AN ABSTRACT OF THE DISSERTATION OF

Todd C. McDonnell for the degree of
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Title: Spatial and Temporal Effects of Atmospheric Deposition, Climate, and Land Management on Forest Nutrient Cycling and Biodiversity

Abstract approved:

Kate J. Lajtha

Emissions largely associated with the combustion of fossil fuels and agriculture has caused elevated atmospheric deposition of nitrogen (N) and sulfur (S) throughout much of the developed world. Increased atmospheric deposition of N and S can lead to soil and surface water acidification and affect forest soil nutrient supply. The critical load (CL) of atmospheric deposition is the level of deposition below which significant harm to ecosystems is not likely to occur. The critical load provides an understanding of the extent to which terrestrial and aquatic systems may be affected by air pollution at present or in the future. This research implements novel methods for generating CL estimates of N and S deposition and evaluates spatial and temporal trends in soil chemistry and plant biodiversity with respect to future climate and land management scenarios. Results are presented in the context of chemical thresholds known to be associated with adverse impacts to terrestrial and aquatic biota of the southern Appalachian Mountains and high elevation Colorado Rocky Mountains.

Forest soils having low exchangeable calcium (Ca) and other nutrient base cation (BC) reserves may induce nutrient deficiencies in acid-sensitive plants and impact commercially important tree species. Past and future depletion of soil BC in response to acidic S deposition, forest management, and climate change alter the health and productivity of forest trees. This research used a process model (Model of Acidification of Groundwater in Catchments [MAGIC]) to address a number of questions related to soil BC status for a group of 65 streams and their watersheds in the southern Blue Ridge physiographic province of the southern Appalachian Mountains. Future S deposition to

the study watersheds used for the Base Scenario was specified according to proposed reductions in S emissions at the time of this study, representing a reduction of 42% of ambient S deposition by 2020. Twenty additional simulations were considered, reflecting four alternate S deposition scenarios (6%, 58%, 65%, and 78% reduction), and various changes in timber harvest, temperature, and precipitation. Base Scenario soil exchangeable Ca and % base saturation showed decreasing trends from 1860 to 2100. Changes in tree harvesting had the largest effect on stream sum of base cations (SBC) and soil BC supply. Each of the scenario projections indicated that median Year 2100 soil exchangeable Ca will be at least 20% lower than pre-industrial values. The simulations suggested that substantial mass loss of soil BC has already occurred since pre-industrial times. Nearly the same magnitude of BC loss is expected to occur over the next 145 years, even under relatively large additional future reductions in S deposition.

Atmospherically deposited S causes stream water acidification throughout the eastern US Southern Appalachian Mountain (SAM) region. Acidification has been linked with reduced fitness and richness of aquatic species and changes to benthic communities. Stream acidification occurs when atmospherically deposited sulfate is conveyed to streams by ground and surface waters, and results in decreased acid neutralizing capacity (ANC) and pH. Maintaining acid-base chemistry that supports native biota depends largely on balancing acidic deposition with the natural resupply of base cations. Stream water ANC is maintained by base cations that mostly originate from weathering of surrounding lithologies. When ambient atmospheric S deposition exceeds the CL the ecosystem can tolerate, stream water ANC conditions may become lethal to biota. This work links results from statistical models that predict continuous ANC and base cation weathering surfaces for streams in the Southern Appalachian Mountain region with established methods for estimating CLs and exceedances. Results showed that 21% of the total length of study region streams displayed ANC values <100 $\mu eq \cdot L^{-1}$, where effects to biota may be anticipated; most were 4th or lower order streams. Nearly one-third of the stream length within the study region exhibited CLs of S deposition < 50 $meq \cdot m^{-2} \cdot yr^{-1}$, which is less than the regional average S deposition of 60 $meq \cdot m^{-2} \cdot yr^{-1}$. Owing to their geologic substrates, relatively high elevation, and cool and moist forested conditions, the percentage of stream length in exceedance was highest for mountain wilderness areas and in national parks, and lowest for privately-owned lands in the valley bottoms. Exceedance results were summarized by 12-digit hydrologic unit code (subwatershed) for use in developing management goals and policy objectives, and for long term monitoring.

To evaluate potential long-term effects of climate change and atmospheric N deposition on subalpine ecosystems, the coupled biogeochemical and vegetation community competition model ForSAFE-Veg was applied to a site at the Loch Vale watershed of Rocky Mountain National Park, Colorado. Changes in climate and N deposition since 1900 resulted in pronounced changes in simulated plant species cover as compared with ambient and estimated future community composition. The estimated CL of N deposition to protect against an average future (2010 – 2100) change in biodiversity of 10% was between 1.9 and 3.5 kg N ha^{-1} yr^{-1}. Results suggest that the CL has been exceeded and vegetation at the study site has already undergone a change of more than 10% as a result of N deposition. Future increases in air temperature are forecast to cause further changes in plant community composition, exacerbating changes in response to N deposition alone.

Spatial and Temporal Effects of Atmospheric Deposition, Climate, and Land Management on Forest Nutrient Cycling and Biodiversity

by
Todd C. McDonnell

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I understand that my dissertation will become part of the permanent collection of Oregon State University libraries. My signature below authorizes release of my dissertation to any reader upon request.

Todd C. McDonnell, Author

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CONTRIBUTION OF AUTHORS

For Chapter 2, William Jackson and Timothy Sullivan contributed to project conceptualization, model input data, and editorial comments. Bernard J. Cosby contributed modeling expertise. Katherine Elliott provided guidance and editorial comments. For Chapter 3, Paul Hessburg, Nicholas Povak, and R. Brion Salter contributed model input data. Keith Reynolds, William Jackson, and Timothy Sullivan provided guidance and editorial comments. Bernard J. Cosby contributed modeling expertise. For Chapter 4, Ellen Porter and Timothy Sullivan contributed to project conceptualization and editorial comments. Salim Belyazid and Harald Sverdrup contributed modeling expertise. William Bowman provided plant parameterizations and editorial comments.

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Spatial and Temporal Effects of Atmospheric Deposition, Climate, and Land Management on Forest Nutrient Cycling and Biodiversity

CHAPTER 1 – INTRODUCTION

Emissions associated with industrial activity, agriculture, and the combustion of fossil fuels have increased atmospheric concentrations of carbon (C), nitrogen (N) and sulfur (S) containing compounds for more than a century (IPCC 2007, U.S. EPA 2009). These changes in elemental cycling have caused increased radiative forcing and increased atmospheric deposition of N and S which have led to climate change and the acidification and eutrophication of soil and surface waters (IPCC 2007, Greaver et al. 2012). Increases in C emissions are expected to continue globally with concomitant future changes in air temperature and precipitation patterns expected throughout the United States (U.S.; Karl et al. 2009). Significant progress has been made in the U.S. to reduce N and S emissions. However, N and S deposition remains above background levels in many areas, particularly at high elevation (Baron 2006, Weathers et al. 2006).

This research presents modeled spatial and temporal trends in soil and surface water chemistry and critical loads (CL) of atmospheric S and N deposition to protect sensitive biotic resources within the context of changes in future climate and land management. Results presented here are focused on the predominantly forested ecosystems of the southern Appalachian Mountains and Colorado Rocky Mountain subalpine zone. Relatively thin soils derived from parent material inherently low in base cations are common in these regions. As a result, soils and surface waters are often sensitive to acidic N and S deposition which can cause soil nutrient depletion and mobilization of potentially toxic dissolved inorganic aluminum (Al_i; Johnson et al. 1993, Sullivan 2000).

Plant community composition in subalpine zones generally reflects the relatively low soil N supply of these areas. Increased soil N availability can alter competitive relationships among plant species, decrease plant species richness, and adversely impact some rare species in favor of others better able to benefit from increased nutrient supply (Bobbink et al. 2010, Pardo et al. 2011, Bowman et al. 2012). High elevation plant communities are adapted to locally specific temperature and precipitation regimes and are particularly susceptible to climate change (Körner 2003).

Changes in climate can also affect the availability of base cations (BC; Ca^{2+}, Mg^{2+}, K^{+} and Na^{+}) in montane systems. Changes in long-term precipitation patterns can affect drought frequency and associated forest productivity. Changes in forest productivity will affect the rate at which BCs are removed from the soil via plant uptake and subsequently removed from the system by timber harvesting. An increase in the long-term ambient air temperature can also increase or decrease forest productivity. Higher temperature may result in a longer growing season, allowing for increased biomass production. In contrast, increased air temperature could increase evapotranspiration and result in water stress causing a negative effect on growth even if precipitation remains the same. Nutrient cycling effects might also occur if tree species composition changes in response to future changes in climate. Additionally, the amount of runoff that moves through the soil profile affects the rate at which BCs are leached from soil exchange sites. An increase in runoff will result in higher BC leaching with the opposite effect under lower runoff regimes

Acidification has been associated with enhanced leaching of sulfate (SO_4^{2-}) and nitrate (NO_3^-) to drainage waters, depletion of calcium (Ca^{2+}) and other nutrient cations from soil, reduced pH and acid neutralizing capacity (ANC) of surface waters, and increased mobilization of potentially toxic Al_i (Sullivan 2000). Resulting biological effects have included toxicity to fish and aquatic invertebrates and adverse impacts on forest vegetation, especially red spruce and sugar maple trees (U.S. EPA 2009).

Aquatic effects have been especially pronounced in the Adirondack Mountains in New York (Driscoll et al. 2001), Monongahela National Forest in West Virginia (Sullivan and Cosby 2004), Shenandoah National Park in Virginia (Sullivan et al. 2003), and other forested mountainous areas of western Virginia (Cosby et al. 1991). Large increases in hydrogen ion (H^+) and Al_i concentration can be directly toxic to fish, including brook trout (*Salvelinus fontinalis*; Bulger et al. 1999), which is the principal game fish native to surface waters of the Appalachian Mountains. Decreased soil solution

Ca and increased Al_i have been associated with winter freezing damage to red spruce (*Picea rubens*; Cronan and Grigal 1995) and effects on root and stem growth in a wide variety of tree species (Sverdrup and Warfvinge 1993). Reduced growth, poor canopy condition and low regeneration of sugar maple (*Acer saccharum*) trees have also been shown to be associated with soil base cation depletion (Long et al. 2009, Sullivan et al. 2013).

The CL is the level of sustained atmospheric deposition of S, N, or acidity below which significant harm to sensitive ecosystems does not occur according to current scientific understanding (Nilsson and Grennfelt 1988). The CL process typically involves selection of a sensitive receptor, a chemical indicator of biological response for the sensitive receptor of concern, and a critical chemical indicator criteria (or threshold) value that has been shown to be associated with adverse biological impacts. For the sensitive receptor stream water, ANC is most commonly selected as the chemical indicator of effects. Soil base saturation and soil solution Ca/Al ratio are typically used as chemical indicators of terrestrial effects. After a CL has been determined, the CL exceedance is calculated by subtracting the current rate of atmospheric deposition from the estimated CL. Negative values indicate that the CL is in exceedance and ecosystem damages are likely.

Process-based models can be used to evaluate the long-term changes in soil and stream water chemistry and to determine dynamic CL of atmospheric deposition. These dynamic models are data intensive. As such, their use is constrained to individual sites that contain sufficient input data. These process models are often implemented on the scale of individual catchments (100 – 500 ha) and require inputs such as the deposition history, physical soil characteristics, soil and stream chemistry, and estimates of vegetation nutrient uptake and harvesting. The MAGIC (Model of Acidification of Groundwater in Catchments; Cosby et al. 1985) model has been used to reconstruct the history of acidification, simulate future trends, and estimate CL in a large number of catchments in both North America and Europe. This research relied on the MAGIC model to evaluate aspects of soil and stream chemistry at select watersheds located in the southern Appalachian Mountains. Dynamic process-based models can also be used to estimate the CL of N deposition for terrestrial systems based on the extent of expected changes in plant biodiversity. The ForSAFE-Veg model is a process-based model that has recently been developed to evaluate such effects (Belyazid and Moldan 2009). This novel approach combines a biogeochemical model (ForSAFE) with a plant competition model (Veg) to simulate the simultaneous effects of N deposition, acidification and climate change on plant biodiversity. Effects of various future climate and N deposition scenarios on plant community composition at a subalpine site located in the Colorado Rocky Mountains were estimated with the ForSAFE-Veg model in this study.

Steady state models assume unchanging environmental conditions and can also be used to generate CL. Data requirements for steady state models are often available at the regional scale which allows for application to a broader spatial extent than process models. The steady state CL for protection of either aquatic or terrestrial resources can be calculated using a simple mass balance, of which there are several approaches. The most commonly used steady state CL modeling approach for evaluating aquatic systems is the Steady State Water Chemistry (SSWC) model (Henriksen and Posch 2001). A modified version of the SSWC model was used here to generate regional estimates of the CL of S deposition throughout the southern Appalachian Mountain region.

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CHAPTER 2 - EFFECTS OF CLIMATE, LAND MANAGEMENT, AND SULFUR DEPOSITION ON SOIL BASE CATION SUPPLY IN NATIONAL FORESTS OF THE SOUTHERN APPALACHIAN MOUNTAINS

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ABSTRACT

Forest soils having low exchangeable calcium (Ca) and other nutrient base cation (BC) reserves may induce nutrient deficiencies in acid-sensitive plants and impact commercially important tree species. Past and future depletion of soil BC in response to acidic sulfur (S) deposition, forest management, and climate change alter the health and productivity of forest trees. This study used a process model (Model of Acidification of Groundwater in Catchments [MAGIC]) to address a number of questions related to soil BC status for a group of 65 streams and their watersheds in the southern Blue Ridge physiographic province of the southern Appalachian Mountains. Future S deposition to the study watersheds used for the Base Scenario was specified according to proposed reductions in S emissions at the time of this study, representing a reduction of 42% of ambient S deposition by 2020. Twenty additional simulations were considered, reflecting four alternate S deposition scenarios (6%, 58%, 65%, and 78% reduction), and various changes in timber harvest, temperature, and precipitation. Base Scenario soil exchangeable Ca and % base saturation showed decreasing trends from 1860 to 2100. Changes in tree harvesting had the largest effect on stream sum of base cations (SBC) and soil BC supply. Each of the scenario projections indicated that median Year 2100 soil exchangeable Ca will be at least 20% lower than pre-industrial values. The simulations suggested that substantial mass loss of soil BC has already occurred since pre-industrial times. Nearly the same magnitude of BC loss is expected to occur over the next 145 years, even under relatively large additional future reductions in S deposition.

INTRODUCTION

The forested terrestrial and aquatic resources of the Blue Ridge Province of the southern Appalachian (SA) Mountains have provided, and continue to provide, valuable ecosystem services to the American people. The SA Mountains have the largest diversity of plant species in the temperate forest region, and some of these species have been utilized for medicinal, cultural, wood products, and other purposes. The forests provide habitat for numerous terrestrial and aquatic wildlife, with some species considered rare or endangered. The SA Mountains also constitute an important source of drinking water for many communities in the region.

Forest soils within the high elevations of the SA region have developed from the slow breakdown of parent rock material that can be inherently low in base cations (BC), including calcium (Ca), magnesium (Mg), potassium (K), and sodium (Na; Elwood et al. 1991). Combustion of fossil fuels and agricultural production within and upwind of the SA region contribute acidic sulfur (S) and nitrogen (N) deposition to SA forests. These intrinsically low soil BC pools have become further depleted as a result of BC leaching to drainage water in response to acidic deposition that began in the mid-1800s and increased to peak values in the early 1990s. Deposition levels have subsequently decreased over the past two to three decades (Greaver et al. 2012) in response to the Clean Air Act (CAA) and its amendments. Negative effects of low soil BC supply on tree growth and health in the SA Mountains and elsewhere have been documented (Elias et al. 2009, Lovett et al. 2009). Partly as a consequence of past land management and acidic deposition, soil BC depletion now poses a threat to sustainable forest productivity within the region (Huntington 2000, Elias et al. 2009).

Ecosystem sensitivity to acidification and the effects of acidic deposition on surface water quality and nutrient status of forest soils have been well studied in this region, especially at the high elevations (Baker et al. 1990a, 1990b, NAPAP 1991, Johnson et al. 1993, 1999, Sullivan et al. 2004, Elliott et al. 2008). Hydrogen and aluminum (Al) ions replace BC lost to leaching from soil cation exchange sites. The BC mobilized and leached to drainage waters leave soils with depleted stores of nutrient cations. Forest ecosystems within the SA mountains have been shown to be especially sensitive to Ca depletion (Huntington 2000). Soils with decreased exchangeable Ca and other BC reserves may induce nutrient deficiencies in acid-sensitive plant species and cause adverse effects on both commercially important tree species and understory plants. Depletion of soil Ca and increases in available Al in response to acidic deposition have altered the health and productivity of forest trees (Schaberg et al. 2006, Halman et al. 2011, Long et al. 2011).

Trees remove nutrient BC (Ca, Mg, K) from the soil and incorporate them into biomass though uptake and assimilation. After a tree dies, these BC are returned to the soil though decomposition and mineralization, making them available for tree uptake. Because of this internal cycling, it can be assumed that there is little or no net loss of BC from the soil in an unmanaged and unimpacted forest over the long-term. However, harvesting of trees in the SA region, with removal of wood from the site, has permanently removed some nutrient BC from biogeochemical cycling at many locations. A variety of factors affect the magnitude of this BC removal, including species, growth rate, and harvest rotation length. The majority of tree BC are stored in tree boles and it has been estimated that tree harvest in the SA region can remove between 7.7 and 25.5 meq/m^2 of nutrient BC per year (McNulty et al. 2007). However, reduced tree growth in response to any stressor is expected to decrease BC uptake into vegetation (Grier et al. 1989, Högberg et al. 2006).

Other factors can also influence soil BC pools and base saturation (BS), such as ongoing change in the climate of the SA Mountains. There is considerable uncertainty, however, regarding the extent to which air temperature or precipitation may change in the future at any given location. If long-term precipitation decreases, then increased drought frequency could reduce forest productivity (Gholz et al. 1991, Hanson and Weltzin 2000) and leaf longevity (Jonasson et al. 1996). Under water stress, there is increased potential for insect infestation (Mattson and Haack 1987), fire (Flannigan et al. 2000), and tree mortality (Kloeppel et al. 2003). Increased temperature can increase evapotranspiration, contributing to water stress on trees and reduced subsurface water flow to streams (Mitchell et al. 1990, Rosenberg et al. 1990). A large majority of stream flow in forested watersheds originates from water routed through subsurface soils and delivered to streams. The amount of water that moves through the soil profile affects the rate at which BCs are leached from soil exchange sites. An increase in precipitation and subsurface flow will result in higher BC leaching. The opposite effect will occur under lower flow regimes.

An increase in the long-term ambient air temperature can also increase or decrease forest productivity (McNulty and Boggs 2010). Higher temperature may result in a longer growing season, allowing for increased biomass production. In contrast, increased air temperature could increase evapotranspiration and result in water stress causing a negative effect on growth even if precipitation remains the same. Furthermore, each tree species contains a unique set of nutrient uptake and environmental requirements to maintain its presence on a site (McNulty et al. 2007, Iverson et al. 2008). Nutrient cycling effects might occur if tree species composition changes in response to future changes in climate.

An understanding of the historical loss of soil BCs and expected future base status of soils under various climate, management, and S deposition scenarios is needed by forest managers responsible for maintaining healthy and productive forests in this region. The principal objectives of the study reported here were to use the Model of Acidification of Groundwater in Catchments (MAGIC; Cosby et al. 1985) for a group of 65 streams and their watersheds in the southern Blue Ridge physiographic province of the SA Mountains to address a number of questions related to soil BC status, including:

- What has been the historical rate of soil BC loss due to leaching in response to anthropogenic S deposition?
- How will decreases in the amount of acidic deposition affect soil BC concentrations in the future?
- Can S deposition be reduced to a level that will allow soil BC pools to recover from historical BC losses?
- What is the expected effect of a changing climate on future soil BC status?
- How will changes in timber harvesting affect soil BC status?
- What amount of BC fertilization will be needed to restore BC pools and recover ecosystem services?

This study is unique among previous applications of MAGIC in the SA region because it is focused on soil condition as it relates to forest health and productivity rather than stream chemistry and associated effects on aquatic biota. Additionally, changes in drivers of BC supply other than S deposition are included in model scenarios. These drivers of BC supply can interact to compound or negate effects and such interactions are addressed. However, the primary objective was to evaluate the potential drivers on an individual basis. The focus of this study was on relatively acid-sensitive watersheds within the SA region. These watersheds are thought to be generally representative of the surrounding national forest land area where forest management actions may be required in order to partly or fully recover historical BC pools and prevent future loss of soil BC caused by continued elevated atmospheric S deposition, as influenced by changes in climate.

METHODS

Study Sites

The study sites included 65 watersheds for which data were available for calibrating the MAGIC model within the Cherokee, Nantahala, Pisgah, and Sumter National Forests (Figure 2-1). These forests are located in the SA Mountains within the Blue Ridge physiographic province. Modeled sites and calibration data were described in the modeling study of Sullivan et al. (2011). At least one watershed was modeled in three of the four federally mandated (according to the CAA amendments of 1977) Class I areas that occur within the study region (Joyce Kilmer-Slickrock, Linville Gorge, and Shining Rock Wilderness) as well as the Class II areas, Ellicott Rock, Middle Prong, Sampson Mountain, and Southern Nanatahala Wilderness. Class I areas receive the highest level of federal protection from air pollution degradation.

Potential modeling sites were pre-screened to remove from consideration streams having high concentration of chloride (Cl^-; > 70 µeq/L) that could have been caused by road salt application, and streams having high concentrations of NO_3^- (> 30 µeq/L) that could have been caused by agricultural or silvicultural fertilization within the watershed. The potential for such anthropogenic disturbance, other than air pollution, was determined by examination of stream chemistry and location of roads, wilderness areas, and agricultural or forestry operations within the watersheds. One site included in the earlier study (Sullivan et al. 2011) was deleted from consideration in this study because of evidence that the modeled stream received geological contributions of S. Samples were also pre-screened to remove sites for which the observed percent mineral soil base saturation (% BS) was > 60%. Such high values of % BS probably represent a sampling or analysis error, or reflect a local (and unrepresentative) heterogeneity in the geology and/or soil matrix at the sampling site (Sullivan et al. 2011).

Land cover in the 65 watersheds is forested, with no agriculture or urban development. Dominant vegetation consists of mountain balds, northern hardwoods, and northern red oak communities. Spruce-fir communities can be found at high elevation (> 1060 m) with hemlock and mixed oak-pine forests occurring at lower elevation. The geology is predominantly granite, sandstone and other rock types that exhibit relatively low BC weathering (Sullivan et al. 2007) and soils are generally shallow (< 1 m; NRCS 2010).

MAGIC Calibrations

MAGIC predicts the long-term effects of acidic deposition and land management on soil and surface water chemistry. A central component of MAGIC is the pool of exchangeable base cations on the soil. As the fluxes to and from this pool change over time due to changes in atmospheric deposition and/or removal via timber harvest, the chemical equilibria between soil and soil solution shift to simulate changes in drainage water chemistry. The influence of naturally occurring organic acids on stream chemistry is included in the model as a triprotic analogue (Driscoll et al. 1994), although there is no provision in the model for changes in organic acidity over time.

The validity of the model has been confirmed by comparison with estimates of surface water acidification inferred from paleolimnological reconstructions (Jenkins et al. 1990a, Wright et al. 1994, Sullivan et al. 1996) and with the results of several experimental watershed acidification and de-acidification experiments (e.g., Wright and Cosby 1987, Cosby et al. 1995, Cosby et al. 1996, Moldan and Wright 1998). MAGIC has been used to reconstruct the history of acidification and to simulate future trends on a regional basis and in a large number of individual catchments in both North America and Europe (e.g., Lepistö et al. 1988, Hornberger et al. 1989, Cosby et al. 1990, Jenkins et al. 1990a, Wright et al. 1994, Sullivan et al. 2008).

The aggregated nature of the model requires that it be calibrated to observed data from a watershed before it can be used to examine potential system response. Calibration is achieved by specifying the values of fixed parameters within the model that can be directly measured or observed. The model is then run using observed and/or assumed atmospheric and hydrologic inputs. Simulated stream water and soil chemical criterion variables are compared to observed values. If the observed and simulated values differ, the values of another set of parameters in the model (called "optimized" parameters) are adjusted to improve the fit. After a number of iterations, the simulated-minus- observed values of the criterion variables usually converge to zero, within some specified tolerance. The model is then considered calibrated and can be used for hindcasting and future scenario modeling. Because estimates of fixed parameters and deposition inputs are subject to uncertainties, a "fuzzy optimization" procedure is implemented during the calibration process. It involves multiple calibrations using random values of the fixed parameters drawn from the observed possible range of values and random values of deposition from a range including uncertainty about the estimated values. Each calibration begins with random selection of values of fixed parameters, deposition, and starting values of the optimized parameters. The optimized parameters are then adjusted using the Rosenbrock (1960) algorithm to achieve a minimum error fit to the target variables. This procedure is undertaken ten times. The final calibrated model is represented by the ensemble of parameter and variable values of the ten calibrations. Projections are reported based on the results derived from the median calibration. The acid–base chemistry modeling for this project was conducted using 2005 as the base year, as this was the year for which the input data were available. MAGIC was calibrated to the available atmospheric deposition, soil chemistry, and water chemistry data and then interpolated or extrapolated to yield base year estimates of stream water chemistry in the year 2005, which served as the starting point for each future scenario.

MAGIC scenario modeling was conducted to estimate the effects on soil chemistry of a variety of future S deposition, management, and climatic conditions. It involved 21 different scenarios, including one Base Scenario and 20 scenarios that altered future conditions. The Base Scenario was run under current management and climate regimes with a reduction in S deposition reflective of current emissions regulations that have been proposed to be phased in by the year 2020. Each of 19 of the additional scenarios modified only one aspect of the Base Scenario in an attempt to isolate the potential effects of future change in either S deposition, stream flow, forest productivity, tree harvest rate, or tree harvest area as described below and in Table 2-1. One final scenario was implemented as a representation of aggressive management, involving the maximum reduction (78%) in atmospheric S deposition considered here and allowing no harvest on any land. Results of this scenario reflect the extent of improvement in base cation supply that might be achievable by aggressively managing S emissions and timber harvest.

Hindcast Simulations

The MAGIC model was used to develop a hindcast simulation reflecting how stream and soil chemistry were expected to have responded to historical changes in S deposition over the period 1860 to 2005. The Advanced Statistical Trajectory Regional Air Pollution Model (ASTRAP; Shannon 1998) was used to provide historical estimates of wet, dry, and cloud deposition of sulfur and nitrogen oxide in and around the study region using a nearest-neighbor approach that included correction for elevation, as described by Sullivan et al. (2004). Shannon (1998) produced wet, dry, and cloud deposition estimates of sulfur and nitrogen oxides every ten years starting in 1900 and ending in 1990. The model outputs were smoothed estimates of deposition roughly equivalent to a ten-year moving average centered on each of the output years. To estimate total deposition from wet deposition, the wet, dry, and cloud deposition estimates provided by ASTRAP for each year were used to calculate dry plus cloud deposition enhancement factors (DDF) for each year and each site. This provided time series of DDF for sulfur and nitrogen oxides for each site extending from 1900 to 1990. These were combined with linear estimates of changes in deposition between 1860 and 1900 from assumed background values. The value of DDF for 1990 was used as the absolute value of DDF for the reference year and was assumed to remain constant in the future.

At high elevation, the inputs of ions from cloud water can be large. In the SAMI project (see Sullivan et al. 2002a, 2002b, 2004), high elevation sites in the Great Smoky Mountain National Park (GSMNP) were determined to have DDF values (reflecting dry and cloud, but particularly cloud water inputs) that were approximately twice as large as those specified by the ASTRAP model. Accordingly, Sullivan et al. (2004, 2011) used the larger DDF values for any site over 1500 m elevation. In this project, the five sites that were over 1500 m elevation were assigned the higher DDF values used by Sullivan et al. (2004). The hindcast was performed using BC uptake rates as described for the Base Scenario below.

Future Base Scenario

Sulfur Deposition

The future rate of S deposition to the study watersheds used for the Base Scenario was specified according to proposed reductions in S emissions at the time of this study, as described by Sullivan et al. (2011). Base scenario deposition represents a reduction of 42% of ambient S deposition phased in between 2010 and 2020, followed by constant deposition from 2020 to the end of the simulation period in 2150.

Forest Management

Some of the study watersheds on National Forest ownership exist largely within areas that are designated as suitable for tree harvest. Suitable land includes areas for which timber harvest (typically bole-only) is allowed according to the land management plan for a particular national forest. Timber harvesting was simulated in all watersheds that contained suitable land (n = 33). Removal of BC from harvesting was approximated with a 65 percent biomass (bark and bole) harvest rate and estimated species-specific BC uptake rates between 7.7 and 25.5 meq/m^2/yr (McNulty et al. 2007).

Simulations of timber harvesting in the Base Scenario only occurred for those portions of the watershed classified as suitable for timber harvesting. The forest land management plans also include other national forest ownership designated as unsuitable for timber harvesting. These include areas designated as wilderness and other areas designated as unsuitable for reasons identified in the forest plan. For example, the Pisgah and Nantahala land management plan lists unsuitable lands that have one or more of the following characteristics:

- The site has threatened, endangered, or sensitive species, and proposed site impacts may affect species viability.
- The site cannot be adequately restocked (regenerated) with a new stand of trees.
- Access is not possible due to terrain and/or state and/or private ownership patterns.
- Unique habitats such as seeps, bogs or rock outcrops exist.
- The area has been designated to occur within the riparian management area.
- Special uses such as power line corridors and access right of ways exist.

Climate

Future changes in air temperature and precipitation patterns are expected to occur globally. It is anticipated that temperatures will increase in the SA mountain region. However, there is uncertainty regarding whether the long-term annual precipitation will increase or decrease (Karl et al. 2009). MAGIC can account for changes in temperature or precipitation by adjusting the simulated forest productivity and stream flow rates. For the Base Scenario, forest productivity and stream flow were assumed to remain constant at values consistent with ambient conditions.

Alternate Future Scenarios

Sulfur Deposition

Four alternate future S deposition scenarios were considered (Table 1). One scenario (Dep4) included a smaller reduction (6% decrease relative to year 2005) in S deposition than the expected changes based on proposed emissions control policy. Each of the other three deposition scenarios reflected further reductions in S deposition beyond those expected to occur in response to proposed emissions control policy (Dep1, Dep2, and Dep3). These scenarios represent 58%, 65%, and 78% reduction, respectively, in ambient S deposition compared to reference year (2005) deposition, phased in between 2010 and 2020, with constant deposition assumed from 2020 to the end of the simulation period.

Forest Management: Biomass Removal Rate

Potential future tree removal rates were selected to bracket the Base Scenario rate of 65% ± 20%, with scenarios of 45% and 85% removal. Two additional scenarios represented extreme tree removals of 0% (no removal) and 100% (removal of all bark and bole; Table 2-1) from all land considered suitable for timber harvest.

Forest Management: Land Management Classification

Under current forest management practices, the Base Scenario simulated harvesting only on the land area that is considered to be suitable for harvesting by the Forest Service. Occasionally, timber harvesting in unsuitable areas outside of a wilderness may occur because of large-scale tree mortality caused by a severe storm event, drought (Kloeppel et al. 2003), insect infestation (Mattson and Haack 1987) or disease outbreak. Tree harvest in these situations is typically described as a salvage operation. It should be noted that under current policies the U.S. Forest Service is unlikely to conduct any timber harvesting or salvage operations in designated wilderness areas of the SA Mountains.

The unsuitable and wilderness areas are commonly found at the highest elevations or ridge tops and can be sensitive to BC removal resulting from acidic deposition. In this study, tree BC removal due to timber harvest was simulated to estimate effects of harvesting on only the suitable lands. The alternative future scenarios based on land management classification included 1) no harvest on any land (equivalent to 0% biomass removal listed above), 2) harvest on both suitable and non-suitable land, and 3) harvest on all forest areas including wilderness (the extreme case).

Climate: Productivity

The Intergovernmental Panel on Climate Change (IPCC) estimated that average global surface temperature will likely rise by 2° to 6° C above 1990 levels by 2100 (IPCC 2007). The spatial distribution of the projected warming is uncertain and some areas may even exhibit a cooling pattern. Further, the effects of a warmer or cooler climate on forest productivity are unknown, although higher temperature will most likely increase productivity. To accommodate this variability and uncertainty, model simulations specified changes of +/- 5% and +/- 15% in forest productivity, as influenced by air temperature (Table 2-1).

Climate: Precipitation and Stream Flow

Along with expected long-term changes in air temperature, the IPCC estimates that future precipitation patterns will also change. Wetter or dryer conditions are expressed in MAGIC through changes in the estimates of stream flow. More/less precipitation will result in more/less water movement through the soil profile and resultant stream flow. Changes of ± 4% and ± 10% in stream flow were simulated to evaluate the effects of potential future changes in precipitation regimes on BC supply (Table 2-1).

RESULTS

Base Scenario

Model results for pre-industrial, ambient, and future soil and stream chemistry under the Base Scenario are presented in Figure 2-2. Soil exchangeable Ca and BS showed decreasing trends throughout the duration of the simulation from 1860 to 2100, while soil pH had a decreasing trend between 1860 and 2005 with little simulated change between 2005 and 2100. These results reflect, in addition to BC inputs via weathering and atmospheric deposition, the removal of BC in harvested trees and BC leaching in response to continued S deposition. The simulated sum of BC (SBC) concentration in stream water increased during the simulation from 1860 to 2005, followed by a decrease which continued to the year 2100. There was also an increase in stream sulfate (SO_4^{2-}) concentration from 1860 to 2005 remaining nearly constant to year 2100.

Ambient acid neutralizing capacity (ANC) of stream water was nearly 30% lower than simulated ANC during the pre-industrial period (Figure 2-2). This trend of simulated decreasing ANC continues to the year 2100, in large part due to decreased S adsorption on soils over time (Turner et al. 1990). As the S adsorption capacity of the soils becomes exhausted, SO_4^{2-} leaching is expected to increase, even under lower levels of atmospheric S deposition (Sullivan et al. 2011).

Each of the modeled watersheds, except one, showed a decrease in soil BC since pre-industrial times. Model results suggested that soil nutrient pool reductions will continue to occur in the future under the Base Scenario (Figure 2-3). For each nutrient base cation, the exchangeable cation pool is expected to continue to decline in the future, in some watersheds to more than a 50% decrease compared with the simulated pre-industrial condition.

Alternative Future Scenarios

Departures from the year 2100 median (across all 65 modeled sites) Base Scenario results for stream ANC and SBC and for soil BS and exchangeable Ca are shown in Figure 2-4 for each of the alternative future scenarios. The largest departure from the Base Scenario for stream ANC was for the simulation based on the Dep4 scenario (CAA Title IV [6%] reduction in S deposition). The largest departures from the Base Scenario for stream SBC and soil BC supply were found for simulations that were based on the two most inclusive harvesting scenarios (all non-wilderness areas harvested and all areas including wilderness harvested). Stream SBC concentrations were also moderately sensitive to simulated changes in stream flow, and stream ANC was moderately sensitive to future reductions in S deposition.

Soil BS and exchangeable Ca were relatively insensitive to simulated changes in the productivity scenarios (which reflected long-term changes in temperature), stream flow, and the highest simulated harvest removal rates of 85% and 100%. However, the more extreme scenarios representing substantially decreased stream flow and harvest removal rate did suggest somewhat larger changes in soil chemistry in response to these perturbations.

The distribution of simulated percent reduction in upper B horizon soil exchangeable Ca from 1860 to 2100 is shown in Figure 2-5 for each of the alternative future scenarios. The scenarios are ranked from lowest to highest according to median simulated reduction. Although only a small number of alternative future scenarios suggested that soil exchangeable Ca might increase appreciably in the future relative to current Base Scenario conditions (Figure 2-4), each of the scenario projections indicated that median Year 2100 soil exchangeable Ca will be at least 20% less than pre-industrial values (Table 2-2, Figure 2-5). Calcium loss is projected to be especially high (more than 50%) for scenarios 19 and 20.

DISCUSSION

Historical and Future Simulations

The impacts of acidic deposition on soil exchangeable Ca and BS, and on stream ANC and SBC have reduced the ecosystem services provided by terrestrial and aquatic ecosystems on national forest lands in the SA Mountains (see report by Sullivan and

McDonnell [2012] on the central Appalachian Mountain region). These include the presence and abundance of brook trout, aquatic species diversity, and likely a range of benefits associated with the presence, health, and growth of red spruce and sugar maple. In addition, historical timber harvests have also reduced soil BC pools, especially in the high elevation lands currently managed as wilderness or otherwise now considered to be unsuitable for harvesting. The simulated increase in stream SBC concentrations from the pre-industrial to the ambient (2005) time period was due to BC leaching from the soils in response to SO_4^{2-} and NO_3^- leaching (Figure 2-2). Furthermore, the simulated decrease in SBC leaching from ambient conditions to the year 2100, as reflected by decreasing SBC concentrations in stream water (Figure 2-2), is a reflection of soil BC depletion, whereby SO_4^{2-} leaching will more commonly be charge balanced by H^+ and Al^{n+} in the future rather than BC. This simulated deterioration of soil BC status began with the harvesting of timber in the early 1900s. The rate of soil BC loss was accelerated by S, and to a lesser extent N, deposition caused largely by the combustion of fossil fuels, reduced S adsorption on soil, and increased desorption of historically retained S.

For the soils considered in this study, substantial mass loss of soil BC has occurred since pre-industrial times (Figures 2-2, 2-6a). Under the Base Scenario, nearly the same magnitude of BC loss is expected to occur over the following 145 years (Figure 2-6b), even under substantial additional future reductions in S deposition. These results represent BC losses in the mineral soil and are based on the soil depth and bulk density used in the MAGIC calibrations along with an assumption that modeled changes in exchangeable BC supply are consistent throughout mineral soil horizons. Median mineral soil BC loss was estimated to be 53, 25, and 28 kg/ha from 1860 to 2005 for Ca, Mg, and K, respectively. These results suggest that the rate of BC deposition and BC weathering have not been sufficient to maintain pre-industrial mineral soil exchangeable BC pools. None of the future individual scenarios considered here are expected to completely recover these median watershed BC losses by the year 2100 and exchangeable Ca under all scenarios evaluated is predicted to remain at least 20% reduced from the historical supply for the median watershed by the year 2100 (Figure 2-5). Under the Base Scenario, the additional median loss of Ca, Mg and K from 2005 to 2150 is expected to be 41, 27 and 46 kg/ha, respectively.

The primary goal of this study was to evaluate potential drivers of soil BC supply on an individual basis. It is recognized that these drivers of BC supply can interact to compound or negate effects and such interactions were addressed by evaluating a combination of two scenarios (Dep3 and Mng1) to reflect the combined effect of the most aggressive management scenarios. The interaction effects were focused on these management scenarios because forest managers have no control over changes in future climate and climate effects on BC supply were relatively small compared to potential changes reflected in the S deposition and forest management scenarios (Figure 2-4). Even under a scenario of aggressive management (decreased S deposition by 78% and no future harvest), soil base cation supply was not simulated to recover to pre-industrial exchangeable Ca conditions**.** Although specific interactions between climate change and management scenarios were not modeled, modeled changes in soil and stream acid-base chemistry (Figure 2-4) may approximately be additive. For example, the effect on exchangeable Ca of increasing the harvest removal rate to 100% might be offset by a change in stream flow of -10%.

In this study, MAGIC was used to evaluate a number of alternative future scenarios to evaluate extreme conditions, understanding that many of the scenarios are not likely to occur in the future. Proposed state and federal air regulations in the eastern United States will likely further reduce S deposition beyond the 6% assumed to occur as a result of Title IV of the CAA amendments of 1990 (Dep4 scenario – Table 2-1). Emissions reductions are anticipated in the future as plans to improve air quality and visibility are implemented according to the national Regional Haze Rule. Proposed additional emissions reductions in the Clean Air Interstate Rule (CAIR) and the Cross-State Air Pollution Rule (CSAPR) have called for an additional approximately 42% reduction in S deposition. Nevertheless, the model simulations suggest that stream ANC, and soil BS and exchangeable Ca would be expected to continue to decline. Additional S deposition reductions at or above 58% are expected to provide soil and stream chemistry improvements relative to what is expected under the Base scenario. Further reductions in S deposition may occur with either additional S emissions controls or with continued replacement of coal-fired power plants with natural gas, biomass, or other alternative energy sources for electricity generation in and upwind of the region.

Timber harvesting in wilderness and other lands classified as unsuitable for harvest was also evaluated, even though it is not expected that this would ever occur. This extreme scenario was evaluated to ascertain the sensitivity of the model projections to harvesting level. Results show the potential for significant impacts of timber harvesting on the acid-base chemistry of streams and soils (Figure 2-4). Land managers may decide to implement management strategies that maintain dead trees on unsuitable lands subsequent to large-scale blowdown, ice storm damage, or insect/disease outbreak, rather than removing bark and boles from the sites in commercial salvage operations. In wilderness areas, existing management policies dictate that there will be no timber harvesting or salvage operations even if there is widespread tree morality. Additional timber harvesting was shown to have the most impact on exchangeable Ca supply (Table 2-2). Since increased rates of timber harvest are not likely in this region, land managers should recognize that future BC supplies may be most strongly affected by changes in precipitation and subsequent soil water movement to streams (Figure 2-4). However, these effects are projected to be relatively minor, with a ±10% change in stream flow corresponding to approximately a ±0.7% change in BS and a ±0.1 meq/kg change in exchangeable Ca.

Results of the individual model scenarios suggested that changes in existing forest harvest practices, climate, or additional reductions in S deposition are not expected to recover historical BC losses prior to the year 2100 (Table 2-2). Complete cessation of tree harvesting was most effective at recovering soil Ca supply, indicating that forest management practices have the largest control on the future of SA forest health as it relates to soil BC status among the scenarios considered. Large (78%) future reductions in S deposition are expected to result in increased BC supply relative to the Base Scenario. However, if these two aggressive management scenarios were to occur simultaneously the combined effect would remain insufficient to prevent further BC losses and recover historic losses (data not shown).

Base Cation Fertilization

Managers can also consider fertilization of watershed soils with BC to ameliorate the historic loss of soil nutrients, restore BS, and prevent future BC losses. For example, land managers may want to consider mitigating the impacts on nutrient base cation supply associated with future timber harvesting by fertilizing the forests to replace BC lost with removal of the bark and boles from the watershed. However, it will be necessary to consider the feasibility of fertilizer application along with potential positive and negative effects of BC fertilization prior to implementation of such a management strategy.

Application of BC fertilizers has been used as a method for restoring exchangeable BC to base-depleted European and North American forest soils (e.g., Huettl 1989, Côtè et al. 1993, Kreutzer 1995, Nohrstedt 2002, Sharpe and Voorhees 2006, Pabian and Brittingham 2007, Cho et al. 2010, Moore and Ouimet 2010, Long et al. 2011). Nutrient base cations have been applied in various forms, including dolomite, wollastonite, and potash of sulfur. Application of BC fertilizer has generally been associated with significant positive effects on soil nutrient status as well as variety of terrestrial ecological metrics including snail and bird abundance, forb ground cover, and tree health (Huettl 1989, Wilmot et al. 1996, Long et al. 1997, Halman et al. 2008, Pabian et al. 2012).

Whole watershed BC fertilization is a potentially viable option for re-establishing pre-industrial BC pools and preventing further BC losses in the SA as well. The interquartile range of simulated BC loss from 1860 to 2150 among the modeled watersheds was 52 to 201 kg Ca/ha, 32 to 99 kg Mg/ha and 44 to 145 kg K/ha. Although some of the modeled watersheds were estimated to exhibit larger BC losses, successful BC fertilization at these rates would recover and prevent BC losses at approximately 75% of the modeled sites.

A liming study was conducted in the Linville Gorge Wilderness (LGW) with an application rate of 247 kg Ca/ha and 129 kg Mg/ha as dolomitic lime on forest plots recently subjected to wildfire (Elliott et al. 2012). Average soil chemistry data one to two years after liming indicated that the moderately burned site with dolomite application showed 15 times more Ca in the upper (0 – 10 cm) mineral soil as compared with the reference site. The Ca and Mg fertilization rates used in the Elliott et al. (2012) study are similar to the estimated losses of these nutrients in the modeled watersheds of this study. However, the fertilization effect of wildfire could not be separated from the effect of the liming application in the LGW study.

Other studies have also shown positive effects of BC fertilization on base-depleted soils. Significant increases in soil BC pools were observed in organic soils after fertilization with 155, 33 and 255 kg/ha of Ca, Mg and K, respectively, in sugar maple stands located in southern Quebec (Côtè et al. 1993). Calcium fertilization with wollastonite ($CaSiO_3$) was applied in an experimental watershed at Hubbard Brook Experimental Forest, New Hampshire (Cho et al. 2010). It was determined that 30.2 tons of wollastonite would be required to raise whole solum BS from 10% to 19% in the 11.8 ha watershed (~850 kg Ca/ha). The actual amount applied was 1,316 kg Ca/ha to account for potential losses and inefficient dissolution and mobility within the soil profile. Marked increases in exchangeable Ca in organic soils were observed following Ca fertilization and mobility within the soil profile. However, mineral soil exchangeable Ca was largely unaffected in the short term at the low to mid watershed elevations. In contrast, a doubling of upper mineral soil exchangeable Ca was observed in soils of a high-elevation red spruce forest three years after the Ca addition (Cho et al. 2010).

Fertilization experiments in a Norway spruce forest located in SW Sweden examined the effects on soil nutrient status of various forms and amounts of BC fertilizer application (Nohrstedt 2002). Addition of 1,135 kg/ha of Ca and 128 kg/ha of Mg, comprised of equal parts crushed calcite and dolomite, caused a 297% increase in exchangeable Ca and Mg in organic soils. There were also 66% and 15% increases in exchangeable Ca + Mg in the top 5 cm and the 5-10 cm depth of mineral soil, respectively, four years after application. The author concluded that movement of Ca and Mg from the soil surface to deeper mineral soil horizons is a slow process that occurs over a period of multiple years. Potassium chloride applied at rates of both 79 and 158 kg K/ha did not result in any appreciable effect on exchangeable K in either organic or mineral soils four years after application. The reason for this result was not known, although it was postulated that it was due to the combined effects of K leaching and plant uptake (Nohrstedt 2002).

Potential drawbacks of BC fertilization may include increased nitrification and associated increased NO_3^- leaching, induced K (in the case of Ca and/or Mg application) or boron deficiency, mobilization of heavy metals, and enhanced fine root development in the surface soil horizons. The latter effect could contribute to increased risk of damage to trees by windthrow, frost or drought (Huettl and Zoettl 1993, Kreutzer 1995), which may be important concerns under a changing future climate. It is not known whether any of these possible effects of BC addition are important in forests in the SA mountain region. Because forest liming entails ecological risks, a careful analysis concerning the need for liming should be conducted in any potential treatment area (Huettl and Zoettl 1993).

Consideration of the form(s) and dose(s) of BC fertilizer can optimize benefits and minimize detriments to forest ecosystems. Given the uncertainly in the movement of surface applied BC fertilizer into mineral soils, it is recommended to develop pilot studies to evaluate the efficacy of BC fertilization on mineral soil nutrient pools in SA forests prior to engaging in widespread fertilizer application in the SA region. Land managers and others need to be aware, however, that the model simulations suggest about 40% of the sites in this study had $\leq$ 10% BS in 1860 and therefore they should be cautious with BC fertilization so that the resulting base cation supplies will not be above the historical levels.

CONCLUSIONS

The relatively acid-sensitive watersheds of the SA region evaluated in this study were considered to be generally reflective of the surrounding national forest land area where forest management actions may be required in order to restore the loss of ecosystem services caused by continued elevated atmospheric S deposition. Soil BC pools in the study region are expected to remain significantly below pre-industrial conditions for hundreds of years into the future, regardless of changes in climate, S deposition, or timber harvest. Forest management actions such as BC fertilization will be required to fully recover BC supply to historical conditions in a time frame relevant to forest managers. Results presented here can be used as the basis for establishing BC fertilization rates in pilot studies to consider the effectiveness of fertilizer application for the recovery of soil BC supply in the SA region.

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TABLES

Table 2-1. Description of scenarios simulated using the MAGIC model for this study.

Type	Scenario ID	Value	Description
Deposition	Dep4	Title IV (-6%)	6% reduction in 2005 sulfur deposition
	Base	Base (-42%)	42% reduction in 2005 sulfur deposition
	Dep1	Scen. A (-58%)	58% reduction in 2005 sulfur deposition
	Dep2	Scen. B (-65%)	65% reduction in 2005 sulfur deposition
	Dep3	Scen. C (-78%)	78% reduction in 2005 sulfur deposition
Climate: Productivity	Clm2	-15%	Large decrease in forest productivity (-15%)
	Clm1	-5%	Small decrease in forest productivity (-5%)
	Base	Base (ambient)	No change in productivity
	Clm3	+5%	Small increase in forest productivity (+5%)
	Clm4	+15%	Large increase in forest productivity (+15%)
Climate: Stream flow	Clm6	-10%	Large decrease in stream flow (-10%)
	Clm5	-4%	Small decrease in stream flow (-4%)
	Base	Base (ambient)	No change in stream flow
	Clm7	+4%	Small increase in stream flow (+4%)
	Clm8	+10%	Large increase in stream flow (+10%)
Management: Harvest Area	Mng1	none	No tree harvesting
	Base	Base (suitable)	Tree harvesting in suitable areas
	Mng2	non-wild	Tree harvesting in suitable and unsuitable nonwilderness areas
	Mng3	all	Tree harvesting in suitable, unsuitable, and wilderness areas
Management: Removal	Mng4	0%	Remove 0% of bark and bole
	Mng5	45%	Remove 45% of bark and bole
	Base	Base (65%)	Remove 65% of bark and bole
	Mng6	85%	Remove 85% of bark and bole
	Mng7	100%	Remove 100% of bark and bole

Table 2-2. Distribution among the 65 modeled watersheds of the simulated percent change in exchangeable Ca from 1860 to 2100 for each scenario, ranked by median change among the modeled watersheds.

	Scenario Description			Change in Exchangeable Ca (Percentile)						
Rank	ID	Perturbation	Value	Minimum	5th	25th	Median	75th	95th	Maximum
1	Mng1	Harvest Area	None	80.2	56.3	-4.0	-20.9	-41.0	-56.0	-73.0
2	Mng4	Removal	0%	80.2	56.3	-4.0	-20.9	-41.0	-56.0	-73.0
3	Clm6	Stream flow	-10%	19.9	2.0	-10.2	-22.0	-36.4	-50.9	-69.2
4	Dep3	Deposition	Scen. C (-78%)	1.3	-10.0	-16.6	-24.5	-39.2	-53.4	-68.3
5	Mng5	Removal	45%	18.7	-0.1	-12.2	-24.9	-41.3	-56.0	-73.0
6	Clm2	Productivity	-15%	8.7	-8.6	-16.7	-25.2	-42.5	-56.5	-73.0
7	Dep2	Deposition	Scen. B (-65%)	0.8	-10.6	-17.2	-26.0	-40.5	-54.4	-69.9
8	Clm5	Stream flow	-4%	7.2	-6.4	-15.1	-26.0	-39.9	-54.7	-71.6
9	Dep1	Deposition	Scen. A (-58%)	0.3	-10.7	-17.6	-27.0	-41.2	-55.0	-70.8
10	Clm1	Productivity	-5%	2.6	-10.8	-18.0	-27.2	-43.0	-57.0	-73.0
11	Base	Base	Base	-0.5	-11.4	-18.2	-29.0	-43.1	-57.0	-73.0
12	Clm3	Productivity	+5%	-3.6	-11.4	-19.3	-29.1	-43.6	-57.0	-73.0
13	Dep4	Deposition	Title IV (-6%)	-3.1	-12.5	-21.1	-31.6	-47.2	-61.6	-77.5
14	Clm7	Stream flow	+4%	-5.2	-15.5	-21.4	-31.6	-45.8	-59.1	-74.3
15	Clm4	Productivity	+15%	-4.0	-11.4	-21.4	-33.7	-45.0	-57.0	-73.0
16	Clm8	Stream flow	+10%	-6.9	-19.7	-26.3	-36.2	-49.3	-62.1	-76.0
17	Mng6	Removal	85%	-4.0	-11.9	-21.4	-36.8	-47.5	-60.3	-73.0
18	Mng7	Removal	100%	-4.1	-11.9	-23.1	-41.3	-52.1	-72.7	-80.8
19	Mng2	Harvest Area	Non-wild	-1.5	-13.5	-27.0	-55.2	-77.5	-100.0	-100.0
20	Mng3	Harvest Area	All	-1.5	-24.6	-44.0	-68.1	-83.8	-100.0	-100.0

FIGURES

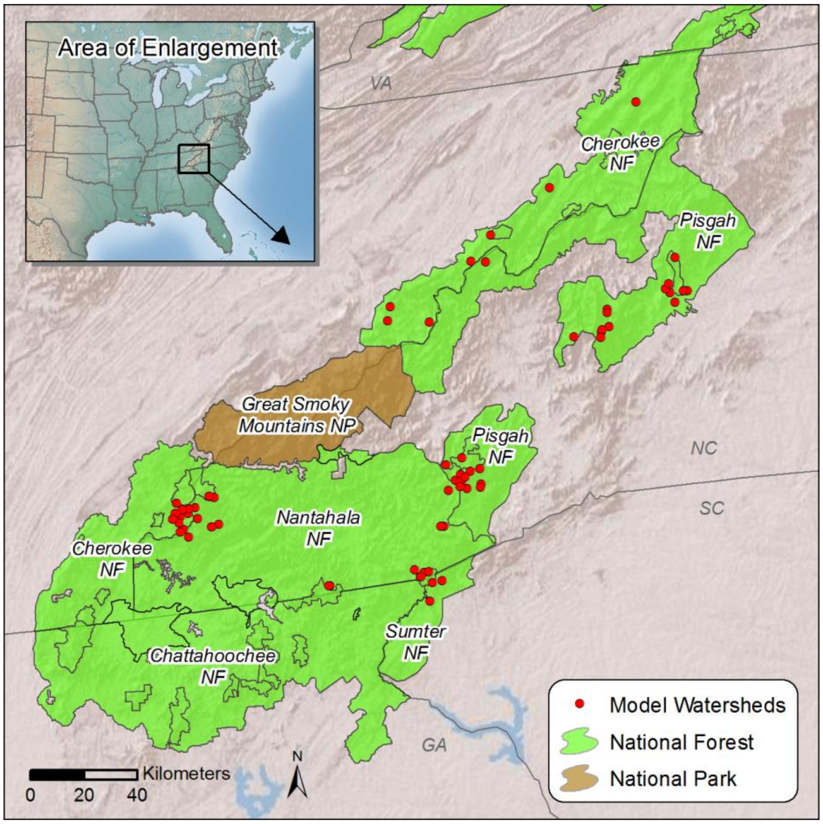

Figure 2-1. Map showing study area and location of modeled sites.

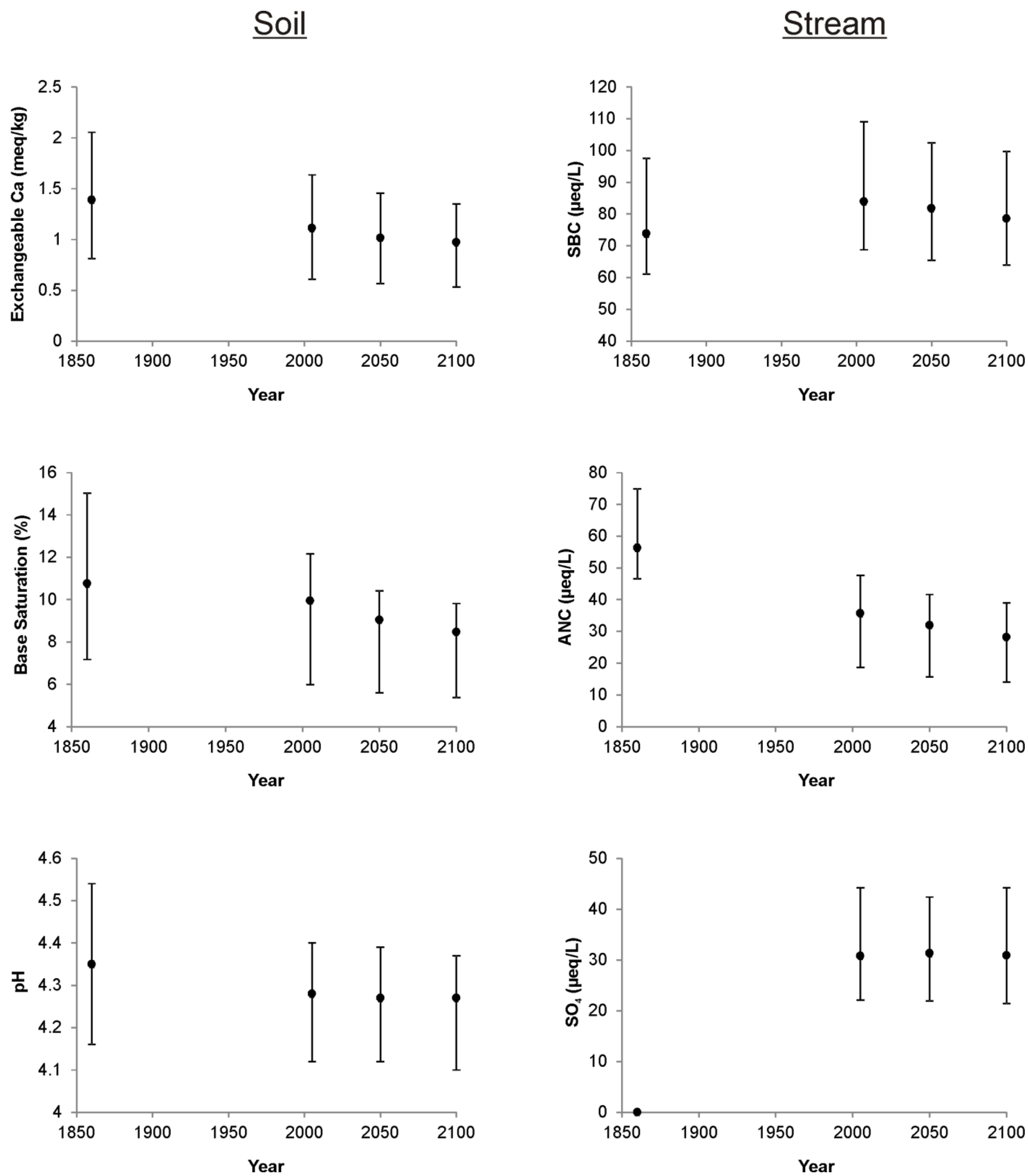

Figure 2-2. MAGIC model simulation results for upper B horizon soil (left panel) and stream water (right panel) for selected years between 1860 and 2100 under the Base Scenario. Data are presented as median (dot) and quartile (bars) simulation results averaged across the 65 modeled sites.

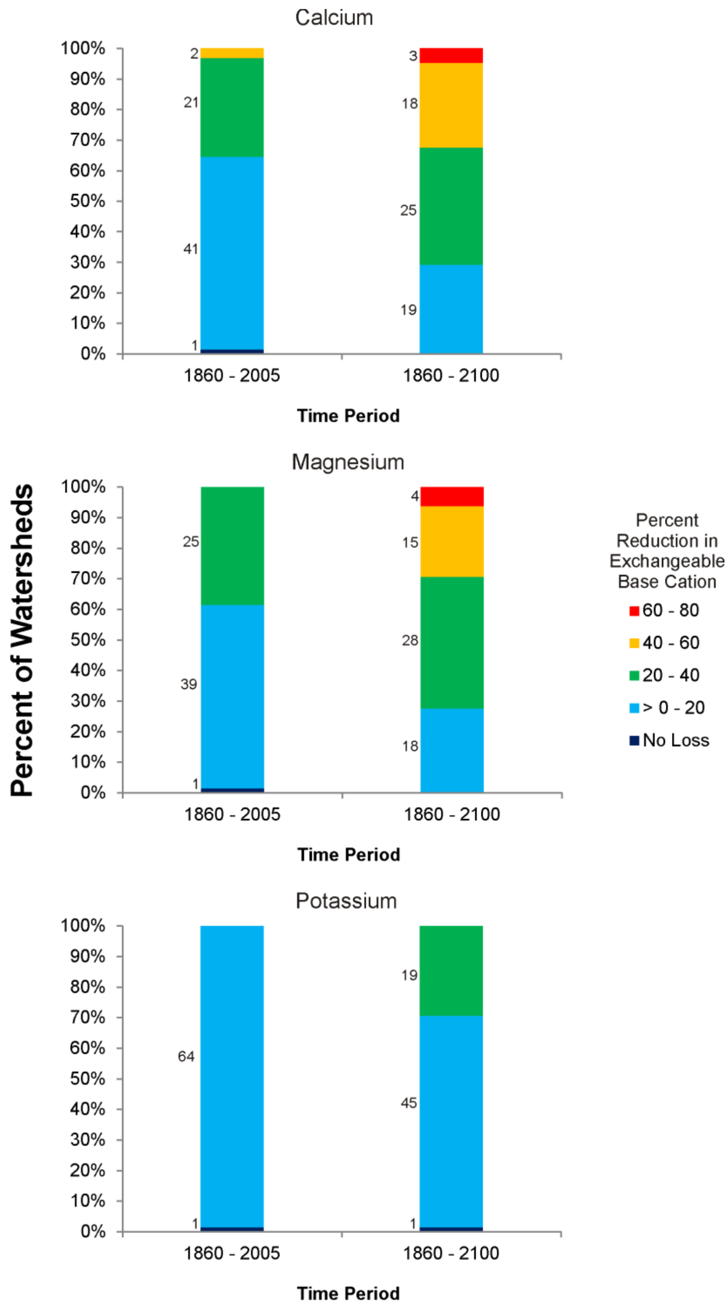

Figure 2-3. Percent of modeled watersheds among various classes showing a reduction in simulated pre-industrial upper B horizon soil exchangeable nutrient base cation concentration (Ca, Mg and K) under the Base Scenario. Nutrient losses are shown for two different time periods; 1860 – 2005 and 1860 to 2100. Values adjacent to the bars indicate the number of watersheds included in each respective class.

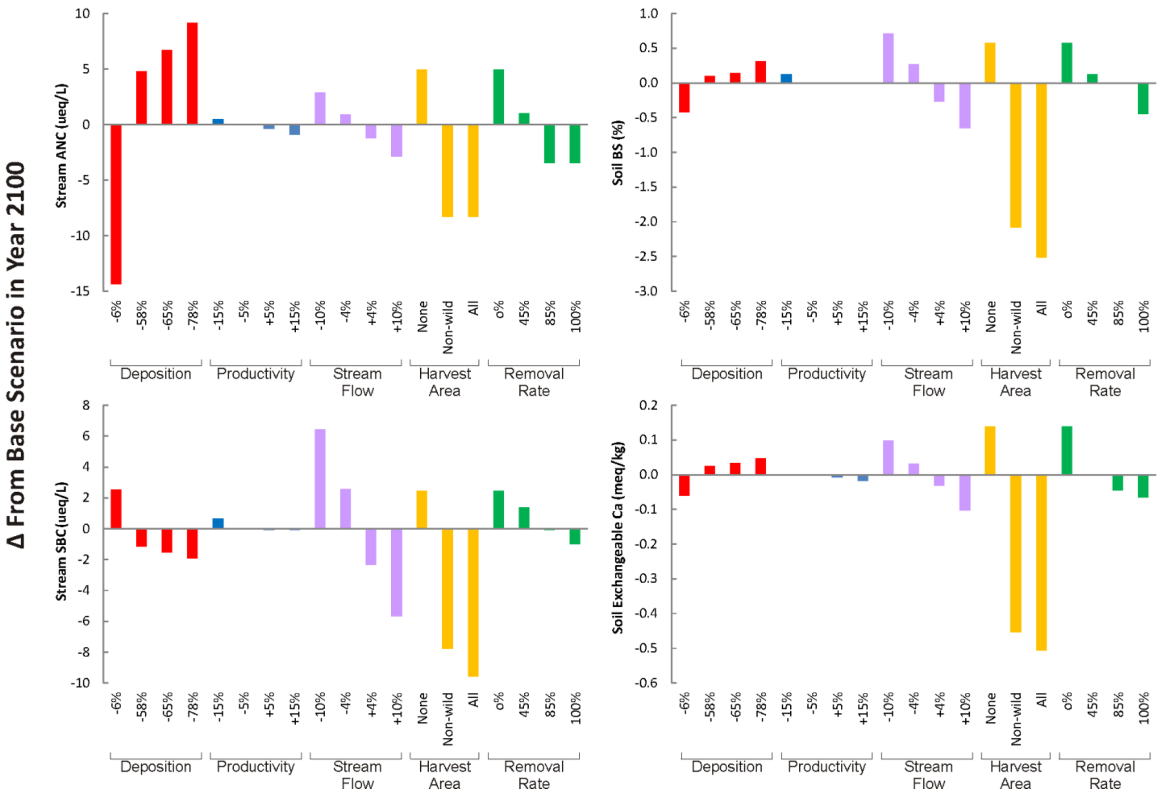

Figure 2-4. Departure from Base Scenario results of model simulations for stream ANC and sum of base cations (SBC), and soil BS and exchangeable Ca in the year 2100 for each alternative future scenario. Scenarios are grouped according to type: Deposition (red), Productivity (blue), Stream flow (purple), Harvest Area (orange) and Removal Rate (green). Positive and negative values indicate an increase and decrease, respectively, in the parameter value relative to the Base Scenario.

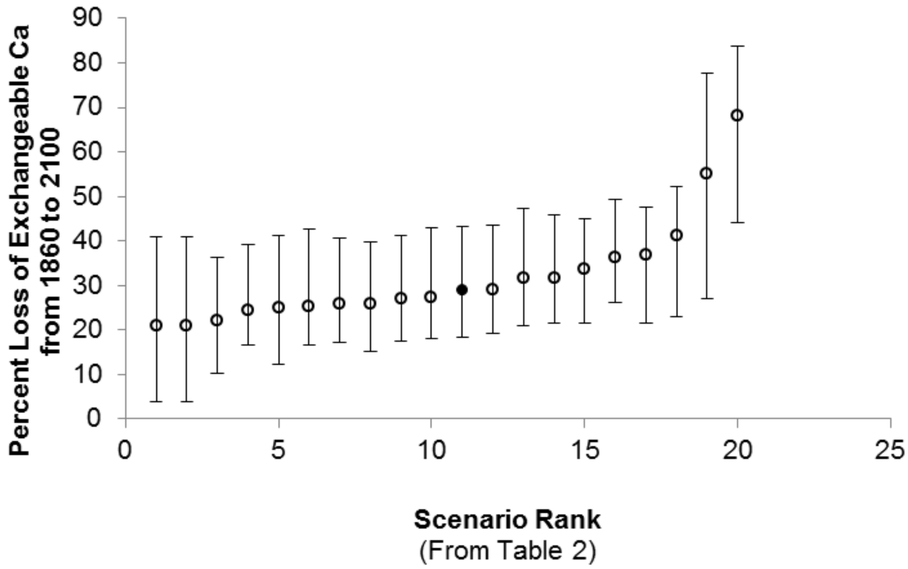

Figure 2-5. Median (circle) and quartile (bars) percent loss of exchangeable Ca from 1860 to 2100 for each scenario. The scenarios are ranked from the lowest to highest median percent loss from left to right. The Base Scenario is shown as a filled-in circle. See Table 2-1 for description of scenarios.

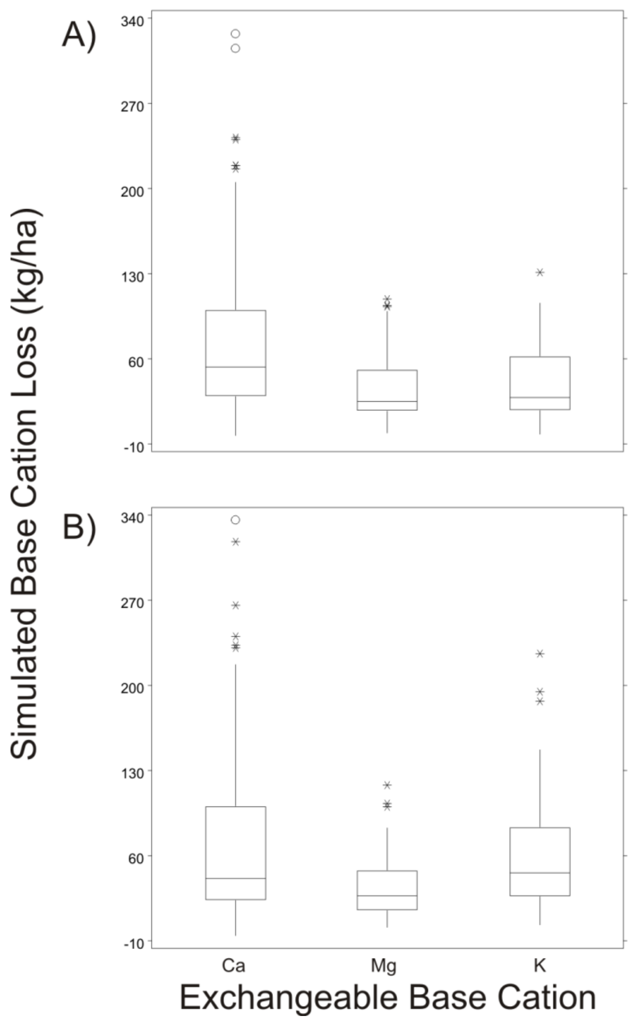

Figure 2-6. Distribution of A) historical 1860 – 2005 and B) future 2005 – 2150 base cation losses (kg/ha) for the 65 model watersheds under the Base Scenario. Asterisks and open circles indicate data points more than 1.5x and 3x, respectively, the interquartile distance.

CHAPTER 3 - PROTECTING AQUATIC ECOSYSTEMS IN THE US SOUTHERN APPALACHIAN MOUNTAIN REGION: ESTIMATING STEADY-STATE SULFUR CRITICAL LOADS AND EXCEEDANCES

Abstract

Atmospherically deposited sulfur (S) causes stream water acidification throughout the eastern US Southern Appalachian Mountain (SAM) region. Acidification has been linked with reduced fitness and richness of aquatic species and changes to benthic communities. Stream acidification occurs when atmospherically deposited sulfate is conveyed to streams by ground and surface waters, and results in decreased acid neutralizing capacity (ANC) and pH. Maintaining acid-base chemistry that supports native biota depends largely on balancing acidic deposition with the natural resupply of base cations. Stream water ANC is maintained by base cations that mostly originate from weathering of surrounding lithologies. When ambient atmospheric S deposition exceeds the critical load (CL) the ecosystem can tolerate, stream water ANC conditions may become lethal to biota. This work links statistical predictions of ANC and base cation weathering surfaces for streams in the Southern Appalachian Mountain region with a steady-state model to estimate CLs and exceedances. Results showed that 21% of the total length of study region streams displayed ANC values <100 $\mu eq \cdot L^{-1}$, where effects to biota may be anticipated; most were 4th or lower order streams. Nearly one-third of the stream length within the study region exhibited CLs of S deposition < 50 $meq \cdot m^{-2} \cdot yr^{-1}$, which is less than the regional average S deposition of 60 $meq \cdot m^{-2} \cdot yr^{-1}$. Owing to their geologic substrates, relatively high elevation, and cool and moist forested conditions, the percentage of stream length in exceedance was highest for mountain wilderness areas and in national parks, and lowest for privately-owned lands in the valley bottoms. Exceedance results were summarized by 12-digit hydrologic unit code (subwatershed) for use in developing management goals and policy objectives, and for long term monitoring.

INTRODUCTION

Atmospheric sulfur (S) deposition, originating largely from coal-fired electrical power generation and other industrial sources causes soil, groundwater, and stream water acidification across broad areas of the southeastern United States (U.S. EPA 2008). Such acidification has been associated with enhanced leaching of sulfate (SO_4^{2-}), depletion of calcium (Ca^{2+}) and other base cations from soils, reduced pH and acid neutralizing capacity (ANC) of surface waters, and increased mobilization of potentially toxic inorganic aluminum (Al_i) from soil to streams (Sullivan 2000). Biological effects have included toxicity to fish and aquatic invertebrates (Cosby et al. 2006, U.S. EPA 2009).

Sulfur is the primary determinant of precipitation acidity and SO_4^{2-} is the dominant anion in streams throughout most of the southern Appalachian Mountain (SAM) region (Sullivan et al. 2004). Nitrate (NO_3^-) is important at some locations, especially in streams that flow from high-elevation old-growth forests in North Carolina and Tennessee (Cook et al. 1994). Although a substantial proportion of atmospherically deposited S can be retained in watershed soils, SO_4^{2-} concentrations in many mountain streams have increased due to atmospheric S deposition and low S retention in soils (Elliott et al. 2008), causing related increases in base cation concentrations and decreased stream water ANC.

Acidic soils and streams have developed in this region over a period of many decades in response to high levels of atmospheric S deposition. Many streams in Great Smoky Mountains (GRSM) and Shenandoah (SHEN) national parks and surrounding national forests show signs of acidification, including streams in wilderness areas. Both of these parks and several wildernesses are federally mandated Class I areas, and receive special protection against air pollution impacts under the Clean Air Act. As a result of emissions controls regulation (U.S. EPA 2009), atmospheric S deposition has decreased throughout the eastern US since the early 1980s, and further decreases are expected. To that end, improvements to stream water acidity predictions are needed for effective resource management.

Ecosystem sensitivity to acidification and the potential effects of atmospheric S deposition on surface water quality are fairly well documented for this region, particularly within the National Acid Precipitation Assessment Program (NAPAP, 1991), the Fish in Sensitive Habitats (FISH) project (Bulger et al. 1999), and the Southern Appalachian Mountains Initiative (SAMI; Sullivan et al. 2004, 2007).

Stream water ANC is one measure that reflects the ability of a watershed to neutralize acidic inputs. As the rate of acidic deposition increases, ANC often decreases in proportion to the natural re-supply of base cations from the soil. At certain levels, increases in the hydrogen ion (H^+) and Al_i concentration are directly toxic to fish, including brook trout (*Salvelinus fontinalis*; Bulger et al. 1999, Baldigo et al. 2007), a favored native game fish found in cold, high-elevation streams in the SAM. Various ANC thresholds are associated with different levels of biological effects. In the SAM region and in mountainous areas of the northeastern US, moderate effects on macroinvertebrate and fish species richness are associated with ANC concentrations between ~50 and 100 $\mu eq \cdot L^{-1}$ (Cosby et al. 2006, Sullivan et al. 2006). More substantial effects have been observed at ANC concentrations <50 $\mu eq \cdot L^{-1}$. Most aquatic species, including the relatively acid-tolerant brook trout, can be extirpated at ANC concentrations <0 $\mu eq \cdot L^{-1}$ (Bulger et al. 1999, Cosby et al. 2006, Sullivan et al. 2006, U.S. EPA, 2009).

Land managers and regulators are concerned about the current and future health of native aquatic species within the SAM, and this concern is legally mandated. Soils in this region have developed from the slow weathering of parent rock material, some of which is inherently low in base cations. Adequate amounts of available Ca, magnesium (Mg), and potassium (K) are all essential to maintain an acid-base chemistry that will support persistence of native fish and aquatic invertebrate species. Where the existing stream water acidity is too high to support the native biota, and where ambient stream water ANC is insufficient for buffering, policy-makers may consider the need to call for added air pollution emissions reductions to allow for the recovery of impacted species, and to prevent further impacts.

To inform public policy regarding air pollutant emissions reductions, it is important to determine: 1) the emissions and atmospheric deposition levels that are associated with varying degrees of chemical effects and 2) the linkages between water and soil chemistry and subsequent biological impacts.

One approach to addressing these issues is to construct model estimates of regional surface water acid-base chemistry and critical loads (CLs). The CL for S acidification is the level of sustained atmospheric S deposition below/above which harmful effects to sensitive ecosystems are unlikely/likely, based on current understanding (Nilsson and Grennfelt 1988). The CL is typically calculated as a steady-state value, using models such as the Steady State Water Chemistry model (SSWC; Henriksen and Posch 2001). However, data with which to inform the steady-state CL calculation may also be derived dynamically using mass balance equations in a process model such as the Model of Acidification of Groundwater in Catchments (MAGIC; Cosby et al. 1985). If ambient S deposition exceeds the identified CL, the ecosystem is said to be in exceedance, and it has a heightened risk of biological harm.

The long-term maintenance of well-buffered aquatic ecosystems depends primarily on maintaining atmospheric S levels that are lower than the natural re-supply rate of base cations through weathering (BC_w). Thus, base cations derived from weathered substrates are generally most influential in determining CLs (McDonnell et al. 2010). However, because BC_w predictions can contribute substantial uncertainty to CL estimation (Li and McNulty 2007, U.S. EPA, 2009), it is essential to continue to improve the certainty of weathering estimates.

Steady-state CL calculations have been developed and applied across northern Europe (Gregor et al. 2004) and eastern Canada (Watmough and Dillon 2002, Ouimet et al. 2006), providing a basis for political and economic negotiations and national and international air pollution legislation. Some recent efforts have focused on process-based dynamic model estimates of critical or associated target loads (cf., Sullivan et al. 2005, 2008). Land managers and regulators are also interested in regional predictions at watershed locations, where current stream water ANC is affecting the health of aquatic biological communities (U.S. EPA, 2009).

This work applies the results of recent regional statistical modeling to predict surfaces of stream water ANC and soil BC_w throughout the SAM region. The BC_w modeling results are used together with other model input parameters to estimate steady-state CLs for atmospheric S deposition and CL exceedances. Stream ANC estimates are used to assess potential biological effects associated with modeled S deposition.

Previous regional efforts to characterize stream water sensitivity to acidic deposition have been based on a stratified random sampling of a subset of streams (cf., Whittier et al. 2002). Results were used to make general statements about ecosystem sensitivity throughout the full population of surface waters; however, methods were insufficient for determining the location of sensitive reaches. This study resolves this problem by generating spatially explicit CL and exceedance estimates for all streams within the SAM region.

As a result of emissions regulation and advances in hydraulic fracturing technology for natural gas production, reduced S emissions at coal-fired power plants and shifts to natural gas-powered electric generation have primarily been responsible for significant reductions in acidic deposition throughout the U.S. (Burns et al. 2011). Results reported here also consider how changes in exposure to S deposition over time relate to the inherent acid sensitivity of the landscape and expected future stream conditions with respect to acidification.

The SAM region comprises an irregular patchwork of land ownerships, protection status, resource management goals, and sensitivity to degradation from S deposition. National parks and wildernesses are home to terrestrial and aquatic ecosystems that are afforded more legal protections than those that exist on other lands (Organic Act 16 U.S.C. §1 (1997); Organic Act 16 U.S.C. §1601(a) (1997); Wilderness Act 16 U.S.C. §1131 (1997)). It is therefore desirable to determine the extent to which acid-sensitive streams occur within these protected areas. Furthermore, present-day land managers need to determine acid-sensitive locations to make informed resource management decisions and recommendations to policy-makers.

METHODS

The study region encompasses a broad geographical extent covering the SAM region and surrounding terrain, from northern Georgia to southern Pennsylvania, and from eastern Kentucky and Tennessee to central Virginia and western North Carolina. The region is primarily comprised of the Blue Ridge, Ridge and Valley, and Central Appalachian ecoregions, but also includes small portions of the Piedmont, Northern Piedmont and Western Allegheny Plateau (Fig. 1; Omernik 1987). The Blue Ridge ecoregion is dominated by metamorphic and igneous parent materials, whereas the Ridge and Valley, and Central Appalachian ecoregions are dominated by sedimentary parent materials characterized by northeast to southwest trending sandstone ridges and limestone valleys. Elevations range from about 300 to 2000 m. Uplands of oak, hickory, pine, and other mesic forests and woodlands are interspersed with crop and pasture lands, which occur primarily in the lowlands. Spruce-fir and northern hardwood forests predominate at the higher elevation sites.

An outline of the primary steps for the development of CLs and CL exceedance results reported here is described below:

1. Aggregate available measured ANC data with which to develop a statistical hurdle model to predict a threshold ANC response (above/below 300 $\mu eq \cdot L^{-1}$) for all streams and a continuous ANC value for those streams with a high probability of low ANC;
2. Calibrate BC_w for a subset of the ANC sample sites that also contain measured soil chemistry data using a process model (MAGIC); and use these calibrated weathering rates to develop a statistical model for generating regional estimates of BC_w for all catchments;
3. Use the ANC threshold model to identify streams within the study region that are most likely to be associated with low-ANC (< 300 $\mu eq \cdot L^{-1}$) conditions;

4. Combine regional estimates of BC_w with gridded surfaces of the remaining SSWC input terms to generate aquatic CLs, based on an ANC target for recovery, for expected low ANC streams;
5. Use spatially continuous estimates of total S deposition to determine the exceedance of the calculated CL using SSWC.

The regional CL and CL exceedance results presented here are based on statistical predictions of ANC for all streams and BC_w for all catchments located within the study region (Povak et al. 2013, 2014).

ANC and BC_w Response Data

Measured ANC data

Stream water chemistry data were obtained from several national and regional databases, including the National Stream Survey (NSS), Environmental Monitoring and Assessment Program (EMAP) stream surveys, Virginia Trout Stream Sensitivity Study (VTSSS), and others. Stream chemistry data sources were described by Sullivan et al. (2004, 2011). A total of 933 sampling locations were included. Sites selected for MAGIC modeling (n = 140) from among the 933 stream sampling sites were those that had soil acid-base chemistry data within the watershed (Figure 3-1).

Water chemistry data were collected between 1986 and 2009, with the largest proportion of the data (43%) collected during the VTSSS survey in 2000. Stream ANC was calculated as the equivalent sum of the base cation concentrations (Ca^{2+}, Mg^{2+}, K^+, Na^+, ammonium [NH_4^+]) minus the sum of the mineral acid anion concentrations (chloride [Cl^-], NO_3^-, SO_4^{2-}). The most recent spring sample was used to characterize ANC status for each site. The distribution of ANC was negatively skewed, with 75% of the sites showing values < 163 $\mu eq \cdot L^{-1}$. Values for ANC had a mean of 188 $\mu eq \cdot L^{-1}$, a median of 72 $\mu eq \cdot L^{-1}$, and the inter-quartile range (IQR) was 33 to 163 $\mu eq \cdot L^{-1}$ (range: -109 to 3,889 $\mu eq \cdot L^{-1}$). A 30-m digital elevation model (DEM) was used to create a synthetic stream network (described below) to which all ANC sample points were georeferenced within a geographic information system.

MAGIC Calibrated BC_w

BC_w was estimated using the MAGIC model for 140 of the 933 sampled water chemistry sites. MAGIC is a lumped-parameter model developed to predict the long-term effects of acidic deposition on surface water chemistry (Cosby et al. 1985). The model simulates soil solution and surface water chemistry to predict the monthly or annual average concentrations of the major ions. Central to MAGIC calculations is the size of the pool of exchangeable base cations in the soil. As the fluxes to/from the pool change in response to atmospheric deposition and biomass export (i.e. tree harvesting), the chemical equilibria between soil and soil solution shift to cause changes in surface water chemistry. MAGIC has been used to reconstruct the history of acidification and to simulate future trends in a large number of catchments in both North America and Europe (e.g., Lepistö et al. 1988, Whitehead et al. 1988, Hornberger et al. 1989, Cosby et al. 1990, Jenkins et al. 1990a, Jenkins et al. 1990b, Jenkins et al. 1990c, Wright et al. 1990, Norton et al. 1992, Wright et al. 1994). For a more complete description of the model, see Cosby et al. (1985, 2001).

Because it is a lumped parameter model, MAGIC must be calibrated with observed stream, soil, and atmospheric deposition data before it can be used to examine potential system responses. The BC_w rate was extracted from MAGIC model calibrations conducted here for each stream watershed and used as the response variable for predictive modeling. Most sites modeled with MAGIC had relatively low measured ANC (< 50 $\mu eq \cdot L^{-1}$). The distribution of BC_w among the MAGIC model sites was also skewed towards relatively low values, with 75% of the sites below 91 $meq \cdot m^{-2} \cdot yr^{-1}$. Simulated values for BC_w at these sites had a median of 66 $meq \cdot m^{-2} \cdot yr^{-1}$ and an IQR of 42 to 91 $meq \cdot m^{-2} \cdot yr^{-1}$ (range: 3 to 257 $meq \cdot m^{-2} \cdot yr^{-1}$), with a mean of 73 $meq \cdot m^{-2} \cdot yr^{-1}$.

ANC and BC_w Predictions

Objectives for ANC and BC_w predictive modeling were to develop and validate statistical models that best explained observed ANC and MAGIC model simulated BC_w

values across the study region. Exploratory studies evaluated statistical performance of ordinary least-squares (OLS), logistic regression (logR), geographically weighted regression (GWR), multivariate adaptive regression splines (MARS), classification and regression tree (CART), boosted classification-regression tree (BCT/BRT) and random forest (RF) methods. Considering up to six statistical performance metrics, RF models generally outperformed all other models. Therefore, RF modeling techniques were used to generate continuous ANC and BC_w estimates for the region (Povak et al. 2013, 2014). The RF model is an adaptation of CART analysis that uses an ensemble of regression or classification trees to produce robust model predictions. Each individual tree within the ensemble is developed using random subsamples of the data and predictor variables (Breiman 2001).

A suite of initial candidate predictor variables was chosen to represent potential broad- to fine-scale climatic, lithologic, topo-edaphic, vegetative, and S deposition variables with the potential to influence ANC and BC_w. In order to incorporate average upslope conditions that potentially influence stream chemistry at specific locations along a stream, it was necessary to express all candidate landscape predictor variables on a grid basis with a cell size of 30 m. This resolution was sufficient to conduct flowpath analyses that were used to develop topographically determined streams and also to prepare the predictor variable datasets for ANC and BC_w regionalization. Values of predictors from the area contributing to each 30-m grid cell were upslope averaged within the study region, based on methodology described in McDonnell et al. (2012).

For predicting ANC, a two-stage hurdle model approach was applied. The hurdle modeling approach used an ANC threshold to preselect locations that were well buffered (high ANC), and a continuous model that generated ANC values for the remaining low-ANC sites. A final model was chosen by identifying the model with the lowest misclassification and root mean squared error (RMSE) values. For each 30-m grid cell of the study region, the predictor variables were entered into the hurdle modeling framework. The threshold model predicted the probability that the grid cell had a low ANC value ($< 300\ \mu eq \cdot L^{-1}$). If the resultant probability value was less than a specified threshold (0.7), then the grid cell was considered well-buffered and assigned an arbitrarily large ANC value. If the probability of encountering low ANC for a particular grid cell was greater than or equal to the specified threshold, the environmental data for that grid cell were entered into the continuous model for prediction.

Predictions of BC_w were generated with a single continuous RF model for each 30-m grid cell within the study region. It was not possible to generate a robust hurdle model for predicting BC_w due to the low number of MAGIC calibrated BC_w rates ($n = 140$). However, results from the ANC threshold model were used to constrain BC_w estimates and resultant CLs to locations that were predicted to exhibit low ANC, as described further in the next section. Streams that had high ANC were assumed to also have high BC_w and CL. Predicted BC_w rates were also generated using water chemistry data at locations where measured water chemistry exist ($n = 933$). These watersheds comprised only 9% of the study region and could not be used for regional analyses. The two statistical methods (with and without water chemistry) and one process-based method (MAGIC) for estimating BC_w were used, where available, to generate a complete surface of BC_w for the study region.

Estimating Sulfur Critical Loads and Exceedances

Watersheds were delineated based on hydrologically conditioned DEM derivatives from the National Hydrography Dataset (NHDPlus; U.S. EPA and USGS, 2005). This process delineated a total of 140,504 watersheds within the study region, with an average size of approximately 1 km^2 (median = 0.8; IQR = 0.5 to 1.3; range = < 0.001 to 17.0). Sulfur critical loads for each topographically determined catchment were then calculated using the SSWC model (Henriksen and Posch 2001), as described below.

The CL process involves selection of one or more sensitive receptor(s), one or more chemical indicator(s) of biological response for the sensitive receptor(s) of concern, and one or more critical chemical indicator criteria values that have been shown to be associated with adverse biological impacts. For the sensitive receptor stream water, the most commonly selected chemical indicator is ANC. A number of critical criteria values of ANC have been used as the basis for CL calculations, the most common of which have been 0, 20, 50, and 100 $\mu eq \cdot L^{-1}$ (cf., Posch et al. 2001, U.S. EPA, 2009). The first two levels approximately correspond in the Appalachian Mountains region to chronic and episodic effects on brook trout, respectively (Bulger et al. 1999). An ANC threshold of 50 to 100 $\mu eq \cdot L^{-1}$ is believed to be protective of general ecological health (cf., Cosby et al. 2006, U.S. EPA, 2009).

SSWC Model for Estimating CL

The SSWC model (Henriksen and Posch 2001) is calculated essentially as a balancing of watershed base cation (e.g., Ca^{2+}, Mg^{2+}, Na^{+}, K^{+}) inputs and outputs. The base cation inputs to the subject watershed are BC_w and atmospheric deposition (BC_{dep}). The base cation outputs include nutrient (i.e., Ca^{2+}, Mg^{2+}, K^{+}; Bc) uptake by tree trunks (boles) that are subsequently removed from the watershed through timber harvest (Bc_{up}). Also included in the model with the base cation output terms is an allowance (or buffer) for the base cations needed to support ecosystem health. In the SSWC and other aquatic CL approaches, this buffer is expressed as an ANC leaching flux (ANC_{limit}), which is calculated as the product of the selected threshold ANC value and water runoff. In this study the ANC threshold value was set at 50 $\mu eq \cdot L^{-1}$.

The watershed supply of base cations due to weathering is the CL model parameter that typically has the most influence on the CL calculation and has the largest uncertainty (Li and McNulty 2007, U.S. EPA, 2009, McDonnell et al. 2010). In essence, the maintenance of long-term aquatic ecosystem acid-base chemical health depends on keeping the atmospheric acid load relatively low compared with the natural re-supply of base cations through weathering and atmospheric deposition. Thus, the CL is controlled largely by BC_w and the desired steady-state ANC (ANC_{limit}).

The CL for S acidity [CL(A)] was calculated for this study (as described by Henriksen and Posch 2001) as:

$$CL(A) = BC_{dep} + BC_w - Bc_{up} - ANC_{limit} \qquad \text{(Eq. 1)}$$

in which all units are in $meq \cdot m^{-2} \cdot yr^{-1}$. Derivation of each term in Eq. 1 can be found described in detail in Appendix 1.

Because each term in Eq 1 can be estimated at a broad spatial scale, it is possible to use the SSWC model to develop regional estimates of the CL of S deposition and associated S deposition exceedances across the landscape. The S deposition exceedance represents the extent to which ambient S deposition exceeds the calculated CL needed for ecosystem protection or recovery. It is calculated as the ambient S deposition minus the CL to protect stream resources against damage caused by excess S deposition. This model function allows assessment of regional patterns in acidification sensitivity (CL) and effects (extent to which ambient acidic deposition exceeds the CL). It also allows for mapping and calculation of the total stream length and percent of stream length within the region or designated subregion of interest that falls within certain CL or exceedance classes.

SSWC CL Exceedance

The CL exceedance was determined by calculating the extent to which ambient estimates of S deposition were above or below the CL. For the purposes of this study, the CL is considered to be in exceedance if the level of S deposition is more than 15% above the CL. If S deposition is more than 15% below the CL then the catchment is not considered to be in exceedance. Catchments where S deposition is within 15% of the CL are considered to be uncertain with respect to exceedance of the CL. Total S deposition was derived from the total deposition (TDEP) project developed by the U.S. Environmental Protection Agency (U.S. EPA 2013). Estimates of S deposition were calculated for two time periods as three-year averages centered on 2001 and 2011. These deposition rates were used to evaluate changes in CL exceedance over time between 2001 and 2011.

RESULTS

ANC

Results from the continuous ANC model showed good agreement with observed values of ANC (R^2 = 0.92, RMSE = 24.9; Povak et al. 2013). Low-ANC conditions tended to occur in areas characterized by siliceous bedrock, relatively wet and cool growing season, low soil pH and clay levels, large amounts of forested land cover, and a small watershed contributing area (Povak et al. 2013).

Regional predictions of stream ANC are shown in Figure 3-2. Portions of both SHEN and GRSM show values less than 100 $\mu eq \cdot L^{-1}$, as do many designated wilderness areas. From a public land manager's perspective, it is important to recognize that 50% of the streams within the study region were predicted to have high ANC and associated high CL. Nevertheless, of the high-ANC area, only 5% occurs on publicly owned land.

Approximately 6% (13,310 km) of the streams within the study region could be characterized using measured ANC based on available stream survey data. Previously, the stream water ANC status of the remaining 210,867 km of stream was largely unknown. Regional statistical modeling results showed that 21% of the stream length in the study region was predicted to have ANC less than 100 $\mu eq \cdot L^{-1}$ (Table 3-1). This is more than three times the total length of stream that had previously been characterized to be in this range using measured ANC data.

The length of streams having predicted ANC sufficiently low as to be associated with probable biological effects varied with stream order. The length of streams with predicted ANC < 50 was highest for 1st and 2nd order streams (Figure 3-3). Streams having ANC between 50 and 100 $\mu eq \cdot L^{-1}$ were also skewed towards low stream orders (4th order or lower).

BC_w

Results from the BC_w predictive modeling showed good agreement with MAGIC calibrated BC_w, particularly for observed BC_w values below 150 $meq \cdot m^{-2} \cdot yr^{-1}$ (Figure 3-4). Above this value there were few data points to inform the model, which led to poor performance at BC_w values above about 150 $meq \cdot m^{-2} \cdot yr^{-1}$. The uncertainty associated with each of these three methods for estimating BC_w is shown in Table 3-2. Overall, uncertainty among the methods was generally low, although uncertainty was largest for the statistical estimates based on landscape characteristics alone. Statistical estimates using both water chemistry and landscape data were more certain. BC_w derived from MAGIC showed the lowest uncertainty; however, these latter estimates only apply directly to 1% of the stream length within the study region. Low BC_w sites (< 150 $meq \cdot m^{-2} \cdot yr^{-1}$) were generally associated with drier areas having siliceous lithology; low S deposition; high conifer cover; and low clay, organic matter, and nitrogen content in soil.

Critical Load

Nearly one-third of the stream length within the study region was determined to have CL of S deposition below 50 $meq \cdot m^{-2} \cdot yr^{-1}$ (Table 3-3). Average 2011 S deposition across all topographically determined catchments was 60 $meq \cdot m^{-2} \cdot yr^{-1}$. Many of the wildernesses and national parks occupy portions of the study region having S critical load values classified in the 0-25 $meq \cdot m^{-2} \cdot yr^{-1}$ or 25-50 $meq \cdot m^{-2} \cdot yr^{-1}$ classes. There were numerous other areas classified in the 0-25 $meq \cdot m^{-2} \cdot yr^{-1}$ CL category (Figure 3-5), and most of these areas are under USDA Forest Service or state management.

Exceedance

Streams throughout the study region were shown to be in exccedance of the CL of S for an ANC threshold of 50 $\mu eq \cdot L^{-1}$ and were commonly located in wilderness areas and national parks (Figure 3-6). The percent of stream length in exceedance was highest for wilderness areas under all federal jurisdictions and non-wilderness areas in national parks (Figure 3-7). Estimates of S deposition in 2001 exceeded the CL for nearly 4,900 km of stream located in wilderness areas and national parks (Figure 3-7a). According to 2011 S deposition, stream length in exceedance within wilderness and national parks was reduced to approximately 3,000 km, which accounts for more than half of the stream length in these areas (Figure 3-7b). Other federal land, managed primarily by USDA Forest Service, showed a similar exceedance response to declines in S deposition between 2001 and 2011 as compared with wilderness areas and national parks. However, the length of stream exceedance in 2011 on other federal land was more than three times greater (9,160 km) than in wilderness and national parks. The majority (57%) of stream length on private land was not in exceedance according to 2001 S deposition and increased to 70% non-exceedance based on 2011 S deposition. However, because of the large amount of private land within the study area, there was more than three times as much stream length in exceedance on private land (40,046 km) as compared with federally managed lands (12,137 km).

A generalized spatial representation of CL exceedance is shown in Figure 3-8. The exceedance results for the topographically determined catchments (n = 140,504) are expressed on the basis of 12-digit hydrologic unit code (HUC) boundaries (n = 1,561) according to estimates of S deposition for years 2001 (Figure 3-8a) and 2011 (Figure 3-8b). These HUC delineations are frequently used by the USDA Forest Service and others for land management. The median values of the exceedance results for all watersheds contained within each 12-digit HUC are shown on the left panels of Figure 3-8 and the 90th percentile exceedance values are shown on right panels. Decreases in the extent of exceedance from 2001 to 2011 are clearly evident. However, portions of all national forests and parks remain in exceedance in 2011 according to these HUC-based percentile metrics.

DISCUSSION

Knowledge of both regional acid sensitivity and spatial patterns in ambient acidic deposition across the landscape allows for holistic land, aquatic, and air resource management. Regional stream ANC and CL results presented here can be used by a variety of land managers, policy makers, and other stakeholder groups. The statistical approaches incorporated in this study provided estimates of ANC for every stream (Figure 3-2) and CL for every catchment (Figure 3-5) in the SAM region. This was accomplished in part because the ANC hurdle modeling approach was able to distinguish between watersheds that contain low ($\leq$ 300 $\mu eq \cdot L^{-1}$) vs. high (> 300 $\mu eq \cdot L^{-1}$) ANC streams. The assumption that streams with high ANC are drained by watershed soils characterized by high (> 300 meq·m^{-2}·yr^{-1}) BC_w was useful for focusing CL modeling with SSWC on locations that are most sensitive to acidic inputs.

Stream ANC less than 100 μeq·L^{-1} is considered to have the potential to cause moderate or substantial impacts on aquatic biota (Bulger et al. 1999, Cosby et al. 2006, U.S. EPA, 2009). The stream surveys included here had identified about 4,800 km of stream length in this category. Statistical techniques employed in this study had identified the locations of approximately an additional 38,000 km of stream length predicted to have ANC less than 100 μeq·L^{-1} (Table 3-1). The measured and predicted locations of low ANC streams are of interest to federal, state, tribal, and private land managers concerned with and responsible for aquatic ecosystem health.

Managers are cautioned when applying these regional results. The S CL (Figure 3-5) and exceedance results (Figure 3-6) were developed for an ANC critical limit of 50 μeq·L^{-1}. However, model simulations reported by Sullivan et al. (2011) suggested that not all streams in the study region had pre-industrial ANC as high as 50 μeq·L^{-1}. Evaluating CL exceedance based on an ANC endpoint that is not achievable in the absence of anthropogenic S deposition may not be appropriate. Land managers will need to select appropriate critical ANC limits for protecting acid-sensitive aquatic biota on the lands that they manage.

Uncertainty

Evaluation of CL exeedance requires consideration of the uncertainties associated with the calculated CL and the estimates of S deposition used to evaluate the extent to which the CL is exceeded. The steady-state CLs presented here are based on dynamic and statistical modeling, each of which include uncertainties as described here and in Povak et al. (2013, 2014). Uncertainty in the CL is lowest for watersheds characterized with MAGIC calibrations and water chemistry. Estimates of CL are more uncertain for the remainder of the streams throughout the region. The spatial variation in uncertainty associated with landscape-scale estimates of BC_w, an approximation of the uncertainty in the CL, is shown in Povak et al. (2014). Research led by the U.S. EPA is ongoing to evaluate uncertainty associated with steady-state CL model parameters and S deposition (U.S. EPA 2013, Blett et al. 2014).

The disparity between the calculated CL and S deposition was used in this study to apply an uncertainty window (± 15%) to the exceedance results. Stream watersheds exposed to S deposition greater than 15% of the CL were considered to be in exceedance. If exposure to S deposition was less than 15% of the CL the stream was considered to not exceed the CL. Exceedance was uncertain for streams with differences between the CL and S deposition that fell within the uncertainty window. For each land management type, uncertainty in CL exceedance was higher based on 2011 deposition as compared with 2001 (Figure 3-7). Streams located in wilderness areas and national parks showed more uncertainty than other public and private land. Resource managers and policy makers using these CL results for decision making can adjust the uncertainty window around CL exccedance as desired.

Natural Resource Management and Policy Implications

The results from this study can help land managers to identify areas at risk or areas where stream acidification is a concern. For example, some managers in the region have active programs to reintroduce the southern strain of brook trout in streams where they have been extirpated. The ANC (Figure 3-2) and exceedance (Figure 3-6) results provide fisheries managers with information regarding the likelihood that stream acidification may hinder the reintroduction of brook trout, or other species in both the short and long-term.

Currently, USDA Forest Service land managers utilize HUC boundaries to classify watershed condition (Potyondy and Geier 2011) and identify watersheds that may benefit from restoration efforts to improve ecosystem heath. Exceedance results were calculated here at the resolution of the topographically determined catchments (Figure 3-6), and then summarized by larger management units (12-digit HUCs; Figure 3-8). The results presented in Figure 3-8 will aid USDA Forest Service managers in the classification of watershed condition (Potyondy and Geier 2011) and can be used for regional forest planning. Currently, the nitrogen (N) plus S CL results from McNulty et al. (2007) are used by the USDA Forest Service for protection of terrestrial resources against impacts caused by soil acidification. The results from this study offer an improvement in the CL predictions for the HUCs by incorporating CLs and exceedances for protection of aquatic resources. Land managers can also use the ANC results in Figure 3-2 to aid in classifying the water quality condition for streams where samples have not been collected. Only about 9% of the watersheds in the region have been sampled and analyzed such that water chemistry data are appropriate to calculate stream ANC (Table 3-2).

It will be important to verify the ANC stream predictions (Figure 2-2) with additional stream water sampling before adjustments to current land management practices are made. Since soil acidification typically precedes surface water acidification, the regional ANC results also provide additional perspective on terrestrial acid-base status. Lands containing streams having low ANC will likely have soils with low concentrations of nutrient base cations. Therefore, in addition to stream sampling, it may be necessary to collect additional soil chemistry data before restoration projects are proposed to improve ecosystem health.

The regional CL and exceedance results can be used to develop public policy related to air emissions reduction strategies. Previously, state and regional air quality agencies and organizations worked cooperatively with federal agencies and other stakeholder groups to evaluate what effect S and N emissions reductions would have on stream ANC (Sullivan et al. 2004). Additional sulfur dioxide emissions reductions are anticipated in the southeastern United States (ARS, 2007) in order to meet the National Ambient Air Quality Standards for fine particulates, and to reach the natural background visibility goal in federally mandated Class I areas by the year 2064. Results from this study provide a scientifically rigorous foundation for air quality agencies to evaluate the regional effectiveness of both current and proposed emissions reductions programs with regard to aquatic ecosystem health. Although decreases in S deposition between 2001 and 2011 have reduced the extent of CL exceedance (Figs. 7 and 8), further reductions in S deposition may be necessary to more fully protect sensitive ecosystems. The CL exceedance results show that federally managed lands, particularly national park and wilderness areas (some of which are Class I areas), are disproportionately in exceedance relative to the remaining study region (Figs. 6 and 7). There is a possibility that current regulatory programs may not be sufficient to reduce S deposition to a level that stream ANC will reach desired targets. The results from this study provide land managers with information on where S deposition may be exceeding the CL (Figs. 6 and 8), providing the basis for evaluation of alternative management options to improve aquatic ecosystem health.

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TABLES

Table 3-1. Stream length by ANC class for streams that are located outside of water chemistry watersheds (Predicted) and streams associated with water chemistry sampling (Measured).

	Stream Length in km (%)			
ANC Class ($\mu eq \cdot L^{-1}$)	Predicted		Measured	
< 50	10,430	(4.9)	1,844	(13.9)
50 - 100	31,961	(15.2)	2,946	(22.1)
100 - 150	38,578	(18.3)	1,502	(11.3)
150 - 200	20,941	(9.9)	1,073	(8.1)
> 200	108,957	(51.7)	5,945	(44.7)
TOTAL	210,867	(100)	13,310	(100)

Table 3-2. Uncertainty and percent of the study region associated with each of the three methods for estimating BC_w.

Type	Method	Percent of Study Region	Uncertainty[a] ($meq \cdot m^{-2} \cdot yr^{-1}$)
Statistical	Landscape Data	100%	11.2
Statistical	Water Chemistry	9%	8.9
Process	MAGIC	1%	4

[a] For the statistical models, uncertainty is represented by the prediction RMSE. Uncertainty in MAGIC is represented by the standard deviation among 10 successful calibrations.

Table 3-3. Steam length (km) associated with various CL classes. The class labeled as "High CL" represents locations predicted by the hurdle model to have ANC > 300 μeq·L^{-1}. Note that units of meq·m^{-2}·yr^{-1} of S can be converted to kg/ha/yr of S by dividing by 6.25.

CL Class (meq·m^{-2}·yr^{-1})	Stream Length in km (%)
0 - 25	13,840 (6.1)
25 - 50	53,623 (23.8)
50 - 75	35,098 (15.6)
75 - 100	6,878 (3.1)
100 - 206	3,187 (1.4)
High CL	112,197 (50)
TOTAL	224,823 (100)

FIGURES

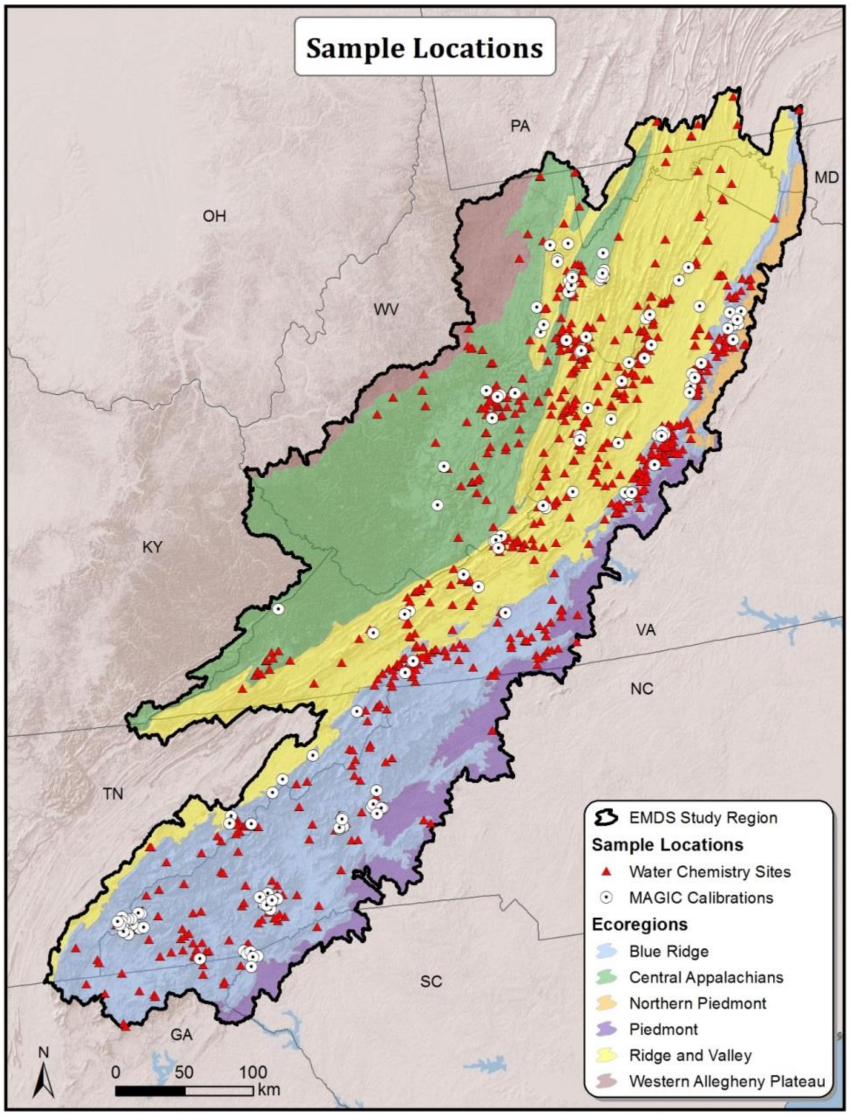

Figure 3-1. Spatial distribution of sampled water chemistry sites, MAGIC calibration sites, and location of Omernik (1987) ecoregions within the study region.

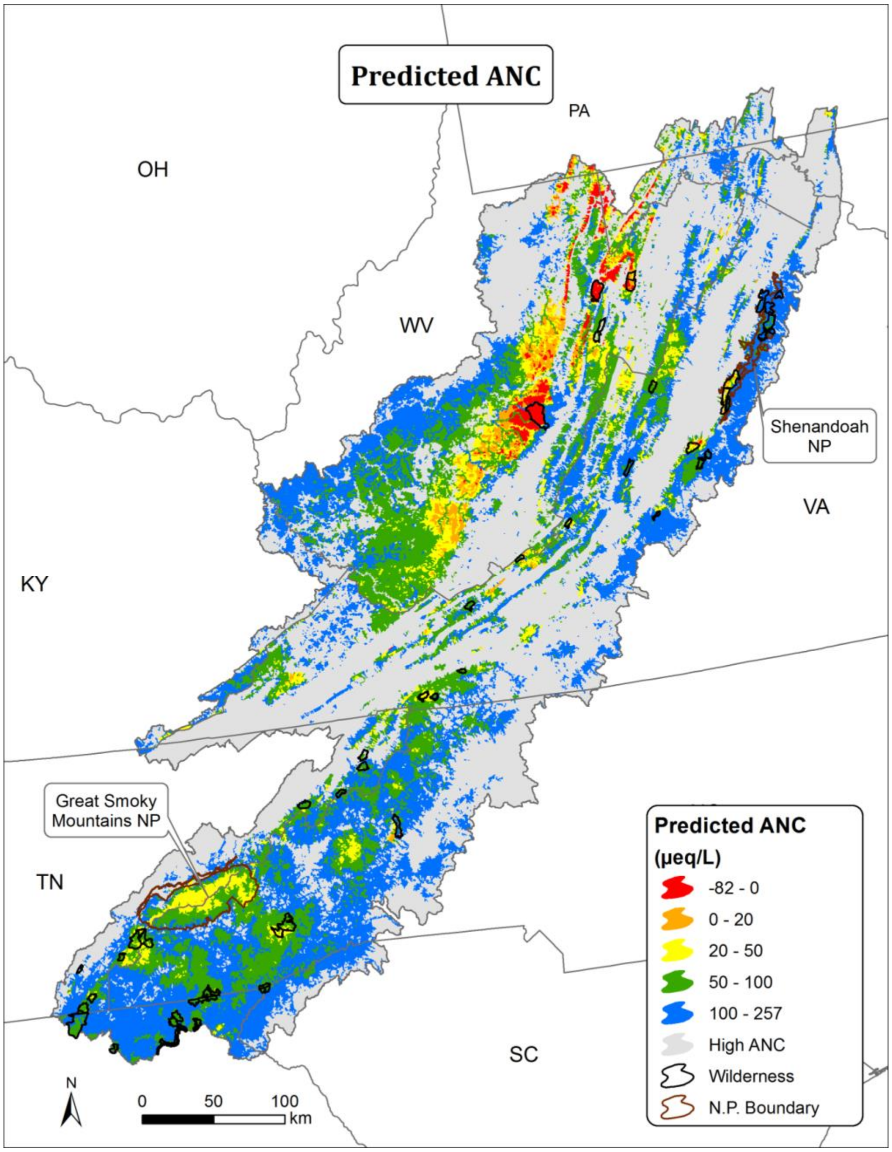

Figure 3-2. Predicted response classes for stream ANC.

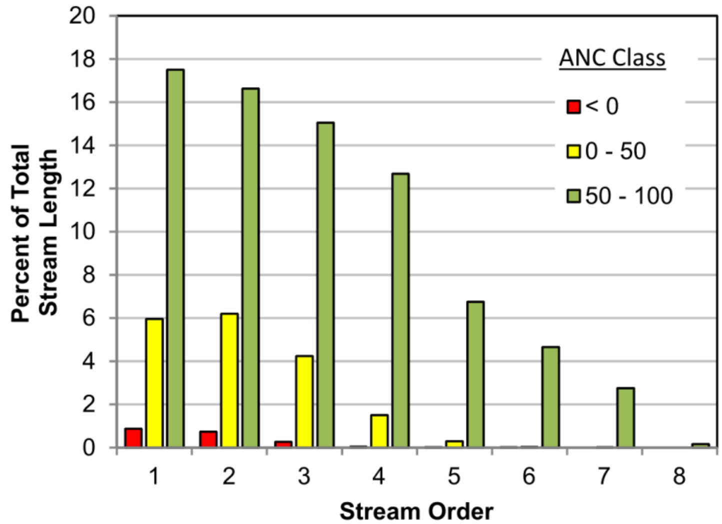

Figure 3-3. Percent of stream length by stream order within ANC (μeq/L) classes expected to be associated with biological effects.

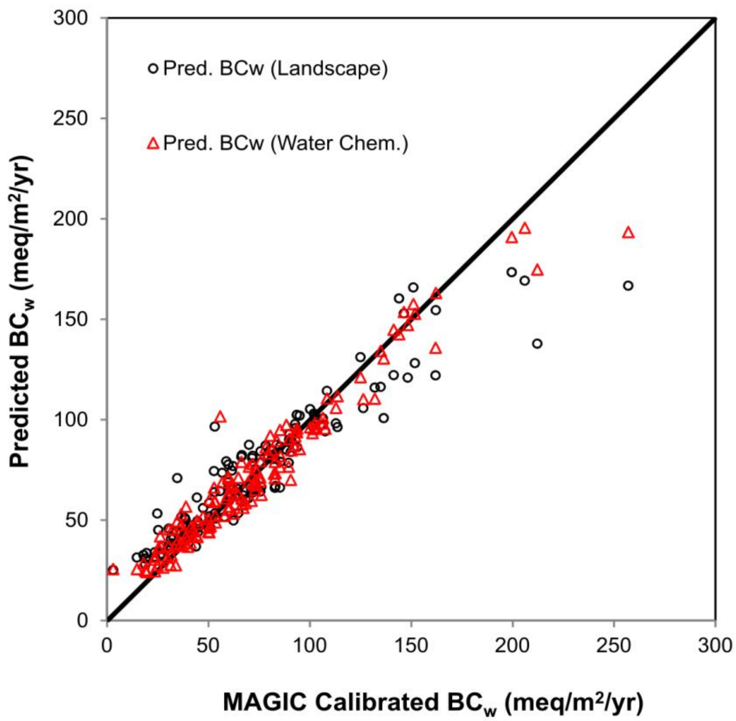

Figure 3-4. Relationship between predicted and MAGIC calibrated BC_w. The black line shows the 1:1 relationship.

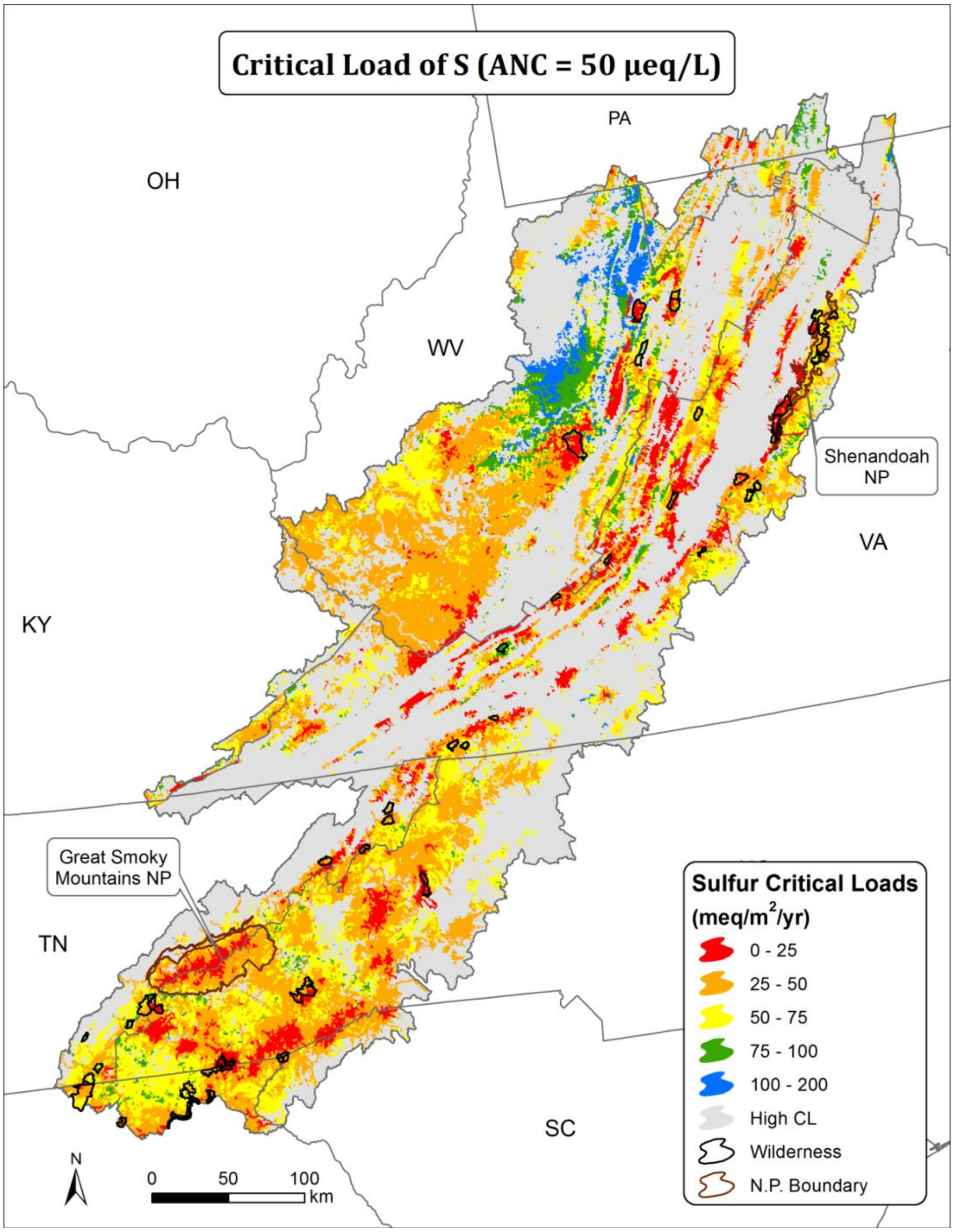

Figure 3-5. Predicted response classes for critical load of atmospheric S deposition to achieve stream ANC = 50 µeq/L under long-term steady-state conditions.

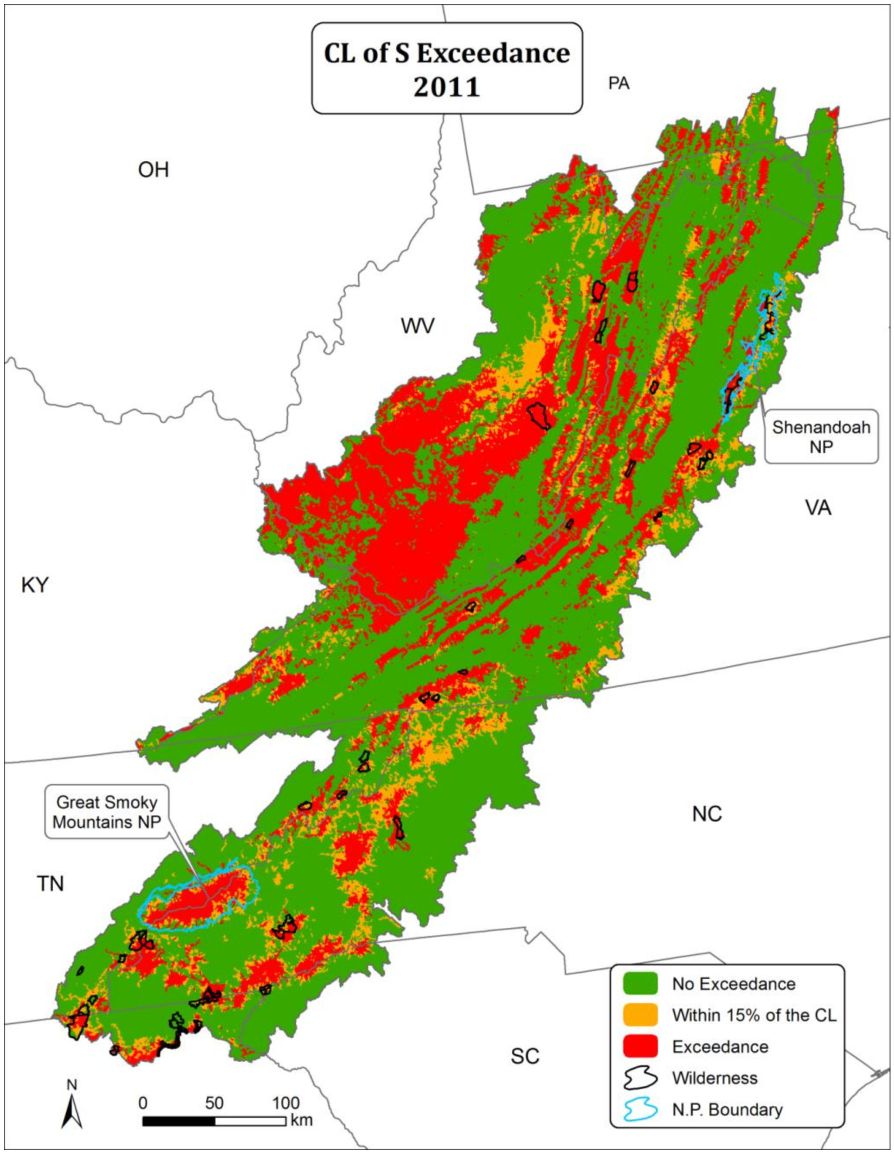

Figure 3-6. CL exceedance across the study region for the ANC critical criterion equal to 50 µeq/L according to year S deposition derived as a three-year average centered on 2011.

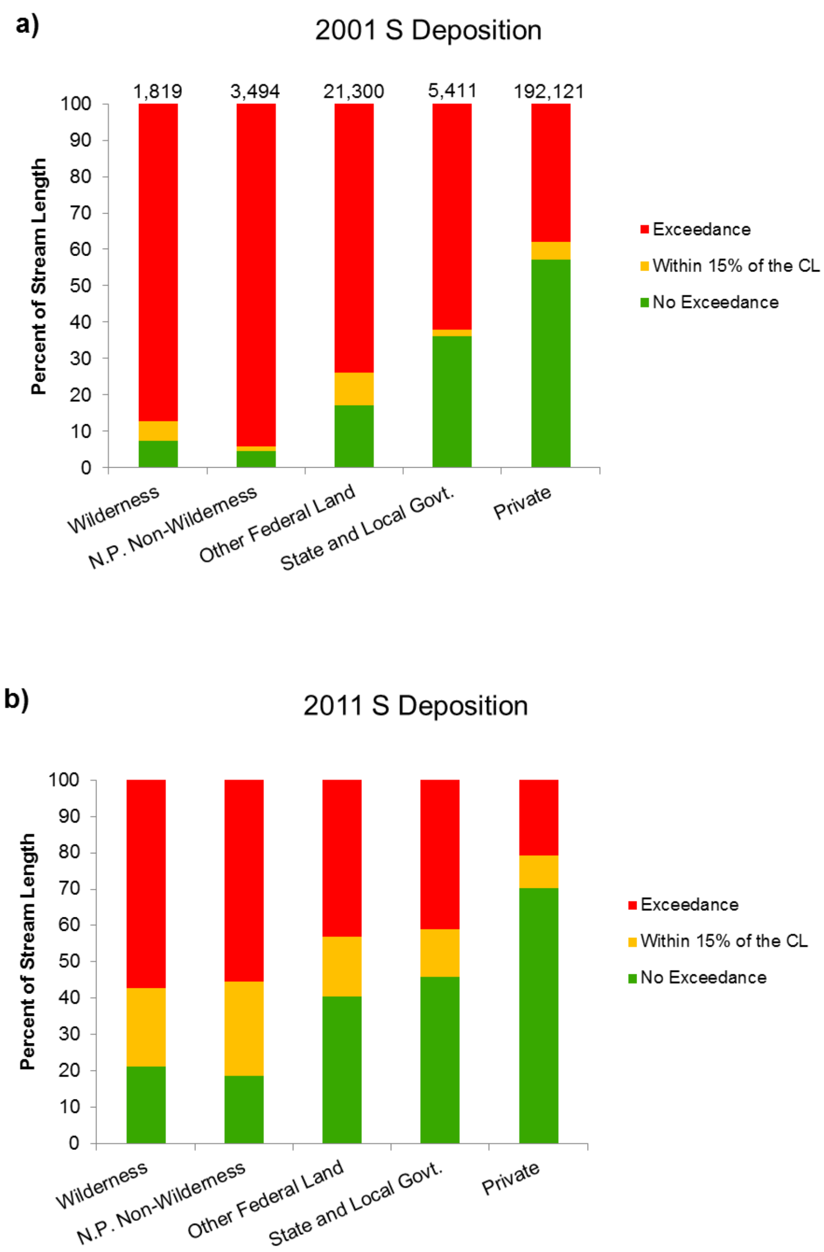

Figure 3-7. Percent of stream length within various CL exceedance classes for the primary land management types found within the study region. CL exceedance was determined from annual S deposition calculated as a three-year average centered on years a) 2001 and b) 2011. An ANC threshold of 50 μeq/L was used to derive the CL. Numbers above the bars indicate the total length of stream (km) within each land management type.

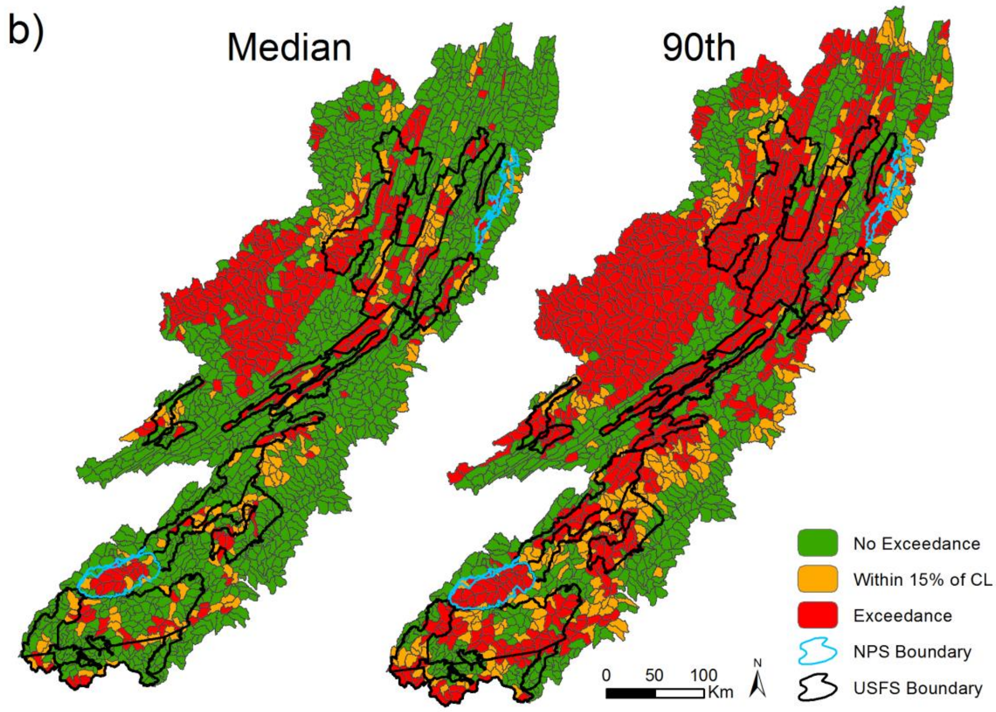

Figure 3-8. Exceedance results aggregated by 12-digit hydrologic unit code (HUC). The median and 90th percentile from the range of exceedance results for the individual study catchments contained within each 12-digit HUC are shown. Exceedance results were based on annual S deposition calculated as a three-year average centered on years a) 2001 and b) 2011. An ANC threshold of 50 µeq/L was used to derive the CL. The USDA Forest Service proclamation boundaries are shown with black outlines and the two national parks in the study area are shown with blue outlines.

CHAPTER 4 - MODELED SUBALPINE PLANT COMMUNITY RESPONSE TO CLIMATE CHANGE AND ATMOSPHERIC NITROGEN DEPOSITION IN ROCKY MOUNTAIN NATIONAL PARK, USA

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Abstract

To evaluate potential long-term effects of climate change and atmospheric nitrogen (N) deposition on subalpine ecosystems, the coupled biogeochemical and vegetation community competition model ForSAFE-Veg was applied to a site at the Loch Vale watershed of Rocky Mountain National Park, Colorado. Changes in climate and N deposition since 1900 resulted in pronounced changes in simulated plant species cover as compared with ambient and estimated future community composition. The estimated critical load (CL) of N deposition to protect against an average future (2010 – 2100) change in biodiversity of 10% was between 1.9 and 3.5 kg N ha^{-1} yr^{-1}. Results suggest that the CL has been exceeded and vegetation at the study site has already undergone a change of more than 10% as a result of N deposition. Future increases in air temperature are forecast to cause further changes in plant community composition, exacerbating changes in response to N deposition alone.

INTRODUCTION

Global nitrogen (N) and carbon (C) cycling have been significantly altered over the previous century, in large part due to growth of human and livestock populations, production and application of N-based fertilizers, and combustion of fossil fuels (Schlesinger 1997, Vitousek et al. 1997). As a result, increased atmospheric emissions and subsequent deposition of N to natural ecosystems have altered competitive relationships among plant species, decreased plant species richness and evenness, and adversely impacted some N-efficient species in favor of nitrophilous ones (Bobbink et al. 2010). Atmospheric emissions of C and N have also very likely contributed to changes in global climate, which are expected to continue into the future (Intergovernmental Panel on Climate Change [IPCC], 2007, U.S. EPA, 2009). Changes in N availability, air temperature, and precipitation patterns influence species distributions (Parmesan 2006), plant community composition, and plant biodiversity (Bobbink et al. 2010, Porter et al. 2013).

Symptoms of nutrient enrichment have been documented in remote regions, including national park and wilderness areas in the mountainous western United States (Burns 2003, Fenn et al. 2003, Geiser and Neitlich 2007, Bowman et al. 2012). Ecosystems at high elevation tend to be especially sensitive to the eutrophying and acidifying effects of atmospheric N deposition because the harsh environmental conditions limit nutrient cycling, soil development, and primary productivity (Williams and Tonnessen 2000, Weathers et al. 2006).

The critical load (CL) is used to identify the level of atmospheric deposition below which undesired effects on sensitive biota are not expected to occur (Nilsson and Grennfelt 1988, Porter et al. 2005). Critical loads have been defined to protect both terrestrial and aquatic receptors against ecological changes associated with acidification and eutrophication caused by atmospheric deposition of sulfur (S) and N (Henriksen and Posch 2001, McNulty et al. 2007, Bobbink et al. 2010). Aquatic CLs of N for protecting high-elevation Rocky Mountain lakes against eutrophication have been estimated based on paleolimnological reconstructions of historical diatom assemblages and nitrate (NO_3^-)

leaching (Baron 2006, Nanus et al. 2012). Estimates of the critical load for protecting alpine plant communities have been based on empirical results of plot-scale experimental N additions (Bowman et al. 2012) and using dynamic modeling (Sverdrup et al. 2012).

Forecasting effects of elevated N deposition on plant communities requires consideration of ongoing changes in climate. The assumption that ecosystems are in steady state, which is inherent in the CL concept (Henriksen and Posch 2001), requires unchanging environmental conditions at some unspecified time in the future at which the CL is estimated. Given current and continuing changes in climate and the complicating effects of climate on nutrient utilization and dynamics, the assumption of steady state that is often relied upon in traditional mass balance CL modeling approaches is problematic when applied to quantify nutrient enrichment impacts on terrestrial ecosystems (Belyazid et al. 2011a).

The potential impacts of changing N inputs and climate on resources within national parks has become an important management concern for the U.S. National Park Service (NPS; Porter et al. 2005, 2012). Atmospheric emissions of N outside national park and wilderness area boundaries, especially under conditions of changing climate, threaten sensitive natural ecosystems. Increased abundance of nitrophilous plant species has been demonstrated in alpine plant communities at Niwot Ridge, about 10 km south of Rocky Mountain National Park (NP; Korb and Ranker 2001). Both locations receive atmospheric pollutants, including N, from multiple emissions source areas, especially those located to the east in the urban corridor between Fort Collins and Denver (Heuer et al. 2000).

The Clean Air Act (CAA) provides special protection for air quality related values in designated Class I areas, which include Rocky Mountain NP and other national park and wilderness areas that meet certain size and establishment date criteria. One objective of CAA provisions is to prevent significant deterioration of air quality in Class I areas while maintaining a margin for industrial growth. Native plant community composition in the alpine and subalpine zones of a high-elevation Class I park such as Rocky Mountain NP represents one of the more sensitive air quality related values under the protection of the NPS. As a consequence, land managers require knowledge of the levels of N deposition at which undesirable changes are likely to occur under ambient conditions and in the future under changing climatic conditions. Public policy decisions regarding whether and to what extent to reduce N emissions depend upon quantitative dose-response relationships that reflect the tolerance of sensitive ecosystem elements to pollution inputs (Sullivan et al. 2005).

The principal objective of this study was to assess the sensitivity of subalpine vegetation in Rocky Mountain NP to simultaneous changes in N deposition and climate using a dynamic model. The resulting information was used to evaluate interactions among changes in climate and nutrient input and to quantify the CL of N deposition that will protect subalpine vegetation in this park against changes in biodiversity in the context of a changing climate. Such information is needed for effective management and preservation of park resources for the enjoyment of future generations. The results of this study may be applicable to subalpine zones elsewhere in the Rocky Mountain region.

METHODS

The coupled biogeochemical and vegetation community model ForSAFE-Veg (Belyazid 2006, Sverdrup et al. 2008, Belyazid and Moldan 2009, Belyazid et al. 2011a, Belyazid et al. 2011b) was used for evaluating past and future terrestrial plant biodiversity at a subalpine site located within the Loch Vale watershed of Rocky Mountain NP. ForSAFE-Veg is an integrated terrestrial ecosystems model, which simulates the cycles of C, N, and base cations as well as tree biomass and nutrition, soil chemistry, soil organic matter and decomposition, hydrology, and ground vegetation composition. Ground vegetation is considered to consist of all plants below the height at which trees species are no longer shaded by other plants. Trees are considered to emerge from the ground vegetation above this height. The model requires inputs of soil physiochemical properties, a plant species list, and time series of climatic and atmospheric deposition data. The geochemical platform of ForSAFE relies on process-based algorithms for chemical equilibria and the preservation of mass balances of elements. It simulates interactions between soil and biota and their integrated responses to changes in climate, atmospheric deposition, and land management (Wallman et al. 2005, Belyazid 2006). The study site was selected to represent a treeline site (elevation of 3,314 m) and was characterized with *Abies lasiocarpa* among shrubs and herbaceous forb and graminoid species. A Front Range Colorado Rocky Mountain treeline site was chosen primarily because species common to this area have been shown to respond to N input and because experimental studies exist with which to parameterize ForSAFE-Veg to this plant community type (Bowman et al. 1993, Bowman et al. 1995, Bowman et al. 2006, Bowman et al. 2012). The representative site vegetation survey used for this study was conducted in the summer of 1999 within a 20 m x 50 m Modified-Whittaker plot (Sara Simonson, Natural Resource Ecology laboratory, Colorado State University; personal communication, 2012). The survey identified 72 species of vascular plants and indicated that both mosses and lichens were also present. Species identified with more than 5% cover (collectively accounting for 75% total cover) included *Salix candida*, *Carex rupestris*, *Carex elynoides*, *Abies lasiocarpa*, and *Geum rossii*.

The Veg module requires that each modeled plant species be associated with a set of attributes which are used to evaluate plant strength under a given set of environmental conditions. Plant strength as affected by competition largely determines the relative abundance of a given species at the site (Belyazid et al. 2011a, Belyazid et al. 2011b). Plant strength is determined by the degree to which environmental conditions overlap with the input niche requirements of a particular species. The relative occupancy of a plant is determined from a multiplicative merge of the individual plant strengths. For this study, 15 species with observed cover greater than 1% each were parameterized for the following traits: responses to soil N availability, moisture, temperature, and soil acidity; shading tolerance; rooting depth; and shading height (see Appendix 2). The model simulates plant response to soil solution N with a species-specific delay defined in the parameters table (sigma). For a detailed description of the specific response equations, see Belyazid et al. (2011a). In addition to species-specific response parameters, ForSAFE-Veg requires data inputs regarding soil physiochemical properties and time series of air temperature, precipitation, and atmospheric deposition of N, S, base cations, and chloride. Plant parameterization, model calibration and derivation of other model input data are more fully described in Appendix 2.

An initial model run was generated, based on expert judgment, using initial vegetation parameterizations for each of the 15 primary plant species. The input climate niche for each of the modeled species was then calibrated to the observed percent cover as represented in the 1999 plant survey. The calibration process was performed manually. Input niche requirements for the species with the greatest deviation from observed cover were adjusted first by considering the plant strength associated with moisture and temperature conditions. For example, given the case in which modeled plant cover was less than observed cover, if moisture conditions were at (or near) optimal conditions according to the plant strength associated with moisture, the temperature niche was adjusted to favor the plant and reduce the deviation between predicted and observed cover. If both of the climate niches were sub-optimal, each niche was adjusted to result in an equivalent contribution toward fitting the model to the observed cover. This process was carried out iteratively and the vegetation parameters were considered to be calibrated when the root mean squared error (RMSE) of predicted versus observed percent cover of each species was less than 1.5%.

Scenario Modeling

For this study, 100 scenarios were simulated, based on all possible combinations of N deposition and climate shown in Table 4-1. In addition to ambient N deposition (3.5 kg N ha^{-1} yr^{-1}, 5-year average centered on 2008), one-half, 2x, and 4x ambient N deposition were considered. Current air temperature and precipitation were modeled along with the lowest, median (temperature only), and highest statistically downscaled (1/8th degree resolution) A2 estimates among 16 global climate models (Maurer et al. 2007). Future changes in temperature of +2.3, +4.6, and +6.6 ºC and precipitation of -10.1% and +22.7% by the year 2100 were simulated. Two other future precipitation scenarios beyond the bounds of the downscaled A2 projections (-50% and +50%) were also considered. Ambient S deposition was maintained to the year 2100 for each scenario. Additional descriptions of the model scenarios are included in Appendix 2.

N Critical Loads

A range of CL estimates was determined for each climate scenario by calculating the rate of N deposition below which various degrees of biodiversity change relative to background (year 1900) N deposition are not exceeded. For the purposes of generating CLs of N deposition in this study, the acceptable level of biodiversity change (calculated as Mondrian units; Belyazid et al. 2011a) is represented as an average value over the period between 2010 and 2100. This average annual difference among plant communities was used to reduce the effect of year-to-year variability and uncertainty associated with modeling N dynamics in the latter portion of the forecasted simulation period.

The area difference between two plant communities is referred to as Mondrians (M), in reference to the famous artwork produced by Piet Mondrian in the 1930s and 1940s. In Mondrian's compositions, each color is confined to a well-defined area of the canvas, and as a result the entire canvas is occupied by one color or another. In the same way, the ground vegetation is modeled with ForSAFE-Veg as multiple species, each covering a given area. If a plant species expands its coverage, it will do so at the expense of other species at the site, which will have to retreat. When a site is simulated under different deposition, climate, or land use conditions, the mosaic of plant area cover will change over time. The unit of area that differs between two scenarios is expressed in M units, where 1 M = 1% of the area covered by a chosen target plant group of interest. If the target plant group represents the entire plant community, M units become equivalent to the classical Bray-Curtis similarity index (Bray and Curtis 1957).

RESULTS

Vegetation Parameter Calibration

Predicted percent plant cover based on the initial vegetation parameters was poorly related to observed percent plant cover (R^2 = 0.02, RMSE = 8.05; Figure 4-1a). Species-specific responses to temperature and soil moisture were therefore adjusted to calibrate the model output to observed relative plant species cover. Model results based on the calibrated vegetation parameters showed good agreement with observed plant cover (R^2 = 0.98, RMSE = 1.49; Figure 4-1b). The adjustments to the vegetation parameters during calibration are shown in Table 4-2. Negative values of delta indicate a shift towards a lower soil moisture or cooler niche; positive delta values represent adjustments in the direction of higher soil moisture or warmer conditions. With the exception of *Carex rupestris* and *Aulacomnium palustre*, initial soil moisture niches were mostly shifted towards drier conditions during calibration. Temperature response was variably adjusted for each species. Favorable temperature ranges for species such as *Salix candida* and *Calamagrostis purpurascans* were shifted to become more tolerant of colder temperature, with adjustment to warmer temperatures made for *C. rupestris* and *A. palustre*.

Hindcast Vegetation Response

Estimated climate and atmospheric N deposition and simulated plant community composition based on M units changed relatively little between 1800 and 1900 (data not shown). Changes in climate and N deposition since 1900 resulted in more pronounced changes in plant species cover (Figure 4-2). Tree sapling (*Abies lasiocarpa*) coverage increased by more than 25%, graminoid response was mixed, and forbs generally decreased in abundance in the simulations between 1900 and 2010 (Table 4-3). By 2010, *Geum rossii* was reduced by more than half of its simulated historical cover.

Forecast Vegetation Response

Ambient Climate

Expected future plant species composition under ambient air temperature and precipitation conditions varied between the two lowest N deposition scenarios and was relatively similar among the three highest N deposition scenarios (Figure 4-3). Background and 0.5x ambient N deposition showed pronounced increases in *Aulacomnium palustre* (moss) abundance, beginning in 2065 and 2080, respectively (Figure 4-3). Higher N deposition scenarios showed suppressed moss cover towards the end of the simulation period. Similar patterns were apparent in the tree response between the lowest two, and highest three, N deposition scenarios (Figures 4-3, 4-4a). The dominant graminoid (*Carex rupestris*) showed decreased cover under background and 0.5x ambient N deposition. Cover of the dominant graminoids and forbs remained relatively constant into the future under higher rates of N deposition (Figure 4-4b,c).

Climate Change

Under ambient N deposition, absolute differences in simulated relative species abundance were generally less than 1% between ambient precipitation scenarios and scenarios in which precipitation was increased or decreased (Table 4-4). Similar magnitudes of change were observed for simulated precipitation change under other rates of N deposition (data not shown). Increased precipitation increases the input of water to the hydrology subroutine. The water mass balance also considers potential evapotranspiration and the hydrological properties of the soil.

Larger changes in simulated community composition occurred with expected increases in air temperature (+4.6 °C by the year 2100; Figure 4-5). Future increases in *Abies lasiocarpa* cover were simulated for all N deposition scenarios under moderate warming (Figure 4-6a). Relative abundances of *Carex rupestris* and *Geum rossii* were reduced by the year 2050 with near complete extirpation from the site by 2100 (Figure 4-6b,c). The other two warming scenarios showed similar trends in vegetation response to warming. All graminoid and several forb species were simulated to be mostly absent from the site by the year 2100 under the most extreme warming scenario (data not shown). Combined effects of changes in precipitation and warming did not yield significantly different future plant communities as compared with warming alone (data not shown). It should be noted that the simulated changes in plant community consider only species with percent cover greater than 1%. Although it is conceivable that species having less than 1% cover may become more dominant in the future, this potential effect is not captured in the model results reported here.

Critical Loads of N

Critical loads of N varied among the climate change scenarios for a specified M unit of acceptable change (Figure 4-7). The estimated CL of N to protect against an average future (2010 – 2100) change in biodiversity of 10 M was between 1.9 and 3.5 kg N ha^{-1} yr^{-1}, depending on the temperature increase scenario. A change of 10 M is roughly equivalent to the amount of cover change that has already occurred (Figure 4-8). Critical loads of N were consistently lower under scenarios with no future climate change compared to scenarios with future increases in ambient air temperature.

DISCUSSION

This study derived N CLs for the protection of terrestrial plant biodiversity in Rocky Mountain NP as represented by changes in relative percent plant species cover. The approach was based on a dynamic biogeochemical and plant competition model informed by empirical N response data. The model is designed to allow for the simulation of integrated effects of N deposition and climate change on plant biodiversity. Model results indicated that the combined effects of increases in ambient air temperature and N deposition resulted in significantly larger changes in subalpine plant community composition as compared with increases in N deposition alone (Figure 4-8). Decreases in ambient N deposition are generally expected to allow current relative plant abundance to remain constant if there are no changes in future climate (Figure 4-8a). Large changes in plant community composition are expected with future increases in air temperature.

These changes can be partially mitigated with reductions in future N deposition (Figure 4-8b).

The ForSAFE-Veg model remains under active development. This study represents the first time that ForSAFE-Veg has been calibrated to observed species abundance data. The calibration process was implemented in this study to refine the vegetation parameters considered to be least accurate due to lack of experimental data with which to inform the parameterization process. This calibration process appears to provide an additional step forward towards generating more reliable model results. However, additional validation of the calibration method is required and future model development should explore more objective approaches to calibration.

Realistic simulation of N deposition effects on subalpine plant species composition depends on the ability of the model to forecast changes in soil solution N concentrations and the provision of accurate N response curves. In the current ForSAFE-Veg formulation, soil solution N concentrations are controlled by the overstory tree vegetation, soil organic processes, and soil chemical transformations. Increased influence on soil solution N from the overstory towards the end of the simulation period of the background N scenarios resulted from the onset of more favorable (warmer) climatic growing conditions during the late 1900s. After a long delay period, tree saplings grew beyond the ground vegetation and the overstory trees participated more strongly in N cycling. In response, simulated soil solution N began to decrease in 2060 due to increased N uptake by the overstory vegetation. As a result of these changes in N dynamics, *Aulacomnium palustre* became the strongest competitor under the new soil solution N regime due to its lack of dependency on the soil environment for N uptake (Figure 4-3a). This effect is less pronounced as atmospheric N deposition increases because the additional N input compensates for the higher rate of N uptake by trees (see Appendix 2).

Critical Loads

Model results presented here suggest that the vegetation at the study site has already undergone a change of more than 10 M relative to plant community composition at estimated background N deposition (Figure 4-8). The CL to protect against an additional 10 M of change under ambient climate is currently in exceedance. The dynamic CL estimates presented here incorporate the effects of climate change, which cannot be circumvented through N emissions controls.

If temperatures continue to rise, the effects of simulated N deposition will become less prominent. Simulated higher temperature increased the CL to protect future plant biodiversity. This suggests that a warmer climate will help to protect against changes in community composition in response to elevated N deposition. Increased temperature promotes tree growth and associated N uptake which counteracts the effects of elevated N deposition on ground vegetation. Nevertheless, changes in plant communities due to increased temperature reduce the margin for acceptable change that may be induced by additional N deposition. Future increases in temperature are forecast to have a substantial impact on plant community composition beyond the expected changes in response to N deposition alone (Figure 4-8b).

Relation to Previous Studies

Colorado Front Range tree species distributions are known to be associated with temperature and moisture gradients as reflected by elevation and topographic position (Peet 1981). Observed establishment of *Picea engelmannii* and *Abies lasiocarpa* in treeline patch forest openings of Rocky Mountain NP has also been related to changes in temperature and precipitation regime (Hessl and Baker 1997). The ForSAFE-Veg model results shown here are consistent with these observations. Hindcast results show increasing cover of *A. lasiocarpa* between 1900 and 2010 at this subalpine location (Figures 4-2, 4-4a). Simulated increases in tree cover at the model site suggest that tree line may continue to advance in an upward direction at locations where appropriate substrate occurs.

Total N deposition in Rocky Mountain NP may increase in the future in response to accelerating oil and gas exploration and production, human population growth, and agricultural development in the Rocky Mountain region. If atmospheric N deposition in the Rocky Mountain area continues to increase in the future, opportunistic graminoid species, such as *Carex rupestris*, will likely continue to increase in cover at the expense of other species, including various forbs. As nitrophilous species become increasingly dominant, more of the rare species will likely be out-competed and richness and diversity will decrease (Suding et al. 2005, Sverdrup et al. 2012). Modeling results in this and other (cf., Sverdrup et al. 2012) studies suggest that such changes will occur over the long term at even lower levels of atmospheric N input. However, it should be noted that the form of the model used in this study did not account for seed dispersal and associated plant migration that may occur with changing environmental conditions. It is possible that species not included in the modeled set may become established and compete with graminoid species that presently occur on the site. Although plant migration may act to buffer changes in diversity resulting from species loss, plant community composition would remain affected.

Changes in alpine plant species composition precede detectable changes in soil chemistry in response to increased N deposition. Changes in species composition of dry meadows (among the most sensitive alpine plant community types) are probably ongoing along the Front Range (Bowman et al. 2006), in response to ambient N deposition, which varies from about 3 to 6 kg N ha^{-1} yr^{-1}. Fertilization experiments at Niwot Ridge suggested that the lowest amount of atmospheric N deposition expected to alter alpine plant communities in the short term is about 4 kg N ha^{-1} yr^{-1} (Bowman et al. 2006). This deposition level is similar to ambient deposition measured at Loch Vale. Our results indicate these changes may be further compounded by changes in climate.

Bowman et al. (2012) previously added inorganic N at various levels to vegetation plots in a species-rich dry alpine meadow community in Rocky Mountain NP. Neither the species richness nor the species diversity changed in response to the N treatments. However, the sedge *Carex rupestris* increased in cover by 34% to 125%. Increased cover of *Carex rupestris* was proportional to experimental N input. Extrapolation of this response function to the point of zero change in cover in response to N addition yielded an estimated CL of 3 kg N ha^{-1} yr^{-1} (Bowman et al. 2012). A similar estimated CL result (4 kg N ha^{-1} yr^{-1}) had been found earlier at Niwot Ridge, an alpine site located about 45 km to the south (Bowman et al. 2006). These empirically determined CL values suggest that vegetative changes are already occurring in response to N addition along the eastern side of the continental Divide within the Denver airshed. Continued elevated N loading for a longer period of time would elicit vegetative change at lower loading levels. Modeling results presented here further support that expectation.

Bowman et al. (2012) estimated a threshold for initiating increased NO_3^- leaching from soils of an alpine dry meadow at Chapin Pass in Rocky Mountain NP in the range of 9-14 kg N ha^{-1} yr^{-1}. Thus, the CL to protect against vegetation changes appears to be much lower than the CL to protect against NO_3^- leaching and consequent soil acidification, at least in the short term. Neither soil pH nor extractable soil base cations were significantly affected by the treatments with up to 30 kg N ha^{-1} yr^{-1} at the Rocky Mountain NP Chapin Pass site.

Empirical CL estimates for alpine plant communities (Bowman et al. 2012) and subalpine lake waters (Baron 2006) in Rocky Mountain NP are lower than those reported for most forest and shrubland ecosystems (Bobbink et al. 2010, Pardo et al. 2011a, Pardo et al. 2011b). Ambient atmospheric N deposition near 4 kg N ha^{-1} yr^{-1} in the park exceeds these experimentally determined CL values. Model results reported here suggest that long-term N CL may be even lower, and that subalpine plant communities will likely respond in a complex fashion to future N supply, changes in climatic conditions, and perhaps other stressors.

Management Concerns

The modeled reductions of subalpine forb species under both current climate and more markedly under warming conditions, may have a negative impact on ecosystem services. As treeline moves up in response to changes in climate and N supply, existing alpine tundra will be replaced by forest. As a result, ecosystem services associated with viewing and photographing alpine flowering plants have the potential to be diminished.

Additionally, Rocky Mountain NP contains habitat for American pika (*Ochotona princeps*), which is restricted to cool rocky alpine environments such as predominate in the Loch Vale watershed. Climatic conditions and vegetation conditions, including increased relative abundance of forb species, were associated with persistence of pika populations at high elevation in the Great Basin (Wilkening et al. 2011). Among other animals that rely on forb species, the winter diet of pika at Niwot Ridge has been shown to be comprised of 60-77% *Geum rossii* leaves that contain high amounts of phenolic compounds which act as preservatives for late winter consumption (Dearing 1996). A future reduction in abundance of forb species such as *G. rossii* may act as an additional stress limiting pika survival. Our modeled future reduction in the primary food source for pika may exacerbate anticipated reductions in habitat suitability due to changes in temperature.

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TABLES

Table 4-1. Climate and N deposition scenarios modeled. Each unique combination among these values was specified in an individual model run. Background N deposition is equivalent to the expected pre-industrial rate of deposition (year 1900; 0.6 kg N ha^{-1} yr^{-1}). Ambient N deposition represents a 5-year average centered on year 2008 (3.5 kg N ha^{-1} yr^{-1}).

	Scenario				
Driver	0	1	2	3	4
N deposition	Background	0.5x	Ambient	2x	4x
Temp (^{o}C)	Ambient	+2.3	+4.6	+6.6	
Precipitation	Ambient	-50%	-10.1%	+22.7%	+50%

Table 4-2. Adjustments to climate-based vegetation response parameters during model calibration. Values are shown for the uncalibrated (UNCAL) and calibrated (CAL) model runs along with the difference between UNCAL and CAL values (Delta) of moisture response (M) and temperature response (T).

		Moisture Response									Temperature Response								
Species	Growth Form	UNCAL Mmin	CAL Mmin	Delta Mmin	UNCAL Mopt	CAL Mopt	Delta Wopt	UNCAL Mmax	CAL Mmax	Delta Mmax	UNCAL Tmin	CAL Tmin	Delta Tmin	UNCAL Topt	CAL Topt	Delta Topt	UNCAL Tmax	CAL Tmax	Delta Tmax
Abies lasiocarpa	tree	1	0.5	-0.5	3	1.5	-1.5	4	4	0	-5	-2	3	5	4	-1	8	8	0
Salix candida	shrub	2	2.5	0.5	3	3	0	4	4	0	-1	-2	-1	5	3.2	-1.8	10	10	0
Rubus parviflorus	shrub	1.5	0.5	-1	2	1	-1	3	2.5	-0.5	-1	-1	0	3	3.2	0.2	8	8	0
Carex rupestris	graminoid	0.5	0.5	0	1.5	2	0.5	2	3	1	-7	-6	1	-2	-0.5	1.5	1	2	1
Carex elynoides	graminoid	0.5	0.5	0	2	1.6	-0.4	3	2.6	-0.4	-7	-7	0	-2	-2	0	2	2	0
Festuca ovina	graminoid	0.5	0.5	0	2.5	1	-1.5	3	2	-1	-3	-2	1	2	2.5	0.5	6	6	0
Calamagrostis purpurascans	graminoid	0.5	0.5	0	1.5	0.75	-0.75	2	1.5	-0.5	-6	-6	0	-1	-3.5	-2.5	2	1.5	-0.5
Poa abbreviata	graminoid	1	0.5	-0.5	2	0.7	-1.3	3	1.25	-1.75	-7	-7	0	-2	-4	-2	2	2	0
Geum rossii	forb	1	1	0	2	1.8	-0.2	3	2.8	-0.2	-7	-7	0	-2	-2	0	1	1	0
Aquilegia caerulea	forb	2	0.5	-1.5	3	1	-2	4	1.5	-2.5	-7	-7	0	0	3	3	4	5	1
Antennaria rosea	forb	0.5	0.5	0	2	1	-1	3	2	-1	-3	-2	1	5	5	0	10	10	0
Arenaria fendleri	forb	0.5	0.5	0	1.5	1	-0.5	2	1.5	-0.5	-7	-7	0	-2	-4	-2	1	0.5	-0.5
Minuartia obusiloba	cushion	1	1	0	1	0.8	-0.2	2	1.8	-0.2	-7	-7	0	-2	-4	-2	1	0.5	-0.5
Zigadenus elegans	forb	1	0.5	-0.5	2.5	1.75	-0.75	3	2.5	-0.5	-2	-0.5	1.5	2	2.5	0.5	5	5	0
Aulacomnium palustre	moss	2	2.5	0.5	3	3.5	0.5	4	4	0	-5	-2	3	0	3	3	5	8	3

Table 4-3. Hindcast absolute and percent changes in species abundance between 1900 and 2010 in response to historical reconstructions of N deposition (Sullivan et al., 2005) and historical climate change (IPCC 2007).

			Relative Abundance (%)			
Growth Form	Species	Code	1900	2010	Absolute Change	Percent Change
Tree	*Abies lasiocarpa*	ABLA	6.3	8.4	2.1	25.1
Shrubs	*Salix candida*	SACA	15.4	25.5	10.1	39.8
	Rubus parviflorus	RUPA	0.0	4.4	4.4	N/A
Graminoids	*Carex rupestris*	CARU	6.8	16.4	9.6	58.7
	Carex elynoides	CAEL	25.1	12.8	-12.3	-95.7
	Festuca ovina	FEOV	1.7	3.2	1.5	45.5
	Calamagrostis purpurascans	CAPU	4.3	3.0	-1.4	-46.1
	Poa abbreviata	POAB	3.2	2.3	-0.9	-38.8
Forbs	*Geum rossii*	GERO	19.7	8.9	-10.8	-121.7
	Aquilegia caerulea	AQCA	4.6	4.8	0.2	4.5
	Antennaria rosea	ANRO	1.5	2.0	0.5	27.0
	Arenaria fendleri	ARFE	4.7	1.8	-2.9	-167.5
	Minuartia obusiloba	MIOB	3.7	1.6	-2.0	-123.8
	Zigadenus elegans	ZIEL	0.0	2.2	2.2	N/A
Moss	*Aulacomnium palustre*	AUPA	3.0	2.6	-0.5	-17.6

Table 4-4. Forecasted absolute changes in species abundance between ambient precipitation and extreme precipitation scenarios under ambient N deposition.

	Year 2100 Relative Abundance			Absolute Vegetation Change from Ambient Precipitation Scenario in Year 2100	
	Precipitation Scenario			*Precipitation Scenario*	
Species	Ambient	-50%	+50%	-50%	+50%
Abies lasiocarpa	10.2	9.3	11.1	-0.9	0.9
Salix candida	26.0	23.8	27.2	-2.2	1.2
Rubus parviflorus	5.2	5.4	5.2	0.2	0.0
Carex rupestris	15.2	18.4	11.3	3.2	-3.9
Carex elynoides	12.3	11.9	13.0	-0.4	0.7
Festuca ovina	3.5	3.9	3.4	0.4	-0.1
Calamagrostis purpurascans	2.5	2.9	2.4	0.4	-0.1
Poa abbreviata	2.0	2.3	1.9	0.3	-0.1
Geum rossii	8.3	7.9	8.8	-0.4	0.5
Aquilegia caerulea	4.6	5.3	4.4	0.7	-0.2
Antennaria rosea	2.5	2.4	2.6	-0.1	0.1
Arenaria fendleri	1.6	1.5	1.7	-0.1	0.1
Minuartia obusiloba	1.4	1.4	1.4	0.0	0.0
Zigadenus elegans	1.4	1.3	1.4	-0.1	0.0
Aulacomnium palustre	3.3	2.4	4.2	-0.9	0.9

FIGURES

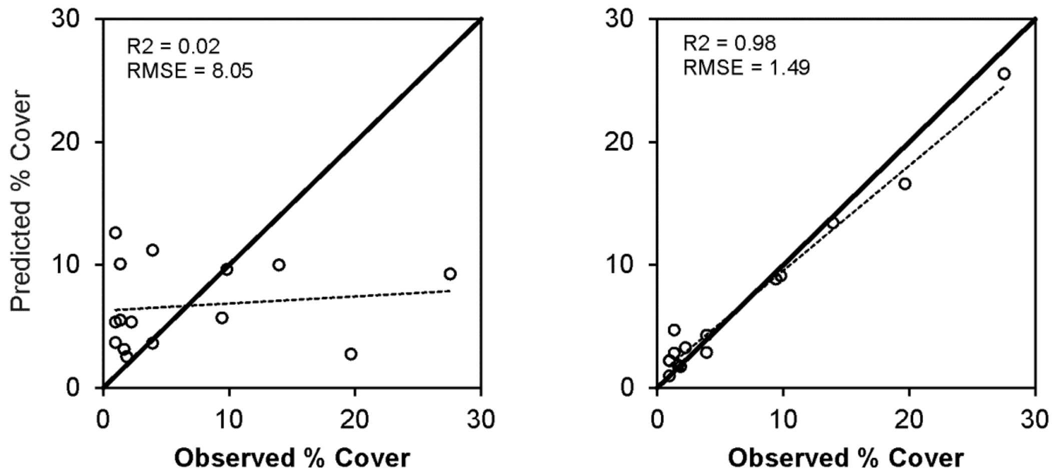

Figure 4-1. ForSAFE-Veg model predicted vs. observed current plant species cover based on a) initial vegetation parameters determined by expert judgment only and b) adjusted vegetation parameters after calibration. Each data point represents one species. The solid line represents 1:1; the dotted line represents best fit regression relationship.

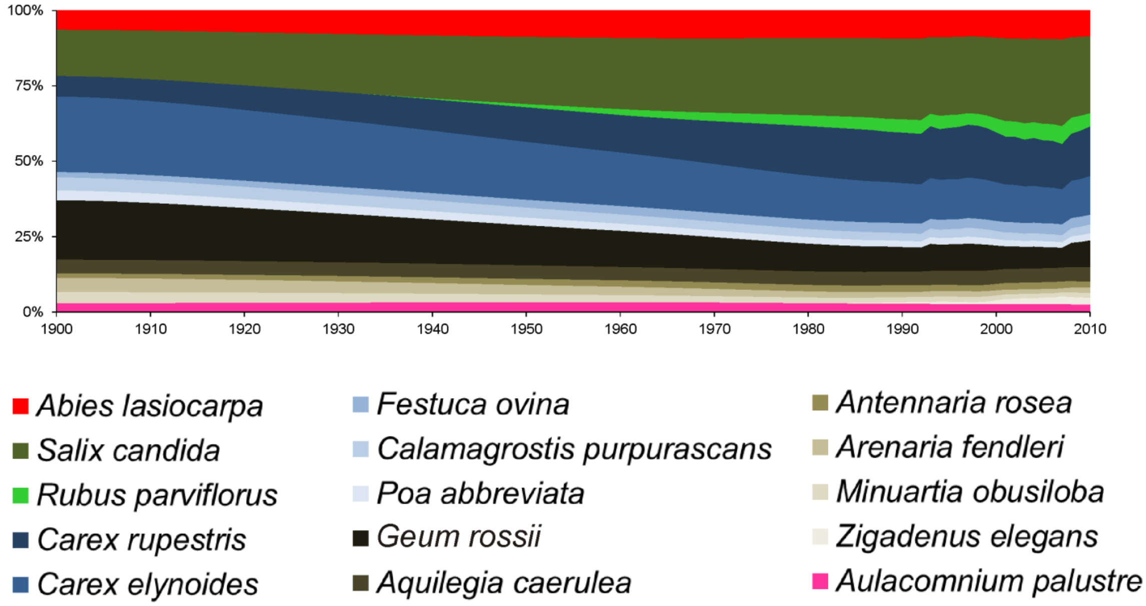

Figure 4-2. Hindcast relative species cover based on estimated historical climate change and back trajectory of ambient N deposition. Species are arranged, from top to bottom, in groupings according to life forms, including trees (red), shrubs (green), graminoids (blue), forbs (brown), and bryophytes (pink). Within each life form group, species are arranged from highest to lowest abundance based on modeled results for year 1999.

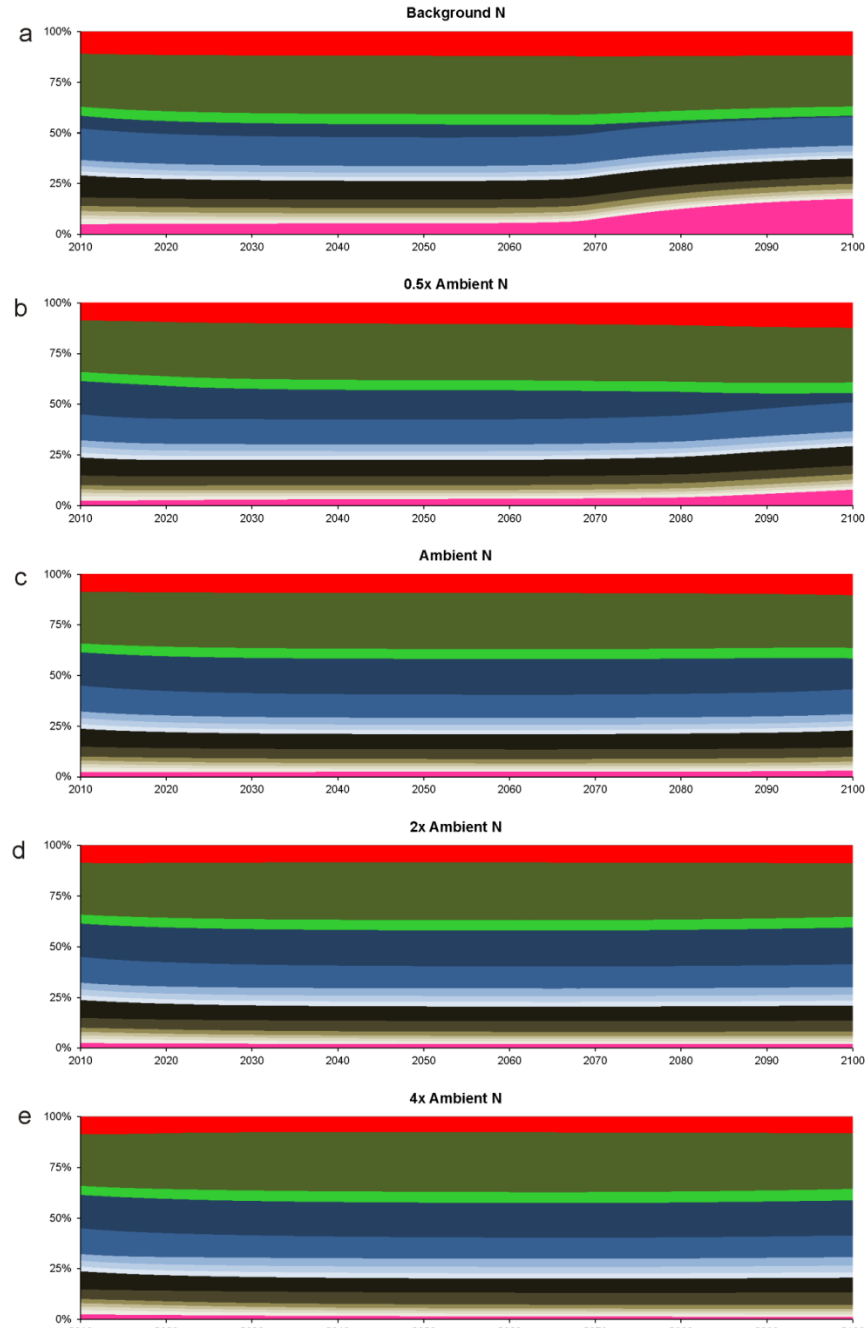

Figure 4-3. Scenario results for relative species abundance based on ambient climate conditions for the set of N deposition scenarios including a) background N, b) 0.5x ambient N, c) ambient N, d) 2x ambient N, e) 4x ambient N. Species are arranged, from top to bottom, in groupings according to life forms; trees (red), shrubs (green), graminoids (blue), forbs (brown), and bryophytes (pink). Within each life form group, species are arranged from highest to lowest abundance based on modeled results for year 1999.

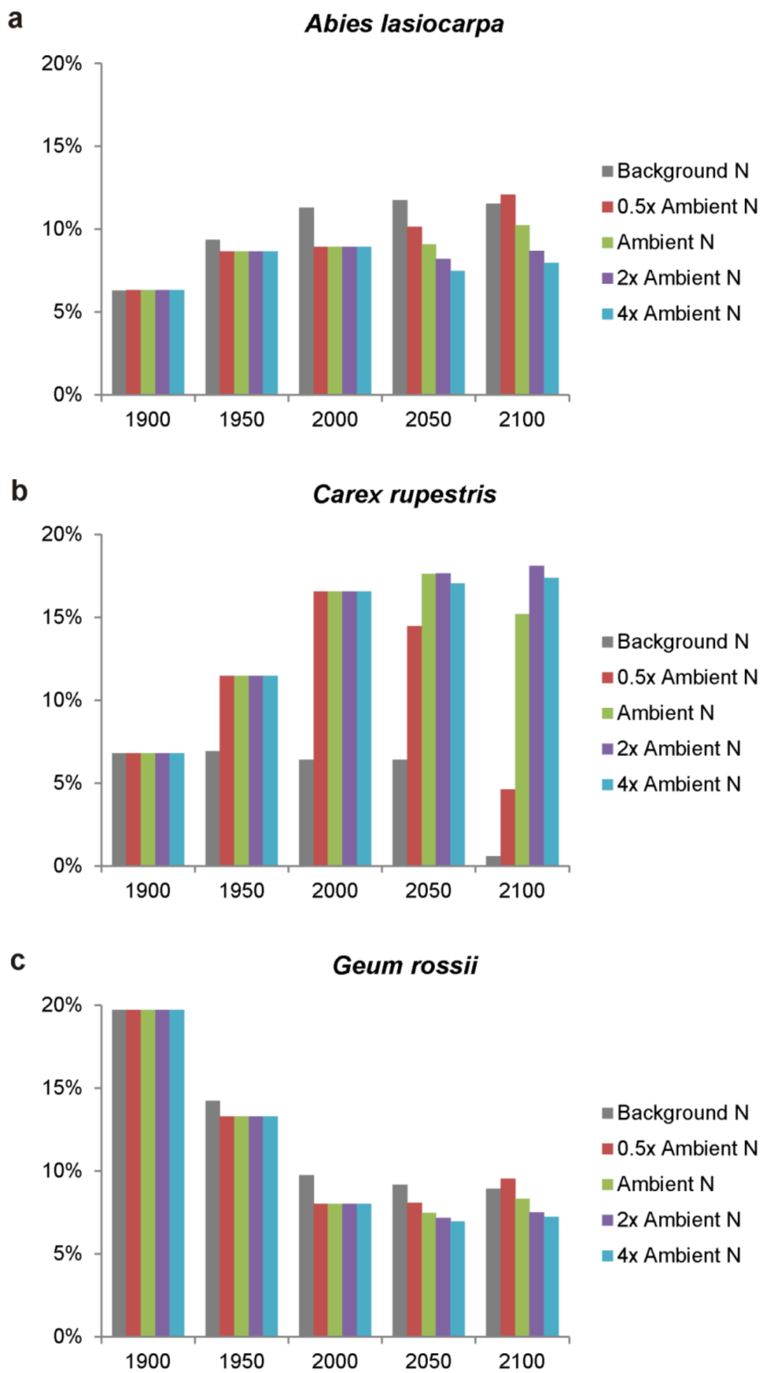

Figure 4-4. Simulated changes in percent cover of a) *Abies lasiocarpa*, b) *Carex rupestris* , and c) *Geum rossii* species, the dominant tree, graminoid, and forb species, respectively, between 1900 and 2100 under ambient climate and different rates of future N deposition. All scenarios include historical reconstructions of N deposition between 1856 and 1984, except for the background N deposition scenario which maintains a constant rate of 0.6 kg N/ha/yr (Baron, 2006) throughout the entire simulation period.

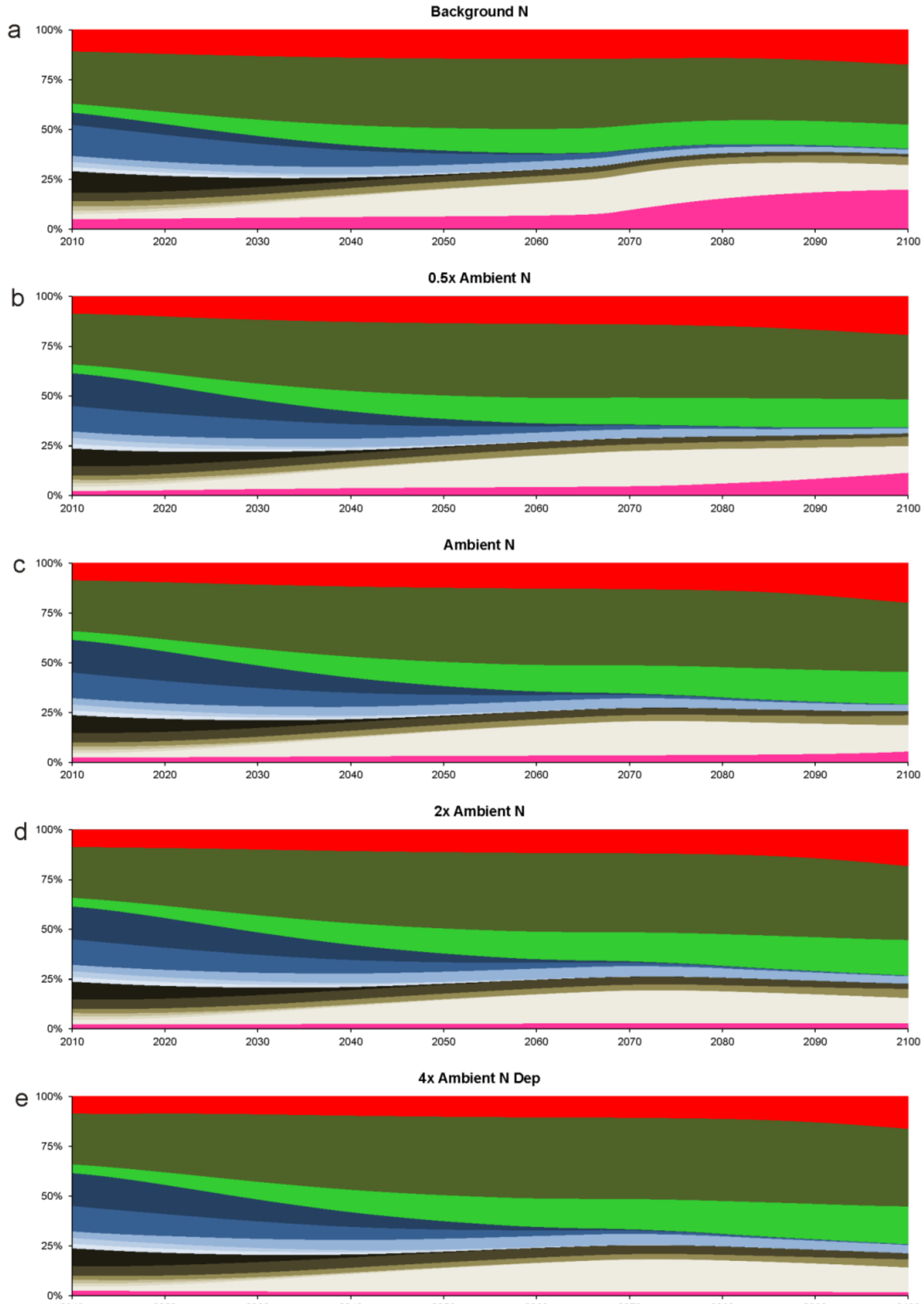

Figure 4-5. Scenario results for relative plant species abundance based on an increase in air temperature of +4.3 °C by the year 2100 for the set of N deposition scenarios, including a) background N, b) 0.5x ambient N, c) ambient N, d) 2x ambient N, e) 4x ambient N. Species are arranged, from top to bottom, in groupings according to life forms, including trees (red), shrubs (green), graminoids (blue), forbs brown), and bryophytes (pink). Within each life form group, species are arranged from highest to lowest abundance based on modeled results for year 1999.

Figure 4-6. Simulated changes in percent cover of dominant a) tree, b) graminoid, and c) forb species between 1900 and 2100 under different rates of future N deposition and a 4.6 ºC increase in temperature by the year 2100. All scenarios include historical reconstructions of N deposition between 1856 and 1984, except for the background N deposition scenario which maintains a constant rate of 0.6 kg N/ha/yr throughout the entire simulation period.

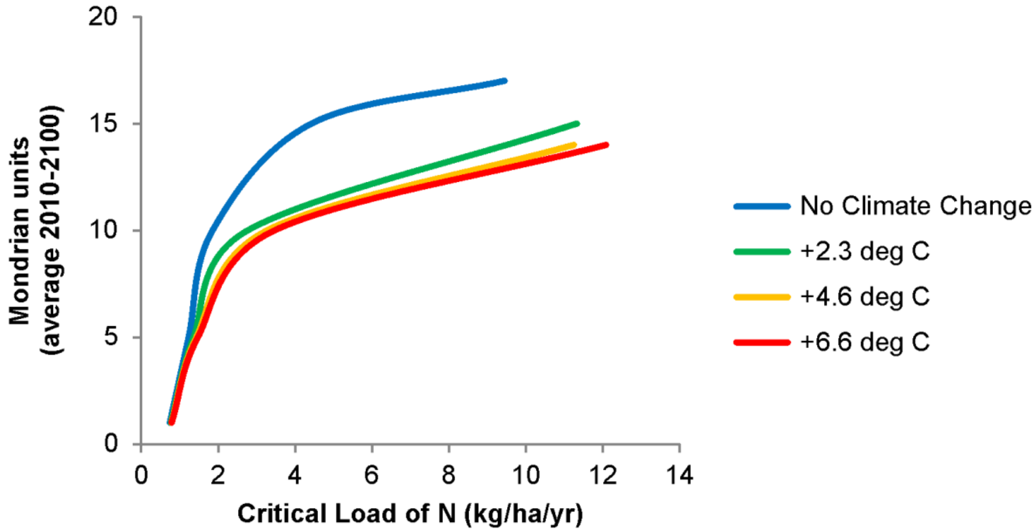

Figure 4-7. Relationship between the critical load of N deposition and level of acceptable biodiversity change, expressed in Mondrian units, for different climate scenarios.

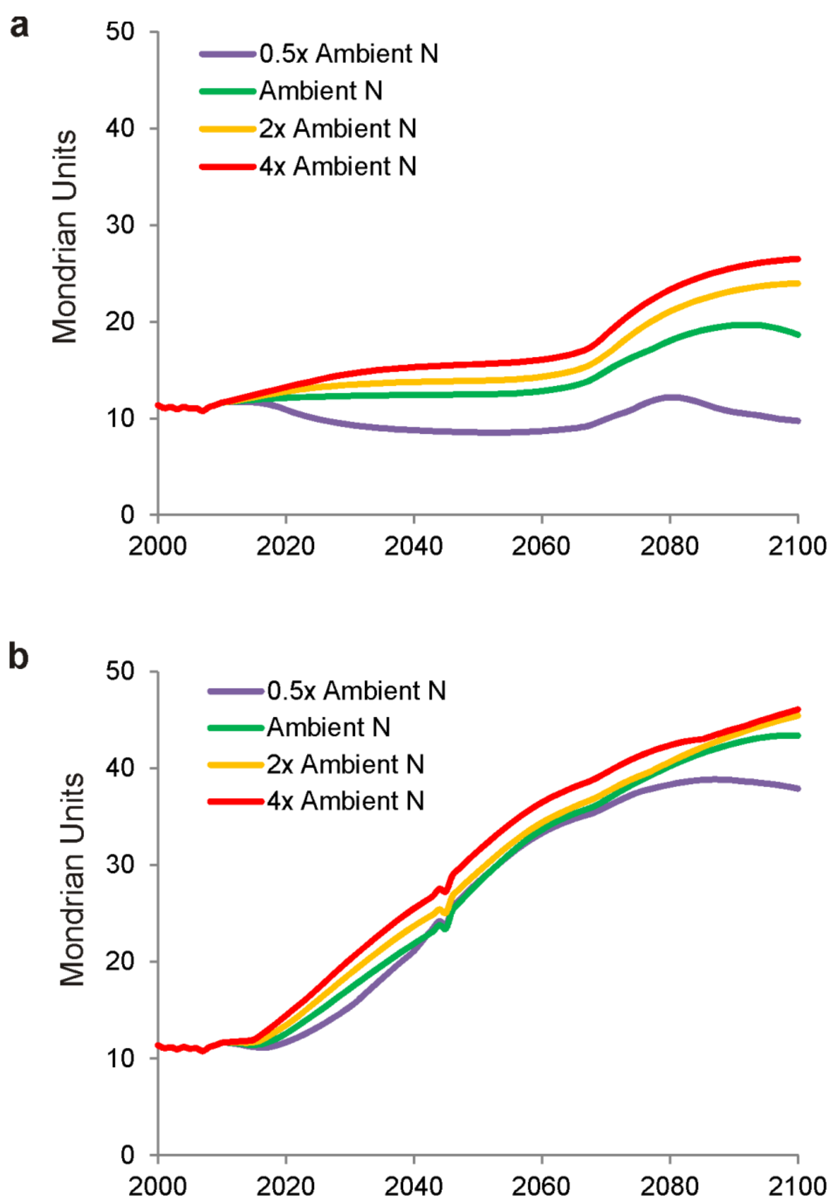

Figure 4-8. Change in species composition (Mondrians) over time for the modeled set of N deposition scenarios under a) no climate change b) +4.6 °C, relative to the scenario of no changes to future climate and assumed background N deposition.

CHAPTER 5 - CONCLUSION

Despite emissions reductions since implementation of the Clean Air Act and its amendments, atmospheric deposition of N and S in the U.S. remains above background levels and is expected to continue to affect forest nutrient cycling and biodiversity. These effects require consideration of alternate future climate and forest management scenarios to understand interactions among these drivers and quantify uncertainty in future ecosystem response. Forest soil and surface waters within the southern Appalachian Mountains are known to be sensitive to the effects of elevated S deposition. In sensitive watersheds of the southern Appalachian Mountains, BC supply is expected to remain below pre-industrial conditions for more than 100 years into the future regardless of changes in S deposition, climate, or tree harvest. Land management options to promote the recovery of BC supply are limited to adjusting tree harvesting practices and fertilizer application. Forests managed for recovery of soil BC supply can be restricted to thinning operations or designated as unsuitable for harvesting. Application of BC fertilizer to watershed soils can also accelerate BC recovery and may be necessary to completely restore historical soil BC supply in acid-sensitive areas. Results presented here can be used as the basis for establishing experimental BC fertilization rates in pilot studies to consider the effectiveness of broad-scale fertilizer application to restore historical soil BC status in the southern Appalachian region.

Estimates of the CL to protect sensitive biotic resources provide an opportunity to evaluate the extent to which decreases in S deposition are sufficient to protect ecosystems from long-term effects associated with acidic deposition. In this study, regional predictions of stream ANC were used to focus CL modeling on streams that were predicted to reflect low-ANC ($< 300\ \mu eq \cdot L^{-1}$) conditions. Although the extent of CL exceedance to protect aquatic resources within the southern Appalachian Mountains has improved between years 2001 and 2011, S deposition in 2011 remained above the CL to protect sensitive aquatic resources in areas found throughout the region. Exceedance was highest in wilderness areas, national parks, and other federal land as compared with private land contained within the southern Appalachian region. The national parks and many of the wilderness areas in this region are federally mandated Class I areas, which receive special protection against air pollution impacts under the Clean Air Act. This information can be used by U.S. EPA to review the adequacy of current NO_x/SO_x National Ambient Air Quality Standards for ecosystem protection.

In addition to acidification effects, N deposition can also affect ecosystems through increased the soil N availability and altered competitive relationships among plant species. Effects of N deposition and climate change on plant biodiversity were estimated based on an application of a process-model. Model results indicated that the CL of N deposition to protect against future changes in subalpine plant community composition in Rocky Mountain NP varied between 1.9 and 3.5 kg N ha^{-1} yr^{-1} based on future climatic conditions. This information can be used by park managers to protect park resources and anticipate future effects on ecosystem services related to tourism and aesthetics that are provided by the high elevation ecosystems of Rocky Mountain NP.

The evaluation of ecosystem impacts in a policy framework requires recognition that the environmental consequences of atmospheric N and S deposition vary with ecosystem type and environmental conditions. The major ecosystem effects known to be associated with elevated N and S deposition are listed in Table 5-1. The CL paradigm can be used to evaluate impacts on ecosystem services associated with these effects (Sullivan and McDonnell 2012). An ecosystem in exceedance of the CL to protect against a given chemical or biological threshold corresponds with a reduction in services provided by the ecosystem. A variety of provisioning, regulating, supporting, and cultural services can be recovered as an ecosystem shifts from exceedance to non-exceedance of the CL (Table 5-2). Thus, understanding impacts of atmospheric deposition on ecosystem services requires knowledge of the inherent ability of an ecosystem to withstand deleterious effects, which can be quantified by the CL and must be considered in the context of other drivers of ecosystem response such as climate change and land management.

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TABLES

Table 5-1. Major chemical and biological effects of atmospheric S and N deposition on natural ecosystems (modified from Sullivan and McDonnell 2012).

Ecosystem	N Effects	S Effects
Low-order streams	Decreased ANC and pH Increased Al_i Eutrophication Decreased fish richness Decreased richness of mayflies and caddisflies	Decreased ANC and pH Increased Al_i Decreased fish richness Decreased richness of mayflies and caddisflies
High-order streams	Eutrophication	None known
Spruce-fir forest	Decreased soil BS Decreased growth and health of red spruce Increased Al in soil solution	Decreased soil BS Decreased growth and health of red spruce Increased Al in soil solution
Northern hardwood forest	Decreased soil BS Increased Al in soil solution Decreased growth, health, and regeneration of sugar maple Decreased growth, health, and abundance of calciphyllic understory plant species Increased risk of insect defoliation	Decreased soil BS Increased Al in soil solution Decreased growth, health, and regeneration of sugar maple Decreased growth, health, and abundance of calciphyllic understory plant species
Wetlands	Changes in plant species composition	Increased methylation of Hg due to increased S supply Increased Hg bioaccumulation in fish and piscivorous wildlife
Grasslands	Changes in plant species composition	None known
Meadows	Changes in plant species composition	None known

Table 5-2. Anticipated ecosystem service benefits to be realized by moving from a state of CL exceedance to non-exceedance (modified from Sullivan and McDonnell 2012).

Ecosystem Service	Anticipated Benefits
Provisioning Services	
Production of maple syrup and related products	Continued or enhanced production of food products
Catch of brook trout and other game fish in sport fishery	Continued or enhanced catch of sport fish
Production of maple wood for furniture and other wood products industry	Continued or enhanced wood production
Production of spruce wood for wood products	Continued or enhanced wood production
Provision of wildlife habitats	Continued provision of habitat for species associated with sugar maple or red spruce trees
Regulating Services	
Climatic regulation	Decreased greenhouse gas production or increased C sequestration can reduce potential for climate warming impacts
Water regulation	Improved tree health in some habitat types can maintain or enhance water storage, reducing the impacts of flooding and providing enhanced stream flow during low flow periods
Erosion regulation	Decreased effect on vegetation cover can limit possible increases in erosion during heavy precipitation events
Supporting Services	
Primary production	Effects on primary production and biomass in N-limited ecosystems can be reduced.

Ecosystem Service	Anticipated Benefits
Nutrient cycling	Decreased soil acidification can limit the depletion from the soil of Ca and other nutrient base cations, which are important for healthy sugar maple, brook trout, and other terrestrial and aquatic species.
	Decreased mineralization, nitrification, and nitrate leaching can limit soil acidification, Al mobilization, and base cation depletion in base-poor terrestrial and aquatic ecosystems.
Trophic interactions	Effects of acidification on aquatic invertebrates, such as mayflies and caddisflies, can be reduced, thereby enhancing food sources for brook trout and other sport fish.
Cultural Services	
	Improved brook trout and other sport fisheries can lead to increased recreational activities and ecotourism.
	Improved sugar maple health can enhance autumn foliage color in forests that contain sugar maple.
	Aesthetic qualities of plant communities can be enhanced through improved health of red spruce, sugar maple, and other acid- and N-sensitive species.
	Iconic species, such as brook trout and sugar maple, can increase in abundance.
	Maple syrup production cottage industry can improve.

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APPENDICES

APPENDIX 1 - DERIVATION OF SSWC MODEL TERMS

Wet base cation deposition was determined from interpolations of National Acid Deposition Program (NADP) measurements (using the approach of Grimm and Lynch [1997] as a three-year average centered on 2002 [data provided by J. Grimm]). Literature values (Baker 1991) of dry:wet deposition ratios for the southern Blue Ridge Mountains were used to estimate dry deposition of base cations from the NADP interpolated wet deposition. Values of wet and dry base cation deposition were combined to yield estimates of total BC_{dep}.

Forest uptake fluxes (Bc_{up}) of the three nutrient base cations (Ca^{2+}, Mg^{2+}, and K^{+}) were estimated from literature values summarized by the U.S Forest Service Southern Research Station (McNulty et al. 2007). To estimate Bc removal from the watershed, estimates of annualized growth rate were used under the assumption that 65% of the tree volume is removed from the site during harvest. This term reflects uptake into woody materials that are removed from the watershed through timber harvest. Uptake into vegetation that subsequently dies on site represents within-watershed recycling; this is not a watershed output. Lands identified as national park, designated Wilderness, and other protected areas were classified as “no harvest”; Bc_{up} was set to zero in such areas. These included areas identified in the Protected Areas Database of the United States (PAD-US v1) constructed by the U.S. Geological Survey (USGS) National Gap Analysis Program (USGS, 2009), corresponding to GAP codes 1 and 2 (Scott et al. 1993).

The ANC_{limit} was calculated as the product of estimated runoff and the designated critical ANC criterion value needed for protection of stream health (e.g., 50 $\mu eq \cdot L^{-1}$). Whereas the ANC criterion value is typically not assumed to vary across the study region, the runoff used together with the critical value to calculate the ANC_{limit} is spatially variable. A robust (R^2=0.95) algorithm was developed using USGS runoff estimates at gaging stations within the study region, combined with elevation, latitude, and orographically corrected precipitation amounts from the Parameter-elevation Regressions on Independent Slopes Model (PRISM; Daly et al. 2002), to estimate fine-scale variation in annual runoff across the study region.

Estimates of BC_w for individual watersheds, to be used as input into SSWC, were generated using three different approaches. These included values extracted directly from the process-based model (MAGIC) calibrations (n=140; Sullivan et al. 2011, McDonnell et al. 2012) and two sets of statistical predictions using RF (as described above and in Povak et al. (2014) and Povak et al. (2013)), one based on available water chemistry plus landscape characteristics and the other based only on landscape characteristics. Water chemistry data were available for 933 streams, representing 9% of the study region, as calculated based on watershed contributing area to each stream sampling site. Therefore, the statistical model that relies on water chemistry can only be used in these limited areas. The statistical model that was based only on landscape characteristics was used to estimate BC_w throughout the entire study region (i.e., all 140,504 topographically determined catchments). Each of these three methods for calculating BC_w was used, where available, to generate CLs using the SSWC model.

Note that the RF statistical model for predicting BC_w across the study region was developed using only 140 spatially unique values of the BC_w response variable, which were generated using the MAGIC model. The RF statistical model used to predict ANC was established with a substantially larger sample size (n=933 stream sampling sites). Furthermore, the sample size used for developing the statistical model for predicting BC_w was dominated by low (< 150 $meq{\cdot}m^{-2}{\cdot}yr^{-1}$, n=132) weathering rates. As a result, the model did not perform well for predicting higher (≥ 150 $meq{\cdot}m^{-2}{\cdot}yr^{-1}$) BC_w (Povak et al. 2014). Regardless of anthropogenic S deposition rates, the general range of stream ANC is controlled mainly by the ability of watershed soils to contribute base cations to drainage water (Charles 1991). This is driven largely by BC_w. Because of this relationship between stream ANC and BC_w and the lack of rigor in the statistical BC_w predictions at high values of BC_w and stream ANC, results from the ANC threshold model were used to focus CL modeling on streams that were predicted to reflect low-ANC (< 300 $\mu eq{\cdot}L^{-1}$) conditions.

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APPENDIX 2 – FORSAFE-VEG MODEL PARAMETERIZATION

Site Description

Rocky Mountain NP is located along the Continental Divide, about 80 km northwest of Denver, Colorado on steep terrain in the Colorado Front Range. Wet atmospheric N deposition in snow and rain in and near this park is among the highest of any area in the Rocky Mountain region (Turk et al. 1992, Heuer et al. 2000). The forest below tree line at Loch Vale is characterized by spruce-fir forest (*Picea englemanii*, *Abies lasiocarpa*) and subalpine meadows. Exposed bedrock, talus, alpine tundra, and permanent snowfields are found at the higher elevations.

The urban corridor that lies along the Colorado Front Range between Fort Collins and Pueblo experienced an increase in human population of about 53% from 1980 to 2000 (Porter and Johnson 2007). The park is exposed to human-caused air pollutants from this corridor due to upslope air flows from the east, as well as reactive N carried from pollutant source regions to the west.

Rocky Mountain NP is one of the most frequently visited national parks, receiving more than 3 million visitors a year. A primary attraction is the biological diversity of high elevation ecosystems, especially those at or above treeline. The colorful summer wildflower bloom is of particular interest to hikers and photographers.

Model Description

Soil nutrient and chemical element concentrations simulated in ForSAFE are dependent on the rates of deposition, ion exchange, mineralization, immobilization, mineral weathering, and nutrient uptake (Belyazid 2006). For this study, uptake processes were adjusted to reflect the relatively undisturbed nature of the study site where tree cover is not managed. Uptake processes were linked to the size of the sapwood, the principal medium for nutrient uptake, and subsequent allocation to plant parts (Dambrine et al. 1995). Sap volume and composition were simulated to determine the maximum size of the potential plant nutrient storage (Sellin 1994, Kravka et al. 1999), which together with actual nutrient storage controls potential uptake.

The dynamic plant community module Veg is based on plant fundamental niches and traits, and uses abiotic drivers from ForSAFE (soil solution chemistry, hydrology, temperature and light at ground level) to simulate plant competition and ultimately the composition of plant communities (Sverdrup et al. 2007, Belyazid et al. 2011a, Belyazid et al. 2011b, Sverdrup et al. 2012). The Veg module requires selection of indicator plant species. Their fundamental ecological niches are specified based on documented and expert knowledge regarding plant growth and reproduction in response to N supply, moisture, temperature, light availability, alkalinity, rooting depth, shading height and palatability to herbivores (Belyazid et al. 2011a). Veg uses these pre-defined fundamental niches and traits to allocate the various plant species to the available space, depending on their strength, which is calculated given the simulated site conditions and competitive ability of each species (Belyazid et al. 2011a). The realized area cover for each simulated plant varies over time as it is influenced by dynamic site condition and competition with other plants at the site.

Nitrogen deposition increases the availability of soil N, which in turn largely regulates species productivity. Under conditions of high N deposition, the pool of N stored in the ecosystem can increase to the point where the internal N cycle can maintain soil N concentrations high enough to change plant responses for decades to centuries, even after the external N input has been reduced or eliminated, unless there are concurrent large removals of N from the reservoirs in the system (Walther et al. 2005, Butterbach-Bahl and Gundersen 2011). Changes in land management, climate, disturbance, or soil acidification may then individually or collectively stimulate N leaching from the terrestrial ecosystem to drainage water.

Vegetation Parameters

The representative vegetation survey used in this study for the Loch Vale treeline site was conducted in the summer of 1999 within a 20 m x 50 m Modified-Whittaker plot

(Sara Simonson, Natural Resource Ecology laboratory, Colorado State University; personal communication, 2012).

The Veg module requires that each modeled plant species be associated with a set of attributes which are used to evaluate plant strength under a given set of environmental conditions. Plant strength as affected by competition largely determines the relative abundance of a given species at the site. For this study, 15 species with observed cover greater than 1% each were parameterized for the following traits: responses to soil N availability, moisture, temperature, and soil acidity; shading tolerance; rooting depth; and shading height (Table A2-1). The plants were parameterized based on results of experimental studies at nearby Chapin Pass and Niwot Ridge (Bowman et al. 1993, Bowman et al. 1995, Bowman et al. 2006, Bowman et al. 2012), with values recommended at an expert workshop in Denver, Colorado (E&S Environmental Chemistry Inc. 2009) and derived from geographic, community, and microhabitat preferences described in floras and range studies (Weber 1976, Komárková 1979, and online Flora of North America (http://www.efloras.org/flora_page.aspx?flora_id=1). Plant temperature and moisture preferences were specified in association with knowledge of microclimate preferences, as well as macroclimate studies of the area (Greenland and Losleben 2001). A cosmopolitan moss species was chosen to represent moss cover at the site and was parameterized using a similar approach as with the plants.

Vegetation Parameter Calibration

The model was run using the initial vegetation parameterizations for the 15 most prevalent plants on the plot (Table A2-1), each having cover of ≥ 1%. The input climate niche for each species was calibrated to observed percent cover for each of the 15 primary species from the 1999 survey. The initial N response parameters were left unadjusted during calibration because these parameters were believed to be most certain based on recent experimental N addition studies (Bowman et al. 2006, 2012). However, soil moisture and temperature niches had been specified using a relatively coarse approach. Therefore, model calibration was achieved primarily by adjusting input soil moisture and temperature niches.

Soils

Soils in the study region were predominately derived from granitic gneiss parent material. Four stratified soil layers with a total depth of 32 cm were modeled, representing the vegetation rooting zone. Soil parameters were derived from data characteristic of a Cryochrept soil group as described by Baron (1992) and Hartman *et al.* (2007). A list of soil properties used for modeling is included in Table A2-2**.**

Air Temperature, Precipitation and Runoff

Daily air temperature and precipitation data for the period 1993 to 2010 were obtained from a weather station located within the Loch Vale watershed (co-located with the National Acid Deposition Program/National Trends Network [NADP/NTN] atmospheric deposition monitoring site CO98; elevation 3159 m). Average minimum and maximum monthly temperature and average monthly precipitation were determined from these data. A temperature decrease of 6^{o} C per 1000 m was applied to account for elevation differences between the Loch Vale weather station and the model site. Estimated average monthly temperature ranged from -8.7^{o} to 12.3^{o} C. Average monthly precipitation ranged from 6.2 to 15.1 cm, with an average total annual precipitation of 107.8 cm. Annual runoff was estimated to be 82% of the annual precipitation (Hartman et al. 2009), with the balance assumed to be evapotranspiration water loss.

Atmospheric Deposition

Background N and S

Total current wet + dry background S deposition was estimated to represent preindustrial conditions in the absence of substantial human impacts. For this study, it was assumed to be 0.25 kg S ha^{-1} yr^{-1} (Sullivan et al. 2005, U.S. Forest Service et al. 2011). Background wet deposition of NO_3^--N and NH_4^+-N were assumed to be 0.40 kg

ha^{-1} yr^{-1} and 0.10 kg ha^{-1} yr^{-1} respectively (Baron 2006, Hartman et al. 2009). Dry:wet deposition ratios (NO_3^- = 0.25, NH_4^+ = 0.07; Hartman et al. 2009) were applied to assumed wet NO_3^- and NH_4^+ background values to estimate background dry deposition, which was summed with wet deposition to determine total background wet + dry deposition (0.61 kg N ha^{-1} yr^{-1}).

Hindcast N and S

Scaling factors for estimating hindcast deposition of S, NO_3^-, and NH_4^+ were taken from Sullivan et al. (2005) and applied to the anthropogenic fractions of deposition reported by Hartman *et al.* (2009). Dry:wet deposition ratios from Hartman et al. (2009) were applied to the scaled wet deposition values for the hindcast years to estimate historical dry deposition, which was summed with wet deposition to determine historical totals of anthropogenic wet + dry deposition. Anthropogenic S, NO_3^-, and NH_4^+ were added to the assumed background total deposition for each ion to obtain estimates of historical total deposition rates from 1856 to 1966.

Recent N and S

Recent (1984–2010) deposition rates for S and N were determined from NADP/NTN wet deposition measurements for each year (http://nadp.sws.uiuc.edu/data/ntndata.aspx). Dry:wet deposition ratios based on nearby Clean Air Status and Trends Network (CASTNET) data (Hartman et al. 2009) were applied to each year of the wet deposition record to obtain recent total deposition rates for each ion. Five-year average (centered on 2008) ambient S and N deposition were estimated to be 1.7 and 3.5 kg ha^{-1} yr^{-1}, respectively.

Base Cations

Hindcast total base cation (BC) deposition rates were obtained from Hartman et al. (2009) and were assumed for this study to be relatively constant from 1856 to 1966 (an untested assumption). Annual wet BC deposition estimates from 1984 to 2010 were obtained from NADP measurements. Wet BC deposition estimates between 1966 and 1984 were estimated by linear interpolation. Dry:wet ratios from Hartman et al. (2009) were used to determine total BC deposition for all years.

Climate

Pre-industrial climate was estimated from the global trends in temperature and precipitation documented by IPCC (2007) . Globally, average surface temperatures have increased by approximately 1 °C between 1900 and 2000 (IPCC, 2007). Prior to 1900, average surface temperatures oscillated within 0.5 °C (IPCC 2007). While the IPCC report provides evidence about the historical evolution of surface temperature, it is less informative about past changes in precipitation. On a global level, it is estimated that precipitation declined by 5% to 20% between 1901 and 2005, and that the proportion of rainfall received as heavy precipitation has increased, suggesting that rainfall is now less evenly distributed than it was in the past (IPCC 2007). This information was used in the simulations to reduce overall historical precipitation and cap extreme precipitation events historically.

Future climate projections of air temperature and precipitation were based on downscaled A2 scenario results from the World Climate Research Programme's (WCRP's) Coupled Model Intercomparison Project phase 3 (CMIP3) multi-model dataset, as referenced in the IPCC Fourth Assessment Report (AR4; IPCC 2007). The downscaled climate projections used in this study were developed using the Bias-Correction Spatial Disaggregation (BCSD) downscaling technique (Maurer et al. 2007) and are available at 1/8th degree resolution (approx. 12 km x 12 km) for the U.S. (http://gdo-dcp.ucllnl.org/downscaled_cmip3_projections).

Sixteen different global climate models have been downscaled using the Maurer et al. (2007) approach. Although the downscaled A2 results vary among models with respect to the magnitude of warming, each model forecasts warmer annual average air temperature for Rocky Mountain NP by the end of this century (Table A2-3).

Downscaled precipitation forecasts predict both drier and wetter precipitation regimes by the end of this century (Table A2-3).

Soil Solution N Dynamics

Simulated soil solution N concentrations among the various N deposition scenarios with no changes in future climate are equivalent over the historical period and become variable into the future in proportion to the level of hypothesized N deposition (Figure A2-1). The reduction in soil solution N towards the end of the simulation period for the scenario of 0.5x ambient N deposition was associated with an increased N uptake efficiency by *Abies lasiocarpa* as result of more favorable climate conditions during the latter portion of the 20th century. This effect on soil solution N is delayed according to the modeled tree growth and establishment period. Under ambient N deposition, continued elevated N deposition causes increased N mineralization. The increased availability of inorganic N partially offsets the increased N uptake by *A. Lasiocarpa.* Under scenarios of elevated N deposition, the soil accumulates sufficient N such that net N mineralization is greater than overstory utilization of the mineralized N. Without this effect from *A. Lasiocarpa,* N leaching towards the latter portion of the 21st century would have been higher under the scenarios of elevated N. Not all of the deposited N does leaches immediately; a portion is taken up and retained in the soil until saturation, which is a gradual process that manifests as increasing availability of soil solution N under the two elevated deposition scenarios.

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TABLES

Table A2-1. Final calibrated vegetation parameters used for scenario modeling at Loch Vale with ForSAFE-Veg.

	Delay Time	Nitrogen			Ca	pH	Moisture			Temperature			Light		Shading	Root Depth	Grazing	
Species	**σ**	**k+**	**k**	**w**	**k_{Ca}**	**pH_{half}**	**W_{min}**	**W_{opt}**	**W_{max}**	**T_{min}**	**T_{opt}**	**T_{max}**	**L_{min}**	**L_{max}**	**h**	**rd**	**k_G**	Growth form
Abies lasiocarpa	10	0.2	3	1	0	3.7	0.5	1.5	4	-2	4	8	0	2	0.3	3	1	tree
Salix candida	10	0.3	15	1	0	4	2.5	3	4	-2	3.2	10	1	4	0.85	3	3	shrub
Rubus parviflorus	10	0.3	20	1	0	4	0.5	1	2.5	-1	3.2	8	1	4	0.5	2	3	shrub
Carex rupestris	10	0.2	15	2	0	4	0.5	2	3	-6	-0.5	2	3	4	0.15	1	2	graminoid
Carex elynoides	20	0.2	8	1	0	4	0.5	1.6	2.6	-7	-2	2	2	4	0.2	2	2	graminoid
Festuca ovina	10	0.4	15	1	0	4	0.5	1	2	-2	2.5	6	3	4	0.2	1	3	graminoid
Calamagrostis purpurascans	10	0.5	15	1	0	4	0.5	0.75	1.5	-6	-3.5	1.5	3	4	0.18	1	1	graminoid
Poa abbreviata	10	0.5	15	1	0	4	0.5	0.7	1.25	-7	-4	2	3	4	0.12	2	3	graminoid
Geum rossii	10	0.2	10	1	0	4	1	1.8	2.8	-7	-2	1	2	4	0.1	2	2	forb
Aquilegia caerulea	10	0.5	15	1	0	4	0.5	1	1.5	-7	3	5	0	4	0.2	2	3	forb
Antennaria rosea	10	0.2	5	1	0	4	0.5	1	2	-2	5	10	2	4	0.08	2	2	forb
Arenaria fendleri	15	0.2	5	1	0	4	0.5	1	1.5	-7	-4	0.5	3	4	0.05	1	3	forb
Minuartia obusiloba	20	0.3	10	1	0	4	1	0.8	1.8	-7	-4	0.5	2	4	0.05	1	3	cushion
Zigadenus elegans	20	0.2	15	1	0	4	0.5	1.75	2.5	-0.5	2.5	5	3	4	0.5	2	0	forb
Aulacomnium palustre	20	0.05	5	1	0	4	2.5	3.5	4	-2	3	8	2	4	0.02	0	1	moss

Table A2-2. Soil properties at Loch Vale used as inputs to ForSAFE.

Soil Properties	Unit	Layer 1	Layer 2	Layer 3	Layer 4
Thickness[a]	m	0.02	0.03	0.12	0.15
Density[a]	kg m^{-3}	750	1000	1250	1500
CEC[a]	keq kg^{-1}	4.7E-4	2.2E-4	2.0E-4	2.0E-4
Mineral surface area	m^2 m^{-3}	814400	1194600	1736640	1336240
Field capacity[a]	m^3 m^{-3}	0.48	0.40	0.35	0.35
Field Saturation[a]	m^3 m^{-3}	0.60	0.50	0.46	0.46
Gibbsite solubility[b]	log(mol $l^{-1})^{-2}$	6.5	7.6	8.2	8.6
Base saturation[c]	Fraction of CEC	0.96	0.74	0.36	0.36
Soil C[c]	%	11.6	2.8	0.9	0.9
Soil N[d]	%	0.77	0.75		
Mineralogy[c]		% of Mineral Surface Area			
Feldspar		16.0	15.0	13.8	13.8
Plagioclase		24.7	26.5	29.6	29.6
Hornblende		0.2	0.5	0.6	0.7
Epidote		0.1	0.5	0.5	0.5
Garnet		0.0	0.1	0.1	0.1
Biotite		4.0	8.8	12.5	12.5
Chlorite		0.0	1.0	2.2	2.2
Apatite		0.0	0.1	0.15	0.15

[a] from Hartman et al. (2009)
[b] generic values based on Warfvinge and Sverdrup (1995)
[c] based on data from Hartman et al. (2007)
[d] from O and A horizon data in Oropeza (2008)

Table A2-3. Climate and N deposition scenarios modeled. Each unique combination among these values was specified in an individual model run.

	Scenario				
Driver	0	1	2	3	4
N deposition	Background	0.5x	Ambient	2x	4x
Temp (deg C)	Ambient	+2.3	+4.6	+6.6	
Precipitation	Ambient	-50%	-10.1%	+22.7%	+50%

FIGURES

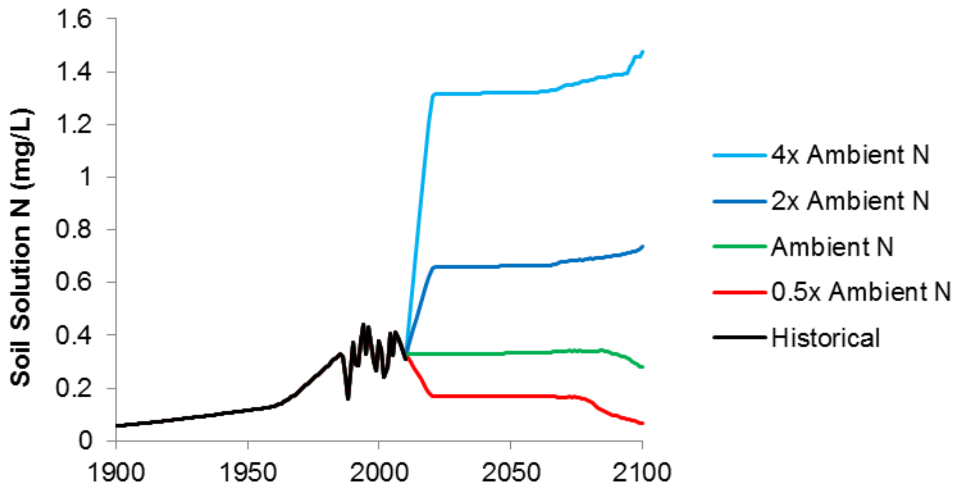

Figure A2-1. Annual average soil solution N concentration for the simulation period 1900 to 2100 among the various N deposition scenarios with no future climate change.