SCALING UP TO THE LANDSCAPE:
EMPIRICAL MODELING OF ATMOSPHERIC DEPOSITION IN MOUNTAINOUS LANDSCAPES

A final report to the National Park Service – PMIS #75187

Acadia and Great Smoky Mountain National Parks

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Abstract

Atmospheric deposition has long been recognized as an important source of pollutants and nutrients to ecosystems. Understanding how these nutrients and pollutants are deposited across the landscape and identifying regions of “hot” (high) and “cold” (relatively low) deposition, as well as quantifying the total deposition within ecosystems, represent critical environmental and scientific issues. How well can we measure atmospheric deposition across landscapes? - Not very. Although we have national monitoring programs that generate wet and dry deposition estimates for many sites in the US, and these monitoring programs generate crucial data, current estimates of total deposition have large uncertainties, particularly across heterogeneous landscapes such as montane regions. The need for reliable, spatially-explicit estimates of total atmospheric deposition (wet + dry + cloud) is central, not only to air pollution effects researchers, but also for calculation of input-output budgets, and to decision-makers faced with the challenge of assessing the efficacy of policy initiatives related to deposition. This project was designed to fill a critical gap in our ability to model atmospheric deposition to heterogeneous terrains.

We developed an empirical modeling approach that predicts total deposition as a function of landscape features such as elevation, vegetation type, slope and aspect. We measured indices of total deposition to the landscapes of Acadia (121 km^2) and Great Smoky Mountains (2074 km^2) National Parks. Using ~300-400 point measurements and corresponding landscape variables at each park, we constructed a statistical (general linear) model relating the deposition index to landscape variables measured in the field. The deposition indices ranged over an order of

magnitude, and in response to vegetation type and elevation, which together explained ~40% of the variation in deposition. Then, using the independent landscape variables available in GIS datalayers, we created a GIS-relevant statistical model (LANDMod). We applied this model to create park-wide maps of total deposition which were scaled from wet and dry deposition data from the closest national network monitoring stations. The resultant deposition maps showed high spatial heterogeneity and a four- to six-fold variation in “hot-” and “cold-spots” of nitrogen (N) and sulfur (S) deposition ranging from 3 to 31 kg N/ha/y and from 5 to 42 kg S/ha/yr across these park landscapes. Area-weighted deposition was found to be 50-70% greater than NADP (wet deposition) plus CASTNet (dry deposition) monitoring station estimates.

Model verification results suggest that the model slightly overestimates deposition for deciduous and coniferous forests at low elevation, and underestimates deposition for high elevation coniferous forests. The spatially-explicit deposition estimates derived from our LANDMod are a large improvement over what is currently available. The NPS can use our LANDMod to update deposition maps to reflect changes in reference (monitoring station) deposition and/or to reflect changes in vegetation. Future research should be focused on testing the LANDMod in other mountainous environments and refining it to account for (currently) unexplained variation in deposition.

Executive Summary

We created deposition maps of sulfur (S) and nitrogen (N) for Acadia (ACAD) and Great Smoky Mountains (GRSM) National Parks. We scaled-up 300+ point measurements made in each of the parks to the respective park landscapes using a combination of statistical and Geographic Information System (GIS) tools. Our deposition maps use national monitoring station data (National Atmospheric Deposition Program—NADP and Clean Air Status and Trends—CASTNET) as a reference point. The resultant maps show “hotspots” and “coldspots” of deposition that are 5+-fold different (from hot to cold) across the park landscapes. Our analysis suggests that vegetation type (e.g., coniferous vs deciduous) and elevation are the variables, quantifiable within a GIS, that most strongly influence deposition at the 10s to 100s of km scale. Using the year 2000 as an example, we compared area-weighted deposition based on our spatially-explicit model, LANDMod, with estimates from wet and dry deposition monitoring data. By comparison, area-weighted, whole-park nitrogen (N) deposition estimated using LANDMod is much greater than park-wide N deposition estimated by local monitoring station data: we estimate that ACAD received 70% greater, and GRSM 50% greater N deposition than the NADP plus CASTNET estimate.

Validation exercises and explorations were revealing: our model appears to overpredict deposition at low elevation (also regions of low deposition) and it underpredicts deposition at high elevation (also regions of high deposition). Overall, the validation results suggest that our model is conservative (i.e. underpredicts) in regard to actual total deposition.

These deposition maps, and the generalizable approach we developed, are a significant improvement over what currently exists for the parks, and other areas of complex terrain.
Our deposition models can be updated based on new monitoring station data (i.e., reference deposition) or changes to the landscape (e.g., reforestation or forest clearing). These maps can also be used as a tool to evaluate the link between deposition and environmental effects of deposition.

(Note: the content of this report is derived from an article published by Weathers et al. in Ecological Applications, August 2006.)

Acknowledgments

This research was funded by the EPA-NPS PRIMENet Program, the NPS Air Resources Division, and through a grant to the Institute of Ecosystem Studies by the Andrew W. Mellon Foundation. We are tremendously grateful for the support and assistance offered to us by personnel from both ACAD (especially D. Manski, B. Breen, B. Gawley, and K. Anderson) and GRSM (especially J. Renfro) National Parks. We received important advice and assistance from the IES Analytical Lab, and field and lab assistance from K. Schwarz, L. Chambers, D. Lewis, and J. Beeler. We thank F. Vermeylen for statistical advice, and H. Ewing, T. Blett, K. Morris, B. Breen, T. Maniero, and J. Renfro for helpful comments both on drafts of the manuscript that arose from this work. Note that the figures and some of the text in this report are from Weathers et al. 2006. Permission to use this information should be obtained from *Ecological Applications*.

Introduction

Atmospheric deposition is an important source of nutrients and pollutants to ecosystems; in fact for many terrestrial ecosystems it is the primary source of nitrogen (N) and sulfur (S) (e.g., Likens and Bormann 1995; Schlesinger 1997). Accurate measures of total atmospheric deposition are critically important for many reasons, including the calculation of input-output budgets, assessment of biological, ecological and watershed responses to pollutant and nutrient loading, and for linking air emissions with actual deposition to landscapes (e.g., Weathers and Lovett 1998; Driscoll et al. 2001; Holland et al. 2005). In fact, for much of Europe, emission control policies and legislation are closely linked to deposition estimates through estimation of critical loads; the amount of deposition above which deleterious ecosystem effects are likely (see for example, Hettelingh 1995a, 1995b). Despite the need for accurate estimates of atmospheric deposition, especially to highly heterogeneous landscapes (i.e., montane ecosystems), our ability to estimate total deposition is limited.

Objectives

We developed an empirical modeling approach to explore controls on deposition across two heterogeneous landscapes at Acadia National Park (ACAD) and Great Smoky Mountains National Park (GRSM). The specific goals of this work were: (1) To develop a generalizable method whereby both total deposition to heterogeneous terrain and its spatial heterogeneity across a landscape could be more accurately measured; (2) to determine which readily measured field and Geographic Information System (GIS)-derived independent variables control deposition rate in an empirically-based statistical model; (3) to develop landscape-scale maps of nitrogen (N) and sulfur (S) deposition (LANDMod) from dry and wet deposition monitoring data ("reference deposition") and a separate empirically-based statistical model using only independent variables that were available as Geographic Information System (GIS) datalayers. (4) Finally, we examined the behavior of our model by performing verification analyses.

A Deposition Primer

Substances are delivered from the atmosphere to the Earth's surface in three forms: wet deposition (rain and snow), dry deposition (gases and particles) and cloud or fog deposition (also known as occult deposition) (e.g., Lovett 1994). We know much about the contribution of wet deposition to nutrient and pollutant budgets, but relatively little about dry and cloud deposition (e.g., Weathers et al. 2000, Weathers et al. 2006).

Wet Deposition. As a result of intensive, site-specific, long-term monitoring efforts (e.g., Likens and Bormann 1995; Kelly et al. 2002) as well as the existence of national monitoring networks, both wet chemistry and deposition data are readily available from more than 250 monitoring stations in the US (National Atmospheric Deposition Program or NADP: nadp.sws.uiuc.edu and AIRMON: nadp.sws.uiuc.edu/airmon). Wet deposition is the product of precipitation amount and chemistry, so point estimates can also be extended with some accuracy to weather stations where precipitation amount is monitored. Thus temporal trends (since the late 1970s) and spatial distribution of wet deposition is well documented in the US (Lynch et al. 1995, 2000; Driscoll et al. 2001; Grimm and Lynch 2004; NADP 2004, Holland et al. 2005, Weathers et al., 2000, 2006); and in many parts of Canada and Europe (Hedin et al. 1994; Summers 1995; van Leeuwen et al. 1996; Holland et al. 2005).

Dry Deposition. Estimates of dry deposition are far less certain than wet deposition, both because of the paucity of air chemistry monitoring stations and the inadequacy of the models used for estimating dry deposition (Weathers et al. 2000, 2006.). Air chemistry is measured routinely at independent research sites (e.g., Kelly et al. 2002) as well as nationally through the Clean Air Status and Trends (CASTNET; www.epa.gov/castnet) and AIRMoN (nadp.sws.uiuc.edu/airmon) monitoring programs. There are relatively few air chemistry monitoring stations in the US, and most are located in the eastern half of the country. Most estimates of dry particulate and gas deposition are based on measured air concentrations which then are used in dry deposition models (Meyers et al. 1998; Finkelstein et al. 2000; Holland et al. 2005) that consider the multiple resistances, canopy characteristics and meteorological variables that influence deposition (see Lovett 1994). Unfortunately, current dry deposition models are limited by simplifying assumptions, which include homogeneous canopies and flat terrain (Weathers et al. 1995, 2000, 2006). In the montane forested areas that are considered to be most at risk from atmospheric deposition of S and N, the assumptions of the dry deposition models are violated routinely. Dry deposition model estimates are reasonably robust in grasslands and, croplands with flat terrain, but perform more poorly in forests and are largely untested in uneven terrain (Baumgardner et al. 2002; Lavery et al. 2003). It is therefore difficult to extrapolate estimates of wet plus dry deposition beyond the immediate vicinity of dry deposition monitoring stations (Weathers et al., 2006).

Cloud or Fog Deposition. The third form of deposition, cloud or fog, represents an important water and chemical flux primarily in montane and coastal regions. However, cloud chemistry measurements and monitoring programs are few. Cloud chemistry data for the United States have been collected from fewer than 20 mountains (e.g., Lovett et al. 1982; Mohnen and Kadlecek 1989; Weathers et al. 1986, 1988; Vong et al. 1991; Mohnen and Vong 1993; Anderson et al. 1999; Baumgardner et al. 2003) and even fewer Atlantic and Pacific coastal areas (e.g., Weathers et al. 1986, 1988; Kimball et al. 1988; Vong et al. 1991, Weathers 1999). Cloud deposition models suffer from the same limitations as dry deposition models: they rely on cloud chemistry data from single stations and measured or modeled meteorologic and canopy variables and do not account for complex topography or heterogeneous vegetation (e.g., Lovett 1994).

Total Deposition. Total deposition (wet + dry + cloud) estimates for complex terrain are either inadequate or lacking, primarily as a result of the vagaries of measuring dry, cloud and/or snow deposition in complex terrain. An S and N deposition map was created in the early 1990s for a limited area of the northeastern US (e.g., Ollinger et al. 1993). It has been extensively used by the research community to determine site-specific deposition estimates, even though it does not have sufficient resolution to accurately discriminate between locations that are 10s of kilometers apart. This map does not include cloud deposition, and its dry deposition estimates do not take into account elevational effects or variations in land cover, reducing its accuracy in complex terrain. In recent years, other important deposition maps have been published for the United States (Holland et al. 2005) and limited areas in the western US (Fenn et al. 2003; Nanus et al. 2003). Despite these efforts, no empirical model exists for total deposition to heterogeneous terrain. The results of the research project described here represent a critical step toward filling this need.

Methods

General Approach

We conducted our research in Acadia National Park (ACAD) in Maine, and Great Smoky Mountains National Park (GRSM) in Tennessee and North Carolina. Atmospheric deposition of pollutants has been identified as a primary threat to natural resources in both of these parks by the National Park Service. Our general approach was to measure indices of total deposition in the field and convert those measures to unitless "scaling factors" by relating them to a "base-level" deposition. Base level deposition = NADP + CASTNET monitoring data for the Park. Using the scaling factors, we developed two statistical models, one that used independent variables that were measured in the field (e.g., elevation, canopy type, diameter at breast height, canopy cover), which we refer to as **StatMod**, and one that used a subset of these variables that were available as data layers in a GIS (e.g., aspect, elevation, vegetation type). We used the latter, plus the NADP and CASTNET monitoring data, in a GIS to create maps of deposition for landscapes that ranged in areal extent from 200 km^2 (ACAD) to 2000 km^2 (GRSM), which we refer to as **LANDMod**. The StatMod identified which independent variables, out of all those available, control atmospheric deposition in mountainous landscapes, while the LANDMod (GIS-statistical model) allowed us to scale up the deposition estimates to the entire area of the parks using a GIS. We have included in Appendix 1 a figure that outlines the complete procedure from collecting data in the field to creating deposition maps; this is our process for scaling up from point measurements to the landscape.

Wet deposition has been monitored since 1981 at ACAD as part of the NADP (site ME98; nadp.sws.uiuc.edu) and dry deposition has been monitored since 1998 as part of the CASTNET program (site ACA416; www.epa.gov/castnet). The wet and dry monitoring stations are co-located on McFarland Hill at 44.3739 º N and 68.2606 º W.

GRSM's Elkmont NADP wet deposition monitoring station (TN11) began operation in 1980 and is located at 35.6645 º N and 83.5903 º W. Data collection at the Look Rock CASTNET dry deposition monitoring site (GRS420) began in 1998; it is located at 35.6331 º N and 83.9422 º W, 31.8 km from the wet deposition station at Elkmont.

GIS Data Acquisition

Vegetation and elevation (DEM) data layers were acquired for both ACAD and GRSM through the National Parks Service. For each park we used a USGS 1:24,000 DEM with 30m x 30m pixels. For GRSM, a vegetation data layer derived from LandSat Thematic Mapper imagery collected in September 1984 was used (see MacKenzie 1993). For ACAD, a vegetation data layer derived from 1:15,840 color infrared aerial photographs collected 27-28 May 1997 was used (Lubinski et al. 2003). All ACAD GIS data that were not already in the UTM (zone 19) projection with NAD83 datum were converted to that projection and datum. All GRSM GIS data were in the Albers Equal Area projection with NAD83 datum.

Field Methods for Acadia National Park

The deposition index (DI, Fig. 1) for ACAD was sulfur (S) in throughfall (TF). Throughfall S has been shown to be a conservative tracer of wet, dry and cloud deposition for forests of the eastern US. Three hundred two (305) throughfall collectors were located on National Park Service property along a set of 10 ACAD trail system loops that could be readily accessed in a short time period (1-2 days) to minimize the risk of precipitation events interrupting the sample collection. The sampling season was from 7 June 2000 to 18 September 2000, subdivided into three sampling periods of 4 to 6 weeks each. For each of the 305 TF locations, vegetation, elevation, slope, and aspect attributes were extracted using GIS data and compared to the attributes of the entire ACAD study area to ensure that each of these variables was adequately represented.

Field Methods for Great Smoky Mountains National Park

The deposition index for GRSM was the concentration of Pb in the soil organic horizon. (Note: we had intended to use the Pb index at both parks, but could not because extensive areas of ACAD burned in the 1940s resulting in a reduction in organic matter and lead loss from the soil.) Pb has been shown to preserve the pattern of long-term total deposition (see below and Johnson et al. 1982; Weathers et al. 1995, 2000, 2006). We collected Pb samples from 378 forest floor locations in GRSM during 1999 and 2000.

The GRSM park is much larger than ACAD (~2500 vs. ~300 km^2), so sampling was constrained to trails in an ~800-km^2 rectangular swath (22 km wide and 37 km long; oriented diagonally from northwest to southeast; NW corner 35.69 º N and 83.65 º W; SE corner 35.54 º N and

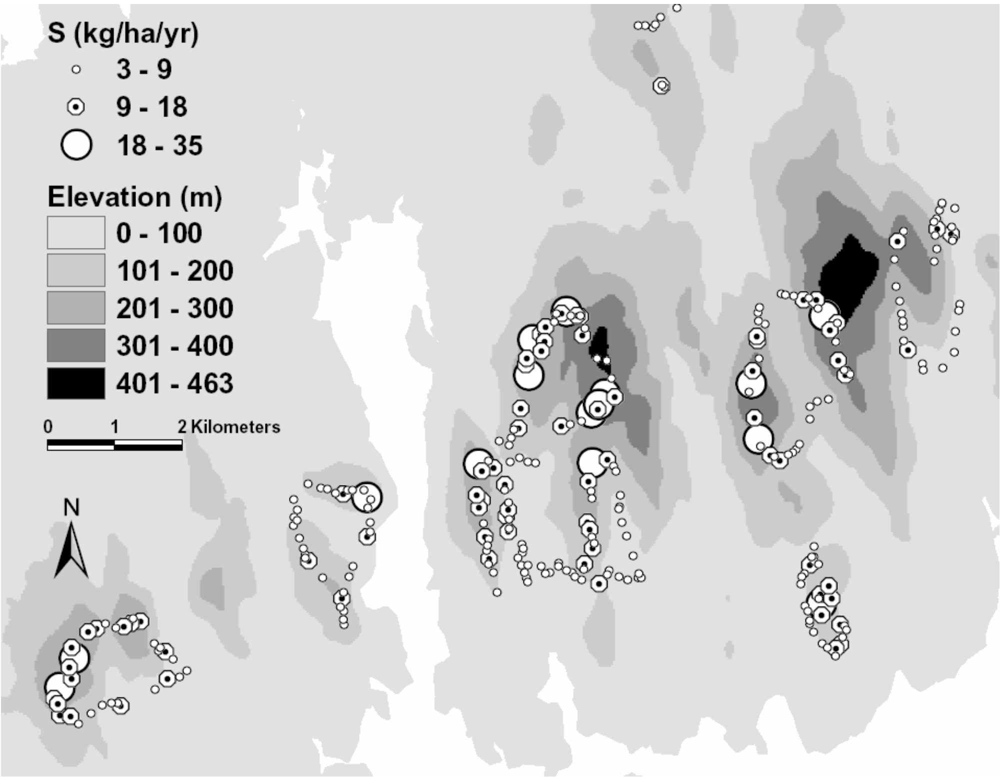
S (kg/ha/yr)
3 - 9
9 - 18
18 - 35
Elevation (m)
0 - 100
101 - 200
201 - 300
301 - 400
401 - 463
0
1
2 Kilometers
N

Figure 1a.

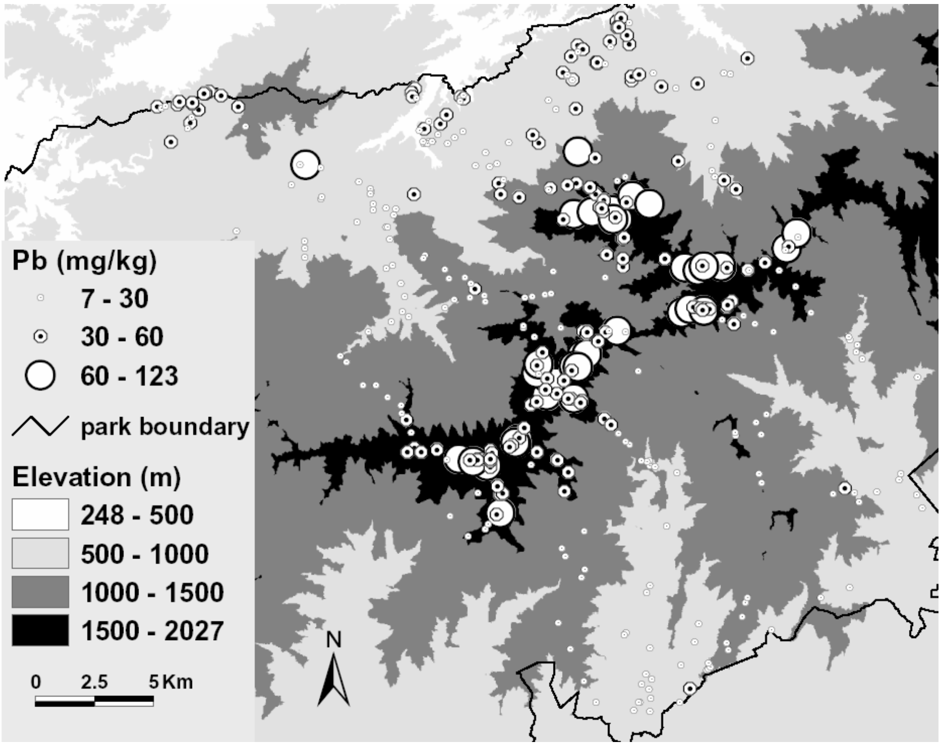
Pb (mg/kg)
7 - 30
30 - 60
60 - 123
park boundary
Elevation (m)
248 - 500
500 - 1000
1000 - 1500
1500 - 2027
0
2.5
5 Km
N

Figure 1b.

Figure 1. Spatial distribution of measured deposition indices (DI) for a) Acadia National Park (ACAD) (dashed lines indicate 100 m elevation contours and gray areas indicate seawater), sulfur deposition in kg/ha/y and b) Great Smoky Mountains National Park (GRSM), Pb concentration in forest floor in mg/kg. Larger bubbles indicate a higher deposition index. (Fig. from Weathers et al. 2006.)

83.23 º W) that passes through the center of the park. Areas < 100 m from primary roads (US 441, US 321, TN 73, TN 71, and US 19) were excluded. For each of the 378 forest floor locations, vegetation, elevation, slope, and aspect attributes were extracted using GIS data and compared to the attributes of the entire GRSM study area to ensure that each of these variables was adequately represented.

All samples were then transported to the Institute of Ecosystem Studies (IES) where they were processed and chemically analyzed (see methods in Weathers et al., 2006, for details).

Creating Deposition Maps

We "normalized" our deposition index measurements (TF or Pb in forest floor) to calculate unitless enhancement factors (relative deposition), to identify how many much more enhanced times deposition was at each sampling point relative to a low elevation "reference" site. We then related all of these enhancement factors to the local monitoring station data, which we considered to be reference or base deposition. In this way, it is possible, for any given year, to estimate how much more, or less deposition any particular 30 x 30m grid cell receives *compared to* the NADP plus CASTNET (wet plus dry deposition) measured at the monitoring stations. It is important to note that the monitoring station data are a key component of this analysis.

We have named our model **LANDMod**. The results of this effort are deposition maps (Figs 2 and 3) for the whole park. We verified the map by withholding data (30 sampling locations for

ACAD, 32 for GRSM) from our statistical model and comparing our modeled results to measured deposition.

Figure 2

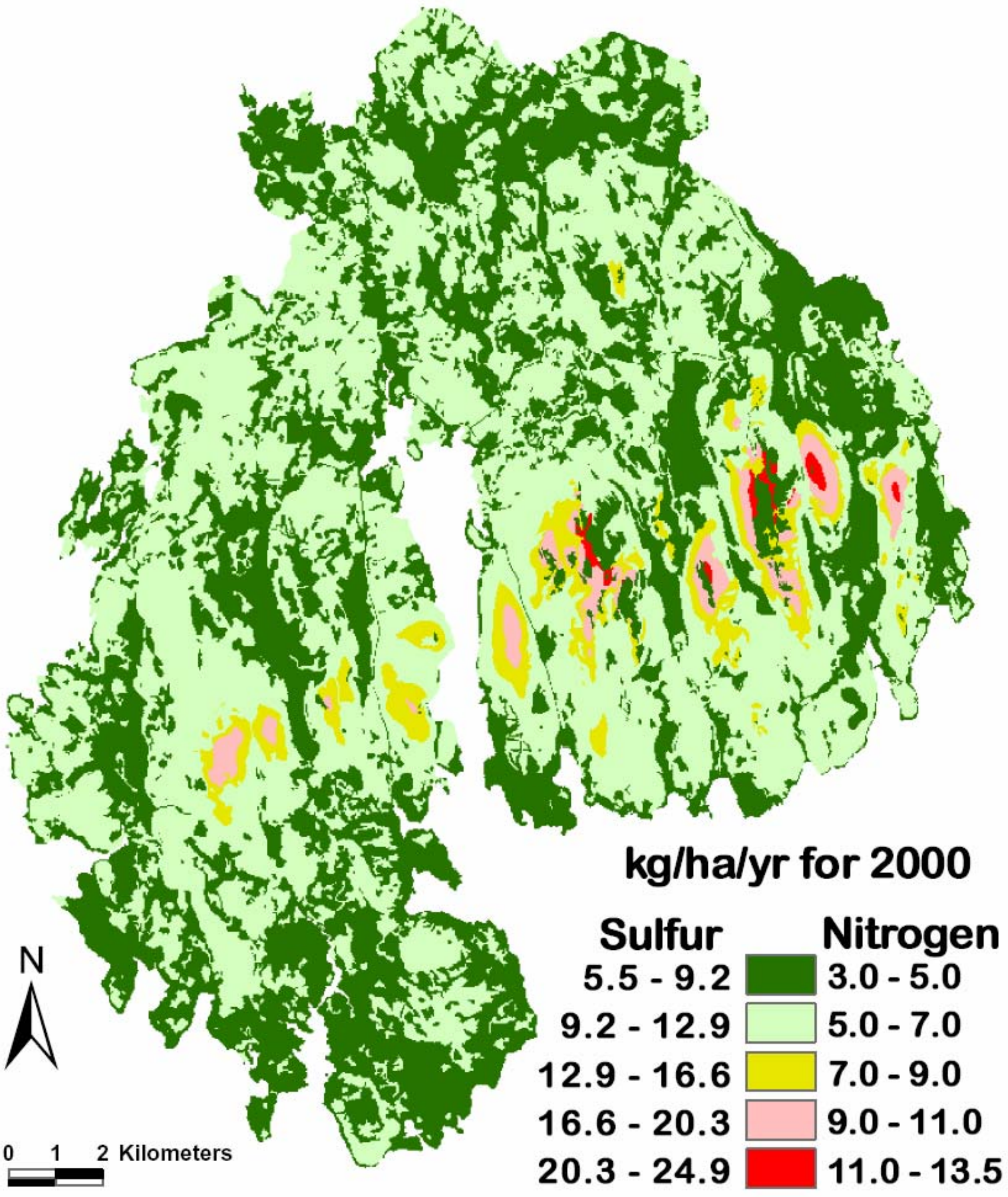
kg/ha/yr for 2000
Sulfur
Nitrogen
5.5 - 9.2 🟩 3.0 - 5.0
9.2 - 12.9 🟩 5.0 - 7.0
12.9 - 16.6 🟨 7.0 - 9.0
16.6 - 20.3 🟥 9.0 - 11.0
20.3 - 24.9 🟥 11.0 - 13.5
N
0 1 2 Kilometers

Figure 2. Modeled (using the empirically-based LANDMod, as described in the text) atmospheric deposition of N (kg/ha/yr) and S (kg/ha/yr) for the year 2000 to Mount Desert Island study area of Acadia National Park (ACAD). Deposition was calculated as deposition scaling factor (unitless) * year 2000 base deposition from 52 weeks of NADP wet and CASTNET dry deposition (5.5 kg/ha/yr of S and 3.0 kg/ha/yr of N). (Fig. from Weathers et al. 2006.)

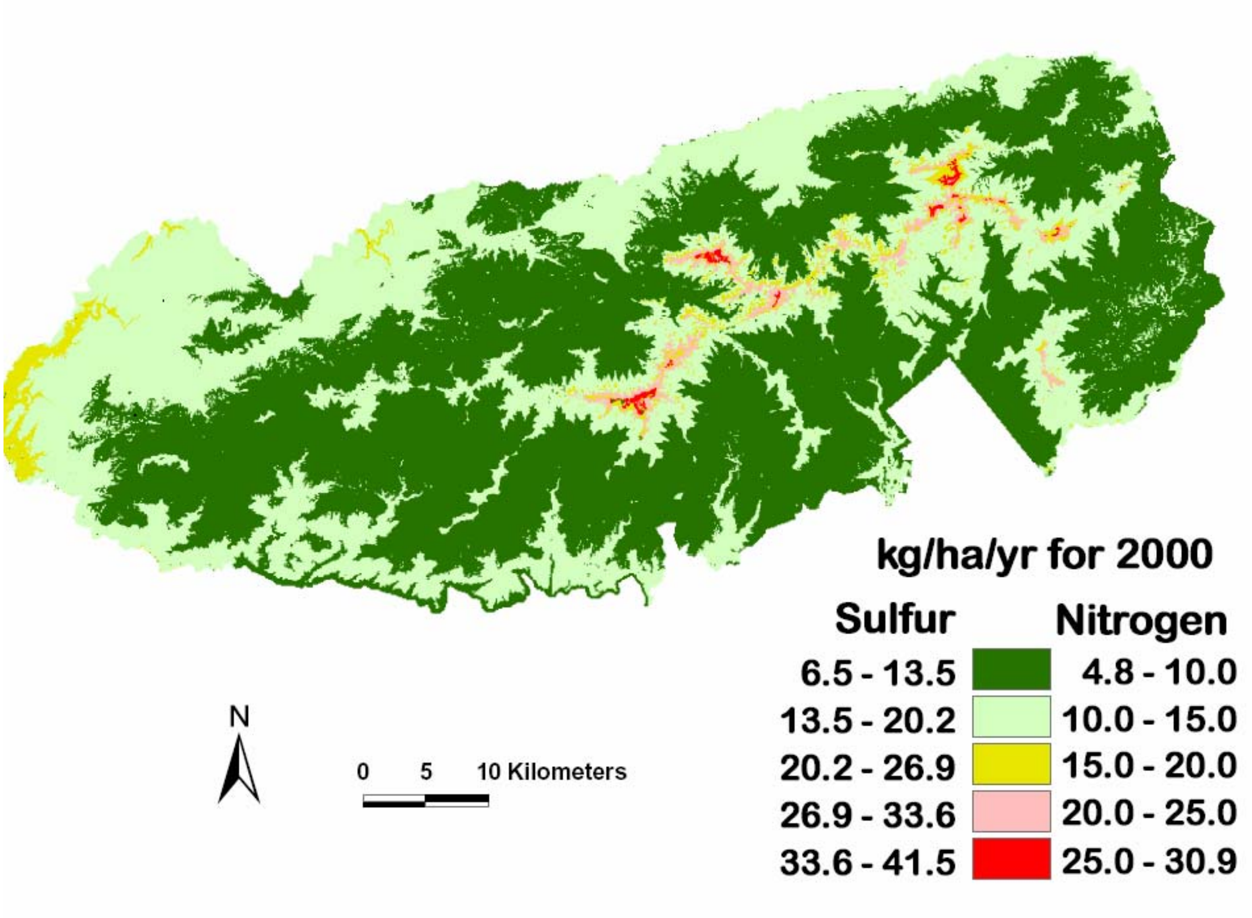
kg/ha/yr for 2000
Sulfur
Nitrogen
6.5 - 13.5
4.8 - 10.0
13.5 - 20.2
10.0 - 15.0
20.2 - 26.9
15.0 - 20.0
26.9 - 33.6
20.0 - 25.0
33.6 - 41.5
25.0 - 30.9
N
0 5 10 Kilometers

Figure 3

Figure 3. Modeled (using the empirically-based LANDMod, as described in the text) atmospheric deposition of N (kg/ha/yr) and S (kg/ha/yr) for the year 2000 to Great Smoky Mountain National Park (GRSM). Deposition was calculated as deposition scaling factor (unitless) * year 2000 base deposition from 52 weeks of NADP wet and CASTNET dry deposition (9.1kg/ha/yr of S and 6.8 kg/ha/yr of N). (Fig. from Weathers et al. 2006.)

Results

Subsampling within the Parks

The regions in which we sampled within the Parks were representative subsets of the Park landscapes (see Tables 1a and 1b).

Patterns of Deposition across the Landscape

Measured forest throughfall S deposition at ACAD ranged from 0.85 - 9.83 kg S/ha (n= 285 resin throughfall sample locations) for the time period 7 June to 18 September 2000, with a mean of 2.5 kg S /ha. Measured deposition indices at the 5 nonforested ACAD locations (i.e. open areas) ranged from 1.05 to 1.56 kg S/ha (Fig. 1a). Measured Pb concentration in forest floor at GRSM ranged from 7.06 to 122.8 mg Pb/kg (n= 378 forest floor sample locations), with a mean of 33.7 mg/kg (Fig. 1b). Our scaling factor analysis (unitless measure that shows how many times greater deposition is at any one point relative to a low elevation site) thus showed that hotspots of deposition receive approximately 12-fold higher deposition than "cold" spots at ACAD and 17-fold higher at GRSM. The spatial distribution of deposition scaling factors for the two parks indicates that many—but not all—of the high elevation locations have high scaling factor values (Figs. 1a and 1b).

Table 1. Summary of landscape features for study area and sampling points at a) Acadia National Park (ACAD) and b) Great Smoky Mountains National Park (GRSM), eastern United States.

Table 1a. Acadia National Park

	study area		sampling points			study area		sampling points	
	km^2	%	number	%		km^2	%	number	%
Vegetation					Elevation(m)				
conifer	93	34	162	56	0-100	230	83	95	33
mixed*	100	36	29	10	100-200	32	11	106	37
deciduous	34	12	94	32	200-300	11	4	54	19
nonforested	52	19	5	2	300-400	4	2	30	10
					400-463	1	0.3	4	1
Slope (deg)					Aspect (deg)				
0-0.001	14	5	1	0.3	NA (no slope)	14	5	9	3
0.001-10	204	73	98	34	0-90	70	25	47	16
10-20	45	16	116	40	90-180	58	21	107	37
20-30	12	4	54	19	180-270	75	27	61	21
30-50	3	0.9	21	7	270-360	62	22	66	23

Table 1b. Great Smoky Mountains National Park

	study area		sampling points			study area		sampling points	
	km^2	%	number	%		km^2	%	number	%
Vegetation					Elevation(m)				
conifer	279.56	13.5	50	13.2	266-500	87.93	4.2	10	2.6
mixed*	50.64	2.4	178	46.8	500-1000	931.08	44.9	156	41.1
deciduous	1695.48	81.7	152	40.0	1000-1500	873.58	42.1	80	21.1
nonforested	48.32	2.3	0	0	1500-2027	181.40	8.7	134	35.3
Slope (deg)					Aspect (deg)				
0	22.43	1.1	3	0.8	NA (no slope)	18.79	0.9	2	0.5
1-10	199.90	9.6	37	9.7	0-90	486.55	23.5	76	20.0
10-20	592.32	28.6	80	21.1	90-180	511.64	24.7	69	18.2
20-30	910.02	43.9	132	34.7	180-270	520.17	25.1	124	32.6
30-65	349.32	16.8	128	33.7	270-360	536.84	25.9	109	28.7

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*conifer and deciduous

Base Deposition

Calculated base deposition at the ACAD monitoring station site for the period 6 June 2000 to 19 September 2000 was 1.14 kg S/ha (0.99 kg/ha of wet S + 0.15 kg/ha of dry S). The base deposition at the forest adjacent to the Great Smoky Mountains NADP site was 16.61 mg Pb/kg.

Statistical Model Results

We considered a host of variables in our statistical models, including such variables as vegetation type, slope, aspect, qualitative canopy openness, and understory type. Our analysis revealed that elevation and vegetation type were the two most important variables in the statistical model used for GIS mapping (Figs. 4, 5 and 6). Such variables as aspect, slope, leaf area, distance from the coast, diameter at breast height (dbh) were not found to be significant drivers of deposition *at the landscape scale* (i.e., 10s to 100s of kms) (Table 2).

The combination of elevation, forest type, and elevation*forest type interaction terms produced a highly significant GIS model that explained about half of the variation in deposition (Table 2).

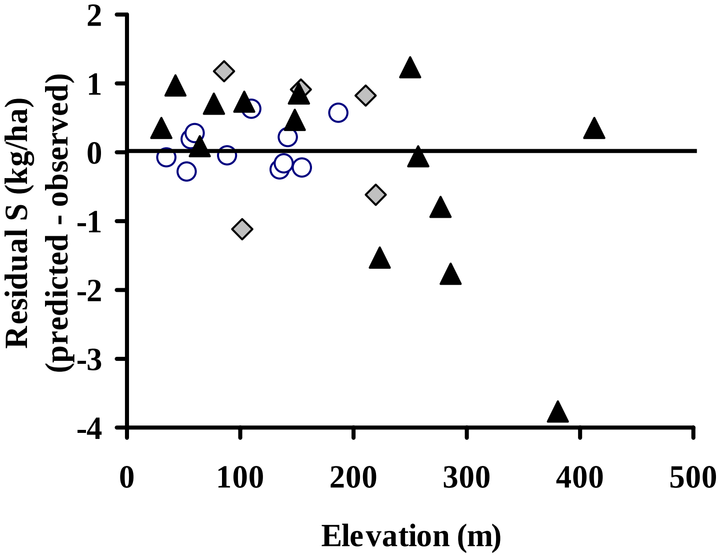

Figure 4a

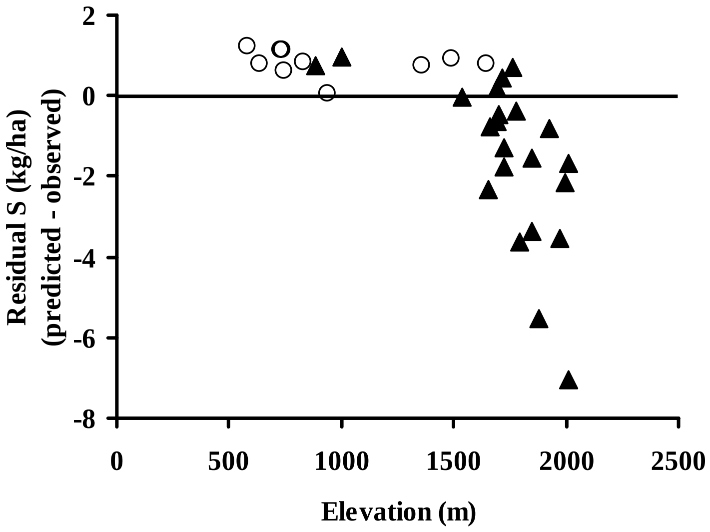

Figure 4b

Figure 4. Residuals of predicted versus observed deposition (S kg/ha) plotted as a function of elevation for (a) ACAD and (b) GRSM. GIS and field vegetation data were matched for this analysis. Deciduous (open circles), coniferous (filled triangles) and mixed (filled diamonds; ACAD only) stands are shown. (Fig. from Weathers et al. 2006.)

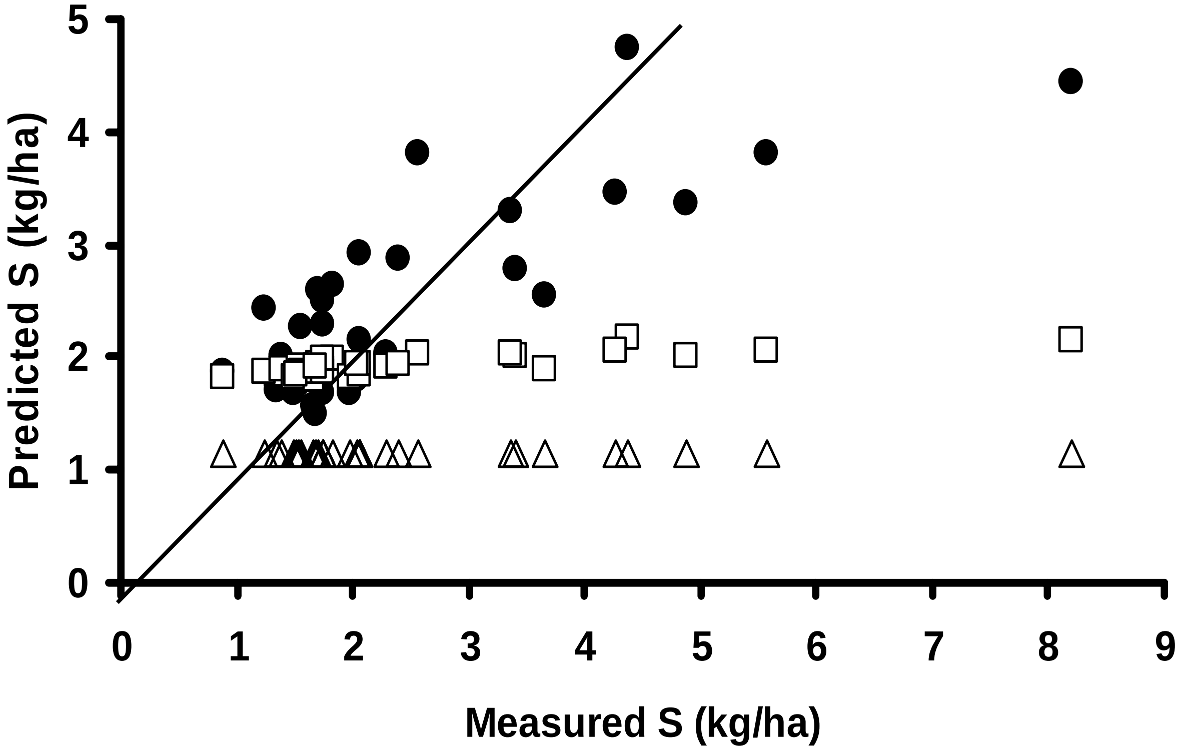

Figure 5

Figure 5. Comparison of S deposition at 32 points validation data points from Acadia National Park, ACAD, (see Results Section for description of validation points) using the LANDMod described in this paper (filled circles), deposition estimates for each of the points based on Ollinger et al. 1993 (open squares), and from the NADP and CASTNet programs (open triangles); these monitoring data have only one value for all locations in the region (see Methods Section). (Fig. from Weathers et al. 2006.)

Table 2. Statistical model parameters of deposition scaling factor (unitless values that show relative deposition, see Fig.1 and Methods Section) as a function of field and GIS-measured landscape variables. Statistical model for field measured landscape variables (STATMod) are shown for a) Acadia National Park, ACAD (n=285), and b) Great Smoky Mountains National Park, GRSM (n=378). Mapping equation parameters of deposition scaling factor as a function of GIS-measurable landscape variables for c) ACAD (n=255) and d) GRSM (n=346).

Table 2a.

variable	coefficient	P	partial r^2	model r^2
intercept	0.98720			
elevation (m)	0.00265	<0.0001	0.212	0.212
conifer *	-0.03922	0.8676	0.105	0.317
tree dbh (cm)	0.01521	0.0098	0.019	0.336
elevation*conifer	0.00482	0.0003	0.031	0.367

Table 2b.

variable	coefficient	P	partial r^2	model r^2
intercept	1.60345			
elevation (m)	0.00263	0.0193	0.171	0.171
conifer *	-0.03248	0.9061	0.080	0.251
mixed forest *	0.48995	0.0380	0.018	0.269
dist. to coast (m)	-0.00015	0.0399	0.006	0.275
elevation*conifer	0.00481	0.0008	0.032	0.307

Table 2c.

variable	coefficient	P	partial r^2	model r^2
intercept	3.32326			
elevation^2 (m)	0.000001995	<0.0001	0.322	0.322
elevation (m)	-0.00427	<0.0001	0.096	0.418
conifer *	-0.15380	0.6113	0.044	0.416
slope (frm DEM)	0.01106	0.0250	0.007	0.468
elevation*conifer	0.000648	0.0070	0.008	0.479

* presence

Table 2d.

variable	coefficient	P	partial r^2	model r^2
intercept	3.69836			
elevation^2 (m)	0.000002195	<0.0001	0.331	0.331
elevation	-0.00464	<0.0001	0.125	0.455
conifer *	-0.26948	0.3737	0.044	0.499
elevation*conifer	0.000748	0.0030	0.014	0.513

LANDMod Deposition Maps

We created LANDMod deposition maps for both S and N (Figs. 2 & 3, see appendix 1). The ACAD scaling-factor map was multiplied by ACAD reference deposition (RD) of 3.00 kg N/ha/yr (2.51 kg N/ha/yr of wet deposition from NADP + 0.49 kg N/ha/yr of dry deposition from CASTNET) for all of calendar year 2000 to produce a 2000 modeled ACAD N deposition map. Nitrogen deposition values ranged from 3.0 to 14 kg N/ha/yr, with an area weighted mean of 5 kg N /ha/yr of N for the entire ACAD study area. For sulfur, the ACAD scaling-factor map was multiplied by ACAD reference deposition of 5.55 kg S/ha/year (4.87 kg/S/ha/yr of wet deposition + 0.68 kg S/ha/yr dry deposition) for all of calendar year 2000 to produce a 2000 modeled ACAD S deposition map (Fig. 2). Sulfur deposition values ranged from 5.6 to 25 kg S/ha/yr, with an area-weighted mean of 9.5 kg S/ha/yr (Fig. 2).

Using the same method as for ACAD, the GRSM scaling-factor map was multiplied by GRSM reference deposition of 6.8 kg N/ha/yr (2.9 kg N/ha/yr wet deposition + 3.9 kg N/ha/yr of dry deposition from the NADP and CASTNet programs, respectively) for calendar year 2000 to produce a 2000 modeled GRSM N deposition map. Nitrogen deposition values ranged from 5 to 31 kg N/ha/yr, with an area-weighted mean of 10 kg N/ha/yr for the entire park area. For sulfur, the GRSM scaling-factor map was multiplied by GRSM reference deposition of 9.1 kg S/ha/yr of S (5.8 kg S/ha/yr of wet deposition + 3.3 kg/ha/yr of dry deposition) for all of calendar year 2000 to produce a 2000 modeled GRSM S deposition map. The resulting LANDMod GRSM S deposition map shows S deposition values that ranged from 7 to 42 S kg/ha/yr, with an area-weighted mean of 14 kg S/ha/year (Fig. 3).

Verification of LANDMod Deposition Maps

Sulfur deposition residuals (LANDMod predicted – throughfall measured) were plotted as a function of elevation to examine model function (Fig. 4). After correcting the mismatches in vegetation type between GIS and field data, the correlation between measured and predicted deposition was significant at ACAD ($r^2 = 0.64$, $P < 0.0001$) and GRSM ($r^2 = 0.80$, $P < 0.0001$), though not 1:1. At both ACAD and GRSM, deposition was slightly overestimated at low elevations and underestimated at high elevations by our model. The residual plots show interesting and consistent patterns between the two sites. The model behaves differently at low elevation than at high (low elevation locations are also low deposition and high elevation are high deposition sites) and for different vegetation types: At ACAD, the model overpredicts at low elevations, more so for coniferous and mixed vegetation than for deciduous. The residuals were closer to zero for deciduous vegetation at low elevations. At high elevation, the model underpredicts for coniferous vegetation. For GRSM, the model also overpredicts at low elevation for deciduous vegetation (the primary vegetation type at low elevation) and underpredicts (relative to field measurements) rather significantly for high elevation conifer forests.

As a further validation of the ACAD model, we compared the LANDMod predicted deposition with sulfur output from a frequently used regional model of deposition (Ollinger et al. 1993, www.pnet.sr.unh.edu/climcalc) as well as with reference data from the NADP and CASTNET monitoring stations (Fig. 5). Although our validation data suggest that the LANDMod underestimates high deposition at high elevations, the existing estimates (e.g. Ollinger et al. 1993) for those locations are considerably lower than what our approach produces.

Discussion

How Good are the Maps?

There are several important factors that determine the accuracy of the deposition maps produced here. Errors in sampling and analyzing the point measurements of deposition chemistry would decrease accuracy, but results from our quality control procedures (+/- 10% precision and accuracy) lead us to believe that this is not a major source of error. When comparing predicted vs. observed data, errors in GIS data (e.g., vegetation) are clearly an important factor that decrease the accuracy of the deposition maps, as indicated by the major improvements in the validation analyses when the erroneous GIS data discussed in the Results section were corrected to match the observed field data. It seems reasonable to assume that as remotely sensed data with a resolution of 1 m or better become available, the number of mismatches between GIS and field vegetation should decrease. The sources of error described above must be taken into consideration. However, we think that the greatest and most interesting sources of uncertainty in the deposition maps arise from as-yet unquantified or unidentified variables that cause real variability in deposition; variables that are not part of our StatMod or LANDMod, but are evident in the pattern displayed in our validation data (NIST/SEMATECH 2005) (Fig. 4).

The relationships between independent variables and deposition indices were highly significant statistically, yet they explained only half of the variance in the data. What explains the rest of the variance in actual deposition? It is likely that the independent variables that explain the rest of the variance, at a broad scale, are those that affect the capture of particles and gases by vegetation, such as (1) canopy roughness and orientation and (2) topographic exposure. Our data suggest an enhancement for predominately coniferous forest—those regions with up to 25% deciduous canopies compared with pure deciduous or coniferous forest at ACAD; however, this was not true at GRSM. Our limited data from ACAD showing large differences in fluxes among coniferous trees suggests that there may be factors in addition to leaf area index (LAI) that control deposition. To wit, the highest measured deposition in the ACAD validation data set—the biggest outlier for predicted vs. measured values—was a large-statured (>3m in height) conifer that was situated on an exposed ridge. This is just the kind of tree-specific (or site specific) depositional environment that no general model can readily replicate.

A major variable controlling small-scale patterns in deposition is likely to be vegetation exposure, including trees whose crowns are exposed above their neighbors and trees growing in unsheltered locations such as ledges and promontories. A tree with high exposure is likely to get much higher deposition than another tree of equivalent LAI and height that is sheltered by neighbors. This has been demonstrated in several studies of forest edges (e.g., Lindberg and Owens 1993; Weathers et al. 1995, 2000). While we were able to generate a GIS index of topographic exposure using elevation models of the ground surface, we believe the horizontal and spatial resolution of those data were not yet good enough to generate an index of the combination of canopy architecture and exposure that is relevant to rates of deposition. However, the development of high resolution remotely sensed LIDAR data shows promise in indicating both the elevation of tree canopies and the density of tree canopies, which should help considerably in filling this data gap. Another challenge is the scale at which topographic exposure might matter, which is likely to be less than 10s of meters rather than the 30x30m pixel resolution of most GIS datalayers. If it were possible to measure canopy architecture and exposure—or surrogates—easily, and better yet, if they were available as GIS data layers, landscape models of deposition could probably be made more accurate. Thus an important next step for deposition research will be to develop a way to quantify different canopy LAIs and architectures along with their exposure, perhaps using remotely sensed data (e.g., LIDAR), and examine their relationship to deposition.

Map Verification

The residual data show that at ACAD, the LANDMod map most significantly underestimated deposition in high elevation regions; these points also represent the highest measured deposition values. The three datapoints that represent high elevation conifer forests show the biggest discrepancies. The GRSM validation plots show much bigger discrepancies for high elevation regions, indicating that high elevation data reduce the homogeneity of variance for the LANDMod model (Fig. 4b). That the variance is high, especially at high elevation, is a somewhat unsurprising result given the rather large scatter of the primary data.

We cannot say definitively how much of an influence our use of Pb in forest floor to create the LANDMod had on the validation comparison. However the consistency between the two validation plots (ACAD and GRSM) suggests that there are missing variables in the LANDMod rather than an effect of using different deposition indices at ACAD vs. GRSM. We might predict that the LANDMod would underestimate leaf-on patterns of S deposition (i.e., the measured values) since the Pb index integrates deposition over seasons and years. It is likely that actual sulfur deposition for a short-time during the season of highest deposition (i.e., when we made the S flux measurements for the validation data) would be higher than predicted deposition. In addition, the LANDMod estimates deposition to a 30x30m pixel that is characterized by an average elevation and vegetation type. We measured deposition to a small area that surrounded individual trees.

Are We Better Off than We Were in Regard to Deposition Estimates?

Yes. Clearly these models need refinement to be able to more accurately predict deposition to heterogeneous landscapes. The validation data sets can be used to suggest next steps, such as the need for identifying and quantifying controls on deposition to high-elevation conifer forests. However, it is important to consider these results within the perspective of the data that are currently available. The LANDMod appears to be conservative: it slightly overpredicts for low elevation, low deposition environments, but significantly underestimates input for high elevation, high deposition environments. The deposition data that have been available for these landscapes prior to LANDMod either show no spatial variability (i.e., as for GRSM with one NADP + CASTNET value for the region), or some spatial variability over the region (i.e., the Ollinger et al. 1993 regional deposition model), but do not capture spatial variability at the scale of 1-100s of kms. Thus, for ACAD, compared with S predictions based on Ollinger et al. (1993), a widely used regional deposition model (see also www.pnet.sr.unh.edu/climcalc), and with the monitoring station data alone, our LANDMod S deposition maps seem to capture the response of measured deposition to both elevation and vegetation type. For GRSM, in addition to the wet + dry deposition estimate, cloud deposition at the peak of Clingman's Dome for the time period over which we sampled was estimated (Lovett 2001). For this high elevation region, the predicted cloud + wet + dry deposition based on the monitoring stations (27 kgS/ha) is more than twice the S measured in the throughfall at the highest measured deposition location, while the LANDMod predicts less than half the measured S in throughfall at that location. Again, both measured and our modeled deposition is quite conservative compared to existing data.

How transferable is this approach?

The empirical scaling factors and models we have created are likely to be applicable to other chemical species that are deposited from the atmosphere in approximately the same proportions and by the same mechanisms as N and S are deposited. In addition, where predictable and strong relationships can be established between chemical species x and S, the models might be extended.

Our approach may also be applicable to other sites. We think that elevation and vegetation type are likely to be important driving variables for deposition in other mountain ranges around the world, albeit the coefficients may differ from landscape to landscape depending upon such factors as the dominant mode of deposition (e.g., fog, dry, and whether snow is a dominant proportion of wet deposition), patchiness of the landscape and its elevational span. Both of these topics warrant further research and comparison.

Updating maps

The statistical relationships in the LANDMod that predict the spatial heterogeneity of the atmospheric deposition maps developed here are not expected to change from year to year. However, reference deposition data from the NADP and CASTNET monitoring stations do change over time. Since reference deposition data are incorporated in the last step of creating the deposition map, it is straightforward to update the deposition maps to reflect this temporal change. The independent variables that drive the LANDMod (vegetation and elevation) are unlikely to change significantly from year-to-year, although major vegetation changes as a result of an anthropogenic disturbance such as logging, or natural disturbances such as fire, major storms or insect outbreaks. Such changes would affect the spatial pattern of deposition and could be accommodated by re-calculating scaling factors. If datalayers with finer resolution or other key data become available, they too could be substituted for the old datalayers at the stage where scaling-factor maps are generated (Appendix 1).

Who cares?

Our park-wide LANDMod deposition maps are based on spatially explicit inputs, and as such they calculate small regions of high, and low deposition (hotspots and coldspots). But, at the scale of the whole park, does it matter? One way to answer this question is to compare our area-weighted total deposition with currently available data, i.e., data from the NADP and CASTNET monitoring stations. Our models indicate that total, area-weighted N deposition is much greater than suggested by the local monitoring station data: ACAD has 70% greater, and GRSM 50% greater deposition (LANDMod compared to NADP + CASTNET).   These results demonstrate that even the average total deposition across mountainous landscapes is substantially underestimated by using data from single-point, low-elevation monitoring stations.

At places such as GRSM where a complete, spatially explicit biologic inventory is being compiled, it may be possible to overlay deposition maps of hotspots with known sensitive species distributions to identify populations at greatest risk of damage from air pollution. Knowledge of deposition patterns can also be used to good advantage in designing effects research. For instance, identifying strong deposition gradients in areas where other factors (e.g., soil type or temperature) are relatively constant could facilitate comparative studies of the effects of deposition on plants and animals. In general, knowing spatial variability in deposition across a landscape can enhance tests of ecological responses to a range of inputs.

Summary

Estimates of atmospheric deposition in complex terrain are currently inadequate; they are either incomplete (spatially representative, but only for wet deposition, for example), or not spatially explicit. In this research, we have developed an empirically-based model for scaling-up to the landscape (10s to 100s of kms) from point measurements to get spatially-explicit estimates of atmospheric deposition. Our LANDMod was created by applying a statistical model to GIS datalayers. Key to this work are the data generated from the NADP and CASTNET national monitoring networks. These data were used for reference deposition; the deposition estimate from which to scale-up. The resultant maps show several-fold variation in deposition across the landscape of two national parks. Vegetation type and elevation are the independent variables that most strongly controlled deposition.

This modeling effort and our validation exercises and explorations were revealing: they confirmed that there is large variance in deposition, especially at high elevations and that, with current tools, landscape scale (10s to 100s of kms) models are unlikely to be able to capture extreme deposition over areas of small spatial extent (e.g., meters to 10s of meters).

Next Steps?

Future research should be focused on exploring the strengths and limitations of the LANDMod and especially refining it to account for (currently) unexplained variation in deposition. There are several ways in which this might be accomplished, including (1) making deposition measurements year round and testing/refining the model; (2) making measurements through additional years; (3) validating the model with new or existing data; (4) testing the model in other park landscapes; (5) relating remote sensing data (e.g., LIDAR) with deposition surrogates as those data become available; (6) linking the modeled deposition fluxes to potential biologic and ecologic response variables.

Literature Cited

Anderson, J. B., R. E. Baumgardner, V. A. Mohnen, and J. J. Bowser. 1999. Cloud chemistry in the eastern United States, as sampled from three high-elevation sites along the Appalachian Mountains. Atmospheric Environment 33.

Baumgardner, R. E., T. F. Lavery, C. M. Rogers, and S. S. Isil. 2002. Estimates of the atmospheric deposition of sulfur and nitrogen species: Clean Air Status and Trends Network, 1990-2000. Environmental Science and Technology 36.

Baumgardner, R. E., S. S. Isil, T. F. Lavery, C. M. Rogers, and V. A. Mohnen. 2003. Estimates of cloud water deposition at mountain acid deposition program sites in the Appalachian Mountains. Journal of the Air & Waste Management Association 53.

Driscoll, C. T., G. B. Lawrence, A. J. Bulger, T. J. Butler, C. S. Cronan, C. Eager, K. F. Lambert, G. E. Likens, J. L. Stoddard, and K. C. Weathers. 2001. Acidic deposition in the northeastern United States: Sources and inputs, ecosystem effects, and management strategies. BioScience 51.

Fenn, M. E., R. Haeuber, K. Tonnessen, J. S. Baron, S. Grossman-Clarke, D. Hope, D. A. Jaffe, S. Copeland, L. Geiser, H. M. Rueth, and J. O. Sickman. 2003. Nitrogen emissions, deposition and monitoring in the Western United States. BioScience 53.

Finkelstein, P. L., T. G. Ellestad, J. F. Clarke, T. P. Meyers, D. B. Schwede, E. O. Hebert, and J. A. Neal. 2000. Ozone and sulfur dioxide dry deposition to forests: observations and model evaluation. Journal of Geophysical Research-Atmospheres 105.

Grimm, J. W., and J. A. Lynch. 2004. Enhanced wet deposition estimates using modeled precipitation inputs. Environmental Monitoring and Assessment 90.

Hedin, L. O., L. Granat, G. E. Likens, T. A. Buishand, J. N. Galloway, T. J. Butler, and H. Rodhe. 1994. Steep declines in atmospheric base cations in regions of Europe and North America. Nature 367.

Hettelingh, J. P., R. J. Downing, and P. A. M. de Smet, (eds.) 1995a. Calculation and Mapping of Critical Loads in Europe. RIVM Report No. 259101004. Coordination Center for Effects. Bilthoven, The Netherlands.

Hettelingh, J. P., M. Posch, P. A. M. DeSmet, and R. J. Downing. 1995b. The use of critical loads in emission reduction agreements in Europe. Water, Air, and Soil Pollution 85.

Holland, E. A., B. H. Braswell, J. Selzman, and J-F. Lamarque. 2005. Nitrogen deposition onto the United States and western Europe: Synthesis of observations and models. Ecological Applications 15.

Johnson, A. H., T. G. Siccama, and A. J. Friedland. 1982. Spatial and temporal patterns of lead accumulation in the forest floor in the Northeastern United States. Journal of Environmental Quality 11.

Kelly, V. R., G. M. Lovett, K.C. Weathers, and G. E. Likens. 2002. Trends in atmospheric concentration and deposition compared to regional and local pollutant emissions at a rural site in southeastern New York, USA. Atmospheric Environment 36.

Kimball, K. D., R. Jagels, G. Gordon, K. C. Weathers, and J. Carlisle. 1988. Differences between New England coastal fog and mountain cloud water chemistry. Water, Air and Soil Pollution 39.

Lavery, T. F., C. M. Rogers, H. K. Howell, and M. C. Burnett. 2003. Clean Air Status and Trends Network (CASTNET) 2002 Annual Report. EPA Contract No. 68-D-03-052.

Prepared by MACTEC, Inc. for US Environmental Protection Agency, Office of Atmospheric Programs. Washington, D.C.

Likens, G. E., and F. H. Bormann, editors. 1995. Biogeochemistry of a forested ecosystem. Second edition. New York: Springer-Verlag.

Lindberg, S. E., and J. G. Owens. 1993. Throughfall studies of deposition to forest edges and gaps in montane ecosystems. Biogeochemistry 19.

Lovett, G. M. 2001. Cloud water deposition to Clingman's Dome in 2000. Report to Harding, ESE.

Lovett, G. M. 1994. Atmospheric deposition of nutrients and pollutants in North America: An ecological perspective. Ecological Applications 4.

Lovett, G. M., W. A. Reiners, and R. K. Olson. 1982. Cloud droplet deposition in subalpine balsam fir forests: hydrological and chemical inputs. Science 218.

Lubinski, S., K. Hop, and S. Gawler. October 2003. US Geological Survey-National Park Service Vegetation Mapping Program: Acadia National Park, Maine. Project Report. US Geological Survey and Maine Natural Areas Program. http://biology.usgs.gov/npsveg/acad/report.pdf.

Lynch, J. A., J. W. Grimm, and V. C. Bowersox. 1995. Trends in precipitation chemistry in the United States: a national perspective, 1980-1992. Atmospheric Environment 29.

Lynch, J. A., V. C. Bowersox, and J. W. Grimm. 2000. Changes in sulfate deposition in eastern USA following implementation of Phase I of Title IV of the Clean Air Act Amendments of 1990. Atmospheric Environment 34.

MacKenzie, M. D. 1993. The vegetation of Great Smoky Mountains National Park: Past, present, and future. Ph.D. Dissertation. The University of Tennessee.

Meyers, T. P., P. Finkelstein, J. Clarke, T. G. Ellestad, and P. F. Sims. 1998. A multilayer model for inferring dry deposition using standard meteorological measurements. Journal of Geophysical Research-Atmospheres 103.

Mohnen, V. A., and J .A. Kadlecek. 1989. Cloud chemistry research at Whiteface Mountain. Tellus 41B.

Mohnen, V. A., and R. J. Vong. 1993. A climatology of cloud chemistry for the eastern United States derived from the mountain cloud chemistry project. Environmental Review 1.

National Atmospheric Deposition Program (NADP). 2004. National Atmospheric Deposition Program 2003 Annual Summary. NADP Data Report 2004-1. Illinois State Water Survey, Champaign, IL.

Nanus, L., D. H. Campbell, G. P. Ingersoll, D. W. Clow, and M. A. Mast. 2003. Atmospheric deposition maps for the Rocky Mountains. Atmospheric Environment 37.

NIST/SEMATECH. 2005. e-Handbook of Statistical Methods, 26 January 2005. http://www.itl.nist.gov/div898/handbook/.

Ollinger, S. V., J. D. Aber, G. M. Lovett, S. E. Millham, R. G. Lathrop, and J. M. Ellis. 1993. A spatial model of atmospheric deposition for the northeastern United States. Ecological Applications **3**:459-472.

Schlesinger, W. H. 1997. Biogeochemistry: an analysis of global change. Second edition. Academic Press. San Diego, CA.

Summers, P. W. 1995. Time trend of wet deposition acidifying potential at five ecological monitoring sites in eastern Canada 1981-1993. Water, Air and Soil Pollution 85.

van Leeuwen, E. P., G. P. J. Erisman, and J. W. Erisman. 1996. Mapping wet deposition of acidifying components and base cations over Europe using measurements. Atmospheric Environment 30.

Vong, R. J., J. T. Sigmon, and S. F. Mueller. 1991. Cloud water deposition to Appalachian forests. Environmental Science and Technology 25.

Weathers, K. C. 1999. The importance of cloud and fog to the maintenance of ecosystems. Trends in Evolution and Ecology 14..

Weathers, K. C., G. E. Likens, F. H. Bormann, J. S. Eaton, W. B. Bowden, J. L. Andersen, D. A. Cass, J. N. Galloway, W. C. Keene, K. D. Kimball, and others. 1986. A regional acidic cloud/fog water event in the eastern United States. Nature 319.

Weathers, K. C., G. E. Likens, F. H. Bormann, S. H. Bicknell, B. T. Bormann, B. C. Daube, J. S. Eaton, J. N. Galloway, W. C. Keene, K. D. Kimball, and others. 1988. Cloudwater chemistry from 10 sites in North America. Environmental Science and Technology 22.

Weathers, K. C., and G. M. Lovett. 1998. Acid deposition research and ecosystem ecology: synergistic successes. Pages 195-219 in M. L. Pace and P. M. Groffman, editors. Successes, Limitations and Frontiers in Ecosystem Science. Springer-Verlag New York, Inc.

Weathers, K. C., G. M. Lovett, and G. E. Likens. 1995. Cloud deposition to a spruce forest edge. Atmospheric Environment 29.

Weathers, K. C., G. M. Lovett, G. E. Likens, and R. Lathrop. 2000. The effect of landscape features on deposition to Hunter Mountain, Catskill Mountains, New York. Ecological Applications 10.

Weathers, K. C., S. M. Simkin, G. M. Lovett, and S. E. Lindberg. 2006. Empirical modeling of atmospheric deposition in mountainous landscapes. Ecological Applications 16.

Appendix

Appendix: Flow chart showing the steps involved in this research to create an empirical model of atmospheric deposition for Acadia (ACAD) and Great Smoky Mountain National (GRSM) parks.