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Acidic deposition and sustainable forest management in the central Appalachians, USA

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Abstract

Long-term productivity of mixed-species forests in the central Appalachian region of the United States may be threatened by changes in base cation availability of the soil. These changes may be due to increased intensity of harvest removals and a shift toward shorter rotations that result in increased removal of calcium and magnesium in aboveground biomass, and through altered nutrient cycling. Atmospheric deposition, particularly of nitrogen, also affects nutrient cycling by increasing leaching of base cations from the soil in soil solution, and by altering the rate of nutrient cycling. In this paper, I present evidence for nitrogen saturation of forested ecosystems in the central Appalachians, discuss the symptoms and implications for productivity as well as the effects of changes in timber harvesting rates and intensity on nutrient cycling, and evaluate the potential for interaction between these two stressors. Indicators of changes in productivity of extensively managed forests due to base cation depletion are also discussed.

Keywords: Acid deposition; Calcium; Forest productivity; Nutrient cycling

1. Introduction

From 1870–1920, $\approx 70 \times 10^6$ m^3 (30×10^9 board feet) of timber were removed from $\approx 4.1 \times 10^6$ ha in the state of West Virginia, USA. At the peak of the timber boom in 1910, 3.5 million m^3 were cut in the state. At that time, little thought was given to sustainability or the future availability of the forest resource. It was assumed that forests would grow back and replenish themselves, and that productivity would be maintained. To a certain extent this has happened. Forests now cover 76% of West Virginia (ca. 5×10^6 ha), and productivity is high, averaging 40 m^3 ha^{-1} (6500 board feet per acre). These figures have been used as evidence of the sustainable long-term productivity of these forests.

However, as explained by Powers et al. (1994), defining and measuring sustainable productivity of managed forest ecosystems is not a straightforward task. This is particularly true in extensively managed forests, due to longer rotations, to natural regeneration which often results in a mix of species, and to a continuously changing environment. The multiple ways in which productivity is measured further complicates the evaluation of sustainability. For example, West Virginia's forests today appear to be productive and sustainable on the basis of extent and per hectare timber yield, as described above. However, species composition of these second growth, naturally-regenerated stands differs from that prior to the turn of the century logging. There are several factors contributing

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to this change in diversity. In the first half of this century, American chestnut (*Castanea dentata*) declined from 40% to 50% of the overstory (Frothingham, 1924; Keever, 1953) to near zero due to the fungal blight *Endothia parasitica*. Slippery elm (*Ulnus rubra*) and butternut (*Juglans cinerea*) have also been decimated by disease (Gibbs, 1978; Ostry et al., 1994). Red spruce (*Picea rubens*) was more extensive prior to 1900 (Brooks, 1911; DiGiovanni, 1990). More recently red oak (*Quercus rubra*) constituted ca. 20% of the sawtimber volume in West Virginia cut in 1995.[1] But throughout the central hardwood region (eastern and central United States, and southern Ontario and Quebec, Canada), come reports of difficulty and failure in regenerating northern red oak on mesic sites. Schuler and Miller (1995) contend that "A continuation of current management practices will likely result in a widespread reduction of oak on mesic sites throughout the region." Is this a sustainable forest ecosystem then, if species composition changes significantly and, in some cases, irreversibly?

This question assumes more than a philosophical importance given current harvesting levels in the central Appalachians. In West Virginia alone, $\approx 3.1 \times 10^9$ m^3 were harvested in 1996, twice that of 1989[2]. The product mix is also changing; what was predominantly sawlogs and veneer now includes more pulp and other fiber, resulting in utilization of smaller diameter trees of less valuable species. Along with the usual concerns about the effects of timber harvesting, e.g., water quality, aesthetics, regeneration, is a new one: sustainability.

The traditional definition of sustainable productivity invoked a constant supply of wood products over a specified period of time. Growth rates and volume production were used to measure the progress toward sustainable productivity. In the last 25 years or so, the products required from forests have changed and no longer is wood production the only acceptable measure of productivity. Recreational opportunities, hydrologic control, ecosystem function, biodiversity, wildlife habitat and carbon sequestration are some of the new 'products' of forests and sustainable productivity of these must also be ensured. Measuring the sustainability of the forest requires an appreciation of these various products and incorporation of them into assessment where possible.

We need to develop meaningful measures of productivity and sustainability for eastern hardwood forests, which may include as many as 20 commercial tree species (Smith et al., 1993) and many more noncommercial, shrub, and herb species. Vanclay (1996) pointed out the deficiencies in using growth indices from successive rotations of natural forests as an indicator of productivity and reminded us of the importance of measuring condition and vitality of residual resources. Measuring net primary productivity (NPP) is one way of evaluating productivity in a mixed-species system that allows comparisons among regions (Grier et al., 1989). It is also important, particularly in the eastern United States, to consider species diversity and ecological function and integrity when measuring productivity. For example, if harvesting alters species composition to favor species such as yellow-poplar (*Liriodendron tulipifera*) or black locust (*Robinia pseudoacacia*), then NPP may increase; conversely, dominance by sugar maple (*Acer saccharum*) would result in lower NPP. Finally, because the soil resource plays such a critical role in determining aboveground productivity, it is essential to consider its physical and chemical condition.

In this paper, I describe two potential threats to the sustainability of central Appalachian forests: (1) increased intensity of harvesting that results in greater removal of biomass and nutrients from forest sites; and (2) acidic deposition. I examine evidence of elevated base-cation leaching from nutrient-poor soils from both acidic deposition and harvesting, and the potential for interactions between the two. Much of the discussion focuses on nitrogen (N) and the base nutrients calcium (Ca) and magnesium (Mg). I also describe a study we have initiated to evaluate these threats, and describe some of the indicators of sustainable productivity that we are using.

2. Effects of harvesting on nutrient pools

Whenever organic material is removed from a site, a net loss of nutrients occurs. Thus timber harvest

[1]Maxey, W.R. 1997. West Virginia's Forest Resource: 1994–1995 Trends. Presentation to the Society of American Foresters. January 30–31, 1997. Charleston, WV.

[2]Same as footnote 1.

Table 1
Harvest removals of nutrients (kg ha^{-1}) and biomass (Mg ha^{-1}) from eastern U.S. hardwood forests

Location	N	P	K	Ca	Biomass	Reference
Sawtimber sale (bole only)						
Coweeta, NC	58	7	48	130	43	Mann et al., 1988
Oak Ridge, TN	110	7	36	410	64	Mann et al., 1988
Cockaponset, CT	162	5	108	442	121	Mann et al., 1988
Fernow, WV	97	7	—	283	51	Patric and Smith, 1975
Whole-tree harvest (above-ground only)						
Coweeta, NC	277	41	216	544	178	Mann et al., 1988
Oak Ridge, TN	323	23	128	1090	175	Mann et al., 1988
Cockaponset, CT	273	19	162	530	158	Mann et al., 1988
Fernow, WV	180	13	—	523	94	Patric and Smith, 1975
Baroga, WI	143	17	131	307	100	Crow et al., 1991

results in removal of nutrients from the site, with the nutrient losses approximately proportional to the volume removed (Grier et al., 1989). Table 1 shows biomass and nutrient removals due to harvesting from several mature, second-growth, eastern U.S. hardwood forests. These data suggest considerable nutrient losses from whole-tree harvesting (removal of bole, branches, and tops of trees; stumps were not removed) and consistently greater removal of nutrients from the site relative to the sawtimber sale. Ca losses are particularly large, because of the high concentrations in wood. Thus, one obvious means of reducing base cation losses from harvesting may be to use a less intensive form of management than whole-tree harvesting, e.g., partial removals, thinnings. Tritton et al. (1987) reported that only 4% of the total organic matter of a site, and <2% of the total nutrients were removed in a commercial thinning of a central hardwood stand in Connecticut. However, according to Patric and Smith (1975), repeated light cuts, such as commercial thinnings and selection harvests, may remove more nutrients from a site during a rotation than a single cut at the end of the rotation (Fig. 1). For example, a commercial clearcut (removal of all trees larger than 12.5 cm in diameter) removed an estimated 280 kg Ca ha^{-1} at the end of a rotation (Fig. 1(B)). An extensive selection cut (selected trees >27.5 cm diameter, to meet species composition and stand structure goals) removed only 120 kg Ca per 10 year cutting cycle (Fig. 1(A)), but over a 75-year rotation, removed an estimated 930 kg Ca ha^{-1} (Fig. 1(B)). Even a diameter-limit cut every 20 years removed more of all of the nutrients listed than a commercial clearcut, over the course of the rotation. These data should be interpreted with caution, however, because Patric and Smith's calculations do not reflect changes in nutrient content with stand age, nor changes in internal cycling and storage.

In addition to removal of nutrients in biomass, harvesting also affects the rate of nutrient cycling. Immediately after harvesting, leaching losses may increase due to decreased transpirational demands by the remaining vegetation and increases in soil moisture (Patric, 1973). Since leaching losses are controlled by hydrologic processes, changes in nutrient leaching will vary with the intensity of overstory removal (Hornbeck et al., 1993). In addition, nitrification often is stimulated after harvest removals, particularly clearcutting, resulting in increased nitrate leaching. Vitousek and Melillo (1979) reported nitrate losses from clearcutting ranging from negligible to 20 times that in uncut stands, with generally greater losses from deciduous than conifer forests. Harvest-induced leaching losses from hardwood forests in the eastern United States have been estimated to range from 6 to 60 kg N ha^{-1} $year^{-1}$, 28–48 kg Ca ha^{-1} $year^{-1}$, and 7–16 kg Mg ha^{-1} $year^{-1}$ (Federer et al., 1989), though much lower losses were recorded at Coweeta Hydrologic Laboratory, NC (2.8 kg N ha^{-1} $year^{-1}$, 2.2 kg Ca ha^{-1} $year^{-1}$, and 0.8 kg Mg ha^{-1} $year^{-1}$ [calculated from Swank, 1988]) and Fernow Experimental Forest, WV (4–9 kg N ha^{-1} $year^{-1}$, 0.7 kg Ca ha^{-1} $year^{-1}$, and 0.4 kg Mg ha^{-1} $year^{-1}$ [Patric, 1980; Kochenderfer

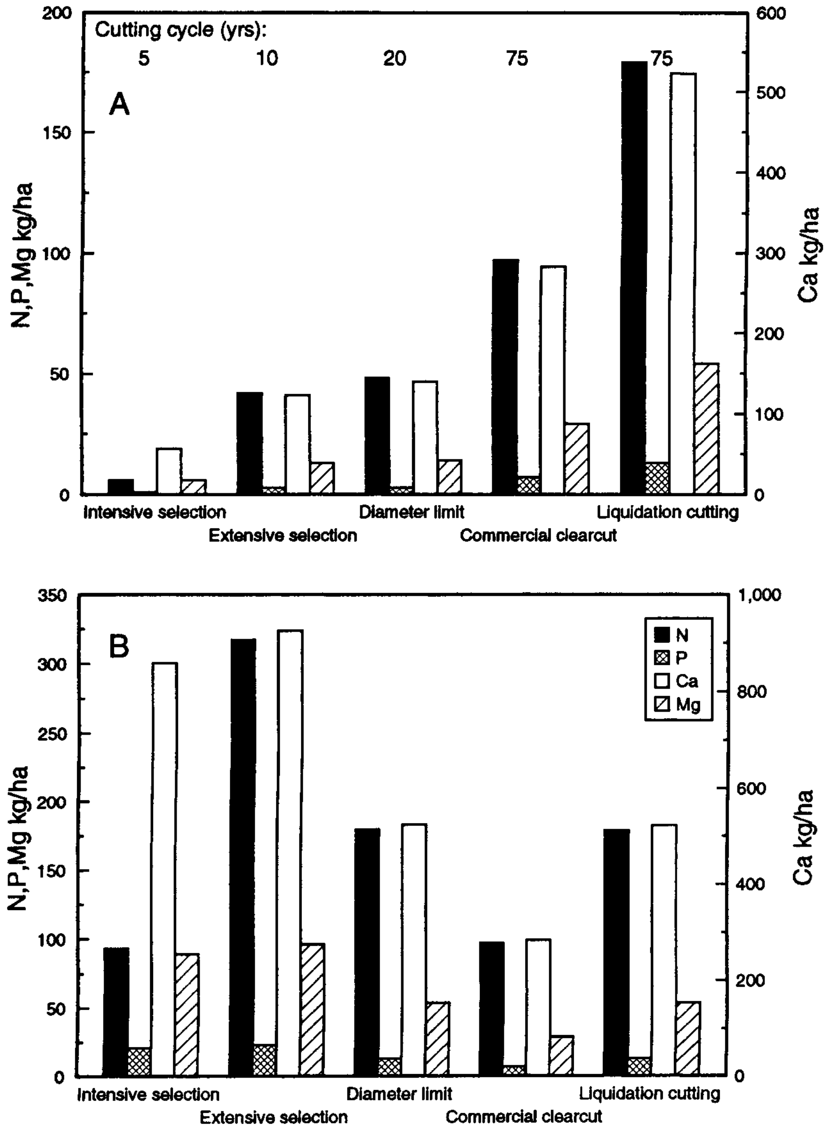

Fig. 1. Nutrients removed (kg ha^{-1}) by different silvicultural techniques per (A): cutting cycle, and (B): 75-year rotation. Intensive selection = selected trees >12.5 cm DBH removed; extensive selection = selected trees >27.5 cm DBH removed; diameter limit = all salable trees over 43 cm DBH removed; commercial clearcutting = all salable trees over 12.5 cm DBH removed; liquidation cutting = all salable trees, branches, and culls removed (analogous to whole-tree harvesting). Calculated from data in Patric and Smith (1975).

and Edwards, 1991]). Note, however, that leaching losses from whole-tree harvesting may be less than from conventional clearcutting, due to the availability of slash and logging residues (and their nutrients) with clearcutting.

Using the model of Federer et al. (1989), I calculated the change in nutrients from eastern hardwood stands due to harvesting. Briefly, the percent change (C) in total pool size of a nutrient during a rotation of 80 years was calculated:

$$C = 100[80(I - O) - n(R + L)]/T$$

where I is the annual input, O the annual leaching output from the uncut forest, R the harvest removal, L

Table 2
Estimated percent change in total nutrient pools of eastern U.S. hardwood forests during an 80-year rotation, with no harvesting, one whole-tree harvest at year 80, or two whole-tree harvests at year 40 and year 80. See text for details of calculations

	No harvest	One harvest	Two harvests	Reference
Nitrogen				
Coweeta, NC	+8	+4	+1	Monk and Day, 1988, Huff et al., 1978
Cockaponset, CT	+13	+7	+1	Federer et al., 1989
Walker Branch, TN	+17	+9	+0.5	Federer et al., 1989
Fernow, WV	+19	+12	+4	Auchmoody, 1972, Adams et al., 1995
Calcium				
Coweeta, NC	−4	−20	−36	Monk and Day, 1988, Huff et al., 1978
Cockaponset, CT	−15	−28	−41	Federer et al., 1989
Walker Branch, TN	−5	−22	−40	Federer et al., 1989
Fernow, WV	0	−15	−26	Adams et al., 1995, Adams et al., 1997, Patric, 1980, Kochenderfer and Edwards, 1991
Magnesium				
Coweeta, NC [a]	−32	−40	−47	Monk and Day, 1988
Cockaponset, CT	+4	−4	−4	Federer et al., 1989
Walker Branch, TN	−5	−6	−7	Federer et al., 1989
Phosphorus				
Coweeta, NC [a]	+2.5	−16	−34	Monk and Day, 1988
Cockaponset, CT	0	−2	−4	Federer et al., 1989
Walker Branch, TN	0	−2	−4	Federer et al., 1989

[a] Calculations based on exchangeable soil levels, rather than total soil values.

the additional harvest-induced leaching, T the initial total nutrient pool, and n the number of harvests during the 80-year rotation. I used $n = 0$ (no cutting), $n = 1$ to represent an 80-year rotation, the current average rotation for central Appalachian hardwoods, and $n = 2$ to represent a 40-year rotation. Results are shown in Table 2. Over 80 years with no harvesting, losses of up to 15% of total Ca pools are possible. With one harvest, losses range from 15% to 30% of the total Ca pool, and with two harvests (40-year rotation) losses range from ≈25% to more than 40%. Because atmospheric inputs of Ca, Mg, and phosphorus (P) are less than stream water outputs, ecosystems are not accumulating these nutrients. Note, however, these calculations do not include other sources of these nutrients, particularly weathering. Although there are no weathering data for Fernow soils, we assume that Ca and Mg inputs from weathering are low, based on the low concentrations of Ca and Mg in the bedrock (3.2% and 0.11%, respectively; DeWalle et al., 1988). We hypothesize that additional base nutrient removals via harvesting will exacerbate cation depletion. This hypothesis is not novel and mirrors the results of Federer et al. (1989). The data are consistent, showing potentially large losses, particularly of Ca. The data also suggest that nitrogen levels will not decline significantly even in relatively short rotations, particularly given predicted increases in N deposition (Galloway et al., 1995). The importance of P to productivity of hardwood forests and potential productivity problems related to removals of this nutrient require further study.

3. Effects of acidic deposition on nutrient pools

Concern over the effects of acidic deposition on forest productivity is not new. In a 1982 review of literature from the previous decade on the effects of acid rain on forest nutrient status, Johnson et al. (1982) concluded that "given sufficiently high inputs on sensitive sites, negative effects of acid rain must occur," but that acute effects were not well documented, and that long-term effects on forest nutrient status "can be either beneficial or adverse, depending on site nutrient status, silvicultural practices and amount of atmospheric inputs." The National Acid Precipitation Assessment Program (NAPAP), a 10-year study mandated by the U.S. Congress, examined the effects of acidic deposition on a variety of resources and human health. It was concluded that "With the possible and notable exception of high-elevation red spruce in the northern Appalachians, acidic deposition has not been shown to be a significant factor contributing to forest health problems in North America." However, the report also contains the admonition that "Possible future impacts resulting from changes in the fertility of sensitive forest soils should be monitored carefully" (NAPAP, 1991).

Since publication of the NAPAP report, there has been continued interest in the effects of air pollution on forest soils and nutrient cycling processes. Focus has shifted from effects of acidity and sulfur (S) to N effects, partly due to changes in emissions and partly to an improved understanding of the effects of acidic deposition on forest ecosystems.

We now know a great deal more about patterns and types of atmospheric inputs to eastern U.S. forest ecosystems. Monitoring has demonstrated the success of some clean air legislation, with decreases in emissions of sulfate and particulates in the eastern U.S. (Hameed and Dignon, 1992; Likens et al., 1996). These changes have significant, but perhaps offsetting, implications for base-nutrient cycling in forests. While decreased S deposition could result in decreased cation leaching from forest soils (due to decreased anion loading), the concurrent decrease in particulate emissions has resulted in decreased atmospheric inputs of Ca and Mg to ecosystems. Declines in Ca inputs of nearly 50% since 1965 have been reported (Hedin and Likens, 1996), and may contribute to slower recovery from acidification (Likens et al., 1996). However, N deposition is expected to increase by 25% during the next 25 years (Galloway et al., 1995), raising concerns about N saturation of forest soils.

Previously it was believed that N was limiting productivity of most forests, and that most forests used or immobilized all of the N available to them. In recent years, N saturation has become a concern relative to atmospheric deposition. N saturation is defined as the availability of ammonium and nitrate in excess of the total combined plant and microbial demand; the term refers to an ecosystem where the biota are unable to utilize all of the N that is being added to the system, either through N fixation, atmospheric N inputs or other sources (Aber et al., 1989; Stoddard, 1994). An N-saturated system may show one or more of the following symptoms: elevated concentrations of nitrate, aluminum and hydrogen in streams, frost damage or other disruptions of physiological function, increased emissions of trace gases such as nitrous oxide, and increased cation leaching from soils (Aber et al., 1989; Stoddard, 1994). Increased cation leaching, which may lead to reduced soil fertility, is of particular concern for sustainable forest productivity.

Numerous watershed-manipulation experiments in Europe and the United States have documented changes in base-cation availability with increased N and S deposition (Feger, 1992; Norton et al., 1994; Wright and Tietema, 1996; Adams et al., 1997; Emmett et al., 1997). As nitrate and sulfate anions move through the soil, cations (most often, Ca and Mg) are removed from exchange sites to maintain charge neutrality of the soil solution. At the Fernow Experimental Forest, five years of experimental additions of ammonium sulfate at twice the ambient input levels (additions of 40 kg S ha^{-1} $year^{-1}$ and 35 kg N ha^{-1} $year^{-1}$) resulted in increased leaching of nitrate from a young ($\sim$25 years old) mixed hardwood forest, along with significant increases in leaching of Ca and Mg (Adams et al., 1997). By the fifth year of treatment, mean annual export of Ca was $\approx$2–3 times greater than inputs (Fig. 2). Weathering was not measured in the Fernow study, but results from other watershed manipulation studies suggest that bedrock weathering did not increase (Norton et al., 1997). Thus, over time elevated inputs could result in significant decreases in soil fertility and nutrient deficiencies. Ca deficiencies have been identified in high elevation red spruce (McLaughlin et al., 1991) and may be linked to sugar maple decline (Long et al., 1997) on non-glaciated soils.

Of particular interest and significance are those forest ecosystems that are believed to be N saturated as a result of ambient N deposition. Forest ecosystems in the United States which are N saturated due to ambient levels of N deposition include the high-elevation spruce-fir forests of the Northeast (McNulty and Aber, 1993), mixed conifer forests of southern California (which receive some of the highest levels of N

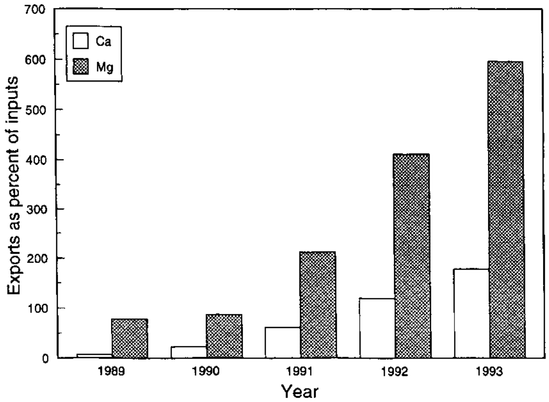

Fig. 2. Calcium and magnesium exports from the treated watershed, Fernow whole watershed acidification study, expressed as percent of inputs. Data are from Adams et al. (1997).

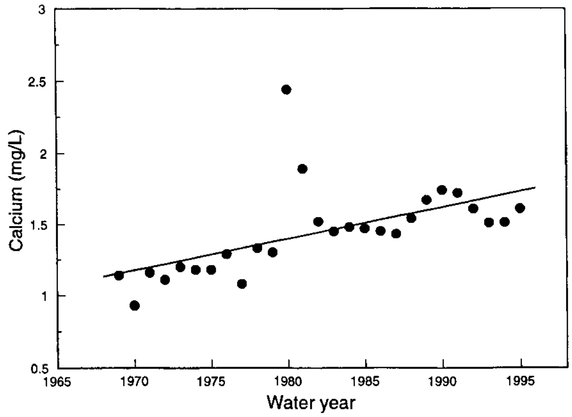

Fig. 3. Mean annual stream water calcium concentrations from the reference catchment, Fernow Experimental Forest ($r^2 = 0.33$). Data are from Peterjohn et al. (1996) and unpublished data (M.B. Adams, 1997).

deposition in the nation, mostly through dry deposition [Fenn et al., 1996]), high elevation catchments in the Rocky Mountains (Williams et al., 1996), and hardwood forests of the central Appalachian region (Adams et al., 1997).

On the Fernow Experimental Forest, an untreated reference watershed has exhibited several symptoms of N saturation (Peterjohn et al., 1996): high relative rates of net nitrification, little seasonable variability in stream water nitrate concentrations, low relative retention of inorganic N, and long-term increases in stream water concentrations of nitrate and base cations. Because the observed changes in stream water chemistry (Fig. 3) do not mirror changes in composition of precipitation over the same period, we conclude that these changes probably represent inherent modifications of the ecosystem and of some ecosystem functions (Edwards and Helvey, 1991; Peterjohn et al., 1996). These changes are attributed to the effects of ambient acidic deposition. Thus these long-term effects can occur at ambient levels.

The various stages of acidification and base-cation leaching that might be used as indicators of change are shown in the hypothesized model adapted from Norton et al. (1997) in Fig. 4. Early in the acidification, concentrations of stream water base cations increase as desorption of cations from soil minerals increases and weathering remains relatively constant. The Fer-

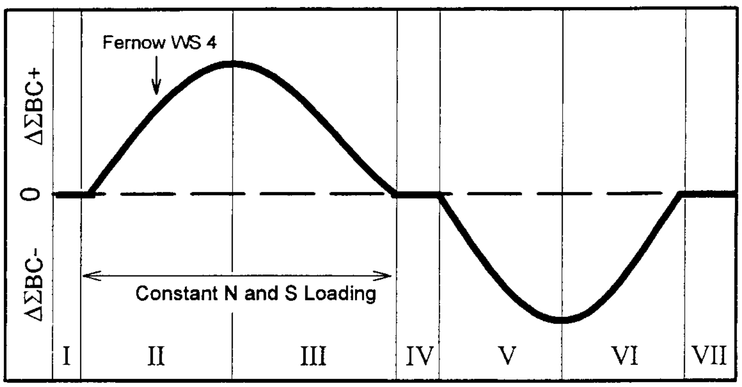

Fig. 4. Conceptual diagram of changes in stream water base cation chemistry with acidification, redrawn from Norton et al. (1997). Vertical axis represents changes in the total base cations in stream water with changes in nutrient loading.

now reference catchment probably is at this stage (Stage II, Fig. 4). As acidic loading continues, base saturation of the soil will decrease until base cations no longer are being desorbed from the soil, and Al concentrations in soil solution and stream water will increase (Stage III). This is of concern because elevated soil solution Al concentrations have been found to interfere with root growth in red oak seedlings (Joslin and Wolfe, 1989), and with Ca uptake in spruce (Shortle and Smith, 1988). Soil solution Ca : Al ratios of ≤ 1 have been suggested as indicative of potential decline or productivity loss (Cronan and Grigal, 1995), although Binkley and Högberg (1997) found no support for this hypothesis for Norway spruce in Sweden. Aluminum saturation of the cation-exchange complex has been found for the majority of forest soils (areal extent) in West Virginia (A.B. Jenkins, personal communication, 1997).

In Stage IV, base saturation reaches a new equilibrium state with respect to inputs, and base-cation export is equal to that of the preacidification stage, controlled only by chemical weathering. Once the acidic inputs are removed, soils rapidly absorb base cations released by chemical weathering, decreasing the flux of cations to the stream (Stage V). At some point, a new steady state is reached among soils, biomass, chemical weathering and atmospheric inputs (Stage VII). Data from the Fernow suggest that some forests in the central Appalachians may be at Stage II, which is the beginning of the cation-loss curve. However, the sensitivity of Appalachian forest soils to acidification and cation leaching is highly variable and poorly understood. We have begun research to identify soils that are most sensitive to acidification and base-cation loss, i.e. those with low base saturation, low C/N ratios, high Al saturation of the cation-exchange complex, and derived from acidic parent material.

Acidification/N saturation also affects the cycling of nutrients by changing rates of litter decomposition (Adams and Angradi, 1996), nutrient uptake and concentration in aboveground biomass (Adams et al., 1995), soil base saturation, and N mineralization and nitrification (Aber et al., 1989).

4. Effects of harvesting and acidic deposition on long-term sustainability

A model of the effects of harvesting and acidic deposition on the long-term sustainability of central Appalachian forest ecosystems that is based on data presented or discussed in this paper is shown in Fig. 5. Acidic deposition can cause increased losses of Ca

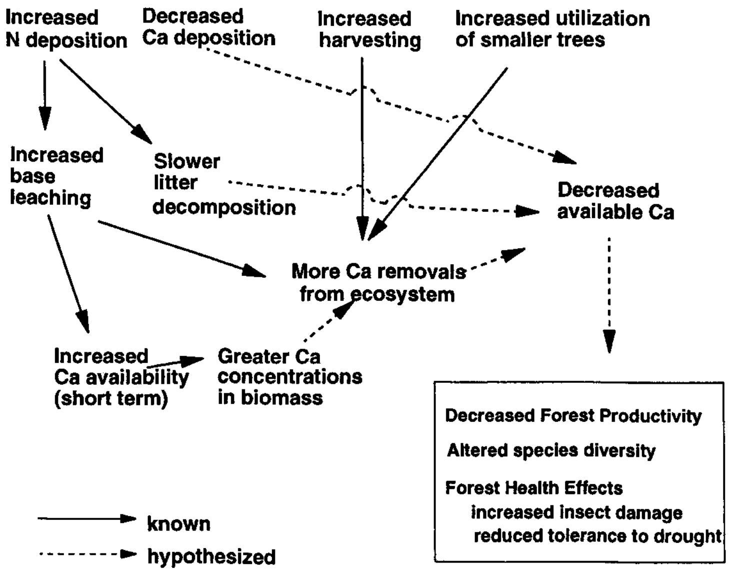

Fig. 5. Conceptual diagram for effects of forest harvesting and acidic deposition on calcium cycling, availability, and long-term forest productivity.

from Appalachian forest soils through the mechanisms discussed previously. We also know that shorter rotations and increased intensity of harvest removal (greater biomass) result in greater losses of Ca and Mg from aboveground pools. Increased harvesting at a time when there is an elevated Ca content in trees (Adams et al., 1995) due to elevated soil solution Ca levels (Stage II, Fig. 4), might accelerate base-cation loss. This suggests more than an additive effect of harvesting and acidic deposition on base-cation losses from these ecosystems.

5. Implications for sustainable forests

Plausible mechanisms and hypotheses exist to link air pollution (notably acidic deposition) and forest harvesting to changes in the long-term productivity of forest ecosystems. However, there are few examples of such declines. No symptoms have been detected of declining forest health, injury to vegetation or nutrient imbalances/deficiencies due to excess N deposition in the montane ecosystems of the Los Angeles Basin (though effects of ozone are well documented) (Fenn et al., 1996). Also, in the central Appalachians, effects of excess N deposition (or acidic deposition in general) such as visible foliar damage, nutrient imbalances, or growth declines have not been demonstrated. Localized declines of hardwood stands are attributable to disease, pests, or past land-use history (WVSAF, 1992). Sugar maple stands in the eastern United States are exhibiting crown die back and growth declines associated with nutrient deficiencies, but the link to acidic deposition is equivocal (Hendershot and Jones, 1989; Ellsworth and Liu, 1994).

This lack of evidence for hypothesized significant decline in these extensively managed systems suggests that: (1) the effects are subtle or delayed; (2) we are measuring the wrong things; (3) we have inadequate controls with which to compare changes; or (4) there are no effects. Research in grassland and tundra ecosystems suggests that one consequence of elevated N inputs is altered plant species diversity (Tilman, 1984; Giblin et al., 1997) and successional pathways (Carson and Barrett, 1988). Numerous fertilization experiments have documented changes in understory plant communities in response to N additions (see review in Binkley and Högberg, 1997). The link between soil productivity and sustainable forest productivity with regards to species composition and effects needs elucidation.

6. Long-term soil productivity in the central Appalachians

With the goal of better understanding the long-term productivity of central Appalachian forests, we have designed a study to evaluate threats to sustainable forest ecosystems in the central Appalachians. We are focusing on the soil as the controller of productivity because we are concerned about losses of base nutrients from these systems as described previously, and the implications for the ecosystem.

We designed this study to compare the effects of acidic deposition (elevated N and S deposition in particular), various levels of harvesting (biomass removal) and interactions between acidic deposition and harvesting on soil processes, aboveground growth and diversity, overall productivity and nutrient cycling. We recognized that to fully replicate all 16 treatments (four levels of biomass removal, four levels of N and S additions) as originally proposed, would overwhelm all of our resources. Although the experiment was subsequently scaled back to three treatments and an untreated control, it is of sufficient size to answer the following questions: Will N deposition remove enough base cations from the soil to result in growth declines within one rotation? Will N deposition and/or base-cation losses lead to shifts in species composition of the various biota? If so, can these effects be easily ameliorated or prevented? What determines the sensitivity to base-cation leaching? Are the effects of harvesting and acidic deposition independent or interactive?

Treatments include: (1) whole-tree harvesting, an extreme harvest treatment (but not unlikely in a pulp-dominated market); (2) whole-tree harvesting plus chemical additions of ammonium sulfate (N and S); and (3) whole-tree harvesting plus additions of N and S and additions of dolomitic limestone (Ca and Mg). Untreated control plots will allow us to compare treatment effects with untreated second-growth succession and growth. The first set of plots, established in 1995, was harvested in the fall and winter of 1996–1997. A second set of plots, on a site of slightly lower

Table 3
Pretreatment measurements as part of long-term productivity study on the Fernow Experimental Forest

	Measurements
Soils	
Soil chemistry	Nutrients, pH, base saturation
Soil physical properties	Bulk density, porosity, texture, coarse fragments
Vegetation	
Overstory	Basal area, diversity, volume, biomass
Woody regeneration	All woody species by origin and height class
Herb and shrub layer	All vegetation less than 2.5 cm DBH, percent cover and number, biomass, diversity
Seedbed potential	30 cm × 30 cm × 10 cm deep samples of forest floor, germinated in greenhouse
Leaf area	
Coarse woody debris	Biomass and nutrient content
Nutrient cycling	Precipitation chemistry
	Soil solution chemistry
	N mineralization/nitrification rates
	Foliar chemistry
Meteorological	Windspeed and direction
	Solar radiation
	Relative humidity
	Air temperature
	Soil temperature (10 cm depth)

site quality, with an additional treatment of whole-tree harvesting and only dolomitic lime, was harvested in the fall of 1998. Pretreatment measurements (Table 3) were collected to provide baseline data on ecosystem productivity and functioning. Vegetation resampling was conducted the first year after harvesting, and will be conducted every five years afterward throughout the 80-year rotation. We will track changes in plant species diversity, biomass, productivity, nutrient cycling, and wood production. A novel monitoring program will also evaluate changes in diversity of mycorrhizal fungi over time and with treatment (J. Cummins, personal communication, 1997).

Detailed pretreatment soil sampling provided information on the physical and chemical condition of the soils prior to harvesting, and will be repeated every five years. Detailed biomass data will be used to estimate NPP and link harvest simulation models (Baumgras et al., 1993) and nutrient-cycling models such as NuCM (Liu et al., 1992) to evaluate the effects of different intensities of harvest removal on nutrient cycling, particularly base-cation leaching. Because one of the treatments is an addition of dolomitic limestone, we also will be able to assess the effectiveness of an amelioration treatment on productivity, diversity and ecosystem function.

References

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

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., 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.

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, 10th central hardwood forest conference. 1995 March 5–8; Morgantown, WV. Gen. Tech. Rep. NE-197. USDA For. Serv., Northeast For. Exp. Stn., Radnor, PA. pp. 119–130.

Auchmoody, L.R., 1972. Nutrient properties of five West Virginia forest soils. Research Note NE-145. USDA For. Serv., Northeast For. Exp. Stn., Upper Darby, PA, p. 4.

Baumgras, J.E., Hassler, C.C., LeDoux, C.B., 1993. Evaluating and validating harvesting system production through computer simulation. For. Prod. J. 43, 65–71.

Binkley, D., Högberg, P., 1997. Does atmospheric deposition of nitrogen threaten Swedish forests? For. Ecol. Manage. 92, 119–152.

Brooks, A.B., 1911. Forestry and wood industries. West Virginia Geol. And Econ. Survey., vol. 5. Acme Pub. Co., Morgantown, WV, pp. 481.

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

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.

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.

DeWalle, D.R., Sharpe, W.E., Edwards, P.J., 1988. Biogeochemistry of two Appalachian deciduous forest sites in relation to episodic stream acidification. Water, Air, Soil Pollut. 40, 143–156.

DiGiovanni, D.M., 1990. Forest statistics for West Virginia – 1975 and 1989. Resource. Bull. NE-114. USDA For. Serv., Northeast For. Exp. Stn., Radnor, PA, pp. 117.

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

Ellsworth, D.S., Liu, X., 1994. Photosynthesis and canopy nutrition of four sugar maple forests on acid soils in northern Vermont. Can. J. For. Res. 24, 2118–2127.

Emmett, B.A., Boxman, A.W., Bredemeier, M., Gundersen, P., Kjonaas, O.J., Moldan, F., Schleppi, P., Tietema, A., Wright, R.F., 1997. Nitrogen – Too much of a good thing? A synthesis of the NITREX project. J. Conf. Abstr. 2(2), 168.

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.

Feger, K.H., 1992. Nitrogen cycling in two Norway spruce (*Picea abies*) ecosystems and effects of a $(NH_4)_2$ SO_4 addition. Water, Air Soil Pollut. 61, 295–307.

Fenn, M.E., Poth, M.A., Johnson, D.W., 1996. Evidence for nitrogen saturation in the San Bernardino Mountains in southern California. For. Ecol. Manage. 82, 211–230.

Frothingham, E.H., 1924. Some silvicultural aspects of the chestnut blight situation. J. For. 22, 861–892.

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.

Gibbs, J.N., 1978. Intercontinental epidemiology of Dutch elm disease. Ann. Rev. Phytopath. 16, 287–307.

Giblin, A.E., Murray, G., Nadelhoffer, K.J., Laundre, J.A., Shaver, G.R., 1997. Controls on methane emission in Alaskan arctic tundra: results of long-term field manipulations of temperature and nutrients. J. Conf. Abstr. 2(2), 185.

Grier, C.C., Lee, K.M., Nadkarni, N.M., Klock, G.O., Edgerton, P.J., 1989. Productivity of forests of the United States and its relation to soil and site factors and management practices: A review. Gen. Tech. Rep. PNW-GTR-222. USDA For. Serv. Pac. Northwest Res. Stn. Portland, OR, pp. 51.

Hameed, S., Dignon, J., 1992. Global emissions of nitrogen and sulfur oxides in fossil fuel combustion 1970-1986. J. Air Waste Manage. Assoc. 42, 159–163.

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

Hendershot, W.H., Jones, A.R.C., 1989. The sugar maple decline in Quebec: a discussion of probable causes and the use of fertilizers to limit damage. For. Chron. 65, 280–287.

Hornbeck, J.W., Adams, M.B., Corbett, E.S., Verry, E.S., Lynch, J.A., 1993. Long-term impacts of forest treatment on water yield: a summary for northeastern USA. J. Hydrol. 150, 323–344.

Huff, D.D., Swank, W.T., Troendle, C.A., Henderson, G.S., Waide, J.B., Haines, T., 1978. Element cycles and water budget analyses applied to forest management in the eastern United States. In: North American forest: gateway to opportunity: proceedings of the Society of American Foresters. Soc. Am. For., Washington DC, pp. 77–89.

Johnson, D.W., Turner, J., Kelly, J.M., 1982. The effects of acid rain on forest nutrient status. Water Resour. Res. 18, 449–461.

Joslin, J.D., Wolfe, M.H., 1989. Aluminum effects on northern red oak seedlings grown in six forest soil horizons. Soil Sci. Soc. Am. J. 53, 274–281.

Keever, C., 1953. Present comparison of some stands of the former oak-chestnut forest in the southern Blue Ridge mountains. Ecology 34, 44–54.

Kochenderfer, J.N., Edwards, P.J., 1991. Effectiveness of three streamside management practices in the central Appalachians. In: Coleman, S.S., Neary, D.G., (comps.), Proceedings of the 6th biennial southern silvicultural research conference. Volume 2. 1990, October 30-November 1; Memphis, TN. Gen. Tech. Rep. SE-70. USDA For. Serv., Southeast. For. Exp. Stn., Asheville, NC. pp. 688–700.

Likens, G.E., Driscoll, C.T., Buso, D.C., 1996. Long-term effects of acid rain: response and recovery of a forest ecosystem. Science 272, 244–246.

Liu, S., Munson, R., Johnson, D.W., Gherini, S., Summers, K., Hudson, R., Wilkinson, K., Pitelka, L.F., 1992. The Nutrient Cycling Model (NuCM): overview and application. In: Johnson, D.W., Lindberg, S.E. (Eds.), Atmospheric deposition and forest nutrient cycling. Springer-Verlag; New York, pp. 583–609.

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.

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.

McLaughlin, S.B., Andersen, C.P., Hanson, P.J., Tjoelker, M.G., Roy, W.K., 1991. Increased dark respiration and calcium deficiency of red spruce in relation to acidic deposition at high-elevation southern Appalachian mountain sites. Can. J. For. Res. 21, 1234–1244.

McNulty, S.G., Aber, J.D., 1993. Effects of chronic nitrogen additions on nitrogen cycling in a high-elevation spruce-fir stand. Can. J. For. Res. 23, 1252–1263.

Monk, C.D., Day, F.P., Jr., 1988. Biomass, primary production and selected nutrient budgets for an undisturbed hardwood watershed. In: Swank, W.T., Crossely, D.A., Jr. (Eds.), Forest hydrology and ecology at Coweeta. Springer-Verlag, New York. pp. 151–159.

Norton, S.A., Kahl, J.S., Fernandez, I.J., 1997. Stream chemistry indicators of acidification at the Bear Brook Watershed Manipulation, Maine. J. Conf. Abstr. 2(2), 260.

Norton, S.A., Kahl, J.S., Fernandez, I.J., Rustad, L.E., Scofield, J.P., Haines, T.A., 1994. Response of the West Bear Brook Watershed, Maine, USA, to addition of $(NH_4)_2$ SO_4: 3-year results. For. Ecol. Manage. 68, 61–73.

NAPAP, 1991. Acidic deposition: state of science and technology. Natl. Acid Precip. Assess. Prog., Washington, D.C., pp. 265.

Ostry, M.E., Mielke, M.E., and Skilling, D.D., 1994. Butternut – strategies for managing a threatened tree. Gen. Tech. Rep. NC-165. USDA For., Serv. North Central For. Exp. Stn., St. Paul, MN, pp. 7.

Patric, J.H., 1973. Deforestation effects on soil moisture, streamflow and water balance in the central Appalachians. Res. Pap. NE-259. USDA For. Serv., Northeast For. Exp. Stn. Upper Darby, PA, pp. 12.

Patric, J.H., 1980. Effects of wood products harvest on forest soil and water relations. J. Environ. Qual. 9, 73–80.

Patric, J.H., Smith, D.W., 1975. Forest management and nutrient cycling in eastern hardwoods. Res. Pap. NE-324. USDA For. Serv., Northeast For. Exp. Stn., Upper Darby, PA, pp. 12.

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

Powers, R.F., Mead, D.J., Burger, J.A., Ritchie, M.W., 1994. Designing long-term site productivity experiments. In: Dyck, W.J., Cole, D.W., Comerford, N.B. (Eds.), Impacts of forest harvesting on long-term site productivity. Chapman and Hall, New York, pp. 247–286.

Schuler, T.M., Miller, G.W., 1995. Shelterwood treatments fail to establish oak reproduction on mesic forest sites in West Virginia – 10 year results. In: Gottschalk, K.W., Fosbroke, S.L. (Eds.), Proceedings, 10th central hardwood forest conference. 1995 March 5–8; Morgantown, WV. Gen. Tech. Rep. NE-197. USDA For. Serv., Northeast For. Exp. Stn., Radnor, PA, pp. 375–387.

Shortle, W.C., Smith, K.T., 1988. Aluminum-induced calcium deficiency syndrome in declining red spruce. Science 240, 239–240.

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, D.C., pp. 141–144.

Stoddard, J.L., 1994. Long-term changes in watershed retention of nitrogen. In: Baker, L.A. (Ed.), Environmental Chemistry of Lakes and Reservoirs. Advances in Chemistry. American Chemical Society, Washington, DC. 237, pp. 223–284.

Swank, W.T., 1988. Stream chemistry responses to disturbance. In: Swank, W.T., Crossely, D.A., Jr. (Eds.), Forest hydrology and ecology at Coweeta. Springer-Verlag, New York. pp. 339–357.

Tilman, D., 1984. Plant dominance along an experimental nutrient gradient. Ecology 65, 1445–1453.

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, 659–666.

Williams, M.W., Baron, J.S., Caine, N., Sommerfeld, R., Sanford Jr., R., 1996. Nitrogen saturation in the Rocky Mountains. Environ. Sci. Technol. 30(2), 640–646.

Wright, R.F., Tietema, A., 1996. Ecosystem response to 9 years of nitrogen additions at Sogndal, Norway. For. Ecol. Manage. 71, 133–142.

WVSAF, 1992. Report of the SAF Ad hoc Committee to evaluate forest health in Rock Creek, West Virginia. Prepared for West Virginia Division, Society of American Foresters.

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.

Vitousek, P.M., Melillo, J.M., 1979. Nitrate losses from disturbed forests: patterns and mechanisms. For. Sci. 25, 605–619.