NUTRIENT BUDGETS OF TWO WATERSHEDS ON THE FERNOW EXPERIMENTAL FOREST

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Abstract: Acidic deposition is an important non-point source pollutant in the Central Appalachian region that is responsible for elevated nitrogen (N) and sulfur (S) inputs to forest ecosystems. Nitrogen and calcium (Ca) budgets and plant tissue concentrations were compared for two watersheds, one that received three years of an artificial acidification treatment and an adjacent reference watershed, both located on the Fernow Experimental Forest, West Virginia. Treatments consisted of ammonium sulfate fertilizer applied aerially three times per year at an annual rate of 61 kg S ha^{-1} and 54 kg N ha^{-1}. Trees of four species (*Betula lenta* L., *Prunus serotina* Ehrh., *Acer rubrum* L. and *Liriodendron tulipifera* L.) were harvested for biomass and nutrient determinations. Some tree species on the treated watershed showed elevated N and decreased Ca levels in some tissues, particularly foliage, but no consistent pattern for any species or tissue component was found. Compared to other watersheds in the central and southern Appalachians, Fernow watersheds are losing more N in streamflow. This loss is almost solely in the form of NO_3, which appears to bring about increased leaching of Ca from watersheds.

INTRODUCTION

During its initial 10-year existence, the National Acid Precipitation Assessment Program (NAPAP) supported a great deal of air pollution research, primarily related to acidic deposition, and its effects on structures, human health, and ecosystems. Although much was learned through NAPAP, many unanswered questions remain, and acidic deposition continues. Nitrogen (N) deposition is expected to increase, despite recent Clean Air legislation (Aber and others 1993), and although reduced sulfur (S) deposition has been reported in some parts of the U.S. (Baier and Cohn 1993), in other regions no change has been detected. The effects of acidic deposition on central Appalachian hardwood forests are not well-understood. Hypothesized effects include nitrogen saturation, altered susceptibility to pests and pathogens, increased tree mortality, and increased growth. None of these hypotheses have been adequately tested, however. Ecosystem level studies can improve our ability to make predictions regarding the effects of acidic deposition on the health and long-term sustainability of central Appalachian forests.

One method to evaluate acidic deposition effects on forest ecosystems is whole-watershed manipulation, i.e. applying a potentially acidifying agent to one watershed and comparing the results with an untreated reference watershed. Because of their expense, whole-watershed experiments are not often replicated. Comparisons between watersheds are therefore based on pseudoreplicated measurements (Hargrove and Pickering 1992). Nonetheless, much can be learned with careful interpretation, and numerous paired watershed studies have been the basis for meaningful ecosystem research (Likens and others 1977, Swank and Crossley 1988). The objective of this paper is to describe and compare nutrient and biomass budgets on two experimental watersheds on the Fernow Experimental Forest, and to evaluate the early effects of an artificial acidification treatment on nutrient cycling. This paper will focus on N and calcium (Ca) because of the recent concerns about N saturation of forest ecosystems (Aber and others 1993) and potential Ca deficiencies, and implications for forest health.

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METHODS

Description of Watersheds

Two experimental watersheds were used for this study. Watershed 3 (WS3) is the treatment watershed, and watershed 7 (WS7) serves as a control or reference watershed. The watersheds are located on the Fernow Experimental Forest near Parsons, West Virginia, USA (39° 3' 15" N, 79° 49' 15" W). The Fernow is located on the unglaciated Allegheny plateau of the Appalachian mountains and is characterized by steep slopes and shallow soils (<1m). Precipitation is distributed evenly between dormant and growing seasons and is among the most acidic in the United States. Average annual pH is 4.20, but pH values below 4.0 are common in summer (Edwards and Helvey 1991). The predominant soil type on the experimental watersheds is Calvin channery silt loam (loamy-skeletal, mixed, mesic Typic Dystrochrept) underlain with fractured sandstone and shale of the Hampshire formation (Losche and Beverage 1967). The experimental watersheds are drained by intermittent, second-order streams. Stands on both watersheds have similar species composition and originated at the same time (1969). Prior to 1969, WS7 was maintained barren with herbicides (Patric and Reinhart 1971), while WS3 was clearcut and permitted to regrow without interference. Other watershed characteristics are given in Table 1.

Table 1. Characteristics of two experimental watersheds on the Fernow Experimental Forest, West Virginia.

	Watershed	
	3	7
Age (yr)	24	24
Stand density (trees ha^{-1})	537.5	376.8
Basal area (m^2 ha^{-1})	3.87	3.27
Area (ha)	34.3	24.2
Aspect	S	ENE
Minimum elev. (m)	735	725
Maximum elev. (m)	860	855
Average annual precipitation (mm)	1480	1417
Average annual streamflow (mm)	666	882
Dominant tree species	Black cherry	Black birch
	Red maple	Red maple
	Black birch	Sugar maple
	American beech	Black cherry

Methods and Measurements

Ammonium sulfate fertilizer was applied to WS3 three times per year by helicopter beginning in 1989. March and November applications consisted of 33.6 kg of fertilizer per hectare, which corresponds to 8.1 and 7.1 kg ha^{-1} of S and N, respectively. Each July we applied 100.8 kg fertilizer per hectare, or 24.4 and 21.2 kg ha^{-1} S and N, respectively. Multiple applications per year were used to more closely mimic seasonal variations in chemical inputs. These application rates were approximately double the amount of S and N deposited on the watersheds in throughfall, which we believe to be a good estimate of bulk deposition. Consequently, the total amount of S and N deposited annually on the treatment watershed was 60.5 kg S (ambient S + 40.6 kg from treatment) and 53.8 kg N (ambient N + 35.4 kg from treatment) per hectare, or approximately three times that received by the reference watershed.

In July 1991, total aboveground portions of five trees from each of four species (black cherry, red maple, black birch and yellow-poplar: *Prunus serotina* Ehrh., *Acer rubrum* L., *Betula lenta* L., and *Liriodendron tulipifera* L., respectively) were sampled from both WS3 and WS7. Only dominant and codominant trees were selected for nutrient

determinations. (For biomass determinations, an additional six trees per watershed per species were selected to insure representative sampling of smaller trees.) All leaves were removed from the felled trees, weighed, then subsampled for nutrient analysis and moisture content determination. Each tree was limbed and divided into stemwood (bole to bottom of live crown), top-wood (remainder of bole), and small and large branches (<1 cm diameter and >1 cm diameter, respectively); all constituent parts were weighed in the field. Subsamples were collected for nutrient analysis and moisture content determinations. Each subsample was weighed to the nearest 0.1 g, dried at 70° C and reweighed. Dead wood biomass was determined on 100 square-milacre plots. On each plot, standing dead and down dead biomass were weighed separately and subsamples collected for nutrient and moisture content determinations. Soil nutrients and herbaceous layer biomass and nutrient methods were described by Gilliam and others (1994).

Aboveground biomass was measured on 0.004 ha (.01 A) plots and stand biomass for each watershed was calculated using biomass equations of Brenneman and others (1978). For those species for which Brenneman and others had no equation, an equation for a species of similar specific gravity was used. Watershed foliar biomass was estimated from litterfall collected from 25 1-m^2 litterfall traps per watershed. Root biomass was estimated from four root cores (45 cm depth) collected from each of 25 plots per watershed during May and September 1991. Roots were washed from the cores, separated into fine and coarse roots, dried (70°C) and weighed. Samples were composited by watershed, horizon and size class to provide sufficient tissue for nutrient analyses. Roots were not separated by species.

Tissue samples were analyzed at the Plant and Soil Analysis Laboratory at the University of Maine. Total N was determined by block digestion using a sulfuric acid-hydrogen peroxide solution, and analyzed using a Wescan 360 ammonia analyzer (Wescan Instruments, Santa Clara, CA)[2]. To determine total Ca, ground tissue samples were ashed at 550°C for 5-6 hrs, dissolved in 50% HCl, and analyzed using inductively coupled plasma emission spectrometry.

Nutrient budgets were constructed using mean watershed nutrient concentrations, multiplied by estimated watershed biomass of each component. Soil nutrient pools were calculated for a 60 cm soil depth. Average root biomass (fine + coarse roots) was calculated for each watershed. Herb layer values were calculated from data of Gilliam and Turrill (1993) and Gilliam and others (1994).

Annual (water year, May 1 - April 30) nutrient exports in stream water were calculated from streamflow volume and weekly ionic concentrations of grab samples to relate to watershed nutrient pools. Streamflow measurement, sampling techniques and analytical procedures were described by Edwards and Kochenderfer (1993) and Edwards and Wood (1993). Effects of the acidification treatment on stream and soil water chemistry were reported by Edwards and Wood (1992) and Adams and others (1993).

Watershed nutrient budgets were calculated for treatment comparisons. Mean mass, concentration and content were also calculated for each of the 4 tree species on each watershed and compared using Student's t tests.

RESULTS AND DISCUSSION

Biomass and Nutrient Budgets

Tree biomass varied between the two watersheds (Figure 1). The treatment watershed, WS3, had greater aboveground woody biomass (stems, tops and large branches) than the control, WS7. Because of different pre-treatment histories, the difference in total tree aboveground biomass between WS3 and WS7 can not be attributed to the acidification treatment. The vegetation on WS7 originated mostly from seed, going through the grass - herbaceous - semi-woody - woody plant successional phases (Kochenderfer and Wendel 1983), while the vegetation that developed on WS3

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[2]The use of trade, firm, or corporation names in this publication is for the information and convenience of the reader. Such use does not constitute an official endorsement or approval by the U.S. Department of Agriculture or the Forest Service of any product or service to the exclusion of others that may be suitable.

originated mainly from stump sprouts, getting a "head start" on biomass accumulation. Moreover, it is unlikely that such a large treatment effect on woody biomass (approximately 12 of the 17 mt ha^{-1} difference between the two stands is in stemwood) would be evident after only three years of fertilization treatment. Changes in production of foliage or small branches have been reported as a short-term growth response to N fertilization (Auchmoody and Smith 1977, Brix and Ebell 1969, Carlson and Preisig 1981). However, foliar biomass did not differ between the two watersheds, and small branch biomass was slightly lower on the treated watershed. The total root biomass was less on WS3 than on WS7.

Total biomass of dead wood was greater on WS3 (Figure 1). Because WS7 was cut earlier (1964-67) than WS3 (1969) and then maintained barren for 5 years, surface temperatures were higher, resulting in greater rates of decomposition (Mattson and others 1989). Also, WS7 was cut several years prior to WS3, allowing more time for woody slash to decompose. Therefore biomass differences between the watersheds are not attributed to fertilization.

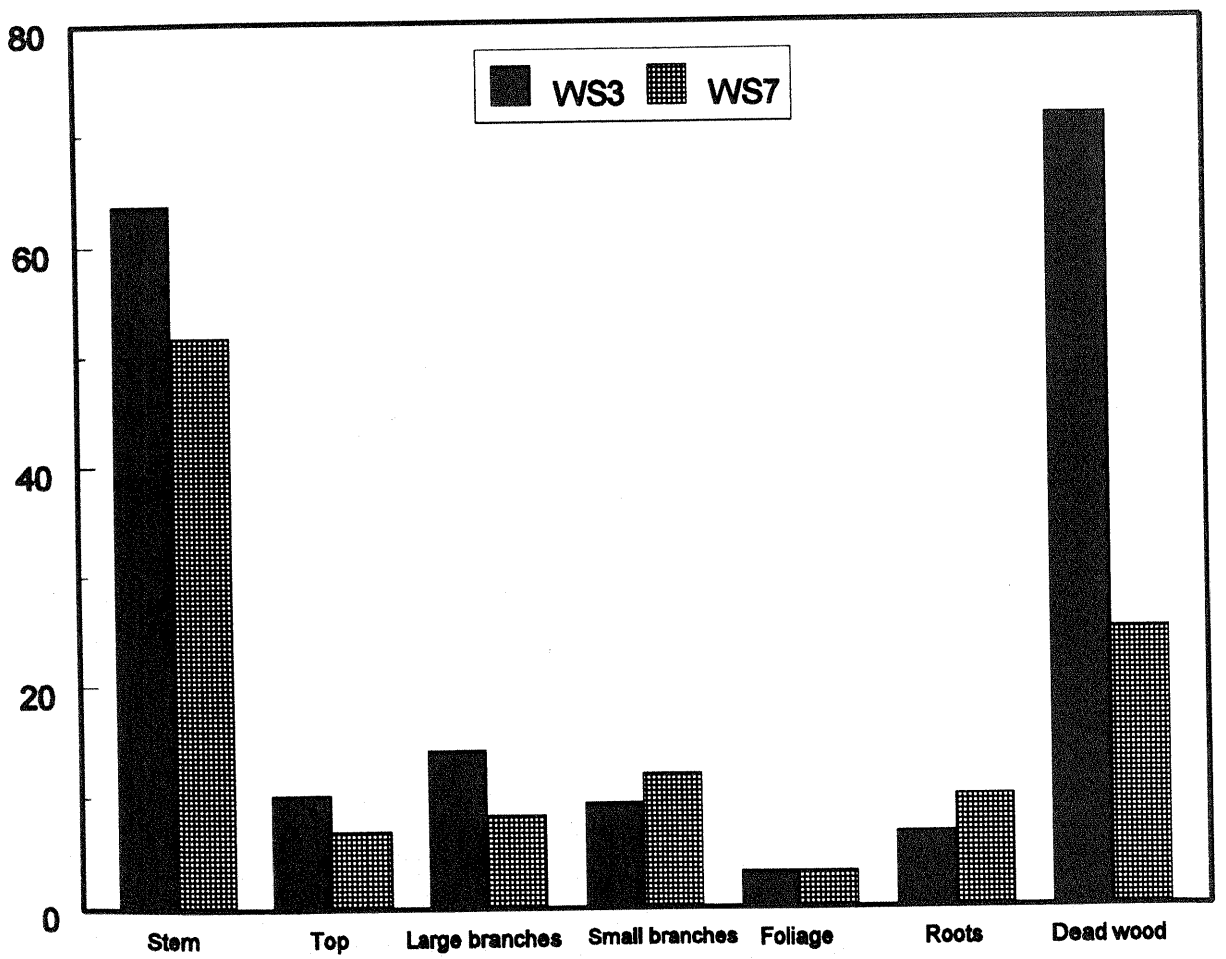

Figure 1. Biomass (mt ha^{-1}) by tree component for 2 experimental watersheds on the Fernow Experimental Forest, West Virginia.

Despite these confounding factors, some observations about the acidification treatment can be made. Aboveground N content was greater for WS3, except in the herb layer (Figure 2). The large herb layer N content in WS7 is due to a large fern community on WS7 which results in twice the herb biomass of WS3 (Gilliam and others 1994). Tissue N concentrations for the herb layer did not differ significantly between the two watersheds, nor did soil extractable N pools or pH differ significantly (Gilliam and others, 1994). Aspect did not significantly affect the response of small stands/watersheds to fertilization (Edwards and others 1991), therefore this is not an alternative explanation. Overall, it did not appear that the treatment had much effect on soil N and vegetation N levels at the watershed scale. Three years after the initiation of treatment, streamwater export of N from WS3 was not significantly different from WS7, although nitrate export from WS3 was significantly greater than that of another reference watershed (Adams and others 1993). This can be explained by examining the differences in pre-treatment N export (Figure 3). For six years prior to treatment, WS3 export of N was consistently less than that from WS7. Within 3 years of initiation of treatment, N export from WS3 increased to a level equal to that of WS7. Because of high variability, this change in trend is not statistically significant, but does agree with results of Adams and others (1993) and Edwards and others (1991) that report significant changes in soil water N and stream water N.

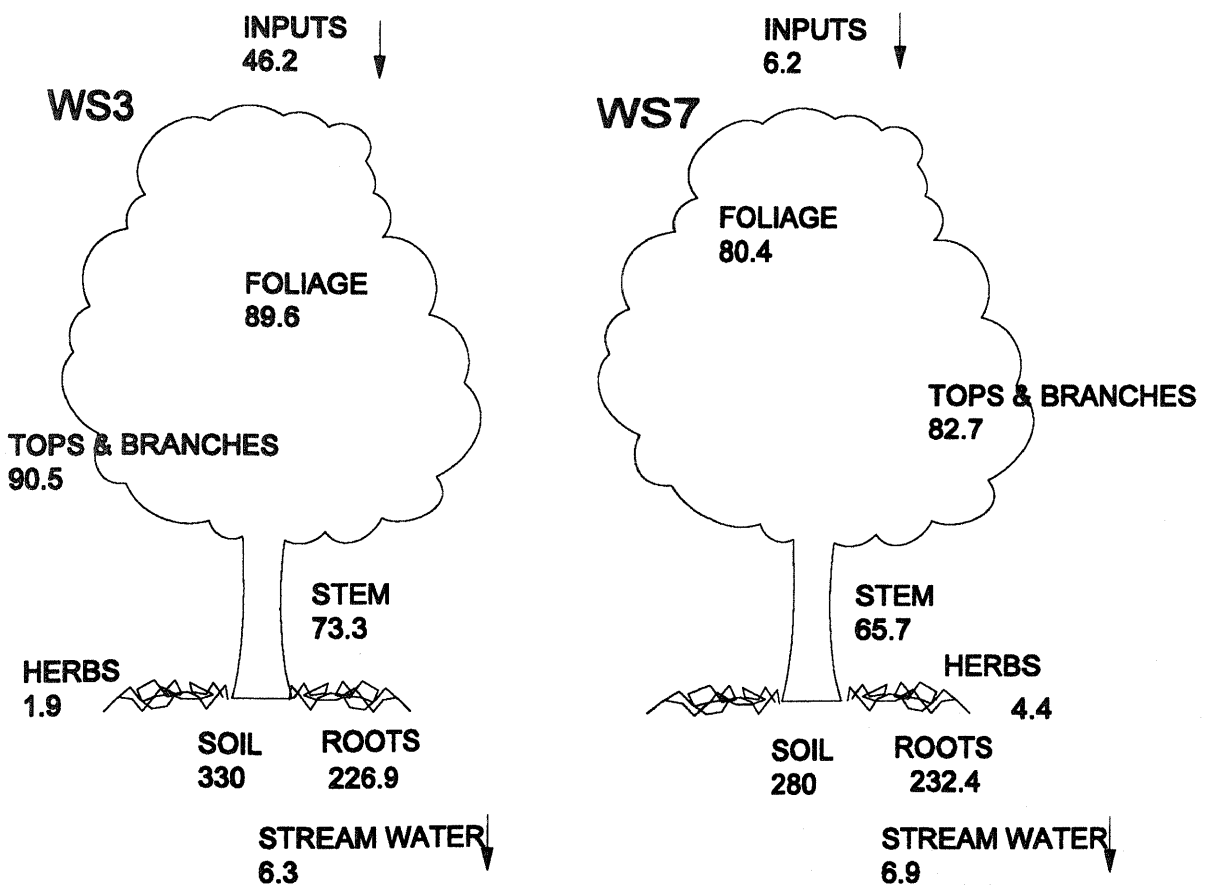

Figure 2. July nitrogen budgets for WS3 (treatment watershed) and WS7 (control watershed), Fernow Experimental Forest. Values are kg ha^{-1}.

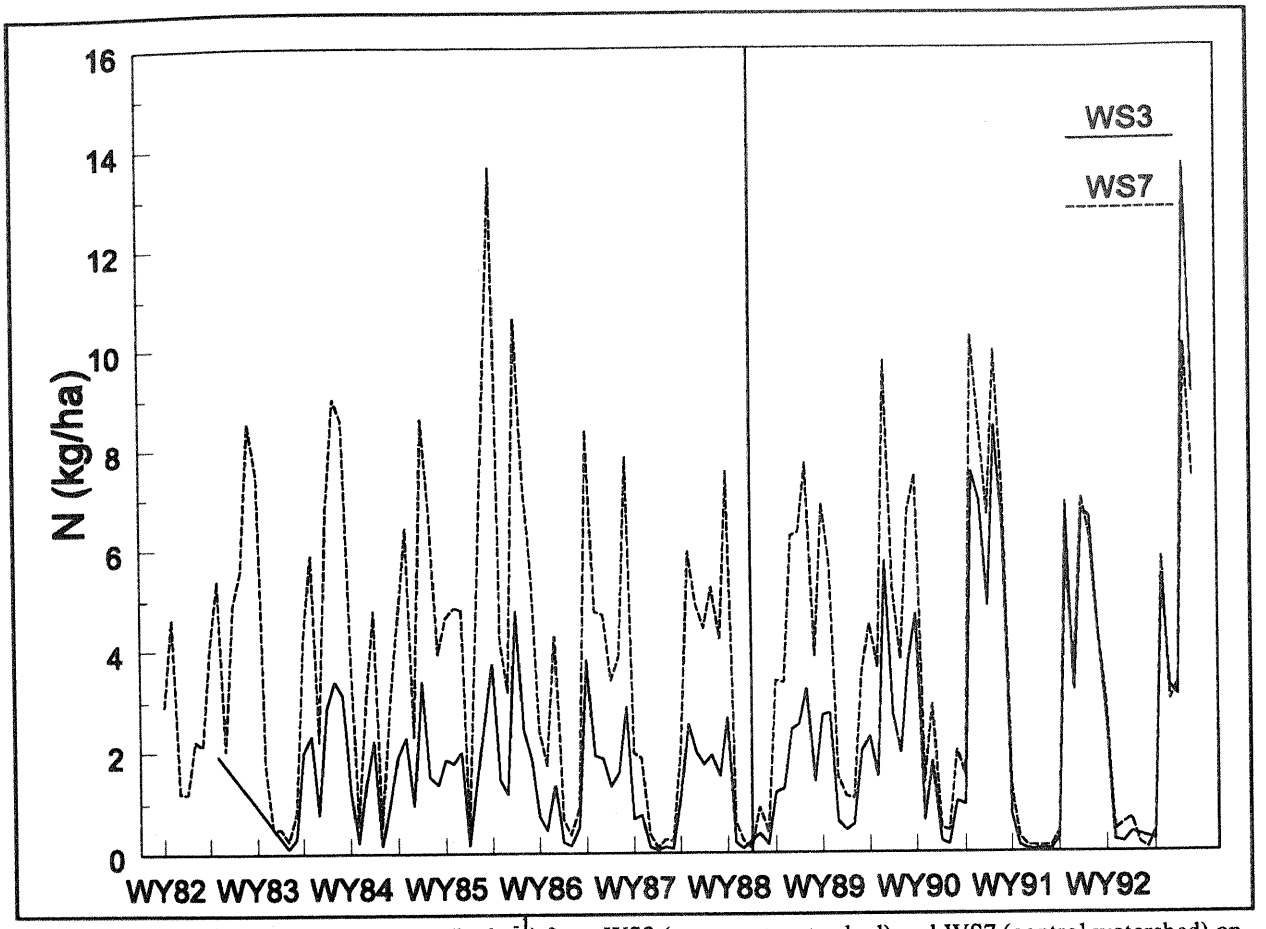

Figure 3. Mean monthly nitrogen export (kg ha^{-1}) from WS3 (treatment watershed) and WS7 (control watershed) on the Fernow Experimental Forest during water years 1982-1992. Vertical line indicates initiation of treatment.

Foliar Ca levels were lower on WS3 than WS7 (Figure 4), reflecting significant differences in foliar Ca concentrations for most species (Tables 3,4). Stemwood Ca did not differ between the two watersheds. Herb layer Ca concentrations for WS3 were twice as high as for WS7, a significant difference (Gilliam and others 1994). However, the difference is masked in the whole watershed budget because of differences in herb layer biomass described previously. Edwards and Kochenderfer (1993) reported ion pairing of Ca and NO_3 in soil and stream water export from WS3. This observation suggests increased availability of Ca on WS3, as Ca is removed from soil exchange sites during export of the mobile NO_3. Although soil Ca levels appear lower on WS3 than WS7, Gilliam and others (1994) found no statistically significant differences in Ca concentrations of soil between the two watersheds, due to high variability. Annual streamwater Ca export from WS3 and WS7 did not differ significantly (Figure 5).

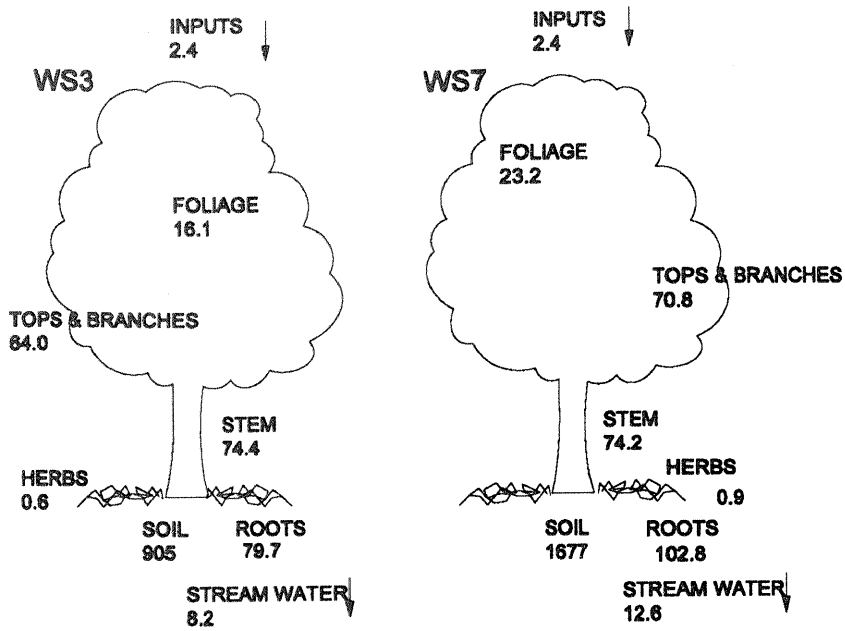

Figure 4. July calcium budgets for WS3 (treatment watershed) and WS7 (control watershed), Fernow Experimental Forest. Values are kg ha^{-1}.

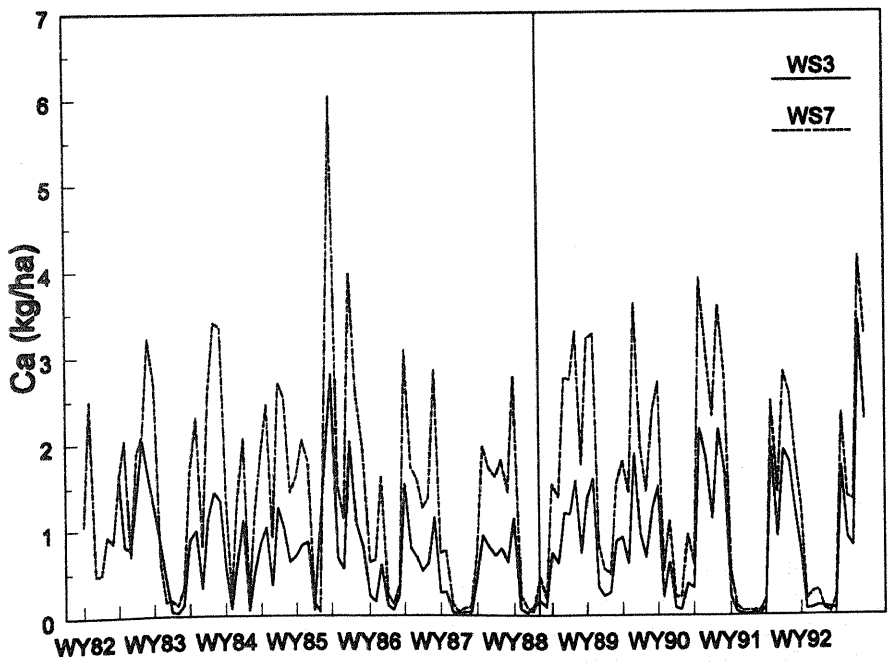

Figure 5. Mean monthly calcium export (kg ha^{-1}) from WS3 (treatment watershed) and WS7 (Control watershed) on the Fernow Experimental Forest during water years 1982-1992. Vertical line indicates initiation of treatment.

Nitrogen and Ca budgets for WS7 (reference watershed) were compared with those of Coweeta's Watershed 18 (Swank and Crossley 1988) and the Walker Branch Watershed in Tennessee (Johnson et al. 1988), both untreated reference watersheds (Table 2) to provide context for the observations from the acidification treatment. It was reasoned that if the three untreated controls showed similar nutrient levels, then the effects of the acidification experiment might be more obvious. The Fernow aboveground vegetation values appear low relative to the other two watersheds. This can be attributed mostly to differences in stand age -- stands on both the Coweeta Watershed 18 and Walker Branch Watershed were approximately 60 years old, while stands on WS3 and WS7 were 24 years old and had not accumulated biomass or nutrients for as long. Nutrient inputs in precipitation are similar among the three sites. However, it should be noted that the 6.2 kg N ha^{-1} reported here for the Fernow for 1991 is below the average value of 13.7 kg N ha^{-1} reported by Helvey and Kunkle (1986), who also reported average Ca inputs of 7.9 kg Ca ha^{1}. This is attributed to lower precipitation in 1991 (125 cm compared with the long-term average of 380 cm). Even during this year of relatively low deposition, N exports were much greater from WS7 than from the Coweeta Watershed 18 or Walker Branch Watershed. Younger, more vigorous stands are normally more conservative of N than older stands, thus this seems a contradiction. For this same year, the export from an 85 year old reference watershed on the Fernow for the same year was 3.45 kg N ha^{-1}. Therefore, N export from the Fernow watersheds does appear elevated, relative to Walker Branch and Coweeta.

Table 2. Comparison of N, P, and Ca budgets for three reference watersheds in the central and southern Appalachian Mountains. (NM=not measured).

	Fernow WS7	Coweeta[1]	Walker Branch[2]	
	kg ha^{-1}			
			YP[3]	OH[4]
Nitrogen				
Input	6.2	8.8	9.8	9.8
Aboveground	233.2	563.2	450	353
Soil	467[5]	6803	5730	2873
Output	6.9	0.09	0.056[6]	0.20[6]
Calcium				
Input	2.4	4.0	8.6	8.6
Aboveground	168.2	550.8	922	1283
Soil	2796	514	1445	668
Output	12.58	7.1	28.1[6]	4.81[6]

[1] Monk and Day (1988)
[2] Johnson and others (1988)
[3] Yellow-poplar stand, mean of 2 values
[4] Oak-hickory stand, mean of 2 values
[5] Extractable soil N, Coweeta and Walker Branch are total soil N
[6] Calculated from soil flux values of Johnson and others (1985)

Nutrient concentrations by species

Nutrient concentrations varied among the four species sampled, and among tree components. Foliar nutrient concentrations appeared the most responsive to treatment (Tables 3, 4). Foliar N concentrations were higher on WS3 for all four species studied, but were only significantly higher ($P \leq 0.10$) for black cherry and red maple. No significant differences in foliar mass were observed for these two species between the two watersheds. Foliar Ca was higher for all species on WS7 and was significantly different ($P \leq 0.10$) between the two watersheds for black birch, yellow-poplar and red maple. Some differences in foliar micronutrient concentrations also were detected, most notably for boron (data not shown). Overall, yellow-poplar showed the most differences, with significant differences in macro- or micronutrient concentrations for all components except small branches, suggesting that this fast-growing species may be more responsive to changes in nutrient supply than the other species examined.

Table 3. N and Ca concentrations by species, watershed and tissue type, from trees from two experimental watersheds on the Fernow Experimental Forest, WV. N is expressed in % and Ca as mg kg^{-1}. Watershed 3 = treatment, watershed 7 = control.

Species	Watershed	Tissue Type	N	Ca
Black birch	3	Foliage	2.720	3985
		Lg.branches	.198	1994
		Sm.branches	.462	2634
		Top	.180	1123
		Stem	.104	1000
	7	Foliage	2.540	5580
		Lg.branches	.170	2480
		Sm.branches	.448	3809
		Top	.130	1118
		Stem	.154	1195
Yellow-poplar	3	Foliage	3.002	6996
		Lg.branches	.212	2405
		Sm.branches	.475	3725
		Top	.193	1579
		Stem	.125	1294
	7	Foliage	2.750	11457
		Lg.branches	.222	2717
		S.branches	.435	4221
		Top	.230	1973
		Stem	.134	1915
Black cherry	3	Foliage	2.990	4354
		Lg.branches	.192	930
		Sm.branches	.508	1723
		Top	.180	776
		Stem	.126	1066
	7	Foliage	2.682	5899
		Lg.branches	.185	1042
		Sm.branches	.510	2413
		Top	.185	840
		Stem	.120	1071
Red maple	3	Foliage	2.310	4536
		Lg.branches	.172	1588
		Sm.branches	.448	3710
		Top	.183	1202
		Stem	.107	1306
	7	Foliage	2.070	6102
		Lg.branches	.178	2151
		Sm.branches	.446	4244
		Tops	.153	1486
		Stem	.101	1555

Table 4. Probability of >/t/, WS3 compared to WS7 for tissue biomass and nutrient concentrations, by tree component and species. Statistically significant values (P ≤ .10) are underlined.

	Mass	N	Ca
Stem			
Black birch	.029	.390	.297
Yellow-poplar	.035	.750	.003
Black Cherry	.848	.780	.974
Red maple	.102	.800	.124
Top			
Black birch	.340	.049	.982
Yellow-poplar	.455	.038	.063
Black cherry	.162	.746	.520
Red maple	.878	.264	.067
Small branches			
Black birch	.099	.796	.027
Yellow-poplar	.184	.163	.301
Black cherry	.499	.962	.195
Red maple	.984	.970	.132
Large branches			
Black birch	.278	.280	.200
Yellow-poplar	.052	.732	.282
Black cherry	.395	.720	.578
Red maple	.237	.806	.089
Foliage			
Black birch	.165	.502	.040
Yellow-poplar	.090	.298	.082
Black cherry	.285	.030	.136
Red maple	.300	.011	.013

Differences in nutrient concentrations were detected in the overstory trees, but not in the herb layer (Gilliam and others 1994). We predicted treatment effects first in the herb layer, particularly in response to N. It may be that the herb layer was limited by factors other than nutrition (e.g., light) (Gilliam and others 1994), and was unable to take advantage of the elevated N inputs. Gilliam and others (1994) reported significant changes in Fe and Al concentrations in herbaceous vegetation which they attributed to increased mobility of these ions in the soil due to the acidification treatment. Differences between watersheds in Al concentrations were significant only for yellow poplar and black cherry stemwood, but no significant differences in Fe concentrations were detected. Thus, the short-term response of these and other ecosystem components is variable and our ability to detect effects of acidic inputs also varies.

CONCLUSIONS

Whether observed differences in plant nutrient content reflect pretreatment differences between the watersheds is unknown. However, the lack of a consistent pattern of N enrichment for any tree species or tree tissue component suggest that plant nutrient status has been only minimally affected by three years of treatment. Soil acidification was not observed, but changes in stream water export of N were measured. The increased export of N from the treatment watershed suggests changes in N cycling at the watershed level. Nitrogen export may be elevated on the control watershed as well, suggesting ambient deposition may be causing effects, though more subtle, to those observed on the treated watershed. This research is ongoing, and will continue to evaluate the changes in nutrient cycling and other processes that result from acidic inputs, and consider implications for long-term sustainability of these important ecosystems.

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