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# Impact of harvesting and atmospheric pollution on nutrient depletion of eastern US hardwood forests

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## Abstract

The eastern hardwood forests of the US may be threatened by the changing atmospheric chemistry and by changes in harvesting levels. Many studies have documented accelerated base cation losses with intensive forest harvesting. Acidic deposition can also alter nutrient cycling in these forests. The combination of increased harvesting, shorter rotations, and more intensive harvesting, along with the potential for N and S saturation due to changing atmospheric chemistry in the eastern US, raises concerns about the long-term productivity of these commercially important eastern hardwood forests. We review the literature describing the effects of intensive harvesting and acidic atmospheric deposition on budgets of base nutrients which presents evidence that the ambient levels of N and S deposition are leading to N and S saturation and elevated base leaching from the soil in some eastern forests, and we discuss potential concerns for long-term productivity. We also discuss criteria and indicators for monitoring sustainability of the soils of these forests. 

*Keywords:* Acidic deposition; Base cation depletion; Calcium; Forest harvesting; Sustainable productivity

## 1. Introduction

The ability to predict forest productivity over a long term is part of the challenge of forest management. Changes in the atmospheric environment, including increasing atmospheric carbon dioxide (Strain and Thomas, 1992) and nitrogen (N) deposition (Galloway et al., 1995), decreasing emissions of sulfate and particulates (cf. Hedin and Likens, 1996) complicate this task. Concurrent with these changes in the atmospheric environment are changes in demand for wood and other forest resources. Because of decreased harvesting on federal lands in the western US, demands in other parts of the US have increased, most notably the northeast and the southeast (Fig. 1). The future demand for wood products will be met from private forest lands, which make up 75–90% of the ownership in the eastern US and most of which is nonindustrial private land. The majority of such forests in the eastern US are extensively-managed hardwood forests.

In intensively-managed plantation forests, species composition and competition are manipulated through selection of improved planting stock of known genetic background, control of competing vegetation, and through alteration of the nutritional environment. This allows forest managers some flexibility to adapt to site quality changes as they occur. However, in the natu-

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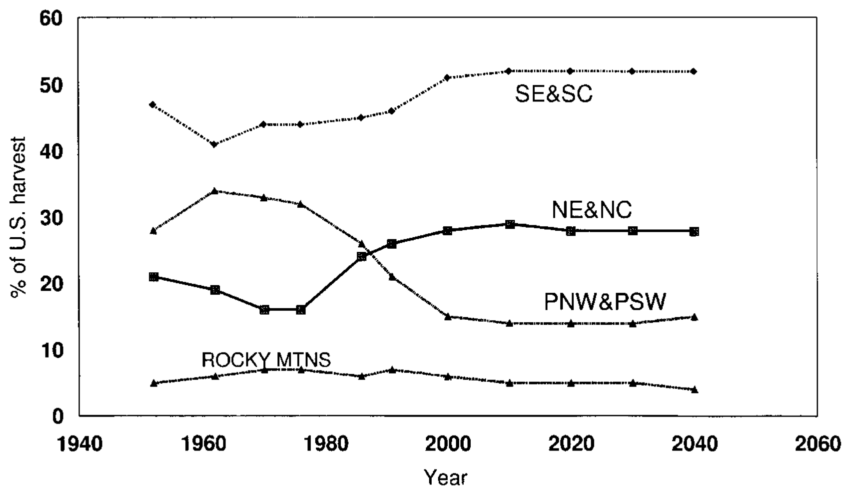


Fig. 1. Timber harvest in the US by region with projections to 2040 (National Research Council, 1998).

rally-regenerated, extensively-managed hardwood forests of the eastern US, there is concern about the effects of such changes on long-term productivity and sustainability, particularly because forest managers have less flexibility to modify the forest environment under less intensive management.

In this paper, we discuss the extensively-managed hardwood forests of the central and eastern US, focusing on the Appalachians, evaluating the potential effects of forest management in a changing chemical environment upon forest productivity. We believe the greatest challenges and opportunities to sustainable forest productivity are found in these forests.

## 2. The resource

The hardwood forests of the eastern US make up the largest forest type of the continental US (Fig. 2). These forests are typified by high spatial variability and high tree species diversity. In the Appalachian mixed hardwoods subtype, for example, there are as many as 20 commercial tree species (Smith et al., 1993), and many more noncommercial tree species. These forests most often are regenerated naturally, from seeds and sprouts, and most are second- or third-growth. Management of these predominantly nonindustrial, privately owned forests consists mostly of manipulation of stand basal area, density or quality, which may include selection for valuable species and/or removal of 'poor quality' trees. Rotation lengths for sawlog-sized trees range from 80 to 120 or more years. Although clearcutting is still used, most harvesting is by partial cutting methods (e.g. diameter-limit cutting, selection cutting) (Stoyenoff et al., 1998), some of which may be unsustainable over the long term (Finley et al., 1997; Fajvan et al., 1998; Miller and Kochenderfer, 1998).

These forests are an important resource for millions of people. The economic value is great, encompassing traditional (timber production) and nontraditional forest products (herbs for medicines, mushrooms, mosses for landscaping), as well as supplying recreational opportunities, including hunting. In West Virginia, the wood products industry alone produced US$ 3.2 billion in 1995, and forest-based tourism contributed approximately US$ 1 billion.

Timber harvesting is expected to increase in the hardwood region in response to shifts in demand (Fig. 1), and has steadily increased since 1976. In West Virginia, 1 billion board feet (~2.4 million m$^3$) of wood was harvested in 1995, approximately the same as was harvested during the peak of the turn-of-the-century timber boom (Fig. 3). Annual growth still exceeds annual harvest removals in West Virginia and Pennsylvania (McWilliams, USDA Forest Service, personal communication, 1998), but the ratio is narrowing. New mills in the Appalachians utilize smaller trees and are oriented toward chip and pulp products rather than traditional sawlog and veneer production.

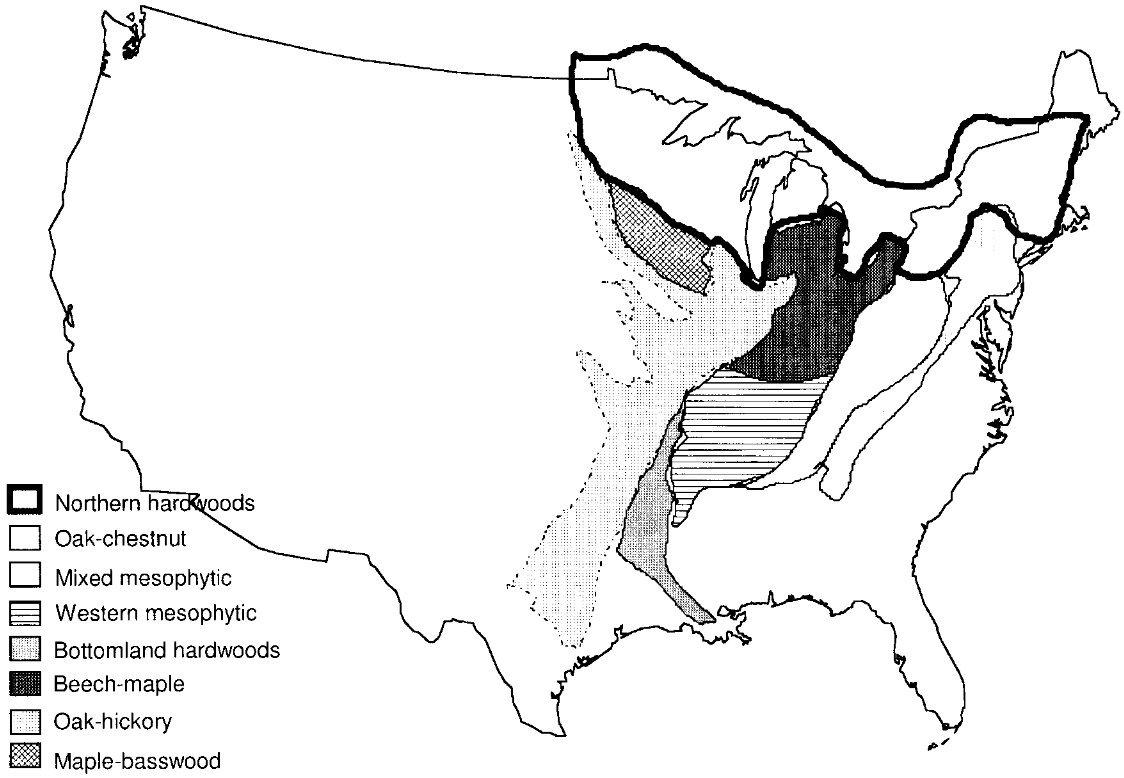


Fig. 2. Extent of eastern hardwood forests. Adapted from Braun (1950) and Barrett (1995).

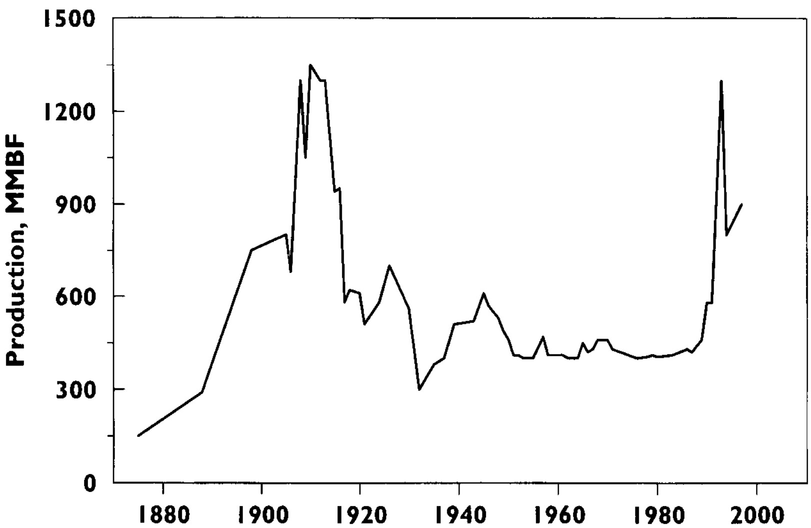


Fig. 3. West Virginia timber harvest levels, 1879–1996. Source: West Virginia Division of Forestry (1990) and Ed Murriner, West Virginia Division of Forestry, personal communication (1998).

## 3. Threats to the resource

### 3.1. Acidic deposition

#### 3.1.1. Mechanisms

One potential threat to the eastern hardwood forest ecosystem is the altered nutrient balance and soil acidity due to changing air quality. At the 1990 North American Forest Soils Conference, Rennie (1990) identified air pollution as a potentially serious, but ill-defined, threat to forest productivity in the US. Today, the threat is perhaps better defined, but is still quite complex, especially because atmospheric chemistry is changing rapidly. We will focus on three aspects of this threat to forest soil productivity related to air quality: nitrogen saturation, sulfate saturation and acidification effects. These three are not independent, but can interact in a fairly complex way.

It is commonly believed that most forests are N-limited, and the response to additional N from acidic deposition would be a positive growth response (Tamm, 1991). This probably is still true for many forests. Recently, however, a number of forested ecosystems in the US have been found to display symptoms of N saturation (Fenn et al., 1998). N saturation occurs when the inputs of N to an ecosystem are greater than the demand by the biota, coupled with decreased terrestrial N retention capacity (Aber et al., 1989). Potential soil effects of N saturation include: increased soil acidification (van Breeman et al., 1982) and aluminum mobility (Johnson et al., 1991), increased nitrate leaching (Aber et al., 1989), altered emissions of greenhouse gases from soil (Steudler et al., 1989; Castro et al., 1994), and elevated base cation leaching (Fenn et al., 1998). One symptom of N saturation is rapid N cycling, particularly in the process of nitrification. As the highly mobile nitrate anion moves through the soil solution, base cations are removed from the soil exchange sites and leached from the soil. These nitrate pulses also can result in considerable increases in aluminum ($Al^{3+}$) in soil solution, to levels that may affect base cation uptake (Johnson et al., 1991; Raynal et al., 1992; Cronan and Grigal, 1995). Since N deposition is expected to increase throughout much of the eastern US during the next two decades (Galloway et al., 1995), N saturation has become a concern.

Saturation of a system with S is not analogous to N saturation because of the different processes driving the two. Whereas N saturation is an ecosystem state, and largely driven by biological processes, S saturation reflects predominantly soil and hydrological processes. As we use the term 'saturation' with respect to S, we are describing a system where S is at steady state under current conditions. Mitchell (1992) reported that biological processes were most important in regulating sulfate fluxes at sites receiving low S inputs. However, at sites receiving higher atmospheric S, the additional S is immobilized by soil adsorption processes which are dependent on the presence of free iron and aluminum oxides and hydroxides (Chao et al., 1962), pH of the soil, and soil solution sulfate concentrations. Sulfate adsorption capacity is also related to the presence of 1:1 clay minerals (e.g. kaolinite), and to soil organic matter content (Jardine et al., 1989). Adsorption is increased when a soil is simultaneously subjected to acidification. Sulfate is an important anion in altering the flux of other ions, especially acidic ($H^+$, $Al^{3+}$) and basic ions ($Ca^{2+}$, $Mg^{2+}$, $K^+$, $Na^+$) in soil solutions (Reuss and Johnson, 1986). Soils with substantial sulfate adsorption are likely to be resistant to accelerated cation leaching (Abrahamsen and Stuanes, 1980). The converse is also true: soils that do not adsorb sulfate are most susceptible to base leaching (Cronan et al., 1978). Thus sulfate saturation or steady state occurs at the point at which soils are no longer adsorbing sulfate onto the soil surface at ambient precipitation concentrations. If precipitation and soil solution concentrations increase, soils may adsorb more sulfate (van Breemen, 1973; Singh, 1984). If adsorption is reversible, then decreases in sulfate input concentrations, as has been recorded in the eastern US (Lynch et al., 1996), could result in decreased adsorption and increased cation leaching (Harrison et al., 1989).

Acidity is the third aspect of atmospheric chemistry which may affect soil productivity. Addition of strong mineral acids (e.g. $H_2SO_4$ and $HNO_3$) in wet and dry deposition were hypothesized to reduce soil pH and base saturation, with other possible detrimental effects relative to elevated $H^+$ activity in soil solutions. However, most forest soils are already acidic and fall within the aluminum or cation exchanger buffering ranges (Table 1), so further acidification can be difficult to observe (Tabatabai, 1985). Markewitz et al.

Table 1
Buffering ranges as defined by Ulrich (1986)

| Buffering range | pH range |
|---|---|
| Carbonate buffering range | 6.5–8.3 |
| Silicate buffering range | All pHs (typically at pH>5) |
| Cation exchanger buffering range | 4.2–5.0 |
| Aluminum buffering range | >4.2 |
| Aluminum/iron buffering range | >3.8 |
| Iron buffering range | >3.2 |

(1998) estimated that 38% of soil acidification observed over three decades under loblolly pine (*Pinus taeda* L.) was attributable to acid deposition, while 62% was attributed to internal ecosystem functions. Acidification processes are also directly related to exchange processes on the soil surface. Base cation depletion occurs from leaching with counterbalancing sulfate and nitrate and as a result of fewer cation exchange sites due to lower pH, and decreased base saturation from higher $Al^{3+}$ availability. When base saturation is low, $Al^{3+}$ is the dominant cation available for exchange, resulting in the release of aluminum in response to acidic deposition. These processes can lead to increased aluminum toxicity or nutrient imbalances (Cronan and Grigal, 1995).

### 3.1.2. Evidence of effects

On two forested watersheds at the Fernow Experimental Forest in West Virginia, we have detected changes in nutrient cycling that we attribute to N saturation (Gilliam et al., 1996; Peterjohn et al., 1996; Adams et al., 1997). One is a relatively young stand of Appalachian mixed hardwoods, regenerated following clearcutting in 1969–1970. Ammonium sulfate fertilizer has been applied to this watershed since 1989 at approximately twice the ambient deposition (as throughfall) rates of 17 kg N $ha^{-1}$ and 20 kg S $ha^{-1}$ per year. We have detected a number of symptoms of N saturation as a result of this treatment (Table 2): elevated concentrations and increased export of nitrate, calcium, and magnesium in stream water, and increased foliar N concentrations in some vegetation. Increased concentrations of calcium and magnesium in the wood of black cherry were also detected (DeWalle et al., 1995). Despite significant changes in cycling of some nutrients, there have been no significant changes in soil N levels (as measured by Kjehldahl N) or tree growth as a result of this treatment. Rates of net N mineralization and net nitrification are high and did not differ between treated and control soils. Although inputs still exceed outputs, retention of the added N decreased significantly over 5 years from 95 to 75% (Adams et al., 1997). It was surprising to us that such a young, apparently vigorous stand of trees would show symptoms of N saturation.

The second stand is on a reference watershed and was last harvested in about 1910. Stream water nitrate and calcium concentrations and conductivity have increased steadily during the last 20 years (Fig. 4; Edwards and Helvey, 1991; Adams et al., 1994, 1997; Peterjohn et al., 1996). Net nitrification is nearly 100% of net mineralization (Gilliam et al., 1996; Williard et al., 1997). Calcium is leaching from this watershed at a rate of 10 kg $ha^{-1}$ per year, approximately two to four times the atmospheric inputs, and magnesium losses are 6 kg $ha^{-1}$ per year, approximately 10 times atmospheric inputs (Adams et al., 1997). Weathering

Table 2
Responses to Watershed Manipulation Treatment of Watershed 3, Fernow Experimental Forest

| | |
|---|---|
| Increased streamwater export of $NO_3$, Ca, Mg | Adams et al. (1997) |
| Elevated foliar N concentrations: | |
| Of *Betula lenta, Liriodendron tulipifera, Prunus serotina, Acer rubrum* | Adams et al. (1995), Gilliam et al. (1996) |
| Of a bioindicator, *Viola blanda* | Gilliam et al. (1996) |
| Decreased foliar Ca and Mg concentrations of *V. blanda* | Gilliam et al. (1996) |
| Lower foliar Ca concentrations in *B. lenta, L. tulipifera, P. serotina, A. rubrum* | Gilliam et al. (1996) |
| No differences in net N mineralization or net nitrification | Gilliam et al. (1996) |
| No differences in soil nutrients after 3 years of treatment | Gilliam et al. (1994) |
| Litter decomposition | |
| Decreased mass loss rates for *L. tulipifera, P. serotina, B. lenta* | Adams and Angradi (1996) |
| Decreased stemwood Ca for *L. tulipifera* | Adams et al. (1995) |
| Increased stemwood Ca for *P. serotina* | DeWalle et al. (1995) |

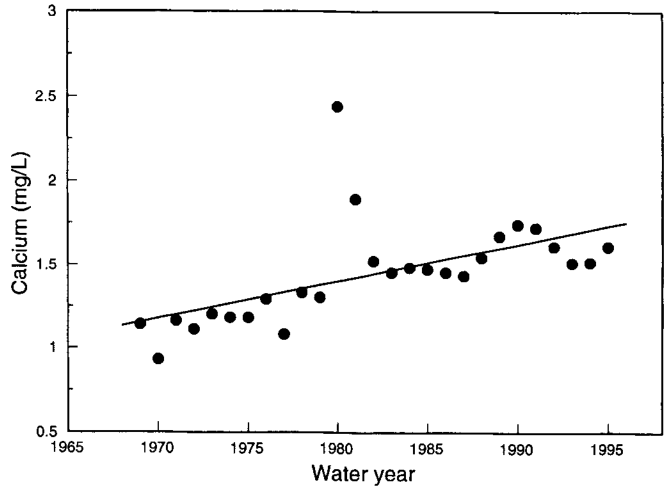


Fig. 4. Streamwater concentrations of calcium, WS4, Fernow Experimental Forest.

rates are not known, but are estimated to be very low from these acidic sandstones and shales. With decreasing inputs and increasing leaching losses of calcium and magnesium, there is concern about the long-term productivity of such forests. Adams (1999) calculated changes in total calcium and magnesium pools in eastern hardwood forests from 0 to −15% and from +4 to −32%, respectively, with no harvesting, over 80 years. Weathering was assumed to be zero in this calculation, based on the low available levels in the parent material.

Sulfur cycling has also been affected. Mass balances suggest these soils are still adsorbing sulfate (Adams et al., 1997). However, soil solution patterns suggest that the soils on the treated watershed may be approaching steady state (Fig. 5). Thin surface horizons, with high organic matter and low iron content have little capacity to adsorb additional sulfate and are prone to leaching (Lusk, 1998). Deeper subsurface horizons with low organic matter and higher iron-oxide contents would adsorb sulfate at high input levels but have little capacity for adsorption at lower concentrations. Lusk (1998) evaluated sulfate adsorption capabilities of soils on the Fernow Experimental Forest in West Virginia and concluded that these soils may already be at steady state (sulfate saturated) at ambient conditions, a conclusion similar to those reached by early region-wide surveys for many parts of the central Appalachians and the mid-Atlantic

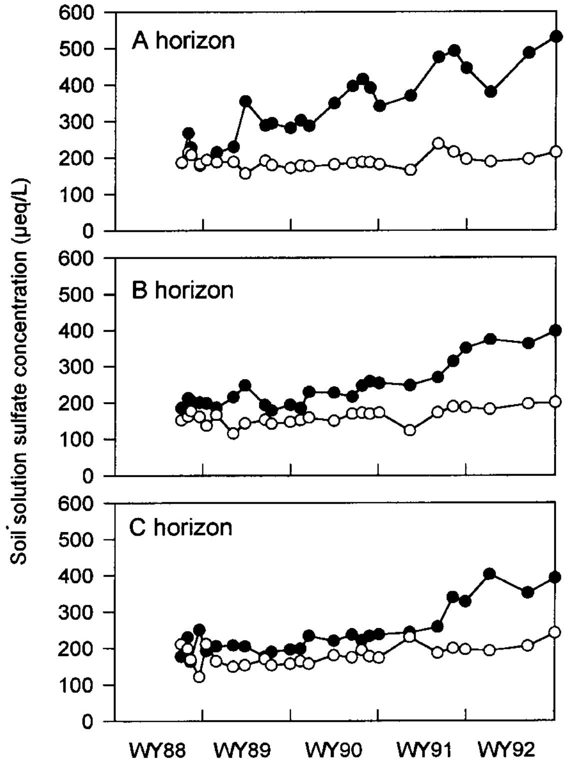


Fig. 5. Soil solution sulfate concentrations for two watersheds on the Fernow Experimental Forest. WS3 (●) has received ammonium sulfate additions since 1989. WS4 (○) is the reference catchment. See text for details of treatment.

region (Turner et al., 1986; Herlihy et al., 1993). Lusk (1998) also predicted that decreasing S inputs may result in decreased sulfate adsorption and increased sulfate leaching, with the consequence of elevated base cation leaching.

Changes in soil pH over time, mostly in upper soil horizons, have been documented (cf. Johnson et al., 1994), but only occasionally has this change been documented in hardwood stands, and attributed, at least in part, to acidic deposition (Droyan and Sharpe, 1997). However, decreases in exchangeable soil calcium and magnesium have been observed in response to elevated N and S deposition (Miller et al., 1993; Johnson et al., 1994; Bailey et al., 1996).

#### *3.1.3. Sensitive forests and soils*

Forests that are sensitive to N saturation are those that are older, relatively undisturbed, with low N demand (Vitousek and Reiners, 1975; Fenn et al., 1998), or with high N turnover (Van Miegroet and Cole, 1984; Johnson and Lindberg, 1992). Data from a number of sites in the mid-Appalachians show high nitrate export from soils with high ratios of net nitrification/net mineralization (Williard et al., 1997). High turnover rates have been reported elsewhere in the hardwoods region (Boerner and Sutherland, 1995; Gilliam et al., 1996).

Soils that might be considered susceptible to N saturation have high soil N stores with low C:N ratios (Williard et al., 1997; Fenn et al., 1998). Soil C:N ratios for typical forest soils in the central hardwood region are generally less than 20 (Jencks, 1969), identified by Van Miegroet et al. (1992) as a critical level for loss of N. Soil exchangeable calcium and soil moisture were also correlated with nitrate leaching and N mineralization rates in hardwood forests (Williard et al., 1997), suggesting that on the more mesic sites N cycling is not limited by moisture or pH, and thus these sites may be susceptible to N saturation. A number of N saturated sites have been identified in Europe (Johnson et al., 1991) and across the US (Fenn et al., 1998) including some in the eastern hardwood forest (DeWalle and Pionke, 1996; Peterjohn et al., 1996).

Forests that receive low inputs of S are those where biological processes are most important in regulating sulfate flux (Mitchell, 1992). Therefore forests with low S demand and high inputs, particularly over many years, are more likely to be saturated. Soils which are likely to be sulfate saturated are young, unweathered soils with low amounts of aluminum and iron oxides, relatively low clay content, and relatively high organic matter (Johnson and Todd, 1983; Rochelle et al., 1987). Surface soils and organic rich subsoils of Spodosols, for example, are relatively inefficient at absorbing sulfate, and more prone to saturation. Subsurface horizons of Ultisols, Alfisols, and some orders of Inceptisols and Entisols are efficient sulfate adsorbers. Sulfate adsorption may increase in response to high nitrification rates due to acidification effects (Nodvin et al., 1986), so N saturated soils may have increased sulfate adsorption capacity.

As forests grow and mature, soils acidify naturally through uptake processes. Therefore, young forests might be more sensitive to effects of acidification. Forests in areas of high acidic inputs are more likely to be affected as well. Soils that are sensitive to acidification and base leaching are those that are poorly buffered (CEC<15 cmol $kg^{-1}$) with moderate base saturation and pH<4.5 (Turner et al., 1986). Soils developed from a parent material with large amounts of available base cations (i.e. limestone, or some glacial parent materials) are less sensitive to these processes. Older, more weathered soils would be more sensitive to these processes.

### *3.2. Harvest removals*

#### *3.2.1. Mechanisms*

Numerous studies have documented that removals of nutrients increase with the mass of the organic matter removed (Smith et al., 1986; Mann et al., 1988; Federer et al., 1989). Nutrient levels in aboveground biomass of mature hardwood forests range from 143 to 323 kg N $ha^{-1}$, 307 to 1463 kg Ca $ha^{-1}$ and 20 to 62 kg Mg $ha^{-1}$ (Table 3). Whole-tree harvesting removes more nutrients than clearcutting because more biomass is removed, but harvest-induced leaching is greater for clearcutting than for whole-tree harvesting. In the short-term, partial harvests remove fewer nutrients than clearcutting or whole-tree harvesting, because of the much lower biomass removed in a single partial cut. However, over the longer term, multiple incursions may actually remove more nutrients over the course of a rotation (Fig. 6). Harvesting may also increase base cation

Table 3
Aboveground biomass, N, Ca and Mg content or hardwood forests

| Forest | Biomass (Mt/ha) | N (kg/ha) | Ca (kg/ha) | Mg (kg/ha) | Reference |
|---|---|---|---|---|---|
| Coweeta, NC | 178 | 277 | 544 | – | Mann et al. (1988) |
| Oak Ridge, TN | 175 | 323 | 1090 | 50 | Mann et al. (1988) |
| Cockaponset, CT | 158 | 273 | 530 | 39 | Mann et al. (1988), Tritton et al. (1987), Federer et al. (1989) |
| Baraga, MI | 100 | 143 | 307 | 23 | Crow et al. (1991) |
| Fernow LTSP, WV | 283 | 633 | 1463 | 62 | Adams (1999) |
| Mt. Success, NH | 111 | 242 | 344 | – | Mann et al. (1988) |
| Wisconsin | 167 | 120 | 241 | 24 | Boyle and Ek (1972) |

leaching through microclimate effects such as increased soil temperature and moisture. Harvesting can also cause short-term acidification of soil solutions, through increased N cycling rates and increased nitrification, leading to release of aluminum.

### 3.2.2. *Evidence of effects*

Soil acidification and base cation mobilization were both evident after clearcutting a northern hardwood forest (Fuller et al., 1987), but were not detected after clearcutting a forested watershed in West Virginia (Aubertin and Patric, 1974). Decreases in sulfate concentrations of streamwater were observed in both studies and attributed to increased sulfate adsorption by Fuller et al. (1987). Research evaluating long-term effects of harvesting on soil base nutrient pools is sparse. Knoepp and Swank (1997) reported significant increases in surface horizon (0–10 cm) exchangeable soil magnesium and potassium concentrations 17 years after clearcutting and significant increases in subsurface (10–30 cm depth) magnesium and potassium concentrations, but no change in calcium concentrations. In a northern hardwood forest, decreases in soil pools in the upper soil horizons 8 years after clearcutting were offset by increases in lower (spodic) horizons (Johnson et al., 1997). Johnson and Todd (1998) reported no changes in exchangeable Ca pools 15 years after harvesting, suggesting some of this replenishment may be due to weathering or deep rooting. The long-term impacts of harvesting on soil cation levels are site-specific and may vary with factors affecting the biogeochemical cycling of nutrients including site fertility, species composition, harvest intensity, and site microclimate.

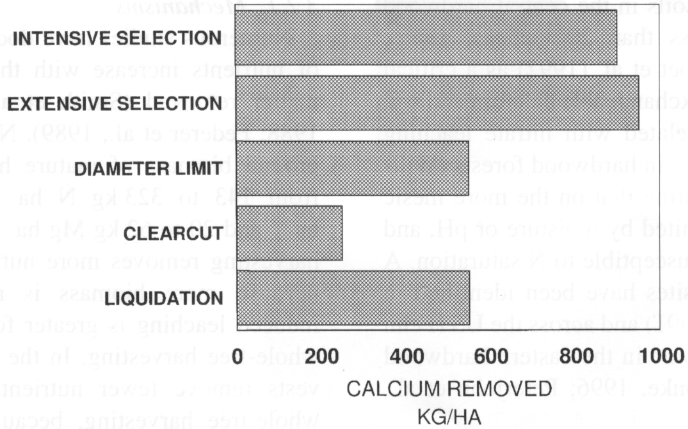


Fig. 6. Calcium removed by five different silvicultural treatments during a 75-year rotation. Intensive selection=selected trees >12.5 cm dbh removed; extensive selection=selected trees >27.5 cm dbh removed; diameter limit=all salable trees >43 cm removed; commercial clearcutting=all salable trees over 12.5 cm removed; liquidation cutting=all salable trees, branches, and culls removed (analogous to whole-tree harvesting). Redrawn from Adams (1999).

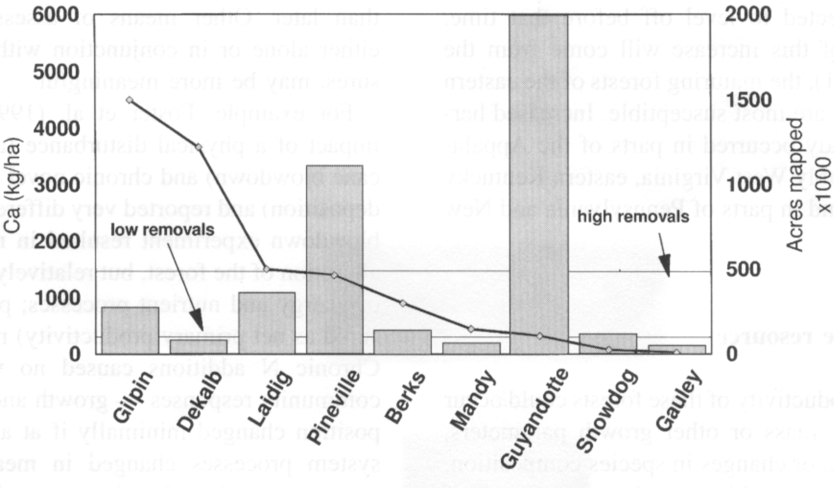


Fig. 7. Comparison of exchangeable soil Ca levels (bars) found in common West Virginia soil series (A.B. Jenkins, unpublished data) with range of Ca removed in harvesting. The line represents the number of acres mapped of each soil series.

### 3.2.3. *Sensitive forests and soils*

Forests that may be particularly susceptible to nutrient depletion effects of harvest removals would be those with a large proportion of species such as hickories (*Carya*), basswood (*Tilia americana*), oak (*Quercus*), and yellow-poplar (*Liriodendron tulipifera*), which store large amounts of calcium in their bole wood (Raynal et al., 1992). Johnson et al. (1988) found significant decreases in subsoil exchangeable calcium due to high uptake rates by the Walker Branch mixed deciduous forest, containing a high proportion of calcium-demanding species. Forests where large amounts of the base nutrients are stored aboveground would be susceptible to base losses from harvesting. Soils that are sensitive to base cation depletion from harvesting include those with low CEC, moderate to low base saturation, those that develop from parent material low in weatherable bases or those that are highly weathered. For some high elevation forest soils in West Virginia derived from variably acid sandstone and shale, total nutrient pools ranged from 400 to 1096 kg Ca ha$^{-1}$ and 1628 to 6662 kg Mg ha$^{-1}$ (Jenkins et al., 1998). Exchangeable values ranged from 156 to 350 kg Ca ha$^{-1}$ and 28 to 126 kg Mg ha$^{-1}$ (Jenkins et al., 1998). Some common forest soils in the Appalachians contain lower levels of exchangeable calcium than is removed in most harvests (Fig. 7).

Hardwood timber harvests have increased in the US since 1952 and are projected to increase through 2040 (Fig. 8), although growing stock and net annual

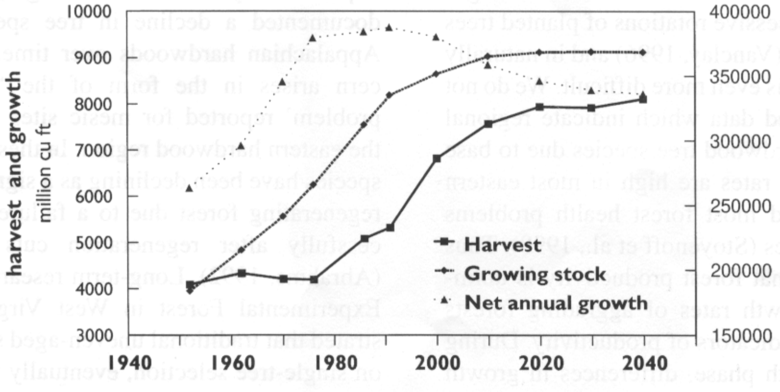


Fig. 8. Hardwood timber harvests, growing stock, and net annual growth in the US. Source: National Research Council (1998).

growth are projected to level off before that time. Because much of this increase will come from the eastern US (Fig. 1), the maturing forests of the eastern hardwood region are most susceptible. Increased harvesting has already occurred in parts of the Appalachians, most notably West Virginia, eastern Kentucky and Tennessee, and in parts of Pennsylvania and New York.

## 4. Effects on the resource

Changes in productivity of these forests could occur as changes in biomass or other growth parameters, changes in health, or changes in species composition. Altered productivity could come about as a result of nutrient deficiencies or imbalances. Although calcium and magnesium deficiencies are rare in trees in extensively managed forests, they have been reported for sugar maple (*Acer saccharum*; Wilmot et al., 1995; Long et al., 1997) and red spruce (*Picea rubens*; Joslin and Wolfe, 1994) in the US. In addition, some hardwood tree species, such as sugar maple, show declines that may be related to acidic deposition stress, but these links have not been conclusively documented. Nitrogen-induced magnesium deficiencies or imbalances have been reported for Norway spruce (*Picea abies*) in Germany (Hüttle, 1990) and for European oaks (*Quercus petrea* and *Q. robur*; Berger and Glatzel, 1994). Nitrogen-induced phosphorus deficiency also has been reported (cf. Mohren et al., 1986; Auchmoody and Smith, 1977).

The traditional measures of forest productivity are aboveground biomass or volume of trees or wood products. Utilizing these measures to detect changes in productivity in successive rotations of planted trees is not straightforward (Vanclay, 1996) and in naturally regenerated forests, it is even more difficult. We do not know of any published data which indicate regional growth declines of hardwood tree species due to base cation losses. Growth rates are high in most eastern hardwood forests, and most forest health problems have identifiable causes (Stoyenoff et al., 1998). Thus one might conclude that forest productivity is unimpaired. However, growth rates of aggrading forests may not be the best indicators of productivity. During the exponential growth phase, differences in growth rates and absolute size are likely to be less obvious than later. Other means of assessing productivity, either alone or in conjunction with traditional measures, may be more meaningful.

For example, Foster et al. (1997) compared the impact of a physical disturbance (a simulated hurricane blowdown) and chronic novel stress (elevated N deposition) and reported very different responses. The blowdown experiment resulted in massive structural alteration of the forest, but relatively minor disruption of energy and nutrient processes; productivity (measured as net primary productivity) recovered quickly. Chronic N additions caused no visible structural, community responses — growth and vegetative composition changed minimally if at all. However, ecosystem processes changed in measurable ways in response to the chronic stress; N mineralization increased and methane consumption decreased, both in a nonlinear fashion. In a similar study, Morris and Boerner (1998) evaluated the effects of chronic (deposition) and acute (thinning) disturbance and concluded that there were significant interactions for many soil chemical parameters and processes related to N cycling. Authors of both studies concluded that assessments of forest health and productivity should be based on key processes rather than structural measurements.

Another possible indicator of changing productivity of these forests is species composition. There already exists a growing concern in the eastern hardwood region about decreasing diversity of tree species. Numerous introduced pests or pathogens have had significant impacts on forest health; some have resulted in near extirpation of some tree species (e.g. *Castanea dentata*, American chestnut, nearly wiped out by the chestnut blight). Schuler (1997) documented a decline in tree species diversity in Appalachian hardwoods over time. Additional concern arises in the form of the 'oak regeneration problem' reported for mesic sites in many parts of the eastern hardwood region. In these areas, some oak species have been declining as a significant part of the regenerating forest due to a failure to compete successfully after regeneration cuts on mesic sites (Abrahms, 1992). Long-term research on the Fernow Experimental Forest in West Virginia has demonstrated that traditional uneven-aged silviculture, based on single-tree selection, eventually reduces the abundance of certain species, including oaks (Smith, 1993;

Schuler and Miller, 1995). Neither diameter-limit harvests (currently, the predominant form of harvest in West Virginia (Fajvan et al., 1998)), nor single-tree selection is recommended in central Appalachian hardwood stands if the goal is to sustain tree species diversity (Miller and Kochenderfer, 1998). This loss of important tree species as a result of harvesting has been documented for both West Virginia (Fajvan et al., 1998) and Pennsylvania (Finley et al., 1997).

Nitrogen saturation could also have implications for diversity. Altered species composition of ground vegetation has been recorded as a result of high levels of N inputs, whether through fertilization or elevated atmospheric N deposition (Carson and Barrett, 1988; Falkengren-Grerup and Ericksson, 1990; Kellner and Redbo-Torstensson, 1995). There is little evidence of altered tree species diversity with elevated N levels, but McNulty et al. (1996) predicted changes in forest species composition, from red spruce to hardwoods, and decreased productivity as a result of N saturation. Because tree species vary in their nutrient utilization capabilities and requirements, it is not unlikely that changes in nutrient availability due to acidic deposition could alter species composition. Further changes in species composition of hardwood forests could exacerbate existing problems, and further limit silvicultural options. Insufficient or nonexistent seed sources or sprouts of some species already limit recruitment in some hardwood stands (Clark et al., 1998).

Diversity of other organisms may also be affected. Diversity of ectomycorrhizal fungi decreased significantly with elevated N additions (Brandrud and Timmerman, 1998). Boerner and Sutherland (1995) posited that changes in N cycling could alter mycorrhizal health and functioning, with implications for tree species composition. Finally, in The Netherlands, declining nesting success of the great tit (*Parus major*) has been attributed to thin egg shells (Graveland et al., 1994). The egg shell defects are caused by calcium deficiency, due to scarcity of snails, whose shell are the main calcium source for the laying female. Snail populations have declined on acidic sites, due to decreased soil calcium levels, which the authors attribute to elevated calcium leaching from very high levels of acidic deposition. Egg shell thinning, of 7–10% since 1850, has also been documented elsewhere (Green, 1998).

## 5. Is there a problem?

The problem of potential base cation depletion of forest soils is not a new one. Researchers began addressing this question back in the 1970s (Weetman and Webber, 1972; Boyle et al., 1973; Voigt, 1979). Our methods and nutrient budgets have been considerably refined through much excellent research, but the conclusions still point to a potential problem. We believe the issue is of increasing urgency because of shifts in wood products markets which are shortening rotations and increasing intensity of harvests throughout the eastern hardwood forest region, at the same time that significant changes in atmospheric chemistry, particularly N, are occurring. Hornbeck (1992) evaluated contributions to soil acidification from acidic deposition and harvesting and concluded that acidic deposition contributions were equal to or greater than the contribution from whole-tree harvesting over the course of a rotation. With both factors being significant in parts of the eastern hardwood forests, we believe that continuing 'business as usual' could lead to declines in long-term productivity. We propose that such declines in productivity over the long-term are not likely to be easily detected with current monitoring programs, which focus almost exclusively on aboveground growth of trees, and that developing research and monitoring programs which include ecosystem processes and indicators is critical. We present a rationale and framework for monitoring sustainability of productivity, focussing on Appalachian forests.

## 6. A rationale and framework for monitoring sustainability of Appalachian forests

### 6.1. Background

There is wide public expectation that forests be managed in a sustainable manner and that they be protected from natural and human caused disturbances that could threaten their long-term productivity, diversity, their function in regulating the hydrologic cycle, and providing wildlife habitat. In the US, a number of federal laws such as the Multiple Use and Sustained Yield Act of 1964 and the Clean Water Act of 1989 resulted from national debates on stewardship. In the

US, the Forest Service has a legal responsibility to monitor the effects of management practices on forests and soils and to ensure sustained productivity within the National Forest System (Powers et al., 1990). In the private sector, member companies of the American Forest and Paper Association are committed to a sustainable forestry initiative of principles, guidelines and performance measures that integrates the perpetual growing and harvesting of trees with the protection of wildlife, plants, soil, air and water quality (American Forest and Paper Association, 1995). Following the United Nations Conference on Environment and Development in Rio de Janeiro in 1992, several international initiatives, one of which is the Montreal Process (1995), ensued on developing criteria and indicators of sustainable forest management at national levels. The US is participating in the Montreal Process with other nonEuropean countries with temperate and boreal forests. These laws and initiatives are clear mandates for working toward the goal of sustainable forestry at local, regional, and national levels. However, accomplishing the goal will involve a long institutional, political, and research process of (1) defining sustainable management; (2) developing criteria and indicators of sustainable management; (3) establishing management guidelines that should achieve the criteria; (4) monitoring the criteria by measuring indicators of sustainability; and (5) adapting management based on the results of monitoring and research programs (Noss, 1990; Karr, 1993; Doran et al., 1994; Burger and Kelting, 1998).

The first step, defining sustainable forestry, involved worldwide debate over a 10-year period. Sustainable forestry is stewardship and use of forests and forest lands in a way and at a rate that maintains their biodiversity, productivity, vitality, and potential to fulfill, now and in the future, ecological, economic, and social functions without damaging other ecosystems (Aplet et al., 1993; Helsinki Process, 1994; American Forest and Paper Association, 1995; Montreal Process, 1995). The second step was accomplished by the Montreal Process which identified six criteria and 67 indicators for sustainable forestry. The fourth criterion includes conservation of soil and its quality. Indicators for this criterion are percent of forest land with significant soil erosion, compaction, diminished organic matter, or changes in other soil chemical and physical properties. These criteria and indicators are still under study by the forestry community, along with the remaining steps for accomplishing sustainable forestry goals. However, this process has provided a framework for local, regional, and national sustainable forestry initiatives, and we believe it is a useful framework for achieving a sustainable Appalachian hardwood forest.

### 6.2. *Criteria and indicators for monitoring sustainability*

As we argued above, the greatest present threats to sustainable Appalachian hardwood forests are changes in nutrient balance and acidity caused by changing levels of nitrate and sulfate additions from air pollution, and accelerated base nutrient depletion from intensive harvesting. Specific sustainability criteria should be (1) maintenance of nutrient balances adequate for forest composition and productivity commensurate with historic levels; and (2) maintenance of a soil acidity/alkalinity balance commensurate with natural levels.

A rationale for defining and selecting indicators of sustainable ecosystem function have been presented by researchers from various disciplines (Noss, 1990; Doran et al., 1994; Nambiar, 1996; Burger and Kelting, 1998). Soil-based indicators act as surrogates for processes controlling water, nutrient, and energy balance and flow. Surrogates are used because a direct relationship between potentially degrading processes (e.g. base cation leaching, aluminum mobilization) and forest composition and productivity is difficult to establish in the short-term. Therefore critical surrogates are chosen that can be related to effects influencing forest vitality, such as soil buffering processes or decreases in fine root production and mycorrhizal frequency that reduce nutrient uptake. To be useful, indicators should (1) have an available baseline against which to compare changes; (2) provide a sensitive, continuous, and timely measure of soil change; (3) be applicable over large areas; (4) be inexpensive and easy to use, collect, and calculate; (5) discriminate between natural changes and those induced by management; and (6) be highly correlated to long-term response in long-lived forest ecosystems.

Early calculations of soil sensitivity to change in nutrient balance and acidity were made by McFee

Table 4
Sensitive forests and soils

| | Forest characteristics | Soil characteristics |
|---|---|---|
| N saturation | Old forests<br>Low N demand<br>High N turnover | Low C:N<br>High nitrification rates |
| Sulfate saturation | Low S demand<br>High S inputs, over long time<br>Native S sources | Young soils<br>Low Al and Fe oxides<br>High O.M.<br>Low clay content<br>4.5<pH<5.5 |
| Acidification | Aggrading forests<br>High acid inputs | Low CEC<br>Moderate B.S.<br>Acidic parent material<br>Low available weatherable minerals |
| Harvesting | Ca, Mg demanding tree species present<br>High leaching rates<br>Low base inputs | Moderate-low B.S.<br>Low available weatherable minerals<br>Low CEC<br>Uptake>outputs |

(1980), Klopatek et al. (1980), and Olson et al. (1982). Reuss and Johnson (1986) reviewed and summarized these and other studies. Based on these assessments and reviews, the most important potential degrading effects of atmospheric deposition of acids and intensive harvests are base cation depletion, aluminum mobilization, and pH depression. Sensitive soils include very acid soils that allow aluminum exchange into solution in response to increased ionic strength, and moderately acid soils with low CEC and moderate to high base saturation with little ability to buffer acidification caused by accelerated leaching (Table 4). This describes the majority of Appalachian forest soils derived from acid shales and sandstones that underlie many of the Ridge and Valley and Allegheny and Cumberland Plateau provinces of the Appalachians. However, within these soils there is still a gradient of sensitivity based on chemistry, mineralogy, and total depth. Maps of these soils based on the above sensitivity criteria are seriously needed. Turner et al. (1986) used available soils data and computer display techniques to map the areal extent of soil characteristics that may be correlated with forest decline. However, regional sensitivity maps such as those produced are simply insufficient for local, forest-level decision making.

Soil indicators that might be used as surrogates for the first criterion of sustainability that we identified, nutrient imbalance and base depletion include: CEC, base saturation, and a surrogate for buffering capital of a site — the product of base saturation$\times$CEC$\times$rooting depth; measures of sulfate steady state and N saturation; and foliar nutrient levels. Many Appalachian soils are well-weathered with much of their pH-dependant charge on 1:1 clays and organic matter; as the soil becomes more acid, pH-dependant charge will drop. Base saturation indicates a relative level of base nutrient availability, especially when multiplied by the CEC and rooting depth. Measures of sulfate and N saturation provide an indication of the potential for cation leaching. Adsorption of sulfate reduces ionic strength of soil solutions and thus the leaching of polyvalent nutrient cations. Nitrate concentration below the rooting zone is an indicator of N saturation of the ecosystem. Nitrate leaching results in base cation leaching due to charge balance chemistry, similar to that of sulfate. Monitoring these soil properties along with foliar nutrient concentrations, and comparing these values with historical values will provide an assessment of nutrient balance and base sufficiency.

The second concern is pH depression and aluminum mobilization and toxicity. Meiwes et al. (1986) selected the following indicators and threshold levels: soil pH for grouping soils into proton buffering categories, base neutralizing capacity, base saturation, Ca/

Al ratio in the soil solution, and the ratio of base cations to acid cations in fine roots and humus. Their threshold values were: (1) soil base saturation (humus layer, $O_H$ layer) values greater than 0.1, in the 0.05–0.1 cm horizon, and less than 0.05 represented little, medium, and high risks, respectively, of acid toxicity to fine roots and mycorrhizal fungi; (2) Ca/Al molar ratios in the soil solution greater than 1 presented no risk, while values less than 0.1 indicated high risk for both spruce and beech; (3) a molar Ca/Al ratio less than 0.3 in fine roots in the surface 5 cm soil horizon, and less than 0.1 in the 5–20 cm horizon indicates a high probability of fine root damage. These criteria and indicators are examples of the type that need to be established for Appalachian hardwood forests.

Most research on acid deposition effects on soils in the eastern US has been in spruce, pine, and sugar maple, with comparatively less research directed to response of oak-hickory, mixed mesophytic, or Appalachian oak forests to soil nutrient imbalances or aluminum toxicity. Because there is evidence to suggest that soil changes could become a problem in these forests (NAPAP, 1990), specific soil indicators and threshold levels for the selected indicators should be identified.

Monitoring soil change is a process of estimating changes in soil condition that have occurred since the last time it was measured. This approach has been criticized (Wagenet and Hutson, 1997) because it gives no indication of future soil conditions that may result from continuing impacts of degrading processes. Wagenet and Hutson (1997) argue that we should anticipate effects in a prospective manner rather than measuring them retrospectively. Political decisions on emission reductions will require determinations of deposition levels of sulfate and nitrate that cause forest decline, and they should be made before the fact. An example of this type of forecasting is provided by De Vries (1993), who calculated average critical loads for N and S for use in acidification abatement policy in The Netherlands. Critical loads were calculated with a combination of steady-state and dynamic models based on critical chemical values for aluminum toxicity, Al/Ca ratio in soil solution, and ratios of ammonium to potassium and magnesium. A logical follow-up is the derivation of critical load maps for spatial interpretations and prescription (De Vries, 1991).

Sustainable management of intensively-harvested hardwood forests that receive high inputs of S and N will require guidelines that are based on the following criteria: (1) cause and effect relationships between harvesting and base nutrient depletion; (2) cause and effect relationships between S and N atmospheric inputs and nutrient imbalances and aluminum toxicity; (3) identification of indicators of soil change that could lead to forest decline; (4) development of a monitoring system to determine if management practices are meeting criteria of sustainability; (5) based on scientific principles and empirical trials, forecasting acceptable harvest levels and critical loads of S and N input on a site-specific basis; and (6) adapting and revising management guidelines as additional experience and research results suggest.

## 7. Recommendations

The hardwood forests of the eastern US are a critically important resource. They provide wood, habitat for humans and wildlife, and important ecological and social functions. We need to understand the components of productivity and the threats to productivity, so that we can better manage these forests to meet the many, conflicting demands placed upon them. Research needs to be interdisciplinary and large-scale, as well as very basic to applied. Development of indicators of soil change related to long-term forest productivity is a high priority.

Finally, we must learn to manage these forests. As alluded to above, ensuring sustainable productivity means management of base levels and nutrient cycling processes. We need to develop means of monitoring the effects of our management (criteria and indicators) as well as become more innovative and proactive in our management. Less intensive and less frequent harvests are often recommended as one way to return more organic matter and nutrients to the forest and prevent productivity loss. But, as predictions suggest the reverse will happen in the eastern hardwoods region, that harvesting will increase in intensity and scope, more active alternatives for managing for important ecosystem processes, like base nutrient cycling or species diversity, are needed. Use of wood residues (wood ash, etc) has been proposed as a way of returning base nutrients to a forest and of effectively 'reducing the N load' from acidic deposition (Lündborg, 1997). Vance (1996) reported that wood-fired boiler ashes can be a successful replacement for agricultural lime and can enhance long-term productivity of forests. Liming is an option (Hornbeck, 1992), although liming can also stimulate nitrate leaching (Johnson et al., 1995), so would need to be carefully done. Fire may also prove to be a valuable silvicultural tool. Burning logging residues returns base cations to the site in the short-term (Boyle, 1973; Raison, 1979). It may also be the answer to the long-unresolved, oak regeneration problem throughout the hardwood region (Abrahms, 1992). Obviously there is much we need to learn in order to use these tools successfully and to develop new ones. All of these suggest a move from extensive forest management to more intensive forest management of eastern hardwood forests is needed to resolve productivity concerns.

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## References

Aber, J.D., Nadelhoffer, K.J., Steudler, P., 1989. Nitrogen saturation in northern forest ecosystems. Bioscience 39, 378–386.

Abrahamsen, G., Stuanes, A.O., 1980. Effects of simulated acid rain on the effluent from lysimeters with acid, shallow soil rich in organic matter. In: Drabløs, D., Tollan, A. (Eds.), Ecological impact of Acid Precipitation. Johs. Grefslie Trykkeri A/S, Mysen, Norway, pp. 152–153.

Abrahms, M.D., 1992. Fire and the development of oak forests. BioScience 42, 346–353.

Adams, M.B., 1999. Acidic deposition and sustainable forest management in the central Appalachians. USA. For. Ecol. Manage. 122, 17–28.

Adams, M.B., Angradi, T.R., 1996. Decomposition and nutrient dynamics of hardwood leaf litter in the Fernow whole-watershed acidification experiment. For. Ecol. Manage. 83, 61–69.

Adams, M.B., Kochenderfer, J.N., Wood, F., Angradi, T., Edwards, P., 1994. Forty years of hydrometeorological data on the Fernow Experimental Forest, West Virginia. Gen. Tech. Rep. NE-184, USDA For. Serv., Northeast. For. Exp. Stn., Radnor, PA, 24 pp.

Adams, M.B., Kochenderfer, J.N., Angradi, T.R., Edwards, P.J., 1995. Nutrient budgets of two watersheds on the Fernow Experimental Forest. In: Gottschalk, K.W., Fosbroke, S.L. (Eds.), Proceedings of the 10th Central Hardwood Forest Conference, Morgantown, WV, 5–8 March 1995, Gen. Tech. Rep. NE-197. USDA For. Serv., Northeast. For. Exp. Stn., Radnor, PA, pp. 119–130.

Adams, M.B., Angradi, T.R., Kochenderfer, J.N., 1997. Stream water and soil solution responses to 5 years of nitrogen and sulfur additions at the Fernow Experimental Forest, West Virginia. For. Ecol. Manage. 95, 79–91.

American Forest and Paper Association, 1995. Sustainable Forestry Initiative. AFPA, Washington, DC.

Aplet, G.H., Johnson, N., Olson, J.T., Sample, V.A. (Eds.), 1993. Prospects for a sustainable future. Defining Sustainable Forestry. Island Press, Washington, DC, pp. 309–314.

Aubertin, G.M., Patric, J.H., 1974. Water quality after clearcutting a small watershed in West Virginia. J. Environ. Qual. 3 (3), 243–249.

Auchmoody, L.R., Smith, H.C., 1977. Response of yellow-poplar and red oak to fertilization in West Virginia. Soil Sci. Soc. Am. J. 4, 803–807.

Bailey, S.W., Hornbeck, J.W., Driscoll, C.T., Gaudette, H.E., 1996. Calcium inputs and transport in a base-poor forest ecosystem as interpreted by Sr isotopes. Water Resour. Res. 32 (3), 707–719.

Barrett, J.W., 1995. Regional Silviculture of the United States. Wiley, New York, 643 pp.

Berger, T.W., Glatzel, G., 1994. Deposition of atmospheric constituents and its impact on nutrient budgets of oak forests (*Quercus petraea* and *Quercus robur*) in lower Austria. For. Ecol. Manage. 70, 183–193.

Boerner, R.E.J., Sutherland, E.K., 1995. Nitrogen dynamics in oak forest soils along a historical deposition gradient. In: Gottschalk, K.W., Fosbroke, S.L. (Eds.), Proceedings of the 10th Central Hardwood Forest Conference, Morgantown, WV, 5–8 March 1995. Gen. Tech. Rep. NE-197. USDA For. Serv., Northeast For. Exp. Stn., Radnor, PA, pp. 523–533.

Boyle, J.R., 1973. Forest soil chemical changes following fire. Commun. Soil Sci. Plant Anal. 4, 364–369.

Boyle, J.R., Ek, A.R., 1972. An evaluation of some effects of bole and branch pulpwood harvesting on site macronutrients. Can. J. For. Res. 2, 407–412.

Boyle, J.R., Phillips, J.J., Ek, A.R., 1973. Whole-tree harvesting: nutrient budget evaluation. J. For. 71, 760–762.

Brandrud, T.R., Timmerman, V., 1998. Ectomycorrhizal fungi in the NITREX site at Gardsjon, Sweden: below and above-ground responses to experimentally-changed nitrogen inputs 1990–1995. For. Ecol. Manage. 101, 207–214.

Braun, E.L., 1950. Deciduous Forests of Eastern North America. Hafner, New York, 596 pp.

Burger, J.A., Kelting, D.L., 1998. Soil quality monitoring for assessing sustainable forest management. In: Davidson, E.A., Adams, M.B., Ramakrishna, K. (Eds.), The Contribution of Soil Science to the Development and Implementation of Criteria and Indicators of Sustainable forest Management, Soil Science Society of America, Special Publication No. 53, pp. 17–52.

Carson, W.P., Barrett, G.W., 1988. Succession in old-field plant communities: effects of contrasting types of nutrient enrichment. Ecology 69, 984–994.

Castro, M.S., Peterjohn, W.T., Melillo, J.M., Steudler, P.A., Gholz, H.L., Lewis, D., 1994. Effects of nitrogen fertilization on the fluxes of $N_2O$, $CH_4$, and $CO_2$ from soil in a Florida slash pine plantation. Can. J. For. Res. 24, 9–13.

Chao, T.T., Harward, M.E., Fang, S.F., 1962. Adsorption and desorption phenomenon of sulfate ions in soils. Soil Sci. Soc. Am. Proc. 26, 234–237.

Clark, J.S., Macklin, E., Wood, L., 1998. Stages and spatial scales of recruitment: limitations in southern Appalachian forests. Ecol. Monogr. 68, 213–235.

Cronan, C.S., Grigal, D.F., 1995. Use of calcium/aluminum ratios as indicators of stress in forest ecosystems. J. Environ. Qual. 24, 209–226.

Cronan, C.S., Reynolds Jr., R.C., Lang, G.E., 1978. Forest floor leaching: contributions from mineral, organic and carbonic acids in New Hampshire subalpine forests. Science 200, 209–311.

Crow, T.R., Mroz, G.D., Gale, M.R., 1991. Regrowth and nutrient accumulation following whole-tree harvesting of a maple-oak forest. Can. J. For. Res. 21, 1305–1315.

De Vries, W., 1991. Methodologies for the assessment and mapping of critical loads and the impact of abatement strategies on forest soil. The Windand Staring Centre for Integrated Land, Soil and Water Research Report 46, DLO, Wageningen, The Netherlands, 109 pp.

De Vries, W., 1993. Average critical loads for nitrogen and sulfur and its use in acidification abatement policy in The Netherlands. Water, Air Soil Pollut. 68, 399–434.

DeWalle, D.R., Pionke, H.B., 1996. Nitrogen export from forest land in the Chesapeake Bay region. In: Proceedings of the 1994 Chesapeake Bay Research Conference, Chesapeake Bay Corporation, Edgewater, MD, pp. 649–655.

DeWalle, D.R., Tepp, J.S., Pickens, C.J., Edwards, P.J., Sharpe, W.E., 1995. Tree-ring chemistry response in black cherry to ammonium sulfate fertilization at two West Virginia sites. In: Gottschalk, K.W., Fosbroke, S.L. (Eds.), Proceedings of the 10th Central Hardwood Forest Conference, Morgantown, WV, 5–8 March 1995. Gen. Tech. Rep. NE-197. USDA Forest Service, Northeast. For. Exp. Stn., Radnor, PA, pp. 179–187.

Doran, J.W., Coleman, D.C., Bezdicek, D.F., Stewart, B.A., 1994. Defining soil quality for a sustainable environment. Soil Science Society of America, Special Publication 35, Madison, WI.

Droyan, J.R., Sharpe, W.E., 1997. Long-term changes in forest soil acidity in Pennsylvania, USA. Water, Air Soil Pollut. 95, 299–311.

Edwards, P.J., Helvey, J.D., 1991. Long-term ionic increases from a central Appalachian forested watershed. J. Environ. Qual. 20, 250–255.

Fajvan, M.A., Grushecky, S.T., Hassler, C.C., 1998. The effects of harvesting practices on West Virginia's wood supply. J. For. 96, 33–39.

Falkengren-Grerup, U., Ericksson, H., 1990. Changes in soil, vegetation and forest yield between 1947 and 1988 in beech and oak sites of southern Sweden. For. Ecol. Manage. 38, 37–53.

Federer, C.A., Hornbeck, J.W., Tritton, L.M., Martin, C.W., Pierce, R.S., Smith, C.T., 1989. Long-term depletion of calcium and other nutrients in eastern US forests. Environ. Manage. 13, 593–601.

Fenn, M.R., Poth, M.A., Aber, J.D., Baron, J.S., Bormann, B.T., Johnson, D.W., Lemly, A.D., McNulty, S.G., Ryan, D.F., Stottlemeyer, R., 1998. Nitrogen excess in North American ecosystems: predisposing factors, ecosystem responses and management strategies. Ecol. Appl. 8 (3), 706–733.

Finley, J.C., Jones, S.B., Pell, J., 1997. Northern Appalachian forest: how are we doing? In: Proceedings of the 1997 Penn State School of Forest Resources Issues Conference Forest Sustainability: Whats it all About? Penn State University, University Park, PA, pp. 34–43.

Foster, D.R., Aber, J.D., Melillo, J.M., Bowden, R.D., Bazzaz, F.A., 1997. Forest response to disturbance and anthropogenic stress. BioScience 47, 437–445.

Fuller, R.D., Driscoll, C.T., Lawrence, G.B., Nodvin, S.C., 1987. Processes regulating sulphate flux after whole-tree harvesting. Nature 325, 707–710.

Galloway, J.N., Schlesinger, W.H., Levy, J., Michaels, A., Schnoor, J.L., 1995. Nitrogen fixation: anthropogenic enhancement, environmental response. Global Biogeochem. Cycles 9, 235–252.

Gilliam, F.S., Turill, N.L., Aulick, S.D., Evans, D.K., Adams, M.B., 1994. Herbaceous layer and soil response to experimental acidification in a central Appalachian hardwood forest. J. Environ. Qual. 23, 835–844.

Gilliam, F.S., Adams, M.B., Yurish, B.M., 1996. Ecosystem nutrient responses to chronic nitrogen inputs at Fernow Experimental Forest, West Virginia. Can. J. For. Res. 26, 196–205.

Graveland, J., van de Wal, R., van Balen, J.H., van Noordwijk, A.J., 1994. Poor reproduction in forest passerines from decline of snail abundance on acidified soil. Nature 368, 446–448.

Green, R.E., 1998. Long-term decline in the thickness of eggshells of thrushes, *Turdus* spp., in Britain. Proc. R. Soc. London B 263, 679–684.

Harrison, R.B., Johnson, D.W., Todd, D.E., 1989. Sulfate adsorption and desorption reversibility in a variety of forest soils. J. Environ. Qual. 18, 419–426.

Hedin, L.O., Likens, G.E., 1996. Atmospheric dust and acid rain. Sci. Am. 88–92.

Helsinki Process, 1994. In: Proceedings of the Ministerial Conferences and Expert Meetings, Liaison Office of the Ministerial Conferences on the Protection of Forest in Europe, Helsinki, Finland.

Herlihy, A.R., Kaufmann, P.R., Church, M.R., Wigington Jr., P.J., Webb, J.R., Sale, M.J., 1993. The effects of acidic deposition on streams in the Appalachian Mountain and Piedmont region of the Mid-Atlantic United States. Water Resour. Res. 29, 2687–2703.

Hornbeck, J.W., 1992. Comparative impacts of forest harvest and acid precipitation on soil and streamwater acidity. Environ. Pollut. 77, 151–155.

Hüttle, R.D., 1990. Nutrient supply and fertilizer experiments in view of N saturation. Plant Soil 128, 45–58.

Jardine, P.M., Weber, N.L., McCarthy, J.F., 1989. Mechanisms of dissolved organic carbon adsorption in soil. Soil Sci. Soc. Am. J. 53, 1378–1385.

Jencks, E.M., 1969. Some chemical characteristics of the major soil series of West Virginia. West Virginia University Agricultural Experiment Station Bulletin 5821, Morgantown, WV.

Jenkins, A.B., Sencindiver, J.C., Bhumbla, D.K., 1998. Biogeochemical relationships of calcium and magnesium in high elevation forest soil of West Virginia. Agronomy Abstracts.

Johnson, A.J., Andersen, S.B., Siccama, T.G., 1994. Acid rain and soils of the Adirondacks. I. Changes in pH and available calcium, 1930–1984. Can. J. For. Res. 24, 39–45.

Johnson, C.E., Romanowicz, R.B., Siccama, T.G., 1997. Conservation of exchangeable cations after clear-cutting of a northern hardwood forest. Can. J. For. Res. 27, 859–868.

Johnson, D.W., Lindberg, S.E., 1992. Atmospheric Deposition and Forest Nutrient Cycling. Springer, New York, 707 pp.

Johnson, D.W., Todd, D.E., 1983. Some relationships among aluminum, carbon, and sulfate in a variety of forest soils. Soil Sci. Soc. Am. J. 47, 792–800.

Johnson, D.W., Todd, D.E., 1998. Harvesting effects on long-term changes in nutrient pools of mixed oak forest. Soil Sci. Soc. Am. J. 62, 1725–1735.

Johnson, D.W., Henderson, G.S., Todd, D.E., 1988. Changes in nutrient distribution in forests and soils of Walker Branch watershed over an eleven-year period. Biogeochemistry 5, 275–293.

Johnson, D.W., Cresser, M.S., Nilsson, S.I., Turner, J., Ulrich, B., Binkley, D., Cole, D.W., 1991. Soil changes in forest ecosystems: evidence for and probable causes. Proc. R. Soc. Edinburgh 97B, 81–116.

Johnson, D.W., Swank, W.T., Vose, J.M., 1995. Effects of liming on soils and steamwaters in a deciduous forest: comparison of field results and simulations. J. Environ. Qual. 24, 1105–1117.

Joslin, J.D., Wolfe, M.H., 1994. Foliar deficiencies of mature southern Appalachian red spruce determined from fertilizer trials. Soil Sci. Soc. Am. J. 58, 1572–1579.

Karr, J.R., 1993. Measuring biological integrity: lessons from streams. In: Woodley, S, Kay, J., Francis, G. (Eds.), Ecological Integrity and the Management of Ecosystems. St. Lucie Press, NY, pp. 83–104.

Kellner, O., Redbo-Torstensson, P., 1995. Effects of elevated nitrogen deposition on the field-layer vegetation in coniferous forests. Ecol. Bull. (Copenhagen) 44, 227–237.

Klopatek, J.M., Harris, W.F., Olson, R.J., 1980. A regional ecological assessment approach to atmospheric deposition: effects on soil systems. In: Shriner, D.S., Richmond, C.R., Lindberg, S.E. (Eds.), Atmospheric Sulfur Deposition. Ann Arbor Science, Ann Arbor, MI, pp. 539–553.

Knoepp, J.D., Swank, W.T., 1997. Long-term effects of commercial sawlog harvest on soil cation concentrations. For. Ecol. Manage. 93, 1–7.

Long, R.P., Horsley, S.B., Lilja, P.R., 1997. Impact of forest liming on growth and crown vigor of sugar maple and associated hardwoods. Can. J. For. Res. 27, 1560–1573.

Lündborg, A., 1997. Reducing the nitrogen load: whole-tree harvesting. Ambio. 26, 387–393.

Lusk, M. G., 1998. Sulfate dynamics and base cation release in a high elevation Appalachian forest soil. M.S. Thesis, Department of Crop and Soil Environmental Science, Virginia Polytechnic Institute and State University.

Lynch, J.A., Bowersox, V.C., Grimm, J.W., 1996. Trends in precipitation chemistry in the United States, 1983–94: an analysis of the effects in 1995 of Phase I of the Clean Air Act Amendments of 1990, Title IV. Nat. Acid Precip. Assessment Program, Washington, DC.

Mann, L.K., Johnson, D.W., West, D.C., Cole, D.W., Hornbeck, J.W., Martin, C.W., Riekerk, H., Smith, C.T., Swank, W.T., Tritton, L.M., vanLear, D.H., 1988. Effects of whole-tree and stem-only clearcutting on postharvest hydrologic losses, nutrient capital and regrowth. For. Sci. 34, 412–428.

Markewitz, D., Richter, D.D., Allen, H.L., Urrego, J.B., 1998. Three decades of observed soil acidification in the Calhoun Experimental Forest: has acid rain made a difference? Soil Sci. Soc. Am. J. 62, 1428–1439.

McFee, W.W., 1980. Sensitivity of soil regions to long-term acid precipitation. In: Shriner, D.S., Richmond, C.R., Lindberg, S.E. (Eds.), Atmospheric Sulfur Deposition. Ann Arbor Science, Ann Arbor, MI, pp. 495–505.

McNulty, S.G., Aber, J.D., Newman, S.D., 1996. Nitrogen saturation in high elevation spruce-fir stand. For. Ecol. Manage. 84, 109–121.

Meiwes, K.J., Khanna, P.K., Ulrich, B., 1986. Parameters for describing soil acidification and their relevance to the stability of forest ecosystems. For. Ecol. Manage. 15, 161–179.

Miller, G.W., Kochenderfer, J.N., 1998. Maintaining species diversity in the central Appalachians. J. For. 96, 28–33.

Miller, E.K., Blum, J.D., Friedland, A.J., 1993. Determination of soil exchangeable-cation loss and weathering rates using Sr isotopes. Nature 362, 438–441.

Mitchell, M.J., 1992. Retention or loss of sulfur for IFS sites and evaluation of relative importance of processes. In: Johnson, D.W., Lindberg, S.E. (Eds.), Atmospheric Deposition and Nutrient Cycling. Springer, New York, pp. 129–133.

Mohren, G.M.J., van den Burg, J., Burger, F.W., 1986. Phosphorus deficiency induced by nitrogen inputs in Douglas fir in The Netherlands. Plant Soil 95, 191–200.

Montreal Process, 1995. Criteria and indicators for the conservation and sustainable management of temperate and boreal forests. Canadian Forest Service, Catalogue F142-238/1995E, Canadian Forest Service, Hull, Que.

Morris, S.J., Boerner, R.E.J., 1998. Interactive influences of silvicultural management and soil chemistry upon soil microbial abundance and nitrogen mineralization. For. Ecol. Manage. 103, 129–139.

Nambiar, E.K.S., 1996. Sustained productivity of forests is a continuing challenge to soil science. Soil Sci. Soc. Am. J. 60, 1629–1642.

National Research Council, 1998. Forested Landscapes in Perspective: Prospects and Opportunities for Sustainable Management of America's Nonfederal Forests. National Academy Press, Washington, DC, 249 pp.

NAPAP, 1990. Acidic deposition: state of science and technology. National Acid Precipitation Assessment Program Report 18. US Government Printing Office, Washington, DC.

Nodvin, S.C., Driscoll, C.T., Likens, G.E., 1986. The effect of pH on sulfate adsorption by a forest soil. Soil Sci. 142, 69–75.

Noss, R.R., 1990. Indicators for monitoring biodiversity: a heirarchical approach. Conserv. Biol. 4, 355–364.

Olson, R.J., Johnson, D.W., Shriner, D.S., 1982. Regional assessments of potential sensitivity of soil in the eastern United States to acid precipitation. ORNL/TM-9374, Oak Ridge National Laboratory, TN.

Peterjohn, W.T., Adams, M.B., Gilliam, F.S., 1996. Symptoms of nitrogen saturation in two central Appalachian hardwood forest ecosystems. Biogeochemistry 35, 507–522.

Powers, R.D., Alban, D.H., Miller, R.E., Tiarks, A.E., Wells, C.G., Avers, P.E., Cline, R.G., Fitzgerald, R.O., Loftus Jr., N.S., 1990. Sustaining site productivity in North American forests: problems and prospects. In: Gessel, S.P., Lacate, D.S., Weetman, G.F., Powers, R.F. (Eds.), Proceedings of the 7th North American Forest Soils Conference, Faculty of Forestry Publication, University British Columbia, Vancouver, BC, pp. 49–79.

Raison, R.J., 1979. Modification of the soil environment by vegetation fires: a review. Plant Soil 51, 73–108.

Raynal, D.J., Foster, N.W., Mitchell, M.J., Johnson, D.W., 1992. Eastern hardwoods. In: Johnson, D.W., Lindberg, S.E. (Eds.), Acidic Deposition and Forest Nutrient Cycling. Springer, New York, pp. 526–534.

Rennie, P.J., 1990. Some threats to sustaining forest yields in North America: research challenges. In: Gessel, S.P., Lacate, D.S., Weetman, G.F., Powers, R.F. (Eds.), Proceedings of the 7th North American Forest Soils Conference, Faculty of Forestry Publication, University British Columbia, Vancouver, BC, pp. 6–22.

Reuss, J.O., Johnson, D.W., 1986. Acid Deposition and the Acidification of Soils and Waters. Springer, New York.

Rochelle, B.P., Church, M.R., David, M.B., 1987. Sulfur retention at intensively studied sites in the US and Canada. Water, Air Soil Poll. 33, 73–83.

Schuler, T.M., 1997. Long-term trends of woody species diversity in a central Appalachian forest. In: Proceedings of the First Biennial North American Forest Ecology Workshop, 24–26 June 1997. Society of American Foresters, Raleigh, NC (Abstract).

Schuler, T.M., Miller, G.W., 1995. Shelterwood treatments fail to establish oak reproduction on mesic forest sites in West Virginia. In: Gottschalk, K.W., Fosbroke, S.L. (Eds.), Proceedings of the 10th Central Hardwood Conference, Morgantown, WV, 5–8 March 1995. Gen. Tech. Rep. NE-197. USDA Forest Service, Northeastern For. Exp. Stn., Radnor, PA, pp. 375–387.

Singh, B.R., 1984. Sulfate sorption by acid forest soil. 2. Sulfate adsorption isotherms with and without organic matter and oxides of aluminum and iron. Soil Sci. 138, 294–297.

Smith, C.T., McCormack, M.L.J., Hornbeck, J.W., Martin, C.W., 1986. Nutrient and biomass removals from a red spruce-balsam fir whole-tree harvest. Can. J. For. Res. 16, 381–388.

Smith, H.C., 1993. Regenerating oaks in the central Appalachians. In: Loftis, D., McGee, C.E. (Eds.), Oak Regeneration: Serious Problems, Practical Recommendations. Symposium Proceedings, Knoxville, TN, 8–10 September 1992. Gen Tech. Rep. SE-84. USDA Forest Service, Southeast. For. Exp. Stn., Asheville, NC, pp. 211–221.

Smith, H.C., Della-Bianca, L., Fleming, H., 1993. Appalachian Mixed Hardwoods. In: Burns, R.M., (Comp.), Silvicultural systems for the major forest types of the United States. Agricultural Handbook No. 445. USDA Forest Service, Washington, DC, pp. 141–144.

Steudler, P.A., Bowden, R.D., Melillo, J.M., Aber, J.D., 1989. Influence of nitrogen fertilization on methane uptake in temperate forest soils. Nature 341, 314–316.

Stoyenoff, J., Witter, J., Leutscher, B., 1998. Forest Health in the Mid-Atlantic States. University of Michigan School of Natural Resources and Environment. Ann Arbor, MI, unnumbered publication.

Strain, B.R., Thomas, R.B., 1992. Field measurements of $CO_2$ enhancement and climate change in natural vegetation. Water, Air Soil Pollut. 64, 45–60.

Tabatabai, M.A., 1985. Effect of acid rain on soils. CRC Crit. Rev. Environ. Control 15, 65–110.

Tritton, L.M., Martin, C.W., Hornbeck, J.W., Pierce, R.S., 1987. Biomass and nutrient removals from commercial thinning and whole-tree clearcutting of central hardwoods. Environ. Manage. 11 (5), 659–666.

Turner, R.S., Olson, R.J., Brandt, C.C., 1986. Areas having soil characteristics that may indicate sensitivity to acidic deposition under alternative forest damage hypotheses. ORNL/TM-9917, Oak Ridge National Laboratory, TN.

Ulrich, B., 1986. Natural and anthropogenic components of soil acidification. Z. Pflanzenernähr Bodenkde 149, 702–717.

van Breemen, N., 1973. Dissolved aluminum in acid sulfate soils and in acid mine waters. Soil Sci. Soc. Am. Proc. 37, 694–697.

van Breeman, N., Burrough, P.A., Velthorst, E.J., van Dobben, H.F., DeWit, T., Ridder, T.B., Reijnders, H.F.R., 1982. Soil acidification from atmospheric ammonium sulphate in forest canopy throughfall. Nature 299, 548–550.

Vance, E.D., 1996. Land application of wood-fired and combination boiler ashes: an overview. J. Environ. Qual. 25, 937–944.

Vanclay, J.K., 1996. Assessing the sustainability of timber harvests from natural forests: limitations of indices based on successive harvests. J. Sustain. For.3, 47–58.

Van Miegroet, H., Cole, D.W., 1984. The impact of nitrification on soil acidification and cation leaching in a red alder forest. J. Environ. Qual. 13, 586–590.

Van Miegroet, H., Cole, D.W., Foster, N.W., 1992. Nitrogen Distribution and Cycling. In: Johnson, D.W., Lindberg, S.E. (Eds.), Atmospheric Deposition and Nutrient Cycling. Springer, New York, pp. 178–196.

Vitousek, P.M., Reiners, W.A., 1975. Ecosystem succession and nutrient retention: a hypothesis. BioScience 25, 376–381.

Voigt, G.K., 1979. Acid precipitation, forest ecosystems, and intensive harvesting. In: Proceedings on the Impact of Intensive Harvesting on Forest Nutrient Cycling, State University of New York, College of Environmental Science and Forestry, Syracuse, NY, 13–16 August 1979, pp. 33–48.

Wagenet, R.J., Hutson, J.L., 1997. Soil quality and its dependence on dynamic physical processes. J. Environ. Qual. 26, 41–48.

Weetman, G.F., Webber, B., 1972. The influence of wood harvesting on the nutrient status of two spruce stands. Can. J. For. Res. 4, 530–535.

West Virginia Division of Forestry, 1990. Forest Resources of West Virginia, 1990. West Virginia Division of Forestry, Charleston, WV, 20 pp.

Williard, K.W., DeWalle, D.R., Edwards, P.J., Schnabel, R.R., 1997. Indicators of nitrate export from forested watersheds of the mid-Appalachians, United States of America. Global Biogeochem. Cycles 11 (4), 649–656.

Wilmot, T.R., Ellsworth, D.S., Tyree, M.T., 1995. Base cation fertilization and liming effects on nutrition and growth of Vermont sugar maple stands. For. Ecol. Manage. 84, 123–134.