

## 5.0 DISCUSSION

### 5.1 Watershed Sensitivity to Acidification

Surface waters that are sensitive to acidification from acidic deposition of sulfur or nitrogen typically exhibit a number of characteristics. Such characteristics either predispose the waters to acidification and/or correlate with other parameters that predispose the waters to acidification. Although precise guidelines are not widely accepted, general ranges of parameter values that reflect sensitivity are as follows (Peterson and Sullivan 1998, Sullivan 2000):

Dilute - Waters have low concentrations of all major ions, and therefore specific conductance is low ( $< 25 \mu\text{S}/\text{cm}$ ).

Acid neutralizing capacity - ANC is low. Acidification sensitivity has long been defined as  $\text{ANC} < 200 \mu\text{eq}/\text{L}$ , although more recent research has shown this criterion to be too inclusive (Sullivan 1990). Waters sensitive to chronic acidification generally have  $\text{ANC} < 50 \mu\text{eq}/\text{L}$ , and waters sensitive to episodic acidification generally have  $\text{ANC} < 100 \mu\text{eq}/\text{L}$ .

Base cations - Concentrations are low in non-acidified waters, but increase (often substantially) in response to acidic deposition. The amount of increase is dependent on the acid-sensitivity of the watershed. In relatively pristine areas, the concentration of  $(\text{Ca}^{2+} + \text{Mg}^{2+} + \text{K}^{+} + \text{Na}^{+})$  in sensitive waters will generally be less than about 50 to 100  $\mu\text{eq}/\text{L}$ .

Organic acids - Concentrations are low in waters sensitive to the effects of acidic deposition. Dissolved organic carbon (DOC) imparts substantial pH buffering and causes water to be naturally low in pH and ANC, or even to be acidic ( $\text{ANC} < 0$ ).

pH - pH is low, generally less than 6.0 to 6.5 in acid-sensitive waters. In areas that have received substantial acidic deposition, acidified lakes (as quantified by analyses of diatom remains in lake sediments) are generally those that had pre-industrial pH between 5 and 6.

Acid anions - Sensitive waters generally do not have large contributions of mineral acid anions (e.g., sulfate, nitrate,  $\text{F}^{-}$ ,  $\text{Cl}^{-}$ ) from geological or geothermal sources. In particular, the concentration of sulfate in drainage waters would usually not be substantially higher than could reasonably be attributed to atmospheric inputs, after accounting for probable dry deposition and evapotranspiration.

Physical characteristics - Sensitive waters are usually found at moderate to high elevation, in areas of high relief, with flashy hydrology and minimal contact between drainage waters and soils or geologic material that may contribute weathering products to solution. Sensitive streams are generally low order.

It was recognized relatively early in acidification research that most of the major concentrations of low-ANC surface waters were probably located in areas underlain by bedrock resistant to weathering. Subsequent compilations of available water chemistry data (e.g., Omernik and Powers 1983, Eilers and Selle 1991) refined and expanded this image of sensitive areas in North America. The extensive research programs conducted in Europe, in Canada, and through NAPAP provided additional insight into factors contributing to the sensitivity of surface waters to acidic deposition by revealing the importance of soil composition and hydrologic flowpath, in addition to geology, in delineating sensitive regions.

The geologic composition of a region plays a dominant role in influencing the chemistry and therefore sensitivity of surface waters to the effects of acidic deposition. Bedrock geology formed the basis for a national map of surface water sensitivity (Norton et al. 1982) and has been used in numerous acidification studies of more limited extent (e.g. Bricker and Rice 1989, Dise 1984, Gibson et al. 1983). Analysis of bedrock composition continues to be an important element for assessing sensitivity of surface waters in mountainous regions (e.g. Stauffer 1990, Stauffer and Whittchen 1991, Vertucci and Eilers 1993).

The presence of large populations of acidic and low-ANC lakes and streams in regions such as northern Florida that are underlain by calcareous bedrock illustrate that if the surface waters are isolated from highly weatherable bedrock minerals, acid-base status is not controlled by bedrock geology (Sullivan and Eilers 1994). Thus, both soil and bedrock composition may exert strong influence on surface water acid-base chemistry, and therefore are important factors to be considered in defining acid-sensitive regions.

The third principal factor now recognized as critical in contributing to the sensitivity of aquatic resources is watershed hydrology. The movement of water through the soils, into a stream, and the interchange between drainage water and the soils and sediments regulate the type and degree of watershed response to acidic inputs. Streams in the same physiographic setting can have radically different sensitivities to acidic deposition depending on the relative contributions of near-surface drainage water and deeper, more highly buffered groundwater (Eilers et al. 1983, Chen et al. 1984, Driscoll et al. 1991). Natural hydrologic events also

radically alter sensitivity to acidification by bypassing normal neutralization processes during high flow periods or changing flowpaths during extended droughts (Webster et al. 1990). The importance of hydrologic factors in influencing the acid-base chemistry of surface waters across the United States was reinforced by Newell (1993), who identified hydrology as a key component associated with changes in the acid-base chemistry of lakes included in EPA's Long Term Monitoring Program.

Watersheds differ in their sensitivity to continued atmospheric loading of sulfur or nitrogen and also to changes in those loading rates. Watersheds that exhibit substantial sulfur adsorption on soils will often show reduced sensitivity to future changes in sulfur deposition. In addition, such watersheds may show continued acidification under dramatically reduced sulfur deposition, in part because of a gradual decrease in sulfur adsorption over time. Watersheds also differ with respect to the extent to which nitrogen deposition is retained within soils and biota versus leached to streamwater as nitrate. Watersheds that currently leach significant amounts of nitrogen are expected to be most responsive to future changes in nitrogen deposition.

Nitrogen dynamics in the SAMI region are also strongly influenced by watershed disturbance. For example, forest defoliation by the gypsy moth in Shenandoah National Park and surrounding areas dramatically increased  $\text{NO}_3^-$  export in some watersheds. In addition, wind damage and forest defoliation by the fall cankerworm resulted in small transient increases in  $\text{NO}_3^-$  export at Coweeta Hydrological Laboratory (Webb et al. 1999).

Gosz and Murdoch (1999) concluded that studies in deciduous forests suggest a threshold for nitrogen saturation (the point at which N inputs exceed biological demand and an appreciable percent of the nitrogen input is exported from the watershed as nitrate in streamwater) at nitrogen deposition rates of about 9-12 kg/ha/yr. Based on a review of recent research and monitoring studies in the southern Appalachian region, Gosz and Murdoch concluded that:

- some areas receiving high levels of nitrogen deposition have been subject to nitrogen saturation, whereas other have not;
- nitrogen saturation is not strictly a function of forest maturation;
- natural disturbance can be associated with elevated nitrogen export; and
- there are relationships between climate and nitrogen cycling.

Nitrogen retention in southern Appalachian watersheds is highly variable, ranging from about 1% to 98% (Webb et al. 1999). At five intensively studied watersheds within the region, nitrogen export was near zero ( $< 0.2$  kg/ha/yr) for the three sites that receive less than about 9 kg/ha/yr of total nitrogen deposition: Coweeta, Walker Branch and White Oak Run (pre-insect defoliation). In contrast, the two sites having higher total nitrogen deposition (each greater than about 13 kg/ha/yr at Fernow and Noland Divide) exhibited elevated nitrogen export, about 4 and 16 kg/ha/yr, respectively (Webb et al. 1999).

Forest soils that are sensitive to acidification typically show low base saturation ( $< 15\%$ ). Such soils, especially those that exhibit very low base saturation ( $< 10\%$ ), have low levels of exchangeable base cations in the soil and tend to have low concentrations of calcium and high concentrations of aluminum in soil solution.

## **5.2 Atmospheric Deposition**

Estimated values for sulfur and nitrogen deposition in the 1995 base year varied by about a factor of five across the SAMI domain. Highest values were found at the high elevation sites of Great Smoky Mountains National Park and in the northern and southern section of the Appalachian Plateau physiographic province in West Virginia and Tennessee (Figure 4-5). There were also large spatial variations in the projected changes in both sulfur and nitrogen deposition within the three Emissions Control Strategies (Figures 4-7 to 4-9). Simulated changes in atmospheric deposition, generated by the Atmospheric Deposition Modeling component of SAMI, in response to the three Emissions Control Strategies, varied spatially across the region and among strategies. In some cases, sulfur deposition was simulated to decrease while nitrogen deposition increased. In most cases, both sulfur and nitrogen deposition were simulated to decline in the future. Sulfur deposition was projected to decline at all sites under all strategies. Nitrogen deposition was projected to increase at some sites, especially under the OTW Strategy, although it was projected to decrease at other sites under all strategies.

These spatial differences in starting point deposition, and in projections of change in deposition both across space and across strategies, contributed to the observed heterogeneity in modeled streamwater response. When coupled with the spatial gradients and variability in watershed sensitivity to acidification with either sulfur or nitrogen, the various Emissions Control Strategies resulted in a wide range of projected streamwater responses.

### **5.3 Aquatic Assessment**

Results of the MAGIC modeling effort were complex, with substantial variation in simulated watershed response within and among Emissions Control Strategies. There were multiple reasons for this complexity, reflecting regional differences in current deposition, future changes in deposition, and watershed sensitivity to acidification. Some of the major complicating factors are discussed below.

Base cation concentration and ANC of streams vary spatially, as do several factors that influence sensitivity to acidification. Within the SAMI region, these variations appear to be partly related to geology, partly related to atmospheric deposition and partly related to physiography. All sites were simulated to exhibit reduced soil base saturation and most sites showed reduced base cation leaching. Thus, the model output suggested continued soil acidification, but a decreased rate of soil acidification, under all strategies.

Some watersheds currently leach significant amounts of nitrate from soils to streamwater and were therefore simulated to be responsive to future changes in nitrogen deposition. These watersheds were scattered throughout the SAMI domain, but were mostly found in West Virginia and near the North Carolina-Tennessee border. The sites most responsive to future changes in nitrogen deposition can be seen as those sites that exhibited the largest decreases in streamwater nitrate concentration in the BYB Strategy (Figure 4-11). Although precise cutoff values are subjective, the sites that showed decreases in streamwater nitrate concentrations that were greater than about 5  $\mu\text{eq/L}$  in the BYB Strategy can be considered sensitive, in general terms, to modest changes in nitrogen deposition. Other watersheds currently retain essentially all nitrogen deposition and would not be expected to respond to modest changes in future nitrogen deposition.

Some watersheds currently leach high concentrations of sulfate, and were simulated to exhibit decreased streamwater sulfate concentration and increased future ANC under reduced sulfur deposition. Other watersheds currently adsorb most deposited sulfur and are expected to be less responsive to reduced sulfur deposition. To further complicate the story, the amount of sulfur adsorbed at many of the modeling sites was projected to decrease in the future under virtually any continued sulfur deposition loading. This caused a situation whereby many sites were projected to continue to acidify in response to the increased streamwater sulfate concentration even while sulfur deposition is being reduced. Such streams were located mainly in the southern portion of the SAMI region and in the Blue Ridge physiographic province in the

north. In some cases, the model projected continued acidification of streamwater while both sulfur deposition and sulfate and nitrate concentrations in streamwater declined. Such a situation was caused by projected large declines in base cation concentrations at these sites. The model output suggested that base cation depletion at these sites was so great as to counteract the benefits that might otherwise accrue from decreased streamwater concentrations of sulfate and nitrate. Because changes in base cation deposition in the Emissions Control Strategies were small, this response can largely be attributed to base cation depletion from soils.

These complications interact spatially and temporally within the SAMI domain to yield significant variability in model responses to the Emissions Control Strategies. Such variations must be considered in order to understand the results of this modeling exercise.

The average projected change in ANC from 1995 to 2040 is given in Table 5-1 for each of the ANC classes in each of the Emissions Control Strategies. Under most strategies, the average ANC of most ANC classes decreased from 1995 to 2040, especially for streams that had ANC > 20 in 1995. The exceptions to this pattern were the acidic streams under the BWC and BYB strategies and the streams in the ANC 0 to 20  $\mu\text{eq/L}$  class in the BYB strategy. Other than the more acidic streams under the more aggressive Emissions Control Strategies, most streams showed projected continued acidification to 2040.

Table 5-1. Projected change in class-average ANC for upper stream reaches between the 1995 reference year and 2040.			
ANC Class	Average Projected Change in ANC ( $\mu\text{eq/L}$ )		
	OTW	BWC	BYB
$\leq 0$	-5.0	0.5	3.9
0-20	-7.9	-3.1	4.8
20-50	-5.3	-3.1	-3.5
50-150	-5.7	-3.9	-2.4

The largest differences in projected change in the percentage of modeled streams in the various ANC classes from 1995 to 2040 occurred for the acidic class (Figure 5-1). The scenario of continued deposition at 1995 levels resulted in more than a doubling of the percent of acidic modeled streams, from 18% in 1995 to 41% in 2040. Each of the Emissions Control Strategies resulted in smaller estimates of percent of modeled streams projected to be acidic in 2040, from 25% in OTW to 15% in BYB. These results suggested that acid-base chemistry would generally deteriorate from 1995 conditions under OTW, but to improve slightly under BYB. Continued improvement was projected to occur beyond 2040, even if deposition was held constant

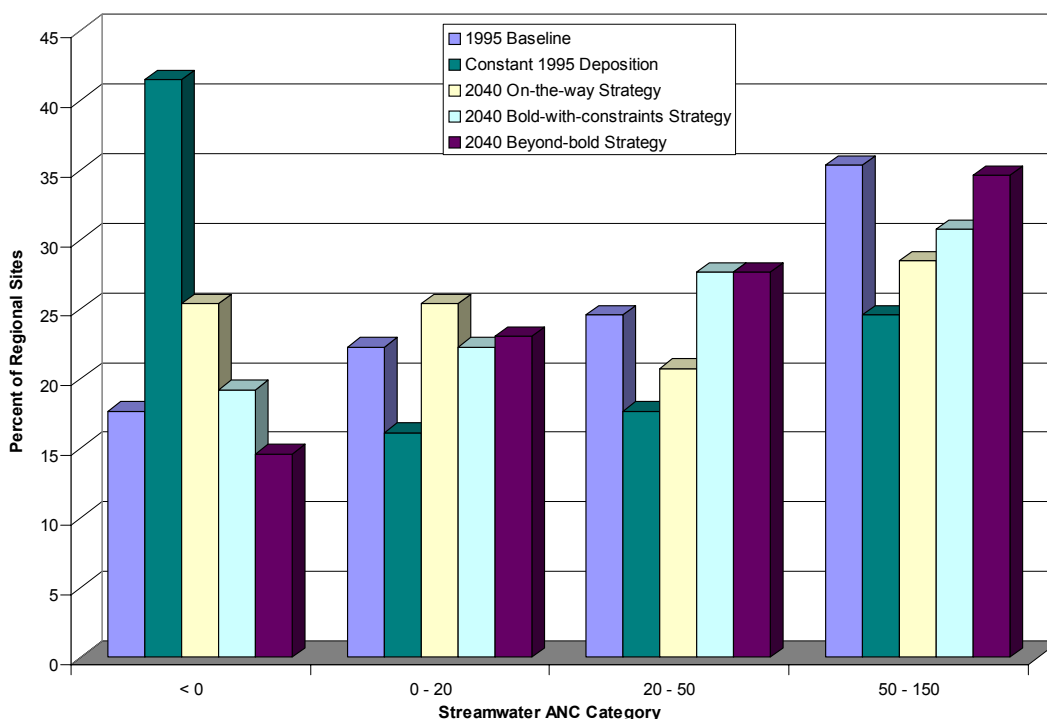


Figure 5-1. Projected percent of regional modeling sites in each ANC class in 2040, based on the scenario of constant 1995 deposition and each of the strategies. Also shown are baseline values for streamwater ANC in 1995.

thereafter. Thus, the model projected that benefits would continue to accrue beyond 2040 from emissions reductions enacted prior to 2040.

The projected differences in streamwater chemistry were small among the three strategies. When these estimates were extrapolated to the population of streams within the SAMI domain, based on the NSS statistical frame, the differences became even smaller. For example, an estimated 7% of the NSS upper node stream population within the SAMI region was acidic in 1995, and this percentage increased to 10% in OTW, but remained constant at 7% in BWC and BYB (Figure 4-20). The lower node streams changed from an estimated 3% acidic in 1995 to 3, 2, and 1% acidic in 2040 in the OTW, BWC, and BYB Strategies, respectively. These extrapolations to the NSS statistical frame help to put the modeling results into the context of the universe of streams that exist within the region. This is important because about 60% of the streams in the SAMI region have ANC > 150  $\mu\text{eq/L}$ . These high-ANC streams are not expected to be acid-sensitive and were not modeled for this assessment. Nevertheless, they constitute the majority of the streams within the region.

### 5.3.1 Intraregional Variability

The SAMI region is highly diverse with respect to its geologic and edaphic (soils) characteristics. Elevation, climate, and vegetation are similarly variable in response to both north-south and elevational gradients in temperature, precipitation, and atmospheric deposition. Some of these differences are reflected in the separation into physiographic provinces (Fenneman 1938).

The geology of the southern Blue Ridge is diverse and complex. To the east, the province is dominated by early Precambrian metamorphic and granitic rocks (Hatcher 1988) and in the west by metamorphosed sedimentary rock, primarily sandstone, with some limestone (King et al. 1968, Elwood et al. 1991). The sandstones cover about 60% of Great Smoky Mountains National Park, and are characterized as massive and of low porosity, with quartz and potassium feldspars as the dominant minerals (Elwood et al. 1991).

In the northern portion of the region, the Blue Ridge Province comprises a narrow (10 to 25 km wide) band of steep-sided mountains that consist of a single major ridgeline, with smaller discontinuous ridges that extend to the east and west. In southern Virginia, the province expands to include a mountainous plateau area that ranges up to 40 km in width. To the southwest, the mountains become closely spaced ridges of greater height and width that end in a complex ridge system that covers portions of North and South Carolina, Tennessee, and Georgia (Harris and Tuttle 1983).

The steep, forested slopes of the Blue Ridge are not highly suitable for agriculture, although soils in the hollows and on gentler slopes were sufficiently deep and fertile to support farms (Gathright 1976). The number of farms has declined in recent decades and much of the Blue Ridge is now covered by second-growth forest (Bryce et al. 1999).

The Ridge and Valley Province represents the eastern margin of the Paleozoic interior sea. Selective erosion is conspicuous throughout this region. Limestones and shales were easily eroded, and siliceous sandstone and conglomerate were more resistant. The combination of folding and selective erosion resulted in a great number of ridges separated by valleys filled with erosional material (Fenneman 1938). The lateral compression that formed the folds also caused thrust faults, especially along the eastern boundary. Since uplift, the entire surface has been eroded, resulting in a marked evenness of crest height.

The limestones, shales, and sandstones that form the Ridge and Valley have been folded and eroded to form steep, parallel ridges of resistant sandstone and rounded shale knobs alternating

with limestone and shale valleys. The ridges are covered with thin, infertile soils and are forested. The valleys experience intensive agriculture and also expanding urban and industrial land use (Bryce et al. 1999).

The geology of the Appalachian Plateau is similar to that of the Valley and Ridge, although the strata are generally flat and undeformed in most places, rather than folded. The plateau is highly dissected, however. Arable limestone areas are not widely distributed, but layers of soft bituminous coal are common. Coal mining and silviculture are the major land uses (Bryce et al. 1999).

The Appalachian Plateau has experienced repeated uplift, but the mountains have been removed by erosion. The province is heavily dissected, and the extent and style of dissection provide the basis for classification into different sections (Fenneman 1938). The edges of the plateau are formed by relatively strong rock strata, but the central syncline also contains a large proportion of shale along with the more resistant sandstone. To the southwest of West Virginia, the rocks are mainly sandstones and conglomerates of the Pottsville series. The beds thicken toward the south until, in Tennessee, a single stratum (Walden sandstone) constitutes most of the mass of the plateau (Fenneman 1938).

The plateau is called Allegheny in the north and Cumberland in the south, although the boundary is somewhat arbitrary. In a general way, however, the rocks to the south are more resistant and less dissected (Fenneman 1938).

Differences in ANC among streams in the southeastern United States appear to be related primarily to differences in the capacity of watersheds to mobilize base cations through weathering and cation exchange and to retain sulfate by adsorption on soil particles (Elwood 1991). Nitrogen retention within the watershed is also important because it is substantially less than 100% in some areas. To a significant extent, these processes vary between physiographic provinces.

The NSS did not sample any acidic streams in the southern Blue Ridge, and estimated that less than 1% of the stream reaches in the southern Blue Ridge were acidic. The results of a study by Winger et al. (1987), however, demonstrated that acidic first-order streams did exist in the region (3% of sampled streams were acidic), although the study was not statistically-based. In addition, the two acidic streams sampled by Winger et al. (1987) drained watersheds that contain the Anakeesta formation, a pyritic phyllite which is a potential geologic source of sulfate in streamwater (Huckabee et al. 1975, Elwood et al. 1991).

Sulfur retention in watersheds in the SAMI region is highly variable, ranging from near zero to well over 75% (Rochelle and Church 1987; Kaufmann et al. 1988; Church et al. 1989, 1992). Similarly, MAGIC model projections suggested considerable variability in the extent to which sulfur retention will change under continued sulfur deposition. Some watersheds were projected to come into steady state (sulfur input  $\approx$  sulfur output) only after more than 150 yr of sulfur deposition at 1984 levels (Church et al. 1992). Soils of the mid-Appalachian Region of the DDRP, which includes the northern portion of the SAMI region, were projected to reach sulfur steady state faster than soils of the southern Blue Ridge Region (35 yrs vs 61 yrs) and also to reach much higher sulfate concentrations in streams (214  $\mu\text{eq/L}$  vs 120  $\mu\text{eq/L}$ ) once steady state is attained (Church et al. 1992).

In the mid-Atlantic Highlands, the net annual retention of atmospherically-deposited sulfur varies among physiographic provinces, and these differences are probably due in part to differences in soils and geology, and in part due to different histories of sulfur deposition loading. Median net annual sulfur retention was estimated to be 11% in the Appalachian Plateau, 35% in the combined Blue Ridge and Valley and Ridge Provinces, and 60% in the Piedmont Province (Herlihy et al. 1993).

Cosby et al. (1991) estimated that the expected average steady state sulfate concentration in VTSSS streams would be about 200  $\mu\text{eq/L}$ . This estimate was based on wet sulfur deposition approximately equal to 8 kg/ha/yr, assumed dry deposition equal to 50% of wet deposition, and an estimated annual runoff of 36 cm/yr. Cosby et al. (1991) concluded that VTSSS streams were retaining, on average, two-thirds of the total deposited sulfur. The sulfate concentrations in VTSSS streams tended to be somewhat higher in the Ridge and Valley (median 92  $\mu\text{eq/L}$ ) than in the Blue Ridge (median 61  $\mu\text{eq/L}$ ). These differences were attributed to the higher sulfur retention capacity of Ultisol soils, which are common in the Blue Ridge, as compared with the Inceptisols that are more common in the Ridge and Valley Province. Potentially higher sulfur deposition in the Ridge and Valley, due to its close proximity to deposition sources to the west, was also suggested as being an important contributor to the observed spatial patterns in streamwater sulfate concentration (Cosby et al. 1991).

Watersheds in the southern Blue Ridge retain a significant fraction ( $\sim$  70% to 80%) of atmospherically-deposited sulfur (Rochelle et al. 1987, Rochelle and Church 1987). The mobility of sulfate in some southern Blue Ridge watersheds has increased, however (c.f., Eshleman and Kaufmann 1988, Elwood et al. 1991). Median sulfate concentration in

streamwater is about twice as high in the northern portion of the Blue Ridge Province, as compared with the southern portion (Elwood 1991).

MAGIC modeling results illustrated large variations in projected change in water chemistry related to site location. These were most strongly related to north-south gradients and differences among physiographic provinces. For example, modeling sites in the northern portion of the SAMI region (Virginia and West Virginia) showed a broad range of projected change in ANC at the 91 modeling sites in this area (Figure 5-2). Some sites showed projected ANC increases of more than 20  $\mu\text{eq/L}$  in response to the OTW strategy. Other sites showed projected decreases in ANC of comparable magnitude. In contrast, the 39 sites in the southern portion of the SAMI region all showed projected decreases in streamwater ANC in response to the OTW Strategy. Differences between north and south were less pronounced for the BYB Strategy, but projected changes in ANC tended to be more commonly positive and of larger magnitude in the north (Figure 5-2).

As expected, the sites that exhibited the highest concentrations of streamwater nitrate in 1995 showed the largest projected changes in streamwater nitrate in response to the strategies (Figure 5-3). Projected changes were generally largest for a given baseline nitrate concentration in the Appalachian Plateau province, and generally smallest in the Blue Ridge. Sizeable (e.g., > 5  $\mu\text{eq/L}$ ) changes in projected streamwater nitrate concentration were generally restricted to the Appalachian Plateau in the OTW Strategy, but to a lesser extent in the BYB Strategy. These are the sites that are most responsive to changes in nitrogen deposition.

The projected percentages of NSS downstream reach ends (lower nodes) having ANC in the year 2040 that was acidic or below 20  $\mu\text{eq/L}$  are listed in Table 5-2, stratified by north and south and by physiographic province. Marked geographic differences were observed, with almost all of the projected acidic and low-ANC lower node streams in 2040 occurring in the northern Appalachian Plateau, regardless of strategy. The projected percentages of acidic and low-ANC upstream reach ends (upper nodes) were somewhat more evenly distributed than the lower nodes, but still mostly confined to the northern section of the SAMI region. Again, the northern Appalachian Plateau was the predominant location of projected acidic and low-ANC streams in 2040. Over 25% of the estimated 4,890 upper node northern Appalachian Plateau stream reaches were projected under the OTW Strategy to be acidic and about 43% were projected to have  $\text{ANC} \leq 20 \mu\text{eq/L}$  in 2040 (Table 5-3).

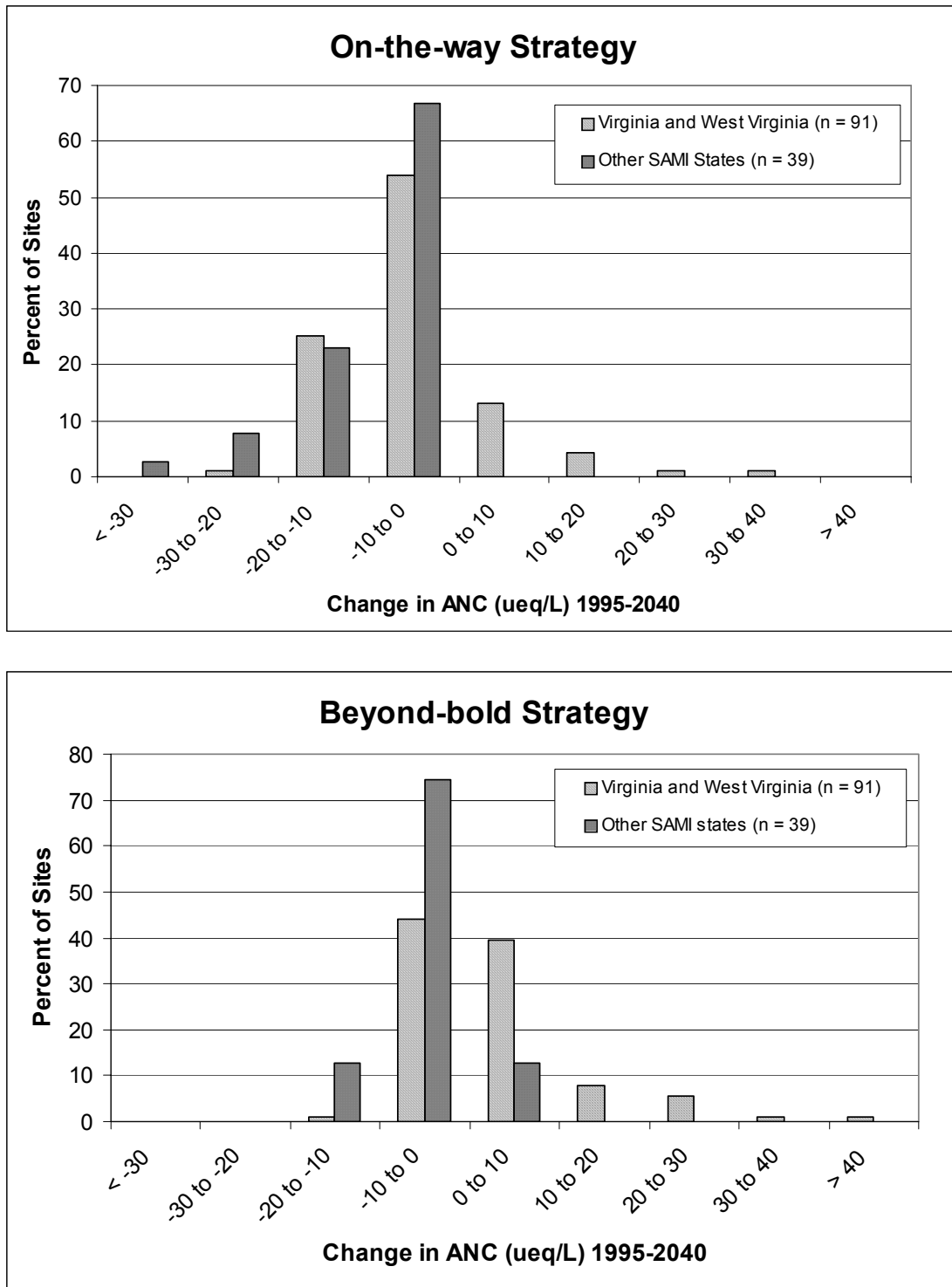


Figure 5-2. Model projection of change in streamwater ANC from 1995 to 2040 under the OTW and BYB Emissions Control Strategies in the northern section of the SAMI region (Virginia and West Virginia) versus the southern portion (all other SAMI states).

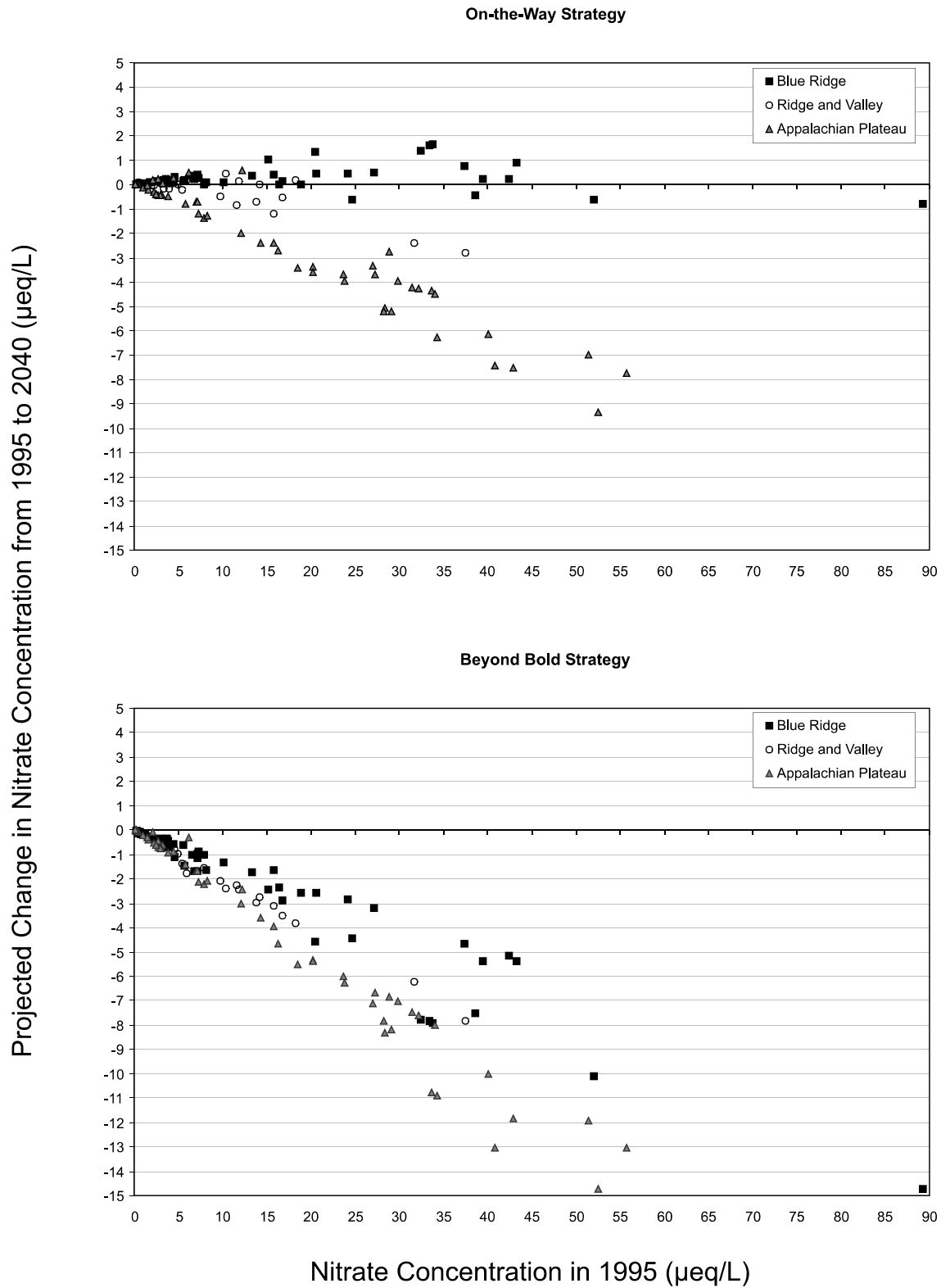


Figure 5-3. Projected change in streamwater nitrate concentrations from 1995 to 2040 for the OTW and BYB strategies versus the nitrate concentrations in streamwater in 1995. Sites are coded by physiographic province.

Table 5-2. Predicted percent of reach ends with ANC below 0 and 20 $\mu\text{eq/L}$ in the year 1995.						
Class*	Downstream Reach Ends			Upstream Reach Ends		
	N	ANC $\leq 0$	ANC $\leq 20$	N	ANC $\leq 0$	ANC $\leq 20$
Northern Blue Ridge	2,033	0	0	2,510	0	12.7
Northern Ridge & Valley	6,022	0.8	0.8	5,068	6.3	7.2
Northern Appalachian Plateau	4,890	12.5	25.9	5,050	19.1	28.3
Southern Blue Ridge	4,289	0	0.6	4,289	0	0.6
Southern Ridge & Valley	1,142	0	0	1,566	0	0
All SAMI	19,185	3.4	7.0	19,026	6.7	11.2

N is the estimated total number of stream reaches in the population.  
\* NSS sample size in the southern Appalachian Plateau is too small to make reliable population estimates so they are not presented.

Table 5-3. Predicted percent of downstream and upstream stream reach ends with ANC $\leq 0$ $\mu\text{eq/L}$ and ANC $\leq 20$ $\mu\text{eq/L}$ in the three SAMI future deposition strategies in the year 2040.							
Class*	N	ANC $\leq 0$			ANC $\leq 20$		
		OTW	BWC	BYB	OTW	BWC	BYB
<b>Downstream</b>							
Northern Blue Ridge	2,034	0	0	0	0	0	0
Northern Ridge & Valley	6,022	0.8	0.8	0	0.8	0.8	0.8
Northern Appalachian Plateau	4,890	12.5	5.9	3.0	25.9	25.9	19.7
Southern Blue Ridge	4,289	0	0	0	0.6	0.6	0.6
Southern Ridge & Valley	1,142	0	0	0	0	0	0
All SAMI	19,185	3.4	1.8	1.8	7.0	7.0	5.4
<b>Upstream</b>							
Northern Blue Ridge	2,034	12.7	0	0	12.7	12.7	12.7
Northern Ridge & Valley	6,022	6.3	6.3	6.3	7.2	7.2	7.2
Northern Appalachian Plateau	4,890	25.4	19.1	19.1	42.9	31.1	28.3
Southern Blue Ridge	4,289	0	0	0	0.9	0.6	0.6
Southern Ridge & Valley	1,142	0	0	0	0	0	0
All SAMI	19,026	10.1	6.7	6.7	15.2	12.0	11.2

N is the estimated total number of stream reaches in the population.  
\* NSS sample size in the southern Appalachian Plateau is too small to make reliable population estimates so they are not presented.

Under the OTW Strategy, many of the low-ANC streams in the northern Appalachian Plateau subpopulation were simulated to decline by about 10  $\mu\text{eq/L}$  in ANC from 1995 to 2040. These projected decreases in streamwater ANC were sufficiently large as to result in quite a large increase in the percent of streams in the northern Appalachian Plateau that were projected to be acidic in 2040 as compared with 1995 estimates. The estimated percent of streams that was acidic in 1985 ranged from 3% (lower node) to 6% (upper node) in the northern Appalachian Plateau. Under the OTW Strategy, the percent of acidic lower node streams was projected to increase by 2040 to 12% and acidic upper node streams to increase to 25%, with most of the increases occurring between 1985 and 1995. These are very large increases and this subregion represents most of the additional acidification associated with the OTW Strategy. This acidification can largely be attributed to base cation depletion in watershed soils. Even under the BYB Strategy, the percent of acidic northern Appalachian Plateau upper node streams was very high (19%; Table 5-3). Such changes in ANC would be expected to have significant adverse impacts on brook trout and other aquatic biota within this subpopulation. Over 40% are expected to be either episodically or chronically unsuitable for brook trout in 2040 as a consequence of the projected low ANC values.

Projected changes in ANC values (as opposed to endpoint ANC values) for the NSS subregional populations in response to the three strategies did not show such large geographical variations (Table 5-4). All subpopulations showed a median decline in projected ANC from 1995 to 2040 for the OTW and BWC strategies. Under the BYB Strategy, the northern Blue

Table 5-4. Median change in ANC ( $\mu\text{eq/L}$ ) from 1995 to 2040 in 130 regional model sites in the three SAMI future deposition strategies. Negative numbers indicate a decline in ANC over time.				
Physiographic Province*	Sample Size	Strategy		
		OTW	BWC	BYB
Northern Blue Ridge	16	-4.0	-1.0	+2.9
Northern Ridge & Valley	41	-6.6	-4.7	-0.4
Northern Appalachian Plateau	34	-4.4	-1.4	+3.9
Southern Blue Ridge	33	-8.0	-5.4	-3.2
Southern Appalachian Plateau	6	-15.9	-13.3	-10.4
ALL	130	-6.7	-4.2	-1.3

\* There were no model sites in the southern Ridge & Valley

Ridge and northern Appalachian Plateau showed small ( $< 4 \mu\text{eq/L}$ ) positive median changes in projected ANC, whereas other subpopulations also showed projected declines under the BYB Strategy. Projected future acidification was generally about twice as high or higher in the southern Appalachian Plateau as compared with other subpopulations, but the sample size of modeled streams was small ( $n=6$ ). Projected acidification was consistently greater in the south, however, than in the north (Table 5-4). This observation can largely be attributed to the consistent projections of positive median changes in streamwater sulfate concentration in southern streams (Table 5-5). Whereas sulfate concentrations in streams of the northern Appalachian Plateau typically decreased by median values of  $-31$  to  $-42 \mu\text{eq/L}$  in the strategies, median changes in streamwater sulfate concentration in the south ranged from  $+3$  (southern Blue Ridge in BYB) to  $+15 \mu\text{eq/L}$  (southern Appalachian Plateau in OTW). These differences in streamwater chemistry reflect differences in sulfur adsorption on soils.

Median projected changes in streamwater nitrate concentration were generally near 0, except in the northern Appalachian Plateau. Changes were most pronounced under the BYB Strategy ( $-5.8 \mu\text{eq/L}$ ; Table 5-6).

It is interesting to note that despite the rather large projected declines in streamwater sulfate and nitrate concentrations in the northern Appalachian Plateau, median projected streamwater ANC declined in two of the strategies. This was because the projected changes in base cation concentrations in streamwater in this subregion were generally larger than the projected changes in sulfate plus nitrate concentrations (Tables 5-5 to 5-7).

Table 5-5. Median change in sulfate ( $\mu\text{eq/L}$ ) from 1995 to 2040 in 130 regional model sites in the three SAMI future deposition strategies. Negative numbers indicate a decline in sulfate over time.				
Physiographic Province*	Sample Size	Strategy		
		OTW	BWC	BYB
Northern Blue Ridge	16	+1.8	-2.7	-7.4
Northern Ridge & Valley	41	-0.5	-5.6	-13.8
Northern Appalachian Plateau	34	-31.2	-36.3	-41.6
Southern Blue Ridge	33	+8.8	+5.6	+3.2
Southern Appalachian Plateau	6	+15.2	+11.6	+7.2
ALL	130	+1.9	-4.0	-8.7

\* There were no model sites in the southern Ridge & Valley

Table 5-6. Median change in nitrate (ueq/L) from 1995 to 2040 in 130 regional model sites in the three SAMI future deposition strategies. Negative numbers indicate a decline in nitrate over time.				
Physiographic Province*	Sample Size	Strategy		
		OTW	BWC	BYB
Northern Blue Ridge	16	0	0	-0.1
Northern Ridge & Valley	41	0	-0.4	-0.4
Northern Appalachian Plateau	34	-3.5	-4.9	-5.8
Southern Blue Ridge	33	0.2	-0.5	-1.0
Southern Appalachian Plateau	6	0.1	-0.2	-0.5
ALL	130	0	-0.6	-1.0
* There were no model sites in the southern Ridge & Valley				

Table 5-7. Median change in base cations (ueq/L) from 1995 to 2040 in 130 regional model sites in the three SAMI future deposition strategies. Negative numbers indicate a decline in base cations over time.				
Physiographic Province*	Sample Size	Strategy		
		OTW	BWC	BYB
Northern Blue Ridge	16	-2.2	-3.0	-5.1
Northern Ridge & Valley	41	-6.8	-8.2	-10.3
Northern Appalachian Plateau	34	-33.8	-38.4	-42.8
Southern Blue Ridge	33	+1.0	-0.6	-2.3
Southern Appalachian Plateau	6	-1.2	-1.9	-3.1
ALL	130	-4.6	-6.5	-7.6
* There were no model sites in the southern Ridge & Valley				

### 5.3.2 Biological Effects

Model results in response to the Emissions Control Strategies did not indicate that large numbers of streams would be likely to shift in biological response categories in the future. For example, the population of upper node streams projected to be chronically acidic ( $ANC \leq 0$ ) only increased from 7% in 1995 to 10% in 2040 in the OTW Strategy, and did not change in the other two strategies. In addition, for the lower ANC classes ( $\leq 0$  and  $0-20 \mu\text{eq/L}$ ), the projected average changes in ANC shown in Table 5-1 were generally small ( $< 8 \mu\text{eq/L}$ ). Nevertheless, projected small decreases in streamwater ANC suggest an increased likelihood of adverse effects

on some species of aquatic biota due to chronic and/or episodic acidification. For some streams in the transitional class (ANC 20-50  $\mu\text{eq/L}$ ), the model results suggest increased possibility of episodic acidification to ANC values that could adversely impact some species. For higher ANC streams ( $> 50 \mu\text{eq/L}$ ), it is not clear to what extent the projected small changes in ANC might impact aquatic biota. Such impacts, if they would occur, would probably be small.

Species loss is a highly predictable result of aquatic acidification. Aquatic species (fish and macroinvertebrates) differ in tolerance of acidic conditions. Acidification lowers species counts by eliminating sensitive species first, followed by species of intermediate tolerance, and finally by the loss of even the most acid-tolerant species as acidification progresses. Regionally, this results in a unambiguous relationship between acid-base status of water bodies and the number of aquatic species they host, such that unacidified lakes and streams host more species than acidified lakes and streams.

Bulger et al. (1999) reported an observed linear relationship between fish species richness and minimum streamwater ANC in streams within Shenandoah National Park. In general, there was one fewer fish species with every decrease in minimum recorded ANC of 21  $\mu\text{eq/L}$ . Other factors besides acid-base chemistry may influence this observed relationship. However, the data reported by Bulger et al. (1999) suggest that the small projected decreases in ANC in response to the SAMI strategies would probably not have an appreciable effect on fish species richness for the majority of streams in the region. For the streams projected to change the most in the future, however, fish species richness may very well be affected. For example, one-fourth of the modeling sites showed projected decreases in ANC of more than 10  $\mu\text{eq/L}$  in the OTW strategy and more than 25  $\mu\text{eq/L}$  under continued constant deposition at 1995 levels. Many of these most responsive streams are located in Class I areas. Over half of the modeled streams in all Class I areas except Shining Rock Wilderness were projected to decline in ANC by more than 10  $\mu\text{eq/L}$  by 2040 under continued constant deposition.

### 5.3.3 Nitrogen Saturation

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 ecosystem compartments (Gunderson 1992).

The nitrogen cycle is extremely complex and controlled by many factors besides atmospheric emissions and deposition. 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<sup>1</sup> 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. (1989a) 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

---

<sup>1</sup> 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 (1989a) 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. 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.

A survey of nitrogen outputs from 65 forested plots and catchments throughout Europe was conducted by Dise and Wright (1995). Below 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).

Similarly, a threshold of nitrate leaching at deposition levels above about 10 kg N/ha/yr was found at experimental sites throughout Europe by Tietema and Beier (1995); no significant leaching was observed at deposition levels below this threshold.

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. 1991, Joslin et al. 1987). Thus, nitrate leaching is partly under biological control and typically shows pronounced seasonal variability.

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. 1991a,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. 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 do not progress toward nitrogen-saturation as rapidly or as far as coniferous stands because they tend to have higher rates of nitrogen uptake and requirement. Decreased growth and increased mortality associated with excess nitrogen have more commonly been observed in coniferous stands (Aber et al. 1998). Indeed, most of the lower elevation deciduous stands, including >90% of all forests in the U.S., are nitrogen-deficient and are therefore likely to benefit (i.e., grow faster) 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, 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.

The MAGIC model does not currently contain a mechanism to determine at what point forested ecosystems that receive elevated nitrogen deposition will begin to leach nitrate to streamwaters, although this is a topic of current research. Available evidence suggests that nitrate leaching is mainly limited to forested ecosystems that receive more than about 10 kg N/ha/yr deposition. This is approximately equal to the average nitrogen deposition estimated for the MAGIC modeling sites for 1995 and under the OTW Emissions Control Strategy for the year 2040 (Appendix G, Table G-5). This may account for the observation that about half of the modeled streams had nitrate concentration > 2 µeq/L in 1995. Under the BYB Strategy, the average nitrogen deposition was estimated to decrease to 8.8 kg N/ha/yr by 2040 (Appendix G, Table G-7). It is not likely that the relatively small nitrogen deposition changes estimated for these Emissions Control Strategies will have an appreciable effect on the proportion of the nitrogen input to forested watersheds in the SA that will leach as nitrate to streamwaters. Although the MAGIC model estimated that streamwater nitrate concentrations might decrease from 1995 to 2040 by up to 15 µeq/L (Figure 5-3), there is no evidence to suggest that the percent of incoming nitrogen that is leached to streamwaters will change appreciably.

#### 5.3.4 Intra-annual Variability

Model projections prepared for this assessment were based on chronic, annual average streamwater chemistry. Superimposed upon that chemistry are seasonal and episodic variability with both natural and anthropogenic (human-caused) components (c.f., Wigington et al. 1993). The magnitude and principal causes of episodic acidification are variable from watershed to watershed within the SAMI domain. For example, O'Brien et al. (1993) compared episodic variability of acid-base chemistry among five mid-Atlantic watersheds and examined the relationships to bedrock type and physiography. The study watersheds were located in three physiographic provinces: Coastal Plain, Valley and Ridge, and Blue Ridge. In general, greater losses of streamwater ANC during episodes were observed in watersheds underlain by reactive bedrock (carbonate, metabasalt), and smaller losses of ANC were observed in watersheds underlain by quartzites and unconsolidated quartz sands and cobbles. Little change in sulfate concentration and ANC was found during episodes at the site that was chronically acidic (Mill Run, VA). At the other four sites, ANC decreased in concert with episodic increases in sulfate concentration, but the slope of the relationship was variable. Nitrate was not the primary acidic anion during stormflow in any of the streams, in contrast to observations commonly found in the Northeast. This finding was attributed by O'Brien et al. (1993) to the fact that mid-Atlantic watersheds do not experience a long dormant period under snowpack and so nitrate is taken up by terrestrial biota, and therefore is retained in the study watersheds year-round.

Published episodic stream chemistry data are available from a number of streams in the northern portion of, and just to the north of, the SAMI region (Wigington et al. 1990). Streams in this area that become episodically acidic have tended to be those that had pre-episode spring baseflow ANC less than about 30  $\mu\text{eq/L}$  (c.f., Figure 3 in Herlihy et al. 1993).