LONG-TERM STREAM CHEMISTRY MONITORING ON THE FERNOW EXPERIMENTAL FOREST: IMPLICATIONS FOR SUSTAINABLE MANAGEMENT OF HARDWOOD FORESTS

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Abstract—Long-term monitoring of stream chemistry of forested watersheds on the Fernow Experimental Forest in West Virginia has been conducted to determine the effects of both human-induced and natural disturbances on nutrient cycling and stream chemistry. We compare mean annual stream water pH, and nitrate (NO_3), sulfate (SO_4), and calcium (Ca) concentrations from 6 gauged Fernow watersheds with different disturbance regimes for the last 30 years. Most disturbances are not sufficiently large in area or extent to have a detectable effect on stream chemistry (diameter-limit or selection harvesting, clearcutting, windstorms). Fertilization, acidic deposition at ambient levels, maintaining watersheds devoid of vegetation, and conversion to conifers significantly affected stream water chemistry. Implications for managing hardwood forests for sustainability are discussed.

INTRODUCTION

Long-term watershed monitoring is a hallmark of the USDA Forest Service Research and Development. Because trees are long-lived organisms and because of the temporal variability associated with climate and other factors, such long-term research is necessary in order to understand the effects of forest management activities and natural disturbances on forest ecosystem processes. In this manuscript we present long-term stream chemistry data from the Fernow Experimental Forest (FEF), one of the few such long-term studies of stream chemistry in the central Appalachians, and discuss effects of both human-induced and natural disturbances on nutrient cycling and stream chemistry. In particular, we evaluate timber harvesting, deforestation, fertilization, acidic deposition and changes in dominant species.

We present data from 6 gaged watersheds (WS) on the FEF (table 1). The stream flow records date back to 1951, and we use stream chemistry data collected beginning in 1971. Older stream chemistry data, and data from other sources, are used to illustrate specific points, as needed. Many of the data have been published previously, as partial data sets; this represents one of the first presentations of more than 30 years of stream chemistry data.

The Watersheds

The FEF (39.03° N, 79.67° W) is located in north-central West Virginia, in the Allegheny Mountain section of the mixed mesophytic forest. Central Appalachian forests have been shaped by a mixture of natural and human caused disturbances including wind, fire, logging, and agricultural use, creating a diverse mosaic of forest stands. More *recently*, several insects and diseases, most of them non-native, have severely impacted Appalachian forests. The chestnut blight (*Cryphonectria parasitica*) has been the most devastating, virtually eliminating American chestnut (*Castanea dentata* [Marsh.] Borkh.), which formerly comprised 25 percent of Appalachian forests, including those of the FEF. Acidic deposition and other air pollutants are a more recent, chronic disturbance (Adams 1999).

Diversity is an important characteristic of central Appalachian forests, and the FEF vegetation fits into Core's (1966) mixed central hardwood forests floristic province. Common tree species on the sites with higher site index are yellow-poplar (*Liriodendron tulipifera* L.), sugar maple (*Acer saccharum* Marsh.), black cherry (*Prunus serotina* Ehrh.), red oak (*Quercus rubra* L.) and basswood (*Tilia americana* L.). Dominant tree species on the poorer sites include white oak (*Q. alba* L.), chestnut oak (*Q. prinus* L.),

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Table 1—Fernow watersheds: treatment histories and descriptive references

WS	Area	Treatment	Treatment date	References
	ha			
1	30	Clearcut to 15 cm d.b.h. except culls	1957–58	Patric and Smith 1978
		Fertilized with 500 kg urea per ha	May, 1971	Kochenderfer and Aubertin 1975
2	15	43 cm diameter limit cut	1958	
		43 cm diameter limit (11 ha)	1972, 1988, 2004	
		Fertilized with 336 kg N/ha and 224 kg/ha P_2O_5, 1.3 ha	April, 1976	Helvey and others 1989
		43 cm diameter limit cut, 5 ha	1978, 1997	
3	34	Intensive selection cut, including cull trees > 12.7 cm d.b.h	1958–59, 1963	Aubertin and Patric 1974, Kochenderfer and others 1990
		0.16 ha patch cuttings totaling 2.3 ha, cut down to 12.7 cm, 2-12 cm stems sprayed with herbicide	July,1968–August, 1968	
		2-12 cm stems treated with herbicide then clearcut to 2.5 cm d.b.h., except for a partially cut 3.0 ha shade strip along the stream channel	July, 1969–May, 1970	
		Shade strip clearcut	November, 1972	
		Natural recovery	November, 1972–present	
		Ammonium sulfate fertilizer applied	December, 1989–present	Edwards and others 2002
4	39	No treatment; natural recovery since 1905		Reinhart and others 1963
6	22	Lower 11 ha clearcut	1964	
		Maintained barren with herbicides	March, 1965–October, 1969	
		Upper 11 ha clearcut	1967–68	
		Maintained barren with herbicides	May, 1968–October, 1969	
		Planted Norway spruce	1973	
		Aerial application, herbicide to release spruce	1975, 1980	
7	24	Upper 12 ha clearcut	1963–64	Patric and Reinhart 1971
		Maintained barren with herbicides	May, 1964–October, 1969	
		Lower 12 ha clearcut	1966–67	
		Entire watershed maintained barren with herbicides	May, 1967–October, 1969	Kochenderfer and Wendel 1983
		Natural recovery	October, 1969–present	Adams and others 1995

WS = watersheds.

hickory (*Carya* spp.), red maple (*A. rubrum* L.), and American beech (*Fagus grandifolia* Ehrh). A partial list of more than 500 species of vascular flora found on the FEF (Madarish and others 2002) illustrates the diversity of these forests.

The growing season on the FEF extends from May through October, and the average length of the frost free season is 145 days. Annual precipitation is about evenly distributed between growing and dormant seasons, averaging 145.8 cm. Precipitation often occurs in the form of snow during the winter but a snowpack usually does not exist for extended periods. Average annual air temperature is 9.2^{o}C (Pan and others 1997), and mean monthly temperatures range from -18^{o}C in January to 20.6^{o}C in July. Potential evapotranspiration on the Fernow was estimated to be 56 cm per year (Patric and Goswami 1968).

The hydrometeorologic network used on the Fernow is described by Adams and others (1994). All of the watersheds are instrumented with 120^{o} V-notch weirs, with FW-1 water level recorders and 7-day strip charts to measure streamflow continuously. Stream water grab samples have been collected from the watersheds on a weekly or bi-weekly basis since 1960. Solution samples are analyzed as described in Edwards and Wood (1993). Watershed 4 (WS4) serves as the reference watershed, against which all others are compared. WS4, and most of the Elk Lick watershed which makes up most of the FEF, was cut around 1905. The watersheds and their respective treatments are described in table 1.

What We've Learned:

Figure 1 shows annual stream water pH from 6 streams on the FEF. Stream pH is often used as an indicator of water quality, particularly in reference to aquatic organisms, such as trout (Cleveland and others 1986). Figure 1 suggests that annual stream pH, at least on the FEF watersheds, is not particularly sensitive to disturbance. Only on WS3 has stream pH decreased significantly over time (Edwards and others, in press), as the result of experimental fertilizer additions. Since 1989 we have been applying ammonium sulfate fertilizer at twice ambient nitrogen (N) and sulfur(S) deposition levels to WS3 to

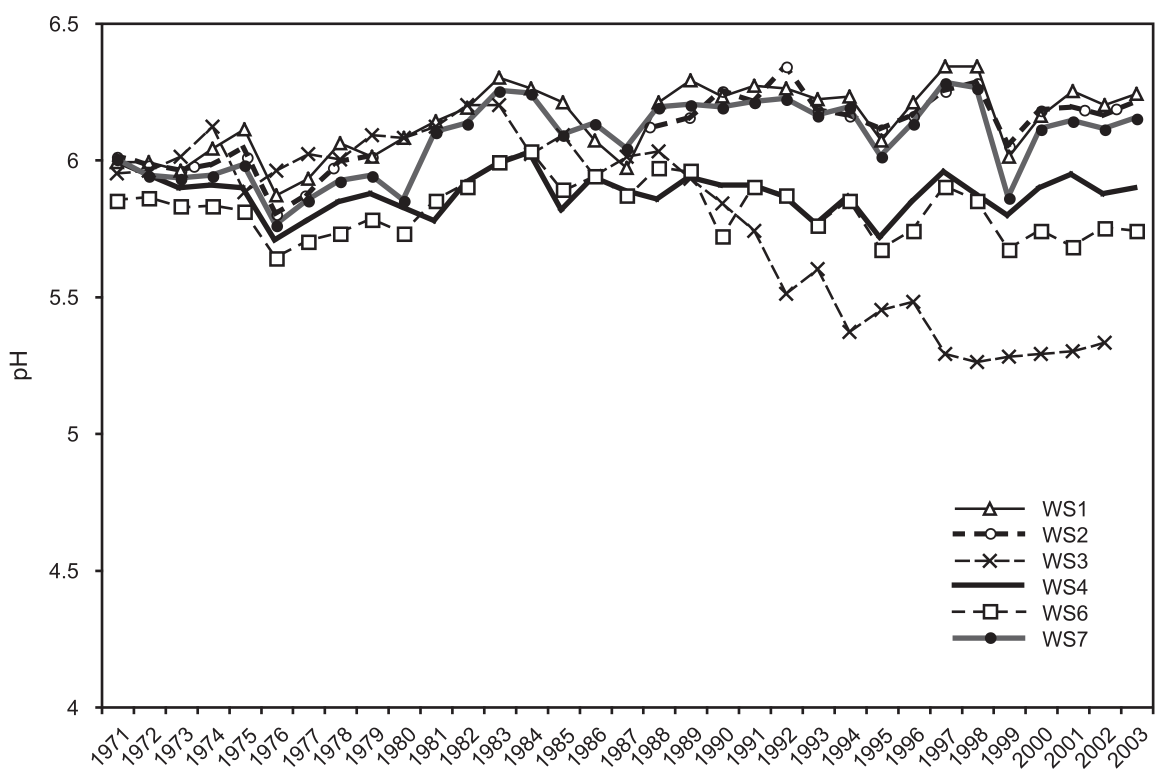

Figure 1—Annual volume-weighted stream pH from six watersheds on the Fernow Experimental Forest.

evaluate watershed acidification responses. Clearly, watershed acidification can be induced after long periods of acidic deposition: treatments to WS3 represent 28 years of ambient deposition. However, most streams in the FEF are still between pH 5.5 and 6.5, despite acidic bedrock, clearcutting (WS3 early years), repeated cuttings (WS2), years of ambient acidic deposition (WS4), repeated blowdown of trees (WS4), and even conversion to conifers (WS6).

Trends in NO_3-N concentrations are shown in figure 2. Nitrate (NO_3) is a concern because elevated NO_3 concentrations in water can have significant implications for ecosystem processes, as well as human health implications. A high rate of NO_3 leaching from a watershed is one symptom of nitrogen (N) saturation (Aber and others 1998, Fenn and others 1998). Research in other forest types has reported elevated NO_3 losses as a result of clearcutting (Likens and others 1970, Niemenen 1998, Waide and others 1988). To evaluate clearcutting effects on the FEF, we might compare concentrations in WS3, WS6, WS7, and WS1 to WS4, the reference WS. WS3 was clear cut in 1969-1970, therefore these concentrations represent the most recent post-clearcutting data. WS6 and WS7 were clearcut and maintained barren in the mid-1960's (table 1), but permitted to regenerate beginning in 1969/1970, the same year as WS3. We know that on WS1, which was clearcut in in 1958, stream flow and water quality returned to pretreatment levels within 2-4 years post-clearcutting (Kochenderfer and Aubertin 1975), so we will not consider WS1 in this discussion of clearcutting effects. Aubertin and Patric (1974) reported that clearcutting WS3 had a negligible effect on most of the stream chemistry analytes they evaluated; they did record that stream water NO_3-N concentrations showed a slight increase in July and August of 1970, then declined to base levels again before increasing in response to a 64 mm rainfall in December 1970. Thereafter concentrations were comparable to those from WS4. Note however, that stream water NO_3-N concentrations draining WS6 and WS7, which regenerated the same year as WS3, are more than twice the concentrations of WS3, for at least the first 4-5 years. We attribute this large difference not

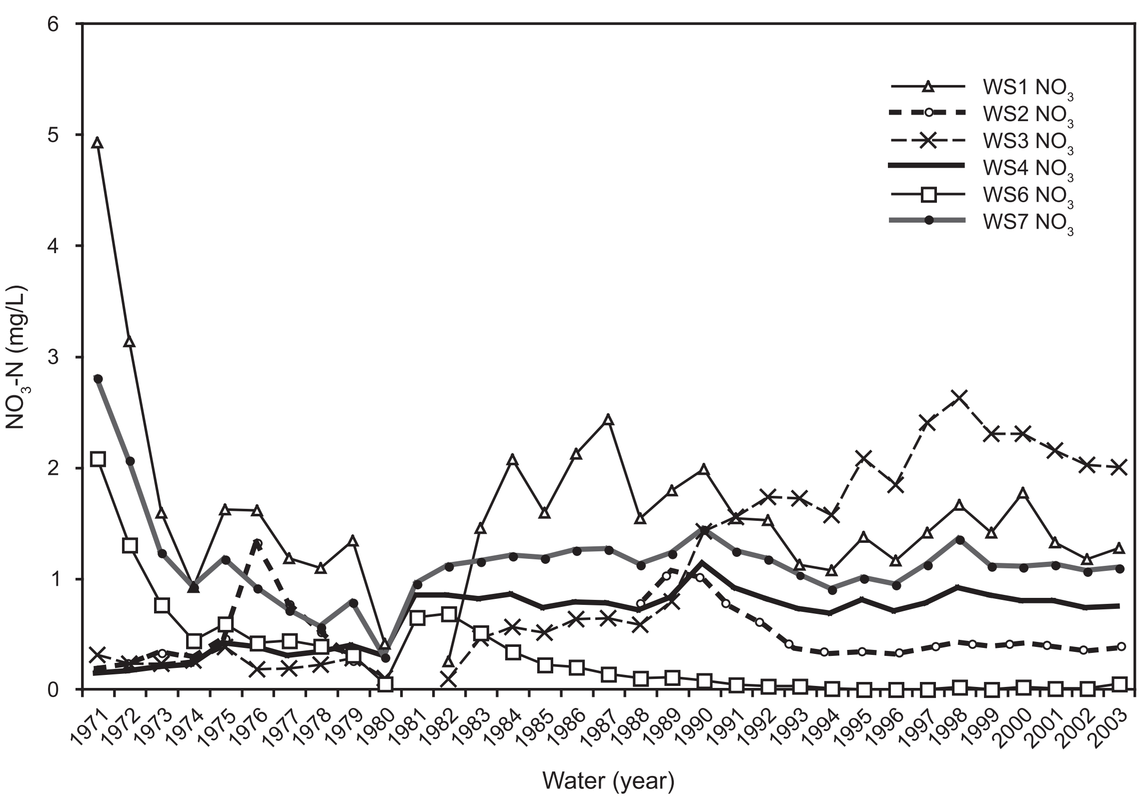

Figure 2—Annual volume-weighted stream NO_3-N concentrations from six watersheds on the Fernow Experimental Forest.

to the clearcutting of WS6 and WS7, but to the herbicide treatments, since WS3 NO_3-N concentrations remained quite low. WS6 and WS7 were maintained barren of vegetation for a number of years through repeated herbicide applications, as part of a study to evaluate water yields and the sources of water flow within a watershed. Thus there was no vegetative sink for the nutrients which were quickly released by the decomposition of slash and organic matter. Once the stands on WS6 and WS7 began to regrow, stream water NO_3 concentrations decreased quickly as N uptake again became an important sink. This distinction is important for forest managers. It is obviously not desirable to maintain a forested area devoid of vegetation, due to concerns about erosion, nutrient losses, habitat, among others, but clearcutting need not result in large increases in N loss from a watershed. Similar results were reported at Hubbard Brook (Likens and others 1970).

Fertilization can significantly increase NO_3 leaching from a watershed, whether it arrives in one large application (WS1, WS2), or chronic levels (WS3). The very high NO_3-N concentrations associated with WS1 can be attributed to a fertilizer application of 500 kg/ha of urea (230 kg/ha of soluble N) in May 1971 (Aubertin and others 1973). A slight, short-lived peak in annual NO_3-N concentration is also observed for WS2 in 1972, which also received a single large application of N fertilizer to a portion of the watershed (Helvey and others 1989). Stream water NO_3-N concentrations have been increasing steadily as a result of relatively low levels of fertilizer additions (35 kg N/ha/year and 40 kg S/ha/year), applied since in 1989 to WS3. Fertilizer applications of ammonium sulfate to WS3 have been rapidly mineralized and nitrified (Gilliam and others 1994, 2001), and move quickly through the soil profile (Edwards and others, in press). Although there is evidence of increased uptake by the vegetation (Adams and others 1993; DeWalle and others, in press), total exports and concentrations of N have increased significantly as a result of the watershed acidification treatment (Adams and others, in press). The elevated leaching of NO_3-N has led to hypothesis that WS4 (Fenn and others 1998, Peterjohn and others 1996, Stoddard 1994), WS3 (Gilliam and others 2001, Peterjohn and others 1996) and even WS7 (May and others 2005) may be N saturated, in response to ambient (WS4, WS7) or artificially elevated (WS3) levels of acidic deposition. While there is still much debate about the implications and definitions of N-saturation, these findings suggest that all forests are not necessarily net N sinks, with near-infinite N retention capacity.

All of the watersheds are leaching some level of NO_3-N (fig. 2), even WS4, the reference watershed, with the eventual exception of WS6. WS4 is one of the few examples of elevated stream NO_3 associated with ambient levels of N deposition. A comparison of WS6 and WS7 early data showed that stream water concentrations draining these 2 watersheds during the devegetated (barren) stage of the study were similar (Kochenderfer and Aubertin 1975). Nitrate concentrations remained elevated through 1971, and declined during the regeneration stage. Stream water concentrations of NO_3-N did not decline as far or as continually on WS7 as they did on WS6. WS6, which was replanted to Norway spruce in 1973, now shows little or no annual export of NO_3. This is due at least in part to statistically significant decreases in stream flow resulting from the change in tree species (Hornbeck and others 1993). Conifer stands have greater transpiration and interception rates than hardwood stands (Swank and others 1988). However, note that both flow-weighted (fig. 2) and non-weighted (data not shown) concentrations have decreased to near detection limits, which suggests a change in N cycling within the watershed as well, independent of changes in water availability. We are continuing to investigate N cycling in WS6 to identify the specific processes by which this watershed is fully retaining N. Note that stream water NO_3-N concentrations from WS2 also are quite low. The stand on WS2 received a diameter limit cut in 1958, 1972, 1988, 1997 and 2004 (table 1), and no evidence of the treatments has shown up in annual stream water concentrations of NO_3-N. One hypothesis to explain this is that by increasing growth, each of the repeated cuttings has created an increased demand for N for growth of the remaining trees, which are probably more vigorous as a result of more light and more growing space; this results in lower stream water concentrations and export. This hypothesis remains to be evaluated, however. It is also probable that the basal area removed by each of the cuts may have been sufficiently small and dispersed through the watershed that effects could not be detected. Hornbeck and others (1993) determined that, to detect a significant effect of removal upon annual water yield, approximately 25 percent of the basal area of a watershed would need to be removed. None of the cuts in WS2 removed more than 20 percent of the basal area.

Figure 3 shows stream water SO_4 concentrations. Sulfate is a concern because it is associated with soil acidification, and because S has historically been the dominant component of acidic deposition. Deposition of SO_4 has decreased significantly during the last 15 years (Likens and others 2001). Yet, because of these changes and the reactions controlling SO_4 adsorption, SO_4 continues to be an important anion in soil and water exchange and acidification processes. Sulfate concentrations from WS3 have increased as a result of the watershed acidification treatment, although retention by the watershed is occurring (Edwards and others, in press). Overall, SO_4 concentrations appear to be declining or leveling off in recent years, despite increases from early levels. Because SO_4 adsorption is a reversible process, it is difficult to determine what the effects of decreasing SO_4 deposition may be in these watersheds. However, other than the fertilization of WS3, which is a direct addition, there seem to be little or no effects of the other disturbances on stream water SO_4 concentrations.

Figure 4 shows the long-term stream water concentrations of Ca from the six watersheds. Base cation depletion from soils has been hypothesized as a concern due to clearcutting (Fuller and others 1987), acidic deposition (Johnson and others 1991), or a combination of the two (Adams and others 2000). For most of the watersheds, the trends are very similar to those of NO_3-N. Ca concentrations have increased on WS3 as a result of the acidification treatment, and while trends are similar to those of both SO_4 and NO_3-N concentrations, analyses suggest that NO_3 is probably the dominant anion driving Ca leaching on WS3 (Edwards and others, in press). Evidence also exists to suggest cation mobilization and depletion occurring in WS3; the evidence is strongest in soil water and peak flow concentration data, and less strong for baseflow concentration data (Edwards and others, in press).

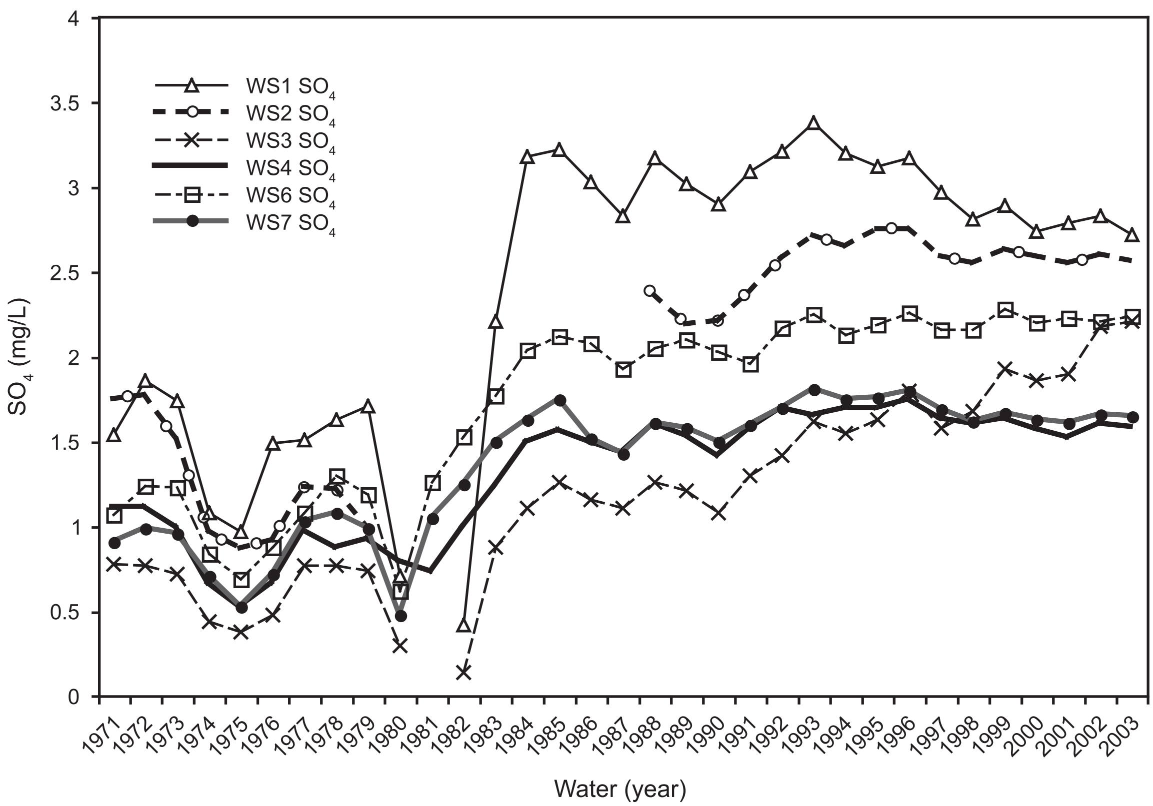

Figure 3—Annual volume-weighted stream SO_4 concentrations from six watersheds on the Fernow Experimental Forest.

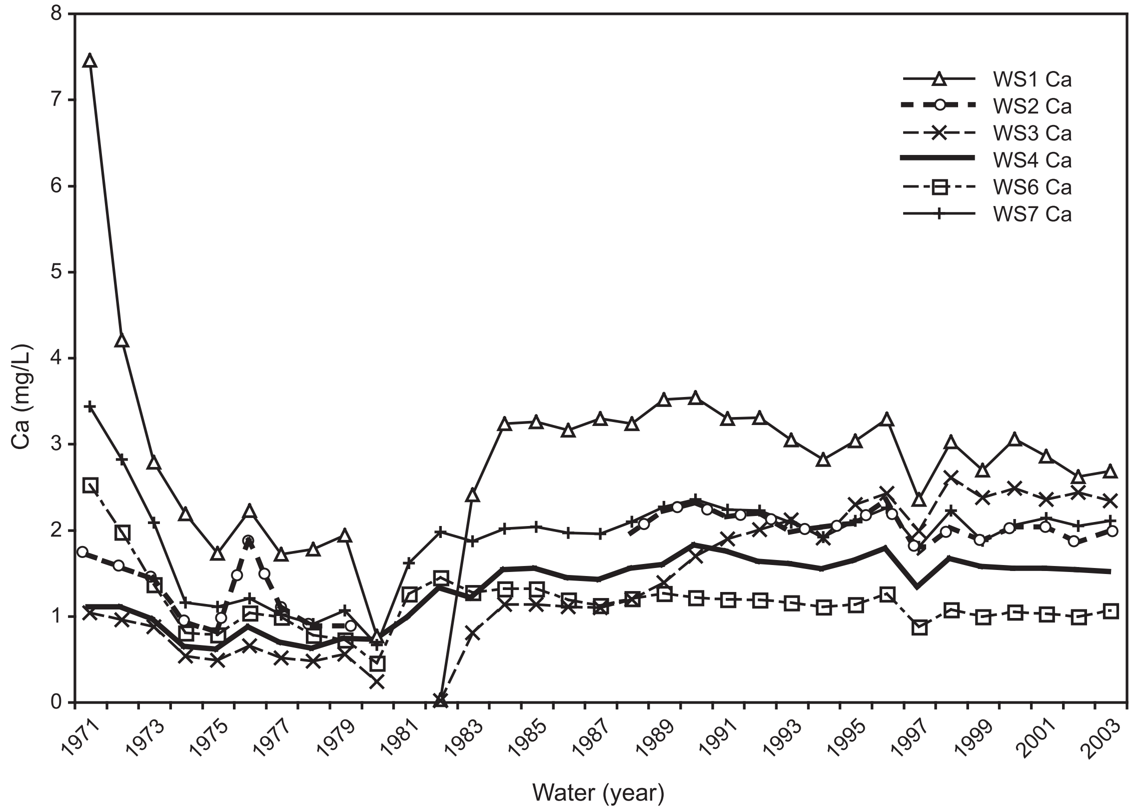

Figure 4—Annual volume-weighted stream Ca concentrations from six watersheds on the Fernow Experimental Forest.

Stream water nutrient concentrations on WS1 are consistently the highest or among the highest for most of those shown in figures 1-4, and for specific conductivity. One explanation could lie with the extreme treatments WS1 has received (the "logger's choice" placement of skid roads during the 1958-1959 clearcutting, a single large application of fertilizer). However, the earliest stream chemistry data available from the FEF show that specific conductance of water draining WS1 was consistently about 1.5-2 times that draining WS4, and the other untreated watersheds (Kochenderfer and Aubertin 1975), even before treatments began.

Significant natural disturbances occurred in 1975, 1985 and 1986 (record rainfall events), and 1993 and 1998 (windstorms which blew down large volume of trees on WS4; Adams and others 2003). These disturbances are not reflected in the annual streamflow chemistry trends in figures 1-4. The windthrow storms, which created 3.5 m^3/ha of new down dead wood each, also did not show up in weekly stream water concentrations or annual sediment exports (data not shown). It is likely that even though these were severe events on a local scale, the effects on the vegetation were not sufficient to significantly change nutrient uptake on the watershed scale. In years with above average rainfall, the watersheds have significantly greater annual flows and nutrient exports, and stream chemical concentrations may be affected as well.

IMPLICATIONS

Stream water chemistry can record and reflect significant disturbances to a forested watershed. Specifically, severe disturbance to the vegetation over a relatively long duration, and fertilization were found to affect stream chemistry. Vegetation appears to significantly regulate or moderate the flux of nutrients from forested watersheds. When vegetation is removed for a long period of time, there are dramatic effects on stream chemistry, as reflected by WS6 and WS7, resulting from the herbiciding treatment. Merely clearcutting, as was done with WS3, proved to have only minor effects on stream chemistry. Repeated partial cuttings, as on WS2 or mimicked in the windstorms on WS4, were neither severe nor long-term enough to produce significant changes in nutrient exports. The conversion of WS6 to spruce had a significant effect on stream water chemistry, and provides further evidence of the importance of the role of vegetation as a regulator of nutrient cycling processes. In this case, the low stream water NO_3 concentrations reflect changes in water movement through a stand as a result of higher interception and transpiration by the spruce relative to the native hardwoods, the effects of greater nutrient uptake due to the potential for year-round physiological activity as temperatures during the dormant season permit, and changes in litter quality and decomposition that can result from species conversion.

The second activity that significantly affected the nutrient cycling of the watersheds, as reflected in annual stream water chemistry, was the application of fertilizer. On WS1, a large single dose of urea fertilizer resulted in an immediate large pulse of NO_3 from the watershed. The treatment of the upper reaches of WS2 also resulted in a small pulse, although not as large as from WS1. Finally, chronic additions of relatively low levels of fertilizer to WS3 resulted in large increases, and continuous, changes in nutrient cycling. The effects on stream water NO_3 concentrations were observed relatively rapidly. Interestingly, chronic fertilization of WS3 produced stream water concentrations similar to those from maintaining WS6 and WS7 barren of vegetation.

Therefore, land managers who wish evaluate the potential for changes in nutrient cycling, which can lead to nutrient depletion and deficiencies due to their management activities should consider the extent of the disturbances to vegetation, and the nutrient uptake processes. Management actions which would result in significant disturbance of the vegetation and its ability to take up and cycle nutrients should be carefully considered, and may require careful planning to minimize the effects. Effects on nutrient cycling are of course not the only management objective that most land managers must consider. However, the results from the FEF show that disturbance need not always result in negative effects on nutrient cycling and water quality. Finally, we add the caveat that some of the effects that we have reported will vary depending on nutrient inputs, vegetation type, geology, soils, and forest health. As always, land managers must consider their particular set of circumstances when making management decisions.

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