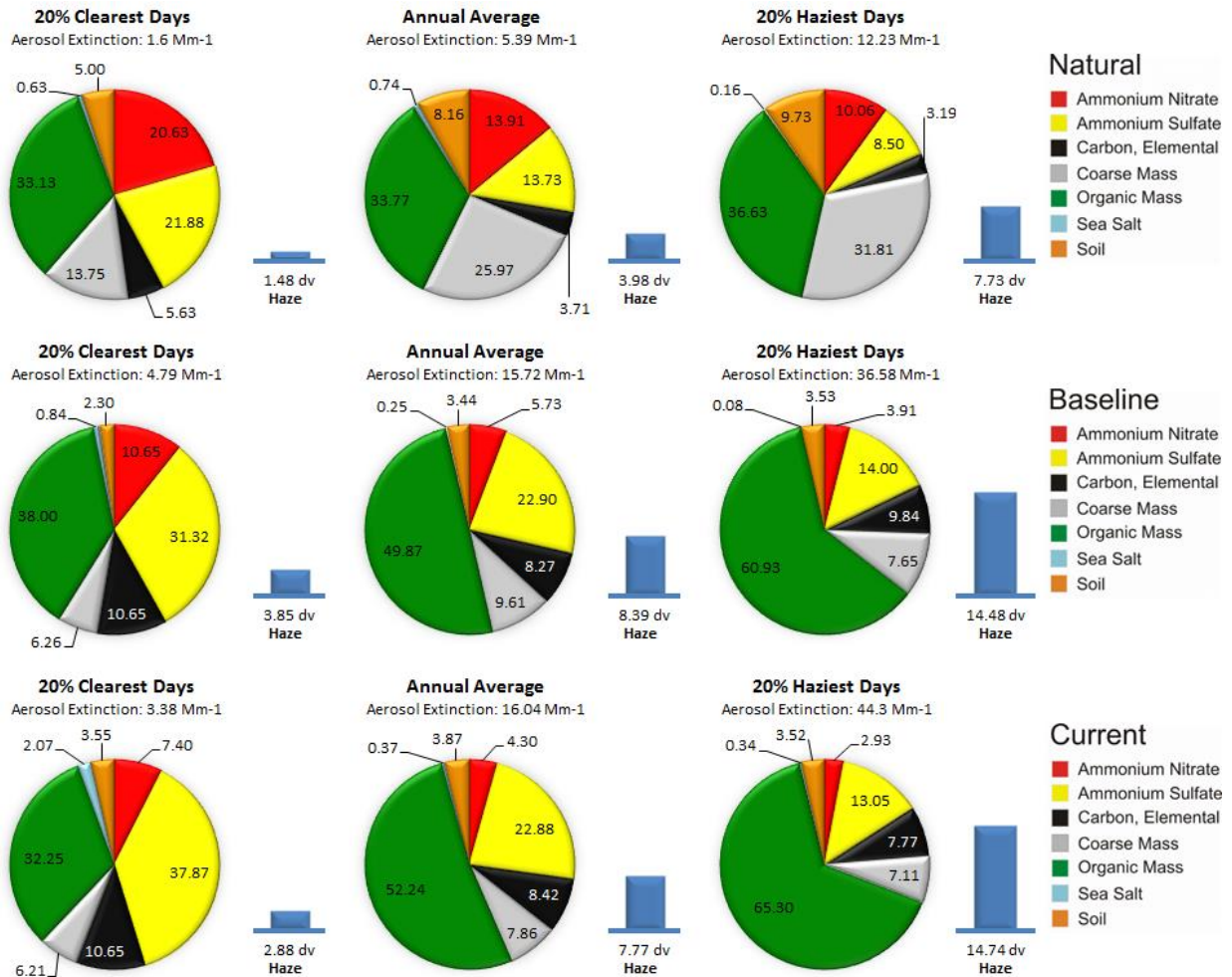


Figure 2b. Estimated natural (pre-industrial), baseline (2000-2004), and current (2006-2010) levels of haze (blue columns) and its composition (pie charts) on the 20% clearest, annual average, and 20% haziest visibility days for GRKO. There were no data available for ROMO for natural haze levels. Data for GRKO were taken from a nearby site. GLAC has no valid, measured data for the years 2002, 2003, and 2009, but has RHR2_VS substituted data for the years 2002 and 2003. (Data Source: NPS-ARD)

GRSA

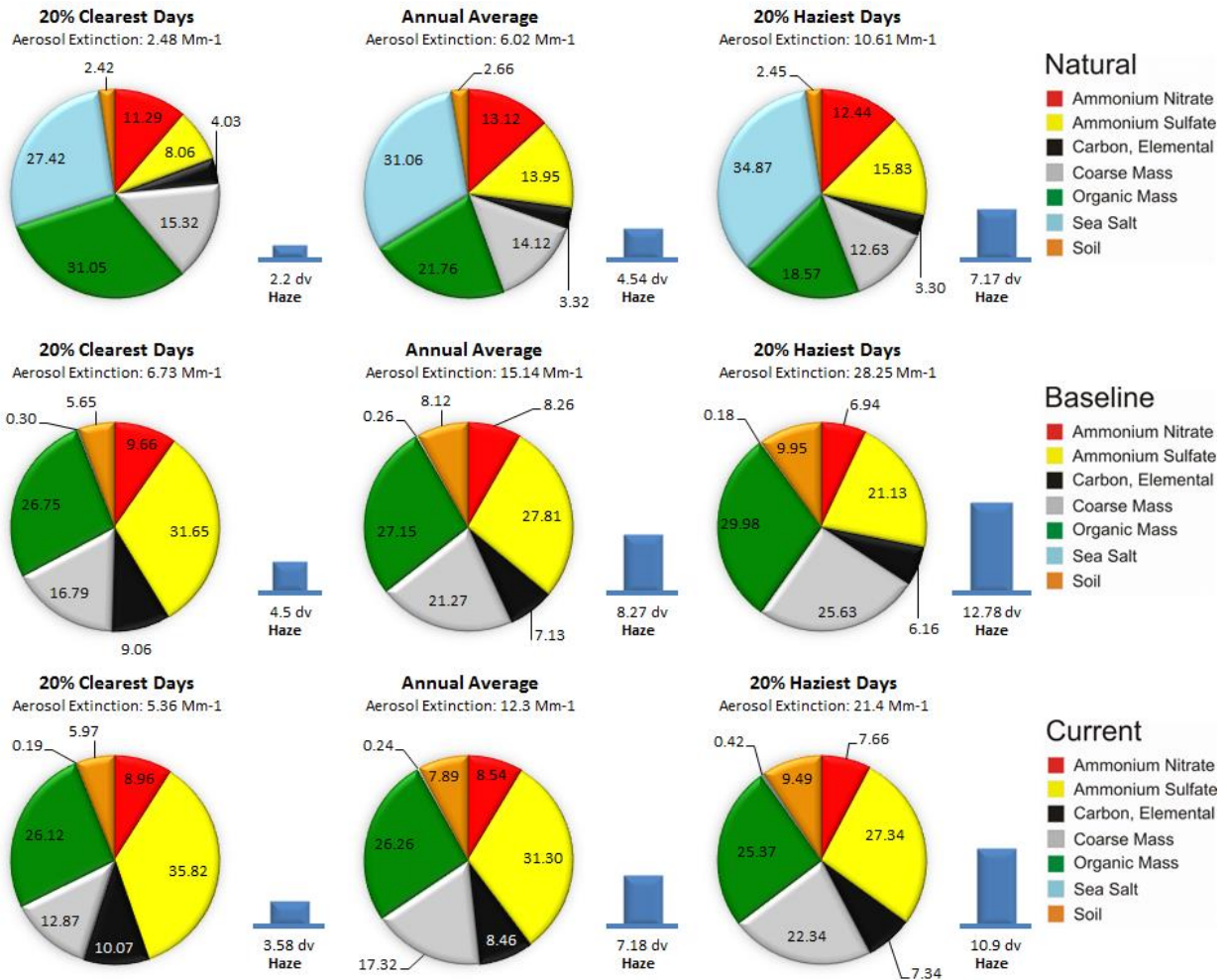


Figure 2c. Estimated natural (pre-industrial), baseline (2000-2004), and current (2006-2010) levels of haze (blue columns) and its composition (pie charts) on the 20% clearest, annual average, and 20% haziest visibility days for GRSA. There were no data available for ROMO for natural haze levels. Data for GRKO were taken from a nearby site. GLAC has no valid, measured data for the years 2002, 2003, and 2009, but has RHR2_VS substituted data for the years 2002 and 2003. (Data Source: NPS-ARD)

ROMO

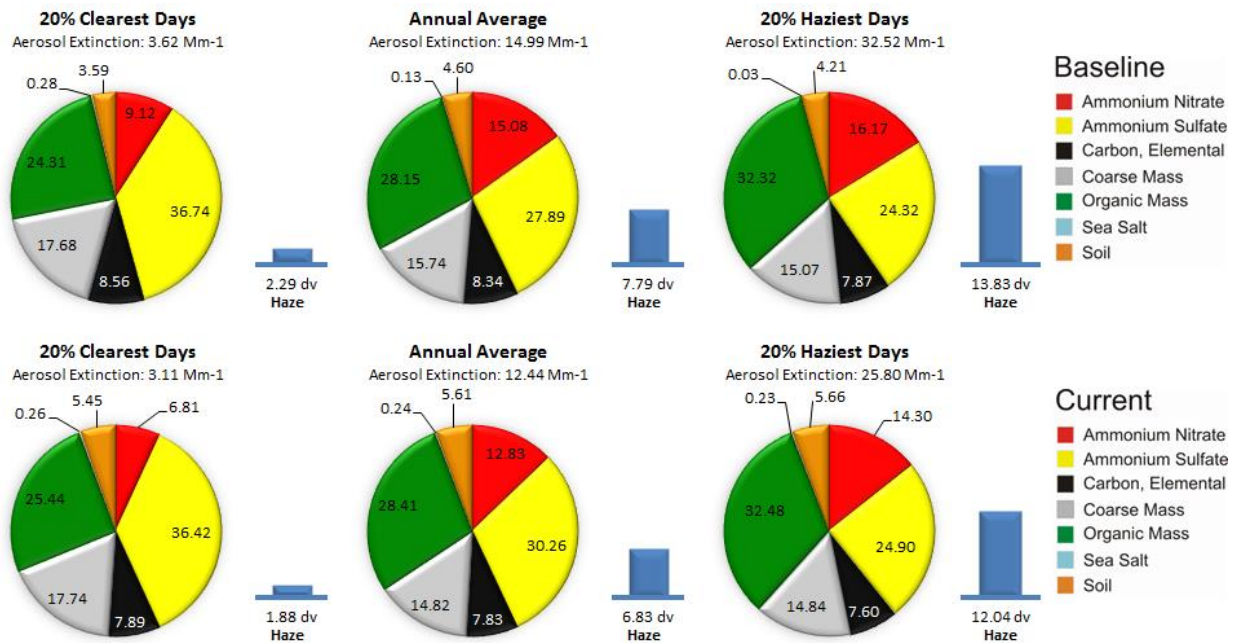


Figure 2d. Estimated natural (pre-industrial), baseline (2000-2004), and current (2006-2010) levels of haze (blue columns) and its composition (pie charts) on the 20% clearest, annual average, and 20% haziest visibility days for ROMO. There were no data available for ROMO for natural haze levels. Data for GRKO were taken from a nearby site. GLAC has no valid, measured data for the years 2002, 2003, and 2009, but has RHR2_VS substituted data for the years 2002 and 2003. (Data Source: NPS-ARD)

McKenzie et al. (2006) modeled the occurrence of wildland fires and consequent haze. They based the likelihood of fire and its severity on environmental conditions conducive to wildfire. They included three modules in their model system:

1. Climate-fire-vegetation module to estimate the effects of climate on fire regimes and vegetation succession,
2. Emissions module to calculate particulate and aerosol emissions,
3. Smoke dispersion module to track spatial patterns of impact.

The highest b_{ext} caused by smoke for Class I areas was in the Bob Marshall Wilderness and GLAC. Values along the crest of the Cascade Mountains were generally lower in response to the typical west-to-east transport of smoke by prevailing winds (McKenzie et al. 2006).

Analyses conducted by the WRAP indicated that organics from natural emissions sources, including wildfire and biogenic sources (vegetation), contribute to substantial visibility impairment throughout the western United States, including within the ROMN. In addition, air pollution sources outside the WRAP domain, including international off-shore shipping and sources from Mexico, Canada, and Asia, can in some cases be substantial contributors to haze (Suarez-Murias et al. 2009).

Trends in Visibility

NPS (2010) reported long-term trends in annual dv on the clearest and haziest 20% of days at monitoring sites in 29 national parks (<http://www.nature.nps.gov/air/who/npsPerfMeasures.cfm>). All 27 parks that showed statistically significant ($p \leq 0.05$) dv trends on the clearest days for the 11-20 year monitoring periods through 2008, including GLAC, GRSA, and ROMO, exhibited decreases in dv over time. None of the sites showed increasing trends on the clearest days.

Available haze monitoring data are shown in Figure 3 for the period of record at each park. In general, haze levels appear to be decreasing at all parks in the network over the last decade or more.

Development of State Implementation Plans

According to the RHR, states and tribes must establish and meet reasonable progress goals for each federal Class I area to improve visibility on the 20% haziest days and to prevent visibility degradation on the 20% clearest days. The national goal is to return visibility in Class I areas to natural background levels in 2064. States must evaluate progress by 2018 (and every 10 years thereafter) based on a baseline period of 2000 to 2004 (Air Resource Specialists 2007).

Progress to date in meeting the national visibility goal is illustrated in Figure 4 using a uniform rate of progress glideslope. Haze on the 20% haziest days at the monitored parks appears to be decreasing sufficiently to comply with the glideslope required by the RHR. Additional data will be needed to ensure continued progress.

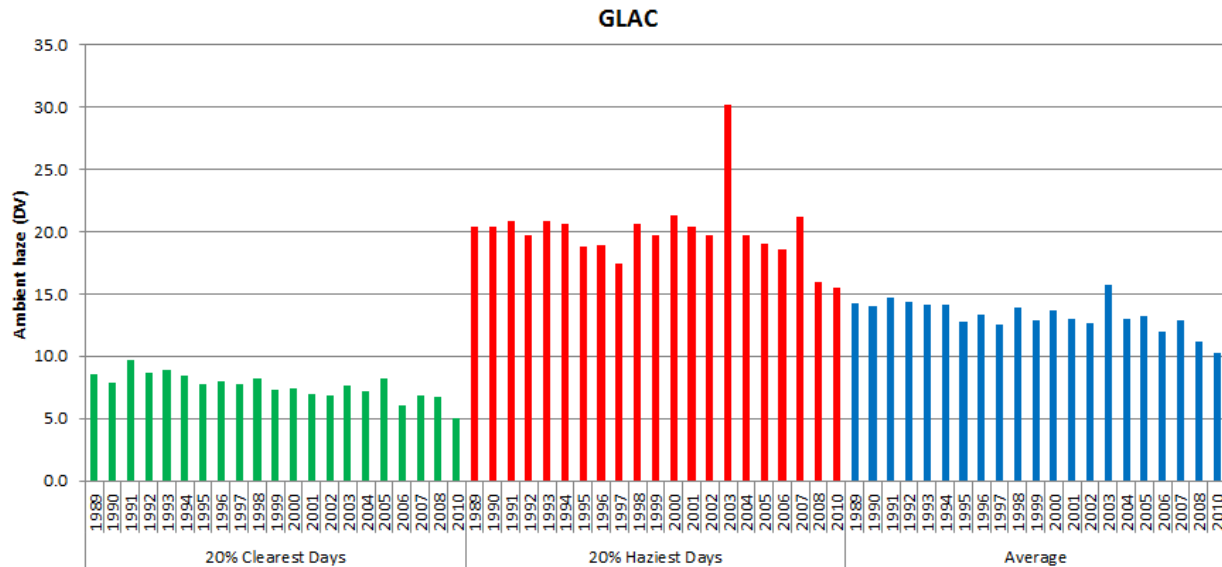


Figure 3a. Trends in ambient haze levels at GLAC, based on IMPROVE measurements on the 20% clearest, 20% haziest, and annual average visibility days over the monitoring period of record. GLAC has no valid, measured data for the years 2002, 2003, and 2009, but has RHR2_VS substituted data for the years 2002 and 2003. (Data Source: http://vista.cira.colostate.edu/improve/Data/IMPROVE/summary_data.htm)

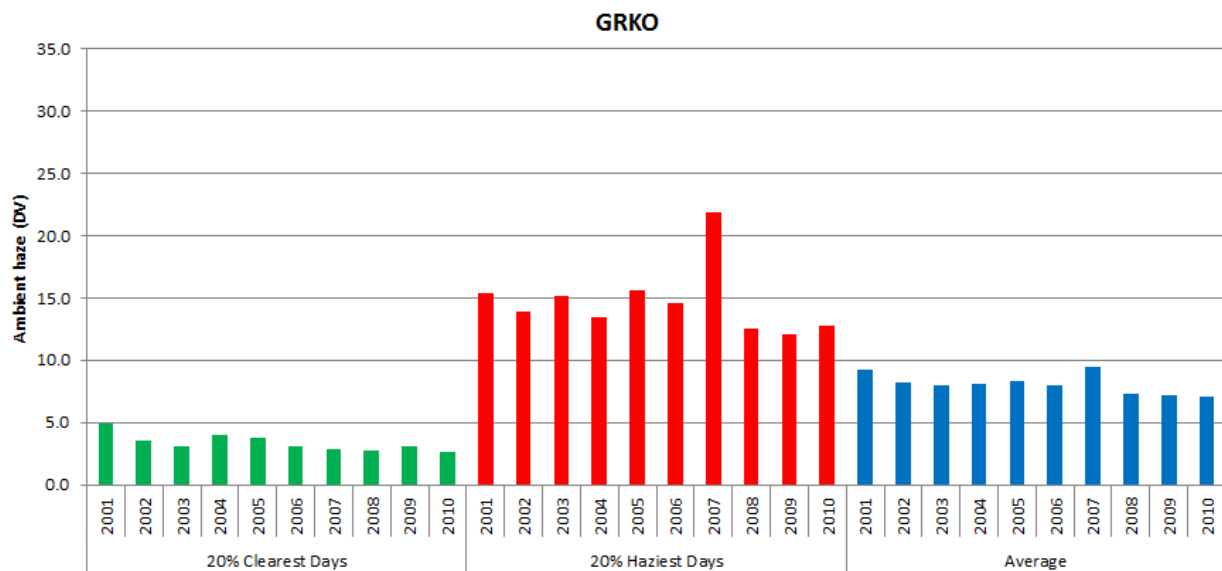


Figure 3b. Trends in ambient haze levels at MONT1 (representing GRKO), based on IMPROVE measurements on the 20% clearest, 20% haziest, and annual average visibility days over the monitoring period of record. GLAC has no valid, measured data for the years 2002, 2003, and 2009, but has RHR2_VS substituted data for the years 2002 and 2003. (Data Source: http://vista.cira.colostate.edu/improve/Data/IMPROVE/summary_data.htm)

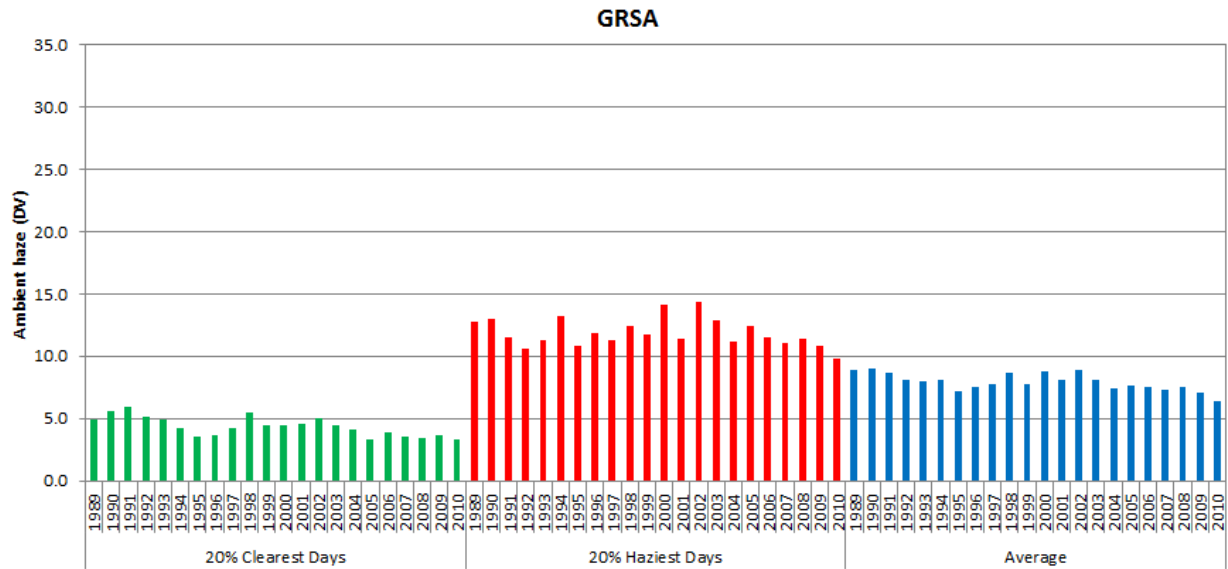


Figure 3c. Trends in ambient haze levels at GRSA, based on IMPROVE measurements on the 20% clearest, 20% haziest, and annual average visibility days over the monitoring period of record. GLAC has no valid, measured data for the years 2002, 2003, and 2009, but has RHR2_VS substituted data for the years 2002 and 2003. (Data Source: http://vista.cira.colostate.edu/improve/Data/IMPROVE/summary_data.htm)

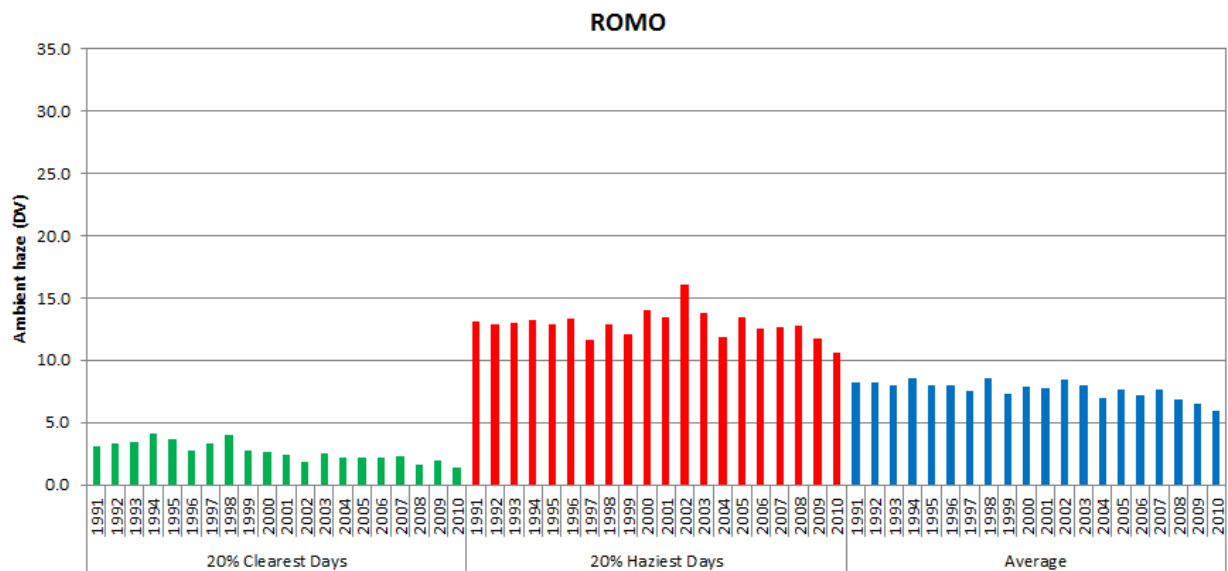


Figure 3d. Trends in ambient haze levels at ROMO, based on IMPROVE measurements on the 20% clearest, 20% haziest, and annual average visibility days over the monitoring period of record. GLAC has no valid, measured data for the years 2002, 2003, and 2009, but has RHR2_VS substituted data for the years 2002 and 2003. (Data Source: http://vista.cira.colostate.edu/improve/Data/IMPROVE/summary_data.htm)

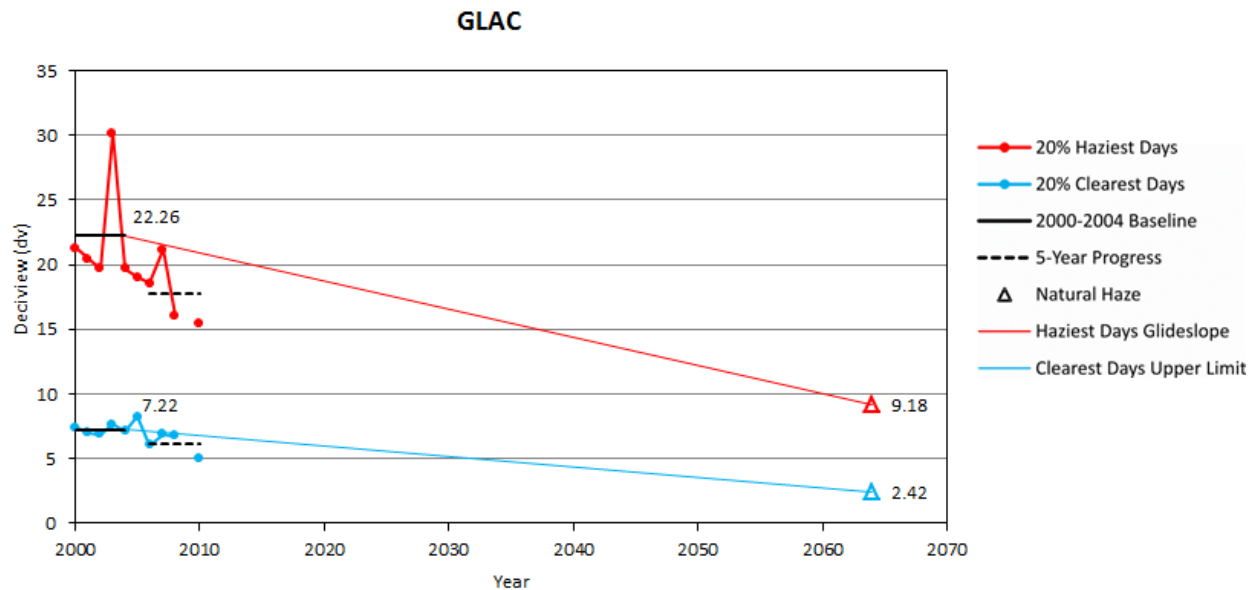


Figure 4a. Glideslopes to achieving natural visibility conditions in 2064 for the 20% haziest (red line) and the 20% clearest (blue line) days in ROMN parks. In the regional haze rule, the clearest days do not have a uniform rate of progress glideslope; the rule only requires that the clearest days do not get any worse than the baseline period. Also shown are measured values during the period 2000 to 2010. GLAC has no valid, measured data for the years 2002, 2003, and 2009, but has RHR2_VS substituted data for the years 2002 and 2003. Data from the MONT1 IMPROVE site in the Lolo National Forest are used to represent conditions at GRKO. (Data Source: http://vista.cira.colostate.edu/improve/Data/IMPROVE/summary_data.htm)

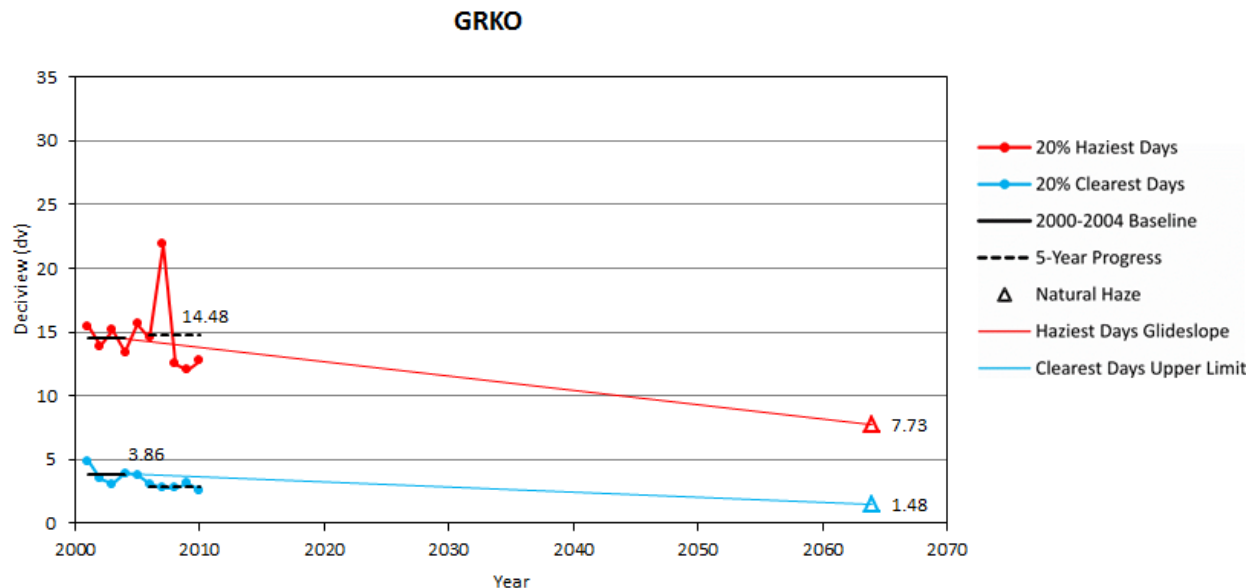


Figure 4b. Glideslopes to achieving natural visibility conditions in 2064 for the 20% haziest (red line) and the 20% clearest (blue line) days in ROMN parks. In the regional haze rule, the clearest days do not have a uniform rate of progress glideslope; the rule only requires that the clearest days do not get any worse than the baseline period. Also shown are measured values during the period 2000 to 2010. GLAC has no valid, measured data for the years 2002, 2003, and 2009, but has RHR2_VS substituted data for the years 2002 and 2003. Data from the MONT1 IMPROVE site in the Lolo National Forest are used to represent conditions at GRKO. (Data Source: http://vista.cira.colostate.edu/improve/Data/IMPROVE/summary_data.htm)

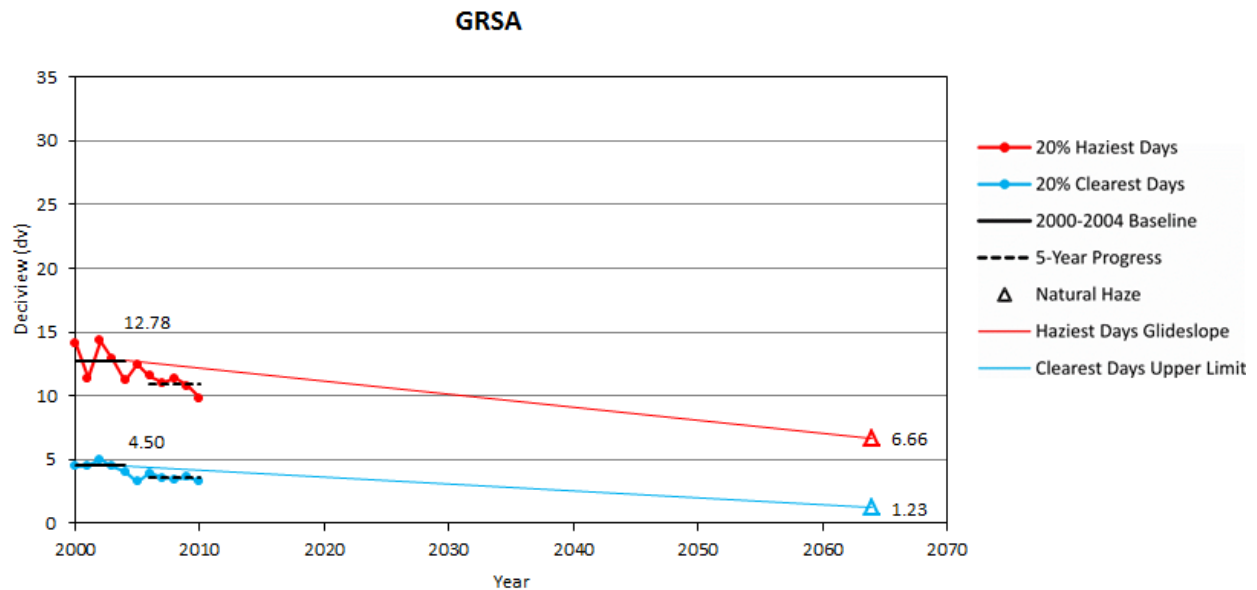


Figure 4c. Glideslopes to achieving natural visibility conditions in 2064 for the 20% hazeiest (red line) and the 20% clearest (blue line) days in ROMN parks. In the regional haze rule, the clearest days do not have a uniform rate of progress glideslope; the rule only requires that the clearest days do not get any worse than the baseline period. Also shown are measured values during the period 2000 to 2010. GLAC has no valid, measured data for the years 2002, 2003, and 2009, but has RHR2_VS substituted data for the years 2002 and 2003. Data from the MONT1 IMPROVE site in the Lolo National Forest are used to represent conditions at GRKO. (Data Source: http://vista.cira.colostate.edu/improve/Data/IMPROVE/summary_data.htm)

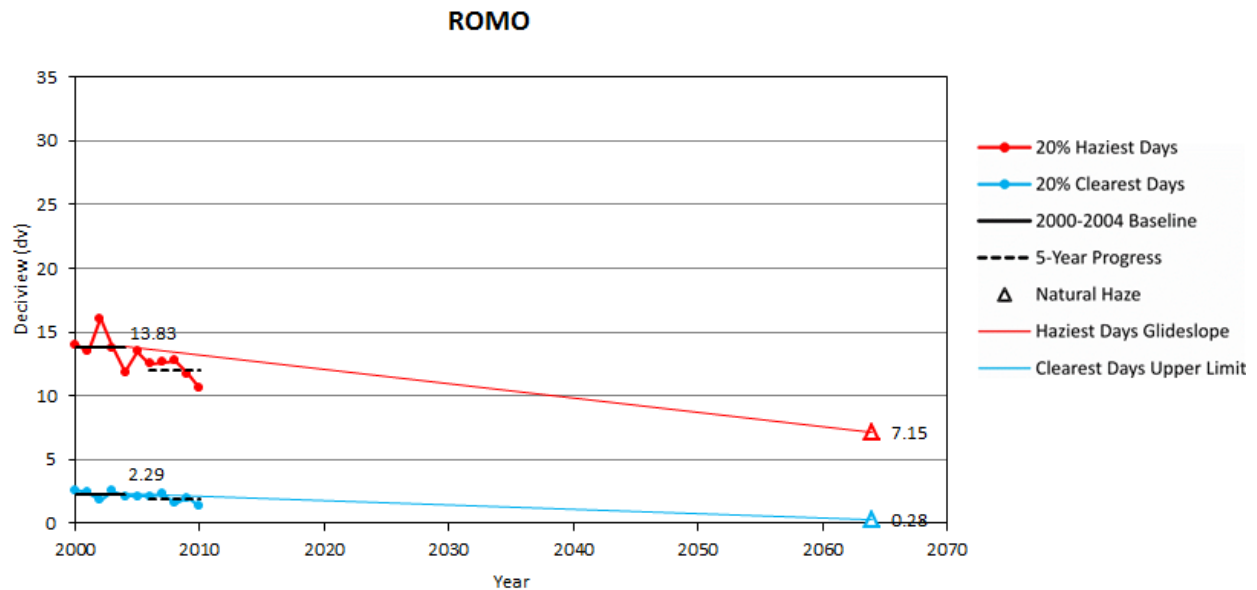


Figure 4d. Glideslopes to achieving natural visibility conditions in 2064 for the 20% haziest (red line) and the 20% clearest (blue line) days in ROMN parks. In the regional haze rule, the clearest days do not have a uniform rate of progress glideslope; the rule only requires that the clearest days do not get any worse than the baseline period. Also shown are measured values during the period 2000 to 2010. GLAC has no valid, measured data for the years 2002, 2003, and 2009, but has RHR2_VS substituted data for the years 2002 and 2003. Data from the MONT1 IMPROVE site in the Lolo National Forest are used to represent conditions at GRKO. (Data Source: http://vista.cira.colostate.edu/improve/Data/IMPROVE/summary_data.htm)

Toxic Airborne Contaminants

Semivolatile Organic Compounds

The WACAP study assessed contaminants in snow, air, water, sediment vegetation, and fish in eight western parks (http://www.nature.nps.gov/air/Studies/air_toxics/wacap.cfm) and found relatively high concentrations of pesticides in GLAC. Among all WACAP parks, GLAC had some of the highest current use pesticide concentrations in snow (along with ROMO and Sequoia/Kings Canyon national parks [SEKI] in the Sierra Nevada Network [SIEN]) and in vegetation (along with GRSA, SEKI, and Yosemite National Park [YOSE] in the SIEN; Landers et al. 2008, 2010).

The dominant CUPs detected were endosulfans and dacthal. PAH concentrations in snow, sediments, and vegetation were 10 to 100 times higher in GLAC than at any other WACAP park. Notably, PAH concentrations were higher in the Snyder Lake watershed than in the Oldman Lake watershed in GLAC, likely the result of emissions from an Al smelter in nearby Columbia Falls, Montana (Usenko et al. 2010). Concentrations of dacthal, endosulfans, hexachlorobenzene, hexachlorocyclohexane- α , chlorpyrifos, and PCBs in GLAC ranged from mid to high as compared with other WACAP parks; along with PAHs, these pollutants were highest on the west side of the park, possibly because of differences across the Continental Divide. On the east side, where there is more agricultural development, the pesticides chlorpyrifos and hexachlorocyclohexane- γ were higher as compared with the west side of the park.

Fish in Snyder Lake on the west side of GLAC had lower concentrations of pesticides than fish in Oldman Lake on the east side (Landers et al. 2008). Concentrations of dieldrin and p,p'-DDE in fish in Oldman Lake exceeded contaminant health thresholds for subsistence fishers; dieldrin in one fish exceeded the threshold for recreational fishers. These concentrations were significantly higher than those in fish from similar ecosystems in Canada (Landers et al. 2008). In Oldman Lake, the contaminant health thresholds for piscivorous birds were exceeded for DDT and for chlordane (in one fish). The DDT concentrations exceeded levels detected in many fish collections around the world, even in regions of Africa that spray DDT as a mosquito control (Landers et al. 2010).

Deposition fluxes for endosulfans and dacthal in snow were comparatively high at ROMO; accordingly, concentrations of these pollutants were also high in fish (Landers et al. 2008). Dieldrin concentrations in fish in ROMO were the highest of all fish analyzed in the WACAP study. All sampled fish exceeded contaminant health thresholds for human subsistence fishers. Some fish also exceeded thresholds for recreational fishers. However, thresholds for wildlife were not exceeded. Fish from similar ecosystems in Canada have significantly lower dieldrin concentrations than fish in ROMO (Landers et al. 2008). In the ROMO snowpack, Hg concentration was relatively high among parks in the WACAP study; however, Hg burdens in fish were relatively low, suggesting low rates of Hg methylation and bioaccumulation. Nevertheless, Hg concentrations in some fish at both ROMO study lakes exceeded the contaminant health threshold for piscivorous mammals and birds (Landers et al. 2008). In both ROMO and SEKI (in the Sierra Nevada Network), there has been a decrease in Hg deposition fluxes to the lakes since about 1990. The fluxes of Hg to all other WACAP lakes have either remained constant or increased since 1990.

Various studies have identified organochlorine compounds in remote tundra ecosystems (cf., Blais 2005, Simonich and Hites 1995, Wania and Mackay 1996). Compounds detected commonly include DDT, HCH, HCB, and PCBs. Such compounds tend to concentrate in aquatic biota in response to biomagnification processes and have been associated with disruption of endocrine systems in aquatic species.

Watras et al. (1995) measured concentrations of Hg and MeHg in 12 lakes within GLAC. They found levels much lower than those found in Wisconsin or New York. Lake DOC explained much of the variability among lakes in GLAC, as well as in Wisconsin and New York. Other factors, including SO_4^{2-} concentration, may also partially explain the differences observed.

Exposure to SOC's can potentially disrupt natural hormonal systems of fish (Kidd et al. 2007). Biological endpoints that can be used to document reproduction abnormalities include the presence of elevated levels of plasma vitellogenin (Vtg; , a protein that indicates estrogen exposure) in males and intersexuality (displaying both male and female reproductive structures; van der Oost et al. 2003). The WACAP study found evidence for endocrine disruption in fish in GLAC and ROMO. Samples from Oldman Lake in GLAC contained one intersex trout. Additionally, two male fish, one from each lake in GLAC, had high concentrations of Vtg. In the fish from Oldman Lake, this high Vtg coincided with the only measured concentration of the endocrine disrupting compound o,p'-DDT and the highest concentration of DDE found in any fish sampled in the WACAP study. Lakes in ROMO also contained intersex trout, male trout with poorly developed reproductive organs, and male fish with elevated concentrations of Vtg. The WACAP data suggested that the occurrence of intersex and increased Vtg in fish may be influenced by contaminants present in the lakes (Landers et al. 2008). However, they also acknowledged possible alternative explanations. Enhanced Vtg levels may be influenced by the species of fish sampled; elevated Vtg was found in all lakes containing fish of the genus *Oncorhynchus* (Mills Lake in ROMO and both sampled lakes in GLAC). Additionally, all intersex fish found in the WACAP study were confined to the Rocky Mountains at ROMO and GLAC. This unexpected result may have been due to low fish sample sizes that have failed to detect intersex fish in other parks.

Based on analyses conducted in WACAP, current-use pesticide concentrations were generally highest in fish collected from lakes in Sequoia National Park, followed by ROMO and GLAC, and lowest in fish from surveyed parks in the Pacific Northwest (Olympic, Mount Rainier) and Alaska (Denali and Noatak; Ackerman et al. 2008). Lake average DDT (plus metabolic breakdown products) concentrations in fish exceeded wildlife contaminant health thresholds for belted kingfisher (*Ceryle alcyon*) in Oldman Lake in GLAC. In addition, chlordane concentrations in several fish from Oldman lake exceeded health thresholds for kingfishers. These results suggest that organic contaminants deposited from the atmosphere to remote national park watersheds in the western United States, including GLAC and ROMO, can accumulate in fish to levels that are of concern regarding wildlife health. Contaminant health thresholds for mink (*Mustela vison*) and river otter (*Lutra canadensis*) were not exceeded by fish contaminant levels in any of the park lakes studied (Ackerman et al. 2008).

Schwindt et al. (2009b) observed intersex male cutthroat trout and brook trout (*Salvelinus fontinalis*) at frequencies of 9 to 33% in a majority of subalpine lakes sampled in ROMO and GLAC. In addition, male cutthroat trout, brook trout, and rainbow trout (*Oncorhynchus mykiss*) produced elevated levels of Vtg. In contrast, reproductive abnormalities were not found in fish in national parks of the Sierra Nevada, Cascade, Olympic, Brooks, or Alaska mountain ranges. Schwindt et al. (2009b) also sampled various fish species of the family Salmonidae collected prior to the era of organic pollutants (pre-1930s). In these museum specimens, they found intersex male greenback cutthroat trout (*O. clarkia stomias*) collected in the late 1800s from Twin Lakes, Colorado. This latter finding suggests that, although SOC's may be associated with some reproductive abnormalities, organic pollutants may not be the only factors involved in reproduction disruption in fish in high-elevation Rocky Mountain lakes (Schwindt et al. 2009b).

Low rates of the intersex condition are probably normal (Schwindt et al. 2009b). It is difficult to determine the levels of intersex occurrence rate that would signify that the fish had experienced an endocrine disrupting exposure. However, Schreck and Kent (2013) judged that, because the sampled water bodies in ROMO show such a high intersex frequency (~ 50%) and because the condition occurs in multiple species, there appears to be a sound basis for concluding that the rate of intersex occurrence in ROMO is not the natural condition.

A follow-on study to WACAP was conducted during the summer of 2009 in high-elevation lakes and streams in several western national parks including ROMO, GRSA, and GLAC (Keteles 2011). The concentrations of pesticides in these surface waters were low and far below the aquatic life benchmarks established by the U.S. EPA (2011).

Some of the highest pesticide concentrations in vegetation in the WACAP study were measured at GRSA, along with GLAC, YOSE, and SEKI. Dominant SOC's included PAHs, endosulfans, dachthal, DDTs, hexachlorobenzene, hexachlorocyclohexane- α , chlordanes, and g-HACH, followed by low concentrations of PCBs.

Keteles (2011) collected surface water samples from high-elevation areas of ROMO, GRSA, GRTE, and GLAC during the summer, 2009 and analyzed them for pesticides. Increased contaminant deposition can occur at high elevation in the Rocky Mountains due to high precipitation, upslope winds, and the process of cold condensation, whereby chemicals volatilize in warm locations (i.e., low-elevation agricultural areas) and subsequently condense when they reach colder areas in the mountains (Blais et al. 1998). Concentrations of pesticides in water were compared with EPA Office of Pesticide Programs Aquatic life benchmarks for freshwater species and to toxicity values determined in other programs (Kegley et al. 2011). The measured concentrations in this screening study suggested low pesticide concentration values in these national parks, far below the aquatic life benchmarks (Keteles 2011).

Organochlorine chemicals can be estrogenic (Garcia-Reyero et al. 2007), contributing to the occurrence of intersex fish, and can accumulate in the aquatic food chains of remote mountain ecosystems (Blais et al. 1998). Biomarkers of exposure to estrogenic chemicals suggest the likelihood of reproductive dysfunction (Harries et al. 1997). Organochlorines are likely transported

as atmospheric contaminants to high-elevation sites (Hageman et al. 2006). Work by Schreck and Kent (2013) followed up on some of the work performed in the WACAP Study. They expanded the number of study sites in ROMO and expanded the range of coverage to additional western parks. The extent of endocrine disruption in fish was assessed in ROMO, GLAC, and GRSA within ROMN, plus eight other parks.

The low observed frequency of intersex fish in most study parks and water bodies may be a natural phenomenon (Schwindt et al. 2009a). The extent to which human-caused contaminants contribute to an increased frequency is difficult to determine (Schreck and Kent 2013). Of the western parks studied, the frequency of intersex fish occurrence was highest in ROMO. Of the five lakes sampled in ROMO, two had intersex fish. No intersex fish were found in GLAC, GRTE, or GRCA (n=21-45 male fish sampled in each). This condition has been documented in ROMO for multiple species and sampling occasions. The significance to fish population dynamics and the cause of the condition are not known.

Hageman et al. (2010) reported results of pesticide analyses of snowpack at remote alpine, arctic, and subarctic sites in eight national parks, including ROMO and GLAC. Various current use pesticides (CUPs; dacthal, chlorpyrifos, endosulfans, and γ -hexachlorocyclohexane [HCH]) and historic-use pesticides (HUPs; dieldrin, α -HCH, chlordanes, and hexachlorobenzene) were commonly measured at all sites and years (2003-2005). The pesticide concentration profiles were unique for individual parks, suggesting the importance of regional sources.

The distribution of CUPs among the parks were explained, using mass back trajectory analysis, based on the mass of individual CUPs used in regions located one-day upwind of the parks. For most pesticides and parks, more than 75% of the snowpack pesticide burden was attributed to regional transport. The authors concluded that the majority of pesticide contamination in U.S. national parks is due to regional pesticide applications.

Fluoride

Potential impacts of atmospheric fluoride (F) deposition constituted a possible source of concern in GLAC due to emissions from the Columbia Falls Aluminum Company plant. Fluoride injury in conifer needles causes tip burn, reduced diameter growth and abscission of older foliage (Shaw et al. 1951). Several plant species are known to be sensitive to atmospheric F, including Oregon grape (*Mahonia aquifolium*), blueberry (*Vaccinium* spp.), common barberry (*Berberis vulgaris*), and young needles of many coniferous trees (Schoettle et al. 1999). Exposure to high levels of atmospheric F has also been shown to cause depletion of lichen populations (Gilbert 1973). Most F tree sensitivity studies have focused on ponderosa pine (both varieties). Early accounts of F injury in ponderosa pine were reported near Kaiser Aluminum Company in Mead, Washington, with symptoms of retarded stem-diameter growth and foliar necrosis (Adams et al. 1956, Lynch 1951, Shaw et al. 1951). A study conducted in the vicinity of Harvey Aluminum Company in The Dalles, Oregon, found foliar injury in ponderosa pine to be attributed to elevated F (Compton et al. 1961). Later studies done in Flathead National Forest and GLAC in 1974 indicated that western white pine appeared to be the most sensitive conifer to F (Carlson and Dewey 1971); ponderosa pine, lodgepole pine (*Pinus contorta*), and Douglas-fir (*Pseudotsuga menziesii*) were moderately sensitive, and Engelmann

spruce, western red cedar (*Thuja plicata*), and subalpine fir (*Abies lasiocarpa*) were most tolerant. Of the shrub species observed, Carlson and Dewey found that chokecherry (*Prunus virginiana*) and serviceberry (*Amelanchier alnifolia*) appeared more sensitive to F than other shrubs. False-lily-of-the-valley (*Maianthemum dillatatum*) and disporum (*Disporum hookeri*) were the most sensitive of the herbaceous species studied. Implementation of emissions control technologies to control F emissions from the Al industry greatly reduced F emissions within the interior Columbia River basin (Schoettle et al. 1999). There are no data to suggest that atmospheric F continues to stress native vegetation at GLAC, since the aluminum plant ceased operation in 2009.

Mercury

Krabbenhoft et al. (2002) sampled 90 lakes in seven national parks in the western United States and analyzed their Hg and methylmercury (MeHg) concentrations. The parks included Lassen Volcano in the Klamath Network, YOSE and SEKI in the Sierra Nevada Network, GLAC and ROMO in the ROMN, and YELL and GRTE in the GRYN. Levels of MeHg were lowest in GLAC (0.02 ng/L) and similar among the other parks (~ 0.05 ng/L).

Elevated levels of Hg have been measured in fish caught from several lakes in GLAC and adjacent Western Lake National Park in Canada, resulting in fish consumption advisories (Downs and Stafford 2009). Relationships between Hg concentrations and fish size varied by lake and species. Especially strong relationships were found for lake trout (*Salvelinus namaycush*) in Harrison, McDonald, St. Mary, and two other nearby lakes (Watson and Flathead; Brinkmann 2007, Stafford et al. 2004).

Downs and Stafford (2009) sampled fish in four lakes in GLAC and analyzed them for Hg content. Study lakes included Lake McDonald, Bowman Lake, and Harrison Lake on the west side and St. Mary Lake on the east side of the park. Atmospheric Hg deposition is presumed to be an important source of the Hg found in park fish. In McDonald and St. Mary lakes, five and seven fish species were sampled, respectively. The concentrations of Hg in fish tissue varied by species and lake. Lake trout were captured in all four lakes. In general, the top predators, such as lake trout and burbot (*Lota lota*), had the highest Hg concentrations (both absolute and normalized by size).

Eagles-Smith et al. (2014) sampled fish in 21 national parks and analyzed them for Hg concentrations in tissue. Results varied substantially by park and by water body. Fish from 19 lakes in ROMO were sampled. The mean Hg concentration in fish collected in ROMO was slightly lower than the study-wide mean across all parks. However, variability was high, with concentrations of Hg in fish ranging from about 20 ng/g ww (Lake Haiyaha) to 121 ng/g ww (Mirror Lake). Relatively large (> 250 mm) brook trout (*Salvelinus fontinalis*) from Mirror Lake and suckers from the Colorado and Fall rivers were judged likely to exceed the EPA human health criterion and the avian reproductive impairment benchmark. Data collected from two sites sampled in GRSA suggested that risk of Hg contamination to fish in this park was small.

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