
**Evaluation of Effects of Acidic Deposition to Terrestrial
Ecosystems in Class I Areas of the
Southern Appalachians**

**A Report to the
Southern Appalachian Mountains Initiative
(SAMI)**

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Contents

1. Introduction	1
2. Southern Appalachian Spruce-Fir Forests	3
2.1 Natural history and ecological characteristics	3
2.2 Environmental characteristics.....	4
2.3 Stressors and factors that contribute to acidic deposition impacts	4
3. Trends and Condition of Spruce-Fir Forests	8
3.1 Tree growth.....	8
3.2 Crown condition	8
3.3 Mortality	9
4. Soils and Nutrient Cycling	11
4.1 Susceptibility of soils to strong acid inputs	11
4.1.1 Soil characteristics	12
4.1.2 Relationship of soil to soil solution chemistry	15
4.1.3 Role of atmospheric deposition in changing soil water chemistry	16
4.1.4 Uncertainties and implications to policy decisions	17
4.2 Nutrient cycling in high elevation spruce-fir ecosystems	18
4.2.1 Atmospheric deposition regime	18
4.2.2 Nutrient pools and fluxes.....	19
4.2.3 Stand dynamics and N saturation	20
4.3 Forest Nutrition	21
4.3.1 Causes of nutrient deficiencies and uncertainties.....	21
4.3.2 Nutrient Concentrations	24
4.3.3 Response to nutrient amendments.....	26
4.4 Impacts at other Southern Appalachian non-Class I Areas	26
4.5 Links between terrestrial and aquatic impacts.....	28
4.6 Summary and Conclusions	29
5. Physiological Diagnosis of the Condition and Causes of Declining Health of Red Spruce and Fraser Fir in the Southern Appalachians	31
5.1 Mature Tree Growth Patterns.....	31
5.2 Changes in Wood Chemistry.....	32
5.3 Physiological Measurements of Red Spruce Saplings.....	32
5.4 Controlled Exposure Studies	33
5.5 Increased Sensitivity to Winter Damage	34
5.6 Fertilization Studies	34
5.7 The Role of Nitrogen	35
5.9 Physiological Measurements on Fraser Fir	35
5.10 Summary.....	36

6. Approaches to an Integrated Assessment of Effects of Acidic Deposition on Terrestrial Ecosystems in the Southern Appalachians	38
6.1 Recommendation	38
6.2 Proposed Collaborators	39
6.3 Description of the Models	39
6.3.1 The MAGIC Models	39
6.3.2 The Nutrient Cycling Model (NuCM)	42
6.3.3 The PnET Models	43
 Literature Cited	 46
 Glossary of Soil Terms	 59

Chapter 1

Introduction

The Southern Appalachian Mountain Initiative (SAMI) seeks to develop a collaborative approach to air quality management in the eight state SAMI region. As part of this process, SAMI has requested Technical Reports that summarize current scientific understanding of the effects of air pollution on selected resources with particular emphasis on Class I Areas, parks and other wilderness areas. This information will be used by SAMI to answer three questions:

1. What is the current status of resources in the Southern Appalachians and what is the contribution of anthropogenic air emissions to current status?
2. What is the relationship between air pollutant exposure and resource response?
3. What changes in resource status are projected to occur as a result of changes in exposure due to implementation of the 1990 Clean Air Act Amendments or other emissions management options being considered by the SAMI Policy Committee?

This report addresses the effects of acidic deposition on terrestrial ecosystems. Our specific objectives are to:

1. Evaluate existing published scientific and technical information on the current status, recent changes, and causes of these changes in terrestrial ecosystems in the Southern Appalachian Mountains.
2. Evaluate available tools and methodologies that are appropriate for use in conducting an assessment of terrestrial effects of acidic deposition under alternative emission management options.

The primary focus will be on high elevation spruce-fir forests. There exists a large body of scientific information for this ecosystem concerning the effects of acidic deposition as a result of a large research program supported by the Forest Response Program, a component of the National Acid Precipitation Assessment Program. Additionally, the Electric Power Research Institute (EPRI) funded research on acidic deposition effects on forest nutrient cycles which included several research sites within the SAMI region, including one that was collocated with research funded by the Forest Response Program. The comprehensive nature of these research programs provides an excellent bases to evaluate acidic deposition effects. Although these ecosystems have a very limited geographic distribution, they represent an important Air Quality Related Value because of their uniqueness (high occurrence of endemic and rare species, high aesthetic quality, etc), because of enhanced susceptibility to adverse impacts from acidic deposition, and their role in regulating chemical outputs to streams.

Southern Appalachian spruce-fir forests are found at elevations above 1370 meters in a disjunct

pattern throughout southwestern Virginia, eastern Tennessee, and western North Carolina. Red spruce (*Picea rubens* Sarg.) occurs on the high peaks in West Virginia, and small stands of red spruce are found in the two Class I Areas in West Virginia (Dolly Sods and Otter Creek). Small remnant stands are located in Shenandoah National Park. Very little research has been conducted on acidic deposition effects on terrestrial ecosystems in the central Appalachians; much of the focus in this region has been on aquatic ecosystem. Results from a 5 year study of an experimentally acidified watershed in Fernow Experimental Forest (West Virginia) are just now beginning to be published and should be valuable to SAMI in the future.

In the Southern Appalachians, research on effects of acidic deposition on spruce-fir forests was conducted in Great Smoky Mountains National Park (Clingmans Dome vicinity), Whitetop Mountain, VA, and Mt. Mitchell, NC. Between 1985 and 1989 surveys were conducted to determine the condition of red spruce stands through evaluation of crown condition, tree growth, and mortality on a network of permanent vegetation plots. There was an intensive effort to characterize the deposition of cloud water at all three sites during this same time. Research at Clingmans Dome and Whitetop Mountain also included evaluation of acidic deposition effects on soil chemistry, nutrient cycles, and physiological processes of red spruce. Physiological studies at Clingmans Dome utilized gradients in acidic deposition to evaluate plant nutrition and physiological status of mature red spruce or saplings. At Whitetop Mountain, native red spruce seedlings were used in controlled exposure studies which excluded cloud water, precipitation, and ozone.

In Chapter 2, we present an overview of spruce-fir forests and important stressors other than acidic deposition that have recently impacted these forests. Most of this information is specific to Great Smoky Mountains National Park which contains 74% of the spruce-fir forests in the Southern Appalachians. The condition of the forest is discussed in Chapter 3 including an update of crown condition of red spruce in 1995 for the Smokies. The considerable amount of research on how acidic deposition influences soil processes and nutrient cycling in spruce-fir forests is summarized in Chapter 4. This is followed by an evaluation of the physiological response of red spruce to acidic deposition and the relationship of these responses to soil processes. In Chapter 6 we describe three models that can be used to assess future changes in spruce-fir ecosystems to changes in acidic deposition.

Chapter 2

Southern Appalachian Spruce-Fir Forests

2.1 Natural history and ecological characteristics

The Southern Appalachian spruce-fir forests are a series of relict island-like stands on high elevation mountains in southwest Virginia, western North Carolina, and eastern Tennessee. Whittaker (1956) suggested that this ecosystem was a refuge sanctuary during the xerothermic period and that spruce-fir was probably restricted to elevations above 1700-1770 m. As the climate cooled down during the late Holocene (approximately 2000 YR BP), the upward retreat of spruce and fir stopped and populations were able to expand downslope, if populations still existed on a specific peaks (Delcourt and Delcourt, 1984). This theory may explain the absence of spruce-fir populations on some Southern Appalachian mountains that seemingly have high enough elevation.

While the southern spruce-fir system is rich in endemic and rare species (White, 1984), there are few dominant tree species besides red spruce, Fraser fir (*Abies fraseri* Pursh. Poir), and yellow birch (*Betula lutea* Mich. f.). White and Renfro (1984) list 46 vascular plant species as characteristic of this forest type with 12 of those species found only in Southern Appalachian spruce-fir and eight of those species are endemic to the southern high peaks.

Whittaker (1956) found that within undisturbed southern spruce-fir forests, species distribution tended to follow an elevation gradient: spruce forest from 1370 to 1675 m, spruce-fir from 1675 to 1890 m, and fir forest from 1890 m and higher. On lower or drier slopes, red spruce shared dominance with yellow birch. At the higher elevations, Fraser fir was often the sole dominant, and mountain-ash often was the only other canopy tree present. Along with the change in composition from low to high elevations and increasing exposure to winds, there was a change in forest structure: average tree size and height decreased as elevation and exposure increased.

White and others (1985a) suggest that, while lightning fire and debris avalanches occur, spatially small canopy gaps ($\leq 200 \text{ m}^2$) dominated the natural disturbance regime of old-growth Southern Appalachian spruce-fir forests. Tree replacement patterns were unpredictable from gap size and age, but the gap event was important for species interactions. Larger disturbance patches have been measured in Northern Appalachian spruce-fir (Reiners and Lang, 1979; Foster and Reiners, 1983).

Most research done on stand dynamics of spruce-fir forests has been carried out on old-growth spruce-fir or almost pure fir stands. However, much of the Southern Appalachian spruce-fir areas involve young second-growth forest dynamics or stands severely impacted by the balsam woolly adelgid and small canopy gaps may not be the dominant form of disturbance. Nicholas and Zedaker (1989) suggested that stand dynamics in second growth or disturbed forests may occur at a more rapid pace and larger scale than in old growth stands.

2.2 Environmental characteristics

Annual precipitation in the Great Smoky Mountains National Park averaged 226 cm at 1524 m elevation for the years 1946-1951 (Shanks, 1954; Stephens, 1969). Recent data indicate somewhat lower precipitation. Annual totals at 1524 m elevation (Newfound Gap) for 1983-1987 averaged 191 cm (Pauley, 1989). Precipitation generally exceeds evapotranspiration (Shanks, 1954), although some moisture stress occurs in summer and early fall (Pauley, 1989). Mean monthly temperatures (at 1536 m) range from a low in January of -1.6°C to a high in July of 17.3°C . The growing season generally lasts 100 to 150 days (White and Cogbill, 1992).

Often enveloped in clouds, the spruce-fir zone has high humidity, high precipitation, and increased moisture inputs via interception of clouds. During winter, moisture in the form of rime ice has the potential to cause severe damage to trees above the cloud base (White and Cogbill, 1992).

Atmospheric deposition occurs via three main pathways: (i) precipitation or wet deposition, where material is dissolved in rain or snow, (ii) dry deposition, involving direct deposition of gases and particles (aerosols) to any surface, and (iii) cloud water deposition, involving material dissolved in cloud droplets, which is deposited when cloud or fog droplets are intercepted by vegetation including forest canopies. One of the principle findings of the Mountain Cloud Chemistry Program, a component of the Forest Response Program, was that deposition of SO_4^{2-} , H^+ , NH_4^+ , and NO_3^- in cloud water represented a significant input to high elevation forest canopies. Exposure to pollutants and deposition of chemicals significantly increases from below cloud base (above 1500 m) to above cloud base. Cloud water deposition can exceed the flux from wet deposition during the growing season at mountain sites that are frequently exposed to clouds. Cloud interception can be the dominant process for the input of sulfur and nitrogen compounds into montane forests (Mohnen, 1992). At Whitetop Mountain, VA cloud water deposition of SO_4^{2-} was 350% greater than precipitation during the growing season (May through October) during the period 1986-1988. Cloud water nitrate deposition was about 2 times higher than precipitation. This high acid anion deposition is also reflected in the lower pH of cloud water, with cloud water being a full unit lower than precipitation in the Southern Appalachians.

2.3 Stressors and factors that contribute to acidic deposition impacts

Soil Characteristics

Soils of spruce-fir forests typically have thick organic horizons, ranging from 5 to 15 cm (Joslin et al., 1992). Depending on parent material and slope characteristics, these organic layers are underlain by (i) shallow bedrock (usually Histosols), (ii) thick dark A horizons, rich in organic matter and formed in loamy or fine-textured soils (usually Inceptisols), and (iii) ashy gray sands over dark sandy loams (Spodosols) (Kelly and Mays, 1989, Joslin et al., 1992). Regardless of soil classification, a high organic matter content is a nearly universal characteristic (Joslin et al., 1992).

Spruce-fir soils are typically naturally acidic due to a combination of high leaching rates (affecting exchangeable acidity) and large accumulations of humus (affecting titratable acidity). Spruce-fir soils typically lie in the aluminum buffering range where little further acidification is expected (Johnson and Fernandez, 1992). Although the observed exchange capacities for many of these

soils are moderately high, the percent base saturation is usually low, especially in the mineral soil (Joslin et al., 1992).

Land Use History

Anthropogenic disturbances in the Southern Appalachian spruce-fir vary widely by mountain area. Because of the inaccessibility of the high mountain forests, the spruce-fir forest was frequented by few people, including native Americans, before the construction of logging railroads. Some livestock grazing occurred and fires occasionally spread into the spruce-fir forest from lower elevation deciduous forest cleared for farming or that were accidentally set by hunters (Pyle and Schafale, 1988). Overall, pre-logging human disturbances to the spruce-fir were minor compared to the damage caused to the deciduous forests downslope.

When the red spruce forests of the northeast became scarce, logging companies moved south. Substantial logging operations of spruce in the Southern Appalachians began in 1905 on Mount Rogers in Virginia and quickly spread south, with the most inaccessible area, the Great Smoky Mountains, cut last (Pyle and Schafale, 1988). Logging in the rugged Southern Appalachians required large capital outlays for railroad construction and land or timber right purchases. In order to profit, large scale mechanized operations were necessary. Regeneration back to a spruce-fir forest did not always happen. Logging fires often destroyed the advance spruce and fir regeneration and removed much of the soil organic layer, increasing erosion susceptibility. Estimates vary, but it is possible that 50-90% of former southern spruce-fir no longer exists because of failed regeneration (Pyle, 1984). The Smokies were the least logged and Pyle estimates that only 25% of the original forest was converted to hardwood forests after the logging era.

Balsam Woolly Adelgid

The balsam woolly adelgid, *Adelges piceae* Ratz. (Homoptera: Adelgidae), is one of the most significant disturbance factors to high elevation Southern Appalachian spruce-fir forests. The insect is a native of Europe where it feeds on European silver fir, causing only minor damage. The adelgid was first detected in the Southern Appalachians on Mount Mitchell in the Black Mountains in 1957 (Speers, 1958), but is suspected to have arrived in the southern mountains in the 1930s via reforestation experiments. Mature Fraser fir is highly susceptible to adelgid attack, with death occurring in two to nine years. Adelgid infestation had spread throughout the Black Mountains a few years after initial detection. Severe mortality of Fraser fir and rapid spread of the adelgid (Amman and Speers, 1965) throughout the Southern Appalachians occurred during the late 1960's and early 1970's.

Community and Ecosystem Impacts. The spruce-fir forests of the Southern Appalachians have changed dramatically due to the balsam woolly adelgid. Directly comparable data from pre- and post-adelgid periods are rare. Stand live basal area has seriously decreased after the death of large fir (Busing et al., 1988; Busing and Clebsch, 1988). Nicholas et al. (1992b) compared data collected from plots in the mid-1980s in the Great Smoky Mountains with measurements from 1946 by Oosting and Billings (1951). Live fir basal area values were only 8-62 percent of those recorded by Oosting and Billings.

There are many implications for the decimation of Fraser fir and subsequent canopy opening. Red spruce, Fraser fir's codominant in the forest, may be exposed to greater amounts of wind and sun with detrimental effects to incremental growth and total density (Nicholas et al., 1992b). Conversely, at sites that originally had low densities of fir trees, spruce may respond positively to stand thinning (Zedaker et al., 1988; Reams et al., 1993). Other, mostly understory, arboreal taxa seem to be increasing in numbers over time since canopy opening, including mountain maple (*Acer pensylvanicum* L.), serviceberry (*Amelanchier arborea* var. *Laevis* [Wiegand] Ahles), yellow birch (*Betula lutea* Michaux f.), and fire cherry (*Prunus pensylvanicum* L.f.) (DeSelm and Boner, 1984). Mountain-ash also seems to respond to increased levels of light but has recently suffered significant mortality (see Section 3.3), possibly due in part to damage from its own exotic insect, the mountain-ash sawfly, *Pristiphora geniculata* Hartig (Nicholas, 1992).

Ecosystem processes, such as nutrient cycling and energy flow and disturbance regimes, will also be affected by fir death. Decomposition processes and the rate of nutrient cycling may accelerate with greater soil insolation. Large amounts of dead fir wood will provide greater nutrient influxes in the short term. However, such changes in ecosystem processes are not limited to the spruce-fir forests, since these high elevation sites are found at the headwaters of major watersheds. Potential changes in disturbance regime include the greater possibility of windthrow and ice storm damage.

Fraser Fir Regeneration and Future Prospects. Perpetuation of Fraser fir requires not only survival of existing seedlings and saplings to cone-bearing age (20-30 years) but also the viability of seeds produced and successful growth of those seedlings in an altered environment. Fraser fir bear large seed crops every two to four years; however, Nicholas et al., (1992a) found that 88-100 percent of seeds were unfilled over a five-year period. Whether this is normal or not is unknown, but Fedde (1973) has found that adelgid infestation impairs seed production.

Observations of the great increase in *Rubus* densities have prompted questions about the ability of this gap-invading species, and possibly others, to out-compete fir seedlings. Data by Pauley (1989) and Pauley and Clebsch (1990) indicate that this is a serious possibility by noting the negative correlation between fir seedling and *Rubus* densities.

The question of the probability of Fraser fir extinction has not yet been answered satisfactorily. This uncertainty is reflected by the USDI Fish and Wildlife Service (1993) review of Fraser fir for possible listing as a threatened or endangered species. Eagar (1984) believed a cycle of infestation followed by regeneration, which survives to produce viable seed before death, is likely. Nicholas et al. (1992a) were more skeptical and suggested that not enough is known about future seed viability and whether sufficient regeneration exists which can grow to maturity and produce a vigorous understory numerous enough for perpetuation. Rheinhardt (1984) suggested that in the case of fir extinction, the remains of the Southern Appalachian spruce-fir ecosystem will resemble Whitetop, Virginia, which is dominated at high elevations by spruce, but no fir.

Other Possible Causes of Forest Stress

Other potential (and possibly interacting) causes of spruce-fir forest deterioration have been considered, including yet unidentified pests, pathogens, and diseases; stand dynamics; and gradual climatic changes and other air quality-related concerns. In a study initiated in 1984, Bruck and others (1989) found a near absence of significant infestations and infections by pests and

pathogens in the Southern Appalachian spruce-fir.

Few studies on stand dynamics of the spruce-fir have considered areas undergoing rapid change (sites with high levels of dead spruce and/or fir or deteriorating crown condition) as case studies. Instead, many have measured sites where the overstory is in composition equilibrium (White et al., 1985a; White et al., 1985b; Busing and Clebsch, 1987). Unlike other studies, Zedaker et al. (1989) used stand-level data at insect-ravaged and/or second growth Southern Appalachian sites and compared them to an early, second-growth northern red spruce study to examine stand behavior. They suggested that no unexplainable shortfall in stand carrying capacity is apparent when adelgid-caused mortality is added to the standing live basal area.

Climatic warming due to increased carbon dioxide concentration in the atmosphere will need to be considered in future assessments of forest condition. Research on water stress impacts indicates that for Fraser fir, water stress caused a significant decrease in biomass (Tseng et al., 1988), while Seiler and Cazell (1990) found red spruce to be very resistant to water stress in terms of photosynthetic rates during dehydration and recovery after watering.

Daily average ozone concentrations in the Southern Appalachian highlands exceeds those occurring at lower elevations. However, red spruce and Fraser fir have been demonstrated in both field and controlled exposure studies to be insensitive to ozone at current concentrations (Tseng et al. 1988; McLaughlin and Kohut, 1992; Edwards et al., 1995).

Chapter 3

Trends and Condition of Spruce-Fir Forests

3.1 Tree growth

While there is no question that Fraser fir populations are deteriorating, the decline of red spruce is not as well defined (Cook and Zedaker 1992). McLaughlin et al. (1987) found an annual growth increment decline in southern red spruce beginning in the early 1960s. However, LeBlanc et al. (1992) suggested that the current decrease in incremental growth of southern red spruce is not greater than historical levels. Reams et al. (1993) quantified the cyclic behavior of radial growth trends and speculate that overall spruce growth decrease is part of a long-term, climate-induced cyclic fluctuation and point out that a local maximum in radial growth preceded the 1960's decline. They concur with LeBlanc et al. (1992) that the recent downward trend is not unusual when compared with data from the past 200 years. If Southern Appalachian spruce is suffering unusual mortality or incremental growth reductions, factors other than acidic deposition may be at least partially responsible, including drought (Adams et al., 1985) and the sudden increase in exposure to wind and ice storms after the death of neighboring fir (Zedaker et al., 1988; Nicholas et al., 1992b).

3.2 Crown condition

Bruck et al. (1989) and Bruck and Robarge (1988) assessed changes in crown condition of spruce at Mt. Mitchell over time. The overall crown conditions of both red spruce and Fraser fir deteriorated between 1984 and 1987; in 1984, 74% of the live spruce were classified as "intact" (0 to 10 percent foliage loss), but by 1987 this proportion had dropped to 34 percent. Interpretation of the results of this survey must be qualified. First, monitored trees were not randomly chosen, but were in plots selected specifically for the good crown condition of trees in 1984; thus, trends may not be representative of the population as a whole. Second, although fir infestation by the adelgid may influence trends in spruce condition, effects of the adelgid were not monitored on the sample plots.

Nicholas, Eagar and Zedaker in Peart et al. (1992) assessed red spruce crown condition in spruce-fir stands at Mt. Rogers, Whitetop Mountain, and the Black and the Great Smoky Mountains from 1985 through 1989, using a similar system for assessing crown condition. Stands at Mt. Rogers and Whitetop showed little change in crown condition from 1985 to 1989 with between 85% and 95% of the trees classified as normal. Both high and low elevation stands in the Black and the Great Smoky Mountains showed deterioration in crown condition between 1985 and 1987 and then a stabilization through 1989. The most consistent change was in the Great Smoky Mountains. In 1985, 86% of the red spruce were rated as normal with respect to percent of crown intact. In 1989, only 52% were considered normal.

The same trees used in the 1985-1989 study were evaluated in 1995 in order to determine if this trend in deterioration of red spruce crowns was continuing or was just part of natural dynamics

(Eagar and Nicholas, unpublished data). There was only a modest improvement in crown condition in 1995 with 63% of the red spruce classified as normal. There was an elevational trend in 1995 which was not apparent in the 1989 data, with 68% classified as normal at 1525 m (low elevation) while 56% were considered normal at the highest elevation. These data suggest that the high elevation spruce-fir forests in Great Smoky Mountains are still experiencing stress.

Zedaker et al. (1989) and Peart et al. (1992) suggest that several stressors influenced changes in red spruce condition. Coincident with deterioration of spruce crown condition was a three year period of drought conditions during the summers of 1986-88. Fraser fir at Mt. Rogers had escaped major damage from the adelgid while populations in the Black and the Great Smoky Mountains had been decimated. Furthermore, both red spruce and Fraser fir are highly shade-tolerant species. Red spruce in fir-dominated stands could be suffering from the shock of sudden exposure (similar to thinning shock) due to high recent fir mortality. In the highest elevation stands, fir accounted for 30% to 90% of total stand basal area. Heavy fir mortality therefore results in sudden exposure of spruce to high light intensity and strong winds which could result in changes in red spruce condition. Adelgid-caused mortality of fir and subsequent thinning shock is not correlated with all changes in red spruce crown condition. However, lower elevation stands in the Great Smoky Mountains experienced changes in red spruce condition similar to those of the high-elevation stands, yet fir is a minor component of these lower elevation stands.

Detailed red spruce crown symptoms were also examined at three Southern Appalachian sites (Nicholas et al., 1989). In the past, dieback from the top of the crown has been frequently noted as a symptom of deterioration (Johnson and Siccama, 1983; Weiss et al., 1985; Innes and Boswell, 1990). Overall, Nicholas et al. (1989) found that only 5% of all live overstory spruce were observed to have such dieback. In branch samples, needle damage was found in 10% of sampled internodes, with some evidence of insect feeding and infrequent occurrences of mining, twisting, or galls. Degree of needle discoloration was greater on the upper surface than the lower surface of needles. No significant trends in needle damage or discoloration by elevation or canopy position were detected. More than 65% of the internodes had some incidence of flecking and more than 25% had some incidence of whole needle discoloration. However, in most internodes, discoloration occurred in less than 10% of the needles (Nicholas et al., 1989).

3.3 Mortality

There are few studies that directly measured tree mortality of eastern spruce-fir forests, and instead, standing dead tree estimates have been widely used as surrogate information. However Nicholas (1992) maintains that enumerations of standing dead trees should not be used to assess tree mortality patterns. She found that a large and variable proportion of overstory trees fall to the ground immediately upon death. Recent studies in both the Northern and Southern Appalachians that concentrated on reporting the amount of dead red spruce stems (Dull et al., 1988; Tritton and Siccama, 1990; Silver et al., 1991; Craig and Friedman, 1991) with no consideration of stem size, offer little insight into previous stand development. Furthermore, assessments of standing dead stems using aerial photography are unable to detect the difference between dead spruce or dead fir (Dull et al., 1988).

Direct measurement of tree mortality, while limited, indicates that overstory annual mortality varied with species. Mortality rates over a five year study (1985-1989) for the Great Smoky

Mountains found that mountain-ash had the highest rate (9.7%), followed by fir (7.4%), birch (1.8%), and spruce had the lowest (1.3%). Mortality rates for those species at Mt. Rogers NRA, Virginia and the Black Mountains of North Carolina were similar (Nicholas, 1992). Busing and Wu (1990) found a red spruce annual mortality rate of 1% over a two decade period (1960-1986) before the balsam woolly adelgid infested most fir in a Mount Collins site in the Great Smoky Mountains. That area was resampled after adelgid-caused fir mortality occurred and red spruce still had a 1% annual mortality rate (measured from 1986 to 1993) (Busing and Pauley, 1994).

The long term mortality rate of one percent was exceeded in 1987 for red spruce in the Black Mountains. Two separate studies indicated that mortality during this year was between 8% (Nicholas and Zedaker, 1989) and 34% (Bruck et al. 1989) A substantial ice storm occurred during the winter of 1986-1987 in the Black Mountains which eliminated many trees from the permanent plots used in these two studies. The wide difference in annual mortality between the two studies is due to the distribution of plots within the Black Mountains. Bruck et al. used plots that were more concentrated around Mt. Mitchell and at higher elevation, therefore, more exposed to ice build up. Nicholas and Zedaker used plots that were distributed throughout the Black Mountains and stratified by elevation. Overall, it seems that long-term mortality rates of red spruce are stable while annual mortality rates for Fraser fir remain above 4 percent (Nicholas, 1992).

Chapter 4

Soils and Nutrient Cycling

Since the end of the National Acid Precipitation Assessment Program (NAPAP) in 1990, research on acidic deposition and nutrient dynamics in high-elevation spruce-fir forests has continued in the Southern Appalachians (SA). There is now a large body of published research on soil processes and nutrient cycles from three spruce-fir sites in the SA. The Integrated Forest Study (IFS) was funded by the Electric Power Research Institute and to a limited extent by the Spruce-Fir Research Cooperative (SFRC). Of the 17 sites in this international program, six were located within or near Great Smoky Mountains National Park. The Smokies had three high elevation sites with two in red spruce stands (Tower and Becking sites) and one in an American beech (*Fagus grandifolia*) stand. Coweeta Hydrologic Laboratory is located just south of the Smokies and had two sites; one dominated by oak (*Quercus spp.*) and one in a white pine (*Pinus strobus*) plantation. At Oak Ridge, TN, a site was located in a loblolly pine (*Pinus taeda*) stand. The SFRC and the Tennessee Valley Authority funded a comprehensive research program in a red spruce forest on Whitetop Mountain, VA (Mount Rogers Recreation Area) which included a considerable amount of work on acidic deposition effects on nutrient dynamics and soil processes. The SFRC also funded some research on spruce-fir soils in the Black Mountains of North Carolina. Results from these studies provides an opportunity to conduct a much more detailed evaluation of acidic deposition effects on soils than was possible during NAPAP.

Important topics which need to be considered in the evaluation of the role of atmospheric deposition in the decline of high-elevation spruce-fir ecosystems in the SA include (1) the processes by which atmospheric inputs can influence soil and soil water chemistry, particularly how and to what extent this differs from processes that occur naturally in the soil; (2) what soil/ecosystem conditions tend to predispose the system to such external alterations; (3) how these deposition-induced changes in soil and solution chemistry in turn can affect nutrient availability and tree nutrition; and (4) how ultimately this combination of natural and anthropogenic factors may adversely affect forest growth. In the following sections only items (1) through (3) will be discussed, while physiological processes and growth are covered in Chapter 5.

4.1 Susceptibility of soils to strong acid inputs

In evaluating the sensitivity of Class I areas, a number of questions need to be addressed: (1) Are there recognizable soil and ecosystem attributes that are common throughout the SA which may render these areas susceptible to the effects of acidic deposition? (2) How are these specific attributes translated into soil water chemistry? and (3) What effect does atmospheric deposition have on soil-water interactions and on the soil as a growth environment?

4.1.1 Soil characteristics

Soil development

There is a high level of consistency in the soil properties among the high-elevation spruce-fir systems in the SA which is linked to similarities in landscape features, climatic conditions, and soil development history (Joslin et al. 1992; Fernandez 1992). Although the southeastern U.S. is typically associated with "old" landscapes uninfluenced by the most recent glaciation, soils in the higher elevations are nevertheless considered relatively immature in terms of soil development. The major forces in keeping these soils young are the frequent disturbances due to slope instability and soil mixing associated with landslides on steeper terrain and windthrow of the shallow-rooted trees (Wolfe, 1967; Fernandez, 1992; White and Cogbill, 1992).

The recalcitrant litter produced by coniferous forests, the cool and wet conditions that slow decomposition, and the absence of high-intensity consumptive fires in these moist high-elevation ecosystems have resulted in a significant accumulation of organic matter in the forest floor (O horizon) and the upper mineral soil (A horizon) (McCracken et al., 1962; Wolfe, 1967, Weaver, 1972; Johnson et al., 1991; Joslin et al., 1992, White and Cogbill, 1992). In some instances the entire soil profile is dominated by organic matter over bedrock with little or no mineral soil present, in which case soils are classified as Histosols. Kelly and Mays (1989) found that approximately 12% of the soils surveyed in the SA fit this latter category. The abundance of organic carbon strongly influences soil solution chemistry through the role it plays in soil formation as well as in curtailing aluminum (Al) toxicity.

Podzolization is an important soil-forming process under these forests. The combination of slow decomposition of conifer litter in a cool environment and high precipitation leads to the production of organic acids that facilitate leaching of cations and translocation of Al as organo-metal complexes from the upper soil horizons into the deeper horizons. The soils in SA Class I areas are primarily classified as Inceptisols (immature soils at the beginning of soil development) which occur in association with less frequent Spodosols (<6% according to Kelly and Mays, 1989) and are, like the latter, undergoing podzolization (Joslin et al., 1987; Lietzke and McGuire, 1987; Binkley, 1989; Kelly and Mays, 1989; Richter et al., 1989; Feldman et al., 1991). Frequent site disturbance inhibits the soils from fully developing into Spodosols. Feldman et al. (1991) suggested that the dominant soil forming processes, and resulting soil chemical properties, make the high-elevation soils in the SA more similar to Northern Appalachian Spodosols than to low-elevation soils of the southern mountains, a conclusion which is of great importance in assessing soil behavior in response to acidic deposition.

Soil Chemical Properties

In Southern Appalachian spruce-fir ecosystems, the production of base-poor litter (which is typical of conifers) and the climatically induced high leaching rates result in naturally acid soils with low base saturation (e.g., Robarge and Smithson, 1989; Smithson, 1990; Fernandez, 1992; Joslin et al., 1992, Johnson, 1992). Soils are typically in the Al buffering range, with base saturation generally less than 10% and the exchange complex dominated by exchangeable Al (Johnson and Fernandez, 1992; Joslin et al., 1992). Cation exchange capacity and base saturation are highest in the upper

horizons and largely associated with organic matter (Wolfe, 1967). The presence of mostly unreactive bedrock (e.g., sandstones) in these areas limits base replenishment through weathering (Elwood et al., 1991; White and Coghill, 1992). Because these soils are already in the Al buffering range, they will not significantly acidify further (Reuss and Jonson, 1986; Johnson and Fernandez, 1992).

Nitrogen status

One issue that has gained in importance in the post-NAPAP era is the growing evidence of nitrogen saturation in high-elevation forest ecosystems of the SA and the influence this condition may have on terrestrial and aquatic chemistry in Class I areas. A number of publications have substantiated through lysimeter or streamwater data the ubiquitous nature of low N retention and high NO₃ leaching export in high-elevation SA spruce-fir ecosystems (Smithson, 1990; Johnson et al., 1991; Joslin and Wolfe, 1992 & 1993; Joslin et al. 1992; Van Miegroet et al., 1992; Jamison et al., 1994; Flum and Nodvin, 1995, Nodvin et al., 1995). There are some distinct differences in this respect between high-elevation ecosystems in the SA and their counterparts in the northeastern U.S. Soils in the SA generally have more N in the mineral soil and consistently lower C/N, which translates into higher N mineralization potential and greater nitrification capacity (Joslin et al., 1992). Interestingly, Wolfe (1967) first noted that in the Great Smoky Mountains, soils under beech had even lower C/N values than under spruce, an observation repeated during the IFS (Van Miegroet et al., 1992a). It may account for the significantly higher N mineralization and nitrification rates that have been measured under beech compared to spruce (Van Miegroet et al., 1992a, Garten and Van Miegroet, 1994). The inherently high N release rates coupled to high N deposition rates in systems with low N requirements and uptake (see Section 4.2.3) are likely to lead to high NO₃ leaching losses below the rooting zone (i.e., N saturation).

Sulfate adsorption capacity

The ability of soils to retain strong acid anion loads, particularly sulfate (SO₄), is of critical importance in assessing their susceptibility to the effects of atmospheric deposition. Yet for many soils, and for forests soils in particular, information on SO₄ adsorption is seldom available (Turner et al., 1986a). There are essentially two approaches to assess the sulfur (S) retention capacity of systems: (1) use soil water or soil extracts to directly measure S retention in the field or laboratory and infer SO₄ adsorption capacity from soil chemical properties (e.g., Harrison et al., 1989 a&b) or (2) use stream water chemistry as an indicator of total watershed S retention through mass balance calculations (e.g., Rochelle et al., 1987). While the first approach primarily focuses on the upper soil (i.e., the rooting zone) and may be most appropriate for this discussion, the second approach reflects the behavior of deeper soil layers and near-stream zones which may have substantially different chemical properties from those in the rooting zone. The latter approach may therefore be more appropriate for assessing susceptibility of aquatic ecosystems and the role of terrestrial watershed components in regulating S input to these systems.

Information from both approaches can be useful in estimating SO₄ retention capacity of ecosystems; however, it is important to recognize that the receptor of interest (aquatic versus terrestrial ecosystem) will influence how soil properties are evaluated (e.g., soil depth, hydrology, role of riparian zones, etc.). The regional assessments that were conducted under NAPAP focussed mainly on the sensitivity of surface waters. They were very broad in approach and coarse

in scale in that they attempted to relate SO_4 adsorption capacity to broad soil taxa in the eastern U.S. (e.g., Olson et al., 1982, Church, 1989, Church et al., 1989 & 1992). Highly weathered Ultisols typical of a large part of the southeastern U.S. were assigned high SO_4 adsorption capacity, Inceptisols moderate adsorption capacity, and Spodosols low adsorption capacity. Because of the spatial scale of the sensitivity analyses, many forest soils, and particularly the small region of the high-elevation spruce-fir zone in the SA, were not explicitly considered in these analyses (Turner et al., 1986a). This led to the conclusion that SO_4 adsorption capacity is generally greater in the southeastern than in the northeastern U.S. (e.g., Turner et al., 1986b, Elwood et al., 1991) based primarily on the predominance of Ultisols in the South. Furthermore, the National Stream Survey (NSS) did not survey acid stream reaches (i.e., high-elevation streams draining watersheds with spruce-fir) and may thus have introduced an inherent bias in the outcome of the analysis related to stream water acidification and its causes (see Elwood et al., 1991).

Some of the conclusions of the NSS with respect to soil sensitivity to the effects of acidic deposition, and especially the rate of SO_4 adsorption, although valid for most of the Southern Blue Ridge (SBR), are unlikely to be applicable to the high-elevation ecosystems of the SA. These nuances are not always recognized and have led to contradictory published opinions on the fate of atmospheric S deposition in the soils in the SA. The essence of the problem is that the name "Inceptisol" is given in soil taxonomy to any pedon (i.e., three-dimensional soil body) with immature soil development, irrespective of the dominant soil forming processes. Inceptisols are one of the most variable soil orders, can be associated with several soil orders, and generally have properties like the more developed soil orders with which they are associated. Broad statements as to the SO_4 adsorptive characteristics of Inceptisols may not be justified or may be erroneous when taken out of spatial context (Rochelle et al., 1987; Binkley et al., 1989). For example, SO_4 adsorption in Inceptisols associated with a low-elevation Ultisols in the Southeast is expected to be much larger than in Inceptisols typical of high-elevation spruce-fir in the SA which occur in association with Spodosols (Rochelle et al., 1987). This distinction is important because it largely resolves the apparent contradiction that exists between modeled or assumed SO_4 adsorption capacity and actual field or laboratory observations (e.g., Cook et al., 1994).

Several parameters contribute to the SO_4 retention capacity of soils. Based on an analysis of several forest soils, Johnson and Todd (1983) found that SO_4 adsorption increases with increasing soil solution SO_4 concentration, is positively correlated with Fe and Al oxides, but negatively correlated with organic matter content. Surface horizons of most soils and carbon-rich subsoils of Spodosols are inefficient SO_4 adsorbers, even if enriched in Fe and Al sesquioxides. The corollary to this conclusion is that Inceptisols associated with Spodosols or undergoing podzolization should likewise have low SO_4 adsorption capacity. Laboratory adsorption studies conducted as part of the IFS verified the positive correlation between oxalate extractable Al and SO_4 adsorption capacity of Inceptisols and Spodosols (Harrison et al., 1989a&b; Harrison and Johnson, 1992). Results showed that Inceptisols in the SA had low amounts of adsorbed SO_4 in the field and low SO_4 adsorption capacity during the laboratory studies, behavior that was consistent with the high total carbon content and the low oxalate-extractable Al levels in these soils. This study also demonstrated the wide range of adsorption that can exist within the same soil order and showed that the SO_4 adsorption in Inceptisols of the SA was often lower than in northeastern spodosols included in the IFS study (Harrison, Unpublished Data).

Sulfur input and SO_4 leaching fluxes measured in a red spruce site in Great Smoky Mountains

National Park during the IFS and stream water measurements at several locations in the Park suggest little atmospheric S retention (<20%) (Mitchell, 1992 a&b, Cook et al., 1994). However, there is evidence from other field studies that a fraction of the incoming S may be retained in some of these high-elevation ecosystems [e.g., decline in soil water SO_4 levels with soil depth (Joslin et al., 1987) or watershed-level mass balance indicating Input > Output (Nodvin et al., 1995)]. It is not clear at this point to what extent this removal is due to SO_4 adsorption, biological immobilization by plants or microorganisms, or to microbial reduction of SO_4 in parts of the soil or watershed that exhibit reducing conditions (e.g., riparian zones). The dominance of the organic S fraction in the total soil S pool would indeed suggest some role of biological processes in S retention (e.g., Harrison et al., 1989a; Mitchell et al., 1992). Fitzgerald and Autry (1992) suggested that the capacity of SA spruce soils to accumulate S through net microbial immobilization could be large, especially when considering the entire soil profile. However, they also indicated that the potential for abiotic SO_4 adsorption generally exceeded organic S formation in this ecosystem type and that there may be large spatial variability in S retention both across sites (horizontal) and with soil depth (vertical). Mitchell et al. (1992) concluded that the role of biological processes in regulating SO_4 flux is most important in sites with low atmospheric S inputs. These seemingly contradictory findings make a general evaluation of the SO_4 retention capacity of high-elevation ecosystems particularly difficult.

4.1.2 Relationship of soil to soil solution chemistry

The combination of climatic conditions (low temperature, high precipitation), vegetation (conifer litter), and parent material (low weatherability and base replacement capacity) results in podzolization as the natural soil forming process in these SA spruce-fir ecosystems and causes soils in these regions to be naturally acid, exchangeable base cation reserves to be low, and Al solution levels to be inherently high (especially in the upper soils). These conditions, in turn, have a profound effect on the sensitivity of soils and soil solutions to atmospheric deposition inputs and predispose the system to potential Al toxicity and Al induced inhibition of cation uptake, rather than to cation depletion per se. Indeed, given the already low base status, it is unlikely that these soils would significantly acidify further. Although further base cation depletion is possible with the input of strong acid anions (e.g., NO_3^- , SO_4^{2-}), this is unlikely to be the major pathway of soil solution change (e.g., Johnson and Fernandez, 1992).

More profound effects are expected in Al chemistry, particularly in terms of amounts of potentially toxic monomeric Al^{3+} forms. The mechanisms of Al mobilization are described in detail in Reuss and Johnson (1986) and Johnson and Fernandez (1992). In acid soils with low base saturation, any increase in mobile anion concentration (SO_4 or NO_3) in the soil solution will preferentially displace Al^{3+} ions (over divalent and monovalent cations such as Ca^{2+} , Mg^{2+} or K^+) from exchange sites because of their abundance on the exchange complex and their charge. Joslin et al. (1987) reported Al levels of 15-35 $\mu\text{mol/L}$ in the organic-rich O and A horizons increasing to 35-50 $\mu\text{mol/L}$ at greater soil depth under birch-spruce forests in the Raven Fork Watershed of the Great Smoky Mountains National Park. Aluminum levels generally in the range of 50 $\mu\text{mol/L}$ and occasionally above 100 $\mu\text{mol/L}$ were measured in the upper A horizons during the IFS study in the Smokies and at Mt. Mitchell, NC. Average Al concentrations did not change considerably with soil depth but peak values declined significantly (Smithson, 1990; Johnson et al., 1991; Van Miegroet, unpublished data). Joslin and Wolfe (1992) reported similarly high Al average and peak concentration for the O and upper mineral soils (0-25 cm) at spruce sites on Whitetop Mountain, VA.

At highly polluted sites, there may be a major anion shift from naturally organic acid-dominated to strong acid anion-dominated soil solutions and a concomitant shift in Al species from organically complexed Al to monomeric Al^{3+} forms. This anion/Al shift can occur to various degrees in the SA soil systems and/or at different parts of the soil profile, and organic acids may continue to exert some influence on leaching and Al chemistry even in polluted areas (e.g., Joslin et al., 1987). Although **total** Al levels in solution may be very similar between pristine and heavily polluted areas (see examples in Fernandez, 1992; Johnson and Fernandez, 1992), significant differences in Al species composition are expected. This issue of Al speciation needs to be considered when evaluating soil solution chemistry data. High Al levels in solution by themselves are not deleterious and often occur naturally in part of the soil profile, but when a high fraction of that Al is in monomeric form the possibility of Al toxicity, inhibition of cation uptake, or leaching loss to streams is increased.

4.1.3 Role of atmospheric deposition in changing soil water chemistry

The two main mobile anions of importance in causing Al mobilization in SA soils are SO_4 and NO_3 , the former because of typically low SO_4 adsorption capacity of SA soils coupled to high atmospheric inputs, the latter because most high elevation ecosystems are N saturated (see discussions in sections 4.1.1, 4.2.1 and 4.2.3). In addition to atmospheric NO_3 inputs, mobile NO_3 also originates from nitrification of other atmospheric N input forms (e.g., NH_4^+) or from mineralization and nitrification of the inherently large organic N pools in these systems (Joslin et al., 1987; Johnson et al., 1991; Van Miegroet et al., 1992). Soil solution measurements at different spruce-fir ecosystems throughout the SA have consistently documented (1) that SO_4 and NO_3 are the dominant anions in rooting zone solutions, (2) that Al dominates the cation fraction, and (3) that most of the Al in the upper mineral soil solution (A horizon) occurs as Al^{3+} (e.g., Joslin et al., 1987; Smithson, 1990; Joslin and Wolfe, 1992 & 1993; Van Miegroet et al., 1990; Johnson et al., 1991). These studies have also indicated that there is less seasonal variability in SO_4 solution concentrations than in NO_3 solution concentration and NO_3 has higher concentration peaks. Because Al mobilization is driven by peak events rather than by annual anion loads or average anion concentrations, the degree of temporal variability in total anion concentration is of critical importance in driving Al levels above potentially toxic levels. This also explains why the role attributed to SO_4 in the mobilization of Al differed among studies with the concentration of SO_4 being either a poor predictor of Al concentrations (where SO_4 concentrations were relatively stable, e.g., Smithson, 1990, Joslin and Wolfe, 1992) or a good predictor (where SO_4 concentrations fluctuated considerably, e.g., Johnson et al., 1991). However, all studies consistently showed a strong correlation between Al levels and NO_3 levels in solution and concluded that internal N dynamics, and particularly of occurrence of periodically large NO_3 peaks were a major driver in Al chemistry.

Comparisons between atmospheric inputs and soil leaching of SO_4 or between SO_4 concentrations in throughfall and those in soil solutions at different SA sites indicate that the SO_4 measured in rhizosphere solutions in the high elevation sites of the SA is primarily derived from atmospheric inputs, and that greater soil SO_4 fluxes are generally associated with greater throughfall inputs to the forest floor (e.g., Joslin and Wolfe, 1992; Mitchell, 1992b; Van Miegroet, unpublished data). Smithson (1990) found that sites at Mt. Michell, NC with greater cloud water impact typically had higher soil solution SO_4 , although differences in soil solution fluxes were less pronounced than

those in total atmospheric S input. Joslin and Wolfe (1992) made a similar observation at Whitetop Mt, VA. In areas that contain pyrite bearing minerals (e.g., Anakeesta formations), SO_4 may also be internally released through weathering (Rochelle et al., 1987; Cook et al., 1994), but this internal source is likely more important within the context of stream water acidification than for rooting zone chemistry.

The situation is somewhat different for N. Although atmospheric input and leaching output fluxes are often similar in magnitude and would suggest that NO_3 leaching losses are entirely derived from atmospheric deposition, a closer look at N fluxes in and out of the upper soil suggest a significant contribution from internal processes (N mineralization + nitrification) to the overall NO_3 signal, and particularly to its temporal variability (Joslin et al, 1987 & 1989; Johnson et al., 1991, Jamison et al., 1994). Cook et al. (1994) found streamwater NO_3 concentrations were typically higher during the dormant season, which provides further evidence for the biological control of N retention and release. In a short-term field study near the IFS spruce site in the Smokies, throughfall and A-horizon solutions were collected at nearly the same location. These paired comparisons showed a strong direct link between throughfall and lysimeter SO_4 levels and a consistent increase in soil solution NO_3 concentration over those in the throughfall solutions (40-100 $\mu\text{mol/L}$). However, it was not possible to unequivocally establish the origin of the soil solution NO_3 based on isotopic N signature (Van Miegroet, Unpub. Data; Garten and Van Miegroet, 1995).

4.1.4 Uncertainties and implications to policy decisions

The observations discussed in section 4.1.3 may have major implications for policy and especially for the effectiveness of regulatory decisions on the health of Class I ecosystems because of the sources of strong anion input and the role in Al mobilization differs substantially between SO_4 and NO_3 . Whereas the SO_4 fluxes in spruce-fir systems are largely due to atmospheric inputs, NO_3 seems to be derived from both atmospheric and internal sources, is largely under biological control, and tends to show much larger seasonal variations. Consequently, high SO_4 levels may be responsible for elevated Al background levels, but it is the NO_3 peaks that are responsible for causing the potentially harmful Al peaks. Curtailing SO_4 input alone may not eliminate the problem as long as N dynamics (especially nitrification) dominate the solution chemistry in the SA soils. At best, such regulatory measures may lower the magnitude of the Al peaks and the frequency with which they exceed set threshold values. Soil and solution chemistry data available to date suggest that more attention should be directed towards understanding internal N dynamics within the framework of local stand dynamics and as affected by anthropogenic influences.

4.2 Nutrient cycling in high elevation spruce-fir ecosystems

Although few mineral cycling studies have been conducted in the spruce-fir zone of the SA, the data that are available point at some interesting general trends with respect to nutrient distribution, atmospheric input fluxes, internal cycling, leaching losses below the rooting zone, and especially the role of stand dynamics and forest disturbances on these cycling patterns. Some of the details can be found in Johnson et al. (1991) and in Johnson and Lindberg (1992).

4.2.1 Atmospheric deposition regime

One of the significant outcomes of the IFS was a more accurate quantification of atmospheric deposition processes and the recognition that the high-elevation spruce-fir systems in the SA are subject to some of the highest atmospheric N and S inputs in the eastern United States. At the IFS spruce Tower site near Clingmans Dome in the Great Smoky Mountains total deposition of N was 27 kg ha⁻¹ yr⁻¹ and of S 36 kg ha⁻¹ yr⁻¹ (Lindberg 1992; Lindberg and Lovett 1992; Lovett 1992; Lovett and Lindberg 1993). A large fraction of these pollutants are deposited via dry deposition (around 25% of total S, little less than 50% of total N) and through cloud and fog impaction (almost 50% for S, 30% for N). In that cloud deposition is generally proportional to cloud immersion times, and the frequency of cloud immersion at the IFS site is lower than at many other mountain summits in the SA, atmospheric deposition levels at other sites could be even higher than those observed in the Smokies (Lovett and Lindberg 1993). Smithson reported annual S deposition rates of 20 kg S ha⁻¹ yr⁻¹ for a low cloud and 49 kg S ha⁻¹ yr⁻¹ for a high cloud impact site on Mt Mitchell, NC, but no N deposition values are available for that location. Throughfall fluxes at Whitetop Mt, VA ranged from 40 to 47 kg ha⁻¹ yr⁻¹ for S and were a little less than 20 kg ha⁻¹ yr⁻¹ for N (Joslin and Wolfe, 1992). Considering that throughfall fluxes generally underestimate total N inputs at high elevation sites by as much as 15% due to canopy retention (Lovett, 1992), actual N deposition rates are probably in the range of those observed in the Smokies.

As the cloud base typically occurs at or above a given elevation [approximately 1800 m elevation in the Smokies (Lindberg and Owens, 1993)], and because of the influence of cloud exposure on deposition in mountainous regions (Lovett et al., 1992), one would expect atmospheric input rates to the forest floor to increase with elevation, especially when comparing locations above versus below the cloud base elevation. While a comparative throughfall study in the Smokies found this trend to be true for S deposition, tree-to-tree variability in throughfall fluxes within each site were too large to reveal any significant N input gradient with elevation (Lindberg and Owens, 1993). More importantly, Lindberg and Owens (1993) showed that canopy gap formation influences the deposition regime within these high-elevation forests by significantly increasing water and ion (SO₄ and NO₃) throughfall fluxes below mature edge trees. The stand structure (i.e., formation of gaps and edges) also has a significant impact on the elevational gradient in throughfall fluxes. Whereas total SO₄ fluxes underneath the edge trees nearly doubled when crossing the cloud base elevation, the elevational gradient for NO₃ followed the reverse pattern. Total throughfall fluxes below the edge trees were about 30% higher below the cloud base compared to the higher-elevation site and this was attributed to greater wind penetration and HNO₃(dry) vapor deposition at the lower site. However, elevational patterns were found to be less consistent with time than those for SO₄. A follow-up study at the Noland Divide Watershed (Shubzda, Unpublished Data) has further verified this elevational trend (increase in SO₄ input and decline in NO₃ input with increasing elevation).

4.2.2 Nutrient pools and fluxes

Only a few intensive nutrient cycling studies have been conducted in the SA (Weaver, 1972; Johnson et al., 1991; Johnson and Lindberg 1992). From these studies and comparisons with other spruce-fir ecosystems in North America (Johnson and Fernandez, 1992) a number of distinct features emerge. Organic matter is a major component in the nutrient pools and nutrient dynamics of this ecosystem type. The forest floor, and to a lesser degree the upper (organic-rich)

portion of the mineral soil, constitute a major reservoir of available nutrients (N, S, Ca, and Mg). Although this observation is common in many ecosystems, the situation is further accentuated for Ca and Mg in the SA by the extremely low exchangeable base cation levels in the mineral soil (see section 4.1.1).

A second distinct feature of these high-elevation ecosystems is the dominance of hydrologic fluxes in the biogeochemistry of most elements: deposition, throughfall, and leaching (all hydrologic fluxes) generally exceed uptake and return by vegetation (biological fluxes). Low uptake rates are typical for mature forests where nutrient increment is primarily associated with wood increment. Recent heavy mortality of fir by balsam woolly adelgid infestation would tend to further decrease the role of vegetation uptake in nutrient retention. Input/output calculations indicate net loss or close to zero retention for N, Mg and S, but net accumulation of atmospherically deposited Ca. Such low biological retention of S and N has major repercussions on anion leaching losses as already discussed in section 4.1.2. At the IFS sites in the Smoky Mountains, there was a net (albeit small) annual loss for Mg through leaching ($\approx 2.5 \text{ kg ha}^{-1} \text{ yr}^{-1}$) which constituted 4% of the soil-exchangeable pool and 0.01% of the total soil pool. Since there is no information on the rate of Mg weathering release in these soils, it is difficult to assess whether or not this process is a significant factor in the Mg supplying power of the soil. Net leaching of all base cations together accounted for an insignificant portion of the exchange capacity and Ca appeared to be accumulating in this system at a rate of $4 \text{ kg ha}^{-1} \text{ yr}^{-1}$ (Johnson, 1992). Collectively, these observations tend to suggest that at this point in time soils in the SA are not significantly acidifying through base cation stripping even at current high atmospheric deposition rates, but that Al is the major cation mobilized from the exchange complex, as predicted by theoretical considerations (see Reuss and Johnson, 1986).

The increase in Ca and Mg flux below the canopy over atmospheric deposition inputs ($2 \text{ kg ha}^{-1} \text{ yr}^{-1}$ for Ca and $< 1.5 \text{ kg ha}^{-1} \text{ yr}^{-1}$ for Mg) supports the contention that these ecosystems are undergoing foliar leaching and that this may be an important factor in causing base cation (Ca and Mg) stress in these forests (see Section 4.3.1). In the case of Mg, net foliar leaching input to the forest floor appears to be larger than litterfall return. These rates could be even more important where hydrologic fluxes increase due to enhanced cloud interception associated with increasing elevation or at forest edges. At Whitetop Mt., VA, Joslin et al. (1988) found foliar leaching during acid cloud episodes equal to 16% of foliar Mg and as much as 36% of foliar Ca.

The predominance of throughfall input to the forest floor (hydrologic input) over litterfall returns (particulate input) is unusual for Mg and rather exceptional for Ca. It illustrates the Ca-poor status of the vegetation (and the foliage in particular), and suggests that deficiencies may be further aggravated in the future through a feedback mechanism between the base status of the foliage and the base supplying capacity of the forest floor that is produced from the foliage.

4.2.3 Stand dynamics and N saturation

Nitrogen saturation in SA spruce-fir ecosystems is indicated by the following signs: (1) high NO_3^- levels in solution throughout the year (Johnson et al., 1991; Joslin and Wolfe, 1992), (2) NO_3^- leaching rates closely in balance with atmospheric N deposition rates (Johnson et al., 1991; Van Miegroet et al., 1992b), (3) N mineralization rates in excess of plant uptake requirements (Johnson et al., 1991; Van Miegroet et al., 1992 a&b), and (4) lack of tree response to N

fertilization (Joslin, 1994). This condition is triggered by the combination of (1) large N pools with low C/N ratios in soil and forest floor which favor net N mineralization and nitrification, (2) high atmospheric deposition rates, and (3) low tree N uptake rates especially in mature forests which have low net primary productivity and/or having recently experienced heavy fir mortality (Johnson et al., 1991; Van Miegroet et al., 1992a).

The role of stand dynamics in belowground N dynamics of the SA was first documented by Silsbee and Larson (1981; 1982; 1983) through stream survey in the Smoky Mountains. The NO₃ concentrations in streamwater consistently increased with elevation, but were generally lower in streams draining watersheds that had been heavily logged prior to the establishment of the National Park compared to those that had not been logged or only lightly logged and contained more old-growth. A recent follow-up survey (Flum and Nodvin, 1995) showed: (1) NO₃ levels are highest and dominate the anion load in streams draining watersheds containing spruce-fir on non-pyrite bearing bedrock (2) NO₃ levels in streamwater tend to decline below the spruce-fir zone, and (3) sharp drops in NO₃ stream concentrations at the spruce-fir/hardwood ecotone more typically occur where the hardwoods have been logged. These observations formed the basis for a comparative study in the Smoky Mountains where N input via throughfall, N mineralization, and NO₃ leaching were compared across spruce-fir ecosystems at different developmental stages (Jamison et al., 1994). A preliminary analysis of the data shows significantly higher NO₃ concentrations in lysimeter solutions under mature to overmature stands compared to those containing considerable regeneration in the understory. This difference in NO₃ leaching can in part be attributed to a difference in stand vitality and tree uptake rates, as N inputs via throughfall were not significantly different among sites. Mature and overmature stands also showed the highest net nitrification rates, whereas net mineralization was not significantly different between stands (Jamison, personal communication).

Because of high nitrification potential in these N-rich soils, changes in stand structure, and especially the formation of gaps through tree mortality and windthrow, are likely to impact belowground N turnover and accentuate N saturation. There are several reasons why an increase in N mineralization and nitrification would be expected. First, any factor causing a decline in tree N uptake (including mortality) will reduce competition for available N and increase microbial N turnover. Indirect evidence to that effect can be derived from IFS. The Becking site had a higher proportion of dead standing biomass, higher N mineralization rates, higher NO₃ baseline levels in the soil solutions, and lower temporal fluctuations than the Tower site (Johnson et al., 1991; Van Miegroet et al., 1992a). In the classification of N saturation stages (see Stoddard, 1994), the decline in biologically induced temporal fluctuations in NO₃ levels and the increase in the overall baseline NO₃ concentrations in streams is used as an indicator for increasing N saturation within the watershed.

A second reason for expecting an increase in N leaching with gap formation is the effect of soil temperature increases on N mineralization rates. Joslin and Wolfe (1993) observed significantly higher NO₃ leaching rates at the warmer, more insolated portion of a gap compared to the shaded (cooler) area. Johnson et al., (1991) also showed the role of seasonal temperature fluctuations in causing periodicity in soil N transformations. A third possible contributor to increased N leaching is soil disturbance caused by windthrow. There is limited direct evidence on the role of soil disturbance on N dynamics, but this issue is frequently debated within the context of sampling methodologies for N availability and especially the relationship between field and laboratory assays

(e.g., Binkley and Hart, 1989). A small experiment in the Smokies suggests that soil disturbance can significantly accelerate nitrification, especially in N-rich soils (Van Miegroet, 1995). Furthermore, it is not uncommon to observe surges in nitrification following lysimeter installation (e.g., Johnson et al., 1991). On the other hand, wild hog rooting in beech forest, which tends to mix the upper horizons and is in that respect similar to soil mixing associated with windthrow, did not significantly affect N mineralization rates (Bloss, 1987). Finally, the presence of edges in the stand structure tends to increase N inputs to the forests floor via throughfall (Lindberg and Owens, 1993), which in turn could lead to greater NO₃ leaching rates.

For all the reasons stated above, areas in the SA that have suffered heavy fir mortality due to infestations by the balsam woolly adelgid or that are experiencing a decline in tree growth (whether naturally due to advanced stand age or induced by cation deficiencies) would seem particularly prone to the negative effect of accelerated NO₃ leaching and a further deterioration of the soil as a growing environment.

4.3 Forest Nutrition

4.3.1 Causes of nutrient deficiencies and uncertainties

There are several hypotheses that relate observed changes in spruce growth rates and carbon (C) allocation to soil chemical properties, and especially atmospherically induced changes in these properties. One hypothesis that is most frequently proposed is that spruce-fir forests in the SA are experiencing base cation deficiencies (especially Ca and Mg), either as a result of cation depletion in the upper soils or through inhibition of cation uptake triggered by elevated Al levels in the rhizosphere. Another hypothesis poses that root activity and function may be adversely affected by Al toxicity. A third hypothesis attributes low foliar base levels to a deposition-induced acceleration in foliar leaching. Central to proving causality between atmospheric deposition and a decline in forest nutrition and health is establishing (1) that trees are experiencing toxicity (e.g., Al) or nutrient deficiencies (Ca, Mg) and (2) that this condition is the direct result of atmospheric deposition.

There are still a number of uncertainties in establishing a direct causal link between atmospheric deposition and tree nutrition, particularly in view of the fact (1) that it is still unknown what portion of the strong anion pulses that mobilize Al in the upper soil of SA forests and/or leach cations out of the upper soil (e.g., Bondietti et al., 1991) are derived from internal processes (i.e., N mineralization and nitrification) versus atmospheric deposition; and (2) that historically these soils have been subject to acidification and cation depletion during soil formation. Low base status may have been a natural condition preceding the advent of atmospheric deposition, and we lack the long-term data required to prove this condition has changed significantly in recent times in the SA.

However, a pre-pollution natural condition of low base, acidic soils provides an important predisposing condition for mobilization of Al by atmospheric deposition of strong anions (Reuss and Johnson 1986). These high elevation forests receive the highest SO₄ deposition in the U.S. which provides an elevated baseline of mobil anions in soil solution. This high baseline makes the seasonal NO₃ peaks that are derived from both internal processes and atmospheric deposition more effective in mobilizing Al. Tree ring analysis and wood chemistry provides circumstantial evidence for a significant change in soil solution chemistry coincident with an increase in pollution (Bondietti et al., 1989).

Proposed deleterious effects of high Al levels in rhizosphere solutions are largely based on seedling studies conducted under controlled laboratory conditions that established threshold values for inhibition of Ca and Mg uptake (100 $\mu\text{mol/L}$; Raynall et al., 1990) or restricting root growth (200 $\mu\text{mol/L}$; Thornton et al., 1987; Joslin and Wolfe, 1988) in red spruce seedlings. Whether these threshold values are the same for mature spruce in a forest environment is not clear. Nutrient deficiency-sufficiency levels in red spruce foliage have likewise been largely based on a single greenhouse study with seedlings conducted in Canada in the early 1970's (Swan, 1971). These standards may or may not apply to field conditions. As noted by McLaughlin and Kohut (1992) comparisons between the nutritional status of Norway spruce seedlings (which behave quite similarly to red spruce seedlings) and mature trees suggest that more restrictive standards may be necessary in the field (see Table 1).

Elevational trends have been used as circumstantial evidence for the role of atmospheric deposition in causing a decline in forest nutrition. For example, McLaughlin et al. (1990 & 1991) reported lower tree growth, lower foliar Ca and Mg levels, and higher foliar Al levels at the highest elevation sites along three elevational gradients in the Smoky Mountains, and found these foliar trends to be strongly correlated with soil extractable levels. Wells et al. (1989) also noted higher Al levels in spruce foliage at the highest elevations in the SA. However, one must exercise caution in attributing causality. First, total atmospheric deposition shows large spatial variability related to stand structure and does not always increase with elevation (e.g., Lindberg and Owens, 1993). On the other hand, cloud exposure does increase with elevation (Lindberg and Owens, 1993) and has been linked to a decline in foliar base cation levels both in field (Joslin et al., 1988) and greenhouse studies (McLaughlin et al., 1993). Secondly, soil solution Al levels (both total and monomeric), Ca and Mg concentrations, and Ca:Al and Mg:Al ratios were not found to be significantly different between the mid- and high-elevation sites at Clingmans Dome (Van Miegroet, et al., 1990). In the Whitetop study, higher soil solution Al levels were not translated into higher Al levels in red spruce foliage (Joslin and Wolfe, 1992). Finally, slope and rockiness may cause significant differences in soil stability, soil depth and soil properties which may in turn affect site fertility status, irrespective of deposition regime (see Johnson and Fernandez, 1992).

A contributing and possibly critical factor to the observed decline in foliar base cations at the higher elevations may be the role of acidic cloudwater in leaching foliar nutrients (Joslin et al., 1988; Thornton et al., 1994). Based on work at Whitetop Mt., Joslin and Wolfe (1992) found significantly higher levels of NO_3 and Al in the soil solution and 40% less fine roots in the upper soil of mature spruce forests under high cloud exposure compared to those having less cloud exposure. Except for Mg in the mineral soils, no consistent and significant differences in Ca and Mg solution concentrations were measured. Solution Ca:Al and Mg:Al ratios were significantly lower at the high cloud site (mostly driven by the differences in Al concentrations). Despite these significant differences in Al solution concentrations, no significant differences in foliar Al levels were observed. For two out of three observation years, newly formed foliage had significantly lower Ca, Mg and Zn concentrations at the high cloud site (Thornton et al., 1994). Elevated soil solution levels of Al have been shown to inhibit the uptake of Ca and Mg by red spruce roots (Thornton et al., 1987). A combination of altered soil and solution chemistry coupled to higher foliar leaching rates seems a plausible explanation for the reported elevational trends in foliar levels and particularly for the sharp decline above the cloud base level (see McLaughlin et al., 1991).

Many of the proposed mechanisms for declining base cation nutrition are based on the assumptions that considerable root activity is located in the upper, most acid soils horizons, that available base cations pools are thus mainly confined to the forest floor, and that the deeper soil has little or no influence on tree nutrition in these forests (e.g., Robarge et al., 1989; Bondietti et al., 1991). Red spruce is a typical shallow rooting species and roots are most abundant in the upper horizons (e.g., Johnson et al., 1991; Joslin and Wolfe, 1992). This characteristic is common in conifer forests and may be related to the abrupt soil chemistry changes that occur between organic matter-dominated and mineral horizons (see discussions in Fernandez, 1992 and Johnson and Fernandez, 1992). However, some data indicate that roots do extend deeper into the soil profile (where base saturation may be somewhat higher) and that such roots should therefore be able to extract nutrients from that part of the profile as well, especially when trees encounter limited base supplies in the upper soil (e.g., Johnson et al., 1991; Johnson and Fernandez, 1992). Root distribution and the influence of organic material on rhizosphere chemistry and seedling growth and nutrition is also discussed in Chapter 5. The differential response of seedlings to acid treatments depending on whether organic matter was part of the growth medium is important to note (e.g., McLaughlin et al., 1993; Thornton et al., 1994), in that it underscores the potentially attenuating influence of organics on Al toxicity and/or Al induced nutrient deficiency.

Another area of uncertainty is the inability to accurately describe rhizosphere chemistry and its implications for tree nutrition. Most soil solutions discussed in the literature were collected with low-tension or zero-tension lysimeters, and essentially reflect gravitational water that drains freely through the soil. Soil water collected in this manner may not accurately describe solutions that roots typically encounter, especially when soils dry out and water is held at greater tension within the smaller soil pores. One study in the Great Smoky Mountains attempted to characterize water held at greater tension in the soil through centrifugation and showed these solutions to be different in composition than solutions from low-tension lysimeters. Contrary to expectations, centrifuged solutions had higher Ca and Mg concentration and Ca/Al and Mg/Al ratios than corresponding lysimeter solutions (Van Miegroet et al., 1990). Although these are the only such data available for the SA, and there may have been disturbance effects associated with the centrifugation technique (see Van Miegroet, 1995), they nevertheless demonstrate that caution should be exercised when discussing rhizosphere solutions.

Finally, there is the issue of Al speciation in organic versus mineral soils and its potential effect on roots. Questions that arise are: Where exactly are roots likely to experience the most deleterious effects of high Al levels? Do elevated Al levels represent the same "toxicity" in organic and mineral soils? What is the role of organic matter in complexing Al and mitigating Al toxicity? To what extent does the presence of strong acid anions diminish the complexing capacity of organics? This issue is relevant if one accepts the assumption that fine roots are preferentially distributed in the upper (organic-rich) horizons and that tree nutrition is for the most part dependent on these surficial roots. This is indeed the zone where organic matter content is high, where most of the decomposition occurs, and where Al is much more likely to occur in the form of Al-organo complexes thought to be less toxic. It is at greater depth, where the mineral soil dominates but less fine roots are present, that Al is more likely to occur in the more toxic monomeric form. Such steep gradients in soil and soil solution chemistry, which also occur under natural conditions, may in fact be a driver in the shallow rooting pattern (e.g., Fernandez, 1992). There is little

information on Al speciation in the SA and even less on the relative toxicity of the different forms.

Smithson (1990) studied the Al fractionation through extraction of soils from several high elevation sites in North Carolina, but did not evaluate how these fraction changed with soil depth. Studies that deal with spruce responses typically discuss Al in terms of total concentrations. Joslin and Wolfe (1992) recognized this limitation in their study of soil chemistry and root distribution across a deposition gradient but argued that (1) when soil solution concentrations of Al are high, most of the Al is in inorganic form, and (2) seedling studies have shown similar correlations between red spruce root growth and total Al as were found for monomeric Al. Such simplification may be valid when comparing changes in chemistry of a particular soil or horizon in response to different external influences or when evaluating changes over time at a given site. They may not be appropriate, however, when the objective is to evaluate the effect of strong acid anion inputs on the soil as a growing environment. As indicated by acid mist applications to red spruce seedlings in pot studies, the presence/absence of organic matter may be critical to seedling growth response (see discussion in Chapter 5). Most trees naturally occur in an acid environment, and many in an environment with high Al levels, yet seem to tolerate these conditions. From this observation it follows that high Al levels per se are not sufficient in explaining the effects of strong acid inputs on tree nutrition. The central question then becomes why Al occurs in a monomeric rather than organically complexed form, even in environments where the formation of the latter should be favored. Alternatively, it may be necessary to reexamine long-held opinions as to the relative toxicity of different Al forms (including those that are organically bound).

4.3.2 Nutrient Concentrations

Foliar chemistry is typically used as a measure of the nutritional status of a tree or forest. Robarge et al. (1989), Johnson et al. (1991), and Joslin et al (1992) have summarized concentrations of base cations, Al and other micronutrients in spruce and fir needles in the SA. Tables 1 and 2 represent a recent update of that summary information. For most conifers that occur in managed forests, there is usually an extensive database derived from fertilizer field trials against which to evaluate existing foliar concentrations for deficiencies. Unfortunately, no such information is available for red spruce and the only guidelines currently available are based on seedling studies conducted under controlled laboratory condition (Swan, 1971). Even less information is available for Fraser fir.

Throughout the SA, foliar Ca levels in spruce are considerably lower than those in high elevation sites in the Northeastern U.S., while other nutrient concentrations are quite similar throughout eastern high-elevation forests. In addition, several studies have found declining Ca and Mg levels and increasing Al levels with elevation and/or with an increase in cloud water deposition (Wells et al., 1989; McLaughlin et al, 1991; Van Miegroet et al., 1993; Thornton et al., 1994). Ascertaining whether low Ca and/or Mg levels are within the deficiency range is problematic in that there are no existing standards for mature trees, and foliar symptoms associated of these possible deficiencies in the SA are neither specific enough nor pronounced enough to allow a reliable diagnosis (e.g., Robarge et al. 1989). Furthermore, lack of historical data for spruce and fir precludes pre- versus post-pollution time trends. It must also be reiterated that nutrient deficiency can occur in absence of substantial anthropogenic pollution and that some high-elevation soils may

be naturally deficient in these elements (see discussion in Robarge et al., 1989).

Based on the thresholds established in Swan's study (1971), most Ca levels in current foliage (range: 2.1 - 1.0 mg/g) would appear well within the sufficiency range, whereas at some sites Mg levels (range: 0.5- 0.9 mg/g) may be approaching the moderate deficiency range for red spruce (Table 1). However if we apply stricter field standards, as suggested by McLaughlin and Kohut (1992), most foliar Mg levels and some of the Ca levels may start falling within the moderate deficiency range. Robarge et al., (1989) compared foliar levels in spruce and fir from different location within the SA to published ranges for unmanaged low-elevation forests in the Northeastern U.S. and Canada and found that some Ca, Mg Mn, and Zn levels fell below published minima. They also concluded that the foliar Al in Fraser Fir and red spruce in the SA are among the highest reported in the literature. Based on the available standards for red spruce seedlings (Swan, 1971) or recommended levels for Fraser fir Christmas trees (Shelton as cited in Robarge et al., 1989), N in spruce and fir would appear to be moderately deficient (Table 1 and 2). This is highly unlikely given that most high-elevation forests in the SA have been found to be N saturated. Robarge et al (1989) and Johnson et al. (1991) concluded that N was probably not deficient since foliar N concentrations did not typically decline in older foliage. Such decline is indicative of internal N translocation which occur when N supplies are limited. This assessment was supported by the lack of foliar N response to N fertilization at Whitetop Mt, VA (Joslin and Wolfe, 1994). Robarge et al., (1989) further concluded that phosphorus (P) and potassium (K) were well within sufficiency range.

Several researchers have used concentration ratios between elements in an attempt to detect possible nutritional shifts. This is a common practice in commercial forestry and the basis for the DRIS system (Diagnosis and Recommendation Integrated System, e.g., Hockman et al., 1989). Robarge et al. (1989) compared the Ca:Mg ratio in the SA ($\text{Ca:Mg} \approx 2$) against published ratios for healthy stands ($\text{Ca:Mg} \approx 4$) and concluded that although both Ca and Mg may be approaching deficiency, Ca limitations appear to be more stringent. Also in that same study, Ca:Al ratios in the SA were among the lowest yet reported (average $\text{Ca:Al}=11$ versus published range 24-114). This is consistent with the low value reported by Joslin and Wolfe (1994) for Whitetop Mt. ($\text{Ca:Al}=18$).

4.3.3 Response to nutrient amendments

Based on the interpretation of foliar concentrations and foliar levels, it would appear that spruce-fir in the SA are currently experiencing some base cation deficiency (Ca and/or Mg). Two fertilization studies that were conducted at Clingmans Dome (Van Miegroet et al., 1993) and Whitetop Mt. (Joslin and Wolfe, 1994), respectively, lend further support to this interpretation. In the first study, red spruce saplings below and above the cloud base received Ca, Mg, and Ca + Mg amendments for two consecutive years. Only at the high site (which was characterized by significantly lower pre-treatment Ca and Mg levels) was there a significant improvement in Ca nutrition with the addition of Ca (Ca and Ca+Mg treatment). This positive response did not persist in the second treatment year, suggesting that the trees were experiencing incipient Ca deficiency. Magnesium fertilization did not cause a significant improvement in Mg nutrition at this site, but appeared to interfere with Ca uptake. At the lower site (generally characterized by better Ca and Mg nutrition) no significant response was observed following either Ca or Mg fertilization. Based on this study, a threshold for incipient deficiency of 1.7 mg/g was proposed for Ca, while Mg levels in the range of 0.6 mg/g were considered sufficient.

A study using mature trees at Whitetop Mt. found similar trends, as well as some distinct differences in fertilizer response (Joslin and Wolfe, 1994). Pretreatment nutrient concentrations in current foliage suggested Mg (0.45 mg/g) and Zn (13.5 $\mu\text{g/g}$) were in the deficiency range and Ca (1.75 mg/g) above deficiency range according to Swan's (1971) standards. Significant foliar growth responses and a concomitant increase in foliar concentrations of Ca, Mg, Zn, and Mn followed the addition of Ca and Mg applied alone or together. Both cations appeared to exert a synergistic effect on the nutrition of other elements in mature spruce trees (including Zn). Calcium and Zn appeared most responsive to the treatments, and were considered most likely to be growth-limiting. Joslin and Wolfe (1994) proposed that the positive response to the addition of Ca was due to the indirect positive effect exerted by Ca upon root growth and elongation and by counteracting Al antagonisms.

4.4 Impacts at other Southern Appalachian non-Class I Areas

To address the question to what extent the situation described for the high elevation spruce-fir is also applicable to other forest ecosystems at lower elevations in the SA, one must compare deposition regimes, soil characteristics, and ecosystems dynamics among these systems. Research as part of the IFS at Coweeta Hydrological Laboratory in southeastern North Carolina and at Oak Ridge, TN provide an opportunity to compare low elevations with higher elevations (Swank and Crossley, 1988; Johnson and Lindberg, 1992).

Forests at lower elevations are much less impacted by cloud water deposition and annual depositions rates of N and S tend to be a considerably lower than those observed at the high elevation sites. While wet deposition rates are generally fairly comparable across low elevation sites, S input via dry deposition is variable among locations and is strongly influenced by concentrations of airborne S (Lindberg, 1992). The concentration of S is determined by the S sources within the pathway of major air masses that reach a particular location. At Coweeta, white pine and hardwood sites located at 720m elevation received around 9 $\text{kg ha}^{-1} \text{yr}^{-1}$ of S and 5-7 $\text{kg ha}^{-1} \text{yr}^{-1}$ of N. A comparable loblolly pine site at 300 m elevation near Oak Ridge, TN received 15 $\text{kg ha}^{-1} \text{yr}^{-1}$ of S and 10 $\text{kg ha}^{-1} \text{yr}^{-1}$ of N (Lindberg, 1992; Lovett, 1992). This compares to high elevation sites in the Smokies which received 36 $\text{kg ha}^{-1} \text{yr}^{-1}$ of S and 27 $\text{kg ha}^{-1} \text{yr}^{-1}$ of N.

Higher base saturation in mid- and low-elevation soils make them susceptible to soil acidification due to cation loss rather than to Al mobilization and its effect on tree health and nutrition. There is evidence of recent change in the exchangeable base capital at the Walker Branch Watershed (Oak Ridge, TN) (Johnson and Van Hook, 1989) and at Coweeta Hydrological Laboratory (Otto, NC) (Knoepp and Swank 1994) which has been attributed both to natural processes (Ca uptake by trees) and atmospheric deposition (SO_4 -induced leaching of Mg). Crucial to these atmospheric deposition effects is the rate of SO_4 deposition and the S retention capacity of the ecosystem. As long as S inputs are below annual plant requirements, S retention is close to complete and there is little danger of SO_4 -mediated leaching. With increasing S deposition, the rate of soil adsorption tends to become critical. In general, Ultisols are considered to be strong SO_4 adsorbers due to higher sesquioxide contents in these highly weathered soils and are not classified as sensitive to the deleterious effect of atmospheric deposition (see Section 4.1.1). There may be considerable variation in the adsorption capacity within this soil order (Harrison and Johnson, 1992), however, and some Ultisols at low elevation were found to incompletely retain atmospheric S inputs (Johnson et al., 1986).

From this information it appears that lower elevation sites are unlikely to show the same base cation deficiency/Al mobilization stresses that typically characterize the high elevation sites. This is likely the results of a combination of lower atmospheric deposition regimes, higher base cation reserves, and larger N and S retention capacity of these systems compared to the high-elevation spruce-fir ecosystems. It should be noted, however, that a decline in S and/or N retention capacity might result in accelerated cation leaching and lowering of the base saturation. Such base cation stripping, if occurring long enough or rapidly enough to exceed replenishment by weathering and decomposition, can ultimately result in soil conditions similar to those currently observed at the high elevation sites.

Although the current information seems to indicate that SO_4 breakthrough is more likely than N breakthrough in these lower elevation systems, especially those that have low soil N capital due to past land use or disturbances, the same impact can be expected whether caused by NO_3 or SO_4 . The streamwater observations from the Fernow Watershed Experimental Forest in West Virginia illustrate this. Since the 1970's, a progressive increase in streamwater NO_3 levels has been measured accompanied by a similar increase in Ca concentrations (Helvey and Kunkle, 1986; Edwards and Helvey, 1991). Using Stoddard's (1994) criteria, the watershed has lost most of the biological control over N retention and has reached stage 2 of N saturation, which is characterized by consistently high NO_3 levels in the stream throughout the year. Although this watershed has

4.5 Links between terrestrial and aquatic impacts

Soil processes that play a critically role in the nutritional quality of the soil solution can also impact stream water quality. Elevated concentrations of specific ions such as NO_3^- or Al^{3+} can have a direct detrimental effect on aquatic ecology. Water quality deterioration can also occur through a decline in the acid neutralizing capacity (ANC) of stream waters caused by a surge in strong acid anion concentrations (irrespective of whether they are NO_3^- , SO_4^{2-} or Cl^-) without an equivalent increase in base cation concentrations (see discussion in Reuss and Johnson, 1986). Thus, in ecosystems where elevated anion loads (from atmospheric deposition or through internal release) cause accelerated cation stripping from the exchange complex, significant soil acidification may result without causing a significant change in stream water acidity (ANC). Conversely, in waters draining terrestrial ecosystems characterized by low base cation supplies (e.g., high elevation spruce-fir) a decline in ANC would be expected with an increase in anion load without a measurable change in soil acidity.

In addition to soil properties, hydrologic flow paths determine what ions are transported from the terrestrial to the aquatic ecosystem and the timing of transport. During stormflow and snowmelt most water bypasses deeper soil horizons and is strongly influenced by the water chemistry of upper, organic-rich horizons with low SO_4^{2-} adsorption capacity (e.g., Joslin et al., 1987; Cook et al., 1994). Baseflow chemistry reflects the chemical transformations of the percolating water along its longer and slower path towards the stream. However, the solution leaving the rooting zone can undergo chemical changes such as denitrification and S reduction in near-stream zones or weathering in deeper soil strata. Consequently, not all the NO_3^- and SO_4^{2-} leached from the uplands portion of the terrestrial ecosystem necessarily ends up in the streams (see discussion Van Miegroet, 1994 and Section 4.1.1).

As discussed in Section 4.1.1, the National Stream Survey (NSS) did not include high elevation (acid) stream reaches and information on possible links between the condition of high-elevation forests and the chemistry of streams draining these areas is scarce. Data collected in the Great Smoky Mountains (e.g., Silsbee and Larson, 1981;1982;1983; Cook et al., 1994; Flum and Nodvin, 1995; Nodvin et al., 1995) substantiate (1) that SO_4^{2-} and NO_3^- are indeed the dominant anions in high-elevation streams, (2) that the ANC in these streams is generally low and declines with each anion peak as predicted, and (3) that spatial patterns in stream water NO_3^- concentration are consistent with the degree of N saturation of the watershed. However, Nodvin et al. (1995) could detect little Al to accompany these strong anion loads, suggesting additional exchanges between the solid and solution phase before reaching the stream.

4.6 Summary and Conclusions

High elevation spruce-fir ecosystems have the following characteristics that make them particularly sensitive to the direct and indirect impacts of atmospheric deposition:

- (1) Spruce-fir forests are often situated at or above the cloud base, on steep slopes and frequently on shallow soils.
 - (2) The soils are generally young with high organic matter content near the surface, low base saturation in the mineral soil, and generally low SO_4^{2-} adsorption capacity.
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- (3) The ecosystems have low N retention capacity due to high soil N contents, low C/N ratio, and low N requirements by the trees leading to high internal nitrification capacity.
- (4) The systems have experienced recent heavy mortality of the Fraser fir component further decreasing uptake and causing gap formation.

The soil solutions in these ecosystems are dominated by the SO_4 and NO_3 . The concentration of these anions is a function of the following factors:

- (1) SO_4 mostly originates from atmospheric deposition; NO_3 is derived from internal and external sources.
- (2) Cloud impact and dry deposition account for a large part of the atmospheric input;
- (3) Cloud water impact and SO_4 deposition generally increase with elevation.
- (4) Biological processes exert a strong control over NO_3 solution levels and are responsible for strong temporal fluctuations.
- (5) Stand condition greatly influences N saturation and spatial NO_3 leaching patterns; (6)

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These ecosystems may be experiencing nutrient stress (especially Ca and Mg and possibly also Zn) indicated by the lower Ca and Mg concentration in the foliage, especially with increasing elevation, and the positive response to Ca and Mg amendments. The extent to which this condition is caused by natural conditions (low base status, nitrification) or by the high atmospheric deposition rates (cation stripping, Al mobilization) is still uncertain. Because of the already low base status, further cation depletion by strong anion inputs is less likely and to this date there is no direct evidence of Al toxicity; however, Al concentrations are frequently in the range that can inhibit uptake of base cations by red spruce. Interference of base cation nutrition due to high Al levels is plausible, but there is still considerable uncertainty with regards to the relative abundance of organically complexed versus inorganic Al forms and their respective effectiveness, especially in the upper organic-rich horizons where the roots are most abundant. There are indications that foliar leaching may be an important mechanism in causing the lower foliar concentrations.

There are still a great number of uncertainties or information gaps that make regional assessment of sensitivity particularly difficult. They include but are not limited to:

- (1) the SO₄ adsorption and desorption behavior of these soils;**
- (2) the relative role of atmospheric versus internal sources of NO₃;**
- (3) reliable foliar guidelines for mature trees against which current base cation levels can be compared for deficiencies;**
- (4) the chemical characterization of rhizosphere solution (versus those collected with lysimeters); and**
- (5) the Al speciation within the rooting zone.**

Nevertheless, despite these uncertainties, the patterns in soil solution, stream and foliar chemistry are rather consistent across the experimental and survey sites, and strongly suggest that the patterns discussed in this report are widespread across the SA.

Chapter 5

Physiological Diagnosis of the Condition and Causes of Declining Health of Red Spruce and Fraser Fir in the Southern Appalachians

The health of forest trees is determined primarily by the balance between the chemical capture of solar energy by foliage and the management of that energy to minimize the effects of environmental stress. Energy capture is determined by uptake and utilization of carbon, water, and nutrient resources. Allocation of that energy is reflected in several ways that indicate how well the tree is in balance with the demands of its environment. Those include growth rate, amount and visible appearance of foliage, resistance to insects and disease, and ultimately, the life span of the tree. While the visible condition may provide a clear signal that the trees are significantly stressed, tree condition may have little diagnostic value unless we understand what processes have been altered to produce the imbalance and under what conditions those processes were altered.

It is the analysis of processes that has been the focus of the physiological research to evaluate the causes of the declining growth and condition of spruce and fir at high elevation sites in the Southern Appalachians. However, the focus of those process-level studies has been heavily guided by the types and patterns of responses that have been significant enough to be measurable at the level of whole trees. Thus, the feedback between observed patterns of response and sequences of processes capable of producing those patterns has been an important feature of this research. The research began with characterizing patterns of growth and tissue chemistry of mature spruce trees and proceeded to exploring the roles of acidic deposition and other factors in producing those symptoms. Contrasts between the tree condition in the Southern Appalachians and tree condition in the intensively studied Northeast have been an important part of this work.

5.1 Mature Tree Growth Patterns

A slowdown in radial growth of red spruce trees has been a significant early indicator of alterations in forest health in both the Northern and Southern Appalachians. However, a distinctive feature of the Southern Appalachians has been the absence of a growth decline below about 1520 m (5000 ft) in elevation (Figure A.)

Much of the analyses of tree growth patterns has centered around evaluating the role of natural stresses, climate and competition in explaining observed growth decreases at high elevation sites. Cook and Zedaker (1992) provided a comprehensive review of dendroecological studies and the issues involved in evaluating the relative roles of climate, competition, and atmospheric pollution in observed growth declines. These conclusions as well as those that McLaughlin et al. (1987) reached in an earlier study indicate that the decline began in the late 1950s to early 1960s in the North and was delayed 5-10 years in the South.

Modeling of red spruce growth responses to annual variations in temperature and rainfall was used to evaluate more recent growth trends. Growth responses to climate at a wide range of test sites across the range of red spruce were examined during the period 1900-1940 to evaluate historical

patterns. Results indicated that trees in both the Northern and Southern Appalachians began to grow more slowly than expected in relationship to climate beginning around 1960 (Figure B). Stand stocking levels at sites from which sampled trees were located were also below levels at which competition would have been a significant contributor to the observed patterns. Thus, based on historical data on expected red spruce growth responses to natural stresses, it was concluded that neither unusual climate nor stand competition appeared to have played a major role in initiating the growth limiting stress in the early 1960's (McLaughlin et al., 1987; Cook and Zedaker, 1992). These results indicated that other regional scale stresses such as air pollution may have played a role in altering the balance of red spruce with its natural environment.

5.2 Changes in Wood Chemistry

A second approach to pattern analysis also involved analysis of tree cores. In this case the chemistry of the cores was analyzed to determine if changes in the trees' chemical environment had occurred. Wood chemistry can serve as an historical monitor of the changing chemistry of soil solutions because wood incorporates chemical elements such as calcium and aluminum taken up from the soil during each year of wood formation. The successive years of wood formed then provide a record of the changes in availability of these elements from year to year (Bondietti and McLaughlin, 1992 and Figure A).

In wood chemistry studies, changes in aluminum and calcium were of particular interest, because acidic deposition acting on naturally acidic soils increases the availability of aluminum relative to that of calcium in the soil solutions (see Section 4.1.2). Analyses of cores from red spruce and other species sampled from high elevation sites in the GSMNP indicated that an increase in aluminum relative to calcium began to occur around the 1950's, at about the same time that emissions of sulfur and nitrogen oxides began to increase in a predominantly upwind direction from the GSMNP (Bondietti et al., 1989). Thus, the tree ring chemistry signal indicated that trees began to experience chemical changes in their soil and atmospheric environment just prior to the period when growth at high elevation sites began to decrease at an unusual rate.

5.3 Physiological Measurements of Red Spruce Saplings

The growth patterns shown in Figure A indicate that mature red spruce trees growing at high elevation sites had lost some growth capacity beginning in the early 1960s. The absence of significant changes at lower elevations afforded an opportunity to identify and evaluate the types of physiological processes that might be responsible for observed differences in growth rates. Saplings were used in these studies in order to allow researchers access to the foliage for the required measurements. A major focus of this work was an evaluation of any changes in the rates of carbon assimilation in photosynthesis and rates of energy loss in respiration.

McLaughlin and Kohut (1992) reviewed the series of published field and laboratory studies that have demonstrated that sapling trees growing at high elevation sites in the GSMNP have experienced a loss in the efficiency of carbon metabolism, expressed as a drop in energy production in photosynthesis (P) relative to energy consumed in respiration (R). This was expressed as a ratio, the P:R ratio, and used as a physiological indicator of a change in carbon metabolism. The drop in P:R shown in Figure C, that was found at the highest elevations sampled on each of three mountains, was associated with increased levels of foliar aluminum, decreased

levels of foliar calcium and reduced soil Ca:Al levels. These gradients also agreed with the pattern of increased cloud exposure and increased acidic deposition found at high elevation sites.

Alterations in carbon metabolism were mostly a function of increased dark respiration rates, which were significantly increased by decreasing foliar Ca and high soil Al (McLaughlin et al., 1991). Such changes would be expected to lead to reduced carbohydrate availability and reduced growth rates of trees growing at high elevation sites.

The discovery that calcium and aluminum availability were significant components of physiological changes at high elevation sites was important to understanding how increased acidic deposition could alter plant health as exposure levels increased with increasing elevation. Calcium is an essential element for many plant processes, notably growth of cell walls, extension growth of root and shoot meristems, membrane stability, and resistance to a wide variety of plant diseases (Bangerth et al., 1979).

There are two primary mechanisms by which acidic deposition can reduce the availability of calcium to plants: (1) through the soil by mobilizing aluminum, which interferes with Ca uptake by roots; and (2) through contact with foliage leading to leaching of Ca and other base cations. Studies of soil solution chemistry in GSMNP have shown that aluminum levels were frequently at levels high enough to interfere with calcium uptake (see Section 4.1.2). Foliar Ca has also been shown to decrease through leaching by the acidity in ambient cloud exposures (Joslin et al., 1988; Thornton et al., 1994). Joslin et al. (1988) followed the changing chemistry of throughfall passing through a red spruce canopy in relationship to the acidity of cloud moisture impacting the canopy, and determined that the greatest losses of foliar cations such as calcium occurred when the acidity of the clouds was highest. Thornton et al. (1994) found that seedlings exposed to acidic clouds at Mt. Rogers had reduced levels of foliar Ca compared to those protected by chambers.

5.4 Controlled Exposure Studies

The association of changes in tree physiology or tree condition with increasing elevation provides circumstantial evidence of a causal linkage to increasing levels of acidic deposition, but this linkage is weakened by the fact that other environmental variables, such as wind, snow, and freezing temperatures also increase with increasing elevation. Thus, it becomes important to test the effects of acidic deposition specifically. This has now been done with both greenhouse and field experiments.

In greenhouse studies, the effects of ambient range acidic rain and mist exposure levels was measured on two-year old seedling trees growing on organic and mineral soils collected from a high elevation red spruce site (McLaughlin, et al. 1993). Both seedling growth and seedling physiology were adversely affected by acidic exposures (Figure D), and the foliar symptoms produced, including both reduced foliar Ca levels and reduced P:R, paralleled responses observed with increasing exposure to acidic deposition in the field. A second study by Thornton et al. (1994) evaluated the effects of exposing red spruce seedlings to combined and separate levels of ambient ozone and acidic mist at Whitetop Mountain. Responses to acidic deposition included reduced levels of foliar calcium as noted above, increased dark respiration, and a loss of cold hardiness (Thornton et al., 1994). Significant growth losses for the treated seedlings, which were potted in soil from the organic soil horizon, were not noted. The lack of a mineral soil

component in the potting medium in which these seedlings were grown may be an important point of distinction. A lack of growth responses has also been noted in other studies in which a predominantly organic substrate has been used as a potting medium (Kohut et al., 1990). Joslin and Wolfe (1992) have reported Ca:Al levels to be approximately two thirds lower in the mineral soil than in the upper organic horizon at the Whitetop site. Reduced growth of fine roots in deeper mineral soil horizons was also noted across a gradient of increasing exposure to acidic clouds at the Whitetop site (Joslin et al., 1992). Growth into the mineral soil was also inhibited in response to acidic deposition in pot studies (McLaughlin et al., 1993). Thus the soil-derived limitations in supply of calcium and other nutrients are an important component of expected response to acidic deposition.

5.5 Increased Sensitivity to Winter Damage

An apparent increased sensitivity of red spruce to damage by low winter temperatures has been an important early visual symptom of the decreasing health of northern spruce forests (Johnson et al., 1988). Increased sensitivity to low temperatures has been tied to exposure of branches to acidity in ambient mist exposures (Vann et al., 1992). In the controlled chamber studies at Whitetop Mountain, a 3°-5° C loss in cold tolerance was noted in seedlings exposed to ambient cloud acidity (Thornton et al., 1994). Winter temperatures in the Southern Appalachians do not typically drop to levels at which a loss of hardiness of this magnitude would be expected to be a significant problem. However, a much less severe, but cumulative, winter discoloration of needles has been found in southern red spruce (Andersen et al., 1991). The symptoms produced closely resemble those attributed to winter damage in Europe and in the Northeastern U.S. (Liedeker and Klein, 1989). Its significance to the red spruce physiology and growth is still unclear.

5.6 Fertilization Studies

Results of exploratory studies across deposition gradients in the field as well as controlled exposures to acidic deposition in a greenhouse and in chambers provided converging evidence that acidic deposition was altering physiological performance of trees through changes in availability of calcium and possibly other nutrients. To test for the specificity of the response under field conditions, fertilization studies were initiated. Evidence of the importance of changes in calcium availability has now come from three different approaches: greenhouse studies with saplings, field studies with saplings, and field studies with mature trees. In greenhouse studies, the effects on acidic deposition (combined exposure to mist at pH 3.0 and rain pH at 3.8) on foliar Ca, carbon assimilation (P:R), and growth were reduced by calcium and magnesium addition to the potting medium (McLaughlin et al., 1992). In this as well as in subsequent studies in the field, the greatest effect was produced by calcium, with magnesium addition causing minimal positive or more typically, antagonistic effects, relative to those of calcium. Additional confirmation of the role of added calcium in overcoming calcium deficiency has been demonstrated by Ca fertilization studies that stimulated red spruce needle growth of saplings on Clingmans Dome in GSMNP (Van Miegroet et al., 1993) and growth of foliage of mature red spruce trees at Whitetop Mountain (Joslin and Wolfe, 1994). More recently, stimulation of stem and twig growth, increased foliar calcium levels, and increased P:R ratios of foliage have been demonstrated with Ca fertilization of red spruce saplings at a high elevation site on Clingmans Dome (Wullschleger et al., in preparation).

5.7 The Role of Nitrogen

Nitrogen is a growth-limiting nutrient in many forest systems, but in the high elevations of the Southern Appalachians, soils are nitrogen saturated (see Section 4.2.3). Foliar nitrogen levels are not high, indicating that the high soil nitrogen levels are not leading to increased nitrogen uptake. Fertilization studies with mature trees (Joslin and Wolfe, 1994) have confirmed that added nitrogen was not taken up by mature trees. The relatively low nitrogen uptake efficiency by these forests raises the question of how much impairment of root function by the high Al:Ca ratios in soil solutions (Johnson et al., 1991) may have diminished nitrogen uptake potential of roots thereby contributing to nitrogen saturation. Recent research using the ratios of two stable isotopic forms of nitrogen (N^{15} and N^{14}) in soil solutions and in foliage offers promise as a tool to monitor the nitrogen dynamics of the high elevation ecosystems and to evaluate the role of reduced nitrogen uptake by vegetation on N accumulation in soils (Garten and Miegroet, 1994).

5.9 Physiological Measurements on Fraser Fir

The presence and dynamics of the balsam woolly adelgid is unquestionable evidence that the adelgid plays a major role in the death of large numbers of Fraser fir in the Southern Appalachians (see Chapter 2). However, it is also important to consider the role of predisposing factors in increasing susceptibility of stressed forests to pathogens (Manion, 1981). Several bits of evidence warrant examination in evaluating factors influencing resistance or susceptibility of Fraser fir to adelgid-induced mortality. First is the fact that some fir trees have a resistance to this introduced pathogen and can survive attack. Within the Southern Appalachians the most resistant population has been at Mt. Rogers in southwestern Virginia, where mortality has been very light and evidence of wound healing pockets in the bark indicates that resistance mechanisms do exist (Eagar, 1984). As the northernmost extension of the range of Fraser fir, the apparent resistance at Mt. Rogers could be due to genetic differentiation within the Fraser fir population. However, there is now increasing evidence of attack in the Mt. Rogers population primarily at low elevation sites (Dull et al. 1988) suggesting that site-related factors may influence sensitivity.

Significant temporal and spatial variability in susceptibility of the closely related balsam fir (*Abies balsameri*) has been recorded across its eastern range since the adelgid was introduced around 1900 (Timmel, 1986). Mortality of this species has been significant in the Canadian Maritime Provinces, particularly in the mid 1980's. Tree vigor, bark characteristics, and the formation of compression wood are apparently related to resistance of fir to the adelgid (Eagar, 1984; and Timmel, 1986).

Several seemingly converging lines of inference have led to examination of the role of acidic deposition on calcium supply and physiological responses potentially related to resistance. First, there is dendroecological evidence that Fraser fir in GSMNP began a growth decline around 1960 in parallel to that experienced by red spruce (McLaughlin et al., 1984). Second, recent physiological measurements on Fraser fir indicate a decline in P:R ratio of foliage with increasing elevation (and acidic deposition exposure) parallel to that documented for red spruce (Stone et al., in preparation). Third, preliminary contrasts between Mt. Rogers and Clingmans Dome indicate higher foliar and soil calcium and lower aluminum levels at the high elevation Mt. Rogers site than found for more sensitive populations at Mt. Rogers low elevation or either high or low elevation GSMNP sites. There was also no significant P:R gradient with increasing elevation at Mt. Rogers

(Stone, in preparation). Finally, the known role of calcium in disease resistance, including wound repair (Bangerth et al., 1989) and formation of lignin, which is a major constituent of compression wood, make calcium supply a likely modifier of resistance of Fraser fir to fatal adelgid infestation.

Collectively this evidence suggests that acidic deposition, through its effects on calcium availability, may be playing a role in the apparent low resistance of Fraser fir to adelgid attack in the Southern Appalachians. Current investigations of Fraser fir ecophysiology are under way to further evaluate the role of calcium nutrition and soil solution chemistry on growth and physiological responses of Fraser fir.

5.10 Summary

The physiological studies discussed above have played an important role in identifying a wide range of essential plant processes that are altered in response to levels of acidic deposition that are now occurring at high elevations in the Southern Appalachians. The responses include reduced uptake and increased leaching of a growth-limiting nutrient, calcium, reduced net carbon assimilation, and reduced growth. Patterns of tree ring chemistry of red spruce and other high elevation species indicate that altered calcium and aluminum availability began to occur coincidentally with both increased regional emissions of SO₂ and NO_x, and patterns of reduced radial growth of mature red spruce trees at high elevation sites.

Figures

Figure A. Composite chronology of red spruce growth, tissue chemistry, and atmospheric emissions: Southern Appalachians. Data and sources are as follows reading top (1) to bottom (5) on left, then top to bottom (8) on the right. (1) mean ring-width chronology of 15 canopy dominant trees near summit (Adams et al. 1990); (2-3) high- and low-elevation ring width chronologies, 15 trees each (McLaughlin et al. 1987); (4-5) ring-width chronologies from two mid-elevation 40-year old Norway spruce stands collected within 10 km of Mt. Leconte, NC, samples (Adams et al. 1990); (6) aluminum/base cation ratios (Bondietti et al. 1989); (7) wood chemistry:cation trend (Bondietti et al. 1990); (8) regionally averaged emissions of SO₂ and NO_x in tons/mi²/y in south and southwest quadrants within a 900 km radius of the Great Smoky Mountains of eastern Tennessee SW of eastern Tennessee (McLaughlin, unpublished data). Dashed vertical line on each graph indicates the year 1960 as a common reference point. After Johnson et al. (1992).

Figure B. Comparison between predicted (dashed line) and actual standardized annual ring widths (solid line) for 10 sites representing differences in latitude, elevation, and aspect. Periods when predictions based on a model developed over the 1900-1940 interval exceeded actual growth are darkened.

Figure C. Mean ratios of light-saturated photosynthesis (PS) to dark respiration (RS) for current red spruce shoots from three elevations on three mountains in the Great Smoky Mountain National Park. Data points are means (± 1 SD) of 6-10 trees per site measured in August of 1988.

Figure D. Comparative responses of physiological and growth responses of red spruce after 12 weeks (foliar chlorophyll) to 18 weeks (remaining variables) of exposure to acidic deposition.

Chapter 6

Approaches to an Integrated Assessment of Effects of Acidic Deposition on Terrestrial Ecosystems in the Southern Appalachians

6.1 Recommendation

Nitrogen and sulfur dynamics must be considered in the assessment of terrestrial effects of atmospheric deposition and the sensitivity of vegetation types to deposition changes in the Southern Appalachians. The NAPAP assessment for the southeastern U.S. primarily focused on the effect and retention of S deposition in low-elevation areas. Since then several studies in the high-elevation spruce-fir have indicated that the fate and retention of N may be equally if not more important than S dynamics in determining atmospheric deposition effects on soil processes. Recently two modeling efforts have incorporated N dynamics into ecosystem process models.

In order for models to be effective as assessment tools they must be geographically extensive. At the same time they should reflect our current understanding of the key processes in the ecosystems of concern. The combination of the following models is proposed for the regional assessment of atmospheric deposition effects in the Southern Appalachian region. We feel that SAMI would benefit substantially from the use of the three models described below. Preliminary budget estimates indicate that all three could be used, if funding were available in the upper end of the range suggested (\$125K and up). Benefits of using three separate modeling efforts and combining the results into the Integrated Assessment include a more robust evaluation of the effects of acidic deposition on terrestrial ecosystems. Utilization of two separate but similar models (MAGIC and PnET) will help reduce the uncertainty that is inherent in application of models to complex ecological processes. The third model, NuCM, provides a more detailed analysis of soil processes and can be used to better interpret the output from MAGIC and PnET. If funding is limited, we suggest that SAMI solicit proposals from the principle investigator for each model and evaluate which one will best accomplish the purpose of the Assessment.

There are currently no models available that link changes in nutrient availability to response of vegetation. The PnET model does simulate forest growth in response to changes in resources, but mainly focuses on carbon/nitrogen/climate interactions. One approach would be to use the soil-based models to simulate changes in nutrient and aluminum availability in red spruce forests and compare these changes with known alteration of physiological processes (mainly the research of Sandy McLaughlin). In doing this it is important to consider the types of changes with respect to the timing of likely shifts in processes affected. For example, changes in soil solution chemistry are rapid and influenced by deposition, whereas changes in the nutrient content of foliage is slower and reflects both soil changes and direct atmospheric effects (e.g., foliar leaching). Changes are slowest in soil nutrient pools. It will also be important to consider the response of foliage to deposition, which appears to be able to cause changes in small pools of foliar nutrients that are very important to physiological processes that influence carbon cycling. Fityproc99 1 7 Tin the n

Terrestrial and Aquatic Effects. The current level of scientific understanding indicates that the principle effect of acidic deposition is on soil chemistry. Alteration of soil chemical properties changes the availability of nutrients and toxic elements on soil exchange sites and in soil solution. These changes can impact the vegetation growing in sensitive areas and alter the chemical characteristics of drainage waters. Both MAGIC and PnET will provide information on watershed output to streams. PnET differs from MAGIC in that PnET provides a more detailed simulation of plant processes and how those processes are impacted by changes in nutrients. PnET can be run within a GIS, thereby providing an assessment of responses to changes in atmospheric deposition over the entire SAMI region. However, this application may be limited by appropriate GIS data layers (or the cost of generating them).

6.2 Proposed Collaborators

MAGIC, MAGIC-WAND and MERLIN were developed by B. J. Cosby at the University of Virginia. The structure of the different models is such that they are mutually compatible and require similar data inputs (e.g., soil parameters, hydrology).

MAGIC and MERLIN are currently being calibrated for the IFS spruce Tower site and the Noland Divide Watershed near Clingmans Dome, NC, in Great Smoky Mountains National Park through collaboration between B.J. Cosby (University of Virginia) and H. Van Miegroet (Utah State University) with funding from USEPA. This is part of a larger effort to apply MERLIN to a number of areas in the eastern U.S. where N saturation is of concern. The application of MERLIN to a high-elevation spruce-fir site, known to be N-saturated, will provide much the basic information that is needed to drive the N dynamics in MAGIC and MAGIC-WAND on a regional level.

John Aber and several associates at the University of New Hampshire have done most of the work on PnET and linking it with GIS data in order to conduct regional assessments. Charles Driscoll at Syracuse University is currently working with Aber et al. to continue to improve the biogeochemistry and soil components of PnET-CN/CHESS. They are currently using PnET for both research and assessment purposes in the Adirondacks and New England. Both Aber and Driscoll are interested in application of PnET-CN/CHESS in the Southern Appalachians.

Dale Johnson, located at Desert Research Institute, Reno, NV, has applied NuCM to several sites in the Smokies and other sites in the Southeast. NuCM has been fully calibrated for red spruce stands in the Smokies.

6.3 Description of the Models

6.3.1 The MAGIC Models

MAGIC (Model of Acidification of Groundwater in Catchments)

MAGIC is a process-based hydrochemical model that combines a number of key processes lumped at the catchment scale to simulate soil and surface water chemistry. It simulates the behavior of the major ions in response to atmospheric deposition, and predicts pH, alkalinity, and ion concentrations (including Al and base cations) in surface water draining the catchment. It also

simulates changes in base saturation with time and in response to the atmospheric deposition regime. Key processes included in the model are: sulfate adsorption, cation exchange, dissolution and precipitation of Al, and dissolution of organic and inorganic carbon. The soil-soil solution equilibria equations describing these processes are assumed to simultaneously govern soil solution chemistry. Mass balance equations quantitatively describe the catchment input - output relationships in which fluxes of major ions are assumed to be governed by weathering and runoff.

Although the model was originally developed to predict fresh water responses, the soil component of the model also allows the assessment of terrestrial effects at the level of the soil. The model can be used for intensive site-specific simulations as well as for regionally extensive assessment of atmospheric deposition effects on water chemistry. It has been used to model and predict fresh water responses under NAPAP. Currently, the model is also successfully used in Europe to assess soil responses to atmospheric deposition, including changes in soil base saturation and Al/Ca ratios in the soil water. The 1996 NAPAP assessment for the eastern U.S. will use information of soil component of MAGIC for terrestrial effects.

Input variables:

Deposition: Atmospheric input of major ions (including SO_4 , NH_4 , NO_3 and base cations)

Soils: The spatial heterogeneity in soil properties within the catchment is lumped to one set of soil parameters per soil layer. In its simplest form, the model considers one (uniform) soil layer, but when needed or where appropriate a two-layer configuration can be implemented. For each of the soil layers, the following input information is needed:

Physical parameters: soil depth, bulk density

Chemical parameters: cation exchange capacity, S adsorption capacity (extrapolated from specific studies), CO_2 and organic acids.

Hydrology: water input and proposed major pathways (NOTE: the hydrology used in this model is a simple flow-through design; how the water is partitioned along the different pathways can be changed at any point within the model to better reflect field conditions)

Output from the model: Soil solution - major ions in the soil solution

Model limitations: MAGIC uses an extremely simplified representation of N retention, with net N retention (the integration of all biological processes) proportional to NO_3 and NH_4 deposition input. The model does not allow for internal release or transformation of N within the catchment.

Regional assessment of N deposition effects, and especially predicting the onset of N saturation requires an integrated understanding of the processes that govern N dynamics and N retention in forest ecosystems. Consequently, for a more accurate description of the N dynamics, other models should be used in conjunction with MAGIC.

MAGIC-WAND (MAGIC with Aggregated Nitrogen Dynamics)

MAGIC-WAND maintains the full S-based chemistry of MAGIC and explicitly incorporates the major terrestrial N fluxes and considers both the inorganic forms of nitrogen separately (NH_4 and NO_3). The primary inputs to the systems are atmospheric deposition of N and mineralization of organic N present in the catchment, and provisions are made for other forms of N input such as N fertilization or N fixation. Nitrogen losses from the system include denitrification and runoff of

inorganic N in the soil and stream water. In addition to external N inputs, the key N sources and sinks incorporated in the model are mineralization, nitrification and plant uptake. These processes are simply used as input sequences to the model, no pools of N accumulation are dynamically considered and there is no feedback between the ecosystem N status and N dynamics in the catchment. MAGIC-WAND is currently used in Europe within the context of critical loads assessment. It lends itself to regional assessment to the extent that input variables are available or can be estimated.

Input variables:

Deposition: External input of major ions (including SO₄, NH₄, NO₃ and base cations)

Soils: same model structure as MAGIC

Hydrology: same flow- through structure as in MAGIC

N fluxes: N mineralization and equations for nitrification and plant uptake; the time series for these processes must be specified a priori (i.e., there are no internal feedback mechanisms incorporated into the model)

Output from the model: **Same as MAGIC**

MERLIN (Model of Ecosystem Retention and Loss of Inorganic Nitrogen)

MERLIN is a process-oriented model for simulating the N dynamics within the forests and evaluation of N saturation and NO₃ breakthrough. The model considers two basic components (1) plants, which consist of active/responsive (foliage) and structural biomass (wood) and (2) soil that consists of a labile (litter) and refractory (humus) compartments. This model allows internal feedback mechanisms; N fluxes are not determined *a priori*, but are calculated from internal state variables. The carbon pools and fluxes are set externally to the system and are used as drivers for the N dynamics. Nitrogen pools and fluxes are linked to the C dynamics by the C/N ratio in the different ecosystem pools. The transfer between compartments is regulated by ecosystem processes such as plant uptake, litterfall, mineralization, immobilization, nitrification, and organic matter decomposition and humification. Key to the MERLIN model is that all N fluxes are assumed to be governed by the C/N ratio in the different ecosystem compartments. The model consists of (1) a bookkeeping procedure that tallies inputs and outputs of C and N to ecosystems and internally among the four major ecosystem components and (2) a series of simultaneous processes that regulate inputs, outputs and fluxes of C and N among the different compartments. The physical characteristics of the soil layers and the hydrological structure of the model are the same as in MAGIC.

MERLIN is currently being calibrated for several European sites where N deposition inputs have been experimentally altered (increased and decreased) as part of NITREX (Nitrogen Saturation Experiments). The model is data intensive and therefore lends itself better to site-specific modeling in a few sites where sufficient data are available. However, the model provides a mechanistic representation of N dynamics and helps to identify the key processes in N retention. As such it can be used to refine the representation of N dynamics in MAGIC-WAND and MAGIC.

Input variables:

Deposition: Wet and dry deposition of NH_4 and NO_3

Soils: same model structure as MAGIC

Hydrology: same flow-through structure as in MAGIC

C fluxes: time sequences for biomass accumulation, litterfall, decomposition, and forest floor accumulation (within time frame of one forest generation)

Ecosystem pools: current C and N pools; C/N ratio of pools; some (low level of uncertainty) can be set a priori, others are calculated during the model runs to make N and C dynamics internally consistent

N fluxes: parameters for equations that describe the N dynamics (adsorption, uptake, mineralization, nitrification)

Output from the model:

N pools: changes in N content of plant and soil compartments.

N fluxes: mineralization, immobilization, plant uptake, N return to forest floor via litterfall, and NH_4 and NO_3 discharge.

6.3.2 The Nutrient Cycling Model (NuCM)

NuCM depicts nutrient cycling at a stand level, where the ecosystem is represented as a series of vegetation and soil components. The model provides for one generic conifer and one generic deciduous species of specified biomass and nutrient concentration (foliage, branch, bole, roots). The model also provides for an overstory that can be divided into canopy, bole, and roots. Tree growth in the model is a function of user-defined stand developmental stage and the availability of nutrients and moisture. The soil includes multiple layers (up to ten), and each layer can have different physical and chemical characteristics. 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. The processes that govern interactions among these pools include user-specified rates for decay, nitrification, anion adsorption, cation exchange and mineral weathering.

Input variables:

Deposition: Atmospheric input of major ions (including SO_4 , NO_3 , and base cations)

Soils: Soil chemistry characteristics, soil temperature data, soil chemistry equilibrium constants and soil process rate coefficients (e.g., organic matter decay, nitrification).

Vegetation: Biomass characteristics and growth coefficients

Output from the model:

Available nutrients in soil strata and vegetation pools

Soil solution chemistry

NuCM has recently been applied to the Tower Site in Great Smoky Mountains National Park to

evaluate the response to changes in S and N deposition. Results indicate that old-growth red spruce forests in the Southern Appalachians are extremely sensitive to changes in Al^{3+} and soil base cation pools as a result of changes in S and N deposition. The simulations also showed that soil solution Ca/Al ratio was much less sensitive to changes in deposition than soil solution Al^{3+} , a finding that has significant implications for mitigation strategies. NuCM could be used by SAMI to assess the effects of 1) changes in base cation deposition and 2) the effects of stand age and growth rate on nutrient fluxes.

6.3.3 The PnET Models

The PnET family of models are generalized, lumped-parameter models that simulate carbon, water, and nutrient interaction in forest ecosystems. PnET operates at a monthly time step and has no specified spatial dimension. A unique feature of PnET is the ability to run it in a GIS and simulate response to acidic deposition over large regional areas. The PnET models are designed to require relatively few and generally available inputs, to run quickly, and to produce outputs that can be validated against available field data. The input requirements are limited so that the models can be run with the types of summary data generally available with a geographic information system (GIS). Input requirements relate to physiological processes of photosynthesis, respiration, allocation, senescence, phenology and decomposition. Whenever possible, generalized relationships are used so that many parameters remain the same for all runs within a broad vegetation type. This means that not all inputs need to be measured for all sites. It is possible to use generic files for broad-leaved deciduous and needle-leaved evergreens, along with site-specific climate drivers, and obtain very reasonable predictions of carbon and water balance.

There are four versions of the PnET model, each incorporating algorithms from the previous version. PnET-Day is a daily time step model of forest canopy carbon balance. It requires data on total canopy biomass, foliar nitrogen concentration and leaf specific weight, and produces daily estimates of gross and net photosynthesis and transpiration. It does not contain a full water balance so that soil-based water stress is not considered. PnET-Day has been validated against daily carbon flux data measured by eddy correlation at the Harvard Forest, Petersham, MA (Aber et al. 1996a).

PnET-II is a monthly time step model that incorporates the photosynthesis routines from PnET-Day and also includes respiration, including total soil respiration, allocation and a full water balance. There are no litter production or explicit decomposition routines, so the carbon cycle is not completed. The soil respiration term is an empirical one that allows estimates of total carbon balance under undisturbed conditions. This version has been validated against data on carbon gain, net productivity and water balance for both Hubbard Brook and Harvard Forest (Aber et al. 1995).

PnET-CN is a monthly time step model and builds on PnET-II by adding litter production and decomposition routines, as well as adding N cycling to all processes. Because of the large size and slow turnover rate of the wood and soil pools, PnET-CN requires some explicitly coded information on land use and atmospheric deposition history. Validation against data from Hubbard Brook and the Harvard Forest has been carried out (Aber et al. 1996b)

PnET-CN has been combined with the CHESS model of soil chemistry to produce a full

biogeochemical model for forest ecosystems (Figure E). CHES is an equilibrium chemistry model that simulates abiotic soil processes such as cation exchange, adsorption and solution speciation. Chemical equilibrium calculations for a given model timestep are made based on inputs to a soil layer from atmospheric deposition, drainage inflow or mineralization. This combined model has been applied to an extensive data set from Solling, Germany, as part of a multi-model comparison (Postek et al. 1995).

Input variables:

Vegetation: physiological processes of photosynthesis, respiration, allocation, senescence, phenology and decomposition. Parameters for broad-leaved deciduous and needle-leaved evergreen can be used for regional simulations.

Climate: Either as input from local datasets or derived from GIS.

Deposition: Current and future deposition levels. Capability exists to generate regional deposition patterns from GIS data and use this as input into the model.

Soil: water holding capacity, chemical characteristics, and equilibrium constants for reactions of interest.

N fluxes: parameters for equations that describe the N dynamics (uptake, mineralization, nitrification, plant use).

Output from the model:

Forest growth (net primary production of foliage, wood and fine roots), hydrology, element cycling soil and drainage water chemistry.

After appropriate validation in the Southern Appalachians, PnET-CN/CHES can be used to predict the changes in nutrient availability and possible impacts on tree productivity. There are data from several studies in the Southern Appalachians that provide both the input parameters for PnET (Integrated Forest Study, Coweeta, Fernow Experimental Forest) as well as validation. Important GIS data include elevation and vegetation type. These data are used to calculate climate and deposition variables as a function of latitude, longitude and elevation based on the approach describe in Ollinger et. al (1993 and 1995).

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Glossary of Soil Terms

The following are Soil Orders - the broadest classification of soils used in the U. S.

Inceptisols: Young soils which form fairly quickly and contain horizons based on alteration of parent materials (in contrast to extreme weathering). In the mountains, the relatively frequent tipping over of trees (up-rooting) results in development of Inceptisols.

Spodosols: Soils which have a spodic horizon, a subsurface zone of accumulation of organic matter and aluminum and/or iron oxides. The spodic horizon is usually located under a highly "bleached" horizon. Leaching of organic acids is primarily responsible for development of Spodosols.

Histosols: Soils with a very high organic matter content. These soils are usually associated with bogs or other water-saturated environments. In the mountains, Histosols might be found in high elevation area that do not have mineral soil, but only organic matter over bedrock.

Ultisols: Highly weathered soils found on old and stable land surfaces in moist, warm and subtropical areas of the U. S. They have a subsurface horizon that is high in clay content which is also high in aluminum and/or iron oxides.

Other soil terms:

Base saturation: The percentage of cation exchange sites occupied by exchangeable base cations (Ca, Mg, K, Na).

Cation exchange capacity: The total amount of exchangeable cations that can be adsorbed by a soil.

Mineralization: The conversion of an element from an organic to an inorganic form by microbial decomposition.

Nitrification: A biological process in which ammonia is oxidized to nitrate. This is often the second phase of nitrogen cycling within the soil following mineralization which produces ammonia from organic nitrogen.
