



Air Quality Related Values (AQRVs) for Eastern Rivers and Mountains Network (ERMN) Parks

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

Natural Resource Report NPS/ERMN/NRR—2016/1165



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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Contents

	Page
Figures.....	iv
Tables.....	iv
Maps.....	v
Summary.....	vi
Background.....	1
Atmospheric Emissions and Deposition	2
Acidification	7
Acidification of Terrestrial Ecosystems	7
Acidification of Aquatic Ecosystems	15
Nutrient Nitrogen Enrichment	20
Ozone Injury to Vegetation	22
Visibility Degradation.....	27
Toxic Airborne Contaminants	28
References Cited	29

Figures

	Page
Figure 1. Ozone injury to milkweed at ALPO	26

Tables

	Page
Table 1. Average changes in S and N deposition between 2001 and 2011 across park grid cells at ERMN parks	8
Table 2. Estimated I&M park rankings ¹ according to risk of acidification impacts on sensitive receptors	13
Table 3. Distribution of steady-state critical loads of sulfur (in kg/ha/yr) to protect stream ANC = 50 µeq/L for parks modeled in the EMDS project within the ERMN	17
Table 4. Empirical critical loads for nitrogen in the ERMN, by ecoregion and receptor from Pardo et al. (2011).....	23
Table 5. Ozone-sensitive and bioindicator plant species known or thought to occur in the I&M parks of the ERMN	24
Table 6. Ozone assessment results for I&M parks in the ERMN 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 1995-1999 ¹	25

Maps

	Page
Map 1. Network boundary and locations of park units and population centers greater than 10,000 people within the ERMN region.	3
Map 2. Total SO ₂ emissions, by county, near the ERMN for the year 2011	4
Map 3. Total NO _x emissions, by county, near the ERMN for the year 2011.....	5
Map 4. Total NH ₃ emissions, by county, near the ERMN for the year 2011.....	6
Map 5. Total S deposition for the three-year period centered on 2011 in and around the ERMN	10
Map 6. Total oxidized inorganic N deposition for the three-year period centered on 2011 in and around the ERMN	11
Map 7. Reduced inorganic N deposition for the three-year period centered on 2011 in and around the ERMN	12
Map 8. Total N deposition for the three-year period centered on 2011 in and around the ERMN	13
Map 9. Steady-state critical load of atmospheric S deposition for maintaining streamwater ANC at or above 50 µeq/L in NERI and GARI	18

Summary

This report describes the Air Quality Related Values (AQRVs) of the Eastern Rivers and Mountains Network (ERMN). AQRVs are those resources sensitive to air quality and include streams, lakes, soils, vegetation, fish and wildlife, and visibility. The ERMN parks that are included in the NPS Inventory and Monitoring (I&M) Program, and discussed in this report, are Allegheny Portage Railroad National Historic Site (ALPO), Bluestone National Scenic River (BLUE), Delaware Water Gap National Recreation Area (DEWA), Fort Necessity National Battlefield (FONE), Friendship Hill National Historic Site (FRHI), Gauley River National Recreation Area (GARI), Johnstown Flood National Memorial (JOFL), New River Gorge National River NERI), and Upper Delaware Scenic and Recreational River (UPDE).

Sullivan et al. (2011a, 2011b) and Kohut (2007) conducted risk assessments for acidification, eutrophication, and ozone (O_3) for all of the ERMN parks; their results are described in this report. This report also describes air pollutant emissions and air quality in the ERMN, 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 O_3 ; haze-causing particles; and airborne toxics. Background for this section can be found in “Air Quality Related Values (AQRVs) in National Parks. Effects from Ozone; Visibility Reducing Particles; and Atmospheric Deposition of Acids, Nutrients and Toxics” (Sullivan 2016).

The ERMN region borders the eastern Great Lakes and contains a number of relatively large human population centers. Portions of this network receive some of the highest levels of sulfur (S) and nitrogen (N) deposition in the United States. The southern portion of the network, in particular, is frequently downwind of the Ohio River Valley, which contains many large point sources of S and N emissions. Although emissions of S and N in and upwind of the ERMN have been very high, they have decreased dramatically over the past several decades, largely in response to emissions controls required by the Clean Air Act and its Amendments.

Sulfur and N pollutants can cause acidification of streams, lakes, and soils. Acidic deposition has been recognized as an important factor causing the observed decreases in concentrations of exchangeable base cations on soils in the region around the ERMN. Base saturation values less than 10% predominate in the B horizon in areas where soil and surface water acidification from acidic deposition have been most pronounced, including hardwood forests on the Allegheny Plateau. Acidic deposition has contributed to a decline in the availability of Ca and other base cations in the soils of acid-sensitive forest ecosystems in the region around the ERMN by increasing the leaching of base cations from foliage and from the primary rooting zone, and by the mobilization of aluminum (Al) from soils to soil solution and drainage water. A long-term decrease in soil pH since 1960 in Pennsylvania hardwood forests has been documented, along with decreases in soil calcium (Ca) and magnesium (Mg).

Biological effects of terrestrial acidification by S and N deposition in the region around the ERMN have included reduced growth and increased stress to overstory trees, and likely changes in the species distributions of understory plants. Included among the direct responses of forest trees to

acidic deposition are increased leaching of nutrients from foliage; accelerated weathering of leaf cuticular surfaces; increased permeability of leaf surfaces to toxic materials, water, and disease agents; and altered reproductive processes. Several studies in Pennsylvania have demonstrated that sugar maple (*Acer saccharum*) decline is linked to the occurrence of relatively high levels of acidic deposition and the presence of base-poor soils. In general, declining sugar maple trees in this region were found at higher elevation, on geologies dominated by sandstone, with base-poor sandy soils having high percentages of rock fragments. Estimates of the empirical critical load (CL) for protecting mycorrhizal fungi and lichens suggest CL exceedance at all parks in the network.

Eastern hardwood forests from Wisconsin to Maine and south to Virginia, including the region around the ERMN, have been shown to have experienced increased tree growth with increasing N deposition across a depositional gradient from about 3 to 11 kg N/ha/yr. Responses were species-specific. Eastern hardwood forests can also respond to increased N deposition with a decrease in biodiversity. The response of the herbaceous layer of the forest can be pronounced, with initial increases in herbaceous plant cover in response to increased N input, followed by decreased species richness and evenness. Effects such as these on hardwood forests have not been studied in the ERMN parks.

Both chronic and episodic stream acidification have been widespread within the region around the ERMN. Most streams draining Dolly Sods and Otter Creek wilderness areas in West Virginia are acidic or low in acid neutralizing capacity (ANC), largely as a consequence of both acidic deposition and acid mine drainage. A thorough characterization of episodic variations in stream chemistry was conducted through the U.S. EPA Episodic Response Project (ERP). Thirteen low-order streams with watershed areas less than 24 km² in the Adirondack and Catskill regions of New York and the Appalachian Plateau in Pennsylvania were monitored from 1988 to 1990. Results of research in the ERP showed that acid episodes reduced the size of fish populations and eliminated acid-sensitive species when median high-flow pH was less than 5.2 and inorganic monomeric Al (Al_i) concentrations exceeded 3.7 µM.

Monitoring data collected during the 1990s in the Long Term Monitoring (LTM) and Temporally-Integrated Monitoring of Ecosystems (TIME) projects illustrated large declines in sulfate (SO₄²⁻) concentrations in surface waters over the 10 years of monitoring. Surface water nitrate (NO₃⁻) concentrations also decreased in the northern Appalachian Plateau. The U.S. EPA's TIME surveys of streams in the northern Appalachian Plateau region suggested that 5,014 km of streams (11.8% of the total stream length) were acidic in 1993-94, but that only 3,600 km of streams (8.5% of the total stream length) remained acidic in this region in 2000. This represented about a 28% decrease in acidic stream length over the preceding decade.

There were many S point source emissions of substantial magnitude within the vicinity of the ERMN in 2002. Some emitted in excess of 40,000 tons of S per year. Most of the ERMN and the surrounding area received S deposition at levels between 15 and 20 kg S/ha/yr, to greater than 30 kg S/ha/yr in 2002 at some locations. Almost the entire network received in excess of 10 kg N/ha/yr of total N deposition. Atmospheric S, and to a lesser extent N, deposition levels have declined at all

ERMN parks except ALPO since 2001, based on TDEP estimates. Decreases in total S deposition over the previous decade for the parks in this network were commonly in the range of 30% to 40%. Estimated total N deposition decreased over that same time period at all parks in the network, in most cases by more than 20%.

NPS (2010) reported long-term trends in concentrations of mercury (Hg) in wet deposition through 2008 for nine national parks. Statistically significant ($p \leq 0.05$) decreases in wet Hg deposition in recent years were reported for only two of the monitored parks, including Allegheny Portage Railroad National Historic Site (ALPO) in the ERMN.

Ozone pollution can harm human health, reduce plant growth, and cause visible injury to foliage. The highest O₃ concentrations in the United States in 2007 were located in six states, including Pennsylvania (U.S. EPA 2008). Ozone exposures were at levels known to injure sensitive vegetation. Nevertheless, there have been widespread recent improvements in atmospheric O₃ levels (U.S. EPA 2008). One of the sites that showed the greatest improvement since the 1990s was near Chambersburg, PA.

Particulate pollution can cause haze, reducing visibility. Sulfate is the most important cause of fine particle pollution and visibility impairment across the Mid-Atlantic/Northeast Visibility Union (MANE-VU) states. A variety of model estimates suggested that emissions from the MANE-VU states accounted for about 25 to 30% of the fine particulate SO₄²⁻ observed at Class I sites within the MANE-VU region. At the Class I sites in this region, SO₄²⁻ accounts for about one-half to two-thirds of total fine particle mass on the 20% haziest days. Within the northeastern United States, visibility improvement has been most pronounced in the western and southern portions of the MANE-VU region, in the area encompassing the ERMN. MANE-VU is focusing primarily on fine particle SO₄²⁻ abatement in its initial efforts to comply with the Regional Haze Rule (RHR; NESCAUM 2006). Controls on other haze-producing constituents, such as organic carbon (C) and both urban and mobile sources of nitrogen oxides (NO_x; mainly during winter) will also be important.

Airborne toxics, including Hg, can bioaccumulate to levels that can harm humans, fish, and wildlife. During the period 2002-2008, based on data from three Hg and precipitation monitoring networks, areas in eastern Pennsylvania (including within the ERMN region), as well as in southern Indiana, Illinois, and central Michigan, had among the highest estimated levels of wet Hg deposition. Within the northern portions of the Great Lakes Network (GLKN) and the ERMN, Hg is well known as constituting a substantial risk to the health of wildlife. The major source of Hg in the watersheds of this region is atmospheric deposition, most of which originates from human-caused Hg emissions sources.

Background

The Eastern Rivers and Mountains Network contains nine parks: Allegheny Portage Railroad National Historic Site (ALPO), Bluestone National Scenic River (BLUE), Delaware Water Gap National Recreation Area (DEWA), Fort Necessity National Battlefield (FONE), Friendship Hill National Historic Site (FRHI), Gauley River National Recreation Area (GARI), Johnstown Flood National Memorial (JOFL), New River Gorge National River (NERI), and Upper Delaware Scenic and Recreational River (UPDE). Two parks (DEWA and NERI) are slightly larger than 100 square miles. 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.

Map 1 shows the network boundary along with locations of each park and population centers with more than 10,000 people. There are no population centers within the ERMN region that are larger than 500,000 people, although Philadelphia is within 125 miles of two network parks (UPDE and DEWA). There are several other cities with populations between 100,000 and 500,000 within the network region (Sullivan et al. 2011b).

Atmospheric Emissions and Deposition

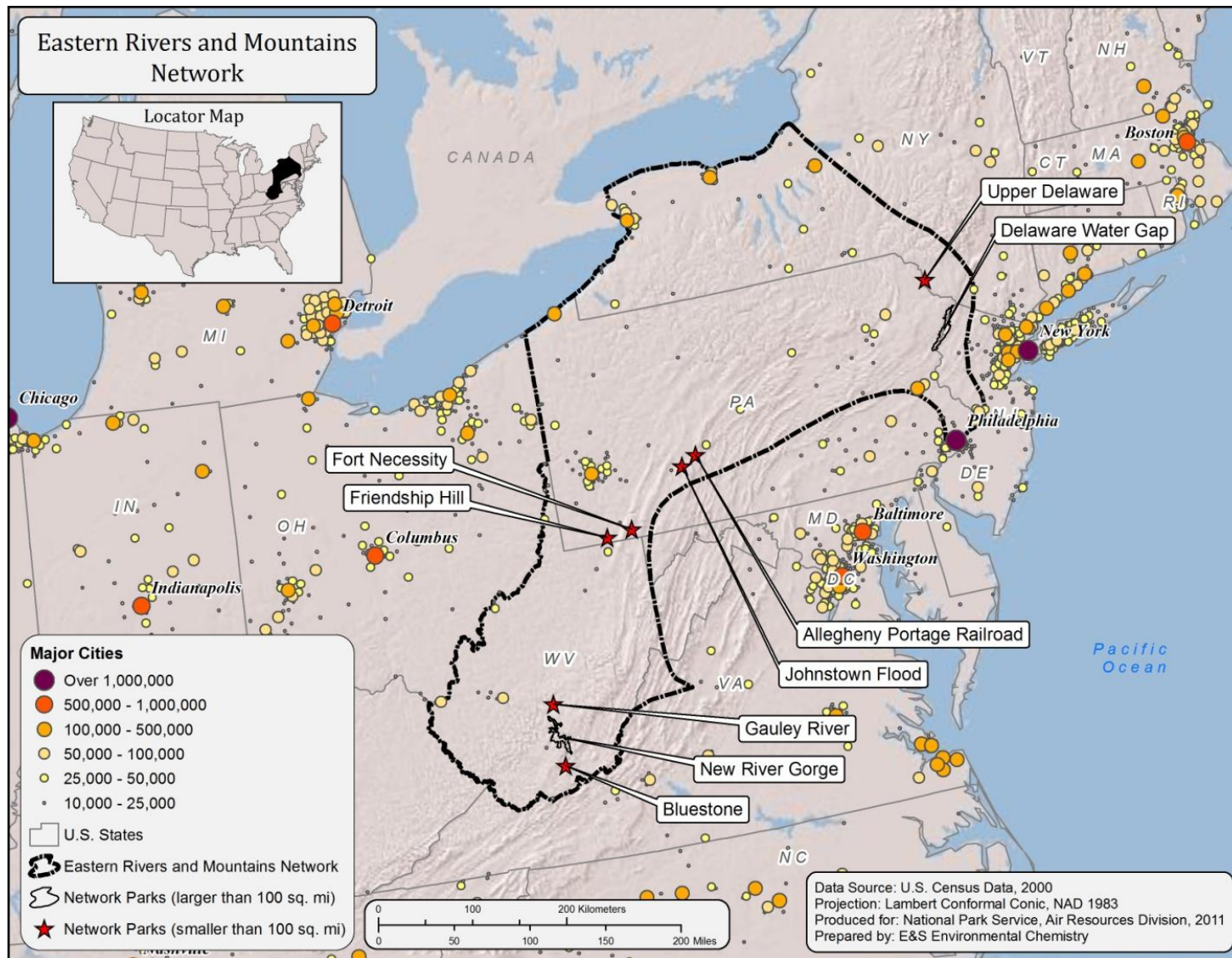
An analysis of 2002 emissions found that there were many sulfur dioxide (SO₂) point source emissions of substantial magnitude within the ERMN region, and a few just west of the network area (Sullivan et al. 2011b). Some emitted in excess of 40,000 tons per year. County-level annual nitrogen (N) emissions within the ERMN region ranged from less than 1 ton per square mile to greater than 100 tons per square mile (Sullivan et al. 2011b). A number of counties to the west of the network had total annual N emissions higher than 50 tons per square mile. The largest point sources were consistently sources of oxidized, rather than reduced, N. Many NO_x point sources around the ERMN were larger than 5,000 tons per year, mainly located in the mid-section of the network region.

County-level emissions near the ERMN, 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 SO₂, oxidized N (NO_x), and reduced N (NH₃), respectively. Many counties near the ERMN parks had relatively high SO₂ emissions (> 50 tons/mi²/yr; Map 2). Spatial patterns in NO_x emissions were generally similar, with high values near some of the ERMN parks (Map 3). Emissions of NH₃ near ERMN parks were somewhat lower, with most counties showing emissions levels below 8 tons/mi²/yr (Map 4).

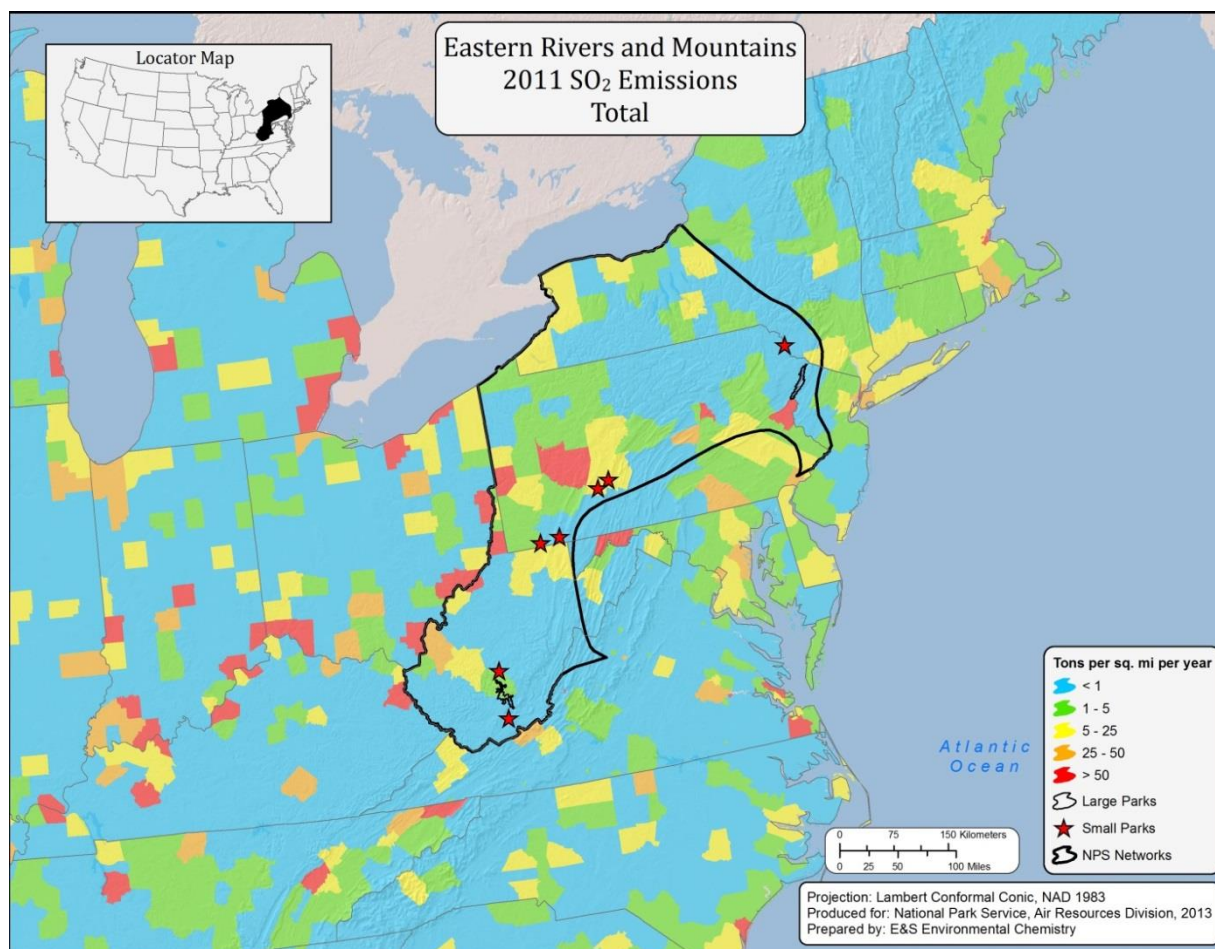
Emissions of SO₂ in the vicinity of the ERMN have decreased dramatically over the past several decades. The largest decrease in yearly 80th percentile atmospheric sulfate (SO₄²⁻) concentration during the period 1988 to 1999 occurred at Dolly Sods wilderness in West Virginia (73% decrease; Malm et al. 2002).

The dominant nonpoint sources of N to watersheds vary across the continental United States (U.S. Geological Survey [USGS] 1999). In most areas, animal manure and fertilizer application account for the vast majority of nonpoint 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 included USGS study watersheds throughout the area around the ERMN (USGS 1999).

Total S deposition in 2002 reported by Sullivan et al. (2011b) was quite high in comparison to other networks, with most of the ERMN region receiving between 15 and 20 kg S/ha/yr, and many areas receiving more than 30 kg S/ha/yr. Only a small portion of the network region (in the southeast) had estimated total S deposition below 10 kg S/ha/yr. Total N deposition within the region ranged from as low as 5 to 10 kg N/ha/yr to as high as 15 to 20 kg N/ha/yr. Almost the entire region received in excess of 10 kg N/ha/yr of total N deposition in 2002 (Sullivan et al. 2011b).



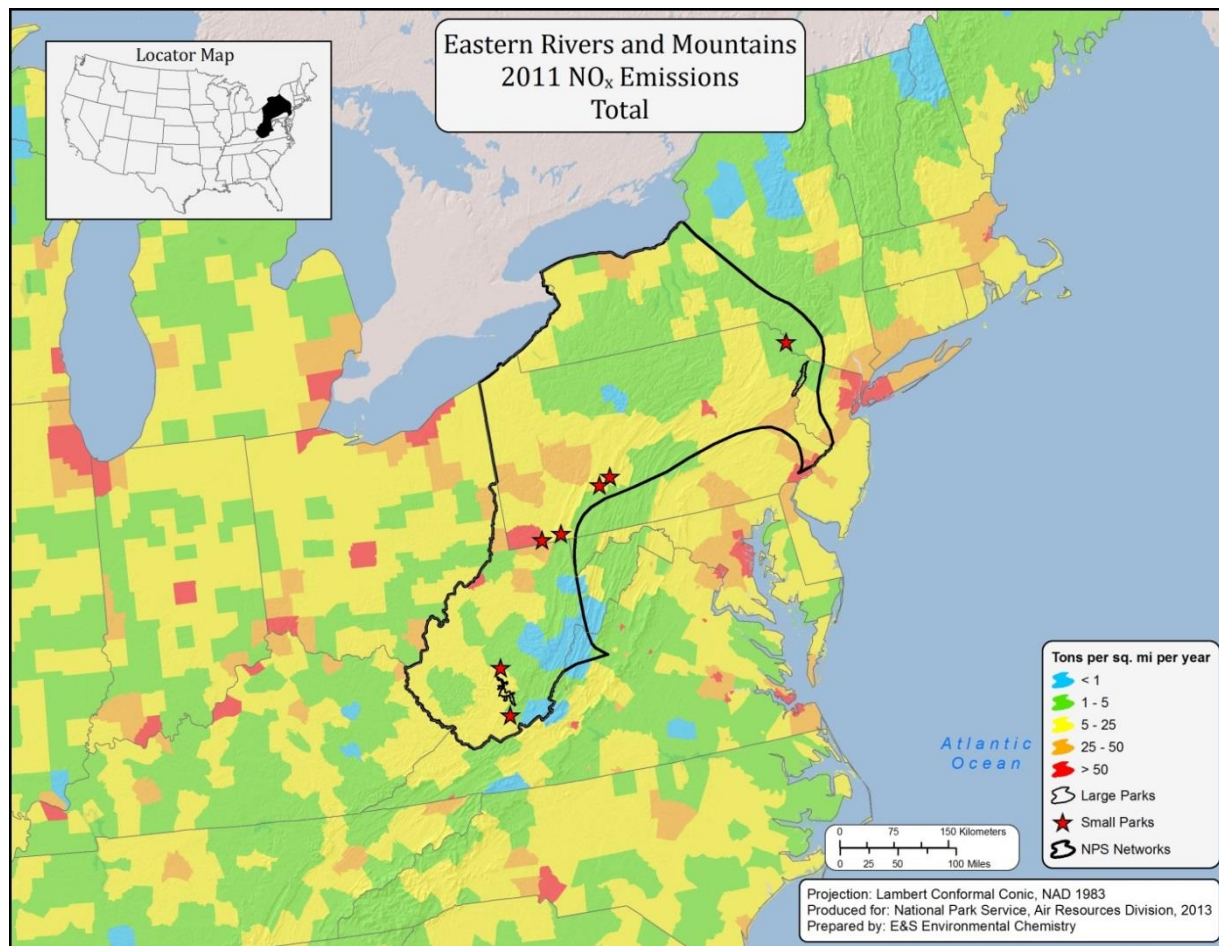
Map 1. Network boundary and locations of park units and population centers greater than 10,000 people within the ERMN region.



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

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 sulfur (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), N₂O₅, NO, NO₂, HONO, and organic NO₃. The total deposition estimates are considered to be dynamic, with updates planned as new information becomes available. The TDEP data reported here were developed in late 2013 and are designated version 2013.02.

Atmospheric S, and to a lesser extent N, deposition levels have declined at all ERMN parks except ALPO since 2001, based on TDEP estimates (Table 1). Decreases in total S deposition over the previous decade for the parks in this network were commonly in the range of 30% to 40%. Estimated total N deposition decreased over that same time period at all parks in the network, in most cases by

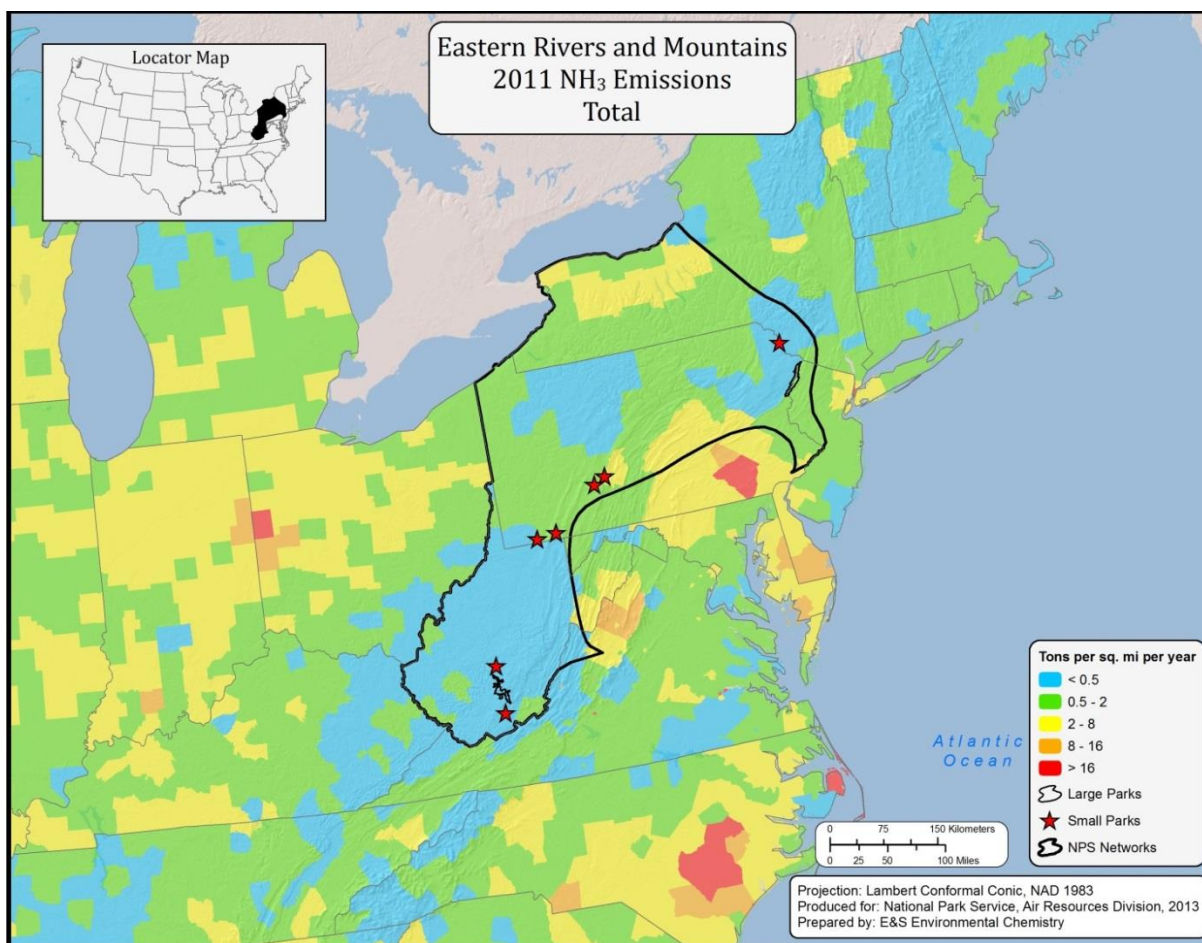


Map 3. Total NO_x emissions, by county, near the ERMN for the year 2011. Data from EPA's National Emissions Inventory.

more than 20%. Decreases in reduced N deposition were small (< 10%), whereas decreases in NO_x were typically larger than 25%.

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Total S deposition in and around the ERMN for the period 2010-2012 was generally highest (> 15 kg S/ha/yr) in the mid-section of the network area and lowest (5 to 10 kg S/ha/yr) in the northeast part of the network area (Map 5). The TDEP output suggested that most parks in the network received more than 10 kg S/ha/yr. Oxidized inorganic N deposition for the period 2010-2012 was in the range of 5-10 kg N/ha/yr throughout the park lands within the ERMN (Map 6). All park lands received less than 5 kg N/ha/yr of reduced inorganic N from atmospheric deposition during this same period (Map 7).



Map 4. Total NH_3 emissions, by county, near the ERMN for the year 2011. Data from EPA's National Emissions Inventory.

Total N deposition was 5-10 kg N/ha/yr at some park locations and 10-15 kg N/ha/yr at others (Map 8).

Risch et al. (2012) reported wet mercury (Hg) deposition throughout the Great Lakes region during the period 2002-2008, based on data from three Hg and precipitation monitoring networks. Areas in eastern Pennsylvania (near ERMN parks), as well as in southern Indiana, Illinois, and central Michigan, had highest estimated wet Hg deposition, ranging from about 10 to 14 $\mu\text{g}/\text{m}^2$. Such high Hg wet deposition levels were caused by high Hg concentration in precipitation ($> 12 \text{ ng}/\text{L}$) and/or high precipitation amount (120-140 cm/yr). There was not a change in deposition observed over the course of the monitoring period; any small decrease that may have occurred in Hg concentration in precipitation was apparently offset by increased precipitation amount (Risch et al. 2012).

NPS (2010) reported long-term trends in concentrations of Hg in wet deposition through 2008 for nine national parks. Statistically significant ($p \leq 0.05$) decreases in wet Hg deposition were reported for only two parks, including ALPO ($-0.17 \text{ ng}/\text{L}/\text{yr}$; 12 years of data) in the ERMN. None of the other parks showed significant trends.

Acidification

The network rankings for acid Pollutant Exposure, Ecosystem Sensitivity to acidification, and Park Protection developed by Sullivan et al. (2011b) yielded an overall network acidification Summary Risk ranking for the ERMN that was relatively high compared with other networks. The geology and surface waters within the ERMN region are known to be acid-sensitive in some areas, and there are acid-sensitive tree species present. While rankings are an indication of risk, park-specific data, particularly data on ecosystem sensitivity, are needed to fully evaluate risk from acidification.

The overall level of concern for acidification effects on Inventory and Monitoring (I&M) parks within this network was considered by Sullivan et al. (2011b) to be High. The two I&M parks in the ERMN that are larger than 100 square miles, DEWA and NERI, were ranked Very High among I&M parks in acid Pollutant Exposure (Sullivan et al. 2011b; Table 2). All of the smaller parks were also ranked Very High for Acid Pollutant Exposure, except BLUE and UPDE, which were ranked High (Table 2). All of the parks in this network ranked Very High in Ecosystem Sensitivity to Acidification except FRHI, which was ranked Moderate for this theme (Table 2). Ecosystem sensitivity to acidification takes into account elevation and land slope, which often influence the degree of acid neutralization provided by soils and bedrock within the watershed. Slopes in the ERMN are steep; the average slope across parklands in this network ranges from 10° to 30°, and there are two parks, NERI and BLUE, that have slopes in the 30° to 40° range. These steep slopes give rise to low-order, relatively high-elevation streams, suggesting increased potential for acid sensitivity of aquatic ecosystems.

Acidification of Terrestrial Ecosystems

Some acid-sensitive forest soils in the United States have been acidified by S and N deposition. Hornbeck (1992) concluded that the contributions of acidic deposition to soil acidification in this region were equal to or greater than the contributions from whole-tree harvesting. Documented effects have mainly been limited to the northeastern United States in the Northeastern Temperate Network and portions of the Appalachian Mountains, including parts of the region around the ERMN. Effects have been most pronounced at relatively high elevation in areas occupied by coniferous forests and on ridgetops having poorly developed soils.

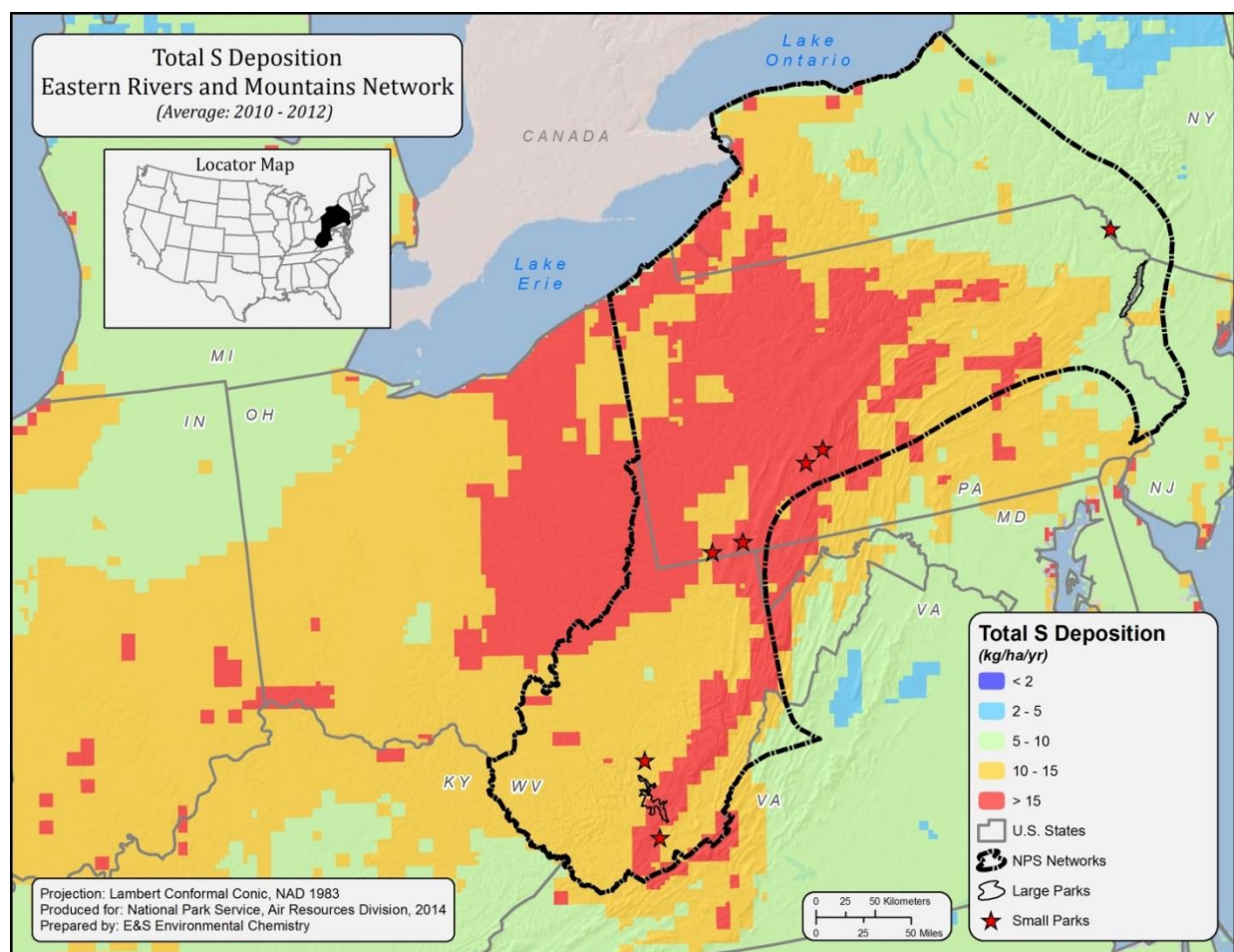
The most important potential effects of acidic deposition on soil are base cation depletion, Al mobilization, and pH depression (Adams et al. 2000, Reuss and Johnson 1986). The more sensitive soils include those that are very acidic and those with low cation exchange capacity, having little ability to buffer acidification that occurs in response to increased leaching of mobile acid anions and nutrient cations. This describes the majority of the Appalachian forest soils derived from acid shales and sandstones that underlie much of the Ridge and Valley, Allegheny, and Cumberland Plateau physiographic provinces (Adams et al. 2000).

Table 1. Average changes in S and N deposition between 2001 and 2011 across park grid cells at ERMN 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.

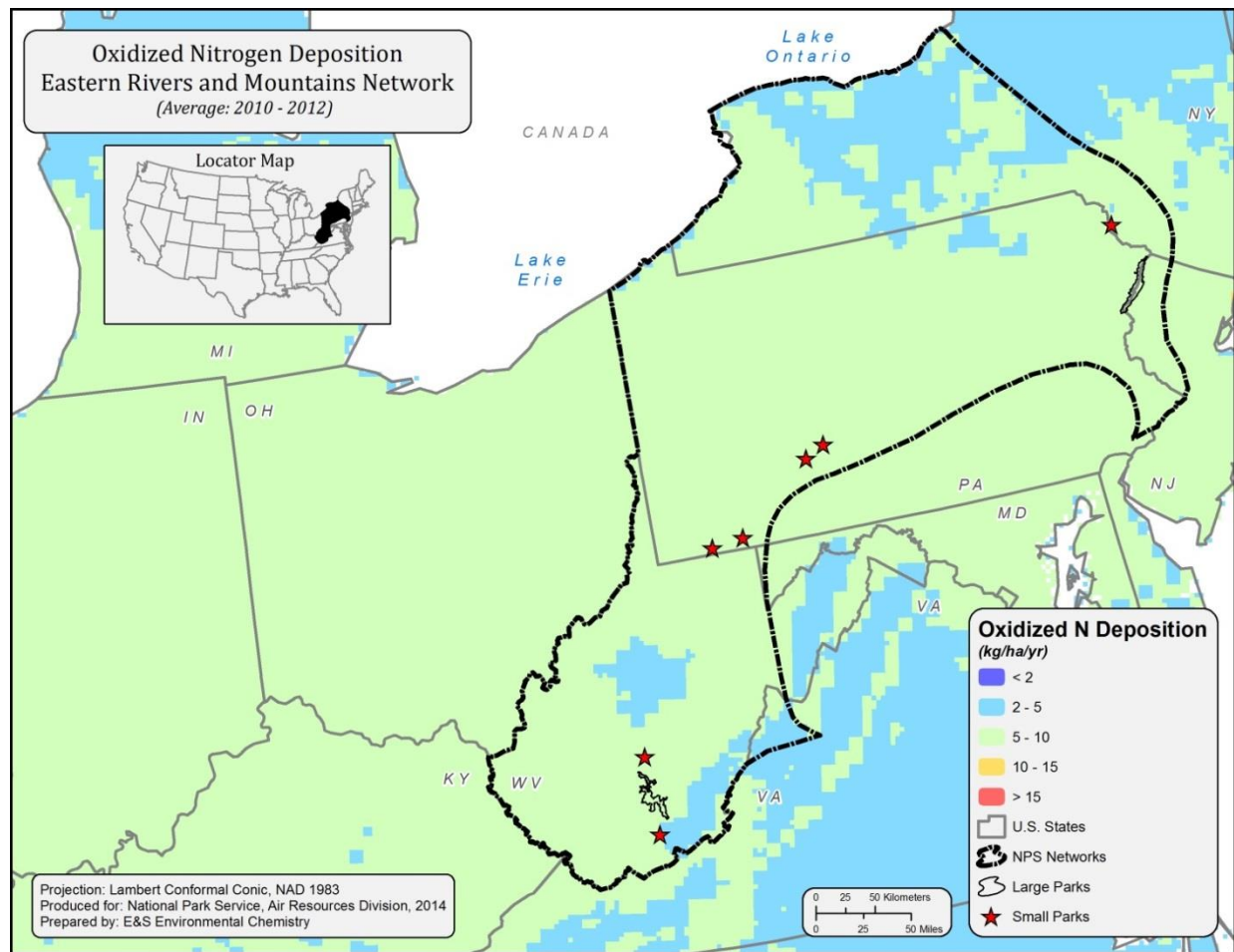
Park Code	Park Name	Parameter	2001 Average (kg/ha/yr)	2011 Average (kg/ha/yr)	Absolute Change (kg/ha/yr)	Percent Change	2011 Minimum (kg/ha/yr)	2011 Maximum (kg/ha/yr)	2011 Range (kg/ha/yr)
ALPO	Allegheny Portage Railroad	Total S	27.02	28.61	1.70	7.6%	20.84	31.41	10.58
		Total N	14.38	11.64	-2.73	-18.9%	10.60	12.16	1.56
		Oxidized N	9.87	7.44	-2.42	-24.4%	6.68	7.82	1.14
		Reduced N	4.51	4.20	-0.31	-6.9%	3.71	4.34	0.63
BLUE	Bluestone	Total S	19.05	15.36	-3.69	-19.4%	14.11	16.67	2.56
		Total N	11.23	7.99	-3.24	-28.9%	7.44	8.55	1.12
		Oxidized N	8.47	5.29	-3.18	-37.6%	4.90	5.69	0.79
		Reduced N	2.76	2.70	-0.06	-2.0%	2.54	2.86	0.33
DEWA	Delaware Water Gap	Total S	14.68	8.87	-5.82	-39.5%	8.07	11.60	3.52
		Total N	14.39	9.91	-4.48	-31.0%	9.52	10.65	1.13
		Oxidized N	11.10	6.49	-4.61	-41.4%	6.30	7.03	0.74
		Reduced N	3.29	3.42	0.13	4.1%	3.19	3.96	0.76
FONE	Fort Necessity	Total S	36.87	20.45	-16.42	-44.5%	18.76	20.52	1.76
		Total N	13.58	10.71	-2.88	-21.2%	10.45	10.83	0.38
		Oxidized N	9.67	6.86	-2.80	-29.0%	6.75	6.96	0.21
		Reduced N	3.92	3.85	-0.07	-1.8%	3.70	3.87	0.17
FRHI	Friendship Hill	Total S	31.35	18.10	-13.25	-42.3%	14.90	18.77	3.87
		Total N	12.33	8.54	-3.79	-30.7%	8.52	8.62	0.10
		Oxidized N	9.18	5.61	-3.57	-38.9%	5.60	5.63	0.03
		Reduced N	3.15	2.93	-0.22	-6.9%	2.92	2.99	0.07
GARI	Gauley River	Total S	20.83	13.20	-7.63	-36.7%	12.21	13.49	1.29
		Total N	11.67	8.22	-3.45	-29.6%	8.03	8.46	0.43
		Oxidized N	8.54	5.37	-3.17	-37.1%	5.16	5.45	0.29
		Reduced N	3.13	2.85	-0.28	-9.0%	2.75	3.02	0.28

Table 1 (continued). Average changes in S and N deposition between 2001 and 2011 across park grid cells at ERMN 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.

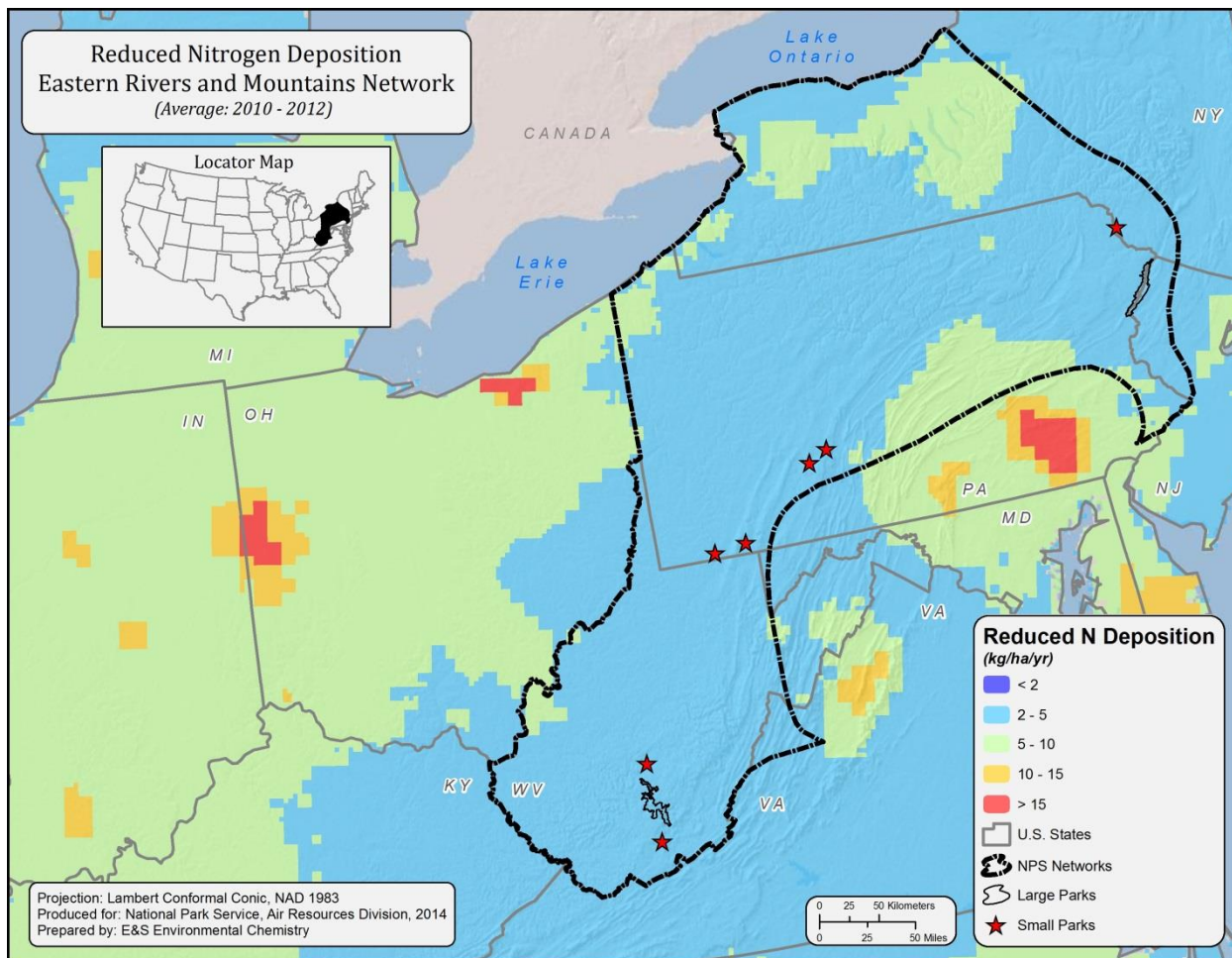
Park Code	Park Name	Parameter	2001 Average (kg/ha/yr)	2011 Average (kg/ha/yr)	Absolute Change (kg/ha/yr)	Percent Change	2011 Minimum (kg/ha/yr)	2011 Maximum (kg/ha/yr)	2011 Range (kg/ha/yr)
JOFL	Johnstown Flood	Total S	32.97	25.47	-7.51	-22.8%	25.47	25.47	0.00
		Total N	14.63	10.62	-4.01	-27.4%	10.62	10.62	0.00
		Oxidized N	10.47	6.81	-3.67	-35.0%	6.81	6.81	0.00
		Reduced N	4.15	3.81	-0.34	-8.2%	3.81	3.81	0.00
NERI	New River Gorge	Total S	20.24	15.23	-5.01	-24.5%	13.06	17.38	4.33
		Total N	11.85	8.40	-3.45	-29.1%	7.67	9.11	1.44
		Oxidized N	8.91	5.51	-3.40	-38.1%	5.07	5.93	0.86
		Reduced N	2.94	2.89	-0.05	-1.7%	2.60	3.18	0.59
UPDE	Upper Delaware	Total S	13.82	7.31	-6.52	-46.9%	6.41	8.62	2.21
		Total N	12.74	8.67	-4.08	-32.0%	8.02	9.87	1.85
		Oxidized N	9.66	5.69	-3.97	-41.2%	5.08	6.54	1.46
		Reduced N	3.08	2.98	-0.11	-3.3%	2.74	3.33	0.59



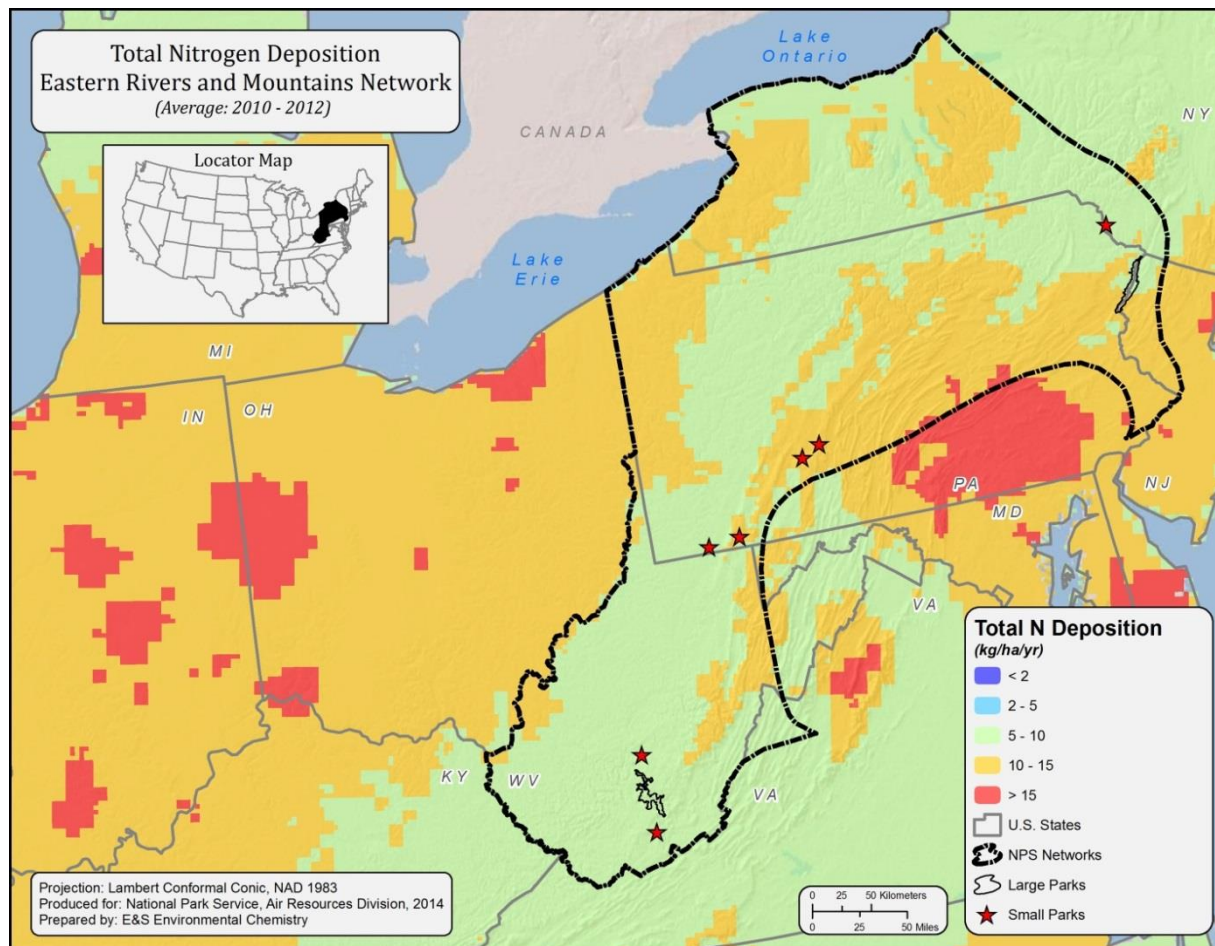
Map 5. Total S deposition for the three-year period centered on 2011 in and around the ERMN. (Source: Schwede and Lear 2014)



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



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



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

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
Allegheny Portage Railroad NHS	ALPO	Very High	Very High
Bluestone NSR	BLUE	High	Very High
Delaware Water Gap NRA	DEWA	Very High	Very High
Fort Necessity NB	FONE	Very High	Very High
Friendship Hill NHS	FRHI	Very High	Moderate
Gauley River NRA	GARI	Very High	Very High
Johnstown Flood NMem	JOFL	Very High	Very High
New River Gorge NR	NERI	Very High	Very High
Upper Delaware SRR	UPDE	High	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.

Decreases in the concentrations of exchangeable base cations in the Oa and B soil horizons over the past several decades have been common and widespread in the eastern United States, including

within the ERMN region. Acidic deposition has been recognized as an important factor causing the observed decreases in concentrations of exchangeable base cations on soils. Base saturation values less than 10% predominate in the B horizon in areas where soil and surface water acidification from acidic deposition have been most pronounced, including hardwood forests on the Allegheny Plateau (Bailey et al. 2004). A long-term decrease in soil pH since 1960 (0.78 pH unit decrease in the O horizon, and 0.23 pH unit decrease in the A horizon) in Pennsylvania hardwood forests has been documented, along with decreases in soil Ca^{2+} and Mg^{2+} (Drohan and Sharpe 1997). To some degree, soils can recover their base cation reserves over time in response to reduced future levels of acidic deposition. However, the recovery potential of soil exchangeable base cation concentrations is dependent on weathering supply, and weathering occurs very slowly.

The ERMN began monitoring soils in 2007 and collected samples at 348 plots throughout the nine ERMN parks. Air pollution, specifically acid deposition of S and N, is considered one of the primary threats to soil health in the parks, potentially causing acidification, disruption of nutrient cycling, and N saturation. In general, forests on well-drained soils over acidic bedrock on upper slopes and ridgetops are more at risk from acid deposition because the soils have depleted cation nutrients, less buffering capacity, and higher concentrations of aluminum (Al) that can be toxic to plants. Forests on lower slopes, more calcareous bedrock, and poor soil drainage are less at risk since these forests tend to have richer soils with more available cation nutrients, more buffering capacity, and lower non-toxic concentrations of Al. Perles et al. (2014) analyzed soils collected from ERMN parks in 2007-2010. They found that, in general, soils in ALPO and JOFL had higher levels of acidity, Al, Fe, C, and N; and lower base saturation, pH, and Ca:Al ratio compared with the other network parks. In contrast, soils in BLUE and FRHI tended to have higher base saturation, Ca:Al ratio, pH, Ca, and Mg; and lower acidity, Fe, and Al. Soil properties in NERI, GARI, FONE, and DEWA tended to have intermediate values. Base saturation is a particularly good indicator of acidification risk. As base saturation decreases below 20%, strong acids cause Al to leach into soil water and convert to forms toxic to terrestrial and aquatic life (Cronan and Schofield 1990). One third of the plots in DEWA and FONE and one half of JOFL plots had base saturation less than 20%. Based on these results, large portions of those parks may be poorly buffered against acidic deposition. About 20% of the plots in GARI and NERI had base saturation less than 20%. In contrast, only 5% of plots in BLUE and FRHI had base saturation less than 20% (Perles et al. 2014).

Biological effects of terrestrial acidification by S and N deposition in the region around the ERMN have included reduced growth and increased stress to overstory trees, and likely changes in the species distributions of understory plants. Included among the direct responses of forest trees to acidic deposition are increased leaching of nutrients from foliage; accelerated weathering of leaf cuticular surfaces; increased permeability of leaf surfaces to toxic materials, water, and disease agents; and altered reproductive processes (Cape 1993, U.S. EPA, 2004).

The main effects of acidic deposition on vegetation, however, are indirect effects which occur through the soil (National Science and Technology Council 1998). Acidic deposition, including both N and S, has contributed to a decline in the availability of Ca^{2+} and other base cations in the soils of acid-sensitive forest ecosystems in the region around the ERMN by the leaching of base cations from

foliage and from the primary rooting zone, and by the mobilization of Al from soils to soil solution and drainage water (Eagar and Adams 1992, NAPAP, 1998). Calcium must be dissolved in soil water in order for it to be taken up by roots. Aluminum mobilization from acidic soils can impede Ca^{2+} and Mg^{2+} uptake by competing for binding sites in the cortex of fine roots; this can potentially induce deficiencies in these nutrients and adversely affect tree health (Garner 1994, Shortle and Smith 1988). A reduction in Ca^{2+} uptake suppresses cambial growth, reduces the rate of wood formation, decreases the amount of functional sapwood and live crown, and predisposes trees to disease and injury, especially when the functional sapwood becomes less than 25% of the cross-sectional stem area (Smith 1990). Foliar Ca^{2+} levels and soil and root Ca:Al ratios are considered low to deficient over large portions of the spruce-fir region in the eastern United States (Cronan and Grigal 1995, Joslin et al. 1992, NAPAP, 1998). Declining sugar maples were shown to be deficient in foliar Ca^{2+} and Mg^{2+} . Decreased Mg^{2+} supply has been associated with crown defoliation and discoloration in Norway spruce (*Picea abies*; Drohan and Sharpe 1997).

Several studies in the United States, mainly in Pennsylvania (near ERMN parks), have demonstrated that sugar maple decline is linked to the occurrence of relatively high levels of acidic deposition and the presence of base-poor soils. In a study of sugar maple throughout the Northeast, Bailey et al. (2004) found threshold relationships between base cation availability in the upper B soil horizon and sugar maple mortality at Ca^{2+} saturation less than 2% and Mg^{2+} saturation less than 0.5%. Horsley et al. (1999) found that dieback at 19 sites in northwestern and northcentral Pennsylvania and southwestern New York was correlated with combined stress from defoliation and soil deficiencies of Mg^{2+} and Ca^{2+} . Dieback occurred predominately on ridgetops and on upper slopes, where soil base availability was much lower than at middle and lower slopes (Bailey et al. 1999).

Drohan et al. (2002) investigated differences in soil conditions in declining versus non-declining sugar maple plots in northern Pennsylvania that were part of the USDA Forest Service's Forest Inventory and Analysis (FIA) program. Soils in plots with declining sugar maple tended to have lower base cation concentrations and pH, and $\text{Ca}:\text{Al} \leq 1$. Regressions between foliar and soil chemistry showed that foliar nutrition was highly correlated with the chemistry of the upper 50 cm of soil. In general, declining sugar maple trees were found at higher elevation, on geologies dominated by sandstone, with base-poor sandy soils having high percentages of rock fragments (Drohan et al. 2002). Such site conditions are representative of the kinds of conditions expected to be most susceptible to adverse impacts of acidic deposition on plants because of probable low initial base cation pools and high base cation leaching losses. More recent research has strengthened understanding of the importance of soil cation nutrition to sugar maple health (Hallett et al. 2006). Because multiple factors, including soil mineralogy and landscape position, affect soil base status, the extent to which sugar maple dieback can be attributed to acidic deposition is not fully known.

Acidification of Aquatic Ecosystems

The larger region encompassing the ERMN, including the Appalachian Plateau in West Virginia and the Allegheny Plateau in Pennsylvania, contains large numbers of low-acid neutralizing capacity (ANC) streams influenced by both acidic deposition and acid mine drainage. Between 2010 and 2011, 53 headwater stream reaches in ERMN parks were sampled to assess stream condition

(Webber 2012). Measurements of major cations and anions, nutrients, organics, and trace metals were performed, in addition to measurements of indicators of acidity including pH, ANC, and SO_4^{2-} . Several park streams did not meet either EPA or state acidity standards, based on pH. In Western Pennsylvania, Ice Pond Run in FRHI had a pH value less than 3.0 during every sampling occasion as a result of its documented history of acid mine drainage. In West Virginia, Dowdy Creek, Richlick Creek, and Little Laurel Creek in NERI did not meet pH standards during one or more sampling period(s). Dowdy Creek also had an ANC less than 50 $\mu\text{eq/L}$. These streams had elevated levels of SO_4^{2-} or metals associated with acid mine drainage (Webber 2012). Although acid mine drainage appears responsible for acidity in many ERMN streams, a history of high levels of acidic deposition in the region likely also contributes to stream acidity.

Monitoring data collected during the 1990s in the LTM and TIME projects illustrated that most regions sampled in the monitoring efforts, including the northern Appalachian Mountains, showed large declines in SO_4^{2-} concentrations in surface waters over the 10 years of monitoring, with rates of change ranging from -1.5 to -3 $\mu\text{eq/L/yr}$. These declines in lake and stream SO_4^{2-} concentration were considered consistent with observed declines in S wet deposition. Surface water nitrate (NO_3^-) concentrations also decreased, but only in the two regions that had the highest ambient surface water NO_3^- concentrations (including the northern Appalachian Plateau).

Population estimates from TIME surveys of streams in the northern Appalachian Plateau region (Stoddard et al. 2003) suggested that 5,014 km of streams (11.8% of the total stream length) were acidic in 1993-94, but that only 3,600 km of streams (8.5% of the total stream length) remained acidic in this region in 2000. This represented about a 28% decrease in acidic stream length over the preceding decade. The approximate rate of estimated change in ANC in the region was +0.79 $\mu\text{eq/L/yr}$.

Critical loads (CL) of S deposition were modeled by McDonnell et al. (2014) in the Ecosystem Management Decision Support (EMDS) regional modeling project. Two national parks in the ERMN (NERI and GARI) were included in this effort. The land slope was partitioned into small watersheds, generally about 1 km^2 , and the steady-state CL to maintain stream ANC = 50 $\mu\text{eq/L}$ was calculated for each. This level of ANC is expected to protect most aquatic biota from adverse effects from water acidification. The CL was based on Model of Acidification of Groundwater in Catchments (MAGIC) estimates of base cation weathering used as input to the Steady State Water Chemistry (SSWC) model of CL. Results for NERI and GARI are shown in Map 9 and Table 3.

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 CL estimates for 15 sites in DEWA. The CL estimates were high at all modeled sites (> 812 eq S/ha/yr).

Modeled aquatic CL for protecting stream ANC to 50 µeq/L in NERI were mostly > 15 kg S/ha/yr along the New River, but were substantially lower (5-10 kg S/ha/yr) in many places along the tributary streams. A few watersheds showed modeled aquatic CL less than 5 kg S/ha/yr in this park.

In most of GARI, modeled aquatic CL for protecting stream ANC to 50 µeq/L were in the range of 5-10 kg S/ha/yr. Atmospheric S deposition is likely in exceedance of these CL values. The northeastern portion of this park showed modeled aquatic CL between 10-15 kg S/ha/yr, and there were a few small watersheds with CL estimates more than 15 kg S/ha/yr.

Table 3. Distribution of steady-state critical loads of sulfur (in kg/ha/yr) to protect stream ANC = 50 µeq/L for parks modeled in the EMDS project within the ERMN (McDonnell et al. 2014).

Park Name	Park Code ¹	# of CLs ²	Minimum	25th Percentile	Median	25th Percentile	Maximum
Bluestone NSR	BLUE	43	11	High	High	High	High
Fort Necessity NB	FONE	3	High	High	High	High	High
Gauley River NRA	GARI	106	4	8	9	16	High
<i>New River Gorge NR</i>	NERI	426	2	7	8	High	High

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

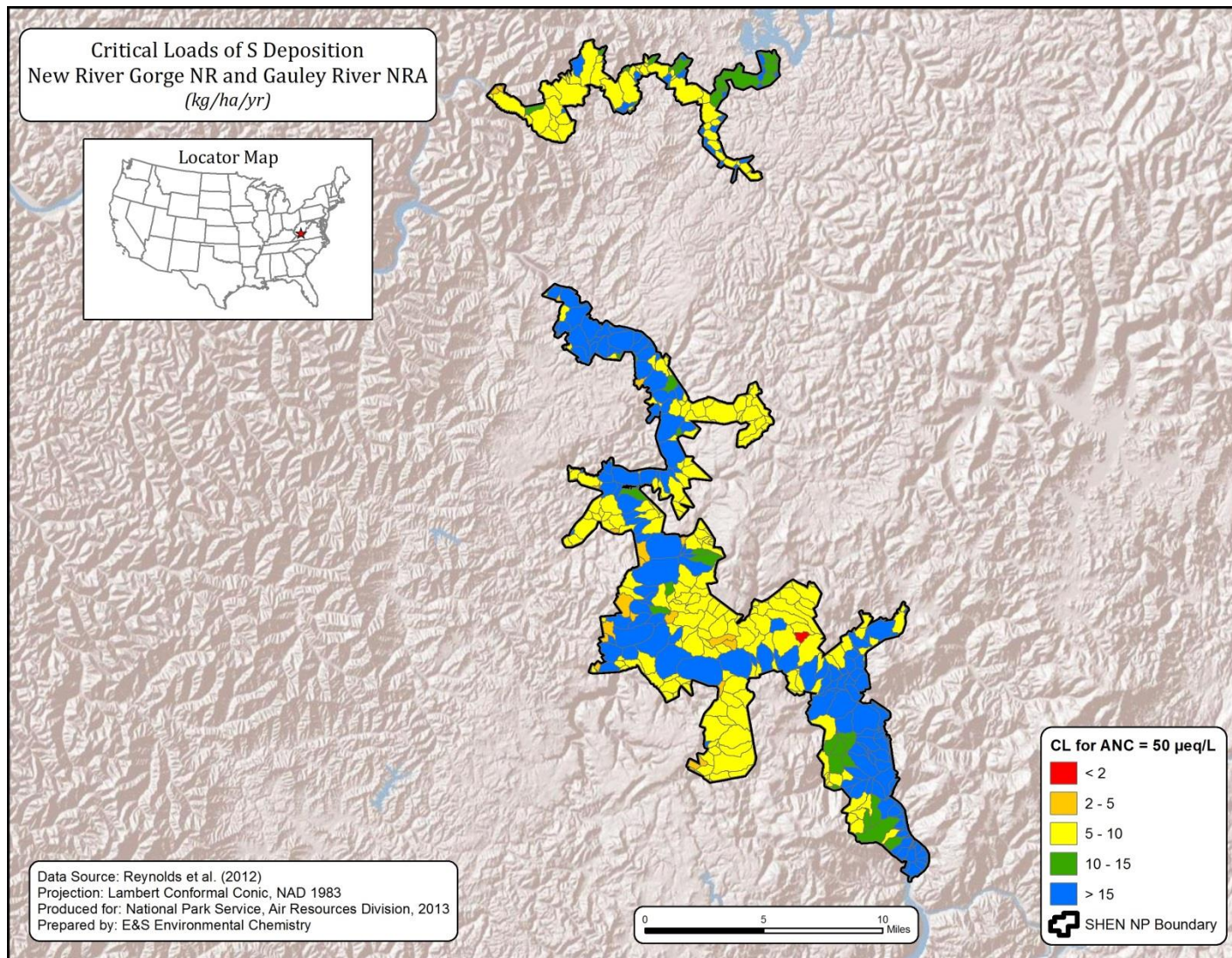
² Number of small watersheds (generally approximately 1 km²), for which CL were calculated by McDonnell et al. (2014), that are wholly or partly within the park

"High" signifies that the CL is significantly higher than ambient deposition

CLs of S in units of kg/ha/yr can be converted to meq/m²/yr by multiplying by 6.25

Temporal Variability in Water Chemistry

Episodic acidification of streams can result naturally from the mobilization of organic acids and from dilution of base cation concentrations, but decreases in pH and ANC associated with increases in SO₄²⁻ and NO₃⁻ are largely attributable to acidic deposition (Wigington et al. 1996). In the Northeast, including areas around the ERMN, the most severe acidification of surface waters generally occurs during spring snowmelt and rainstorm events (Charles 1991). Episodic acidification events are most common in the early spring and late fall and least common in summer, when high water flows tend to be infrequent. Seasonal variation in stream flow also results in seasonal patterns of surface water chemistry at base flow. Lakes and streams at base flow tend to be more acidic in early spring than at other times of the year, and least acidic in late summer (Lawrence et al. 2007).



Map 9. Steady-state critical load of atmospheric S deposition for maintaining streamwater ANC at or above 50 µeq/L in NERI and GARI. Estimates were generated within the Ecosystems Management Decision Support modeling effort of McDonnell et al. (2014).

Using data from the TIME surveys, Stoddard et al. (2003) found that, on average, spring season ANC values during the period 1990 to 2000 in several northeastern regions, including the northern Appalachian Plateau, were about 30 $\mu\text{eq/L}$ lower than summer values. This implied that lakes and streams in these regions would need to recover to chronic ANC values above about 30 $\mu\text{eq/L}$ before they could be expected to not experience acidic conditions during spring (Stoddard et al. 2003). However, the estimate of 30 $\mu\text{eq/L}$ is certain to be low because the comparison did not specifically target episodic sampling of streams during spring snowmelt.

The transient nature of high flows makes episodic acidification difficult to measure. Therefore, assessments have generally estimated the number of lakes and streams prone to episodic acidification by combining episode information from a few sites with base flow values of ANC determined in large surveys (Bulger et al. 2000, Driscoll et al. 2001, Eshleman et al. 1995). Inclusion of episodically acidified water bodies in regional assessments substantially increases estimates of the extent of surface water acidification.

A thorough characterization of episodic variations in stream chemistry was conducted through the U.S. EPA's Episodic Response Project (ERP), in which 13 low-order streams with watershed areas less than 24 km^2 in the Adirondack and Catskill regions of New York and the Appalachian Plateau in Pennsylvania (including the area around the ERMN) were monitored from 1988 to 1990 (Wigington et al. 1996). Acid episodes involved decreases in ANC of up to 200 $\mu\text{eq/L}$, decreases in pH of up to one unit, and increases in concentrations of Al_i of up to 15 μM (Wigington et al. 1996).

Stream acid-base chemistry in the ERMN can change in response to disturbances other than acidic deposition. For example, defoliation by elm spanworm (*Ennomos subsignarius*) larvae in old-growth hemlock-hardwood forests on the Allegheny High Plateau of northwestern Pennsylvania increased streamwater NO_3^- concentrations from pre-defoliation levels of about 29 $\mu\text{eq/L}$ to peak values of about 100 $\mu\text{eq/L}$ during the summer after defoliation (Lewis and Likens 2007).

Effects on Aquatic Biota

Much of the available *in-situ* long-term monitoring and dose-response data with which to evaluate biological responses to surface water acidification have been generated from studies of streams in Virginia and Pennsylvania, within or near the ERMN, and streams and lakes in New York. There is a general scarcity or absence of biological long-term monitoring studies in most other acid-impacted regions, resulting in limited data with which to evaluate acidification dose-response relationships.

Results of research in ERP (Baker et al. 1996) showed that acid episodes reduced the size of fish populations and eliminated acid-sensitive species when median high-flow pH was less than 5.2 and inorganic monomeric Al (Al_i) concentrations exceeded 3.7 μM . Baker et al. (1996) concluded that impacts on biota from episodic acidification were likely to be similar to those from chronic acidification. When an acid episode is synchronous with a sensitive life stage in fish, it can result in the elimination of an annual age class. Surface water acidification, both chronic and episodic, can result in the loss of acid-sensitive species. As there are more species lost with more pronounced acidification, the overall species pool can decrease as surface water acidity increases. Thus, acidification can result in decreasing species richness in streams and lakes. Largely on the basis of

the results of Baker et al. (1996), the EPA concluded that reversal of effects from episodic acidification could be used as a key ecological endpoint for an acid deposition standard for protection of the aquatic environment (U.S. EPA 1995).

The ERMN monitors stream benthic macroinvertebrates in all parks except UPDE to determine status and trends of benthic biota and habitat condition. Data are used to calculate macroinvertebrate biotic index of integrity (MBII) scores for each stream. Webber (2012) found that the MBII was both accurate and responsive in predicting condition and ecological integrity in ERMN streams. The MBII distinguished streams meeting water quality standards and reference conditions from disturbed, impaired streams.

Nutrient Nitrogen Enrichment

Atmospheric N deposition poses a risk for nutrient enrichment to temperate forest ecosystems. This is partly because the growth of trees in these systems is often N-limited (Vitousek and Howarth 1991), and forest canopy structure can enhance rates of N deposition to terrestrial ecosystems by scavenging dry N deposition from the atmosphere (Lovett 1992). Forests considered especially sensitive to N deposition include red spruce (*Picea rubens*) and sugar maple forests which receive high rates of N deposition, and where effects of N deposition on root allocation or late-season growth may exacerbate other stresses from acidic deposition and harsh climate. The effects of excess N on sensitive forests can be exacerbated in forests that have already experienced base cation depletion from soils due to the acidifying effects of acidic deposition. However, Perles et al. (2014) estimated that the forests that are more at risk for N saturation and subsequent negative consequences in ERMN parks are located on lower slopes, over more calcareous rock, and are more poorly drained. This is opposite from acidification risk that more strongly affects the higher slopes, acidic rock, and well-drained soils.

In some areas, particularly in high-elevation terrestrial ecosystems which have become N-saturated, high levels of N deposition have caused elevated levels of NO_3^- in drainage waters, making these areas potentially more sensitive to additional N deposition (Aber et al. 1989, Aber et al. 1998, Bowman et al. 2006, Campbell et al. 2002, Stoddard 1994). Elevated NO_3^- leaching contributes to depletion of base cations from forest soils, causing adverse effects on sensitive tree species in addition to acidification of drainage waters in base-poor soils (Aber et al. 1989, Aber et al. 1998, Stoddard 1994). In general, sites receiving high N deposition also are subject to other pollutant exposures. Interactions of the nutrient enrichment effects of N deposition with ozone (O_3) exposure and acid rain make it difficult to attribute specific effects to nutrient N alone.

Gilliam (2006) and Gilliam et al. (2006) proposed that high NO_3^- in soil can increase invasibility of sites by exotic plants. They predicted a loss of biodiversity with increasing N availability. Nevertheless, research at the Fernow Experimental Forest in West Virginia, in the ERMN region, indicated that applications of 35 kg N/ha/yr applied as $(\text{NH}_4)_2\text{SO}_4$ for six years had no significant impact on the herbaceous layer in an Appalachian hardwood forest (Gilliam et al. 2006). However, Fernow has been the recipient of high levels of N deposition for decades, raising the possibility that the herbaceous vegetation layer responded long ago to changes in N availability.

May et al. (2005) reported the effects of N fertilization on a mixed hardwood forest at Fernow. Plots were amended with 35 kg N/ha/yr over 11 years. Stem diameter growth, foliar N content, N:P ratios, and nutrient resorption were determined for red maple (*Acer rubrum*), yellow poplar (*Liriodendron tulipifera*), and black cherry (*Prunus serotina*). Tree responses to fertilization were slow and nonlinear, but suggested declining growth and vigor in red spruce and, to a lesser extent, in black cherry and yellow poplar.

Lichens have been heavily impacted by air pollution throughout eastern hardwood forests. High levels of both S and N have played important roles (Showman and Long 1992, Smiley and George 1974). More recently, epiphytic lichen diversity and cover has increased in response to recent decreases in S and N emissions and deposition in south central Pennsylvania (McClenahan et al. 2007). Returning lichen species in Ohio and Pennsylvania have mainly been S-sensitive, but they have also been somewhat N-tolerant (Geiser et al. 2010), likely because N deposition was still relatively high (about 6.2 to 7.0 kg N/ha/yr; Gilliam et al. 2011).

The C:N ratio in soil provides one index of the effects of N deposition on forest ecosystems. It decreased from 30 to 17 over a gradient of increasing atmospheric N deposition from about 5 to 10 kg N/ha/yr in North Carolina and West Virginia, but the ratio did not continue to decrease at higher N deposition levels (Brookshire et al. 2007).

Eastern hardwood forests from Wisconsin to Maine and south to Virginia have been shown to have experienced increased tree growth with increasing N deposition across a depositional gradient from about 3 to 11 kg N/ha/yr (Thomas et al. 2010). Responses were generally weak and were species-specific. Growth increases were most pronounced for red maple, sugar maple, white ash (*Fraxinus americana*), yellow poplar, black cherry, balsam fir (*Abies balsamea*), pignut hickory (*Carya glabra*), eastern white pine (*Pinus strobus*), quaking aspen (*Populus tremuloides*), northern red oak (*Quercus rubra*), and scarlet oak (*Quercus coccinea*). Other tree species showed negative response to N addition. Growth decreased to a statistically significant level with increasing deposition for red pine (*Pinus resinosa*), red spruce, and northern white cedar (*Thuja occidentalis*). Mortality increased with increasing deposition for yellow birch (*Betula alleghaniensis*), eastern white pine, basswood (*Tilia americana*), quaking aspen, bigtooth aspen (*Populus grandidentata*), scarlet oak, chestnut oak (*Quercus prinus*), and northern red oak.

Eastern hardwood forests can respond to increased N deposition with a decrease in biodiversity (Gilliam et al. 2006). The response of the herbaceous layer of the forest can be pronounced, with initial increases in herbaceous plant cover, followed by decrease in species richness and decrease in species evenness. The response time is typically shorter if ambient N deposition is relatively high (Fraterrigo et al. 2009, Gilliam et al. 2006, Gilliam et al. 2011, Royo et al. 2010). Because the plants in the herbaceous layer of eastern hardwood forests tend to have foliage with relatively high nutrient content, herbaceous plants influence N cycling to a level that is disproportionate to their biomass (Gilliam et al. 2011, Moore et al. 2007, Muller 2003, Welch et al. 2007).

Pardo et al. (2011) 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. Available data suggested that the empirical CL for resource protection was consistently less than 12 kg N/ha/yr at all parks in the ERMN for protecting mycorrhizal fungi, lichens, and forests and for preventing NO₃⁻ leaching in drainage water (Table 4). Ambient N deposition at the parks in the ERMN reported by Pardo et al. (2011) was higher than that level at nearly all parks (only at BLUE was the reported deposition slightly lower). In addition, the CL for protecting herbaceous plants was apparently exceeded at the two ERMN parks that are within the Northern Forests ecoregion (DEWA and UPDE). Thus, based on the empirical CL data compiled by Pardo et al. (2011), ambient N deposition may exceed the nutrient CL for a variety of sensitive receptors across this network.

Ozone Injury to Vegetation

The O₃-sensitive plant species that are known or thought to occur within the I&M parks found in the ERMN are listed in Table 5. Those considered to be bioindicators, because they exhibit distinctive symptoms when injured (e.g., dark stipple), are designated by an asterisk. Each of the parks in this network contains a number of O₃-sensitive and bioindicator species.

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 6, along with Kohut's (2007) O₃ risk ranking. The two ranking systems differ. The NPS ranking system (NPS 2010) is a quick assessment of O₃ condition and simply ranks O₃ exposure levels over a 3-month period according to injury thresholds from the literature (Heck and Cowling 1997), using a 5-year average of the W126 or SUM06 index. The W126 was classified as Moderate at values between 7 and 13 parts per million (ppm)-hr, as defined by NPS (2010). Values higher than 13 ppm-hr were classified as High, and values lower than 7 ppm-hr were classified as Low. The SUM06 was classified as Moderate at values between 8 and 15 ppm-hr values. Higher than 15 and lower than 8 ppm-hr were classified as High and Low, respectively, as defined by NPS (2010).

Kohut's approach constitutes a rigorous assessment of potential risk to plants. 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. Ozone condition, as rated by NPS, is Moderate in all ERMN parks. Kohut's evaluation of risk to plants ranges from Moderate to High. In each park, there was an inverse relationship between O₃ exposure and soil moisture; when O₃ concentrations were high, soil moisture tended to be low. This inverse relationship reduces the likelihood of foliar injury when exposures are high.

Table 4. Empirical critical loads for nitrogen in the ERMN, by ecoregion and receptor from Pardo et al. (2011). Ambient N deposition reported by Pardo et al. (2011) 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
Allegheny Portage Railroad NHS	Eastern Temperate Forests	13.8	5 - 12	4 - 8	17.5	3 - 8	8
Bluestone NSR	Eastern Temperate Forests	11.2	5 - 12	4 - 8	17.5	3 - 8	8
Delaware Water Gap NRA	Eastern Temperate Forests	14.4	5 - 12	4 - 8	17.5	3 - 8	8
Delaware Water Gap NRA	Northern Forests	14.4	5 - 7	4 - 6	7 - 21	3 - 26	8
Fort Necessity NB	Eastern Temperate Forests	14.4	5 - 12	4 - 8	17.5	3 - 8	8
Friendship Hill NHS	Eastern Temperate Forests	14.4	5 - 12	4 - 8	17.5	3 - 8	8
Gauley River NRA	Eastern Temperate Forests	12.7	5 - 12	4 - 8	17.5	3 - 8	8
Johnstown Flood NMem	Eastern Temperate Forests	13.8	5 - 12	4 - 8	17.5	3 - 8	8
New River Gorge NR	Eastern Temperate Forests	12.7	5 - 12	4 - 8	17.5	3 - 8	8
Upper Delaware SRR	Eastern Temperate Forests	12.5	5 - 12	4 - 8	17.5	3 - 8	8
Upper Delaware SRR	Northern Forests	12.5	5 - 7	4 - 6	7 - 21	3 - 26	8

Table 5. Ozone-sensitive and bioindicator plant species known or thought to occur in the I&M parks of the ERMN. (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								
		ALPO	BLUE	DEWA	FONE	FRHI	GARI	JOFL	NERI	UPDE
<i>Aesculus flava</i>	Yellow buckeye		x			x				
<i>Aesculus octandra</i>	Yellow buckeye								x	
<i>Ageratina altissima</i> var. <i>altissima</i> *	White snakeroot		x	x		x	x	x		
<i>Ageratina altissima</i> *	White snakeroot	x			x					
<i>Ailanthus altissima</i> *	Tree-of-heaven	x		x	x	x	x		x	
<i>Alnus incana</i> spp. <i>rugosa</i> *	Speckled alder			x	x					
<i>Alnus rugosa</i> *	Speckled alder								x	x
<i>Apios americana</i> *	Groundnut	x	x	x		x	x		x	x
<i>Apocynum androsaemifolium</i> *	Spreading dogbane	x		x	x	x		x	x	x
<i>Apocynum cannabinum</i>	Dogbane, Indian hemp	x	x	x	x		x	x	x	x
<i>Asclepias exaltata</i> *	Tall milkweed			x					x	x
<i>Asclepias incarnata</i>	Swamp milkweed		x	x	x	x	x	x	x	
<i>Asclepias syriaca</i> *	Common milkweed	x	x	x	x	x	x	x	x	x
<i>Aster acuminatus</i> *	Whorled aster								x	x
<i>Aster macrophyllus</i> *	Big-leaf aster								x	
<i>Cercis canadensis</i> var. <i>canadensis</i> *	Redbud		x				x			
<i>Cercis canadensis</i> *	Redbud			x		x			x	
<i>Clematis virginiana</i>	Virgin's bower		x	x	x	x	x		x	x
<i>Corylus americana</i> *	American hazelnut	x	x	x	x	x	x		x	
<i>Eupatorium rugosum</i> *	White snakeroot									x
<i>Eurybia macrophylla</i> *	Big-leaf aster	x	x	x	x					
<i>Fraxinus americana</i> *	White ash	x	x	x	x	x	x	x	x	x
<i>Fraxinus pennsylvanica</i>	Green ash	x	x	x	x		x	x	x	x
<i>Gaylussacia baccata</i> *	Black huckleberry	x	x	x	x	x	x		x	x
<i>Liquidambar styraciflua</i>	Sweetgum			x		x	x		x	
<i>Liriodendron tulipifera</i> *	Yellow-poplar	x	x	x	x	x	x		x	x
<i>Lyonia ligustrina</i> var. <i>ligustrina</i> *	Maleberry						x			
<i>Lyonia ligustrina</i> *	Maleberry			x					x	x
<i>Oclemena acuminata</i> *	Whorled aster	x		x				x		
<i>Parthenocissus quinquefolia</i>	Virginia creeper	x	x	x	x	x	x	x	x	x
<i>Philadelphus coronarius</i>	Sweet mock orange			x						
<i>Pinus banksiana</i>	Jack pine	x								
<i>Pinus pungens</i>	Table-mountain pine		x							
<i>Pinus rigida</i>	Pitch pine	x	x	x		x	x		x	x
<i>Pinus virginiana</i>	Virginia pine	x	x	x			x		x	
<i>Platanus occidentalis</i> *	American sycamore	x	x	x		x	x		x	x
<i>Populus tremuloides</i> *	Quaking aspen	x		x	x			x		x
<i>Prunus serotina</i> var. <i>serotina</i> *	Black cherry		x				x			
<i>Prunus serotina</i> *	Black cherry	x		x	x	x		x	x	x

Table 5 (continued). Ozone-sensitive and bioindicator plant species known or thought to occur in the I&M parks of the ERMN. (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								
		ALPO	BLUE	DEWA	FONE	FRHI	GARI	JOFL	NERI	UPDE
<i>Prunus virginiana</i>	Choke cherry	x		x	x			x	x	x
<i>Rhus copallina</i>	Winged sumac								x	
<i>Rhus copallinum</i>	Winged sumac	x		x			x			
<i>Robinia pseudoacacia</i>	Black locust	x	x	x	x	x	x	x	x	x
<i>Rubus allegheniensis</i> *	Allegheny blackberry	x	x	x	x	x	x	x	x	x
<i>Rubus canadensis</i> *	Thornless blackberry			x					x	
<i>Rudbeckia laciniata</i> var. <i>laciniata</i> *	Cutleaf coneflower		x				x			
<i>Rudbeckia laciniata</i> *	Cutleaf coneflower	x		x		x		x	x	x
<i>Sambucus canadensis</i> *	American elder								x	x
<i>Sambucus nigra</i> spp. <i>canadensis</i> *	American elder	x	x	x	x	x	x	x		
<i>Sambucus racemosa</i> var. <i>racemosa</i> *	Red elderberry	x								
<i>Sambucus racemosa</i> *	Red elderberry			x						x
<i>Sassafras albidum</i>	Sassafras	x	x	x	x	x	x	x	x	x
<i>Solidago altissima</i>	Goldenrod	x	x	x	x	x	x	x	x	
<i>Spartina alterniflora</i>	Smooth cordgrass									x
<i>Symphoricarpos albus</i> *	Common snowberry			x						
<i>Verbesina occidentalis</i> *	Crownbeard		x				x		x	
<i>Vitis labrusca</i> *	Northern fox grape			x						x

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

* Bioindicator species

Table 6. Ozone assessment results for I&M parks in the ERMN 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	
Allegheny Portage Railroad NHS	ALPO	10.63	Moderate	14.12	Moderate	Moderate
Bluestone NSR	BLUE	10.29	Moderate	13.63	Moderate	Moderate
Delaware Water Gap NRA	DEWA	11.10	Moderate	13.62	Moderate	High
Fort Necessity NB	FONE	10.66	Moderate	14.07	Moderate	High
Friendship Hill NHS	FRHI	10.71	Moderate	14.13	Moderate	High
Gauley River NRA	GARI	9.85	Moderate	12.68	Moderate	Moderate
Johnstown Flood NMem	JOFL	10.63	Moderate	14.11	Moderate	Moderate
New River Gorge NR	NERI	10.14	Moderate	13.24	Moderate	Moderate
Upper Delaware SRR	UPDE	11.53	Moderate	11.30	Moderate	High

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.

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 affect plant response to O₃ exposure, but O₃ exposures have likely changed since the time of the assessment (1995-1999). The NPS approach considers more recent O₃ exposure (2005-2009), but not the modifying environmental conditions.

A survey in 2007 at ALPO found light O₃ injury on milkweed and black cherry (Figure 1). No other surveys have been conducted in the ERMN parks.

Documented O₃ exposures in the region around the ERMN have been high, but some recent reductions have occurred. Atmospheric O₃ concentrations at 568 monitoring sites nationwide were generally stable throughout the 1990s, but declined between 2002 and 2007, largely due to reductions in NO_x emissions (U.S. EPA 2008). Although there have been widespread improvements in atmospheric O₃ levels, there were 51 sites classified as nonattainment areas in 2007 because of failure to meet the O₃ National Ambient Air Quality standard (U.S. EPA 2008). The highest O₃ concentrations in the United States in 2007 were located in six states, including Pennsylvania, most of which is in the ERMN region (U.S. EPA 2008). Despite these high exposure levels, one of the sites that showed the greatest reduction in O₃ concentration since the 1990s was near Chambersburg, PA.



Figure 1. Ozone injury to milkweed at ALPO. (Photo courtesy of Dr. Robert Kohut)

Visibility Degradation

Various pollutants make up the haze that causes visibility degradation, including SO_4^{2-} , NO_3^- , elemental carbon, coarse mass, organic mass, sea salt, and soil. Sulfates form in the atmosphere from SO_2 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 human sources, including agriculture, industry, and fires. Soil can enter the atmosphere through both natural processes and human disturbance. Haze measurements are not provided here for any I&M parks within the ERMN because there are no IMPROVE sites in the parks or considered to be representative of the parks. Nevertheless, some information regarding visibility in the ERMN is available from Mid-Atlantic/Northeast Visibility Union (MANE-VU) modeling efforts. MANE-VU was formed by the mid-Atlantic and northeastern states, tribes, and federal agencies to coordinate regional haze planning activities for the region. MANE-VU was formed to encourage a coordinated approach to meeting the requirements of EPA's Regional Haze Rule (RHR) and reduce visibility impairment¹ in major national parks and wilderness areas in the Northeast and Mid-Atlantic region. MANE-VU provides technical assessments and assistance to its members, evaluates linkages to other regional air pollution issues, provides a forum for discussion, and encourages coordinated actions. MANE-VU also facilitates coordination with other regions. MANE-VU's technical support committee assesses the nature of regional haze, the sources that contribute to regional haze, and the technical tools that states will use to develop their programs.

Sulfate is the most important cause of fine particle pollution and visibility impairment across the MANE-VU states. At the Class I sites in this region, SO_4^{2-} accounts for about one-half to two-thirds of total fine particle mass on the 20% haziest days (NESCAUM 2006). On the 20% clearest days, SO_4^{2-} typically accounts for 40% or more of total fine particle mass. MANE-VU is focusing primarily on fine particle SO_4^{2-} abatement in its initial efforts to comply with the RHR (NESCAUM 2006). Controls on other haze-producing constituents, such as organic C and both urban and mobile sources of NO_x , especially during winter, will also be important.

In the MANE-VU region, regional SO_4^{2-} emissions generally control visibility during summer. During winter, visibility depends more on both regional and local SO_4^{2-} sources and also on local meteorological conditions (including temperature inversions). Organic C is, after SO_4^{2-} , the next most important contributor to regional haze within MANE-VU (NESCAUM 2006).

A number of model simulations suggested that emissions from the MANE-VU states accounted for about 25 to 30% of the fine particulate SO_4^{2-} observed in Class I sites within the MANE-VU region. The Midwest Regional Planning Organization (MWRPO) and Visibility Improvement State and Tribal Association of the Southeast (VISTAS) states each accounted for another approximately 15%

¹ *Visibility impairment* means any humanly perceptible change in visibility (light extinction, visual range, contrast, coloration) from that which would have existed under natural conditions.

to 25% of total fine particulate SO_4^{2-} at the Class I areas within the MANE-VU region (NESCAUM 2006).

MANE-VU is using a weight of evidence approach to meeting the requirements of the RHR. Multiple independent methods are used to assess the relative contribution of different emissions sources and source regions to regional haze in the Class I areas within and near the MANE-VU region (NESCAUM 2006). These approaches include application of Eulerian grid-based emissions source models, Lagrangian (air parcel-based) source dispersion models, back trajectory calculations, and analysis of monitoring data.

Within the northeastern United States, visibility improvement during recent years has been most pronounced in the western and southern portions of the MANE-VU region, in the area near the ERMN. The area of greatest visibility improvement has been in proximity to large power plant sources of SO_2 emissions in the Ohio River and Tennessee Valleys (NESCAUM 2006).

Toxic Airborne Contaminants

Research in the vicinity of the ERMN on the potential effects of airborne toxics on sensitive resources has primarily focused on Hg. The northern Great Lakes region, and eastward into northern New York, is especially sensitive to Hg bioaccumulation, due in part to relatively high Hg deposition and in particular due to watershed and lake characteristics that exacerbate Hg transport, methylation, and bioaccumulation (Evers et al. 2011).

Myers et al. (2007) proposed a national map showing the relative sensitivity of aquatic ecosystems to Hg contamination. Areas showing the highest estimated sensitivity were mainly confined to the East Coast (in the vicinity of the ERMN), Gulf Coast, and Upper Midwest, with scattered small areas in other regions.

Within the Great Lakes region, including near the ERMN, Hg is well known as constituting a substantial risk to the health of wildlife. Much of the landscape is Hg-sensitive. The major source of Hg in the watersheds of this region is atmospheric deposition (Mason and Sullivan 1997, Rolfhus et al. 2003, Wiener et al. 2006), most of which originates from human-caused sources (Lamborg et al. 2002, Lorey and Driscoll 1999, Swain et al. 1992).

Some work has been done on other toxic contaminants. Analysis of osprey (*Pandion haliaetus*) eggs from locations along the Delaware River and Bay, including a site just south of DEWA, found a variety of contaminants. Organochlorine pesticides and their metabolites (e.g., dichlorodiphenyltrichloroethane [DDT], dichlorodiphenyldichloroethylene [DDE]), polychlorinated biphenyls (PCBs), other pesticides, polybrominated diphenyl ethers (PBDEs; flame retardants), and Hg were detected in the osprey eggs (Toschik et al. 2005).

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