ASSESSMENT OF THE EFFECTS OF ACIDIC DEPOSITION ON FOREST RESOURCES IN THE SOUTHERN APPALACHIAN MOUNTAINS

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List of Acronyms

ANC	Acid neutralizing capacity
BWC	Bold-with-Constraints Emissions Control Strategy
BYB	Beyond Bold Emissions Control Strategy
CALK	Calculated ANC
CEC	Cation exchange capacity
DDF	Dry and occult deposition enhancement factor
DDRP	Direct Delayed Response Project
DOC	Dissolved organic carbon
EMAP	Environmental Monitoring and Assessment Project
EPA	U.S. Environmental Protection Agency
FISH	Fish in Sensitive Habitats Project
MAGIC	Model of Acidification of Groundwater in Catchments
NADP	National Atmospheric Deposition Program
NAPAP	National Acid Precipitation Assessment Program
NSS	National Stream Survey
NSWS	National Surface Water Survey
NuCM	Nutrient Cycling Model
OTW	On-the-Way Emissions Control Strategy
QA/QC	Quality assurance/quality control
SAA	Sum of mineral acid anions
SAMAB	Southern Appalachian Man and the Biosphere program
SAMI	Southern Appalachian Mountains Initiative
SBC	Sum of base cations
SIP	State Implementation Plans
USDA	U.S. Department of Agriculture
USGS	U.S. Geological Survey
VTSSS	Virginia Trout Stream Sensitivity Study

Abstract

Concerns have been cited since the early 1970s about potential forest declines that could result from soil acidification and nutrient deficiency brought about by acidic deposition. Forest resources that are potentially sensitive to the adverse impacts of acidic deposition are found throughout the Southern Appalachian Mountains Initiative (SAMI) domain, in particular at higher elevation sites. Spruce-fir forests are most sensitive, and cover an estimated 0.3% of the area, generally at elevations above 1,400 m.

Acidic deposition in the SAMI region has contributed to a decline in the availability of calcium and other base cations in the soils of acid-sensitive forest ecosystems by the leaching of base cations from foliage and from the primary rooting zone and by the mobilization of aluminum from soils to soil solution and drainage water. Both nitrogen and sulfur deposition have contributed to these mechanisms. Foliar calcium levels and soil solution and root calcium-to-aluminum ratios are considered low to deficient over much of the southern spruce-fir region. Aluminum mobilization from soils that are already acidic can also impede calcium and magnesium uptake and potentially induce deficiencies in these nutrients.

The principal objectives of this terrestrial assessment were to provide quantitative estimates of future change in the acid-base characteristics of forest soils in selected forest stands in response to multiple sulfur and nitrogen emissions control strategies provided by SAMI, and provide a limited qualitative assessment of potential changes in terrestrial ecosystems throughout the SAMI domain based on results for the modeled watersheds and published literature. The Nutrient Cycling Model (NuCM) was applied to three spruce-fir sites, two northern hardwood sites, and two mixed hardwood sites. The assessment involved use of two indirect measures as indicators for forest response modeling: the simulated molar ratio of calcium-to-aluminum in soil water and the base saturation of soils. These indicators have been shown to be associated with sensitivity to acidification and with adverse impacts on forest productivity or forest health.

Model forecasts of future soil solution chemistry were developed for each of the watersheds to which NuCM was calibrated. The dynamics of future atmospheric deposition were specified for these simulations in two ways. An initial analysis was conducted by assuming that future deposition of all ions would stay constant at 1995 levels. In addition, a suite of simulations was based on three regional strategies of emissions controls provided by SAMI. The strategies represent air regulatory requirements being implemented at the time of SAMI's formation,

expected reductions under recent federal regulatory actions, and additional strategies that SAMI might recommend for regional, state, or community-based actions.

Results indicated future soil acidification, as represented by projected reductions in the base saturation of soils at the modeling sites, and decreases in the calcium-to-aluminum molar ratio in soil solution at most sites under all strategies. This result suggests a future deterioration of soil conditions for forest growth and health, especially in the spruce-fir forests, which generally exhibited low calcium-to-aluminum ratios in soil solution in the reference year. All of the modeled spruce-fir sites and one of the modeled hardwood sites displayed calcium-to-aluminum ratios well below the published threshold of 1, indicated in previous studies to be an index of warning that an ecosystem might be entering a zone of increased stress.

Results of the NuCM modeling exercises conducted for this assessment, together with the results of NuCM simulations published for other watersheds in the Southern Appalachian Mountain region, suggest that spruce-fir forests in the region are likely to experience decreased calcium-to-aluminum ratios in soil solution under all strategies of future acidic deposition considered. This is partly because sulfur adsorption on soils is likely to decline, even with dramatically reduced sulfur deposition. In addition, many spruce-fir forests in the region are nitrogen-saturated, and continued nitrogen deposition will contribute to elevated nitrate concentrations in soil water, which will further enhance base cation leaching and mobilization of aluminum from soils to soil solution. These processes will be facilitated by the already low values of base saturation in the soils of many of these forests.

It is not clear, however, to what extent these changes in the chemistry of soils and soil solutions might actually impact forest growth or health. The state of scientific understanding on this topic would suggest that such chemical changes would increase the likelihood that the growth and/or health of spruce-fir forests would be adversely impacted, perhaps making them more susceptible to other stressors associated with such factors as insect pests, pathogens, or extreme climatic conditions. However, the occurrence of low base saturation and calcium-to-aluminum ratio in soil solution will not necessarily be sufficient to cause widespread impacts. Many factors in addition to soil base saturation and soil solution acid-base chemistry are important in this regard.

The state of scientific understanding is less clear with respect to the lower elevation hardwood forests within the SAMI region. Although NuCM projected decreasing soil base saturation and soil solution calcium-to-aluminum ratio at some sites, there are limited data available that would associate such projected chemical changes with adverse forest effects. Available information is not sufficient to draw conclusions regarding the increased likelihood of future effects on the condition of hardwood forests in the region. Certainly, such effects are less likely for hardwood forests than for spruce-fir forests.

Although the modeled sites, especially the spruce-fir sites, generally showed projected future deterioration in soil solution acid-base chemistry, the differences among strategies were small. There are also large uncertainties associated with interpreting the importance of the model projections with respect to the probability of forest impacts. Variability within and among watersheds is extremely high, especially in association with elevation and species mixes. Therefore, the results of these projections are not adequate as a basis for recommending one Emissions Control Strategy over another.

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1.0 INTRODUCTION

Concerns have been cited since the early 1970s about potential forest declines that could result from soil acidification and nutrient deficiency brought about by acidic deposition. The Swedish Report to the United Nations forecasted forest growth declines in northern Europe of 1.5% per year from accelerated calcium leaching caused by acidic deposition (Engstrom et al. 1971, Johnson et al. 1992). This estimate and others like it were based upon treating the soil as a homogeneous, unvegetated block with certain exchangeable cation reserves that would become more acidic, given sufficiently large acid inputs over a sufficient length of time. In addition, concerns have arisen regarding the mobilization of aluminum in forest soils due to inputs of acidic deposition, and the potential toxicity of that aluminum to forest stands (Cronan and Grigal 1995).

Reuss (1976), Sollins et al. (1980), and Ulrich (1980) helped to introduce an ecosystem-level perspective to the acidic deposition-soil acidification issue by noting that acidic deposition is merely an increment to a variety of internal acidification processes (carbonic, organic, and nitric acid production within the soil; humus formation; and plant uptake). These authors proposed that internal acid-generating processes could be described and quantified by constructing "hydrogen ion budgets," a concept later elaborated by Binkley and Richter (1987).

Acidic deposition has contributed to a decline in the availability of calcium and other base cations in the soils of acid-sensitive forest ecosystems by the leaching of base cations from foliage and from the primary rooting zone and by the mobilization of aluminum from soils to soil solution and drainage water (Eager and Adams 1992, NAPAP 1998). Both nitrogen and sulfur deposition have contributed to these mechanisms. Foliar calcium levels and soil and root calcium-to-aluminum ratios are considered low to deficient over much of the southern spruce-fir region (Joslin et al. 1992, Cronan and Grigal 1995, NAPAP 1998). Aluminum mobilization from already acid soils can also impede calcium and magnesium uptake and potentially induce deficiencies in these nutrients.

The lack of recent glaciation in the Southeast has contributed to acid-sensitive, base-poor soils. Because the Southeast was not glaciated during the last glacial period (about 100,000 to 10,000 years ago), low- to mid-elevation soils are largely residual, relatively deep, and highly structured vertically, versus those in glaciated areas to the north. The lower horizons of southeastern soils are also rich in iron and aluminum, which can strongly affect solution chemistry via efficient retention of many negatively charged solutes (e.g. sulfate, phosphate, organic anions associated with dissolved organic carbon [DOC]), of which sulfate is of major interest in the context of acidic deposition. Ultisols represent one of the dominant soil groups in lower elevations of the Southeast; these are characterized by sandy or loamy surface horizons and subsurface horizons that are loamy or clayey in texture. They are typically acidic and are low in base saturation (Adams and Hackney 1992). The mountainous regions of the southeast are dominated by acidic Inceptisols with some incipient acidic Spodosols.

Forest resources that are potentially sensitive to the adverse impacts of acidic deposition are found throughout the SAMI domain, in particular at higher elevation sites. NAPAP (1998) concluded that the only cases of significant forest damage in the United States for which there was strong scientific evidence that acidic deposition was a primary cause included the observed reduced growth of red spruce in the Southern Appalachian Mountains (SA) and increased mortality and decline of red spruce in the Northeast. Within the SAMI domain, spruce-fir forests cover an estimated 0.3% of the area and are generally found at elevations above 1,400 m.

There are no data to suggest that forest resources in the SA, other than high-elevation spruce-fir forests, are currently showing indication of widespread damage directly attributable to acidic deposition. However, there is evidence to suggest that these less sensitive forests can experience gradual loss of calcium and other base cations from forest soils as a consequence of acidic deposition up to the point where the soils become so acidic that further losses are minimal. At that point, these soils are very sensitive to the mobilization of aluminum into soil solution by the introduction of sulfate and/or nitrate. This may reduce forest growth and nutrition over extended periods of time.

High-elevation areas in the SA are often dominated by sandstone and other unreactive bedrock. Base cation production via weathering is limited (Elwood et al. 1991). These soils are largely colluvial and not residual, and they have less iron and less clay and hence lower sulfate retention capacity than most Ultisols. Soils of spruce-fir forests in the SA region tend to have thick organic horizons, high organic matter content in the mineral horizons, and low pH (Joslin et al. 1992). Because of the largely unreactive bedrock, base-poor litter and organic acid anions produced by the conifers, high precipitation, and high leaching rates, soil base saturation in these high-elevation forests tends to be below 10% and the soil cation exchange complex is generally dominated by aluminum (Johnson and Fernandez 1992, Joslin et al. 1992).

The forest Nutrient Cycling Model (NuCM - Liu et al. 1992) model simulates the processes that alter the acid-base chemistry of precipitation as it moves through the forest canopy, into and through watershed soils, and into surface waters. These processes include vegetation growth, litter fall and decay, soil biogeochemical processes, and water routing. The model is calibrated on the basis of input data that include atmospheric deposition, soil chemistry and process rate coefficients, vegetative biomass characteristics, and forest growth coefficients. Model outputs include soil solution chemistry and available nutrients in soils and vegetation. The model can be used to simulate a forest plot or a forested watershed with a stream. The interactions of the various processes simulated by the model determine the ultimate acid-base characteristics of soil solutions and surface waters and the size of forest nutrient pools. For this assessment, NuCM was used to project the responses of seven forest stands in the SAMI region to various emissions control strategies specified by SAMI (Sullivan et al. 2002).

2.0 OBJECTIVES

The principal terrestrial assessment objectives were to:

1. provide quantitative estimates of future change in the acid-base characteristics of forest soils in selected forest stands in response to multiple sulfur and nitrogen emissions control strategies provided by SAMI, and

2. provide a limited qualitative assessment of potential changes in terrestrial ecosystems throughout the SAMI domain based on results for the modeled watersheds and results from previous studies.

The SAMI terrestrial assessment is site-specific and the NuCM model was calibrated to available soil solution data for seven sites (Table 1), and used to project future soil solution chemistry. There was not a statistical basis for site selection or a statistical frame from which to make regional extrapolations for terrestrial systems. However, the model output is discussed within the context of what is known about acidification sensitivity and effects within the region. NuCM was also calibrated to streamwater chemistry data for seven watersheds. The results of these NuCM projections were not considered adequate for examining forest soil response, but are provided in Appendix A.

Table 1. Forest effects simulation watersheds.

Spruce/Fir	Northern Hardwood	Mixed Hardwood
Noland Divide Raven Fork White Top	Fernow WS #4 Raven Fork (Hughes Ridge)	Coweeta WS #2 Joyce Kilmer

3.0 METHODS

Forest resources were stratified using forest vegetation community type. Three sensitivity classes were selected:

- spruce-fir forest
- northern hardwood forest
- mixed hardwood forest

These classes correspond with the expected extent of sensitivity of forest soils to adverse impacts from acidic deposition. Soils and associated vegetation in spruce-fir forests were expected to be most sensitive and mixed hardwood forests least sensitive.

The NuCM model was applied to three spruce-fir sites, two northern hardwood sites, and two mixed hardwood sites. The assessment involved use of indirect measures as indicators for forest response modeling. The simulated ratio of calcium-to-aluminum in soil water (in moles per Liter) and the base saturation of soils were represented quantitatively in response to changes in acidic deposition. These indicators have been shown to be associated with sensitivity to acidification and with adverse impacts on forest productivity or forest health.

Data were obtained, where available, to support application of the NuCM model on a plot-scale. These data included:

- watershed physical characteristics (e.g., aspect and slope);
- forest cover distribution, standing biomass, age, growth rate, and canopy cover;
- mineral composition of soil and bedrock; and
- soil concentrations of exchangeable cations and sulfate adsorption properties.

3.1 Site Description

The availability of data needed for application of NuCM proved to be a significant factor in selecting sites for modeling. The ideal data set for assessing forest effects of acidic deposition is one that includes soil solid phase characteristics and soil solution characteristics at several points in the soil profile based on measurements from soil pits and lysimeters. However, sites with lysimeter data are relatively rare. Consideration was also given to issues of geographic distribution and inclusion in Class I areas in the selection of sites for simulation. The final site selections are listed in Table 1. A map showing the area of the SA considered for this assessment and the locations of all simulated sites is presented in Figure 1. All of these sites are described in detail in the literature. Brief descriptions of each site are presented below.

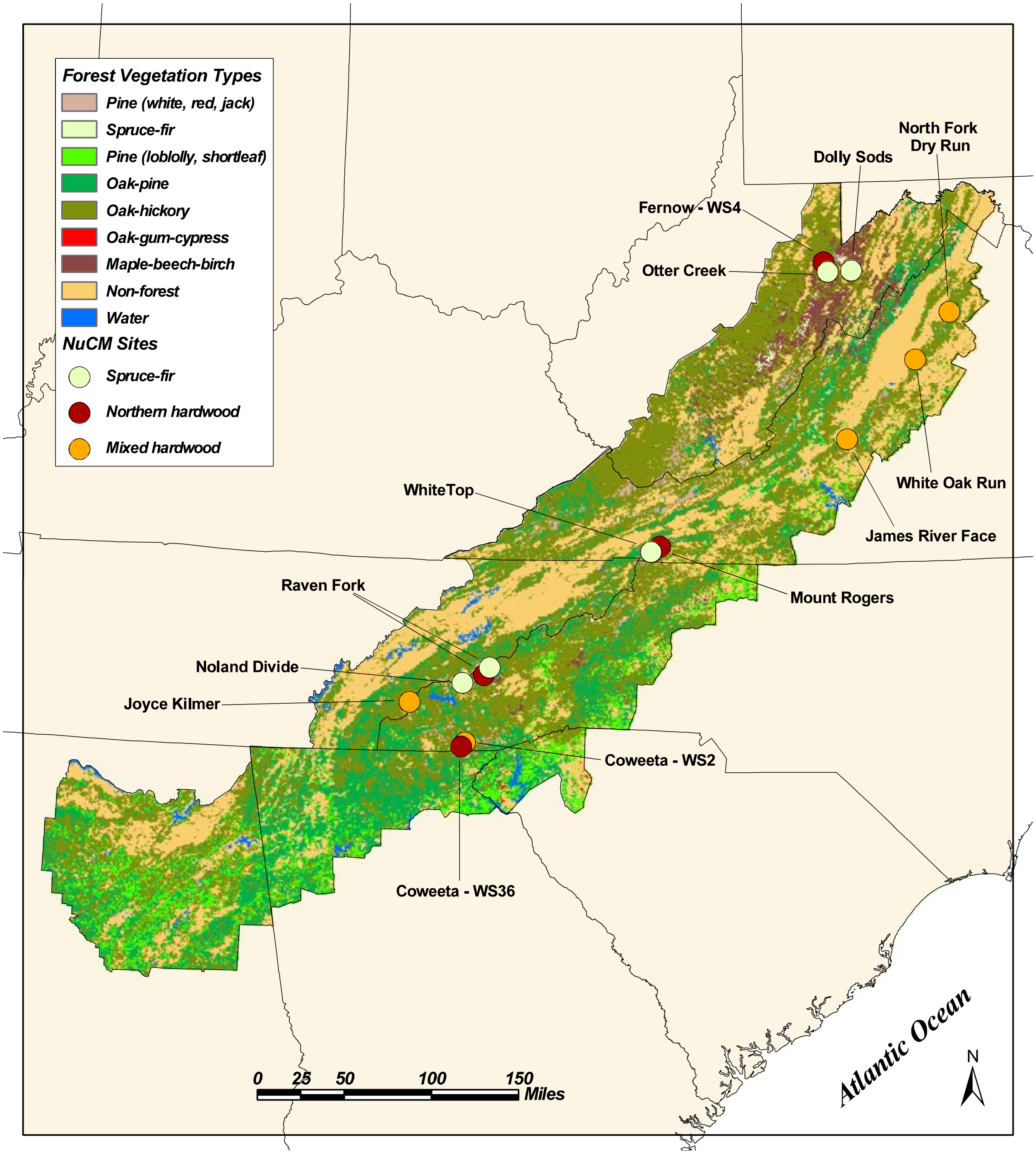


Figure 1. Map showing forest vegetation types (Source: USDA Forest Service) and location of forest sites simulated with the NuCM model.

3.1.1 Noland Divide Spruce

The Noland Divide site consists of an old-growth (200-300 years old) spruce-fir forest in Great Smoky Mountains National Park at an elevation of 1740 m. Annual precipitation at the site averages just over 200 cm. The watershed is underlain by Thunderhead Sandstone, with some phyllite inclusions and quartz veins. Soils have high surficial organic matter, low pH, low base status, and high exchangeable aluminum. The slope at the site is 47%, and observed base saturation values range from 8.6 to 13.6%. The site has no history of commercial logging or fires. Model calibration for Noland Divide was based upon lysimeter data (Johnson et al. 1999).

3.1.2 Raven Fork Spruce

The Raven Fork watershed contains both a spruce-fir site and a northern hardwood site. The spruce site consists of an old-growth spruce-fir forest in Great Smoky Mountains National Park at an elevation of 1730 m. Annual precipitation at the site averages just over 165 cm. The watershed is underlain by colluvium of greywacke and slate. The slope at the site is 58%, and observed base saturation values range from 17 to 36%. The site has no history of commercial logging or fires, but was infested by the balsam wooly adelgid in the 1980s. Model calibration for the Raven Fork spruce site was based upon lysimeter data obtained by the Tennessee Valley Authority.

3.1.3 White Top Mountain Spruce

The White Top Mountain site contains a spruce forest at an elevation of 1658 m in southwestern Virginia. Annual precipitation at the site is just under 130 cm. The site is underlain by the Mt. Rogers formation interspersed with small amounts of greenschist. The Mt. Rogers formation includes multiple rock types, including greenstone, rhyolite, felsic volcanic rock, and conglomerate. The slope at the site is 40%, and the observed base saturation values range from 9 to 30%. The site contains two age classes of trees (100 and 170 years old), but here has been no major logging in the past 100 years, and there have not been any major burns. There was, however some selective logging in the 1950s. The area was grazed, however, from approximately 1800 – 1970. Model calibration for the White Top Mountain site was based upon lysimeter data (Joslin and Wolfe 1992).

3.1.4 Raven Fork Northern Hardwood

As indicated above, the Raven Fork watershed contains both a spruce-fir site and a northern hardwood site. The northern hardwood site consists of a beech, birch, red maple, buckeye forest in Great Smoky Mountains National Park at an elevation of 1555 m. Annual precipitation at the site averages just over 165 cm. The northern hardwood site is underlain by colluvium from phyllite over interbedded phyllite and greywacke residuum. The slope at the site is 70%, and observed base saturation values range from 24 to 55%. The site has been selectively cut, but the fire history is unknown. Model calibration for the Raven Fork northern hardwood site was based upon lysimeter data obtained by the Tennessee Valley Authority.

3.1.5 Fernow Watershed 4 Northern Hardwood

The Fernow Watershed 4 site consists of a hardwood forest made up predominantly of maple with beech and cherry and a minor component of oak in the Fernow Experimental watershed in West Virginia at an elevation of 737 m. Annual precipitation at the site averages just over 150 cm. The site is underlain by the Hampshire formation, which contains mudrock classified as argillaceous. The slope at the site is 11%, and observed base saturation values range from 2 to 11%. The site was last logged in 1911, and dead chestnut was removed in the 1930s. There was a major blowdown at the site within the last 10 years. Model calibration for the Fernow Watershed 4 site was based upon lysimeter data.

3.1.6 Coweeta Watershed 2 Mixed Hardwood

The Coweeta Watershed 2 site consists of an oak forest with minor components of sourwood, maple and hickory in the Coweeta Experimental watershed in western North Carolina at an elevation of 720 m. Annual precipitation at the site averages just over 138 cm. The site is underlain by quartz minerals with minor components of plagioclase, muscovite, and biotite. The slope at the site is 30%, and observed base saturation values range from 14 to 80%. The site has been undisturbed since 1923. Model calibration for the Coweeta Watershed 2 site was based upon lysimeter data (Johnson et al. 1999).

3.1.7 Joyce Kilmer Mixed Hardwood

The Joyce Kilmer site is a mixed hardwood forest adjacent to Santeetlah Creek in the Joyce Kilmer/Slickrock Wilderness Area. The NuCM calibration for this site was provided by the

Forest Service (Jim Vose and Kitty Elliot) and was used unaltered for the NuCM projections reported here.

3.2. Model Description

The NuCM model was designed by a team of investigators in the Integrated Forest Study (see Johnson and Lindberg 1992) and the code was written by TetraTech, Inc (Liu et al. 1991). NuCM depicts the cycling of nitrogen, phosphorus, potassium, calcium, and magnesium at a stand level but also includes the fluxes of major cations (hydrogen, ammonium, calcium, magnesium, potassium, sodium), anions (nitrate, sulfate, ortho-phosphate, chloride, bicarbonate, organic anion), and silica in precipitation, throughfall, and soil solution. Because NuCM was designed primarily for simulating the effects of atmospheric deposition on nutrient cycling processes, its construction emphasizes soil and soil solution chemistry (Liu et al. 1991).

Vegetation is represented in NuCM as one generic conifer and one generic deciduous species in the overstory and one generic understory species. Biomass and nutrient concentrations (foliage, branch, bole, roots) must be specified by the user. Maximum potential vegetative growth in the model is defined by the user and is reduced when nutrients become limiting. An important point in this context is that plant growth in NuCM will be reduced when the availability of any nutrient in any horizon is reduced to a low level regardless of how luxuriant the nutrient levels are in all other horizons; NuCM does not allow root migration to compensate for the depletion of nutrients in any given horizons. Uptake by horizon is specified by the user for each major nutrient and is unchanged during the simulation regardless of whether one horizon becomes depleted or not. Thus, a very important part of calibration is to insure that uptake by horizon for each nutrient is set such that no horizon becomes depleted while other horizons remain high. This requires repeated trials for each nutrient. Even after calibration, NuCM will show forest growth reduction under certain scenarios when one horizon is depleted.

NuCM simulates the forest floor from user-specified litterfall inputs (which are assumed constant over time) and user-specified litter decay. Litter decay is represented as a four stage process each of which is represented by a temperature-dependent first order equation of the form:

$$\frac{-\,dC}{dt} = -\,k\theta\,t^{(T-20)}\mathbf{C_o} \qquad (1)$$

where C = the component in question; t = time; k = reaction rate coefficient (user-defined); θ = temperature correction factor; C_o = initial content of component C; and T = soil temperature (°C).

The soil includes multiple layers (up to 10) and up to 4 soil minerals, and each layer can have different physical and chemical characteristics. The user defines bulk density, cation exchange capacity, exchangeable cations, adsorbed phosphate and sulfate, and four soil minerals and their composition. These inputs define the initial soil exchangeable/adsorbed pools and total pools. Initial total soil nitrogen pools are simulated from litterfall and decay, as described above, and user-defined carbon to nitrogen ratios. One limitation of the NuCM model is that it does not allow for coarse fragments (>2mm), and thus the user must reduce soil nutrient pools to allow for coarse fragment content by either reducing horizon thickness or by reducing concentrations. Reducing horizon thickness is preferable, but also raises concerns over potential artifacts introduced into the hydrology calibration and simulations. Vegetation, litter, and soil pools change over a simulation in response to growth, litterfall and decomposition, and nutrient fluxes via deposition, leaching and weathering, as described below.

Translocation, defined as the removal of nutrients from foliage prior to litterfall, is user-specified (as a percentage of foliage nutrient content). Translocation operates only on the deciduous species in the model at present (Kvindesland 1997). Maximum uptake is calculated from biomass and nutrient concentrations; actual uptake is equal to this maximum value when sufficient nutrients are available and reduced when nutrients become limiting. Reduced uptake first allows reduced nutrient concentrations in plant tissues (by a user-specified percentage) then causes a reduction in growth. Net foliar exudation and leaching rates are simulated in the model as proportional to foliar concentrations and user-defined coefficients. The user must define the fraction of annual uptake that occurs during each month of the year. In practice, this is never known and must be estimated. It has proven to be a very sensitive calibration parameter as well as a very useful research tool.

Within the soil, nitrification is represented in the form of a pH- and temperature-dependent Michaelis-Menton rate expression:

$$u = \alpha v\ ([NH_4^+]/Ks + [NH_4^+]))\theta_N^{(T-20)} \qquad (2)$$

where u = nitrification rate; α = pH dependency factor; v = maximum nitrification rate at 20^{o}C; Ks = half saturation constant; and $\theta_N^{(T-20)}$ is the temperature-dependency factor where T= soil temperature in ^{o}C. Mineral weathering reactions are described in the model using rate expressions with dependencies on the mass of mineral present and solution-phase hydrogen-ion concentration taken to a fractional power. The user must specify four soil minerals, their stoichiometry, and their weathering rates, and the hydrogen ion dependency of the weathering rates. The weathering algorithms in NuCM are similar in structure to those in the ILWAS model. This formulation was based on the belief that weathering rates cannot be either accurately measured or simulated, and thus should be user-specified in the NuCM model. This is one way in which the NuCM model contrasts significantly with the MAGIC model, wherein the latter depicts weathering and cation exchange from one generic mineral pool and base cation availability is determined in the calibration process.

Cation exchange in the soil is represented by the Gapon equation:

$$\frac{[EC^{a+}]\,(C^{b+})^{1/b}}{[EC^{b+}]\,(C^{a+})^{1/a}} = Kgp \tag{3}$$

where parentheses designate soil solution activity, E = exchange phase equivalent fraction, C^{a+} = cation of valence a, C^{b+} = cation of valence b, and Kgp = selectivity coefficient. The model simulates the noncompetitive adsorption of sulfate, phosphate, and organic acid. Phosphate adsorption in the model is represented by a linear isotherm. Sulfate adsorption can be simulated using either linear or Langmuir adsorption isotherms; the Langmuir isotherm was used in these simulations. Unlike most models of its kind (Prenzel 1994), NuCM simulates pH-dependent sulfate adsorption, which is a feature that we feel is both realistic and important for SAMI predictions. There is no question that sulfate adsorption is pH-dependent, and thus as soil acidity changes over time, so will sulfate retention in soils.

Precipitation in the NuCM model is defined by input meteorological files (typically 1 to 5 years long) which are repeated in order to generate long-term simulations. Precipitation is routed through the canopy and soil layers, and evapotranspiration, deep seepage, and lateral flow are simulated. Potential evapotranspiration (ETp) is calculated as:

$$Etp = (Fet/n)(TmCeHc) \tag{4}$$

Where Fet = the evapotranspiration factor, which is a function of latitude (r) (Hargreaves 1974); Fet = 2.322 - 0.0115 r, at $r \leq 35^o$; Fet = 3.434 - 0.0434 r, at $r \geq 35^o$; Tm = mean ambient daily temperature (°F); Hc = a humidity correction factor (unitless); Ce = a unitless calibration factor. The movement of water through the system is simulated using the continuity equation, Darcy's equation for permeable media flow, and Manning's equation for free surface flow. Percolation occurs between layers as a function of layer permeabilities and differences in moisture content. Nutrient pools associated with soil solution, the ion exchange complex, minerals, and soil organic matter are all tracked explicitly by NuCM. Leaching is calculated from soil water flux and simulated soil solution concentrations using the soil chemical and biological algorithms defined above for each soil horizon.

NuCM output can be viewed on user-selected screens displaying biomass, solution concentrations, soil concentrations, water fluxes, and nutrient pools and fluxes. Output files (ASCII) can also be produced which contain values displayed on the screens. NuCM provides outputs of solution chemistry on a daily, weekly, or monthly basis, according to user preference. Simulated biomass and soil exchangeable/adsorbed nutrients are provided on a monthly basis, and simulated nutrient pools and fluxes are provided on a yearly basis.

3.3 Model Calibration

Application of the NuCM model follows a fixed sequence. The first step involves preparation of the meteorological and deposition quality data files. The NuCM hydrologic module operates on a daily timestep, and thus daily precipitation quantities and maximum and minimum temperatures are required for the meteorological data input file. Estimates of current and recent wet deposition of sulfur and nitrogen for calibration NuCM were provided by Jim Lynch, using the model of Grimm and Lynch (1997). Dry and occult (cloud) deposition estimates were derived using the ASTRAP atmospheric transport and deposition model (see Sullivan et al. 2002 for details). The deposition quality data file contains monthly flow-weighted mean concentrations of all constituents simulated by NuCM in wet, dry, and cloudwater deposition.

After the meteorological and deposition input files have been prepared in a format suitable for input to the model, the characteristics of the forested watershed system to be simulated are input. These data include the physical characteristics of the system (e.g., surface area, slope, elevation, stream length), biomass characteristics of the stand (e.g., standing biomass, percent cover, vegetation stoichiometric characteristics), and soil chemical characteristics (minerals present, cation exchange capacity, soil organic matter characteristics). Ideally these data should all be directly from the system being simulated. However if any specific parameters are missing, data from nearby, similar systems can be substituted.

Once all of the input data have been entered, the hydrologic calibration process begins. The first step in hydrologic calibration is establishing the value of the annual evapotranspiration coefficient. This is done by running the model and comparing simulated and observed cumulative outflow, and then altering the value of the coefficient until the values match. If observed outflow data are not available or if some of the data are missing, comparison to runoff values for nearby basins can be used to help establish this parameter value. Once the value of the annual evapotranspiration coefficient is established, hydrologic calibration proceeds with seasonal and instantaneous comparisons. The goal in this process is to match peaks in flows and flow recessions following storms or snowmelt events. This insures proper routing of water through the soil layers and that the proper biogeochemical processes are invoked.

Following hydrologic calibration, the chemical calibration process begins. The NuCM chemistry module can be operated on a daily, weekly, or monthly basis, depending on the frequency of the observed data. The chemical calibration process begins by balancing the soil organic matter pools such that they are neither aggrading nor being depleted at too high a rate. This is done by adjusting litterfall rates and the rates of decay of organic matter. Once these pools are in balance, the soil solution concentrations are evaluated to determine whether the soil selectivity coefficients are properly set for each individual layer. Soil solution concentrations are relatively stable over short (3-5 year) periods, and major deviations in simulated concentrations (aside from within-year variability) indicate that the cation exchange complex has not been properly equilibrated. The next phase of chemical calibration is to compare simulated and observed soil solution concentrations versus time. Deviations between simulated and observed values are typically addressed by changing rate constants for rate-limited reactions. Allocation of the percent of annual nutrient uptake for each month is an especially important, user-specified aspect of calibration, both in terms of forest growth and soil solution chemistry.

Both of the Raven Fork sites were calibrated based on soil solution observations made in 1981. Deposition data from 1990 were used as a basis for calibration because 1981 deposition data were unavailable. The White Top site was calibrated based upon deposition data and soil solution observations from 1993. The simulated concentrations of aluminum and Ca/Al ratio were consistent with field observations. Calibration of the Noland Divide spruce site was based upon soil solution and deposition data collected as part of the Integrated Forest Study from 1986-1989 (Johnson and Lindberg 1992). The Fernow northern hardwood site was calibrated based upon soil solution and deposition data for the period 1991-1995. Observations from this site also underscore the wide variability seen in soil solution concentrations for a single site. Data used for the calibration of the Coweeta Watershed 2 mixed hardwood site were from the same study as the Noland Divide site. The data are presented as comparisons of the observed 3-year average concentration and the simulated 3-year average concentration, because this was the form in which the data were available. One of the significant differences between the mixed hardwood site and the generally higher elevation spruce and northern hardwood sites is in the observed base saturation values. In general, and at Coweeta Watershed 2 specifically, the base saturation values are significantly higher for mixed hardwood sites.

3.4 Emissions and Deposition Strategies

Model forecasts of future soil solution chemistry were developed for each of the watersheds to which NuCM was calibrated. The dynamics of future atmospheric deposition were specified for these simulations in two ways. An initial analysis was conducted by assuming that future deposition of all ions would stay constant at 1995 levels. Results of this constant deposition scenario were used as a base case against which the results of emissions control strategies could be evaluated. In addition, a suite of simulations was based on three regional strategies of emissions controls provided by SAMI.

SAMI characterized atmospheric emissions of 12 emission precursors to ozone, acid deposition, and atmospheric aerosols, including SO_2, NO_x, and NH_3. These emissions were examined for five source sectors:

- utilities
- industry
- highway vehicles
- non-road engines, and
- area sources.

It was assumed for the reference strategy that current land use patterns and lifestyle behaviors would continue into the future. For the bold strategies, however, it was assumed that highway vehicle use would decline. Emissions projections for nitrogen and sulfur are shown in Figure 2 for the eight SAMI states (Odman et al. 2002a).

The strategies represent air regulatory requirements being implemented at the time of SAMI's formation, expected reductions under recent federal regulatory actions, and additional strategies that SAMI might recommend for regional, state, or community-based actions (Ogburn et al. 2001). Emissions inventories were developed for precursors of acidic deposition and ozone for the major source categories throughout the eastern U.S. Nine, week-long episodes of air quality were selected to represent air quality and meteorological conditions over the five-year period, 1990-1995. Air quality modeling results for these episodes were combined to represent seasonal and annual air quality metrics, which were projected to the years 2010 and 2040. Future growths of emissions source subcategories were estimated based on projections of

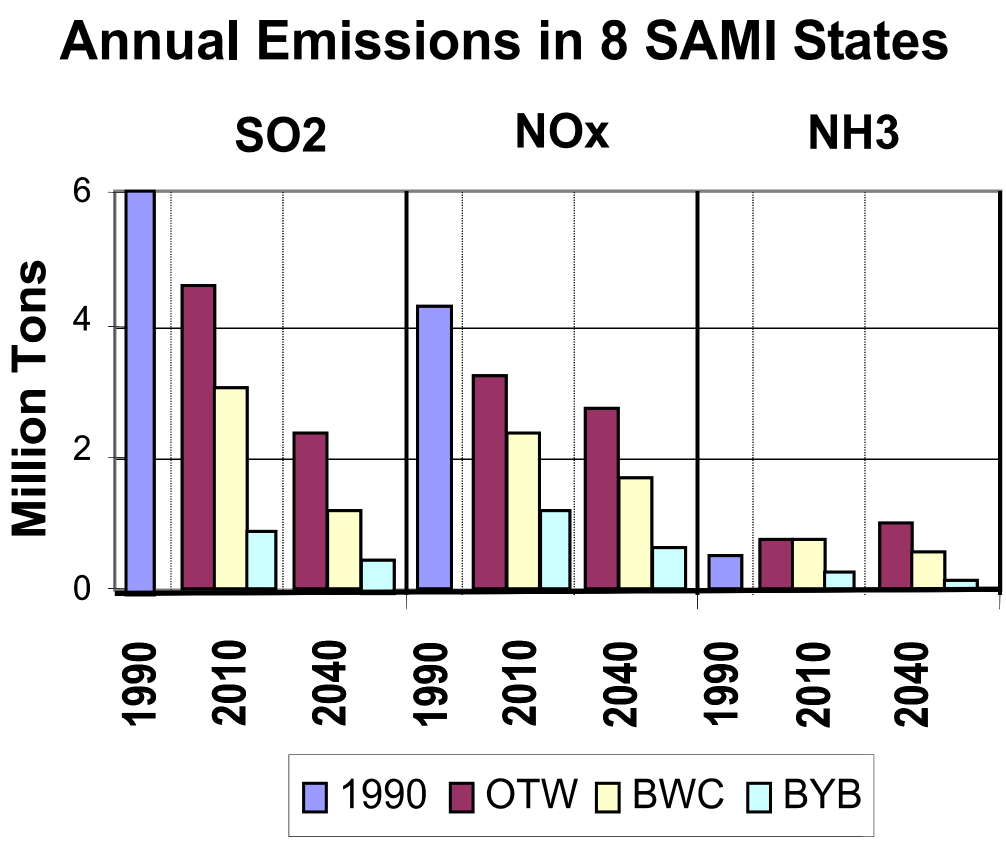


Figure 2. Annual emissions estimates for the eight SAMI states (Odman et al. 2002a)

population, industrial growth, electricity generation, non-utility point sources, vehicles, and area sources, using projections from the Bureau of Economic Analysis, U.S. EPA, U.S. Department of Transportation, U.S. Department of Energy, and Power Systems Research (Ogburn et al. 2001).

The spatial dynamics of these emissions control options resulted in varying estimated future changes in sulfur and nitrogen deposition at different locations within the SAMI domain. The strategies that were implemented for this assessment were designated On-the-Way (OTW), Bold-with-Constraints (BWC), and Beyond Bold (BYB). OTW is the reference strategy that represents SAMI's best estimates for acidic deposition controls that have been promulgated and are relatively certain. These include the 1990 Clean Air Act Amendments and several mobile source reductions, including EPA's call for revised State Implementation Plans (SIP). OTW was applied to all eastern states, focused on utility and highway vehicle sectors. BWC and BYB strategies assume progressively larger emissions reductions. They are targeted only to the eight SAMI states, but cover all emissions sectors. All strategies assume existing or anticipated new technology, and replacement of utility units at age 65 years with cleaner technology.

Under the OTW strategy, sulfur dioxide emissions are projected to decline substantially, especially by 2040. Nitrogen oxide emissions are also projected to decrease, but they are offset by projected emissions increases in ammonia-nitrogen. Under the BWC and BYB strategies, nitrogen emissions are projected to decrease.

3.4.1 Analysis of Constant 1995 Deposition Scenario

A scenario was implemented that involved maintaining constant deposition into the future at 1995 levels for all major ions. Scenario results provide a metric for judging which watersheds would be most responsive to continued constant and/or future changes in sulfur and nitrogen deposition. Results of the model simulations that assumed constant deposition at 1995 levels also provided a baseline against which the results of emissions control strategy simulations could be compared.

3.4.2 Strategy Runs

In conjunction with SAMI's atmospheric modeling contractor, an approach was developed for supplying future deposition inputs sufficient for the NuCM input requirements (Odman et al. 2002b). The approach utilized specified emissions levels for the years 1995, 2010, and 2040, under baseline assumptions, and several alternative emissions control strategies, and interpolation (ramping) procedures for specifying deposition for the intervening years. These estimates of deposition, shown in Figure 3 for sulfur and nitrogen, were then used in our modeling to represent deposition to each modeled forest stand. The SAMI strategies did not predict substantial changes in base cation deposition. For each year of the simulation and each strategy, NuCM output included soil base saturation, Ca:Al molar ratio in soil solution, and the concentrations of major ions in soil solution. Outputs were saved of the responses for the years 1995, 2010, 2020, 2030, and 2040.

4.0 RESULTS

NuCM was successfully calibrated to soil solution data for six sites. Comparisons between simulated and observed solution chemistry were judged to be acceptable, and are presented below. However, there was considerable variability in most of the measured soils data, as is commonly found. The plots present all of the observed concentrations available during the calibration period for each site in order to illustrate the uncertainties. Simulations were based upon a scenario of constant deposition at 1995 levels and each of the three Emissions Control Strategies. Forest response was evaluated in terms of changes in projected soil base saturation, and projected soil solution nitrate and sulfate concentrations and calcium-to-aluminum molar ratio in the rooting zone of the trees (A-horizon). Simulated and observed values are illustrated in Figures 4 through 16. Observed values were not available for the Joyce Kilmer calibration.

Nitrate concentrations in the soil solution within the rooting zone of all three spruce-fir stands were projected to decline under all strategies (Figures 17 through 19). The differences in projected soil solution nitrate concentration among strategies were generally small, however, and concentrations were projected to remain high (greater than about 100 μeq/L) in the future at all spruce-fir sites. Results of the simulations for the hardwood forest sites were more variable, but differences among strategies were also relatively small. At the two northern hardwood sites, simulated soil solution nitrate concentrations were above 100 μeq/L in the base year. They declined to below 20 μeq/L in all strategies at the Raven Fork site, and declined in all strategies at the Fernow site until 2010, after which they increased again to near base year values (Figures 20 and 21). Base year nitrate concentrations at the mixed hardwood site were low (< 20 μeq/L) and remained low through the simulations (Figure 22).

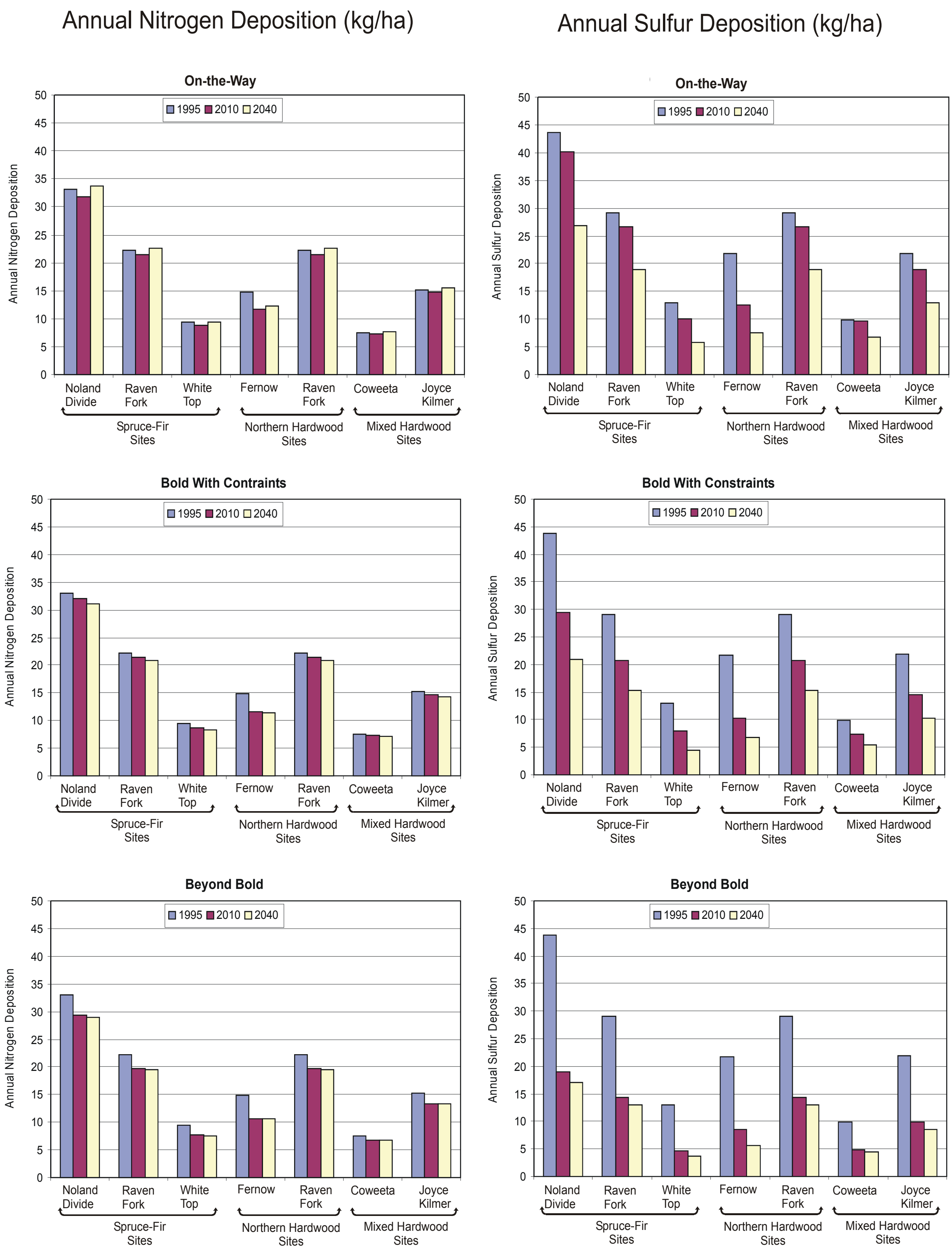


Figure 3. Estimated total sulfur and nitrogen deposition at each of the terrestrial modeling sites and for each of the strategies in the years 1995, 2010, and 2040.

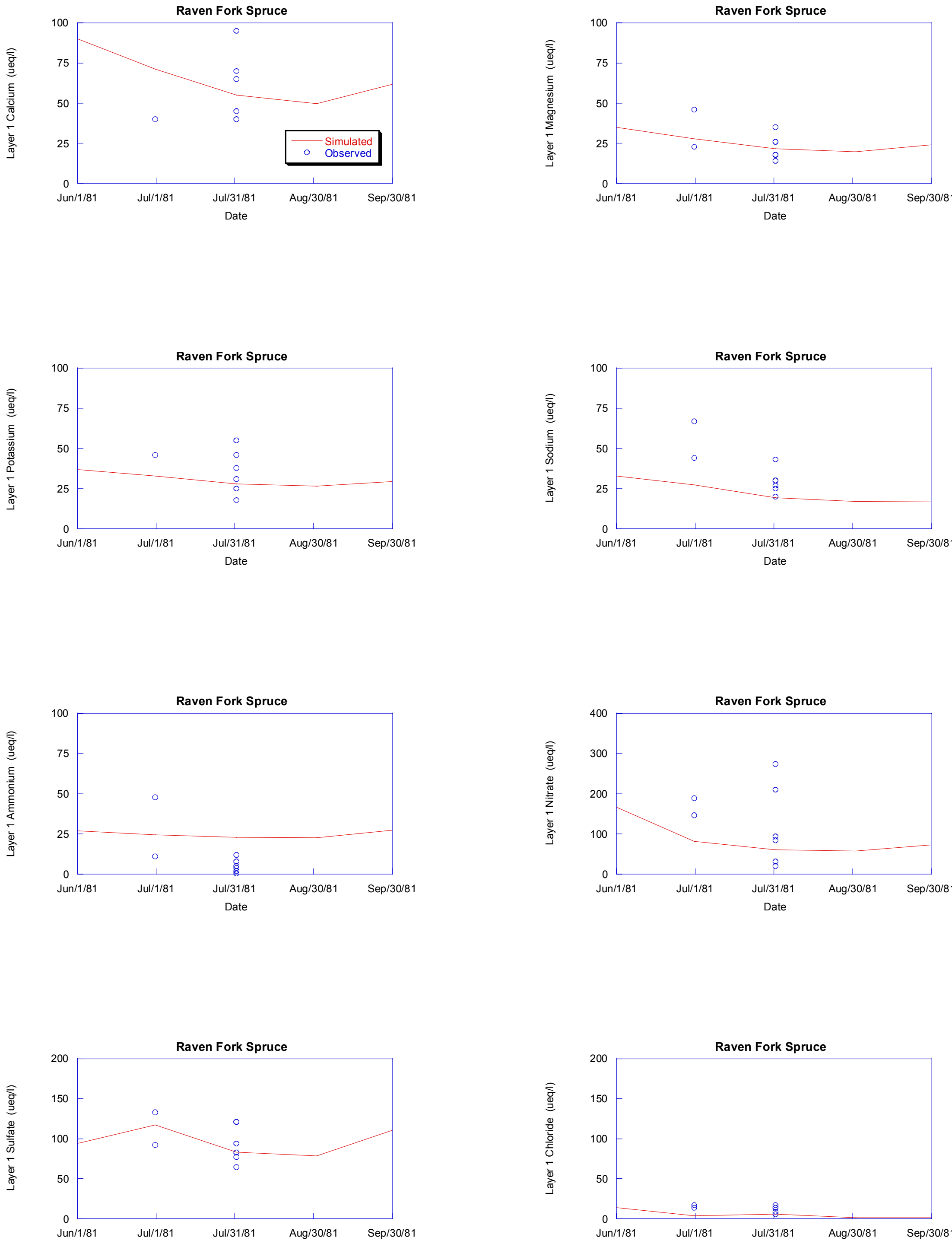


Figure 4. Simulated versus observed concentrations for the O horizon at the Raven Fork Spruce Site.

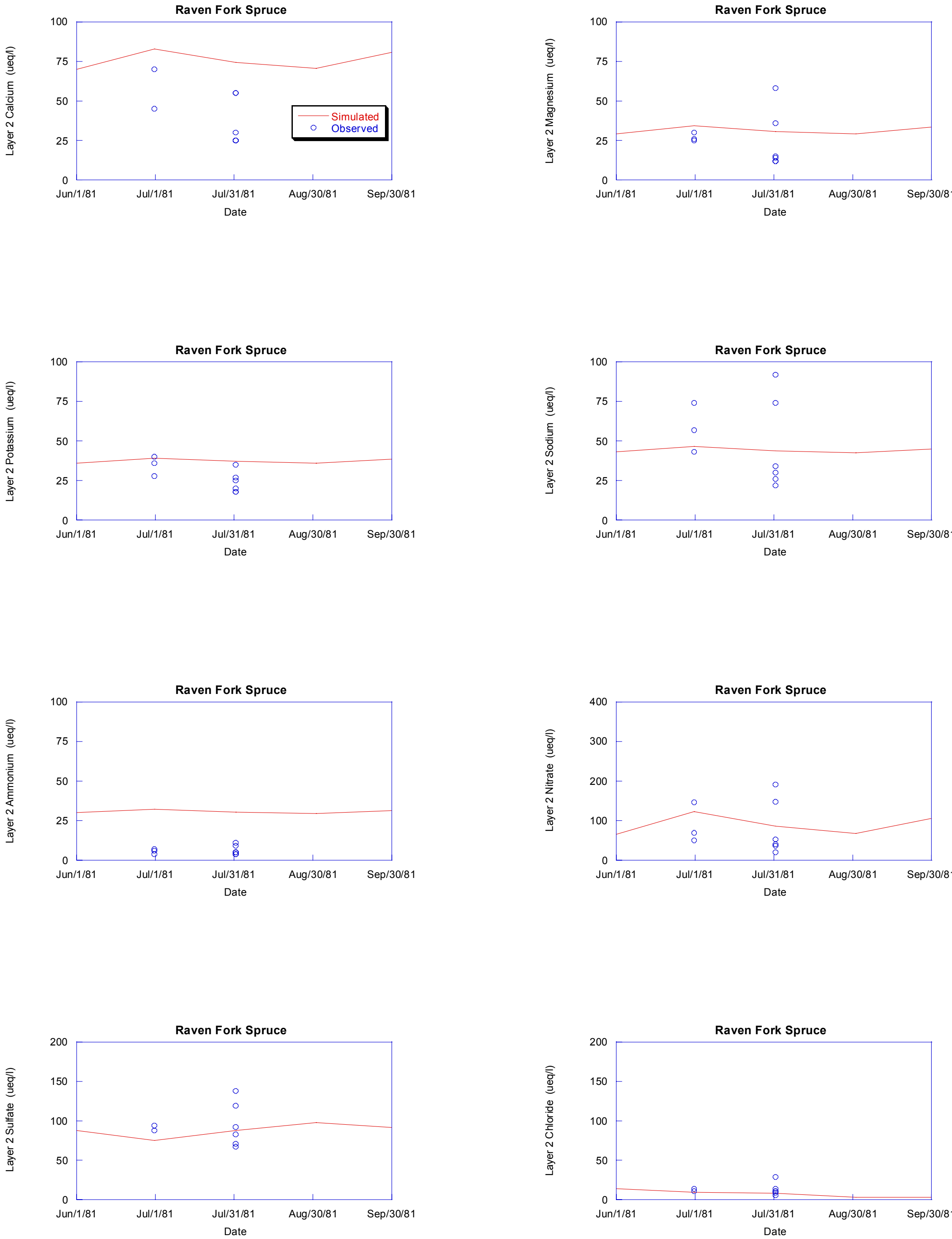


Figure 5. Simulated versus observed concentrations for the A horizon at the Raven Fork Spruce Site.

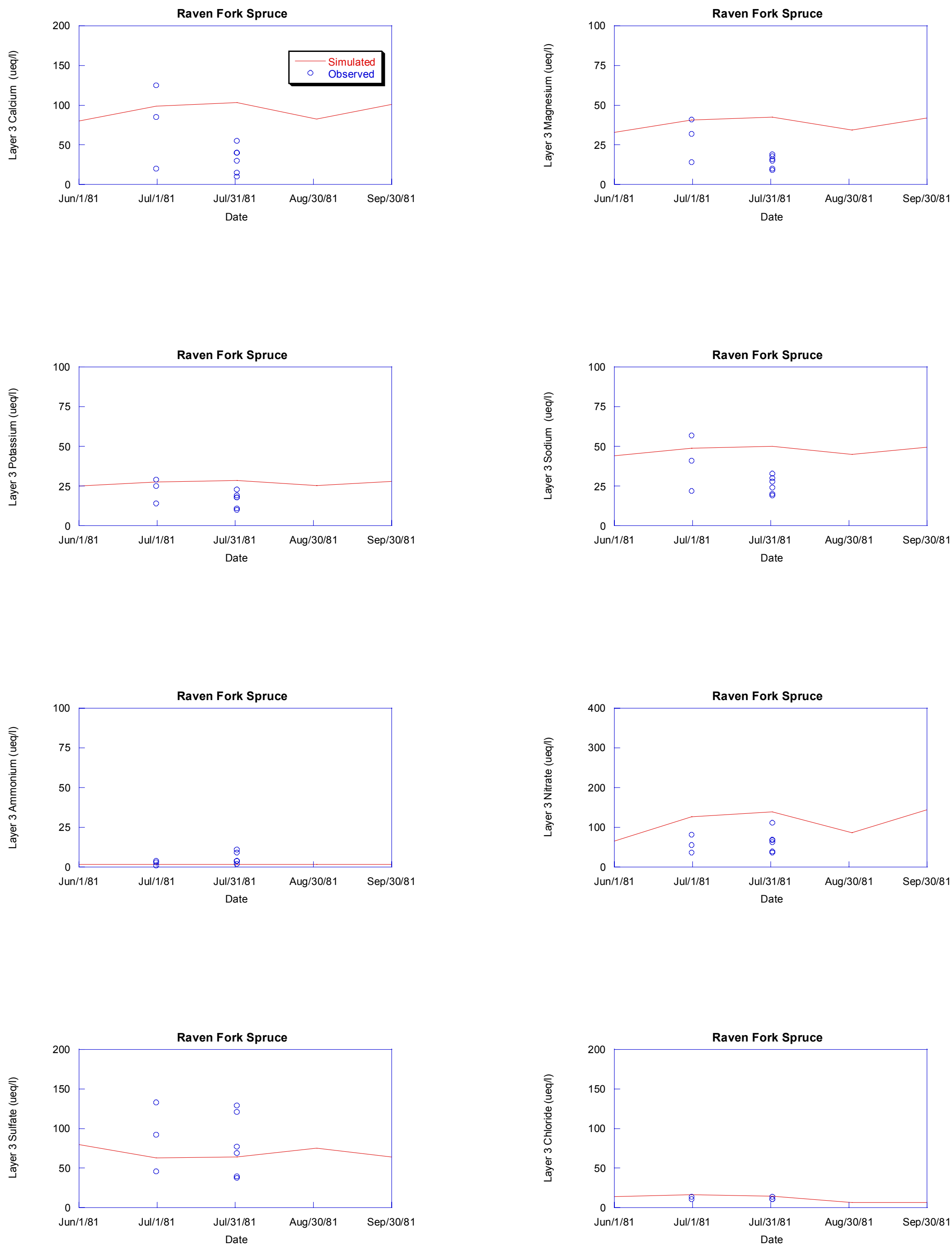


Figure 6. Simulated versus observed concentrations for the B horizon at the Raven Fork Spruce Site.

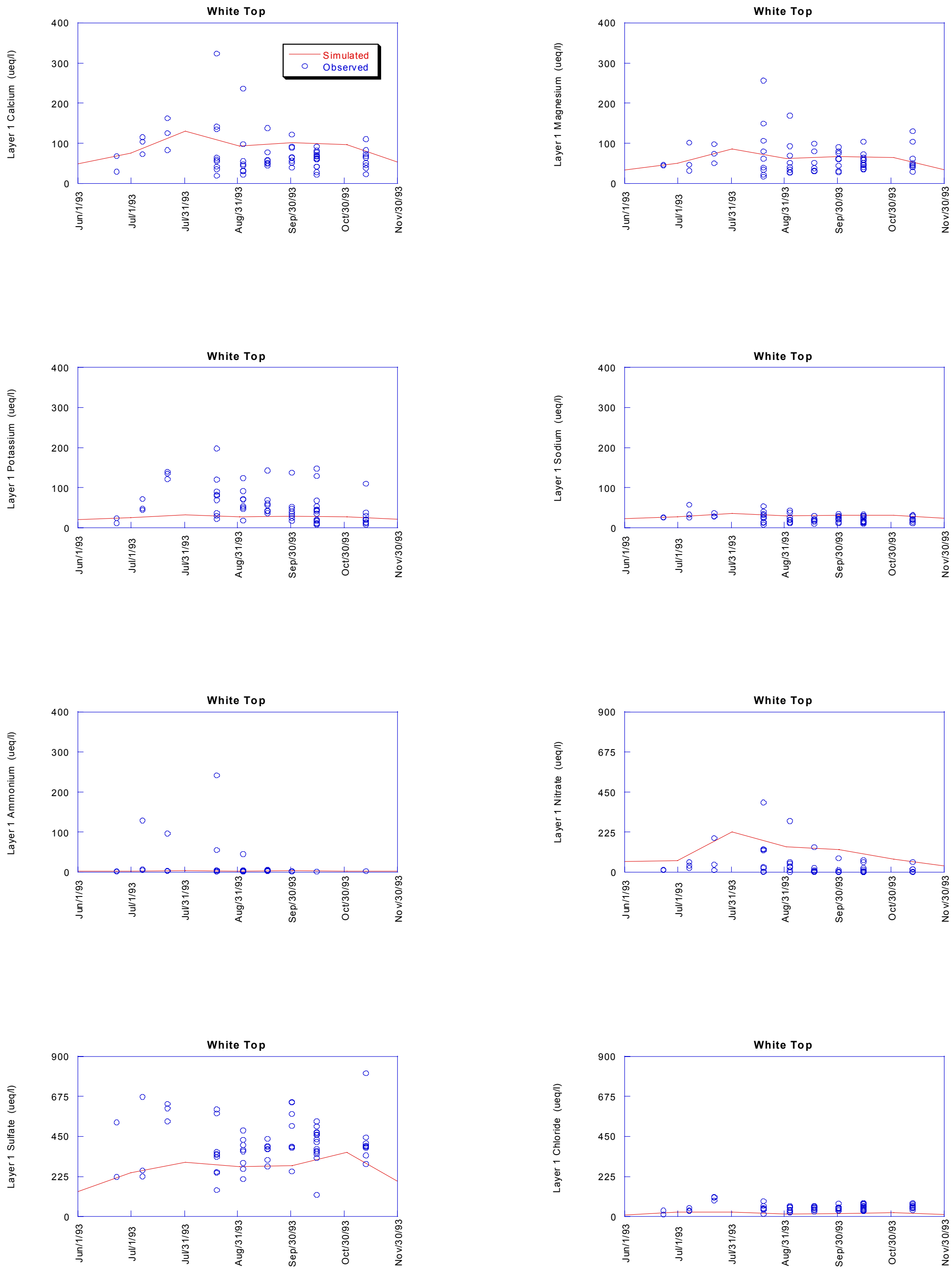


Figure 7. Simulated versus observed concentrations for the O horizon at the White Top Spruce Site.

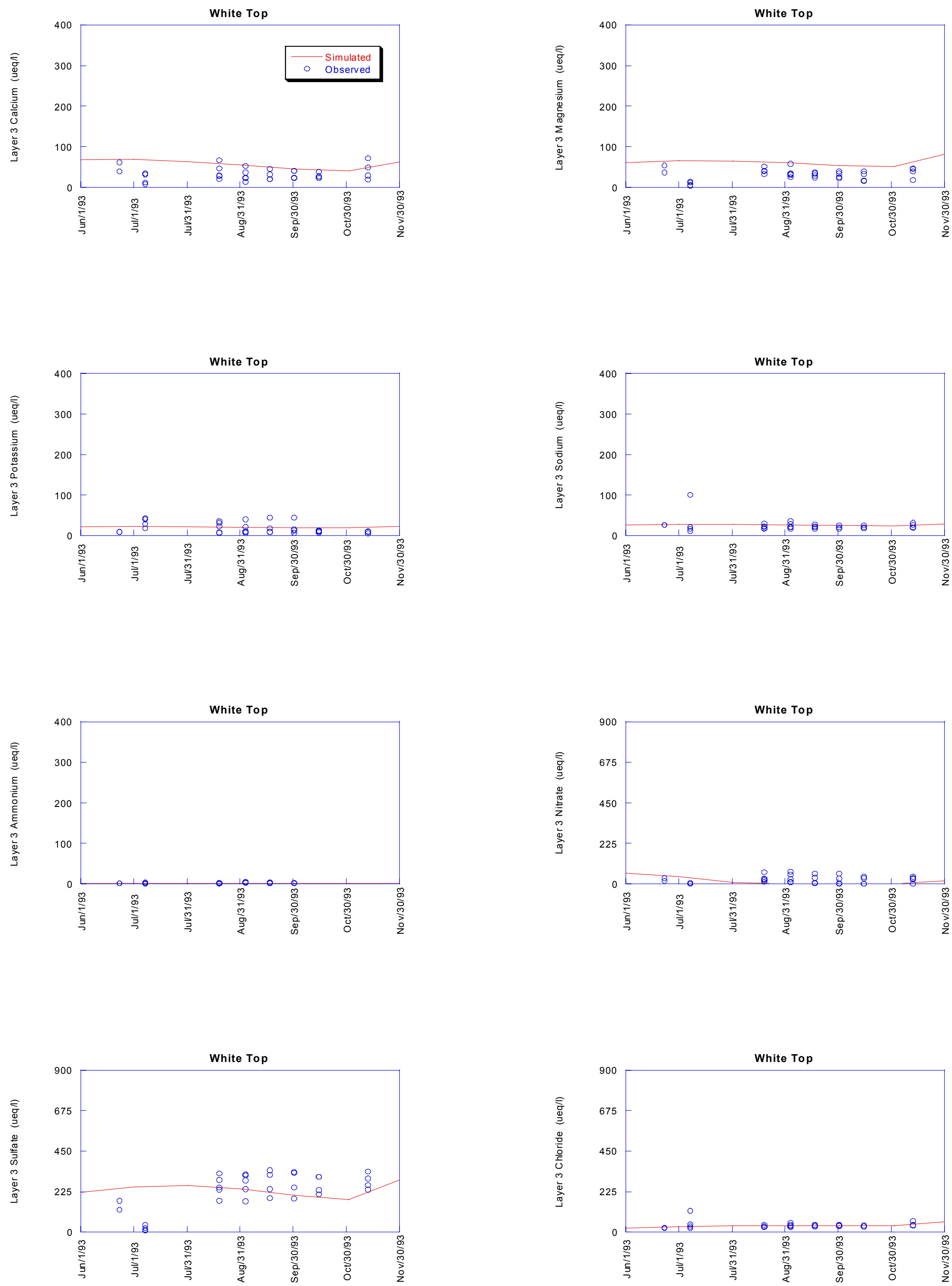


Figure 8. Simulated versus observed concentrations for the B horizon at the White Top Spruce Site.

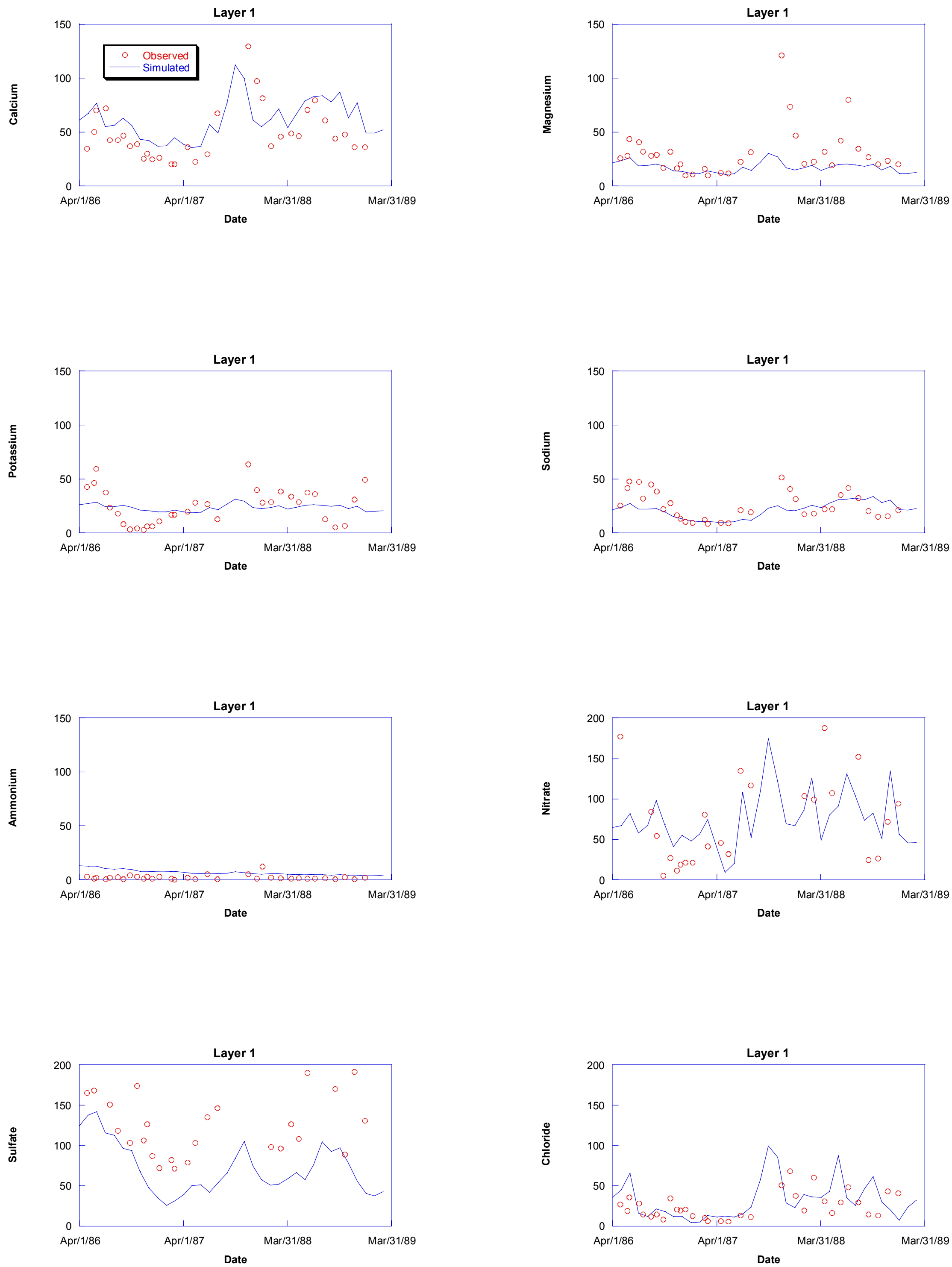


Figure 9. Simulated versus observed concentrations for the A horizon at the Noland Divide Spruce Site.

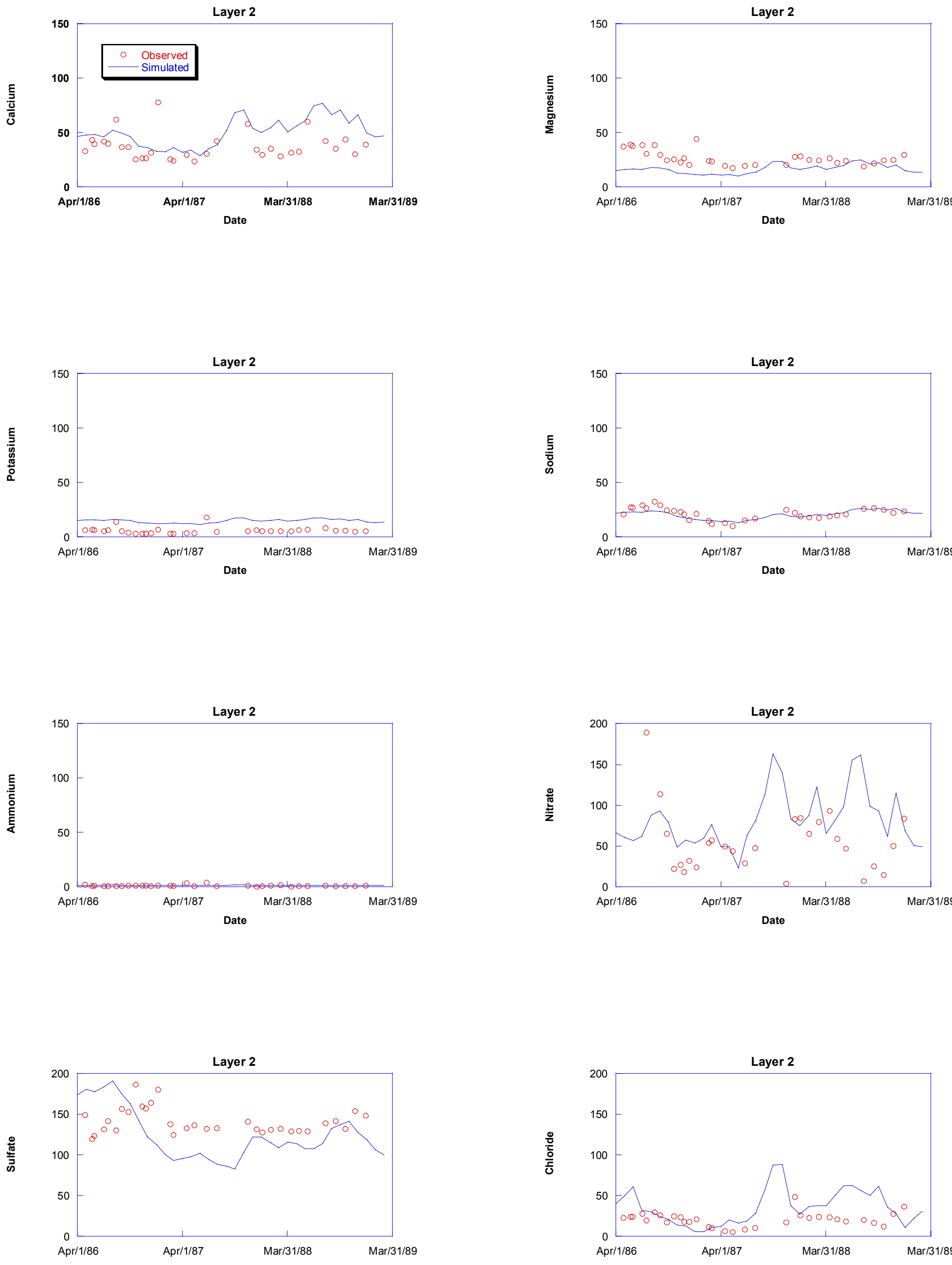


Figure 10. Simulated versus observed concentrations for the B horizon at the Noland Divide Spruce Site.

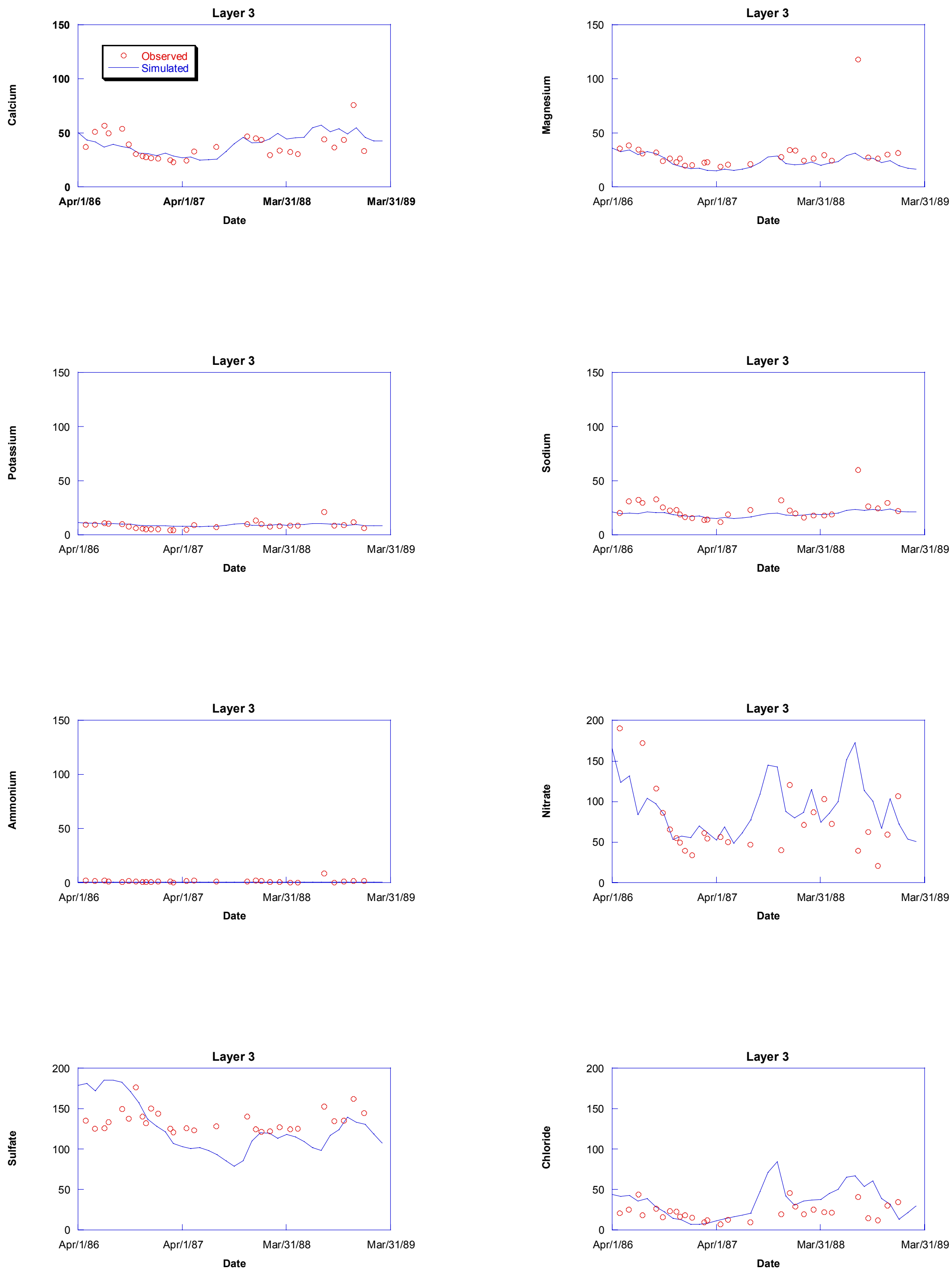


Figure 11. Simulated versus observed concentrations for the BC horizon at the Noland Divide Spruce Site.

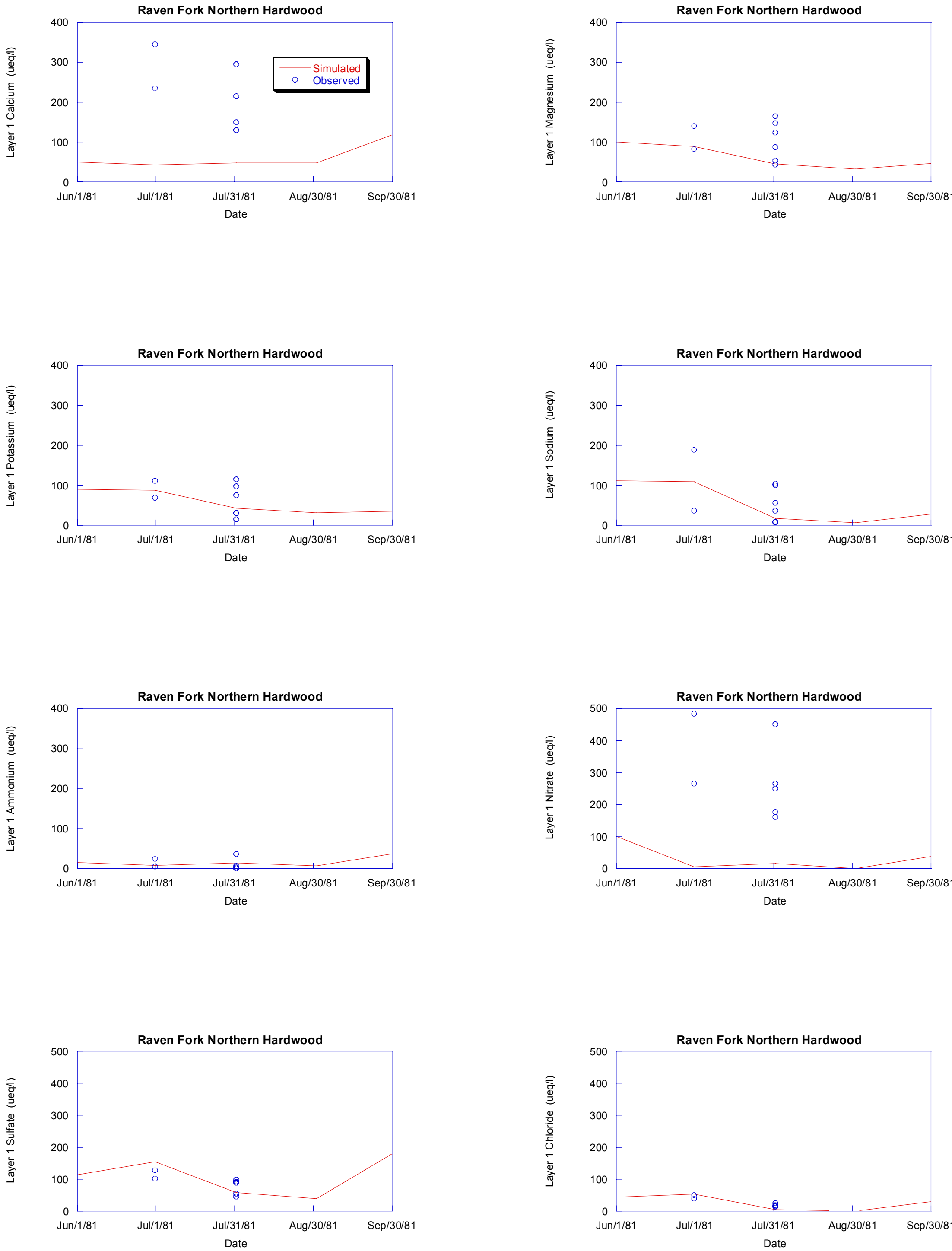


Figure 12. Simulated versus observed concentrations for the O horizon at the Raven Fork Northern Hardwood Site.

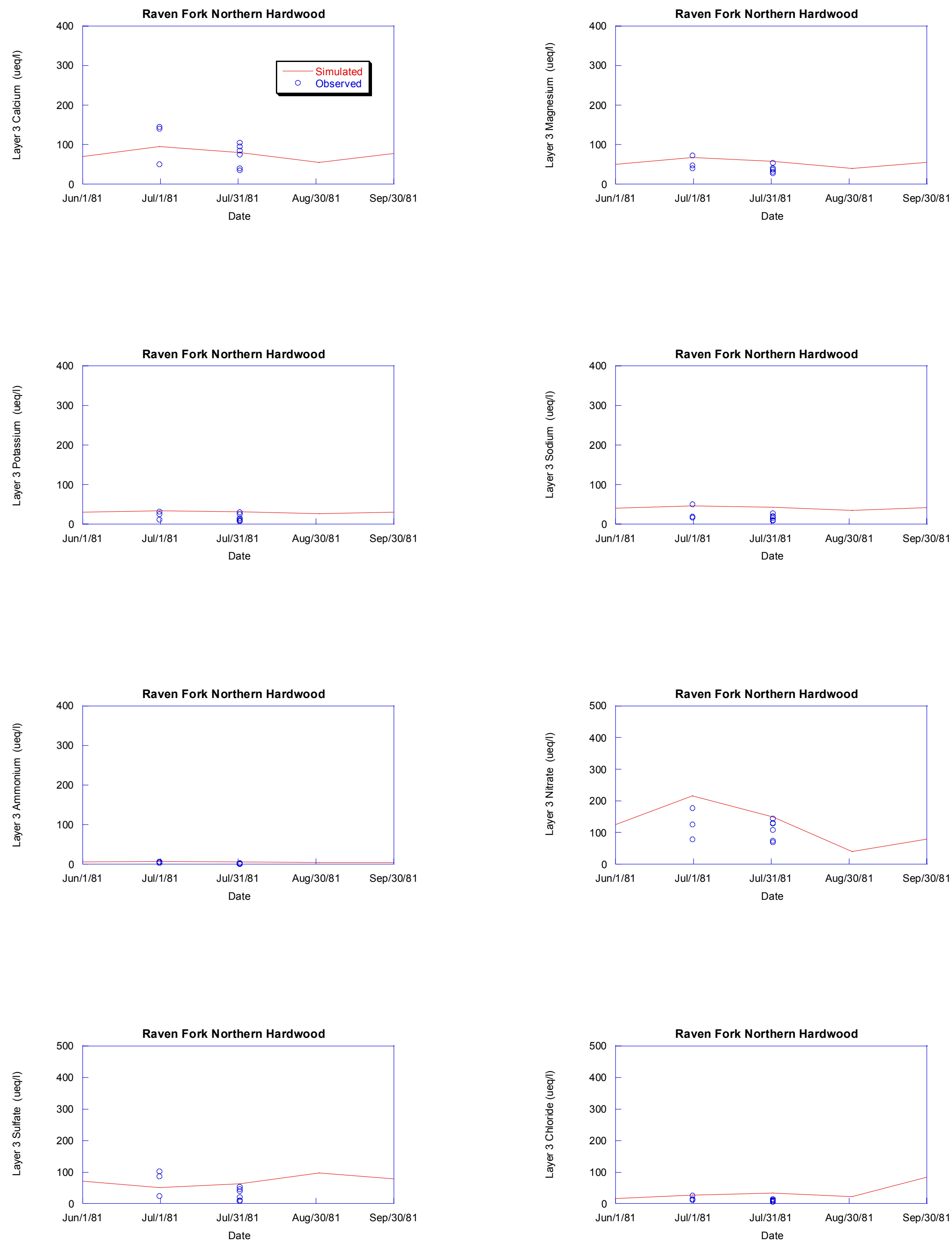


Figure 13. Simulated versus observed concentrations for the B horizon at the Raven Fork Northern Hardwood Site.

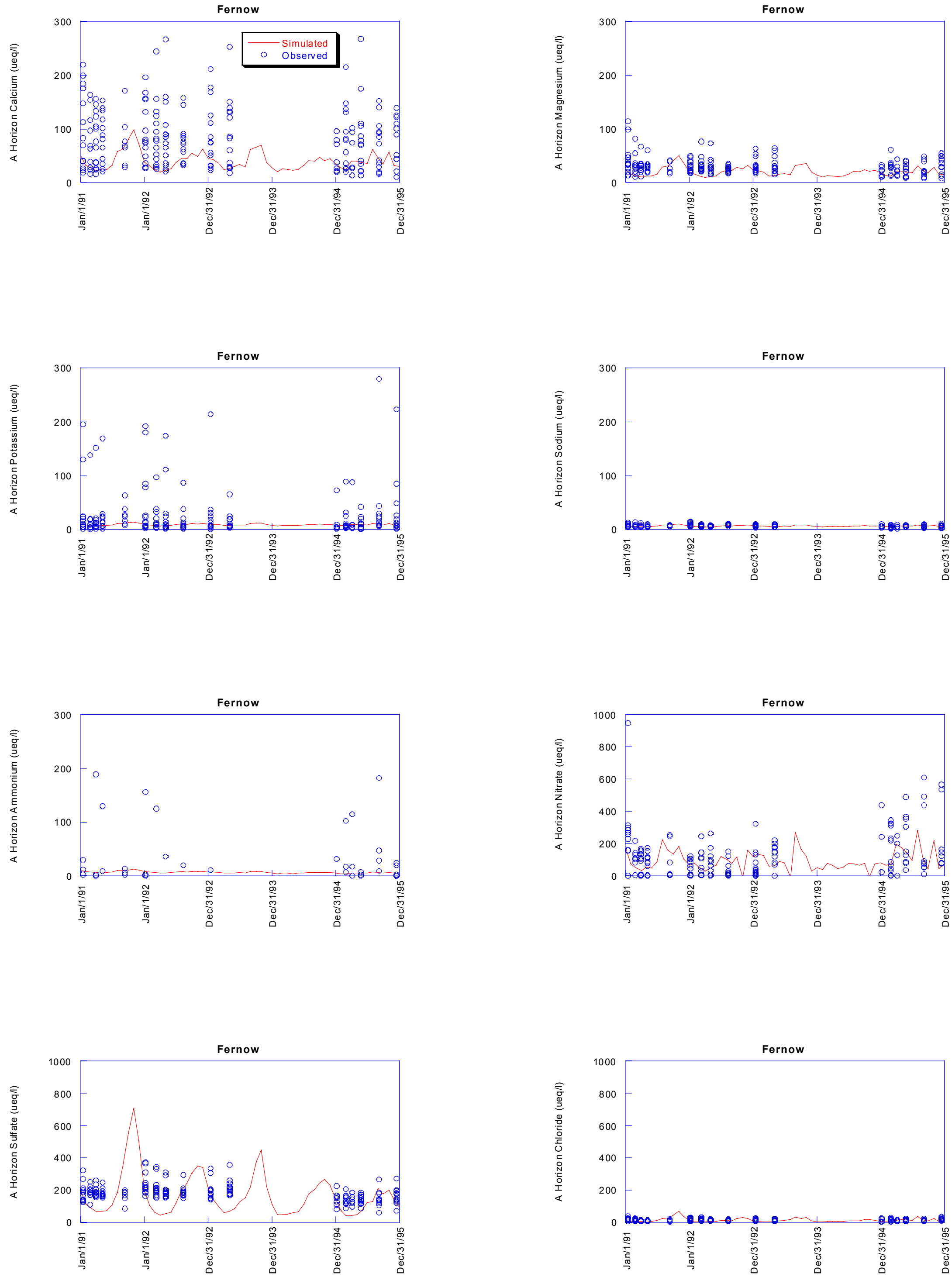


Figure 14. Simulated versus observed concentrations for the A horizon at the Fernow Northern Hardwood Site.

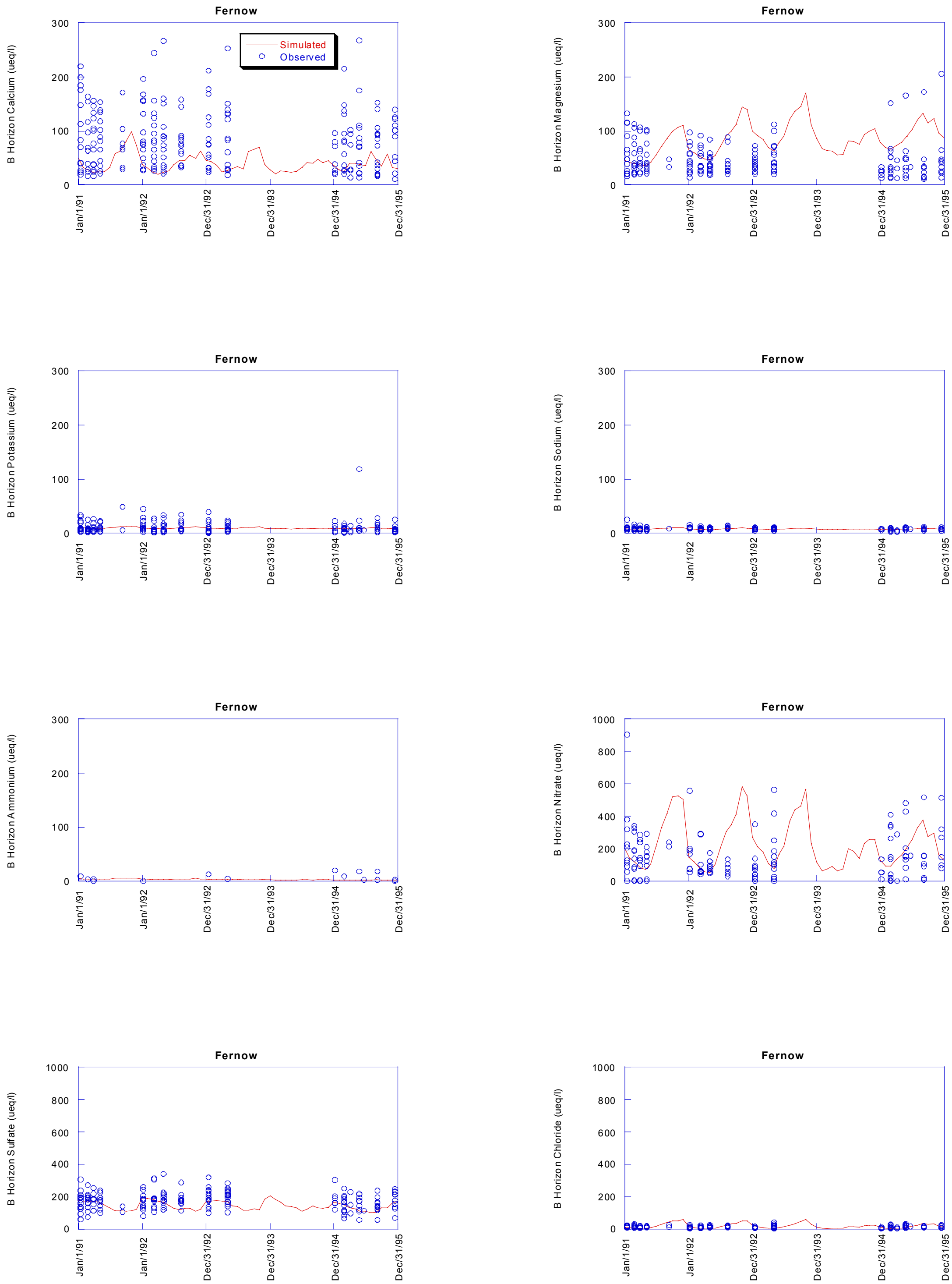


Figure 15. Simulated versus observed concentrations for the B horizon at the Fernow Northern Hardwood Site.

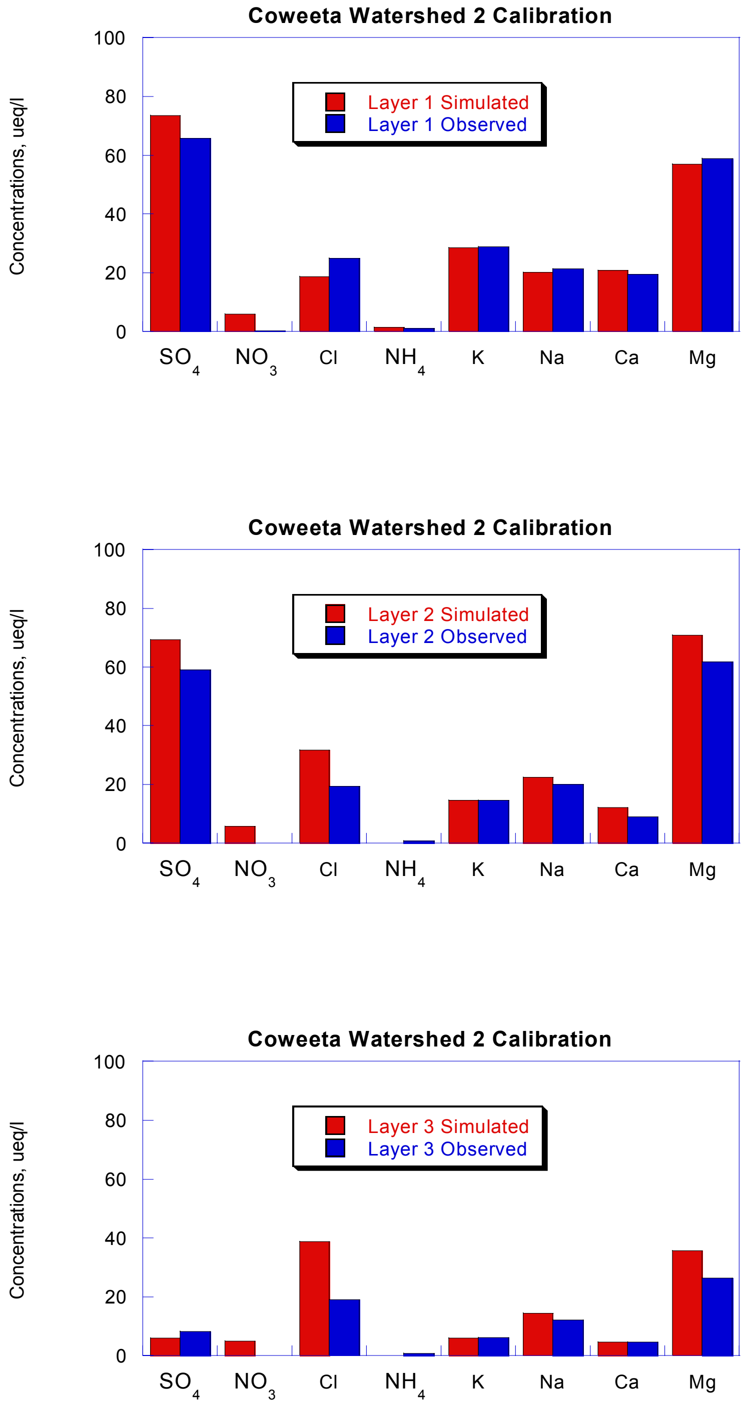
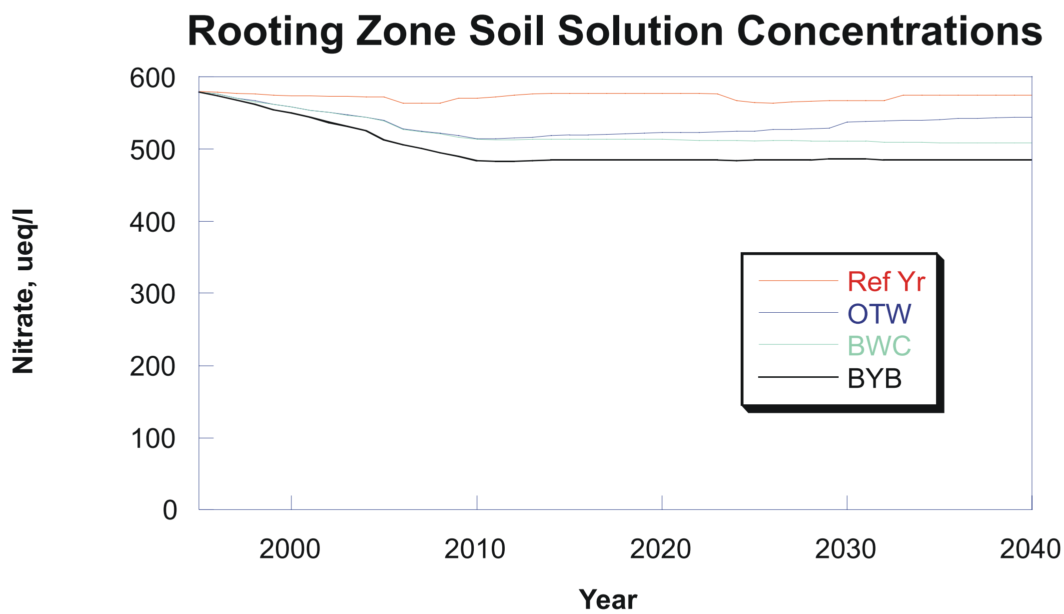


Figure 16. Simulated versus observed concentrations at the Coweeta Watershed 2 Mixed Hardwood Site.



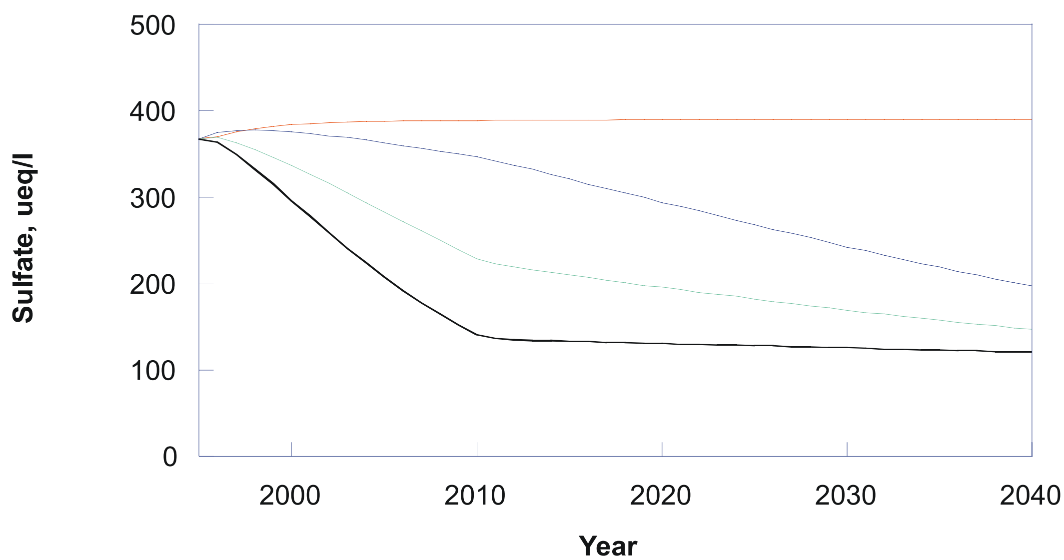
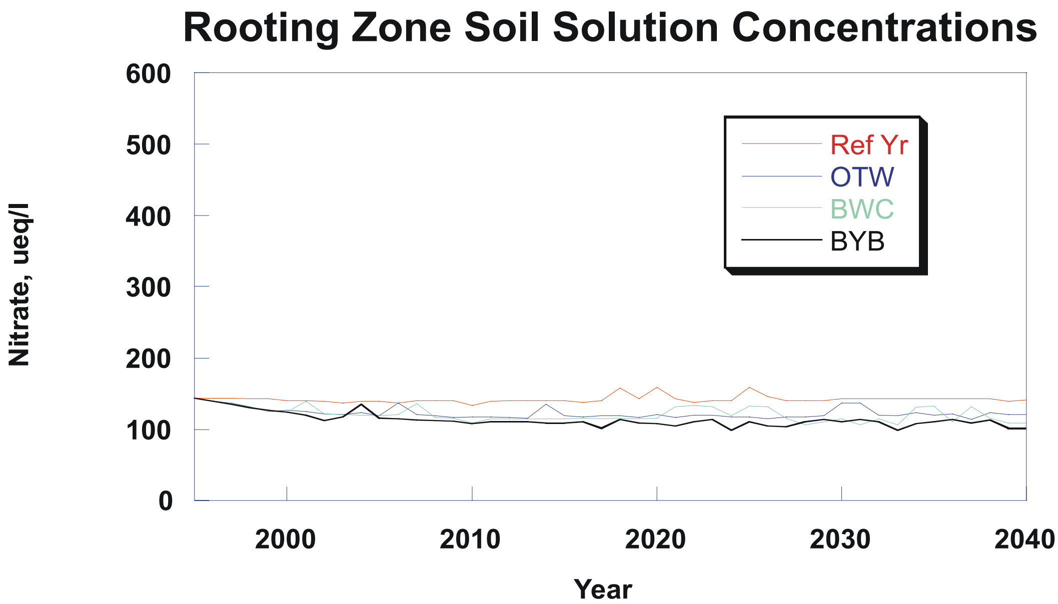


Figure 17. Raven Fork spruce site rooting zone modeling results for nitrate and sulfate concentrations in soil solution. Ref Yr refers to simulation using reference year deposition throughout the simulation period.



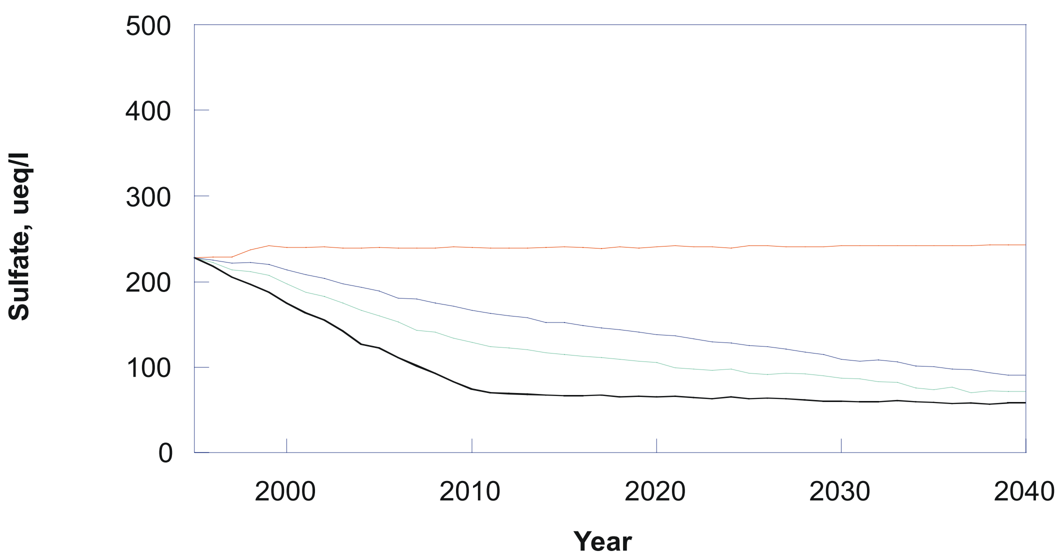
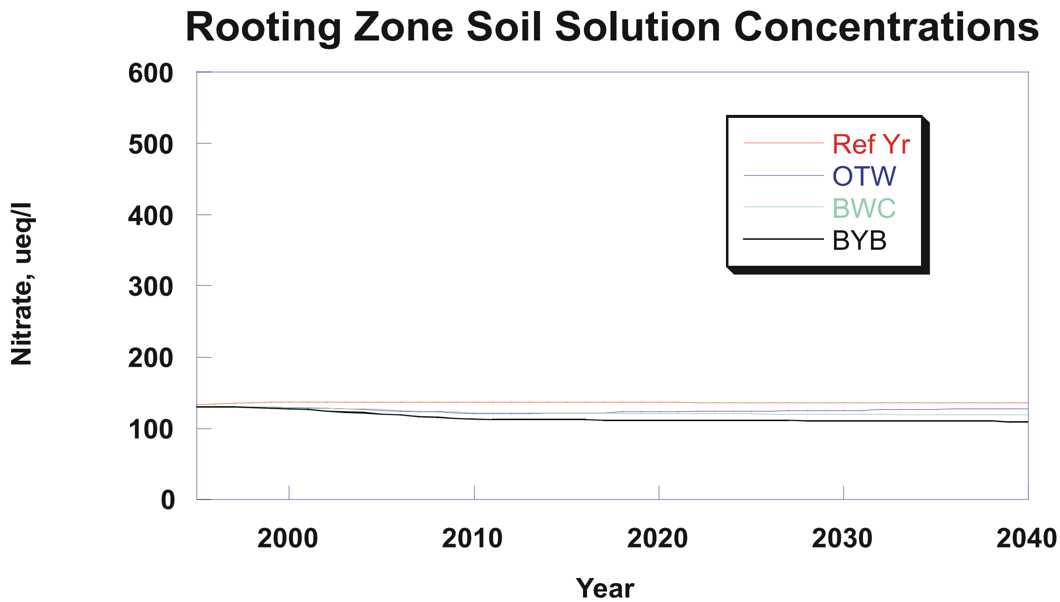


Figure 18. White Top spruce site rooting zone modeling results for nitrate and sulfate concentrations in soil solution. Ref Yr refers to simulation using reference year deposition throughout the simulation period.



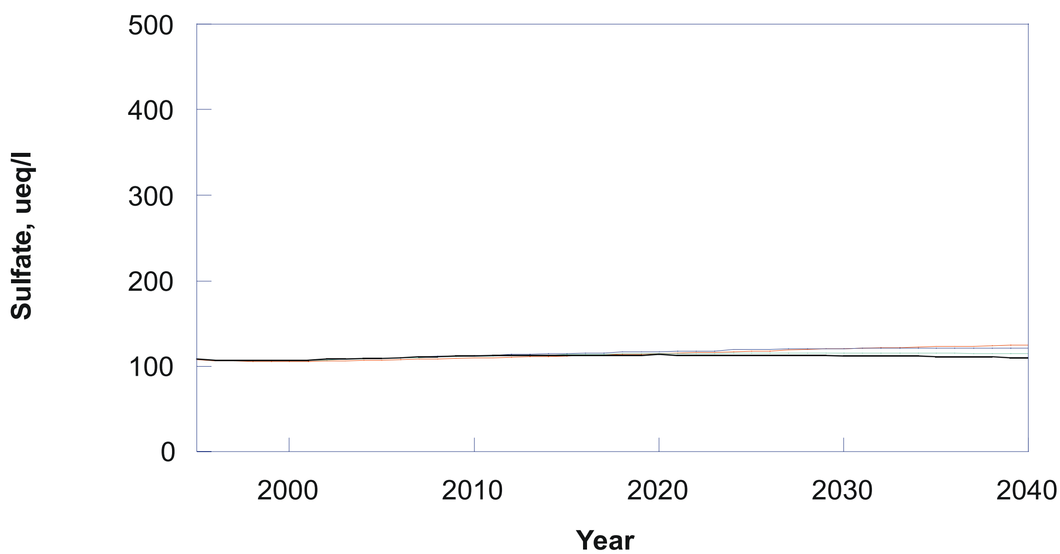


Figure 19. Noland Divide spruce site rooting zone modeling results for nitrate and sulfate concentrations in soil solution. Ref Yr refers to simulation using reference year deposition throughout the simulation period.

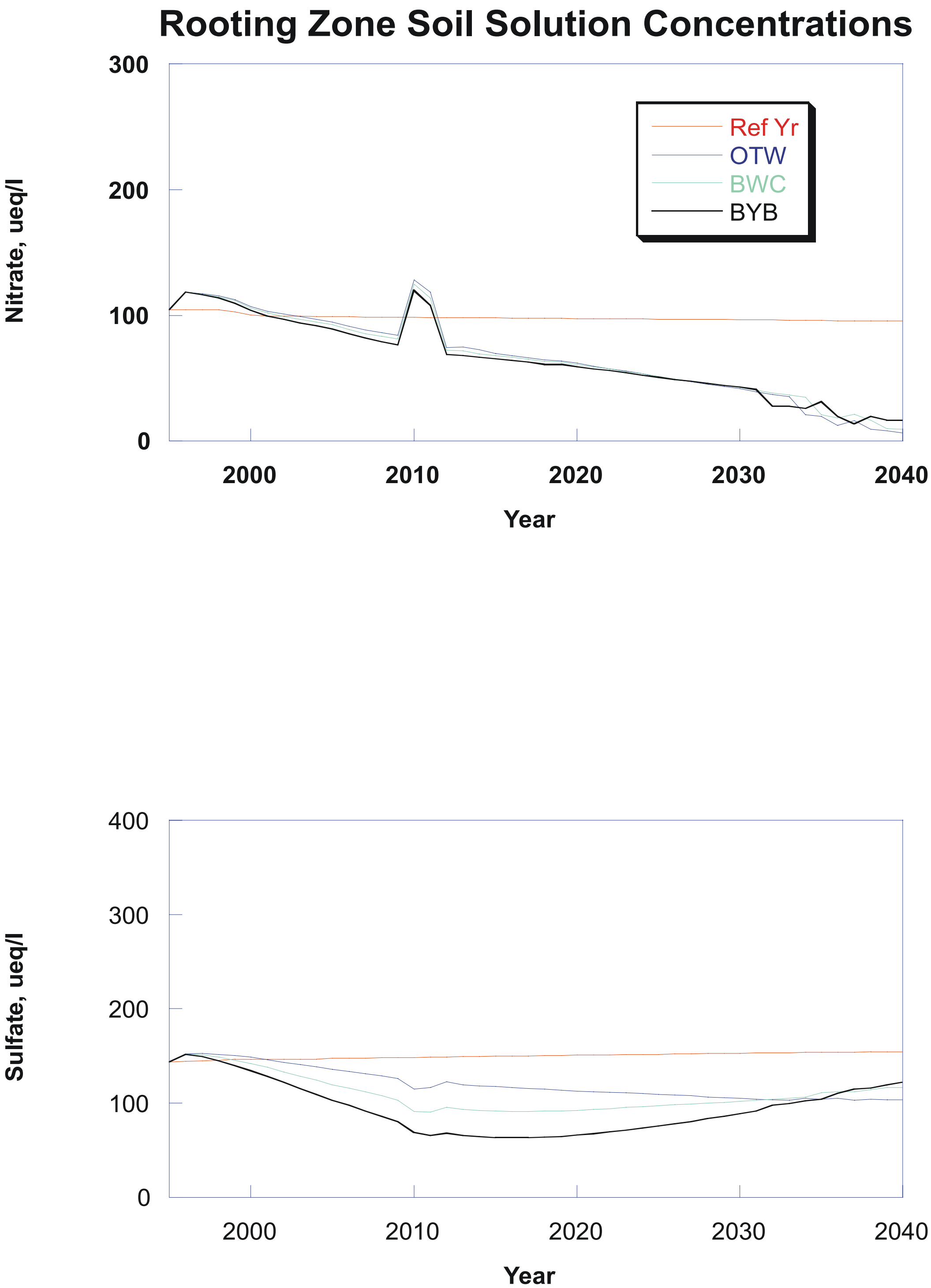


Figure 20. Raven Fork northern hardwood site rooting zone modeling results for nitrate and sulfate concentrations in soil solution. Ref Yr refers to simulation using reference year deposition throughout the simulation period.

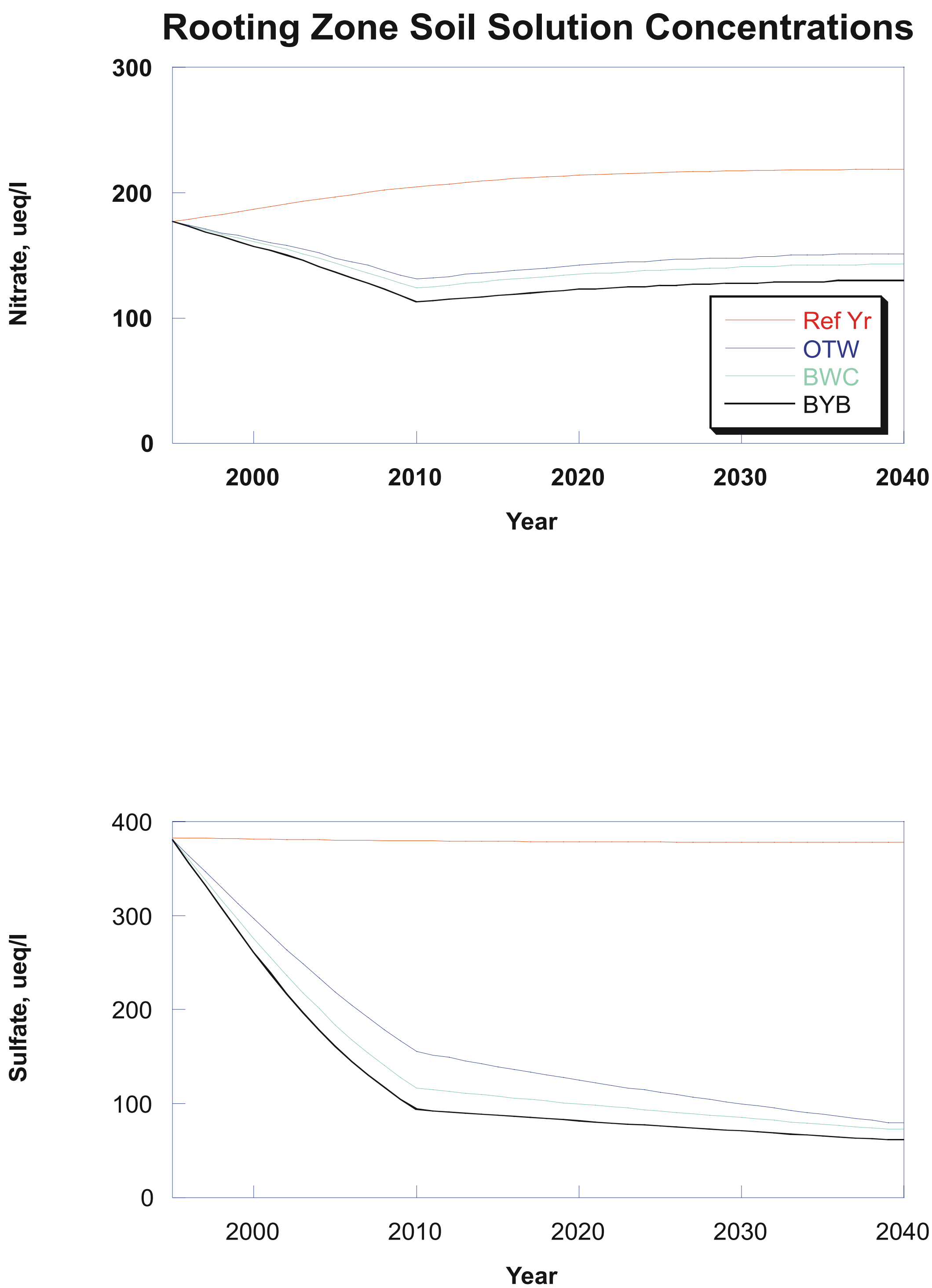
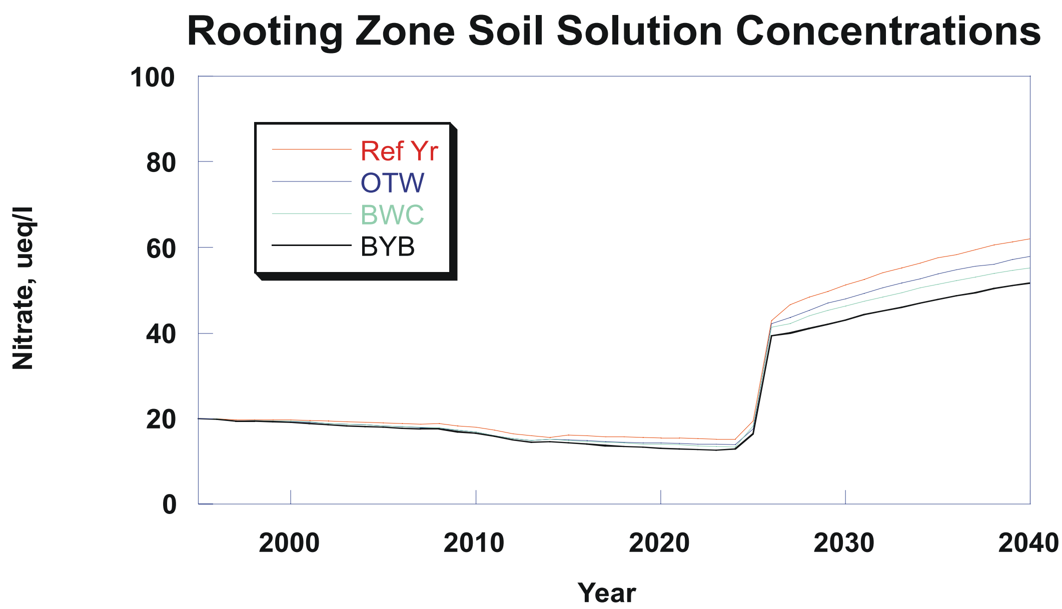


Figure 21. Fernow northern hardwood site rooting zone modeling results for nitrate and sulfate concentrations in soil solution. Ref Yr refers to simulation using reference year deposition throughout the simulation period.



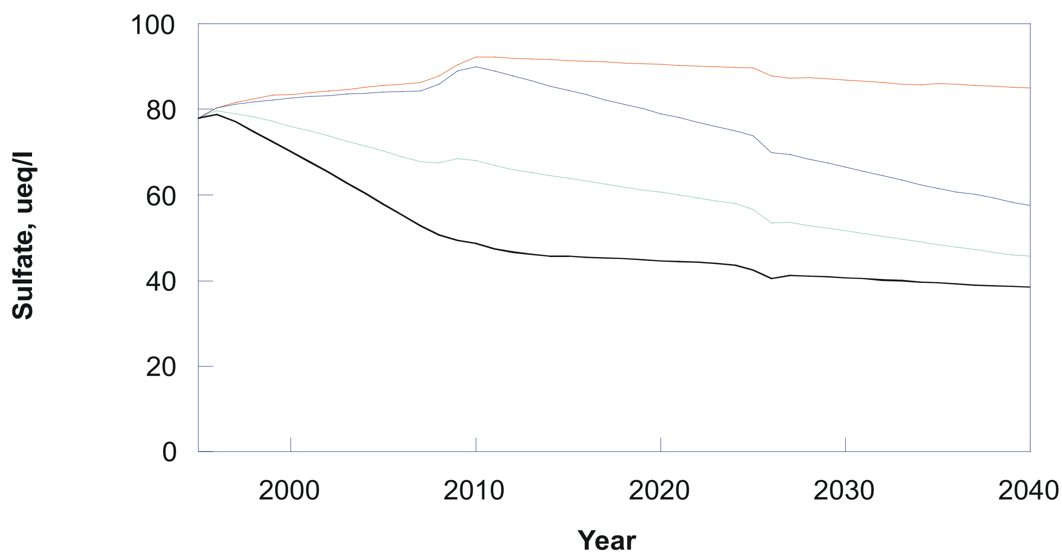
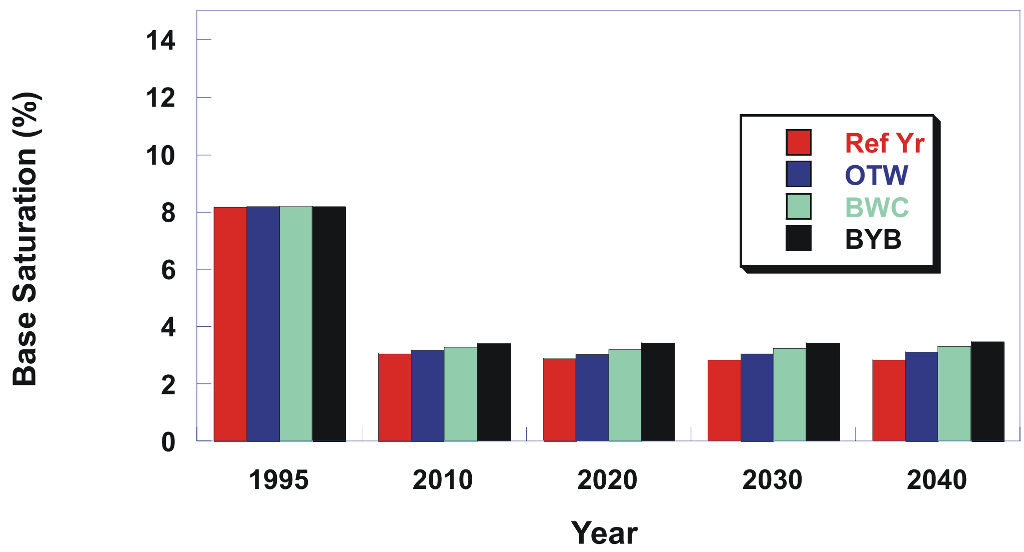


Figure 22. Coweeta WS2 mixed hardwood site rooting zone modeling results for nitrate and sulfate concentrations in soil solution. Ref Yr refers to simulation using reference year deposition throughout the simulation period.

Simulated sulfate concentrations in soil solution varied from < 100 µeq/L to near 400 µeq/L in the reference year at the modeled sites (Figures 19 through 22). The three sites that showed high soil solution sulfate concentration (> 200 µeq/L) in 1995 (Raven Fork and White Top spruce stands and Fernow northern hardwood stand) were all projected to show substantial declines in response to the strategies (Figures 17, 18, and 21). Projected differences among strategies in the year 2040 were relatively small, although the differences were more pronounced in the intervening years, especially at the Raven Fork and White Top spruce sites. In general, the largest projected declines in soil solution sulfate concentration occurred in the BYB Strategy and the smallest projected declines occurred in the OTW Strategy.

The three spruce-fir sites all showed simulated declines in base saturation over the simulation period (Figures 23 through 25). The magnitude of the declines ranged from about 2 to 5%. Base saturation values in the reference year were variable at the three spruce-fir sites, ranging from about 3% at Noland Divide to 13% at White Top. Given the variability in solid phase characteristics within watersheds, measurement of a change of less than 5% in the field would be difficult. Differences in base saturation among strategies were even smaller than differences among sites. Variations in the timing and magnitude of deposition reductions had little impact on the simulated changes in forest indicators. In terms of the calcium-to-aluminum ratio, all of the spruce-fir sites had simulated ratios less than 0.3 and showed no indication of recovery to values greater than 1 by 2040 regardless of deposition reductions. The simulated values of calcium-to-aluminum ratio were consistent with the few values published for upper mineral horizon soils at these sites (Johnson et al. 1999, Joslin and Wolfe 1992). These low values of calcium-to-aluminum ratio would seem to suggest that the forest communities should be significantly stressed under current conditions. Alternatively, an absence of stress might be partly a consequence of root migration into organic horizons where different processes influence aluminum mobility. Furthermore, the calcium-to-aluminum criterion value cannot be interpreted as an absolute threshold for biological response.

The simulated responses at the northern hardwood sites (Figures 26 and 27) were more variable than for the spruce-fir sites. Projected base saturation values were substantially higher (10% and 22%) than at the spruce-fir sites. They showed small declines at Raven Fork, but were steady or increased slightly at Fernow. Differentiation among strategies was very slight in terms of base saturation, but greater in terms of calcium-to-aluminum ratio. The Raven Fork site had a calcium-to-aluminum ratio greater than 1 in the base year, but the ratio declined below 1 given



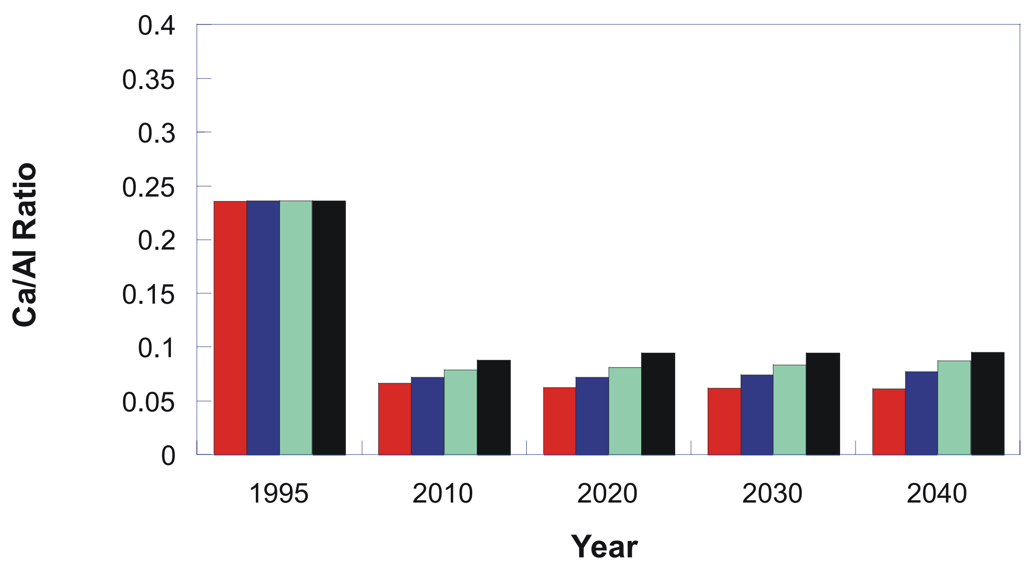


Figure 23. Simulated response to changes in atmospheric deposition in the rooting zone (A horizon) at the Raven Fork spruce site. The height of each bar on this and subsequent figures represents annual average simulated values.

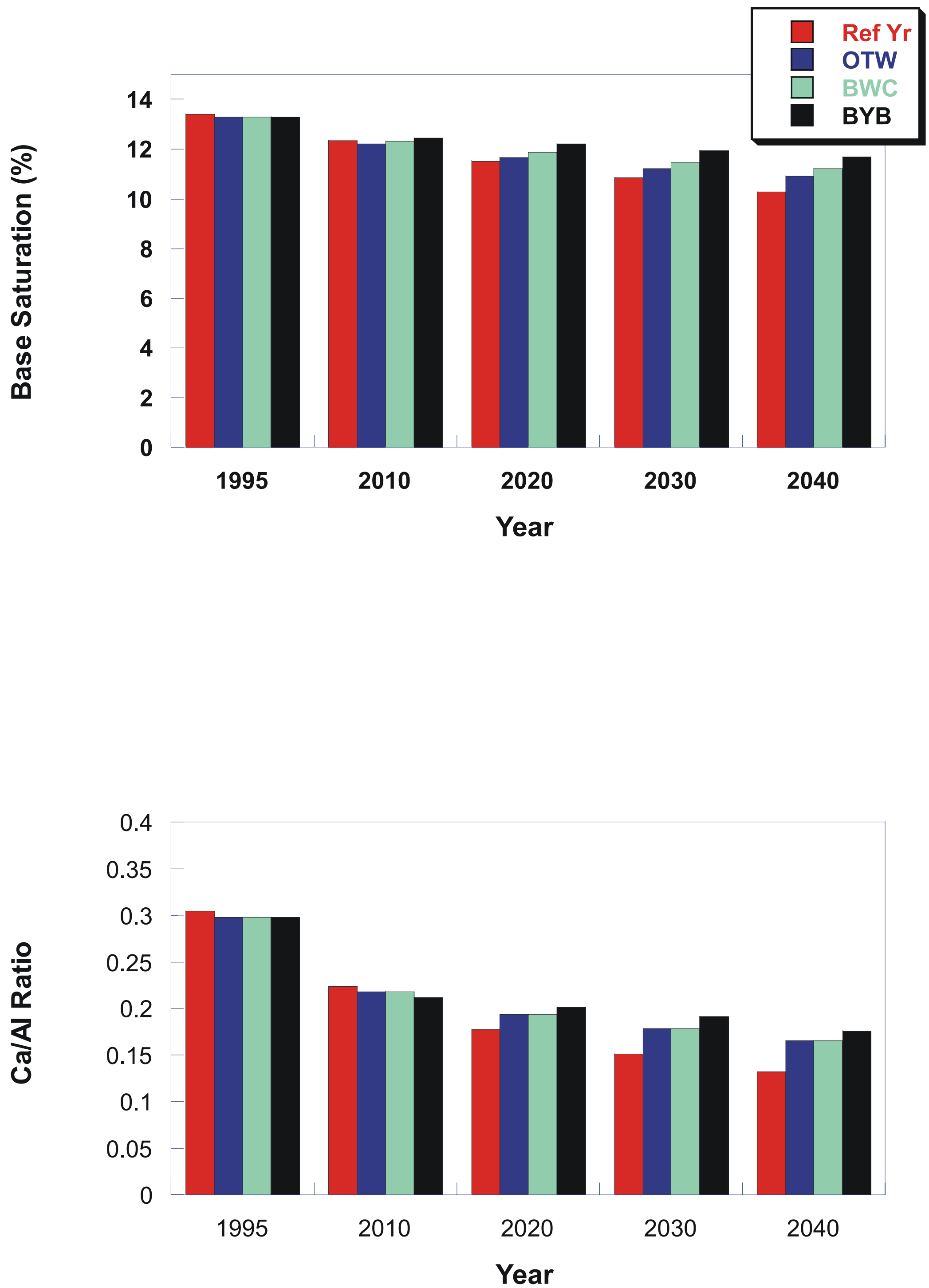
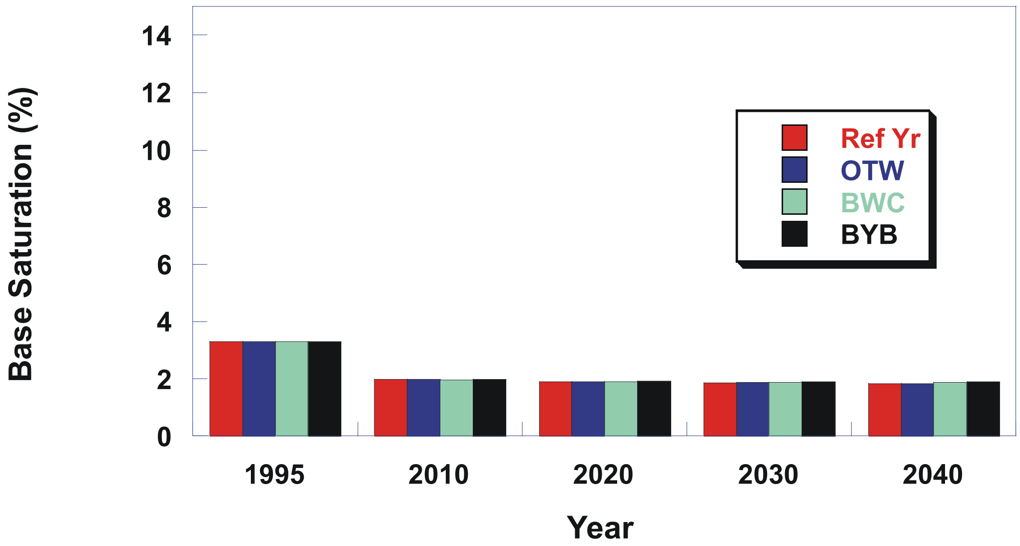


Figure 24. Simulated response to changes in atmospheric deposition in the rooting zone (A horizon) at the White Top Mountain spruce site.



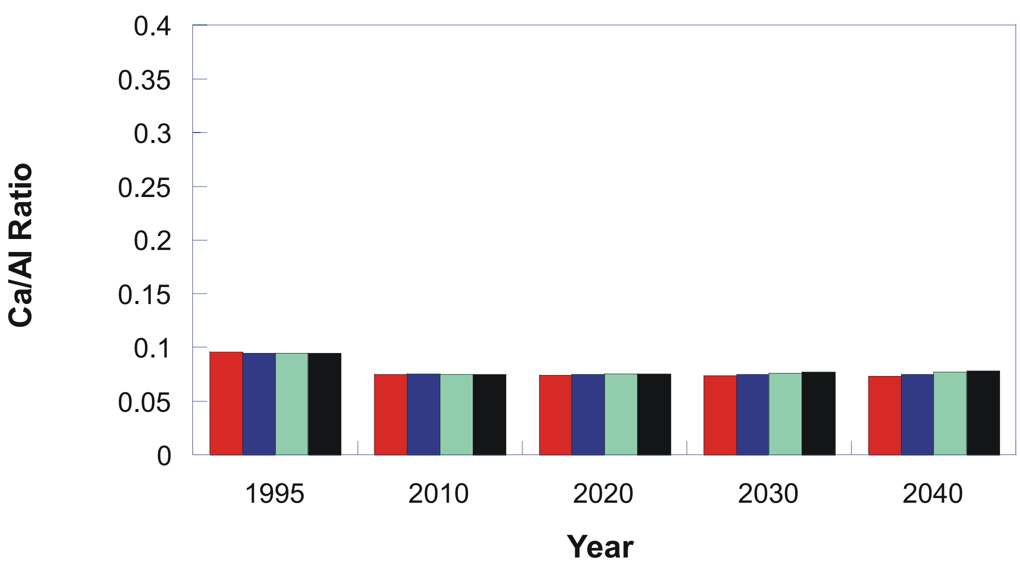
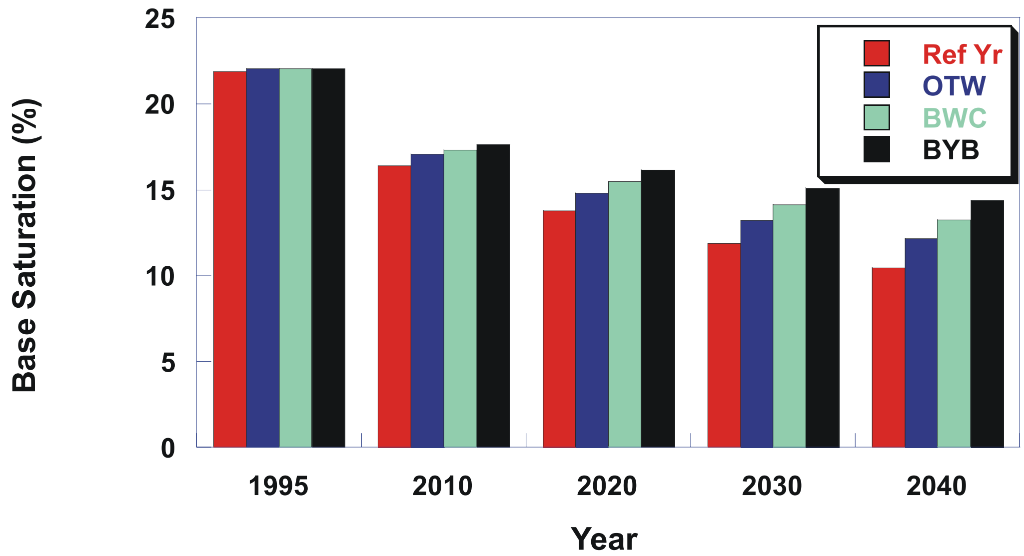


Figure 25. Simulated response to changes in atmospheric deposition in the rooting zone (A horizon) at the Noland Divide spruce site.



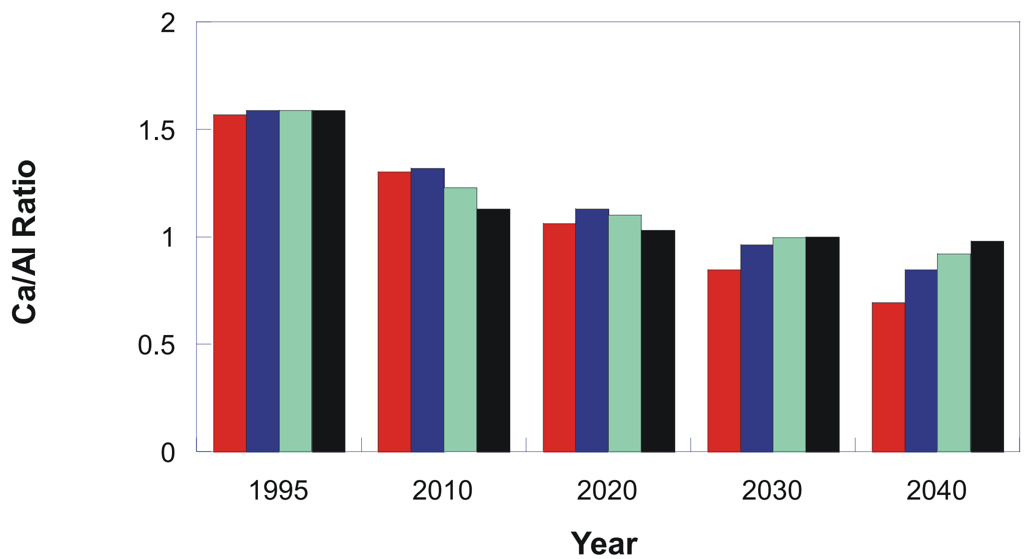
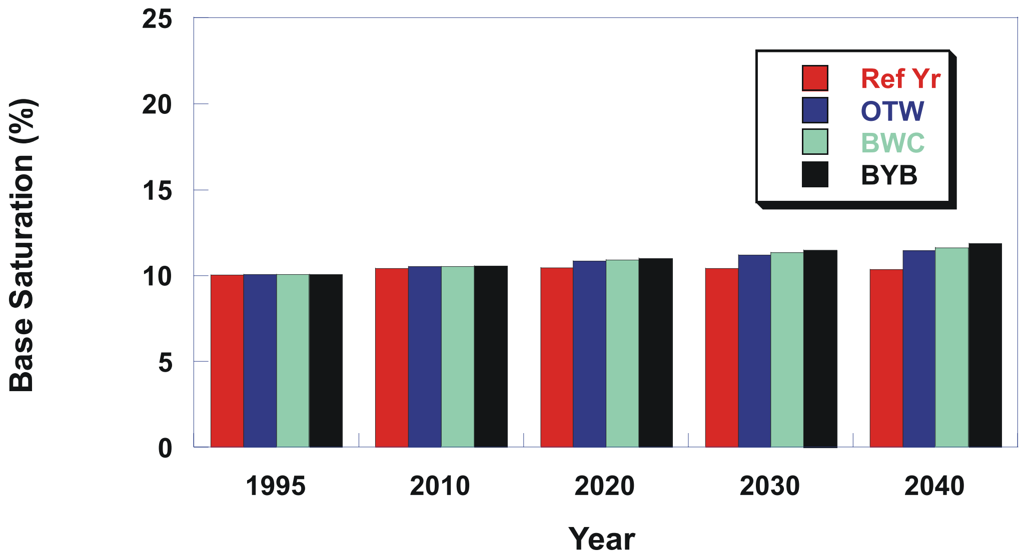


Figure 26. Simulated response to changes in atmospheric deposition in the rooting zone (A horizon) at the Raven Fork northern hardwood site.



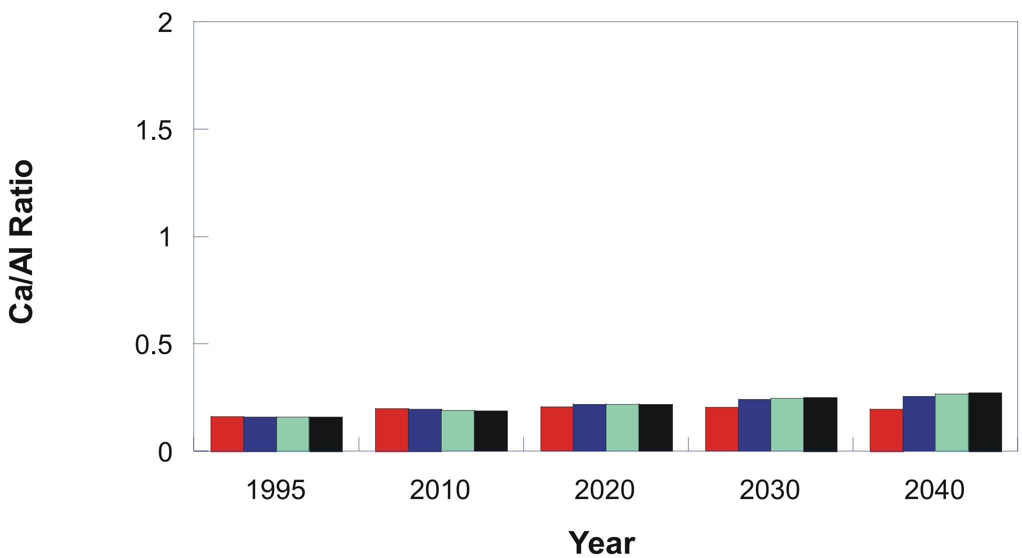


Figure 27. Simulated response to changes in atmospheric deposition in the rooting zone (A horizon) at the Fernow Watershed 4 northern hardwood site.

constant 1995 deposition. Deposition reductions served to ameliorate this effect. The Coweeta and Joyce Kilmer mixed hardwood stands showed large and rapid declines in both base saturation and calcium-to-aluminum ratio. In spite of this, the projected calcium-to-aluminum ratios were greater than 1 at the end of the simulation period at both sites for all strategies (Figures 28 and 29).

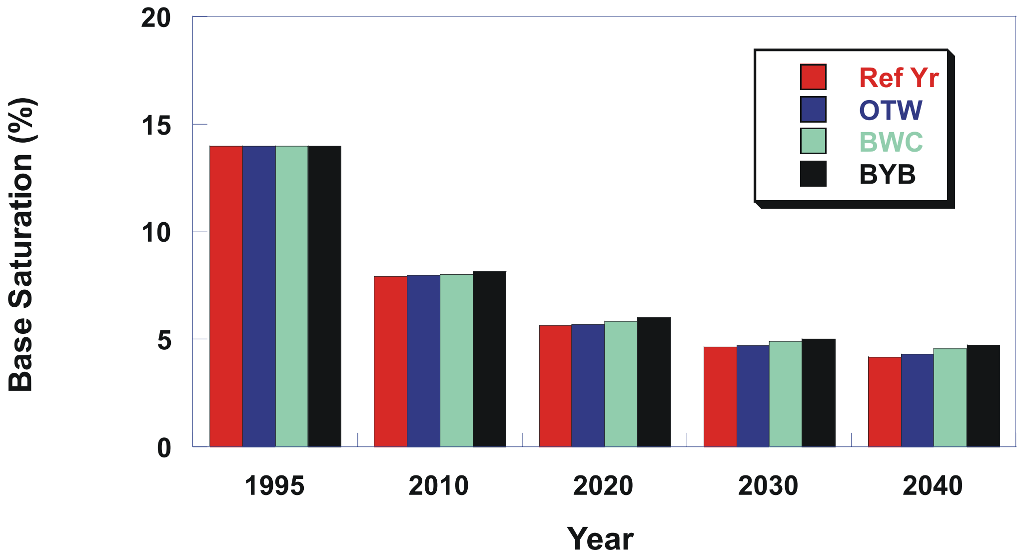
5.0 DISCUSSION

5.1 Effects of Acidic Deposition on Base Cation Supply

Calcium and other base cations are major components of soil water and surface water acid-base chemistry, and are also important nutrients that are taken up through plant roots in dissolved form. Base cations are typically found in abundance in rocks and soils, but a large fraction of the base cations stored in soils are bound in mineral structures and are unavailable to plants. The pool of dissolved base cations resides in the soil as cations that are adsorbed to negatively-charged exchange sites. They can become desorbed in exchange for hydrogen or aluminum, and are thus termed exchangeable cations. The process of weathering gradually breaks down rocks and minerals, returning their stored base cations to the soil in dissolved form and thereby contributing to the pool of adsorbed base cations. Base cation reserves are gradually leached from the soils in drainage water, but are constantly being resupplied through weathering and deposition.

It is well known that elevated leaching of base cations by acidic deposition might deplete the soil of exchangeable bases faster than they are resupplied via weathering and base cation deposition (Cowling and Dochinger 1980). However, scientific appreciation of the importance of this response has increased with the realization that surface waters are generally not exhibiting much ANC and pH recovery in response to recent decreases in sulfur deposition. In many areas, this lack of substantial recovery can be at least partially attributed to decreased base cation concentrations in surface water.

After passage of the Clean Air Act in 1970 and subsequent Amendments in 1990, emissions and deposition of sulfur were reduced and the concentrations of sulfate in lake and streamwater in many areas in the eastern United States and Canada decreased (Dillon et al. 1987, Driscoll et al. 1989, Sisterson et al. 1990). Long-term monitoring data confirmed that much of the decrease in surface water sulfate concentration was accompanied by rather small pH and ANC recoveries (Driscoll and van Dreason 1993, Kahl et al. 1993, Driscoll et al. 1995, Likens et al. 1996). The



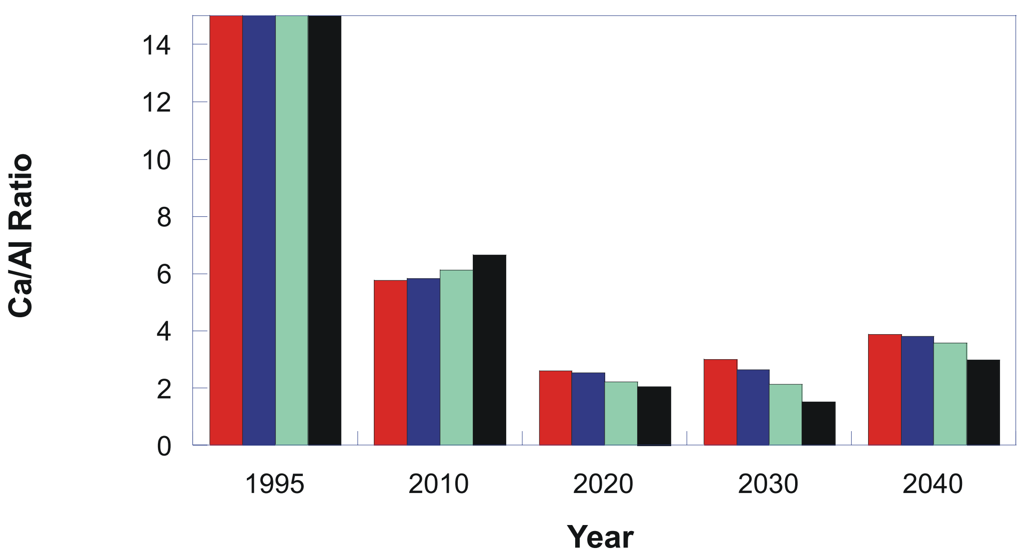
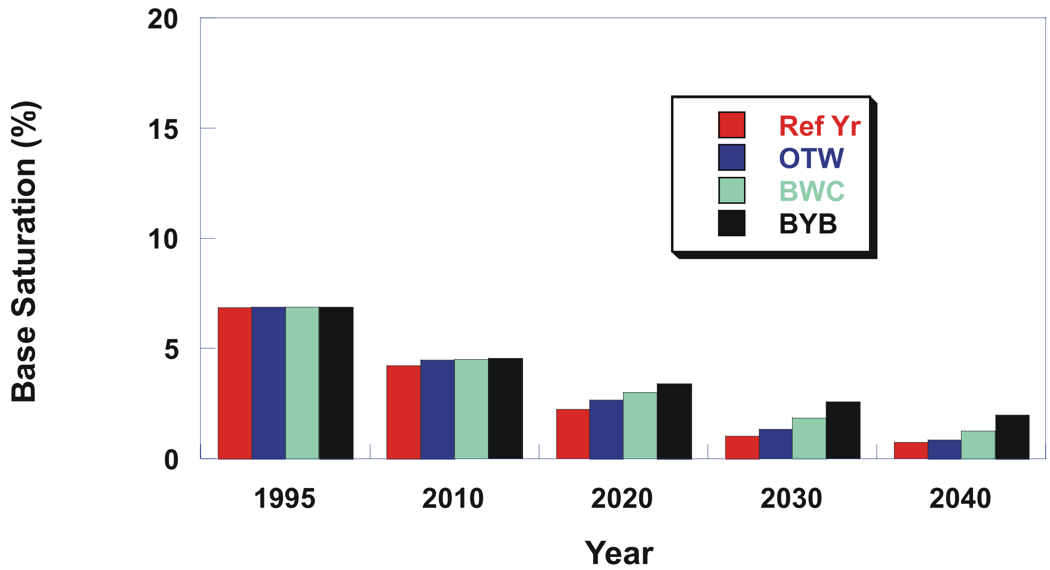


Figure 28. Simulated response to changes in atmospheric deposition in the rooting zone (A horizon) at the Coweeta Watershed 2 mixed hardwood site.



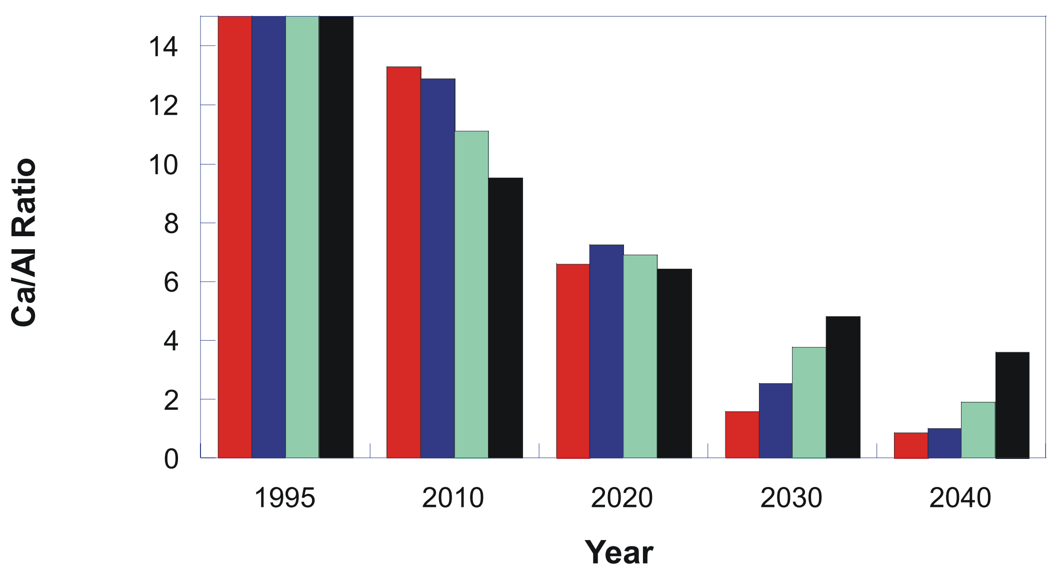


Figure 29. Simulated response to changes in atmospheric deposition in the rooting zone (A horizon) at the Joyce Kilmer mixed hardwood site.

most significant response, on a quantitative basis, was decreased concentrations of calcium and other base cations.

Thus, as sulfate concentrations in lakes and streams have declined, so too have the concentrations of calcium and other base cations. There are several apparent reasons for this. First, the atmospheric deposition of base cations in the eastern United States has decreased in recent decades (Hedin et al. 1994), likely due to a combination of air pollution controls, changing agricultural practices, and the paving of roads (the latter two affect generation of dust which is rich in base cations). Second, decreased movement of sulfate through watershed soils causes reduced leaching of base cations from soil surfaces. Third, soils in some sensitive areas have experienced prolonged base cation leaching to such an extent that soils may have been depleted of their base cation reserves. Such depletion greatly prolongs the acidification recovery time of watersheds and may adversely impact forest productivity (Kirchner and Lyderson 1995, Likens et al. 1996).

As aquatic effects researchers have revised their understanding of the quantitative importance of the various acidification processes, terrestrial effects researchers have also turned greater attention to the importance of the response of base cations to acidic deposition and the interactions between base cations (especially calcium and magnesium) and aluminum. Likens et al. (1996) concluded that acidic deposition enhanced the release of base cations from forest soils at the Hubbard Brook Experimental Forest in New Hampshire (HBEF) from the mid-1950s until the early 1970s, but that, as the labile pool of base cations in soil became depleted, the concentrations in streamwater decreased from 1970 through 1994 by about one-third. The marked decrease in base cation inputs and concomitant increase in net soil release of base cations at HBEF have likely depleted soil pools to the point where ecosystem recovery from decreased sulfur deposition will be seriously delayed. Moreover, Likens et al. (1996) suggested that recently-observed declines in forest biomass accumulation at HBEF might be attributable to calcium limitation or aluminum-toxicity, which can be expressed by the calcium-to-aluminum ratio in soil solution (Cronan and Grigal 1995).

5.2 Indices of Acidic Deposition Impacts

Cronan and Grigal (1995) proposed that a molar ratio of calcium-to-aluminum concentration equal to 1.0 in soil water be used as a general index, suggesting an increasing probability of stress to forest ecosystems. It cannot be interpreted as a single cutoff point, and variability is high. Tree species vary widely in their sensitivity to aluminum stress. In addition, aluminum concentrations in soil solution often exhibit pronounced spatial and temporal variability. Finally, the form of aluminum present in solution plays an important role in determining toxicity. For example, organically-complexed aluminum, which predominates in upper, organic-rich soil horizons, is essentially nontoxic.

The calcium-to-aluminum molar ratio of 1.0 may be interpreted as an index of warning that an ecosystem might be entering a zone of increased stress. However, many tree species may not exhibit obvious symptoms of stress until this ratio reaches levels substantially lower than 1.0. Conversely, tree roots exposed to a ratio higher than 1.0 may experience mild chronic stress without exhibiting above-ground symptoms.

A variety of factors predispose soils of high-elevation spruce-fir forests to potential aluminum toxicity and aluminum-induced inhibition of cation uptake. These factors include features of the climate (high precipitation, low temperature), vegetation (coniferous litter), bedrock (low base cation production), and soil forming processes such as podzolization (Eagar et al. 1996). Because base saturation in spruce-fir forests in the SA tends to be very low, continued input of nitrate and/or sulfate from atmospheric deposition might further acidify the soils, but is more likely to contribute to further aluminum mobilization from soils to soil solution (Johnson and Fernandez 1992). Increased mineral acid anion concentrations (nitrate, sulfate) in soil will cause the mobilization of aluminum ions from the exchange sites of acid soils. Although base cations normally dominate exchange sites in the soil compared to aluminum, base cation reserves are so low in acidic soils that aluminum exchange dominates. Dissolved aluminum concentrations in soil solution at spruce-fir study sites frequently exceed 50 µM and sometimes exceed 100 µM (Joslin and Wolfe 1992, Johnson et al. 1991b, Eagar et al. 1996). All studies reviewed by Eagar et al. (1996) showed a strong correlation between aluminum concentrations and nitrate concentrations in soil solution. They speculated that the occurrence of periodic large pulses of nitrate in solution were important in determining aluminum chemistry.

5.3 Evidence for Nitrogen Saturation in Southern Forests

Nitrogen is an essential nutrient for both aquatic and terrestrial organisms, and is a growth-limiting nutrient in most ecosystems. Thus, nitrogen inputs to natural systems are not necessarily harmful. For each ecosystem, there is an optimum nitrogen level which will maximize ecosystem productivity without causing significant changes in species distribution or abundance. Above the optimum level, harmful effects can occur in both aquatic and terrestrial ecosystems (Gunderson 1992, Aber et al. 1998).

The nitrogen cycle is extremely complex and controlled by many factors besides atmospheric emissions and deposition (Aber et al. 1991, 1998). Also, nitrogen inputs that may be beneficial to some species or ecosystems may be harmful to others. Increased atmospheric deposition of nitrogen does not necessarily cause adverse environmental impacts. In most areas, added nitrogen is taken up by terrestrial biota and the most significant effect seems to be an increase in forest productivity (Kauppi et al. 1992). However, under certain circumstances, atmospherically-deposited nitrogen can exceed the capacity of forest ecosystems to take up nitrogen. In some areas, especially at high elevation, terrestrial ecosystems have become nitrogen-saturated[1] and high levels of deposition have caused elevated levels of nitrate in drainage waters (Aber et al. 1989, 1991; Stoddard 1994). This enhanced leaching of nitrate causes depletion of calcium and other base cations from forest soils and can cause acidification of soils and drainage waters in areas of base-poor soils.

Analyses have been conducted in the northeastern United States and Europe to examine the relationships between nitrogen deposition and nitrate leaching to surface waters. The relationship between measured wet deposition of nitrogen and streamwater output of nitrate was evaluated by Driscoll et al. (1989) for sites in North America (mostly eastern areas), and augmented by Stoddard (1994). The resulting data showed a pattern of nitrogen leaching at wet-inputs greater than approximately 400 eq/ha/yr (5.6 kg N/ha/yr). Stoddard (1994) presented a geographical analysis of patterns of watershed loss of nitrogen throughout the northeastern United States. He identified approximately 100 surface water sites in the region with sufficiently intensive data to determine their nitrogen status. Sites were coded according to their presumed stage of nitrogen retention, and sites ranged from Stage 0 (background condition) through Stage 2 (chronic impacts). The geographic pattern in watershed nitrogen retention depicted by Stoddard (1994) followed the geographic pattern of nitrogen deposition. Sites in the Adirondack and Catskill Mountains in New York, where nitrogen deposition is about 11 to 13 kg N/ha/yr, were typically identified as Stage 1 (episodic impacts) or Stage 2. Sites in Maine, where

[1] The term nitrogen-saturation has been defined in a variety of ways, all reflecting a condition whereby the input of nitrogen (e.g., as nitrate, ammonium) to the ecosystem exceeds the requirements of terrestrial biota and a substantial fraction of the incoming nitrogen leaches out of the ecosystem as nitrate in groundwater and surface water.

nitrogen deposition is about half as high, were nearly all Stage 0. Sites in New Hampshire and Vermont, which receive intermediate levels of nitrogen deposition, were identified as primarily Stage 0, with some Stage 1 sites. Based on this analysis, a reasonable threshold of nitrogen deposition for transforming a northeastern site from the "natural" Stage 0 condition to Stage 1 would correspond to the deposition levels found throughout New Hampshire and Vermont, approximately 8 kg N/ha/yr. This agreed with Driscoll et al.'s (1989) interpretation, which suggested nitrogen leaching at wet inputs above about 5.6 kg N/ha/yr, which would likely correspond to total nitrogen inputs near 10 kg N/ha/yr because total deposition is often nearly double the wet deposition amount. This is likely the approximate level at which episodic aquatic effects of nitrogen deposition would become apparent in many watersheds of the eastern United States.

Analysis of data from surveys of nitrogen outputs from 65 forested plots and catchments throughout Europe were conducted by Dise and Wright (1995) and Tietema and Beier (1995). Below the throughfall inputs of about 10 kg N/ha/yr, there was very little nitrogen leaching at any of the study sites. At throughfall inputs greater than 25 kg N/ha/yr, the study catchments consistently leached high concentrations of inorganic nitrogen. At intermediate deposition values (10-25 kg N/ha/yr), Dise and Wright (1995) observed a broad range of watershed responses. Nitrogen output was most highly correlated with input nitrogen (r^2=0.69), but also significantly correlated with input sulfur, soil pH, percent slope, bedrock type, and latitude. A combination of input nitrogen (positive correlation) and soil pH (negative correlation) explained 87% of the variation in output nitrogen at 20 sites (Dise and Wright 1995).

Nitrate leaching losses from soils to drainage waters are governed by a complex suite of ecosystem processes in addition to nitrogen inputs from atmospheric deposition. In particular, mineralization and nitrification processes play important roles in regulating the quantity of, and temporal variability in, the concentration of nitrate in soil solution, and consequently leaching losses from the rooting zone (Johnson et al. 1991a,b; Joslin et al. 1987; Reuss and Johnson 1986). Thus, nitrate leaching is mostly under biological control and typically shows pronounced seasonal variability (Van Miegroet et al. 1993). Peak concentrations of nitrate in soil solution appear to be largely responsible for the potentially toxic peaks in aluminum concentration that sometimes occur in soil solution, although sulfate may also play a role by serving to elevate chronic aluminum concentrations (Eagar et al. 1996).

High leaching of nitrate in soil water and streamwater draining high-elevation spruce-fir forests has been documented in numerous studies in the SA region (c.f., Joslin and Wolfe 1992; Joslin et al. 1992; Van Miegroet et al. 1992a,b; Nodvin et al. 1995). This high nitrate leaching has been attributed to a combination of high nitrogen deposition, low nitrogen uptake by forest vegetation, and inherently high nitrogen release from soils. Forest age is another major factor affecting uptake, with mature forests requiring minimal nitrogen for new growth and, hence, often exhibiting higher nitrate leaching that younger, more vigorous stands (Goodale and Aber, 2001). Old-growth red spruce stands in the Southern Appalachians have been demonstrated to have significantly slower growth rates than stands younger than 120 years (Smith and Nicholas, 1999). The latter feature is associated with low carbon to nitrogen ratios in mineral soil, high nitrogen mineralization potential, and high nitrification (Joslin et al. 1992, Eagar et al. 1996).

In general, deciduous forest stands in the eastern U.S. have not progressed toward nitrogen-saturation as rapidly or as far as spruce-fir stands in part because they tend to be located at lower elevations and receive lower atmospheric inputs of nitrogen. Many deciduous forests have higher rates of nitrogen uptake and requirement than spruce-fir forests. Decreased growth and increased mortality have more commonly been observed in high-elevation coniferous stands than in lower elevation hardwood forests, and have been attributed by some to excess inputs of nitrogen (Aber et al. 1998). Indeed, most of the lower elevation deciduous stands, like >90% of all forests in the US, are nitrogen-deficient and are therefore likely to benefit (i.e., grow faster), at least up to a point, with increased inputs of nitrogen.

There are examples of nitrogen saturation in lower-elevation forests of the SA, especially in West Virginia. For example, progressive increases in streamwater nitrate and calcium concentrations were measured at the Fernow Experimental Forest in the 1970s and 1980s (Edwards and Helvey 1991; Peterjohn et al. 1996; Adams et al. 1997, 2000). This watershed has received higher nitrogen deposition (average throughfall input of 22 kg/ha/yr of nitrogen deposition in the 1980s) than is typical for low-elevation areas of the SA, however (Eagar et al. 1996), and this may explain the observed nitrogen saturation.

5.4 Influence of Acidic Deposition on Forest Growth and Health

Forest health is an elusive concept. It can be reflected by a variety of physiological indicators, including, for example, changes in the growth rate of trees, foliar damage, susceptibility to insects or disease, and tree mortality. Similarly, forest health can be affected by a host of potential stressors, of which air pollution is only one possibility. Climate, stand competition, outbreak of non-native pathogens, and forest management (alone or in combination) often contribute greatly to observed forest health problems. Attempts to document, and in particular to quantify, the effects of air pollution on forest health have encountered considerable complexity and uncertainty. Nevertheless, such efforts have produced some evidence that suggests that red spruce in the SA has experienced declining health as a consequence of acidic deposition. Available evidence was reviewed by Eagar et al. (1996) and is summarized here.

Radial growth of spruce trees has decreased at high elevation in the SA, but this effect has apparently been limited to elevations above 5000 ft (1520 m). The decline began about 1960, and could not be attributed to unusual climate or stand competition (McLaughlin et al. 1987, Cook and Zelaker 1992). Wood chemistry has changed in parallel with growth. An increase in aluminum relative to calcium began to occur in wood produced by high-elevation red spruce trees in Great Smoky Mountains National Park in about the 1950s (Bondietti et al. 1989).

Sapling trees growing in the same area experienced a reduction in photosynthesis relative to respiration (P:R; reflecting decreased carbon metabolism efficiency) in association with increased foliar aluminum, decreased foliar calcium, and decreased calcium-to-aluminum ratio in soil solution (McLaughlin et al. 1993). In fact, soil solution chemistry data collected at high elevation sites in the park have frequently shown aluminum concentrations sufficiently high as to interfere with calcium uptake by trees.

Decreased foliar calcium has been attributed to exposure to acidic cloudwater (Joslin et al. 1988, McLaughlin et al. 1993, Thornton et al. 1994, Eagar et al. 1996). At least two studies have shown that acidic cloudwater affects membrane-associated CA in leaf cells, leading to consistent reductions in foliar cold tolerance (Jiang and Jagels 1999, Schaberg et al. 2000). Greenhouse studies have also shown that red spruce tree seedling P:R ratios were reduced by acidic exposure (McLaughlin et al. 1993).

Joslin et al. (1992) further documented reduced growth of roots into deeper mineral soils, as compared with upper organic soils which tend to have more favorable aluminum chemistry. It is also clear that changes in calcium availability are important for tree growth at these high-elevation sites. This evidence has come from greenhouse and field fertilization studies (Van Miegroet et al. 1993, Joslin and Wolfe 1994). Soil aluminum treatments have also been repeatedly demonstrated to reduce foliar concentrations of calcium, magnesium, and other nutrients, as well as reducing growth and net photosynthesis (Raynal et al 1989, Schaberg et al. 2000). McNulty et al. (1996) demonstrated that nitrogen additions of 16 to 31 kg N/ha/yr to sites receiving low nitrogen atmospheric deposition in Vermont (5 kg N/ha/yr) resulted in (1) reduced foliar Ca:Al ratios in spruce, fir and birch trees and (2) increased rates of decline and reduced basal area growth. In contrast, Jacobson et al. (2000) found no impact 13 years later on plant nutrient concentrations of 20 years of nitrogen additions (at 24 to 120 kg N/ha/yr) to Norway spruce stands in Sweden.

Despite evidence suggesting impacts of acidic deposition on red spruce health, widespread decline of red spruce in the SA has not been documented (Cook and Zadaker 1992, LeBlanc et al. 1992). Decreased crown condition of red spruce in the 1980s was noted in several areas (Peart et al. 1992, Bruck et al. 1989), but long-term mortality rates have appeared stable (Nicholas 1992, Eagar et al. 1996, Smith and Nicholas 1999). Pauley et al. (1996) also found no unusual mortality or health symptoms in red spruce in the Great Smoky Mountains. Further, in places where red spruce mortality has occurred in the Northeast, red spruce appears to be maintaining its composition fraction in the recovering forest, though balsam fir and birches are growing faster than red spruce (Battles and Fahey 2000). It is also possible that recent observed decreases in the growth rate of high-elevation southern red spruce may be part of a long-term, climate-induced pattern of fluctuation, and may not be unusual compared with data from the past 200 years (LeBlanc et al. 1992, Reams et al. 1993, Eagar et al. 1996). In addition to acidic deposition, other important factors contributing to red spruce growth and health might include drought and increased exposure to wind and ice storms subsequent to the death of neighboring fir trees from balsam wooly adelgid infestation (Zedaker et al. 1988, Nicholas et al. 1992, Eagar et al. 1996). Nevertheless, there is considerable evidence that spruce growth and health have declined at high elevation in the SA, and that acidic deposition has played a role in such effects.

The evidence for adverse effects of acidic deposition on Fraser fir in the SA is less compelling. Eagar et al. (1996) concluded that Fraser fir populations were deteriorating throughout the region. There is evidence from tree ring studies that Fraser fir in Great Smoky Mountains National Park began a growth decline in about 1960 (McLaughlin et al. 1984). Inadequate calcium supply could play a role in the resistance of Fraser fir to adelgid infestation, but no scientific evidence has been presented to support this relationship (c.f., Manion 1981). While Fraser fir seedlings are rapidly re-occupying many areas of adult fir mortality, there is considerable evidence that the shortage of reproducing Fraser fir adults, due to adelgid-caused mortality, will eventually lead to progressive decreases in replacement (Smith and Nicholas, 2000). In contrast to red spruce, studies on Fraser fir Christmas trees have demonstrated the ability of the species to take up and effectively utilize nitrogen at high rates of application (up to 170 kg N/ha/yr; Hinesley et al., 2000).

5.5 Role of Disturbance

Spruce-fir forests throughout the SA have been subjected to significant disturbance, especially from the balsam wooly adelgid (*Adelges piceae*), a European pest which has infested Fraser fir since about the 1960s. Severe fir mortality has occurred in many areas. This disturbance factor has the potential to interact with acidic deposition and other ecosystem stresses, and contribute to multiple-stress tree mortality and to changes in biogeochemical cycling.

5.6 Sensitivity to Acidification of Different Forest Types in the SAMI Region

5.6.1 High-Elevation Spruce Fir

A number of studies suggest that acidic deposition has impacted high-elevation spruce-fir forests in the SA. These ecosystems are geographically limited within the region, occurring primarily at elevations above about 1400 m in southwestern Virginia, eastern Tennessee, and western North Carolina. Much of the research conducted to date has focused on Great Smoky Mountains National Park (which contains 74% of the spruce-fir forests in the region), White Top Mountain, VA, and Mt. Mitchell, NC (Eagar et al. 1996). This ecosystem experiences high precipitation (near 200 cm per year), high humidity, and frequent cloud cover. Atmospheric deposition of sulfur and nitrogen can be very high and can include substantial cloud deposition (Mohner 1992).

Soil processes and nutrient cycling have been intensively studied at a number of sites in the SA, including three spruce-fir sites. The Integrated Forest Study (IFS) included five sites in or near Great Smoky Mountains National Park, two of which were red spruce (Tower and Becking sites). The SFRC and Tennessee Valley Authority sponsored research on a red spruce stand on White Top Mountain, and the Spruce-Fir Research Cooperative also sponsored research on spruce forest soils in the Black Mountains, NC. High-elevation areas in the SA are often dominated by sandstone and other unreactive bedrock. Base cation production via weathering is limited (Elwood et al. 1991). Soils of spruce-fir forests in the SA region tend to have thick organic horizons, high organic matter content in the mineral soils, and low pH (Joslin et al. 1992). Because of the largely unreactive bedrock, base-poor litter and organic acid anions produced by the conifers, high precipitation, and high leaching rates in these high-elevation forests, soil base saturation tends to be below 10% and the soil cation exchange complex is generally dominated by aluminum (Johnson and Fernandez 1992, Joslin et al. 1992).

Spruce-fir forests in the SA, especially those at high elevation, receive high atmospheric deposition of nitrogen and show a number of signs of approaching nitrogen saturation, including:

1. high concentrations of nitrate in soil solution and streamwater throughout the year,
2. nitrate leaching losses that sometimes approach atmospheric inputs (Nodvin et al. 1995, Van Miegroet et al. 2001),
3. nitrogen mineralization in excess of nitrogen-uptake requirements of plants,
4. lack of tree growth response to nitrogen fertilization (Johnson et al. 1991b, Joslin and Wolfe 1992, Eagar et al. 1996), and
5 Sensitivity of red spruce to moderate levels of soil solution aluminum and moderately low Ca/Al ratios (Raynal et al. 1989, Cronan and Grigal, 1995).

The consistency of the results for the modeled sites suggests that many high-elevation spruce-fir sites within the SAMI domain may respond to changes in acidic deposition inputs in a similar fashion. Areas that have experienced a decline in tree growth or increased mortality (for example from balsam wooly adelgid infestation) are expected to be particularly susceptible to accelerated nitrate leaching and associated adverse impacts of nitrogen saturation on forest health. However, the dense reoccupation of some of these sites with rapidly-growing Fraser fir regeneration, which is capable of taking up considerable nitrogen and incorporating it in new growth, may minimize this adverse impact (Hinesley et al. 2000, Smith and Nicholas 2000).

5.6.2 Hardwood Forests

Conditions described for spruce-fir forests are not generally applicable to lower elevation hardwood forest sites throughout the SAMI domain. IFS research at Coweeta Hydrological Laboratory, NC and at Oak Ridge, TN illustrated some key differences in soil and vegetation that are associated with elevational differences (Swank and Crossley 1988, Johnson and Lindberg 1992). In general, lower elevation forest soils in the SA differ in a number of important ways from high-elevation soils. Because of their greater age and stability on the landscape, low-elevation soils are mostly Ultisols and Inceptisols associated with Ultisols. They generally have higher base saturation and smaller nitrogen pools in the soil due to more intensive land use, greater age, warmer climate, and greater disturbance histories (Eagar et al. 1996). Therefore, lower-elevation forest ecosystems generally have much larger nitrogen retention capacity, and substantial nitrate leaching to drainage water is usually only found in association with severe disturbance (c.f., Swank and Crossley 1988, Webb et al. 1995, Eshleman et al. 2001).

Lower-elevation soils are more susceptible to soil acidification than high-elevation soils. This is because they tend to release base cations, rather than aluminum, in response to the movement of mineral acid anions through the soil column. However, in general they are less likely, at present, to show cation deficiency or aluminum stress because they still generally retain sufficient base cations on the soil exchange complex. Nevertheless, prolonged leaching of nitrate and/or sulfate through low-elevation soils can potentially contribute to accelerated cation loss, eventually leading to lower soil base saturation, and consequent increase in aluminum concentrations in soil solution and aluminum stress to tree roots.

There is no evidence to suggest widespread forest decline in response to nitrogen inputs in deciduous forests in the southeastern United States (Raynal et al. 1992). However, the complex relationships between atmospheric inputs and forest health are nevertheless of concern, especially in view of the demonstrated effects of acidic deposition in some situations on soil acidity, nutrient supply, and metal toxicity. Any forest stand, even a deciduous forest with high nitrogen uptake rates, can become nitrogen saturated with high enough nitrogen deposition for a sufficient period of time (Peterjohn et al. 1996, Adams et al. 2000). Gilliam et al. (1996) has demonstrated that nitrogen additions to a hardwood forest in West Virginia can result in reductions in plant tissue calcium across a variety of species. In Pennsylvania oak stands, oak mortality was significantly correlated with low soil base saturation, low Ca/Al ratios, low levels of foliar calcium, and foliar deficiencies of potassium. In France, moderate liming of oaks growing in strongly acidic soil produced improved root and shoot growth, as well as the uptake of nutrients by roots (Bakker et al., 1999).

There are few hardwood forests in which actual soil changes have been measured in association with acidic deposition (Raynal et al. 1992). Acidification and/or nutrient loss can be inferred, however, from studies of elemental budgets (c.f., Binkley and Richter 1987). There is a general tendency for some hardwood trees to accumulate calcium, especially oak and hickory species, and this can cause calcium depletion and soil acidification (Johnson and Todd 1990). Such effects can be exacerbated by both tree harvesting and acidic deposition, both of which contribute to removal of base cations from soils (Raynal et al. 1992). Calcium limitation has not been shown to be significant in eastern hardwood forests to date, but several studies have suggested impending calcium depletion with intensive harvesting (Johnson et al. 1988, Federer et al. 1989, Johnson and Todd 1990). The issue therefore merits continued monitoring. It is not clear whether low-elevation forest soils in the SAMI region will ultimately develop calcium deficiency. Soils may stop changing their base cation status before deficiencies develop as sulfur and nitrogen deposition levels decline, or weathering rates may be sufficient to maintain adequate cation nutrient supplies.

5.7 Use of the NuCM Model to Evaluate Forest and Soils Conditions

Johnson et al. (2000) recently reviewed the performance of the NuCM model in various applications, including several in the SA. Some of the salient features of NuCM as related to the objectives of SAMI are reviewed here. The reader is referred to Johnson et al. (2000) for additional details.

NuCM has been applied to a variety of sites and manipulations since its inception, with varying degrees of success. Table 2 lists the sites calibrated prior to the SAMI exercise and Table 3 lists the manipulations which have been simulated using NuCM. The interested reader is referred to Johnson et al. (2000) for a review of the "successes" and "failures" of various NuCM applications as of 1999; since that time, the model has been applied to the issue of climate change in several forests (Johnson et al., 2001) and to a site where a nitrogen-fixing species (red alder) caused high rates of nitrate leaching and soil acidification in forests of western Washington (Verburg et al., in press). By successes, we mean that the model output mimicked the general patterns observed in the field, sometimes even before field observations were made. Among the successes for NuCM, we include: (1) harvesting and species change in mixed deciduous and loblolly pine forests, (2) response to liming in a deciduous forest, (3) prediction of high nitrate in ponderosa pine forests, (4) response to artificial acid rain and nitrogen fertilization in a soil column experiment, and (5) nutrient cycling in a Norway spruce stand. It is noteworthy that, from a scientific point of view, much more was gained by analyzing the failures than was gained from the successes in these applications.

Table 2. Sites calibrated for NuCM.

Site	Species	References
Clingman's Dome, NC	*Picea rubens* *Fagus grandifolia*	Liu et al. 1992 Johnson et al. 1996, 1999
Coweeta, NC	Mixed deciduous, *Pinus strobus*	Johnson et al. 1993
Walker Branch, TN	Mixed deciduous	Johnson et al. 1998
Duke, NC	*Pinus taeda*	Johnson et al. 1995a
Bradford, FL	*Pinus elliotii*	Johnson et al. 2001
Barton Flats, CA	*Pinus ponderosa*	Fern et al. 1996
Little Valley, NV	*Pinus jeffreyii*	Johnson et al. 2000
Nordmoen, Norway	*Picea abies*	Kvindesland 1997
Ås, Norway (Soil columns)	*Pinus sylvestris*	Sogn and Abrahamsen 1997 Sogn et al. 1997
Thompson, WA	*Alnus rubra* *Pseudotsuga menziesii*	Verburg et al., in press

Table 3. Manipulations simulated by NuCM.

Manipulation	Site	References
Atmospheric deposition (N, S, base cations)	Barton, Coweeta, Clingman's, Duke, Nordmoen, Ås	Johnson et al. 1993, 1995a, 1996 Fenn et al. 1996 Sogn and Abrahamsen 1997 Sogn et al. 1997 Kvindesland 1997
Harvesting	Duke, Coweeta	Johnson et al. 1995a
Liming	Coweeta	Johnson et al. 1995b
Species change	Duke, Coweeta	Johnson et al. 1995a, 1997
Precip. amount	Walker Branch	Johnson et al. 1998
Elevated CO_2 (via N cycle)	Duke, Walker Branch	Johnson 1999
Climate change	Walker Branch, Coweeta, Clingman's, Bradford, Duke, Little Valley	Johnson et al. 2000
Nitrogen fixation	Thompson	Verburg et al. in press

In the following paragraphs, some of the results of simulations for forests in the SAMI region are reviewed. NuCM was used to simulate responses of red spruce at Noland Divide to reduced sulfur and nitrogen deposition (c.f., Johnson et al. 1996). Johnson et al. (1996) hypothesized that reducing nitrogen and sulfur deposition would cause 1) large reductions in soil solution nitrate, sulfate, aluminum and increased calcium-to-aluminum ratios, but 2) small changes in exchangeable base cation reserves. Hypothesis 1 was supported in part; simulated reductions in atmospheric deposition had substantial and nearly immediate effects upon simulated soil solution mineral acid anions, calcium, and aluminum concentrations.

Calcium-to-aluminum molar ratios were much less sensitive to changes in deposition and soil solution ionic strength than either calcium or aluminum separately. Hypothesis 2 was not supported; although the ameliorations of losses in base saturation and exchangeable cation pools were small relative to cation exchange, they were large relative to initial exchangeable base cation pools. These changes in base cation pools and base saturation were of sufficient magnitude to affect simulated soil solution composition over a very short time.

With some exceptions, NuCM simulated the temporal variations in soil solution aluminum, calcium, and calcium-to-aluminum ratios over the three-year monitoring period reasonably well (Figures 30 and 31). Although the exact temporal patterns in the field were not matched, NuCM showed the same basic patterns as observed in the field. In both the field data and the simulations, B horizon calcium concentrations were very low and relatively unresponsive to variations in soil solution ionic strength and aluminum concentrations. NuCM overestimated peak aluminum concentrations in the B horizons, but the baseline values were quite close at ≈ 200 μM. Simulated calcium-to-aluminum ratios were generally within the range of variability of field measurements (Figure 31).

An important aspect of these simulations was that while soils remained extremely acidic in all scenarios, the relative changes in exchangeable calcium and base saturation (compared to initial values) were large and had substantial projected effects upon soil solution calcium and aluminum concentrations. In the A horizon, exchangeable calcium and base saturation declined by 20 to 70% less in these scenarios when compared to the baseline scenario. In the Bw1 and Bw2 horizons, exchangeable calcium and base saturation declined by 20 to 150% less compared to baseline. These large relative changes in base saturation caused substantial projected effects on soil solution calcium-to-aluminum ratios even though the soils remained very acidic. This occurred because these soils are in the range where calcium-to-aluminum ratios are extremely

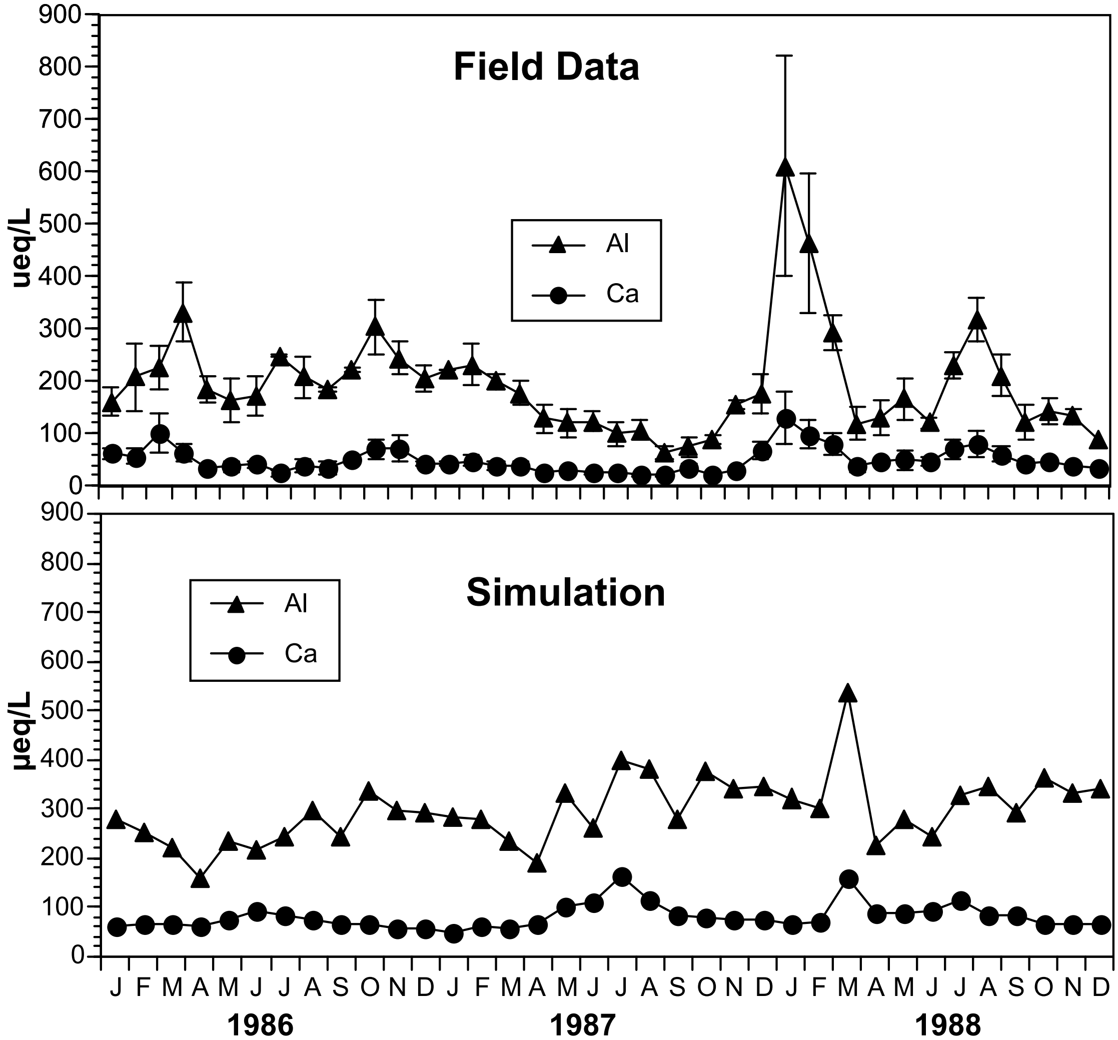


Figure 30. Actual and simulated soil solution aluminum and calcium in A horizons of the Tower Site, Noland Divide. (from Johnson et al. 1996)

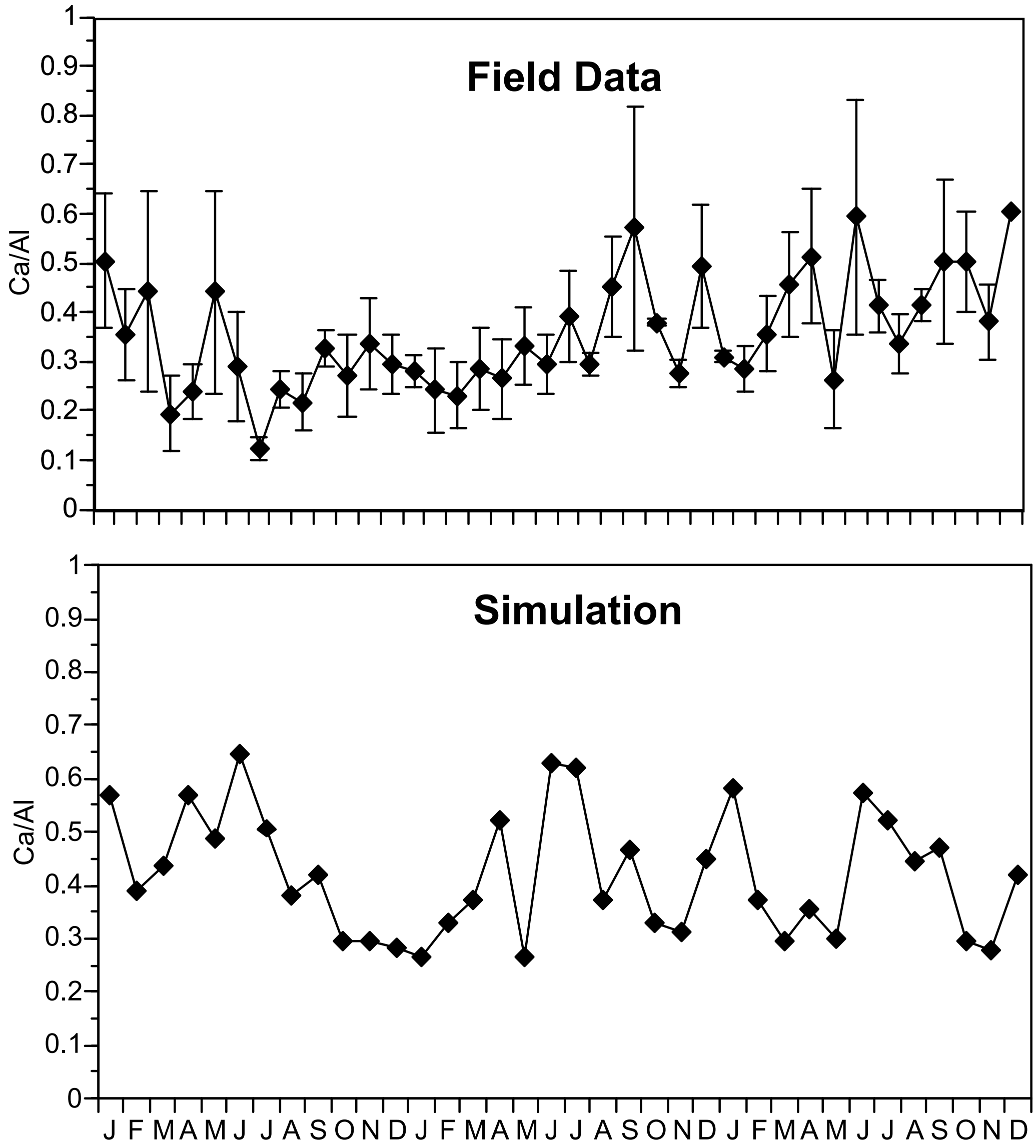


Figure 31. Actual and simulated soil solution calcium-to-aluminum molar ratios in A horizons of the Tower Site, Noland Divide. (from Johnson et al. 1996)

sensitive to changes in both base saturation and soil solution mineral acid anion levels (Reuss and Johnson 1986). This extreme sensitivity is a simple consequence of the cation exchange equations used in the models, which predict that soil solution aluminum will vary by the 3/2 power of soil solution calcium concentration. The interested reader should consult Reuss and Johnson (1986) for details.

These results led to a follow-up analysis of the effects of reduced sulfur, nitrogen, and base cation deposition, focusing on interactions between changes in solution concentrations and base saturation. Reduced base cation deposition has been noted in many areas throughout the eastern U.S. and Europe (Hedin et al. 1994), including the SAMI region. Whereas the reduced sulfur and nitrogen deposition might be expected to cause reduced soil leaching of base cations and facilitate recovery of base saturation in soils, reduced base cation deposition may offset these benefits. The objective of this analysis was to explore the potential effects of combined reductions in the deposition of nitrogen, sulfur, and base cations on both the extremely acidic Noland Divide Site and a relatively non-acidic system at Coweeta. These simulations suggested that, for the extremely acidic Noland Divide system, sulfur and nitrogen deposition were the major factors affecting soil solution mineral acid anion and thus aluminum concentrations. Base cation deposition was the major factor affecting soil exchangeable cations, which in turn most strongly affected soil solution base cation concentrations. Both had substantial effects on the calcium-to-aluminum ratio, but in opposite directions. This caused the net effect of the combined reduction in nitrogen, sulfur, and base cation deposition to be closer to baseline than either of the two scenarios separately. In no case did the molar calcium-to-aluminum ratio rise above 1.0 (Figure 32). However, these simulations suggest that SAMI should consider the potential future effects of changes in base cation deposition as a complicating factor in determining the mitigative effects of emissions reduction strategies on extremely acidic sites such as Noland Divide.

One interesting result was that the scenario that specified 50% reduction in nitrogen and sulfur deposition caused increased base saturation in the surface horizons, as expected, but decreased base saturation in the deepest (Bw3) horizon. This counter-intuitive result was apparently a function of reduced leaching of base cations from upper to lower soil horizons with reduced deposition and is discussed further below. This reversal in base saturation in the deepest horizons was a key factor in the responses observed in that horizon. It is not known whether such patterns occur under field conditions, but they could have substantial and counterintuitive implications for the acidity of drainage water in response to SAMI emissions control strategies.

In the moderately acidic, sulfur-accumulating mixed deciduous forest at Coweeta, reduced base cation deposition by 50% caused a very slight (<4%) reduction in base cation leaching as a result of slightly reduced base saturation and increased soil sulfate adsorption. The effects of reducing sulfur and nitrogen deposition by 50% on base cation leaching (16% over the

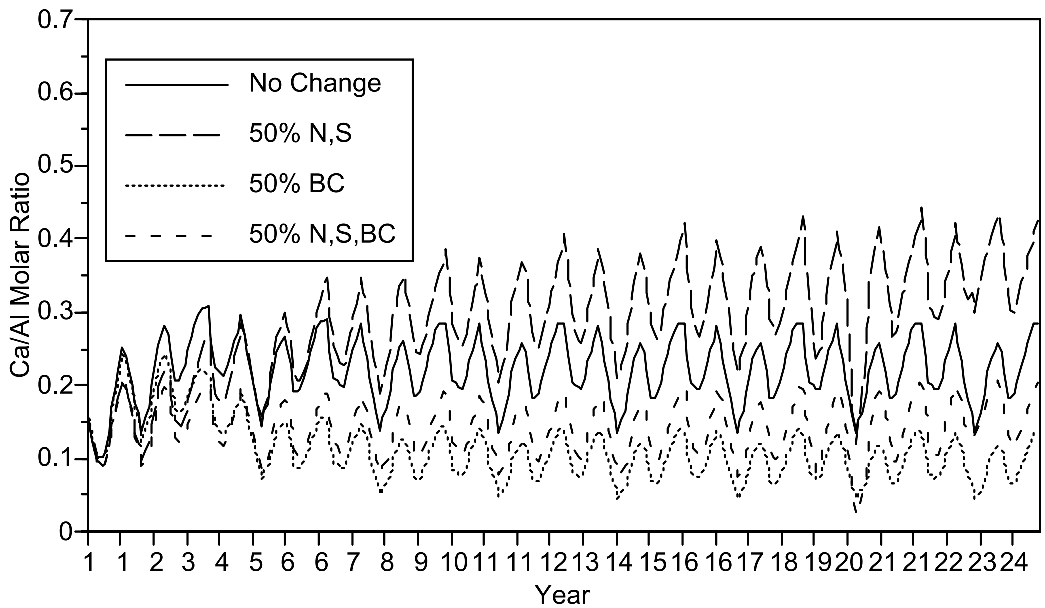


Figure 32. Simulated molar calcium-to-aluminum ratios in A horizon solutions from Noland Divide under no change, 50% reduction in nitrogen and sulfur deposition (50% N, S), 50% reduction in base cation deposition (50% BC), and 50% reduction in nitrogen, sulfur, and base cation deposition (50% N,S,BC). (After Johnson et al. 1999).

simulation period) were greater than those of reduced base cation deposition. The Coweeta system continued to accumulate both sulfur and nitrogen with reduced deposition levels, even though growth and vegetative uptake were slightly reduced (-5%) by decreased nitrogen availability.

As was the case at Noland Divide, the reduced sulfur and nitrogen deposition scenarios caused less reduction in base saturation in the A and AB soil horizons when compared to the baseline condition of "No Change", but reduced base saturation in the BC horizon (Figure 33). This pattern has been seen often in NuCM simulations (Johnson et al., 2000). In the early days of assessing acid deposition effects on soils, many predicted that soils were very inert and slow to change because simple budget calculations indicated that pool sizes were large compared to fluxes into and out of the soil profile (McFee et al. 1980, Reuss and Johnson 1986). However, several field studies have shown declines in soil exchangeable bases over periods of decades, even in some sites with relatively low atmospheric inputs (Bergkvist and Folkeson 1995; Falkengren-Grerup and Eriksson 1990; Falkengren-Grerup and Tyler 1992; Johnson et al. 1988, 1991a; Binkley et al. 1989; Knoepp and Swank 1994). Vegetation uptake of base cations has proven to be a major factor in causing soil change in many cases (Johnson et al. 1988, Johnson

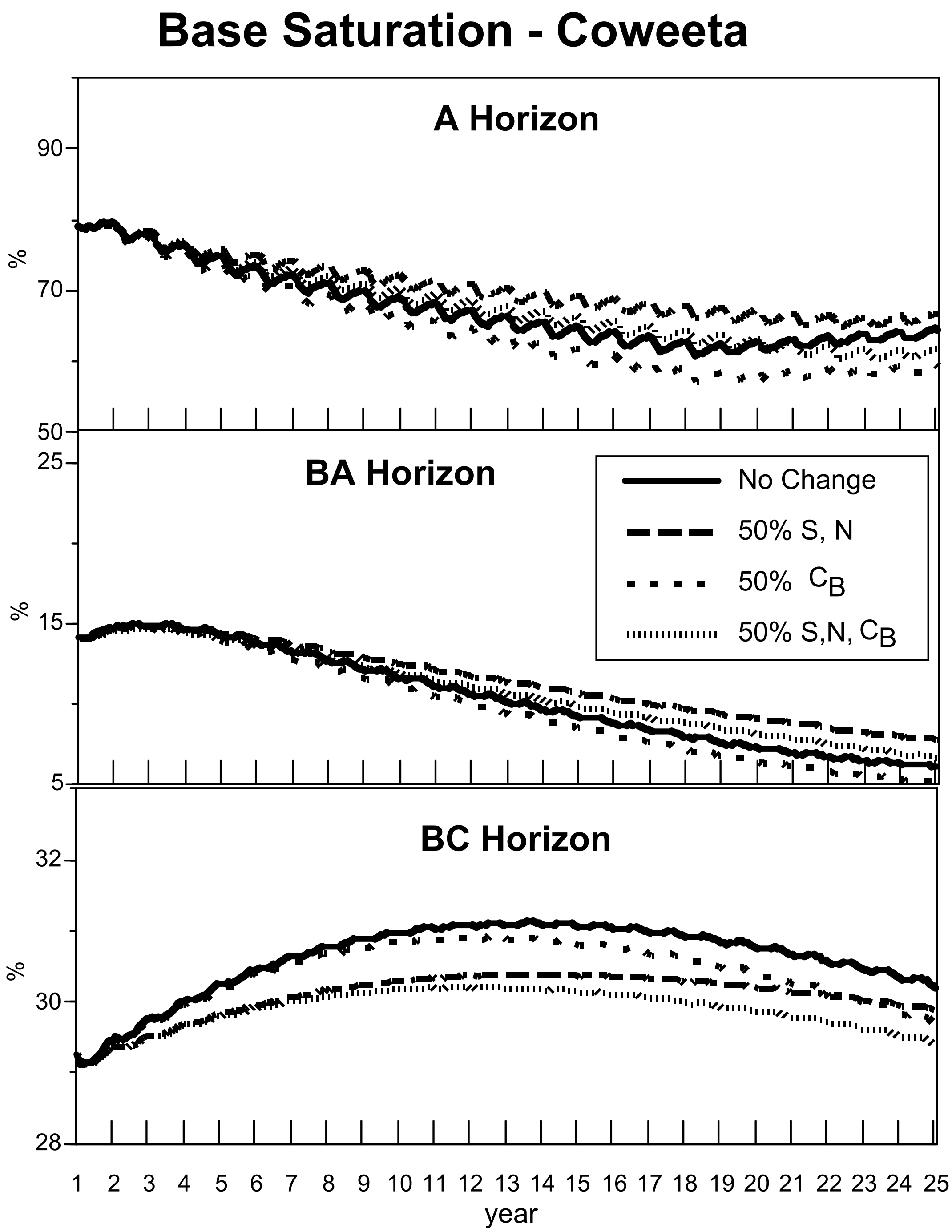


Figure 33. Simulated base saturation at Coweeta under no change, 50% reduction in nitrogen and sulfur deposition (50% N,S), 50% reduction in base cation deposition (50% BC), and 50% reduction in nitrogen, sulfur, and base cation deposition (50% N,S,BC). (After Johnson et al. 1999).

and Todd 1990, Johnson et al. 1991a, Richter et al. 1994). These discrepancies seem to reflect differences in scaling. If fluxes into and out of the entire soil profile are compared with soil nutrient contents, then the possibilities for change over periods less than centuries seem very limited. However, if soils are divided into their natural horizons and fluxes into and out of each horizon are estimated, then the possibility of change within a given horizon over much shorter time periods becomes quite obvious. If the soil is viewed as a continuous chromatographic column, the time frame for local change becomes short.

NuCM has added insights into the ways in which soils might change in response to acidic deposition, some of which have not yet been tested with field data. NuCM nearly always shows projected seasonal fluctuations in soil exchangeable base cations, as illustrated in the example from Coweeta in Figure 33. NuCM also suggests that long-term trends in soil change may not be uni-directional; simulated exchangeable bases often show both increases and decreases over time, reflecting adjustments in the overall cycling of these nutrients. This frequency of change may not be visible on the decadal time scale of most soil sampling programs. NuCM has also suggested inter-horizonal interactions in response to changes in deposition (such as reductions in base saturation in deep horizons) that are counterintuitive at first glance, but in fact reasonable upon reflection.

The example from Coweeta in Figure 33 is typical. In the A horizon, there is a continuous downward trend (with seasonal oscillations) in exchangeable calcium that is slightly mitigated by reduced sulfur deposition. In the BA horizon, the seasonal oscillations are damped, but there appears to be another longer-term oscillation of greater amplitude and much longer frequency (first an increase then a decrease). In the BC horizon, both seasonal and longer-term oscillations are nearly damped out. However, reduced sulfur deposition has the counterintuitive effect of causing reduced exchangeable calcium in lower horizons because of lower inputs from upper horizons.

One of the most illuminating "failures" of NuCM was the failure to reproduce the cation leaching patterns in a beech forest soil in the Great Smoky Mountains National Park. In the autumn of 1985, three months after establishing lysimeter plot, a large nitrate pulse occurred which caused concurrent pulses of cations in soil solution. The cause(s) of this nitrate pulse are not known but also are not particularly relevant for the testing of NuCM in this instance. As the nitrate pulse leached through the soil profile, it caused pulses in all cations in soil solution, as theory would predict. As the pulse passed through successively deeper horizons, the peaks in cation concentrations separated, just as in a chromatographic column. Figure 34A illustrates the patterns for magnesium, calcium, and aluminum observed in the Bw2 horizon field (Johnson 1995). NuCM failed to reproduce this chromatographic effect: all cations were simulated to peak simultaneously with nitrate in all horizons (Figure 34B). This response occurred even when the soil was broken into 10 horizons (as many as the model will allow). The consequences of this failure could be significant in terms of aluminum toxicity; because of the separation of peaks, calcium-to-aluminum ratios in the deepest horizons of the beech site were much lower than they would have been if all cations had peaked simultaneously (Figure 34A). NuCM greatly underestimated the responses of calcium-to-aluminum ratios to the nitrate pulse (Figure 35B), suggesting that the resolution of soil layers in NuCM might be too coarse to accurately predict the short-term temporal dynamics of soil solution.

NuCM, like other models of its kind, has too many adjustable parameters in the calibration process to produce consistent predictions. There are usually far more parameters needing values than there are values to apply to them. For example, NuCM requires that the user provide estimates of monthly uptake of nutrients (fraction of total annual uptake that occurs in each month). This information is almost never available, yet it is vital to the behavior of the model in an intra-annual time scale. It is of no consequence, however, in terms of the decadal interests of the SAMI assessment. Users of NuCM must make educated guesses at these and other required but unknown input parameters. These educated guesses are nearly always incorporated into the initial calibration of the model, leaving open the possibility of getting a wide variety of model output values for a given site from different investigators and/or calibrations. In short, the results of NuCM modeling should not be considered unique for a given set of site conditions. This is not a flaw particular to NuCM; other similar models which operate on comparable time scales require this kind of information also and perhaps make fixed assumptions about it in the code that are invisible to the user.

NuCM has proven to be relatively "successful" in mimicking decadal-scale changes in nutrient pools and soils in response to harvesting, species change, and liming. It has not proven so successful in mimicking shorter-term (intra-annual) variations in soil solution chemistry, despite the fact that soil chemistry is one of the model’s strong aspects. The reason for the latter is the lack sufficient knowledge of controls on soil chemical and biological processes over short time periods. Like all models of its type, NuCM does not adequately portray several important processes simply because we do not have enough knowledge to write code for these processes.

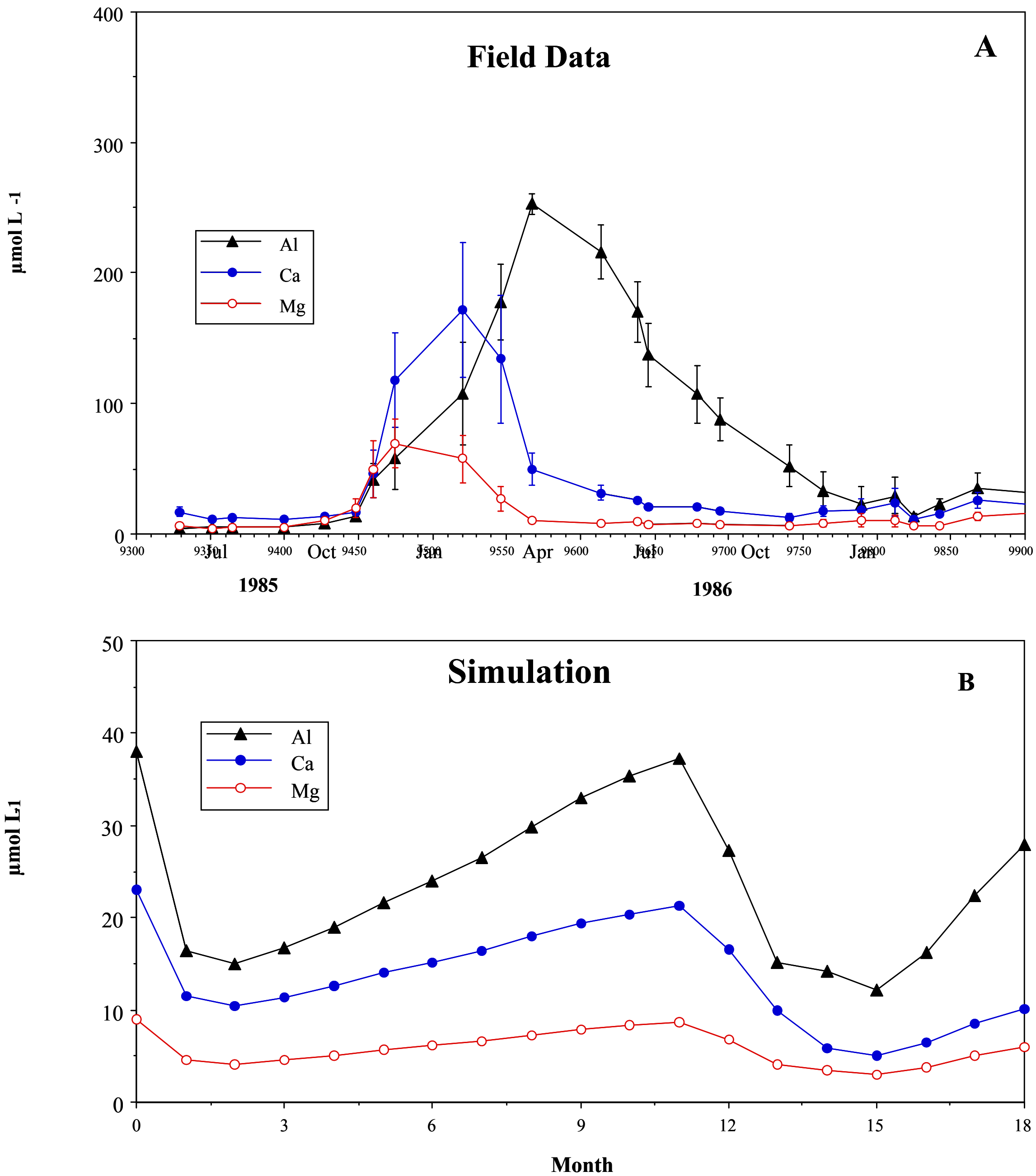


Figure 34. Field data (A) and simulation output (B) of soil solution aluminum, calcium, and magnesium in the Bw2 horizon of the Beech site near Noland Divide (after Johnson 1995)

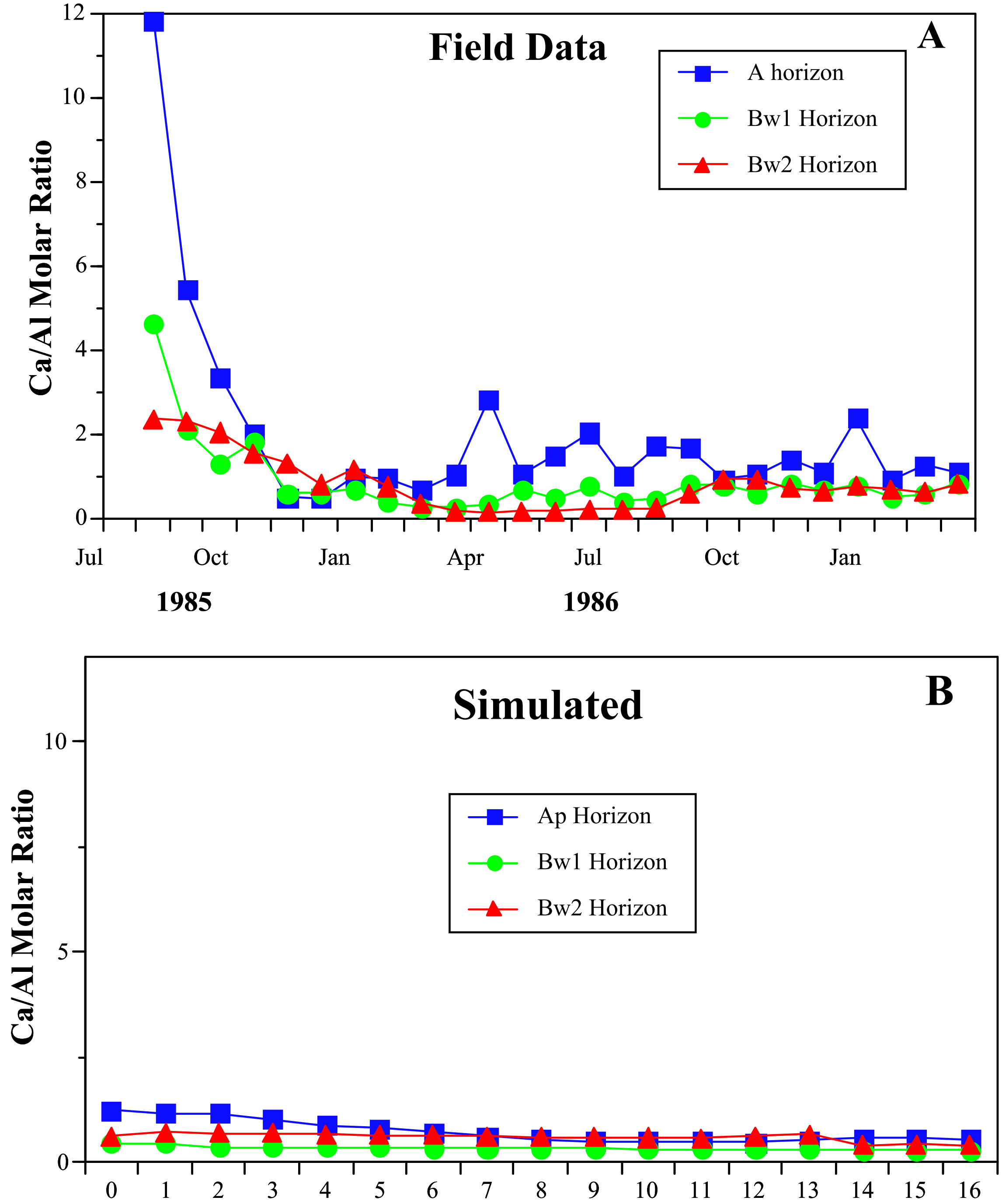


Figure 35. Field data (A) and simulation output (B) of soil solution Al/Ca ratios in the Beech site near Noland Divide (after Johnson 1995)

For example, nitrogen fixation is not represented in the model, and must be included with atmospheric deposition as an input to the terrestrial system. The soil weathering module in NuCM is complex and probably as good as any available, but soil weathering remains an elusive quantity to measure, let alone simulate. Soil weathering is not something that we can suggest as a need for improvement, however, because our knowledge of weathering processes and how to quantify them is inadequate.

Despite its shortcomings, which we believe are no worse than any other model of its kind, we find that NuCM has considerable heuristic value. For example, the model has provided us with considerable insight regarding the interactions of leaching and uptake in long-term nutrient budgets through species change simulations. It has confirmed that our basic understanding of the reactions of lime with acidic soils can account for the long-term retention of calcium on the exchange sites in a numerically rigorous fashion. The mixture of successes and failures of the NuCM model – even within the same set of simulations – has in itself provided valuable insight into modeling. These successes and failures have shown that a model which fails to mimic short-term temporal variation in soil solution chemistry (as in the beech simulations) can nevertheless provide reasonable assessments of long-term changes in soils. In areas where no field data are available, NuCM has provided interesting hypotheses which can help guide research, such as its predictions regarding the rapidity and nature of soil change and the effects of translocation on ecosystem nitrogen retention and cycling.

Finally, the availability of the various adjustable parameters in the model that cause problems for consistent prediction at the same time offer great opportunities for the exploration of the interactions among nutrient cycling processes that may not exist in less flexible models. For example, we have only begun to explore the implications of spatial (within litter and soil) and temporal (within season) variations in uptake that can be easily investigated with NuCM. As we continue to test and explore the model, many more such opportunities will certainly arise.

6.0 CONCLUSIONS

Results of applying the NuCM model to seven forest stands indicated future soil acidification, as represented by projected reductions in the base saturation of soils, at six of those sites under all strategies. The model projections also suggested that future decreases in base saturation would most likely be accompanied by decreases in the calcium-to-aluminum molar ratio in soil solution at most sites under all strategies. This model result suggests a future deterioration of soil conditions for forest growth and health, especially in the spruce-fir forests, which generally exhibited low calcium-to-aluminum ratios in soil solution in the reference year. All of the modeled spruce-fir sites and one of the modeled hardwood sites displayed calcium-to-aluminum ratios well below the published threshold of 1, indicated in previous studies to be an index of warning that an ecosystem might be entering a zone of increased stress (Cronan and Grigal 1995). The simulated continued decline of the calcium-to-aluminum ratio in soil solution at many of these sites in response to continued acidification and nutrient leaching suggests an increased likelihood of future adverse effects of soil conditions on forest growth and health.

Deposition of both sulfur and nitrogen are of concern with respect to potential acidification effects on forest soils and vegetation throughout the SA. However, the forest resource at greatest risk is the spruce-fir ecosystem that is found at high elevation. This resource is of greatest concern because it:

1. contains a tree species (red spruce) that is highly sensitive to damage by acidification;
2. exhibits soil properties that render it particularly sensitive to soil solution aluminum mobilization, including low pH and base saturation even in the absence of acidic deposition, and shallow rooting zone;
3. receives very high levels of cloud deposition, and therefore total deposition, of sulfur and nitrogen; and
4. has been shown to exhibit symptoms of nitrogen saturation.

Potentially sensitive spruce-fir forest resources are found at high elevation (generally above about 1,370 m) in the SA and in some areas of West Virginia and Virginia. In particular, Great Smoky Mountains National Park contains extensive spruce-fir forests, with almost three-fourths of the 20,000 ha being old-growth spruce-fir forest. The suspected mechanisms of acidification damage include increased mobilization of aluminum from soils to soil waters (Joslin and Wolfe 1992, Johnson et al. 1991b, Eagar et al. 1996), accelerated leaching of base cations from tree canopies, and decreases in the calcium-to-aluminum ratio in soil solution to levels that can be harmful to sensitive tree species (Eagar et al. 1996). It is likely that spruce-fir forests throughout the Appalachian Mountains have been adversely impacted by acidic deposition, based on field data that suggested positive response of red spruce to experimental additions of calcium at Great Smoky Mountains National Park, White Top Mountain (VA), and Mt. Mitchell (NC).

Results of the NuCM modeling exercises conducted for this assessment, together with the results of NuCM simulations published for other watersheds in the SA, suggest that spruce-fir forests in the region are likely to experience decreased calcium-to-aluminum ratios in soil solution under all strategies of future acidic deposition considered. This is partly because sulfur adsorption in soils is likely to decline, even with dramatically reduced sulfur deposition. In addition, many spruce-fir forests in the region are nitrogen-saturated, and continued nitrogen deposition will contribute to elevated nitrate concentrations in soil water, which will further enhance base cation leaching and mobilization of aluminum from soils to soil solution. These processes will be facilitated by the already low values of base saturation in the soils of many of these forests. Results of modeling efforts conducted to date are consistent in suggesting that such changes to forest soils and soil solutions will likely continue to occur.

It is not clear, however, to what extent these changes in the chemistry of soils and soil solutions might actually impact forest growth and health. The state of scientific understanding on this topic would suggest that such chemical changes would increase the likelihood that the growth and/or health of spruce-fir forests would be adversely impacted, perhaps making them more susceptible to other stressors associated with such factors as insect pests, pathogens, or extreme climatic conditions. However, the occurrence of low base saturation and calcium-to-aluminum ratio in soil solution will not necessarily be sufficient to cause widespread impacts. Many factors in addition to soil base saturation and soil solution acid-base chemistry are important in this regard. Recent evidence indicates that mortality in red spruce in the SA is not abnormal when compared to historical rates, and that Fraser fir stands killed by the balsam wooly adelgid are largely being replaced by vigorous re-growth of young stands of that species. To what extent spruce or fir mortality will be replaced with a species mix similar to that existing prior to the mortality remains to be seen.

The limited available empirical data suggest that the kinds of changes in soil solution chemistry projected by NuCM for spruce-fir stands in the SAMI region will be consistent with the kinds of changes that have been associated in the past with reductions in forest growth. The weight of evidence for spruce-fir forests suggests that adverse impacts on soil solution chemistry are likely, and adverse impacts on forest growth and health are possible. Changes in red spruce growth rates are potentially attributable, at least in part, to base cation deficiencies caused by inhibition of base cation uptake by trees due to elevated aluminum concentration in soil solution within the rooting zone. Other factors that could also be important include depletion of base cations in upper soil horizons by acidic deposition, aluminum toxicity to tree roots, and accelerated leaching of base cations from foliage as a consequence of acidic deposition, especially cloud deposition.

The state of scientific understanding is less clear with respect to the lower elevation hardwood forests within the SAMI region. Although NuCM projected decreasing soil base saturation and soil solution calcium-to-aluminum ratio at some sites, there are limited data available that would associate such projected chemical changes with adverse forest effects. Available information is not sufficient to draw conclusions regarding the increased likelihood of future effects on the condition of hardwood forests in the region. Certainly, such effects are less likely for hardwood forests than for spruce-fir forests.

Based on published research findings, it is unlikely that terrestrial ecosystems at lower elevations within the SAMI region have experienced sufficiently high deposition of sulfur or nitrogen so as to cause widespread adverse acidification-related impacts to forests. There are several reasons why such widespread impacts are unlikely. First, the vegetation type that appears to be most susceptible to damage from acidic deposition, spruce-fir forest, is largely absent from lower elevation sites. Second, cloud deposition of sulfur and nitrogen becomes a large contributor to total sulfur and nitrogen deposition only at high elevation. For example, Sigmon et al. (1989) estimated total warm season cloud deposition at the Pinnacles of Shenandoah National Park, Virginia (1,014 m elevation) equal to 0.9, 0.7, and 0.2 kg/ha/mo, respectively, for sulfate, nitrate, and ammonium. These deposition estimates contrast with the estimates by Lindberg et al. (1988) at Great Smoky Mountains National Park (1,740 m elevation) that were more than three-fold higher for nitrate, eight-fold higher for sulfate, and five-fold higher for ammonium (c.f., Vong et al. 1991). As a consequence, total sulfur and nitrogen deposition levels are much higher at the higher elevation Great Smoky Mountains site. Third, total deposition levels of both sulfur and nitrogen in most low-elevation areas within the region are considerably below levels that have been shown to cause adverse impacts on forest ecosystems in Europe (Tietema and Beier 1995, Dise and Wright 1995, Dise et al. 1998, Fenn et al. 1998). Fourth, because forests throughout the region are mostly second growth, and soils are low to deficient in nitrogen supply subsequent to widespread timber harvesting and localized agricultural land use, the nitrogen-demand of the regrowing forest is likely to be high. This precludes nitrate leaching at current (and probably moderately higher) nitrogen deposition levels in the absence of significant disturbance such as was seen with gypsy moth infestation in

Virginia (Webb et al. 1995). Fifth, it is likely that most lower elevation forests, as compared with the high-elevation spruce-fir forests, have higher base saturation and soil pH and would therefore not be expected to show the same level of cation deficiency and aluminum stress that sometimes characterize the higher elevation sites (Eagar et al. 1996).

Thus, although there is some evidence that high-elevation spruce-fir forests have experienced adverse effects from nitrogen deposition, it is important to note that total nitrogen deposition at such sites is often very high, due to the high amounts of cloud deposition received in those high elevation areas. As a consequence, the streams that drain undisturbed watersheds in the Great Smoky Mountains exhibit some of the highest recorded nitrate concentrations in the United States and also nitrate concentrations that can be comparable to or higher than sulfate concentrations (Silsbee and Larson 1982).

Because of the limited number of sites modeled with NuCM for this study and because of the paucity of available information to link soil solution chemistry and forest health, results from this study are not adequate for quantifying regional responses of forests within the SAMI domain to acidification stress. In particular, there are considerable uncertainties associated with interpretation of the base saturation and calcium-to-aluminum ratio indices. Variability within and among watersheds is extremely high, especially in association with elevation and species mixes.

Although the modeled sites, especially the spruce-fir sites, generally showed projected future deterioration in soil solution acid-base chemistry, the differences among strategies were small. There are also large uncertainties associated with interpreting the importance of the model projections with respect to the probability of forest impacts. Therefore, the results of these projections are not adequate as a basis for recommending one Emissions Control Strategy over another.

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APPENDIX A

NuCM MODELING FOR SITES CALIBRATED TO STREAM DATA

A.1. Background

The NuCM model was calibrated to seven watersheds in the SAMI region for which soil solution data were not available. For these sites, the calibrations were based on streamwater chemistry data. Calibration results and projected values of soil base saturation and Ca:Al raio in soil solution are provided here. However, because the calibrations were not based on soil solution, the projections are not considered reliable for evaluating forest effects.

A.2. Site Descriptions

Dolly Sods

The Dolly Sods site consists of a second-growth spruce-fir forest that also contains yellow birch, red maple and black cherry in the Dolly Sods Wilderness area of West Virginia at an elevation of 933 m. Annual precipitation at the site averages just over 130 cm. The watershed is underlain by Pennsylvanian Pottsville bedrock with local colluvium over sandstone-shale over moderately weathered residuum from shale-siltstone. The slope at the site is 22%, and observed base saturation values range from 2 to 15%. The site was logged from 1870-1900, and then burned following the logging. There was also coal mining activity during and shortly after the logging. The site was grazed in the early 1900s, and army artillery maneuvers in 1943 left behind artillery and mortars. Model calibration for the Dolly Sods site was based upon streamwater data from Little Stonecoal Run.

Otter Creek

The Otter Creek site consists of a second-growth spruce-fir forest that also contains yellow birch, red maple and black cherry in the Otter Creek Wilderness area of West Virginia at an elevation of 1160 m. Annual precipitation at the site averages just over 138 cm. The watershed is underlain by Pennsylvanian Pottsville bedrock with slightly weathered residuum from shale-siltstone. The slope at the site is 4-17%, and observed base saturation values range from 5 to 25%. The site was logged from 1909-1915, and then burned following the logging. Model calibration for the Otter Creek site was based upon streamwater data from Devils Gulch.

Mt. Rogers

The Mt Rogers site consists of a maple/beech/birch forest in southwestern Virginia at an elevation of 1325 m. Annual precipitation at the site averages just over 125 cm. The site is underlain by the Mt. Rogers formation and contains high silica rhyolite. The slope at the site is 40%, and observed base saturation values range from 9 to 26%. The solid phase data used for model application to the Mt. Rogers and James River Face sites were generated as a part of the SAMI effort and are presented in Table G.1. The site was logged from 1905-1930. A small area of the upper watershed had heavy slash burning for about 10 years after logging and then was grazed from the 1930s through the 1960s. Cattle grazing continues today, but at a much lower intensity. Model calibration for the Mt. Rogers site was based upon streamwater data from Lewis Fork.

Coweeta Watershed 36

The Coweeta Watershed 36 site consists of an hickory/oak forest in the Coweeta Experimental watershed in western North Carolina at an elevation of 1021 m. Annual precipitation at the site averages just over 270 cm. The site is underlain by quartz minerals with minor components of plagioclase, muscovite, and biotite. The slope at the site is 65%, and observed base saturation values range from 7 to 25%. The site has no history of fire or logging. Model calibration for the Coweeta Watershed 36 site was based upon streamwater data (Gherini et al. 1989).

James River Face

The James River Face site consists of a mixed oak/pine forest in the James River Face Wilderness area of Virginia at an elevation of 475 m. Annual precipitation at the site averages just over 115 cm. The site is underlain by the Chilhowee group and contains quartz and phyllite. The slope at the site is 40%, and observed base saturation values range from 9 to 19%. The solid phase data used for model application to the Mt. Rogers and James River Face sites were generated as a part of the SAMI effort and are presented in Table G.1. These data include exchangeable cation and aluminum concentrations in the A horizon (U) and B horizon (L). The site has not been logged, but a large outbreak of southern pine beetle in the mid 1990s has left a large area of dead timber. Model calibration for the James River Face site was based upon streamwater data from Belfast Creek.

North Fork Dry Run

North Fork Dry Run is a 223 ha catchment in Shenandoah National Park that ranges in elevation from 503 to 1036 m. Average annual rainfall in the area ranges from near 100 cm at a nearby lower elevation site to nearly 130 cm at a nearby higher elevation site. The vegetation in the watershed is a mixed-mesophytic oak-hickory forest with some hemlock at higher altitudes. The watershed lies on the Pedlar Formation, which contains some basaltic intrusions, but comprises primarily granulites made up of quartz, potassium feldspar, plagioclase feldspar, and pyroxene. The soils occur chiefly as well-drained, very rocky, old, strongly acid, sandy Hapludults. Maximum soil depths of greater than 153 cm have been observed. The Shaver Hollow watershed was settled and partially logged before 1936. There is also evidence that farming was practiced historically in the watershed (Pauley et al. 1996, Ryan et al. 1989, Cozzarelli 1986, Currie 1992).

White Oak Run

White Oak Run lies in a 515 ha catchment in Shenandoah National Park that ranges in elevation from just under 500 to greater than 900 m. Average annual rainfall in the area ranges from near 100 cm at a nearby lower elevation site to nearly 130 cm at a nearby higher elevation site. The vegetation in the watershed is second growth forest 50 to 80 years old, consisting mostly of white oak and chestnut oak. The watershed lies primarily on the Hampton Formation, an early Cambrian brown to greenish gray shale and siltstone with interbedded ferruginous quartzite. The soils are classified as typic dystrochrepts that are 60 to 200 cm in depth, are strongly acidic, and are described as silty or silty clay loams (Pauley et al. 1996, Ryan et al. 1989, Cozzarelli 1986, Currie 1992).

A.3. Calibration Results

Results of calibrating NuCM to streamwater chemistry data in these seven watersheds are shown in Figures A-1 to A-7. In general, simulated streamwater chemistry matched observed values rather well. Observations were very limited, however, for the Dolly Sods and Otter Creek sites.

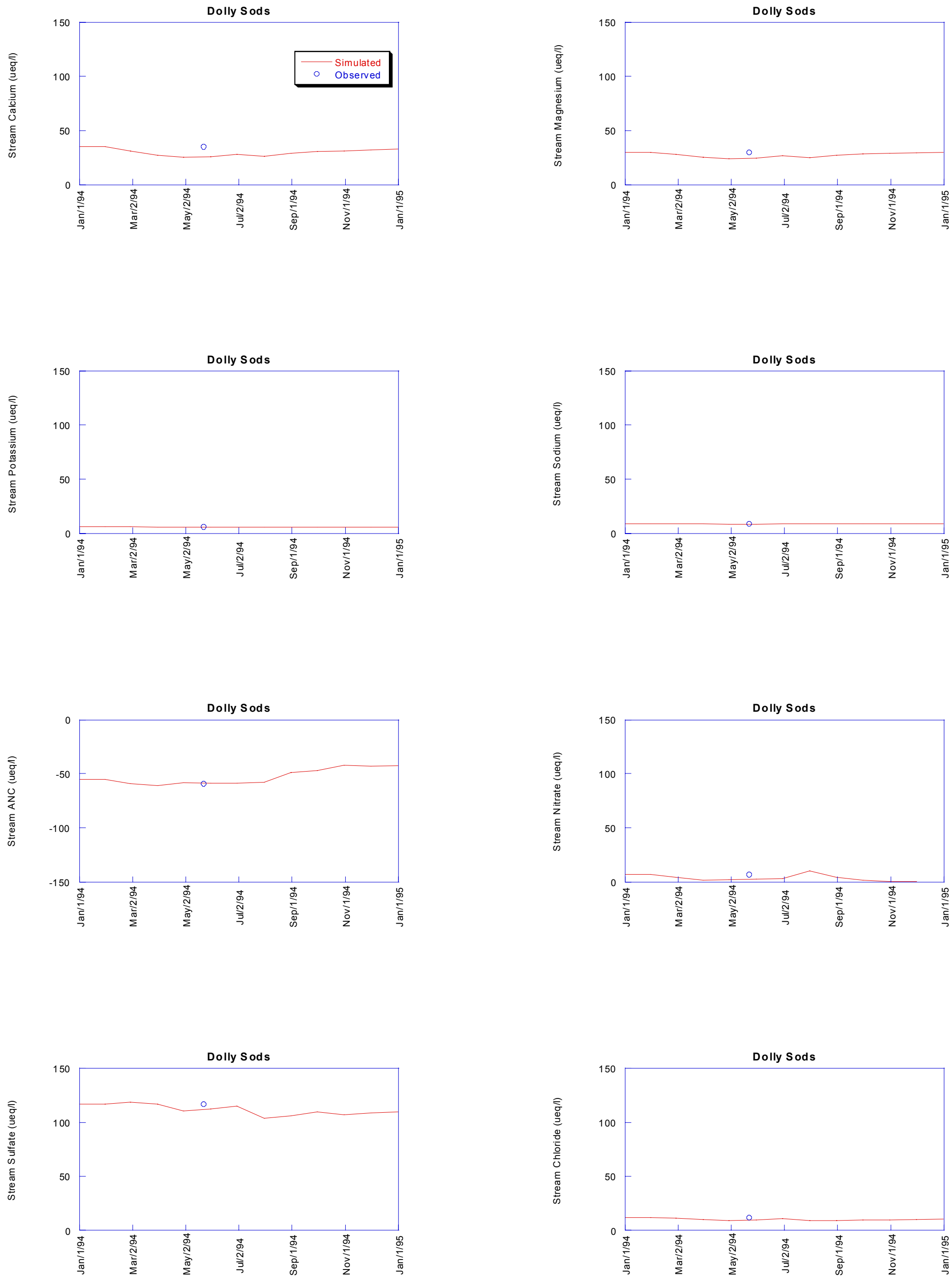


Figure A-1. Simulated versus observed concentrations for the stream at the Dolly Sods Spruce Site.

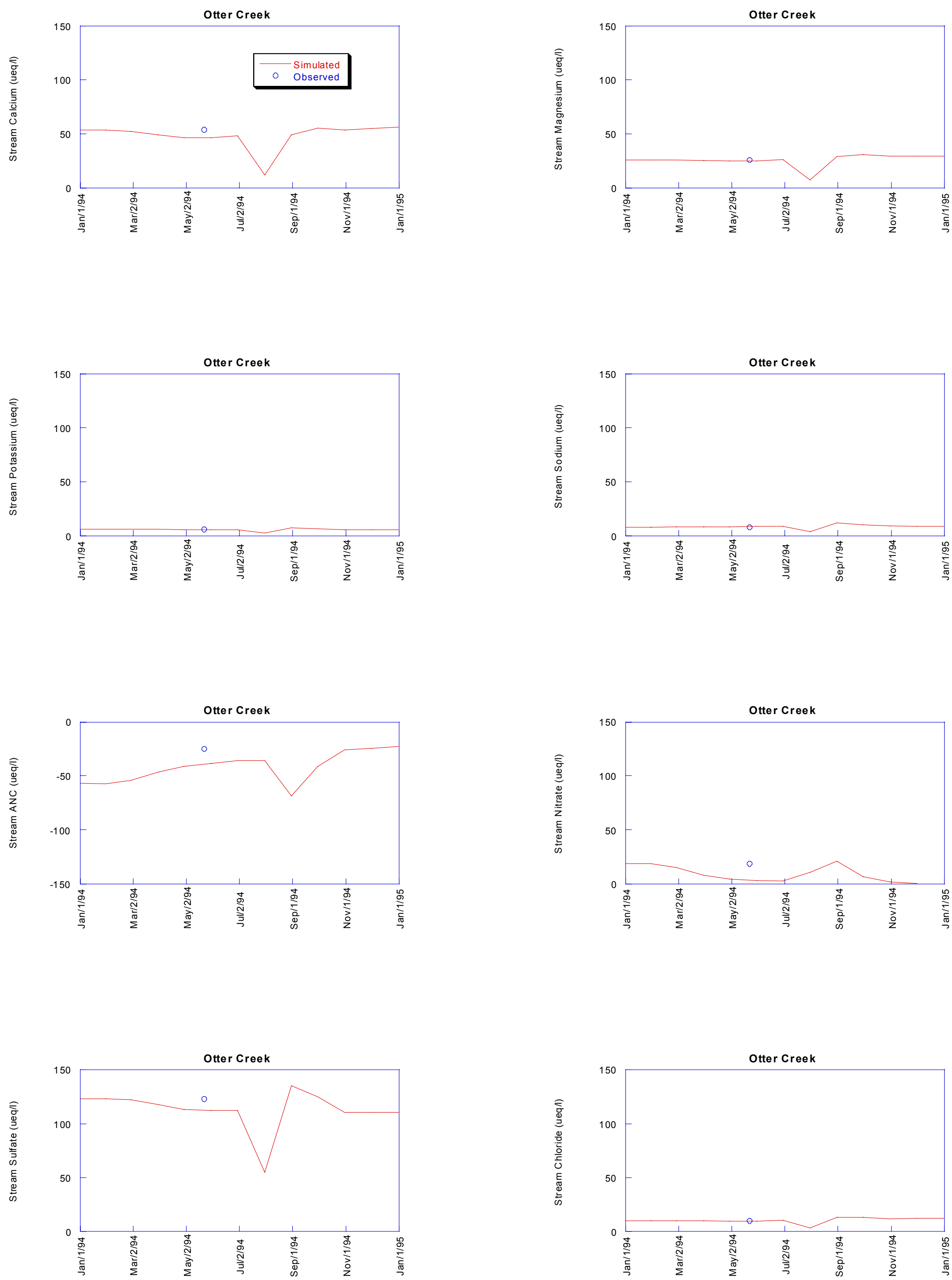


Figure A-2. Simulated versus observed concentrations for the stream at the Otter Creek Spruce Site.

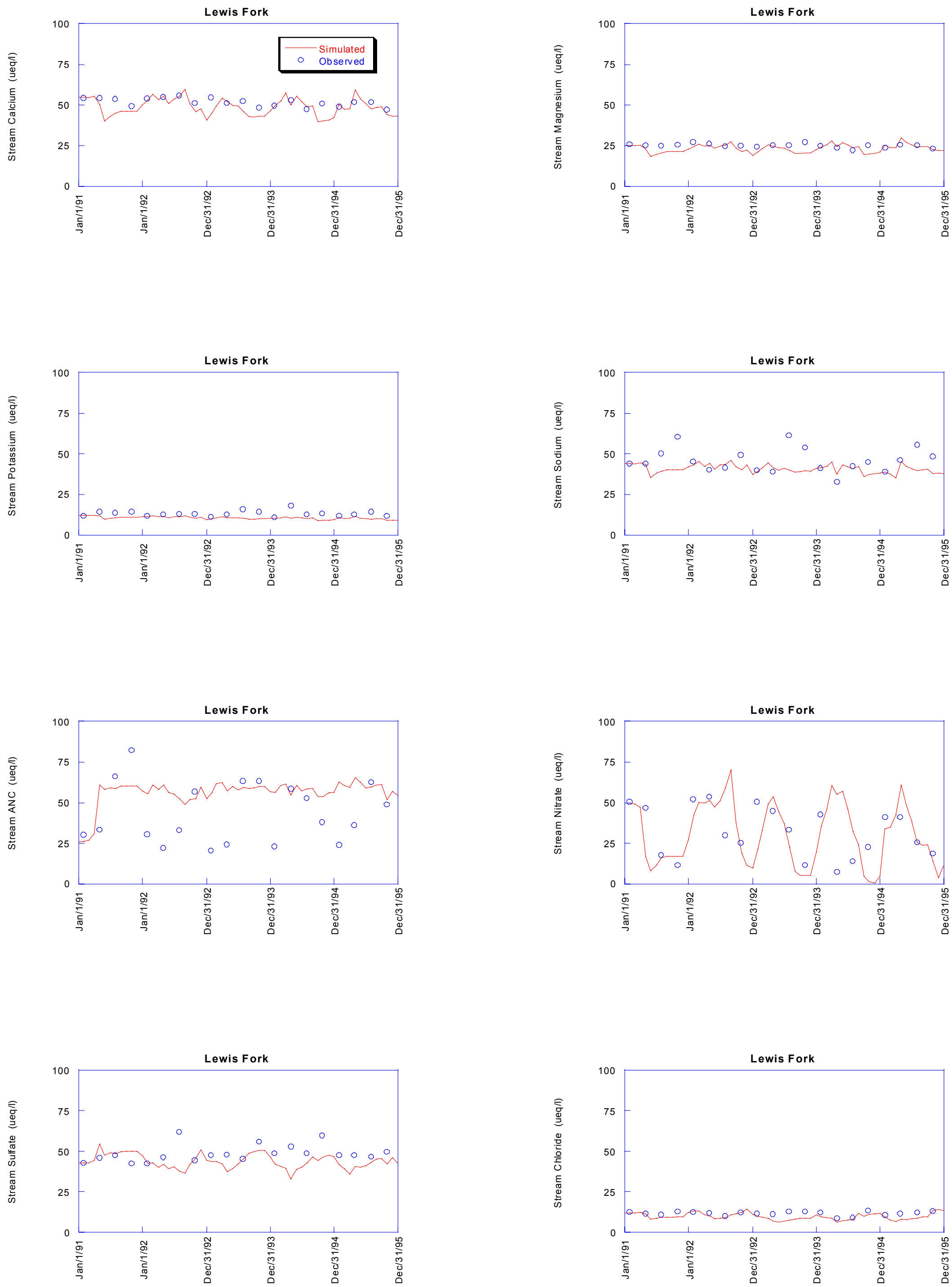


Figure A-3. Simulated versus observed concentrations for the stream at the Lewis Fork-Mt. Rogers Northern Hardwood Site.

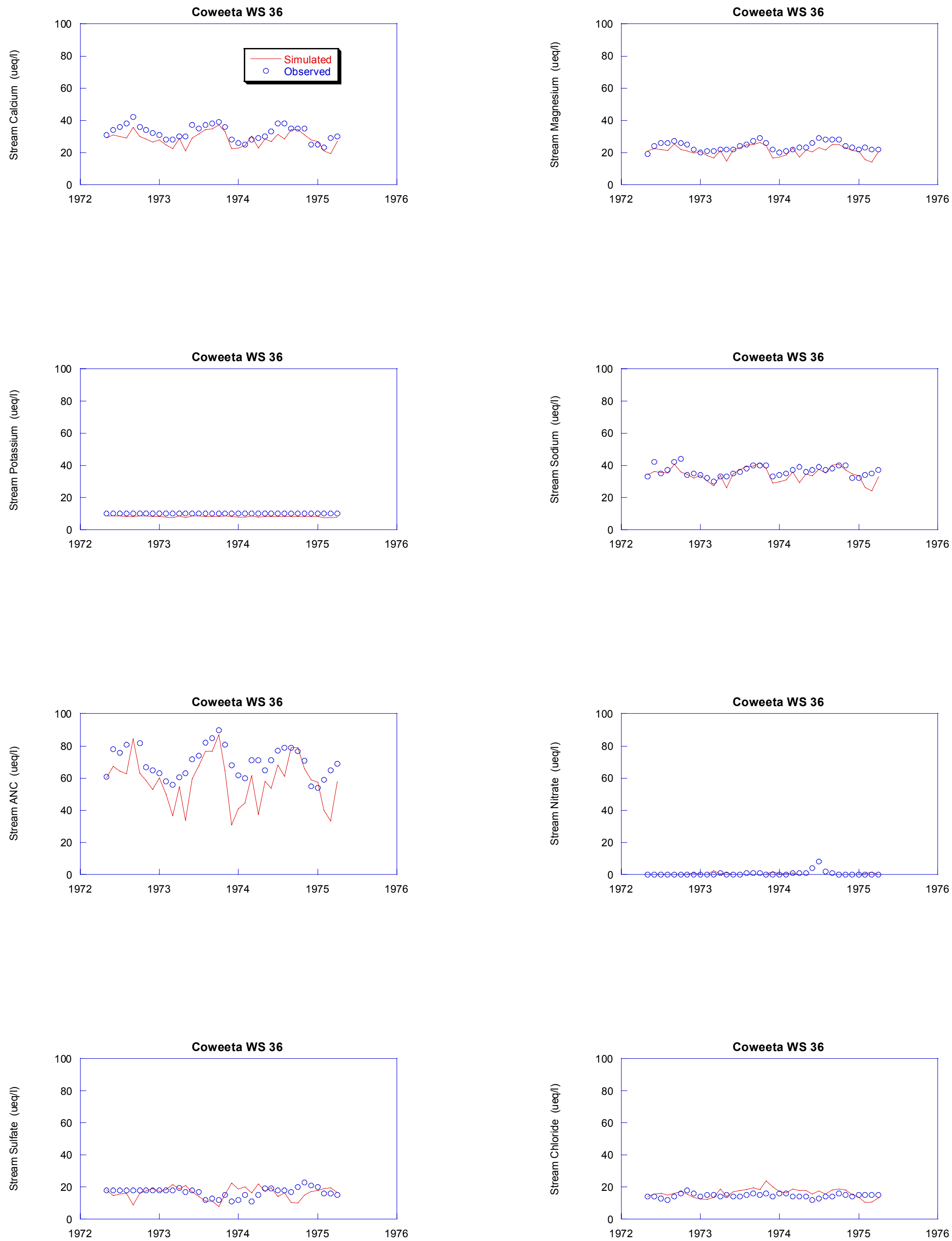


Figure A-4. Simulated versus observed concentrations for the stream at the Coweeta Watershed 36 Northern Hardwood Site.

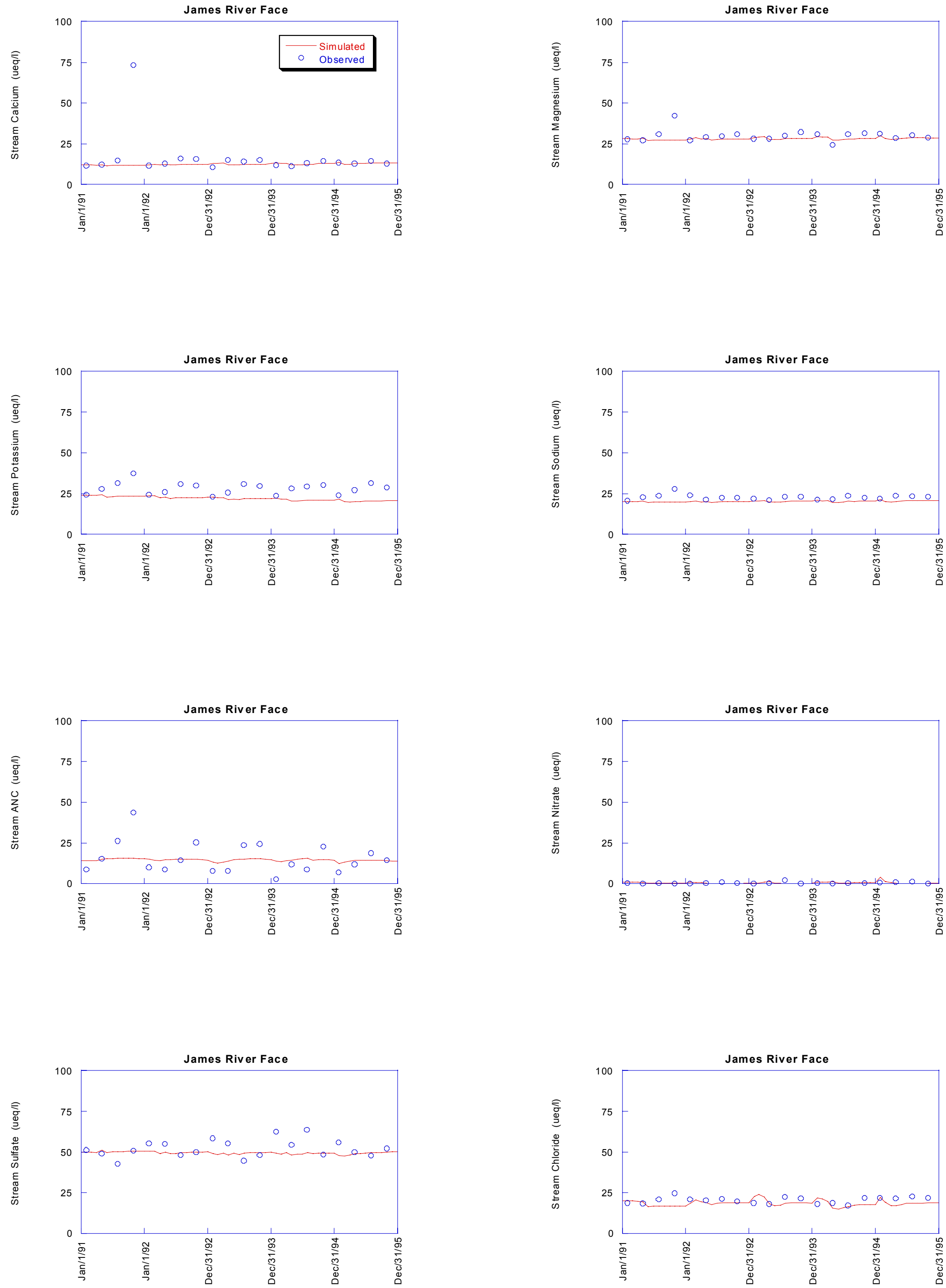


Figure A-5. Simulated versus observed concentrations for the stream at the James River Face Mixed Hardwood Site.

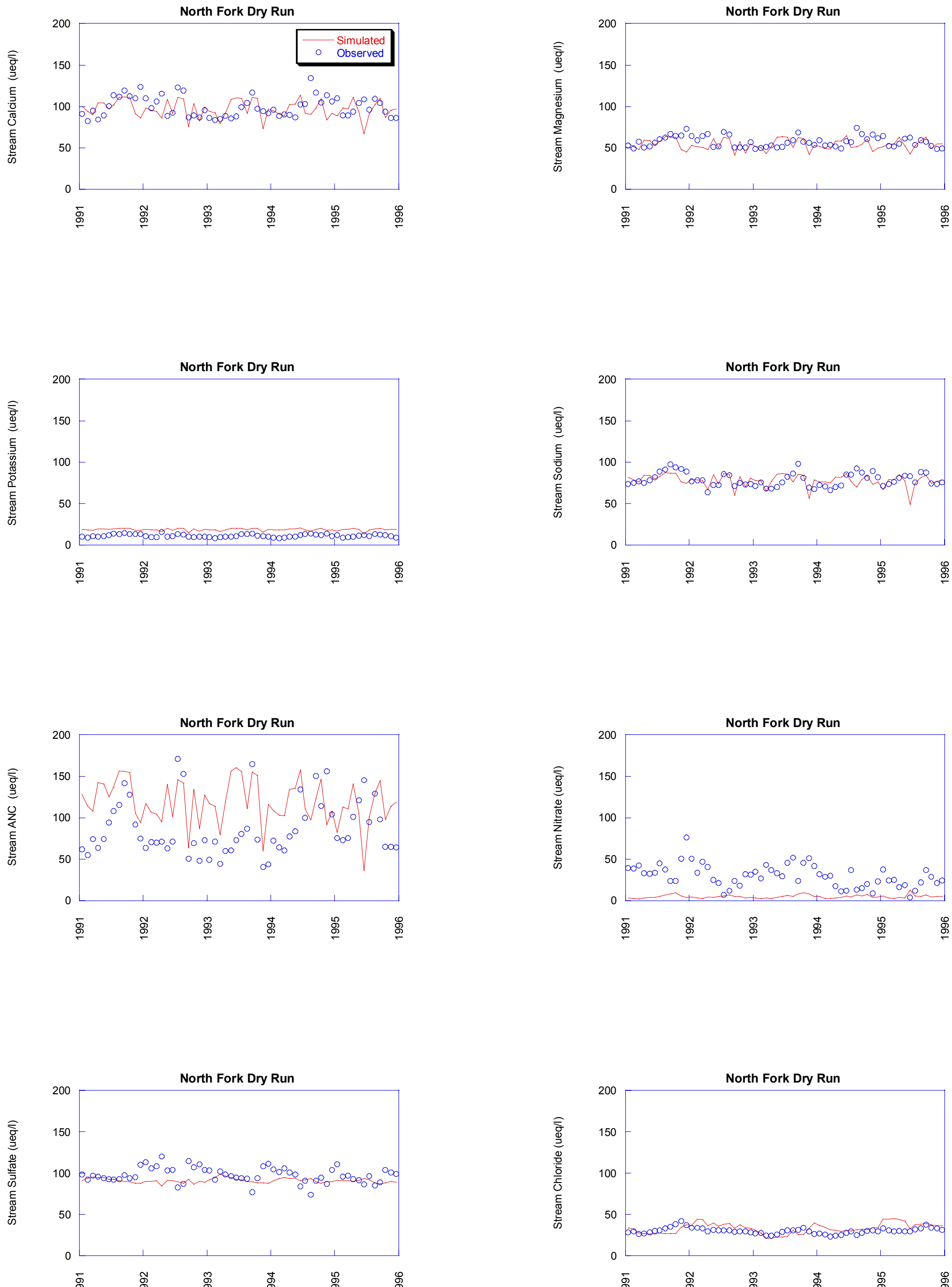


Figure A-6. Simulated versus observed concentrations for the stream at the North Fork Dry Run Mixed Hardwood Site.

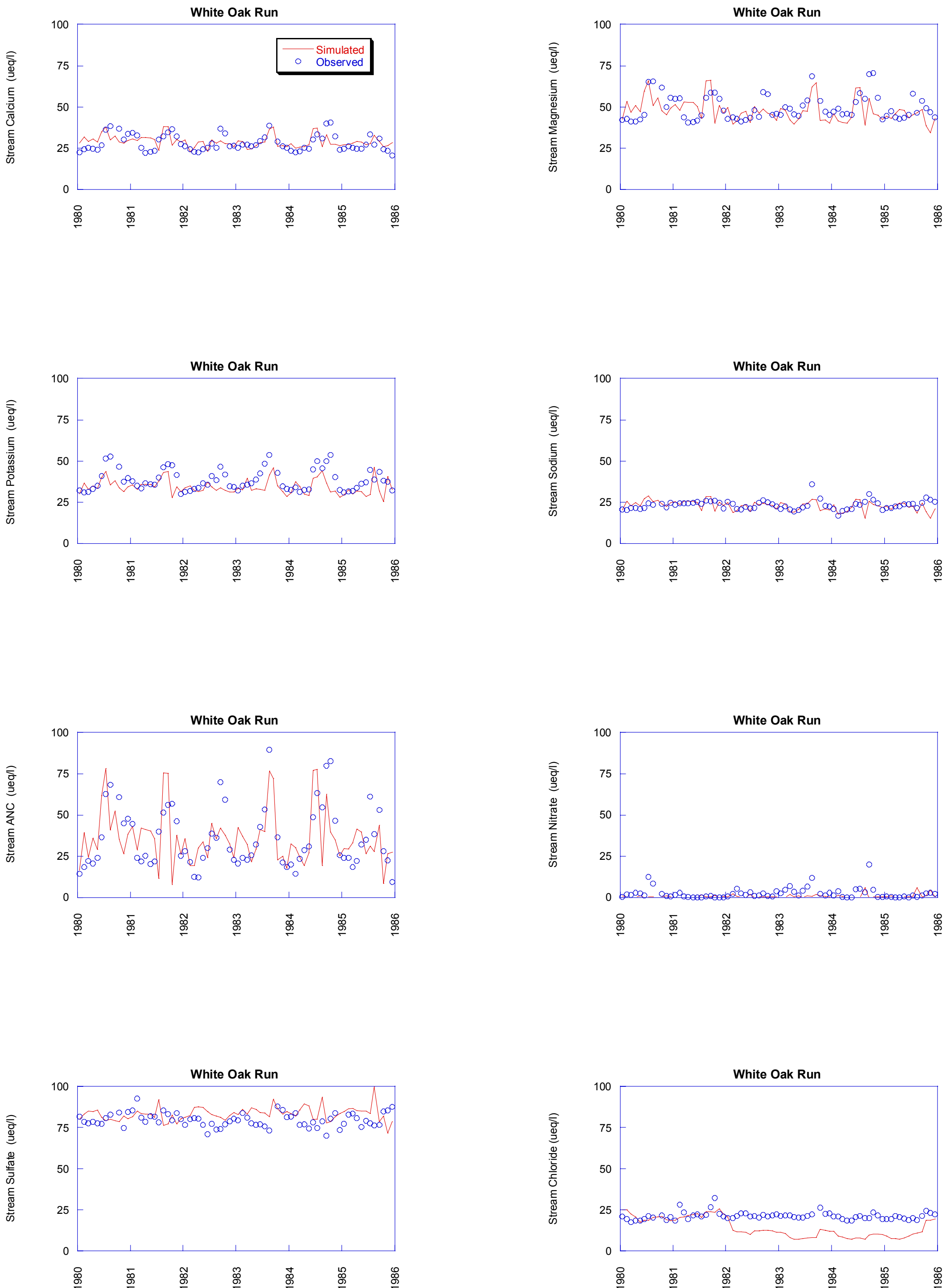


Figure A-7. Simulated versus observed concentrations for the stream at the White Oak Run Mixed Hardwood Site.

A.4. Model Projections

Simulated response of the seven watersheds to the scenario of constant deposition at reference year (1995) values and in response to each of the three Emissions Control Strategies are shown in Figures A-8 through A-14.

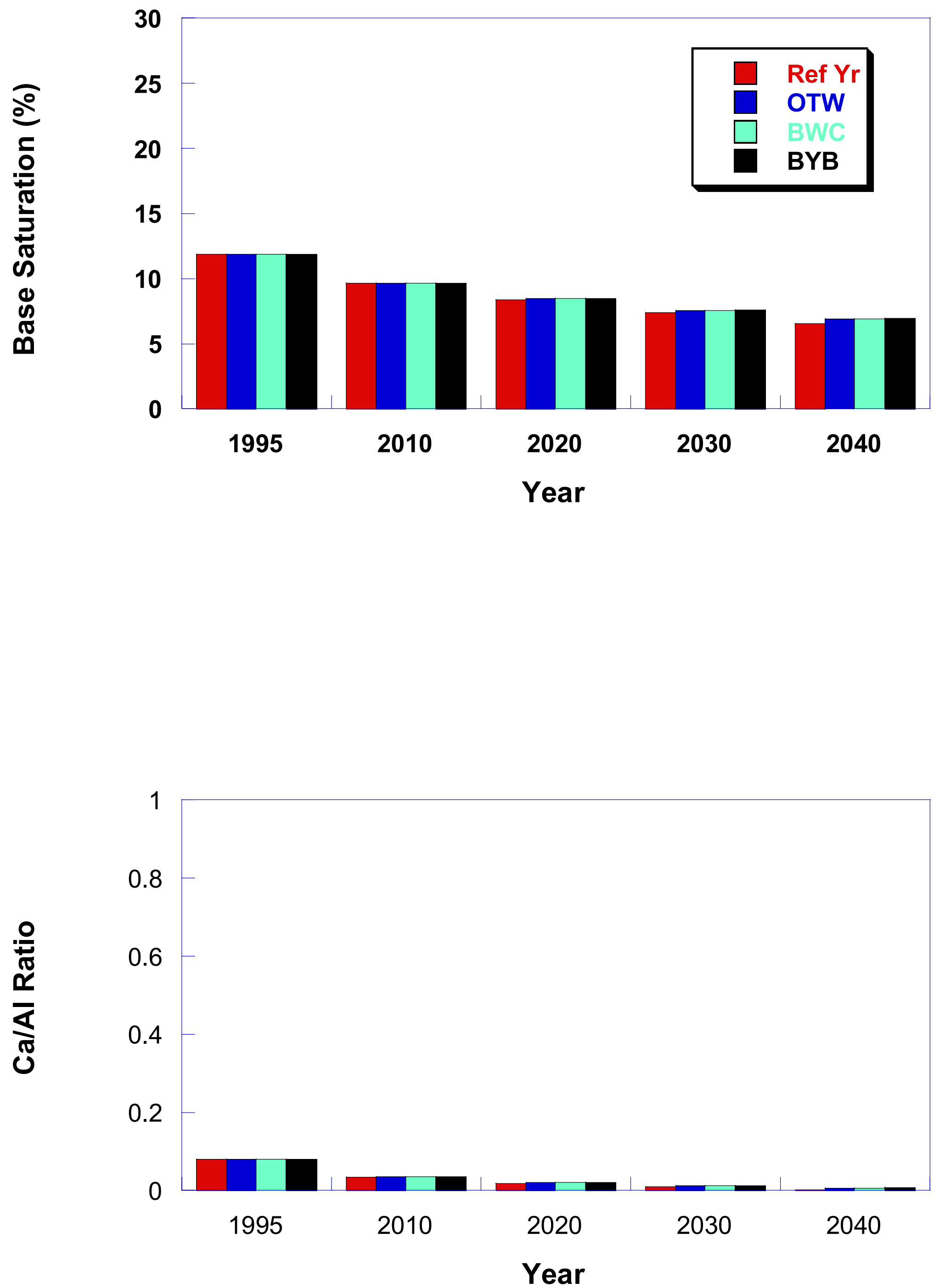
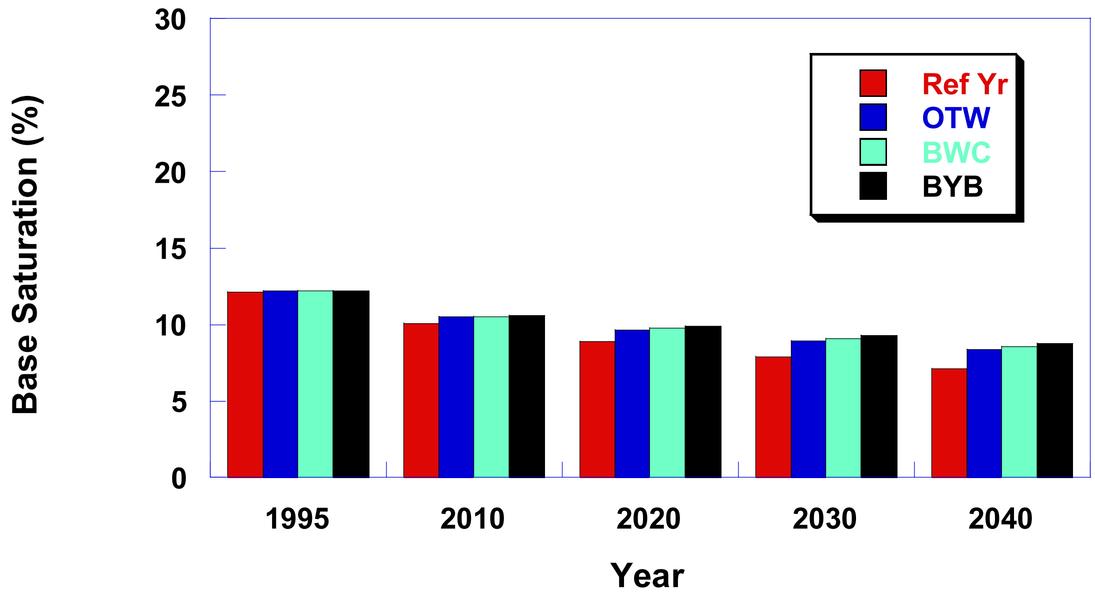


Figure A-8. Simulated response to changes in atmospheric deposition in the rooting zone (A horizon) at the Dolly Sods Spruce Site.



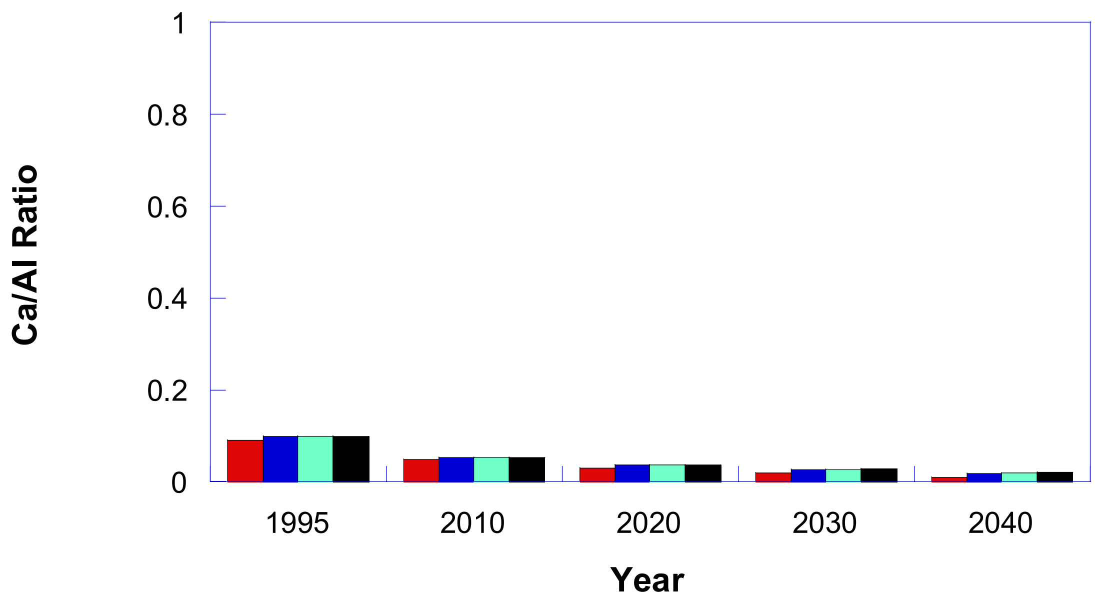


Figure A-9. Simulated response to changes in atmospheric deposition in the rooting zone (A horizon) at the Otter Creek Spruce Site.

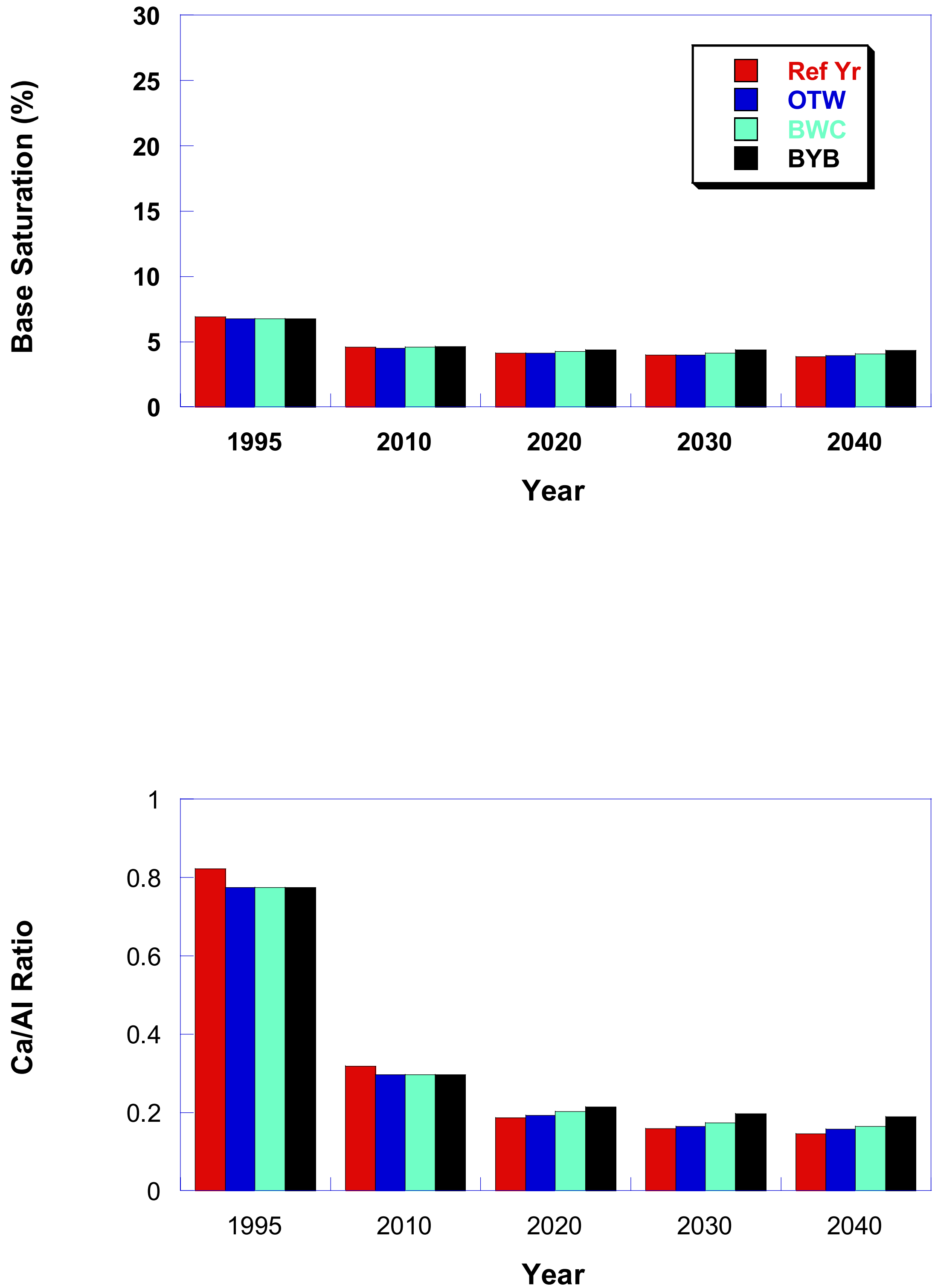
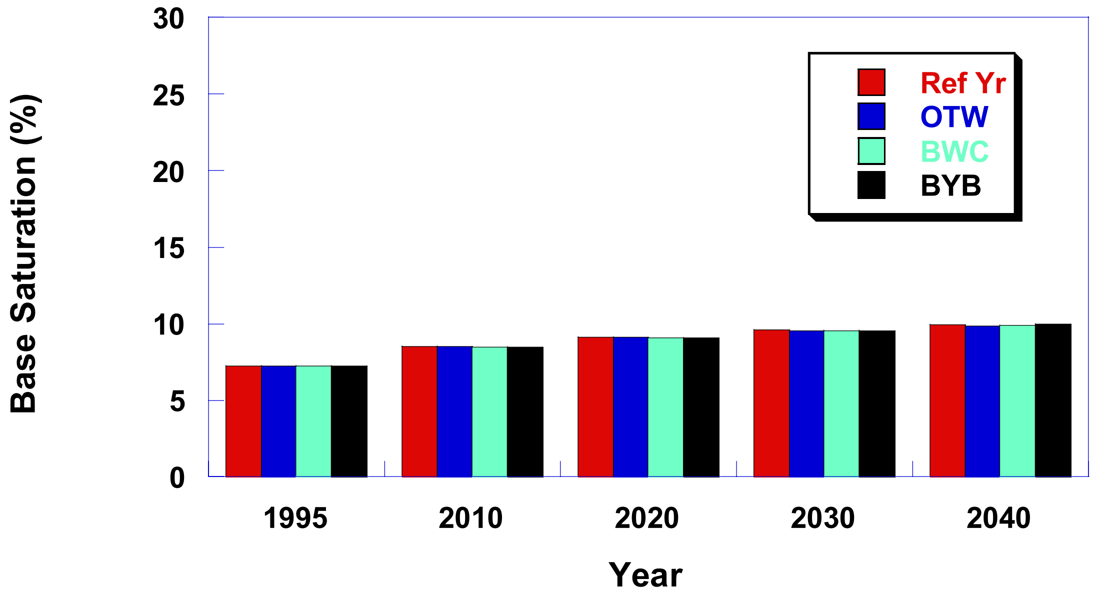


Figure A-10. Simulated response to changes in atmospheric deposition in the rooting zone (A horizon) at the Lewis Fork-Mt. Rogers Northern Hardwood Site.



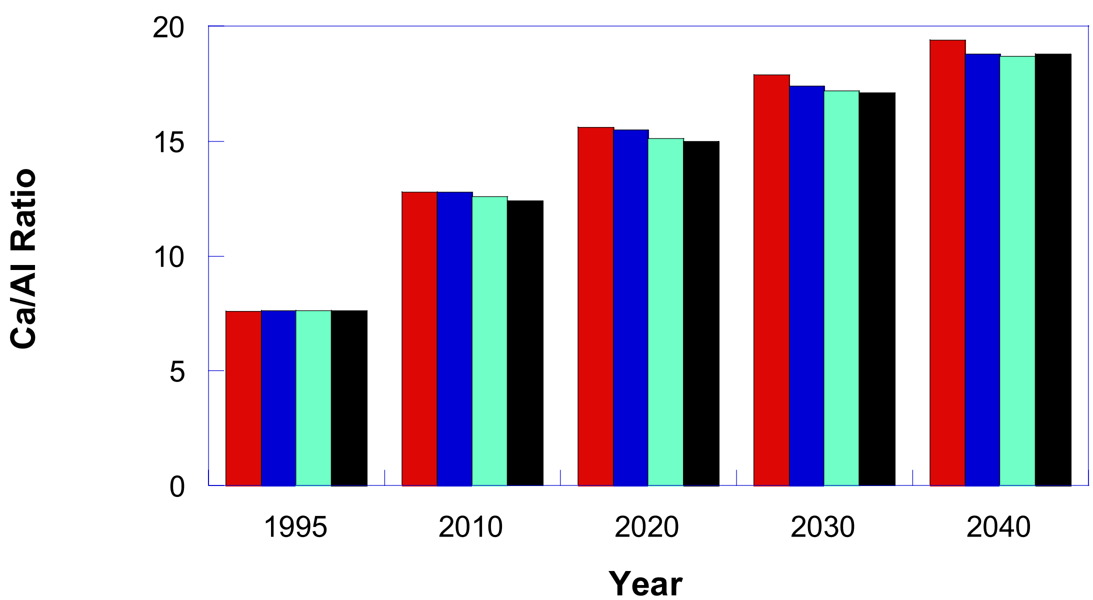


Figure A-11. Simulated response to changes in atmospheric deposition in the rooting zone (A horizon) at the Coweeta Watershed 36 Northern Hardwood Site.

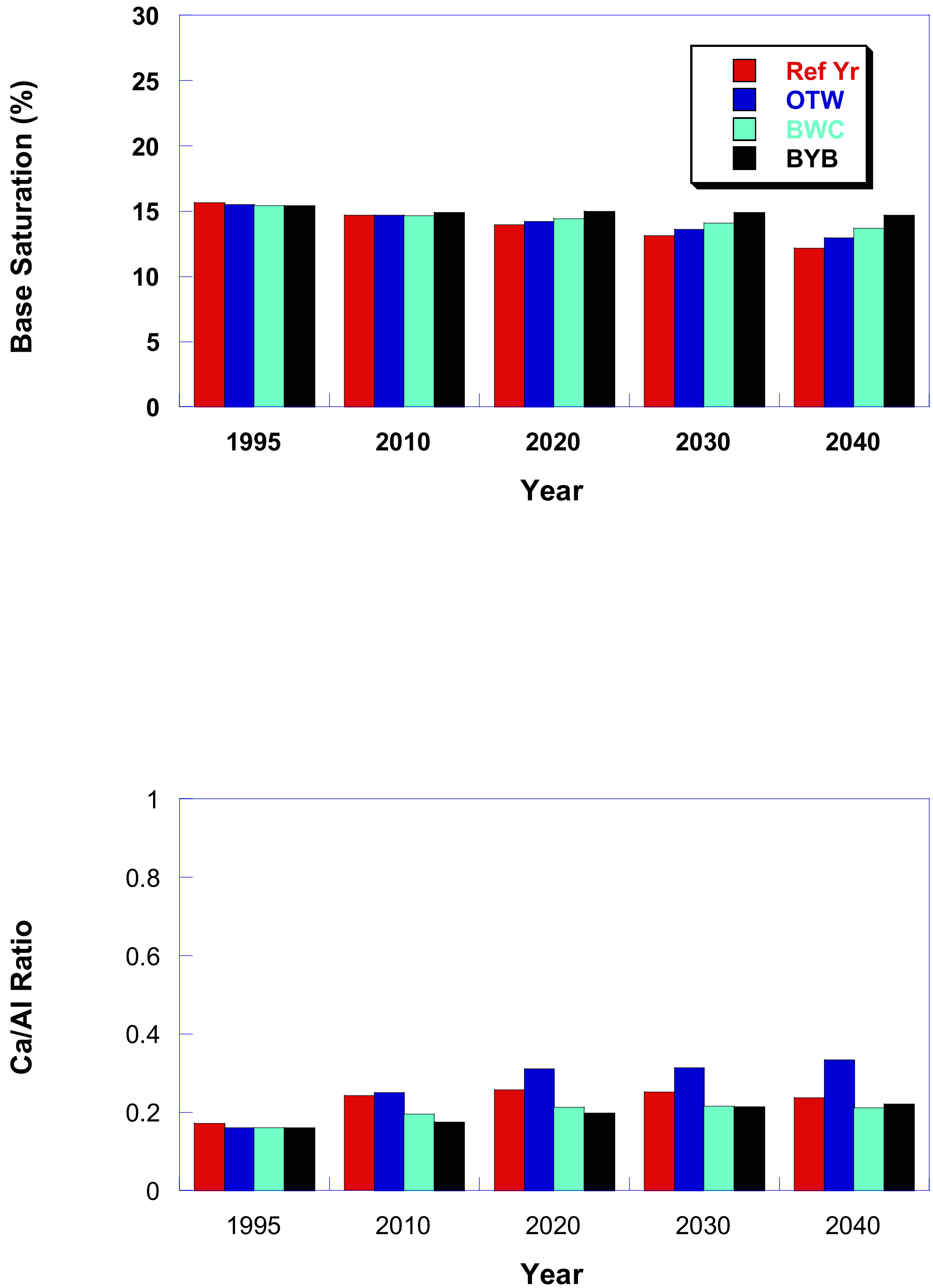
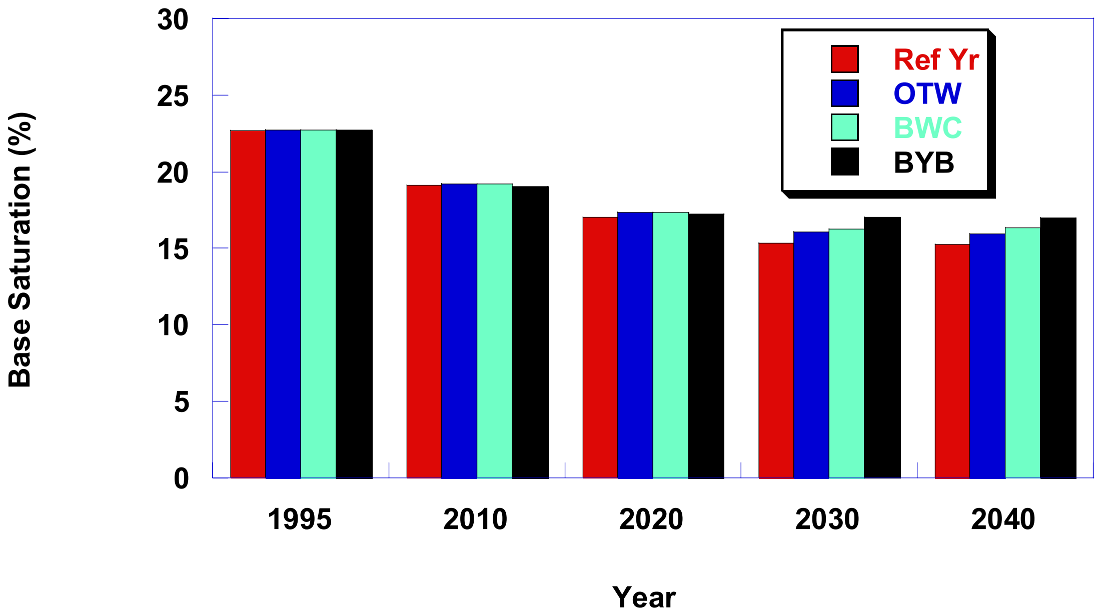


Figure A-12. Simulated response to changes in atmospheric deposition in the rooting zone (A horizon) at the James River Face Mixed Hardwood Site.



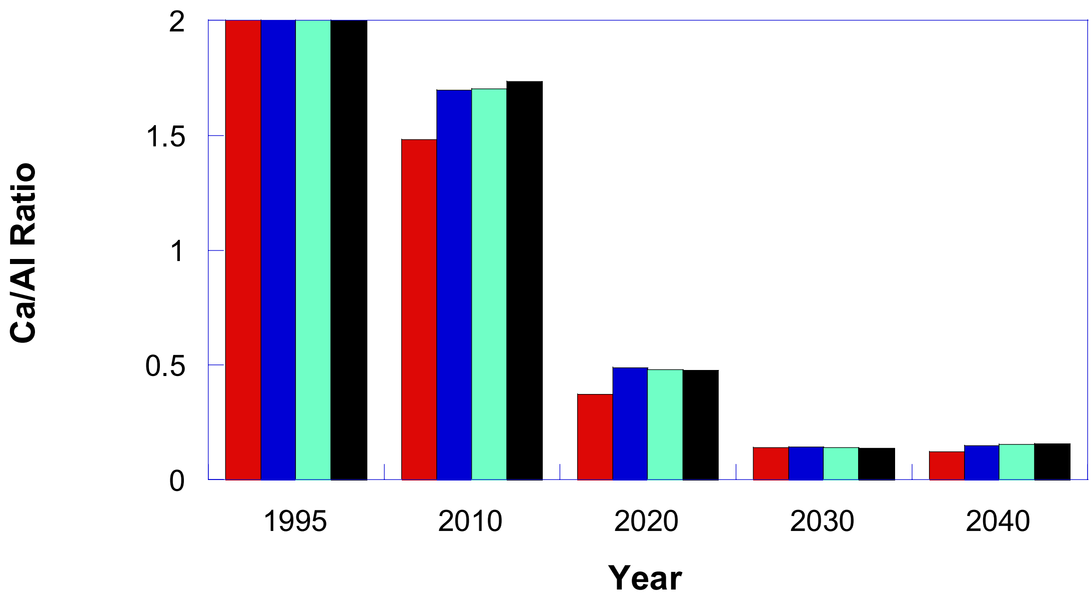
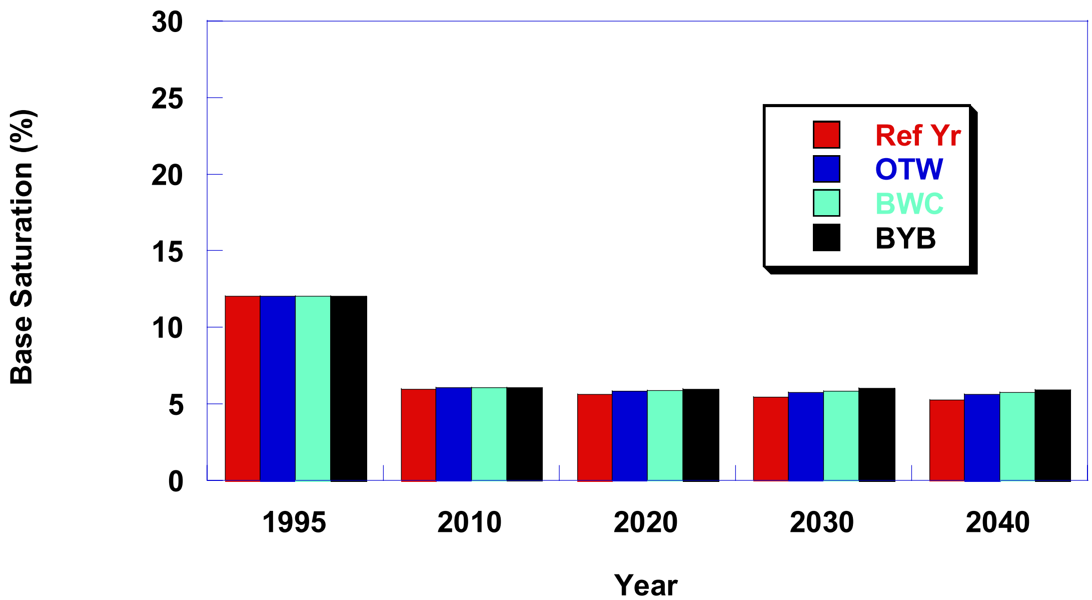


Figure A-13. Simulated response to changes in atmospheric deposition in the rooting zone (A horizon) at the White Oak Run Mixed Hardwood Site.



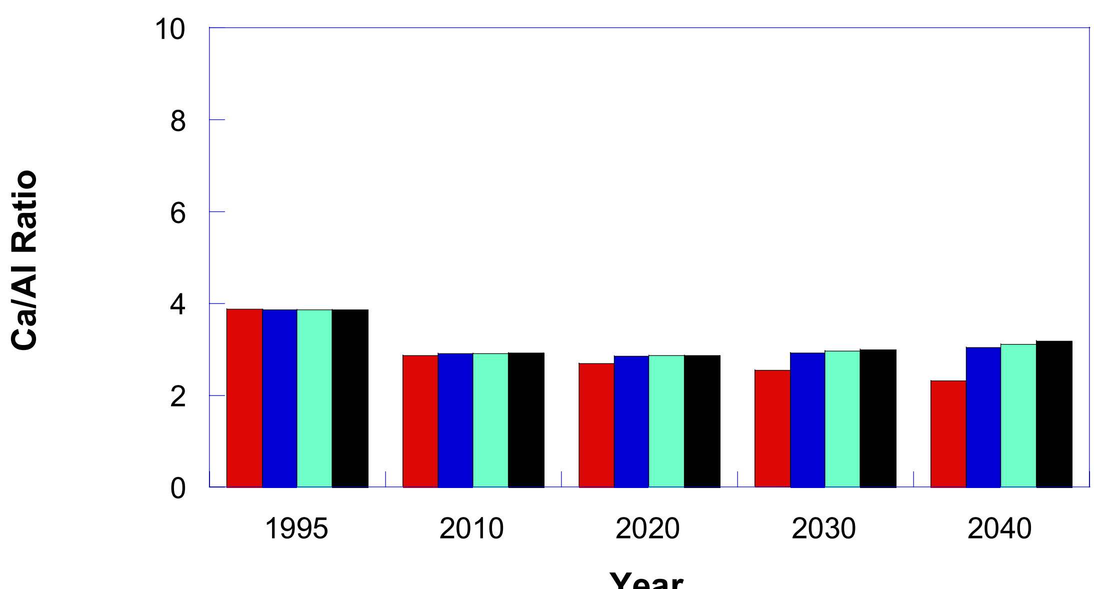


Figure A-14. Simulated response to changes in atmospheric deposition in the rooting zone (A horizon) at the North Fork Dry Run Mixed Hardwood Site.