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Artificial watershed acidification on the Fernow Experimental Forest, USA

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Abstract

A whole-watershed manipulation project was begun on the Fernow Experimental Forest in West Virginia, USA, in 1987, with the objective of increasing understanding of the effects of acidic deposition on forest ecosystems. Two treatment watersheds (WS9 and WS3) and one control watershed (WS4) were included. Treatments were twice-ambient N and S deposition, applied via NH_4SO_4 fertilizer, with three applications per year. Three years of pretreatment data were collected and used for calibration. Stream water chemistry data collected during 3 years of treatment were evaluated. Stream water pH and electrical conductivity were not significantly affected by the elevated N and S inputs on either treatment watershed. On WS9, there were no statistically significant treatment effects on stream water export of Ca, SO_4, or NO_3. On WS3, however, stream export of both NO_3 and Ca have increased as a result of acidification treatments. The implications of these results are discussed. Research is continuing so that the processes involved may be elucidated. In addition, effects on vegetation, aquatic invertebrates and amphibians also are being evaluated.

Introduction

Since the previous International Symposium on Forest Hydrology in 1965, watershed research has expanded in scope and subject area. Water quality and land management issues continue to be important research topics. In addition, watershed research techniques have proven valuable for ecosystem research. Whole-watershed manipulations offer a larger areal basis for study, incorporating natural variability. The chemical composition of water draining from forested watersheds can be used to gauge the health of forested ecosystems. The catchment serves to integrate the many processes operating within an ecosystem, and offers a distinct unit with definable surface boundaries.

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In many areas of the world, atmospheric deposition of pollutants to natural ecosystems remains a concern. Despite recent Clean Air legislation in the USA, total nitrogen deposition is expected to remain elevated over much of the northeastern USA (Aber et al., 1993). There is particular concern about the effects of acidic deposition on forested ecosystems. Despite nearly two decades of research, few studies can be considered realistic enough to mimic natural ecosystems, extensive enough to incorporate a large number of variables, or of sufficient duration to generate long-term results. Although models can increase our understanding, uncertainties exist about how well they represent actual rates, capacities and interactions among important processes. Studies utilizing whole-watershed manipulations can address some of these shortcomings, if conducted over a sufficiently long period of time. In 1987, the US Environmental Protection Agency (EPA) funded a watershed manipulation project on the Fernow Experimental Forest. In this study, an acidifying agent was applied to a forested watershed over time to simulate acidic deposition above that received from natural deposition of pollutants. EPA funding was discontinued after 3 years as a result of Congressionally mandated budget cuts. However, this project is now in its fifth year, supported in full by the US Department of Agriculture (USDA) Forest Service. This paper describes stream chemistry responses of two treated watersheds to 3 years of treatment.

Location and methods

Description of watersheds

The experimental watersheds used in this study are located on or near the Fernow Experimental Forest near Parsons, West Virginia, USA (39°3′15″N, 79°41′15″W; Fig. 1). This area is part of the unglaciated Allegheny plateau of the Appalachian mountains and is characterized by steep slopes and shallow soils. Precipitation is distributed evenly between dormant and growing seasons on these watersheds. Precipitation in the area is among the most acidic in the USA. Average annual pH is 4.20, but pH readings below 4.0 are common during summer months (Helvey et al., 1982). On all three watersheds, the predominant soil type 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). All of the streams are small second-order streams. Because of shallow soils (1 m depth or less), steep slopes and relatively little near-surface groundwater storage, flow is high during periods of high precipitation and falls off quickly during periods

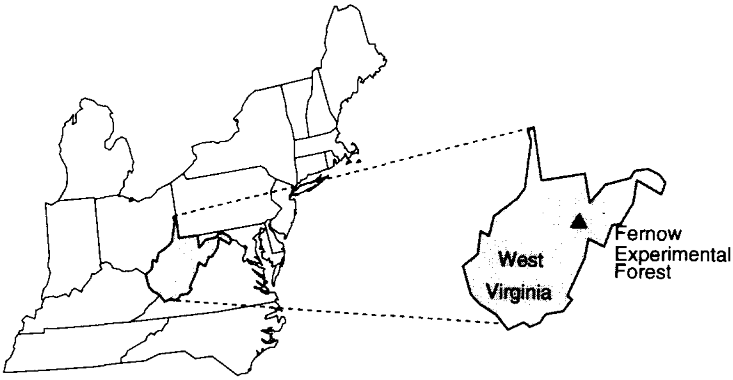

Fig. 1. Location of Fernow Experimental Forest in West Virginia, USA.

with little or no precipitation (Reinhart et al., 1963). Other watershed characteristics are described in Table 1.

Watershed 9 (WS9) was farmed during the 1800s until the 1920s, leaving a nutrient-poor, eroded site, which was revegetated by poor-quality hardwoods (Lima et al., 1978). In November 1983, the area was prepared for planting: a bulldozer was used to remove standing trees and push the residue into windrows (Kochenderfer and Helvey, 1989). Japanese larch (*Larix leptolepis* Sieb. and Zucc.) seedlings were planted the following spring.

Watershed 3 (WS3) was clearcut to 2.5 cm diameter at breast height (dbh) between July 1969 and May 1970 except for a 2.99 ha buffer strip along the stream channel in which a light selection cut was made. In November 1972 the

Table 1
Characteristics of three experimental watersheds

	Watershed		
	3	4	9
Area (ha)	34.3	38.7	11.6
Aspect	S	ESE	S
Minimum elevation (m)	735	750	760
Maximum elevation (m)	860	870	840
Average annual precipitation (mm)	1480	1450	1550
Average annual streamflow (mm)	666	640	760
Average slope (%)	27	20	25
Dominant tree species	*Prunus serotina* *Acer rubrum* *Betula lenta* *Fagus grandifolia*	*Acer saccharum* *Acer rubrum* *Fagus grandifolia* *Quercus rubra*	*Larix leptolepis*

buffer strip was clearcut and all debris in the channel and within 2.5 m on either side of the stream channel was manually removed. After each cut the stand was allowed to revegetate naturally. Dominant tree species include *Prunus serotina* Ehrh., *Acer rubrum* L., *Betula lenta* L. and *Fagus grandifolia* Ehrh.

Watershed 4 (WS4) serves as a reference (control) catchment. The forest stand on this watershed has been relatively undisturbed since approximately 1905, when the area was heavily cut, but not clearcut. In the 1940s, dead American chestnut (*Castanea dentata* Marsh.) was removed from the watershed. Current dominant vegetation is approximately 85 years old, but some residual trees may be up to 200 years old. Dominant tree species include *Acer saccharum* Marsh., *Acer rubrum* L., *Fagus grandifolia* Ehrh. and *Quercus rubra* L.

Description of treatments

Granular ammonium sulfate fertilizer (21–0–0–24, relative proportions of N (nitrogen), P (phosphorus), K (potassium) and S (sulfur)) was used to simulate N and S inputs to WS3 and WS9. The proportions of N and S in this fertilizer closely mimic N and S in throughfall on these watersheds. Fertilizer was applied at a rate double the ambient N and S inputs in throughfall, which are considered to approximate the combined inputs of wet and dry deposition. Because ambient deposition varies seasonally, with approximately half of the annual S deposition occurring from May to August, three applications were made per year, in March, July and November. The March applications were double the average historical deposition rates for the January–April period, July applications were double the May–August rates, and November applications were double the September–December rates (33 kg ha^{-1} for March and November, and 101 kg ha^{-1} for July). These rates correspond to 8.1 kg S ha^{-1} and 7.1 kg N ha^{-1} for the March and November applications, and 24.4 kg S ha^{-1} and 21.3 kg N ha^{-1} for the July applications.

Treatment of WS9 began in April 1987, and of WS3 in January 1989. Treatments will continue through 1993, when the need for further treatment will be evaluated. Three years of pretreatment data (calibration data) and 3 years of treatment data are included in this analysis.

Methods and measurements

A 120° V-notch weir in combination with an FW-1 water-level recorder (Belfort Instrument Co., Baltimore, MD) is located at the mouth of each of

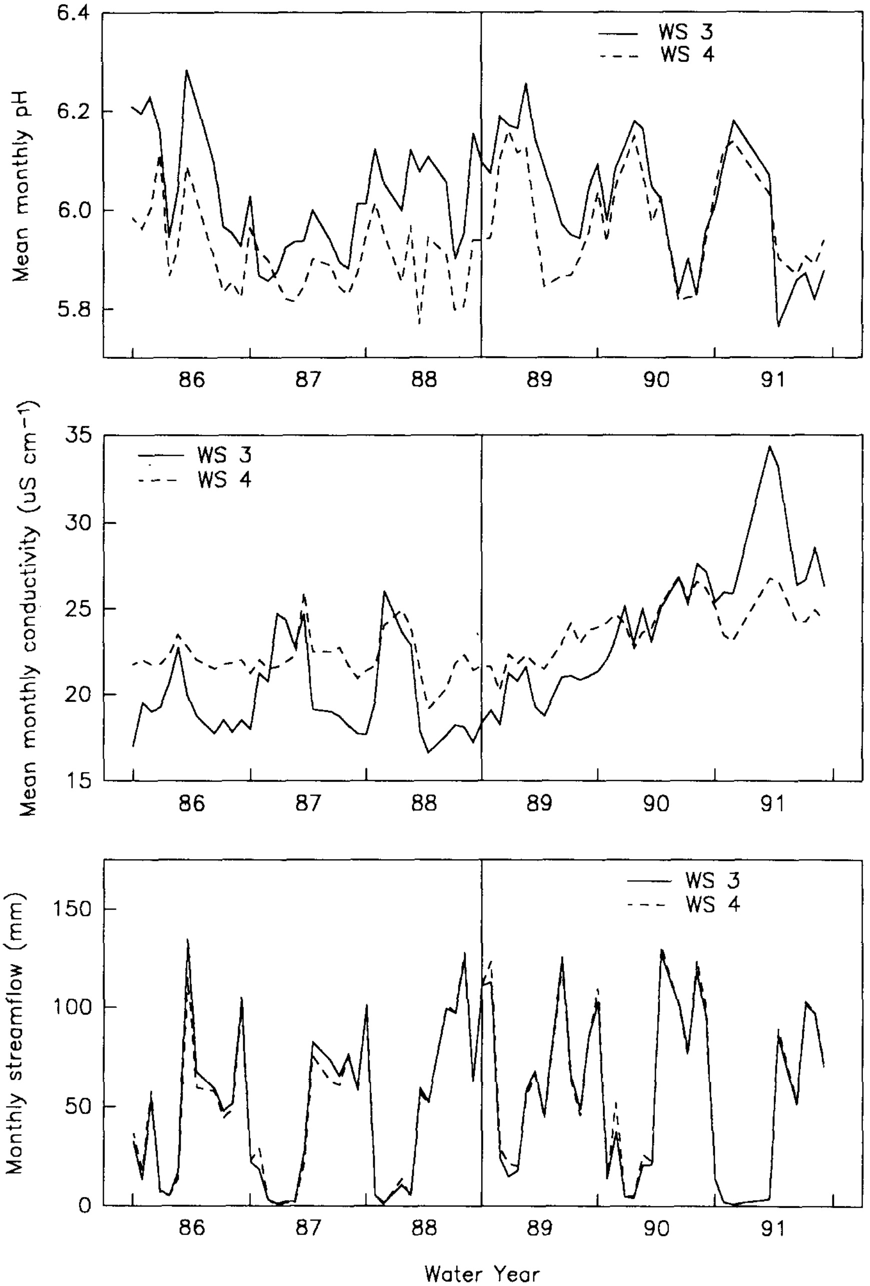

Fig. 2. Mean monthly pH, electrical conductivity and streamflow for a stream draining a treated watershed (WS3) compared with that draining an untreated control (WS4). Vertical line indicates initiation of treatment.

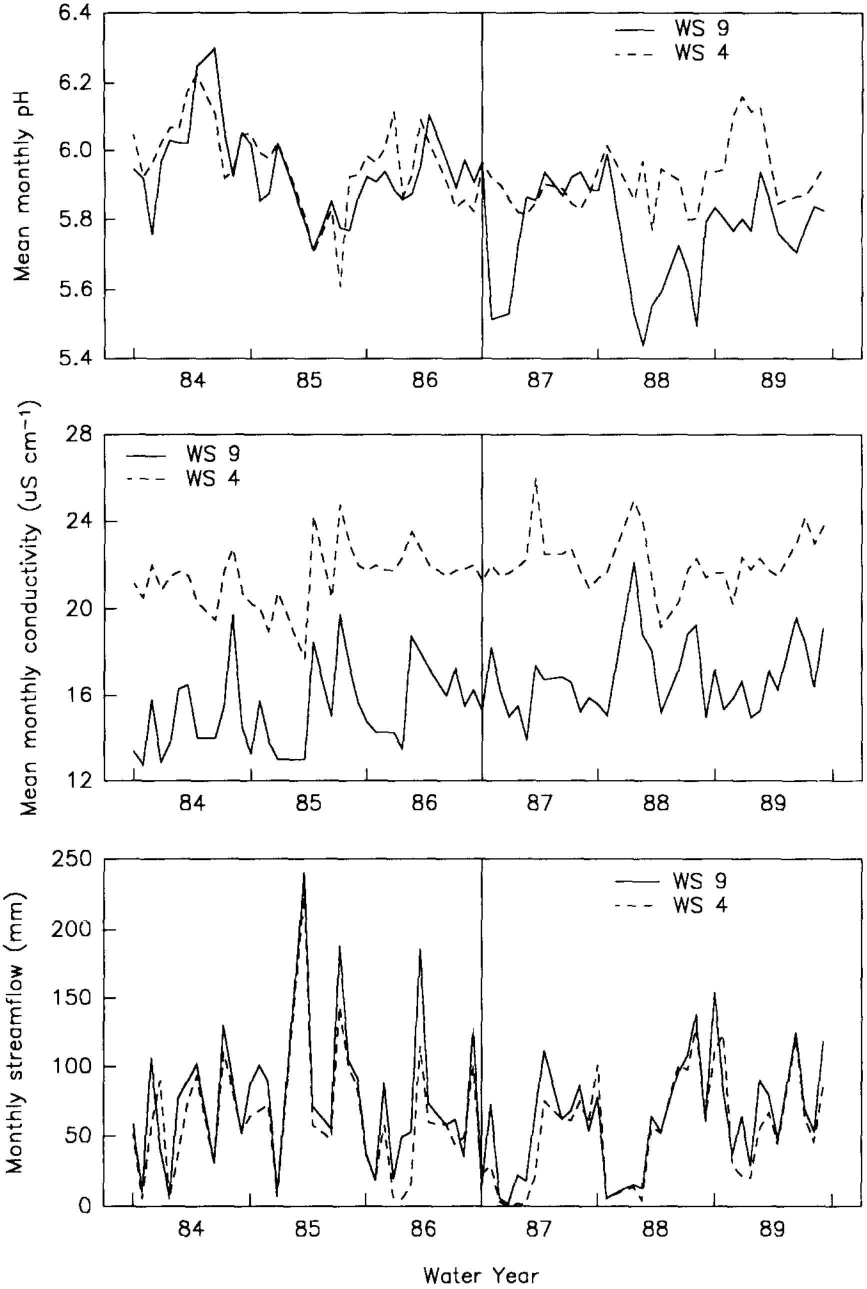

Fig. 3. Mean monthly pH, electrical conductivity, and streamflow for a stream draining a treated watershed (WS9) compared with that draining an untreated control (WS4). Vertical line indicates initiation of treatment.

the watersheds for continuous measurement of streamflow. Streams have been gaged since 1951 on WS3 and WS4, and since 1957 on WS9. Weekly or every 2 weeks grab samples of stream water were collected from a permanently marked location just upstream from each weir. All water samples were analyzed at the USDA Forest Service's Timber and Watershed Laboratory in Parsons, using EPA approved protocols, holding times, and quality assurance/quality control procedures (Edwards and Wood, 1993). Specific analyte and instrumentation information have been given by Edwards and Kochenderfer (1993). This paper focuses on pH, electrical conductivity, SO_4, NO_3, and Ca.

Stream chemistry loadings (total nutrient export on an equal area basis; kg ha^{-1} month^{-1}), were calculated as mean weekly or every 2 weeks concentration × flow × watershed area, with the exception of pH and electrical conductivity, for which mean monthly values were used. Monthly totals were used in data analysis to eliminate the problem of unequal sample sizes or missing data. Data were split into two groups – a 3 year pretreatment or calibration group and a 3 year treatment period group. Data were analyzed using either linear regression or time series analysis depending on the presence of serial correlation in the error terms. Bartlett's Kolmogorov–Smirnov test further showed that all WS9 and WS4 paired series, except pH, were white noise (i.e. not stationary time series), so linear regression analysis was used (Fuller, 1976). Separate regression equations with WS3 or WS9 loadings or means as the dependent variables and WS4 loadings or means as the

Table 2
Regression results, pretreatment and treatment regressions

Analyte	Calibration intercept/slope	Treatment intercept/slope	Calibration R^2	Treatment R^2	Slope significant[a]	Intercept significant[a]
WS 3/4 comparisons						
pH	0.884/0.879	0.524/0.920	0.57	0.60	N	N
Electrical conductivity	−9.951/1.341	−15.014/1.641	0.55	0.64	N	N
Calcium	0.001/0.800	0.003/0.934	0.97	0.92	Y	N
Sulfate	−0.015/0.813	−0.086/0.801	0.97	0.99	N	N
Nitrate	0.018/0.846	−0.051/1.260	0.98	0.92	Y	N
WS 9/4 comparisons						
pH	2.539/0.570	3.302/0.417	0.34	0.06	N	N
Electrical conductivity	−7.702/1.075	0.151/0.751	0.55	0.33	Y	Y
Calcium	0.085/0.760	0.085/0.797	0.74	0.83	N	N
Sulfate	0.437/0.704	0.401/0.807	0.70	0.76	N	N
Nitrate	0.027/0.161	0.011/0.150	0.78	0.80	N	N

[a] Indicates statistically significant difference (Y, yes; N, no; $P = 0.95$) between treatment and calibration values.

independent variables were developed for each 3 year period. Slopes and intercepts were then tested to determine if they were statistically different ($P = 0.95$).

Results and discussion

Mean monthly stream pH for the treatment and control streams ranged from approximately 5.4 to 6.3, fluctuating during the year and with flow (Figs. 2 and 3). Apparent changes in stream water pH of WS9 after treatment are not statistically significant, nor are there any statistically significant changes in pH on WS3 as a result of 3 years of treatment (Table 2). Mean monthly electrical conductivity appears to have increased slightly on WS9 and WS3 since treatment began (Figs. 2 and 3); however, there were no significant

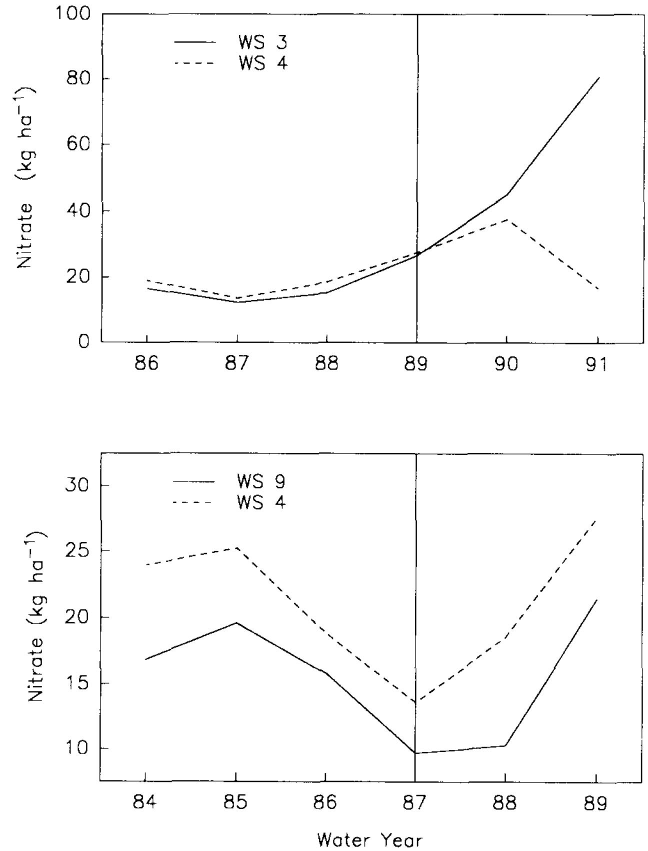

Fig. 4. Annual NO_3 load in stream water draining two treated watersheds (WS3 and WS9) compared with that draining an untreated control (WS4). Vertical line indicates initiation of treatment.

treatment effects on electrical conductivity. Although a statistically significant difference was detected for WS9/4 electrical conductivity, using ordinary least squares, the fit of the regressions was so poor that the validity of this difference is questionable. Also, the increase parallels that of WS4, suggesting this was not a treatment effect. The reason for the increase in electrical conductivity on WS4 is not well understood, but the trend of increasing electrical conductivity is a continuing one (Edwards and Helvey, 1991), and has been observed on another similar control watershed nearby (M.B. Adams, unpublished data, 1992). It should be noted that electrical conductivity was generally greater for WS9 relative to WS3 and WS4.

Stream water SO_4 concentrations have not increased on WS3 or WS9 since treatment began (Table 2), despite expectations that soils on these watersheds were near SO_4 adsorption capacity. Nitrate export appears to have increased on the treatment watersheds (Fig. 4), but the change is significant only for WS3, where the slopes of the two regression lines are significantly different (Fig. 5). Stream water Ca export similarly increased on WS3 (Fig. 6).

Table 3 lists annual precipitation inputs and stream water outputs from these watersheds. Calcium and NO_3 export from WS3 increased during the 3 years of treatment, with considerably higher values in water year 1991.

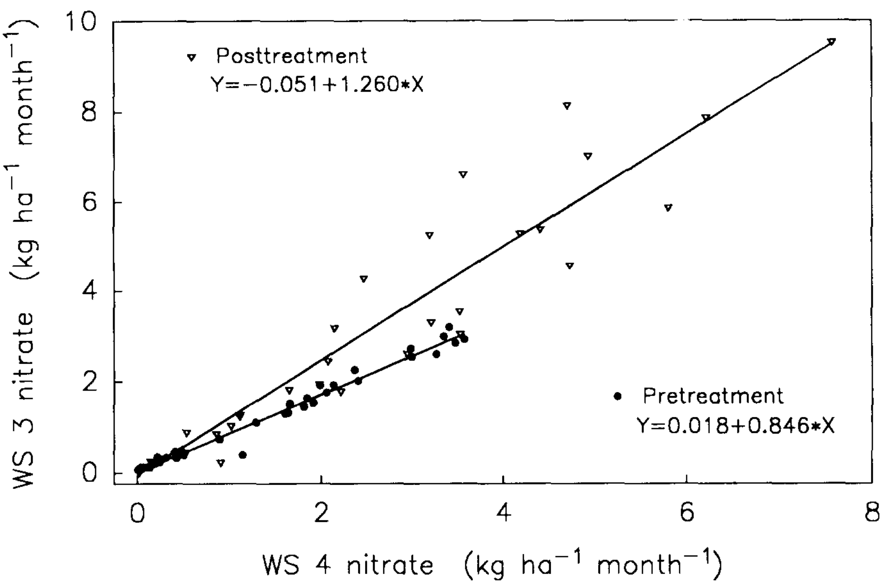

Fig. 5. Regressions of monthly NO_3 loading in stream water for treated watershed (WS3) vs. untreated control watershed (WS4); 3 years of calibration data are compared with 3 years of treatment data.

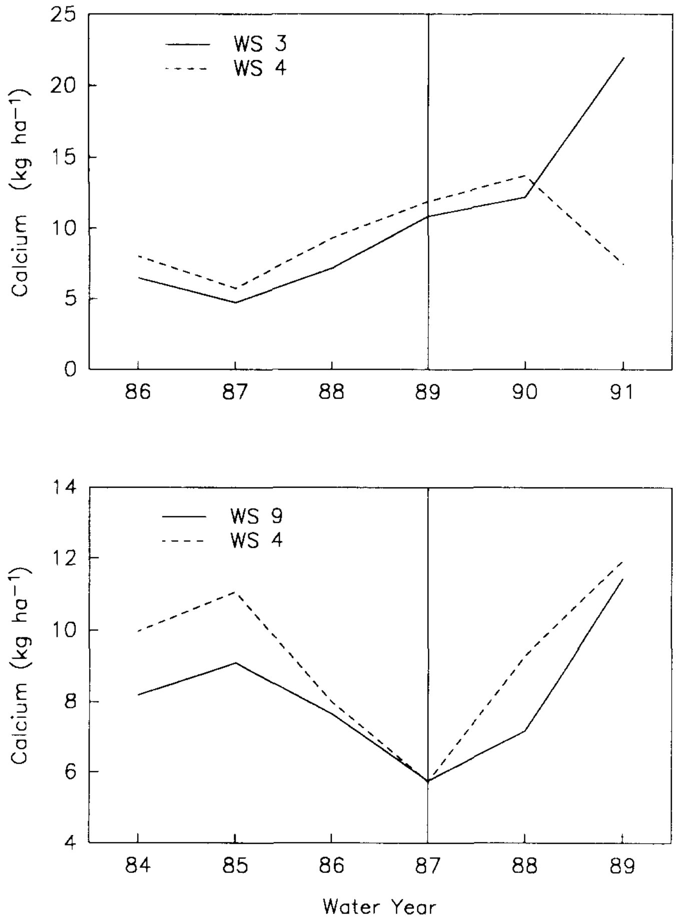

Fig. 6. Annual Ca load in stream water draining two treated watersheds (WS3 and WS9) compared with that draining an untreated control (WS4). Vertical line indicates initiation of treatment.

Annual throughfall averages 16 kg N ha^{-1}. Of that, approximately one-third is NO_3-N (Helvey and Kunkle, 1986). Therefore, approximately 22 kg of NO_3 ha^{-1} is deposited to the forest floor. If we assume that all of the N applied as fertilizer (36 kg N ha^{-1}) is converted to NO_3, then 158 kg NO_3 ha^{-1} is added from the experimental treatment, for total inputs of 190 kg NO_3 ha^{-1}. In water year 1991, about half of that left WS3 via streamflow. The 3 years of treatment of WS9 did not produce such high rates of NO_3 export, however. Lima et al. (1978) detailed the history and characteristics of WS9 and described 'dangerously depleted organic matter, erosion and sedimentation'. Severe erosion was estimated for about 90% of the area (Losche and Beverage, 1967), and revegetation predominantly consisted of low-value

Table 3
Annual (water year; 1 May–30 April) loadings in precipitation and streamflow for three experimental watersheds (kg ha^{-1} per water year)

	WS3		WS4		WS9	
	Precipitation	Streamflow	Precipitation	Streamflow	Precipitation	Streamflow
Calcium						
1984			6.16	9.97	5.81	8.20
1985			3.55	11.07	3.73	9.10
1986	3.65	6.45	3.23	8.00	3.17	7.65
1987	4.11	4.73	3.64	5.73	3.70	5.76
1988	3.31	7.17	2.93	9.30	2.83	7.16
1989	2.59	10.87	2.29	11.94	3.62	11.44
1990	4.32	12.22	3.83	13.72		
1991	2.41	21.96	2.14	7.44		
Sulfate						
1984			41.78	29.37	55.82	24.71
1985			53.05	34.65	68.58	28.28
1986	43.26	20.32	38.28	25.12	59.88	24.04
1987	29.77	14.45	26.34	17.62	45.46	18.37
1988	29.66	22.84	26.25	29.43	36.96	26.06
1989	40.25	28.02	35.61	36.33	64.79	36.43
1990	41.95	23.88	37.12	32.74		
1991	32.18	43.88	28.48	20.06		
Nitrate						
1984			22.09	23.97	32.58	16.87
1985			31.70	25.25	38.60	19.63
1986	24.24	16.46	21.69	18.88	31.96	15.83
1987	19.74	12.24	17.44	13.61	27.31	9.72
1988	16.51	15.30	14.61	18.59	27.05	10.28
1989	23.02	26.71	20.36	27.66	37.05	21.49
1990	23.06	45.11	20.41	37.70		
1991	18.28	80.88	16.16	16.86		

hardwoods typical of poorer sites (Lima et al., 1978). All of these observations suggest an infertile site, which may explain the lack of response to the acidification treatments. Feger (1992) also reported little N loss on a low-fertility site (Table 4). Calcium and Mg concentrations (weak acid extraction) for soils on WS9 were 103 and 15 mg kg^{-1}, respectively (J.N. Kochenderfer, unpublished data, 1992). These compare with 121 mg kg^{-1} Ca and 20 mg kg^{-1} Mg, respectively, for soils on WS3 (F.S. Gilliam, unpublished data, 1992). Thus, with respect to Ca and Mg, the two sites do not differ greatly. One explanation for this relative fertility of WS9 was suggested by the presence of limestone fragments, an indicator that burnt lime was applied when the watershed was farmed. This was a common practice in the area.

Table 4
Results of artificial acidification field studies

Reference	Location	Soil type	Treatment	Results
Wright et al. (1988)	Sogndal, Norway	Lithic Haplumbrept	Acid addition H_2SO_4 + HNO_3, or H_2SO_4 70 mequiv m^{-2} $year^{-1}$ in 1984, 100 mequiv m^{-2} $year^{-1}$ in 1985–1987	HNO_3: minor increase in NO_3 runoff, increased Al runoff, increased SO_4 runoff and increased base cation export
Feger (1992)	Schluchsee, Germany	Iron-humus podzols	$(NH_4)_2SO_4$, 1 year 170 kg S ha^{-1} and 150 kg N ha^{-1}	Increased N export (mainly NO_3), decreased soil solution pH, increased export of H, Al, Mg, no effect on S cycling
	Villingen, Germany	Dystric cambisols	$(NH_4)_2SO_4$, 1 Year 170 kg S ha^{-1} and 150 kg N ha^{-1}	Slight increase in N export (mainly as NO_3), no effect on S cycling
Christopherson et al. (1982)	Storgama, Norway		'Unpolluted' rain 9 μequiv H^+ l^{-1}, 11 μequiv SO_4 l^{-1}	'Unpolluted' precipitation caused lowered levels of SO_4 in runoff within a week
Kahl et al. (1993)	Maine, USA	Typic Haplorthods	$(NH_4)SO_4$, applied once every 2 months since 1989, 25 kg N ha^{-1} $year^{-1}$	Decreased NO_3 retention; NO_3 flux increased from 200 to more than 500 equiv, ha^{-1} $year^{-1}$
Norton et al. (1993)	Maine, USA	Typic Haplorthods	Same as Kahl et al. (1993)	Increased SO_4 export in streamwater

Unfortunately, soil N and S values are not available, so no comparisons can be tested for these elements. However, given its history, WS9 may be severely N deficient, which would explain why there is no increased export of NO_3 from this watershed. Further analyses of soil and plant tissue on WS9 will address this question. It should also be noted that the regressions for WS9 and WS4 have lower R^2 values than those for WS3 and WS4 (Table 2). This may be partially explained by location; WS3 and WS4 are located adjacent to each other, and WS9 is approximately 12 miles away. Thus, subtle treatment differences might not be detected.

The observed increase in Ca outputs from Fernow watersheds is probably due to pairing of Ca with the highly mobile NO_3 ions, as has been reported elsewhere (Table 4). Soil solution data support this hypothesis. Although NO_3 concentrations in soil leachate from WS3 and WS4 exhibit strong seasonal patterns, comparisons between the watersheds suggest that the amplitude of these seasonal cycles has been elevated on the treatment watershed owing to the amendment (Edwards et al., 1993). Calcium and Mg also closely track NO_3.

Because this area of the country receives some of the highest loadings of S (Cowling, 1983), there was concern about SO_4 losses from the watersheds, and the resulting cation leaching that would occur. However, SO_4 inputs are generally greater than outputs in streamflow; thus, these soils do appear to be adsorbing much of the SO_4 inputs (Helvey and Kunkle, 1986). Changes in chemistry of A-horizon leachate have been observed on WS3 (Edwards and Wood, 1992). Leachate SO_4 concentrations have increased steadily since early 1990 on WS3, but have remained relatively constant on WS4. As no increases occurred in the B- or C-horizon leachate or in stream water, most of the mobile SO_4 probably was adsorbed in the lower horizons. WS9 lysimeters are located at 45 cm, therefore the fate of ions below that depth is unknown. However, data from WS9 lysimeters suggest a situation similar to that for WS3 (P.J. Edwards, unpublished data, 1993). Sulfate is moving through the top layers, but because there are no significant differences in stream chemistry, it is assumed that most of the applied SO_4 is being retained by the lower horizons.

After 3 years of additions of twice-ambient levels of nitrogen and sulfur deposition, one treated watershed may be beginning to show changes in chemistry, whereas the other has not been significantly affected. Sulfate is still being adsorbed by the soils of both watersheds, but NO_3 is leaching from WS3 and carrying Ca with it. The differences may be due to different species' nutrient requirements, site fertility, or other unknown factors. Research to examine internal nutrient budgets of these watersheds is continuing and should shed some light on these findings. The impacts on aquatic and terrestrial biota also are being evaluated, to provide further information on other aspects of ecosystem function and stability.

Other researchers also have used whole-watershed acidification experiments to evaluate N and/or S movement through ecosystems (Table 4). Although soils, climate and treatments vary among these studies, some similar results have been recorded. On sites that are not severely N limited, NO_3 export generally increased in response to nitrogen applications. A concurrent increase in cation leaching also was observed. Feger (1992) reported no effects on sulfur cycling, whereas Wright et al. (1988) and Norton et al. (1993) did observe increased SO_4 runoff. The increase in N export (mostly as NO_3) suggests that there may be many forest ecosystems which may be susceptible to 'nitrogen saturation', if the definition of Aber et al. (1989) is accepted. Because conventional wisdom tells us that most forests are N limited, a seeming contradiction appears, which requires further investigation.

Simulation models such as MAGIC (Cosby et al., 1985; Wright et al., 1988) were developed to evaluate changes in water chemistry as a result of changes in precipitation chemistry. Generally, such models were designed to address S deposition impacts, although N may also be taken into account in some. However, on the basis of studies such as that described here, and the knowledge that whereas S deposition may decrease, N deposition is likely to continue to increase, it may be more important to develop and refine models to address the problems of N saturation. Information from whole-watershed studies can lend support to such modeling efforts.

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