Nutrient Dynamics

EFFECTS OF HARVESTING ON SOIL NITROGEN (N) DYNAMICS IN A N-SATURATED HARDWOOD FOREST

Frank S. Gilliam and Mary Beth Adams[1]

Abstract—Recent evidence suggests that soils of some central Appalachian hardwood forests have become nitrogen (N) saturated, a condition that develops when availability of soil N exceeds demand for N by plant roots and soil microbes. Among many environmental concerns associated with N saturation are the following: (1) greatly altered N cycle seen as a de-coupling of N dynamics from the high degree of biotic control that occurs in N-limited forest ecosystems and (2) greatly enhanced nitrification and leaching of NO_3^- along with calcium (Ca^{++}) and magnesium (Mg^{++}) from the soil and into streams, depleting base cation availability in the soil. Both of these have been demonstrated for hardwood forests of the Fernow Experimental Forest (FEF), West Virginia. Because increases in rates of nitrification and loss of soil cations often are associated with forest harvesting, it is important to determine whether the simultaneous effects of N saturation and harvesting may accelerate ecosystem degradation. Accordingly, we established a plot-based study to examine the effects of harvesting with and without addition of N and cations on soil N dynamics at FEF. Our treatments (each with four replicate 0.2-ha plots) were as follows: (1) Control (no treatment), (2) Harvest only (whole-tree harvesting [WTH] of entire plot), (3) Harvest+N (WTH plus 36 kg N/ha/yr), and (4) Harvest+N+cations (WTH plus 36 kg N/ha/yr plus 22.4 and 11.9 kg/ha/yr each of Ca and Mg, respectively). The cation addition treatment was done to simulate a mitigative treatment to replace loss of base cations to leaching. As expected at this N-saturated site, even our control plots exhibited high rates of N mineralization, mostly as net nitrification. However, these high rates were increased significantly by all harvest treatments. The addition of N in combination with harvesting did not increase nitrification significantly. By contrast, the addition of cations in combination with harvesting and N resulted in nitrification rates significantly higher than the other harvest treatments. These results illustrate the increasing complexities and challenges facing management of hardwood forests in the context of pollutant conditions.

INTRODUCTION

Deciduous forests of the eastern United States have exhibited an extremely high degree of resilience to a variety of historical perturbations and stresses. For example, although most of these forests originated after the heavy cutting and fires that occurred during the era of railroad logging (1880-1930), present forests have generally high species richness and productivity (Marquis and Johnson 1989, Gilliam and others 1995). While overall forest productivity remains reasonably high despite a growing demand for wood, the question of sustainable productivity, biodiversity, and health of forest ecosystems of the eastern U.S. is one of increasing concern. Indeed, recent data indicate that several eastern deciduous forest species are experiencing higher-than-normal rates of mortality (Twardus 1995). Despite on-going debate among forest ecologists, foresters, and resource managers, the observed mortality has been attributed to a variety of factors, alone or in combination. Some of these are acidic deposition, nutrient cation depletion, and excess nitrogen deposition, all of which have the potential to decrease survivorship of several forest species by weakening the trees and increasing their vulnerability to other stresses, including temperature and moisture extremes, disease, and insects (Adams and Eagar 1992, Eagar and Adams 1992).

One of the primary hypothesized agents of observed declines is base cation depletion of poorly buffered soils via forest harvesting and atmospheric inputs of nitrogen and sulfur (Eagar and Adams 1992, Gilliam and Richter 1991). Although typical central Appalachian soils are not considered infertile (Auchmoody 1972), recent concerns about sustainability of central Appalachian forests have been linked with soil fertility. Declines in nutrient cation levels in soil, particularly calcium (Ca) and magnesium (Mg), from intensive harvesting of forest products have been documented in some parts of the United States (Federer and others 1989, Mann and others 1988). Because timber harvesting is expected to increase in this region, and because the shift in forest utilization is toward fiber and more intensive harvesting, cation export in forest biomass could amount to a significant loss from some forest sites.

Earlier research on the effects of acidic deposition on forest systems indicated that the productivity of many soils in the humid east could decline significantly within 50-70 years (Binkley and others 1989), the span of a typical timber rotation for many stand types (Marquis and Johnson 1989). The central Appalachian region receives some of the highest inputs of acidic deposition in the United States (Adams and others 1993, Gilliam and Adams, 1996). The Clean Air Act contained provisions primarily targeting sulfur (S) emissions, not N emissions. Indeed, U.S. emissions of sulfur in 1990 were >25 percent less than those in 1970, while emissions of N were 2

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percent higher in 1990 compared to 1970 (World Resources Institute 1994). In short, pollutant emissions of oxides of N are not expected to decline greatly in the near future. Thus, considering the likelihood of several eastern deciduous forests becoming N saturated in the future and the increased demand on these forests for the production of wood products, information on interactions between N saturation and forest harvesting will be of great importance for forest ecologists, forest and resource managers, and policy makers. Research suggests strongly that some sites have reached or are nearing N saturation (Stoddard 1994, Gilliam and others 1996, Fenn and others 1998, Aber and others 1998). Furthermore, studies of stream chemistry at other eastern forest sites have shown that elevated N additions resulted in elevated losses of N (as NO_3) and base cations, especially calcium (Ca) and magnesium (Mg), suggesting a direct connection between N saturation (N additions) and the leaching of nutrient cations (Kahl and others 1993, Rustad and others 1993, Norton and others 1994).

The effects of forest harvesting on ecosystem nutrient cycling have been studied for many forest types and harvesting practices. Indeed, much has been learned to indicate that response of the biota (recovering vegetation) is largely responsible for regulating nutrient change as the forest recovers from the disturbance of the harvest regime (Binkley 1986, Tritton and others 1987, Martin 1995, Gilliam and Adams 1995, Johnson 1995). It also has been learned, however, that there is great inter-site variability in these responses, precluding broad generalizations across forest ecosystems. Despite their great ecological and economic importance, forests of the central Appalachian region have not been studied extensively with respect to nutrient responses to forest harvesting.

We know of no studies that have examined N saturation in the context of commercial forest harvesting. Accordingly, the purpose of this study was to examine the effects of harvesting on N mineralization and nitrification in a N-saturated forest. Harvesting was carried out alone and in combination with additional treatments to simulate different pollutant and mitigation scenarios. In addition, because mineralization and nitrification rates are often found to be directly related to soil temperature, the effect of harvesting on the temperature of surface soils was examined in a sub-set of sample plots. Finally, we will review current knowledge on N dynamics at the Fernow Experimental Forest as an example of a central Appalachian forest under conditions of N saturation.

METHODS

Study Site

This study was done on the Fernow Experimental Forest (FEF), Tucker County, West Virginia, a 1900-ha area of montane hardwood forests within the unglaciated Allegheny Plateau (39°03'N, 79°49'W). Prior to treatments, stands of all sample plots were typical of mature forests of the region (Gilliam and others 1995), dominated by sugar maple (*Acer saccharum* Marsh.) and northern red oak (*Quercus rubra* L.), with occasional large stems of black cherry (*Prunus serotina* Ehrh.) and yellow poplar (*Liriodendron tulipifera* L.). Soils of the study site within FEF have been described in great detail by Lusk (1998). In general, they are acidic (pH 4.1-4.7), shallow (30-40 cm to C horizon), and loamy in texture (Lusk 1998), similar to characteristics of soils of other watersheds of FEF (Gilliam and others 1994).

Field Design and Sampling

A single 4-ha mixed hardwood forest site was located on uniform soil at FEF. Slopes were gentle with a southeast aspect. Sixteen 0.2-ha plots, separated by 15 m buffer areas, were located and treatment assignments made prior to harvest. Twelve of the 16 plots were designated as harvest plots; four plots remained unharvested as controls. The treatments (each with four replicate plots) were as follows: (1) Control (C, no treatment), (2) Harvest only (H, whole-tree harvesting of entire plot), (3) Harvest+N (HN, harvesting as in 2 plus 36 kg N/ha/yr), and (4) Harvest+N+cations (HNC, as in 3 plus 22.4 and 11.9 kg/ha/yr each of Ca and Mg, respectively). The base cation treatment was carried out by adding dolomitic limestone by hand; rates were chosen to be twice the annual loss in stream flow from the reference watershed (WS4) at FEF (Adams and others 1997). All sampling and analyses were carried out during a one-year pre-treatment period and, for this paper, a one-year post-treatment period.

Available (extractable) N pools and rates of N mineralization and nitrification were measured with in situ incubations (buried bag technique of Eno 1960), based on the method described by Gilliam and others (1996). Following removal of forest floor material at a single sample point within each subplot, mineral soil was sampled by hand trowel to a 5-cm depth. Each sample was divided into two sub-samples and placed in polyethylene bags, one bag buried in the mineral soil at 5 cm for ~28 d and the other brought back to the lab. To minimize disturbance to the soil, samples were not sieved, but pebbles, rocks, and larger roots were excluded from sub-samples. All bags were refrigerated immediately in the field (using icepacks within backpacks) and then stored at 4 C prior to extraction. Sampling was carried out from approximately the middle of May to the middle of September for each of the 2 years.

Soil temperature was measured with HoboTemp (Onset Corporation) thermister temperature probes. Temperature was monitored continuously from 1 May to 30 November by placing probes at a 10-cm depth in four locations in each of two control and two harvest plots.

Laboratory and Data Analyses

Sub-samples of soil from paired sample bags were extracted for determination of net N mineralization and nitrification. Moist soils were extracted with 1N KCl at an extract:soil ratio of 10:1 (v:w). Extracts were analyzed colorimetrically for NH_4 and NO_3 with a Bran+Luebbe TrAAcs 2000 automatic analysis system. Net N mineralization was calculated as post-incubation (NH_4 and NO_3) minus pre-incubation (NH_4 and NO_3); net nitrification was calculated as post-incubation NO_3 minus pre-incubation NO_3.

To determine the relative amount of N mineralized that was eventually nitrified, N mineralization rates were compared to net nitrification rates using Pearson product-moment correlation and linear regression. Monthly rates of N mineralization and nitrification were averaged across all sample times in each year for each plot, four plots per treatment (i.e., N=4 for all means). Pre-treatment means were compared among treatments using ANOVA, whereas pre- versus post-treatment means were compared using t-tests for each treatment separately (Zar 1994).

RESULTS

Response of Soil Temperature to Harvesting

Seasonal patterns of soil temperature were similar among all four plots in which temperature was measured during the pre-treatment (pre-harvest) period (data not shown). Harvesting greatly increased maximum daily temperature of surface soil, especially from early June to mid October (fig. 1). Seasonal patterns of maximum soil temperature were highly similar between the two plots within each treatment type (harvest or control).

Response of Soil N to Harvesting and Additional Treatments

There was a highly significant correlation ($P<0.01$) between N mineralization and nitrification during the pre-treatment period ($r = 0.93$) (fig. 2). This was also highly significant for the post-treatment period ($r = 0.95$) (fig. 3).

Analysis of variance revealed no significant differences ($P<0.05$) among treatments during the pre-treatment period for net N mineralization (fig. 4). T-tests showed no differences in net N mineralization in pre- versus post-treatment periods for control (C) plots. However, net N mineralization was significantly higher ($P<0.05$) in post-treatment than pre-treatment plots for all harvest treatments, particularly for the HNC treatment.

The pattern for net nitrification was similar to that for mineralization. There were no significant differences ($P<0.05$) among treatments during the pre-treatment period, but there were significant differences between pre- and post-treatment means for most of the harvest treatments (fig. 5). Post-treatment nitrification rates were slightly (by about 25 percent), but significantly ($P<0.10$) higher in C plots. Net nitrification increased 2- to 2.5-fold from pre- to post-treatment periods on all harvest plots. This was significant ($P<0.10$) for all harvest treatments, except HN plots (fig. 5).

DISCUSSION

Although not always measured, soil temperature is assumed to increase following whole-tree harvesting. Because activity of nitrifiers is usually highly temperature-dependent,

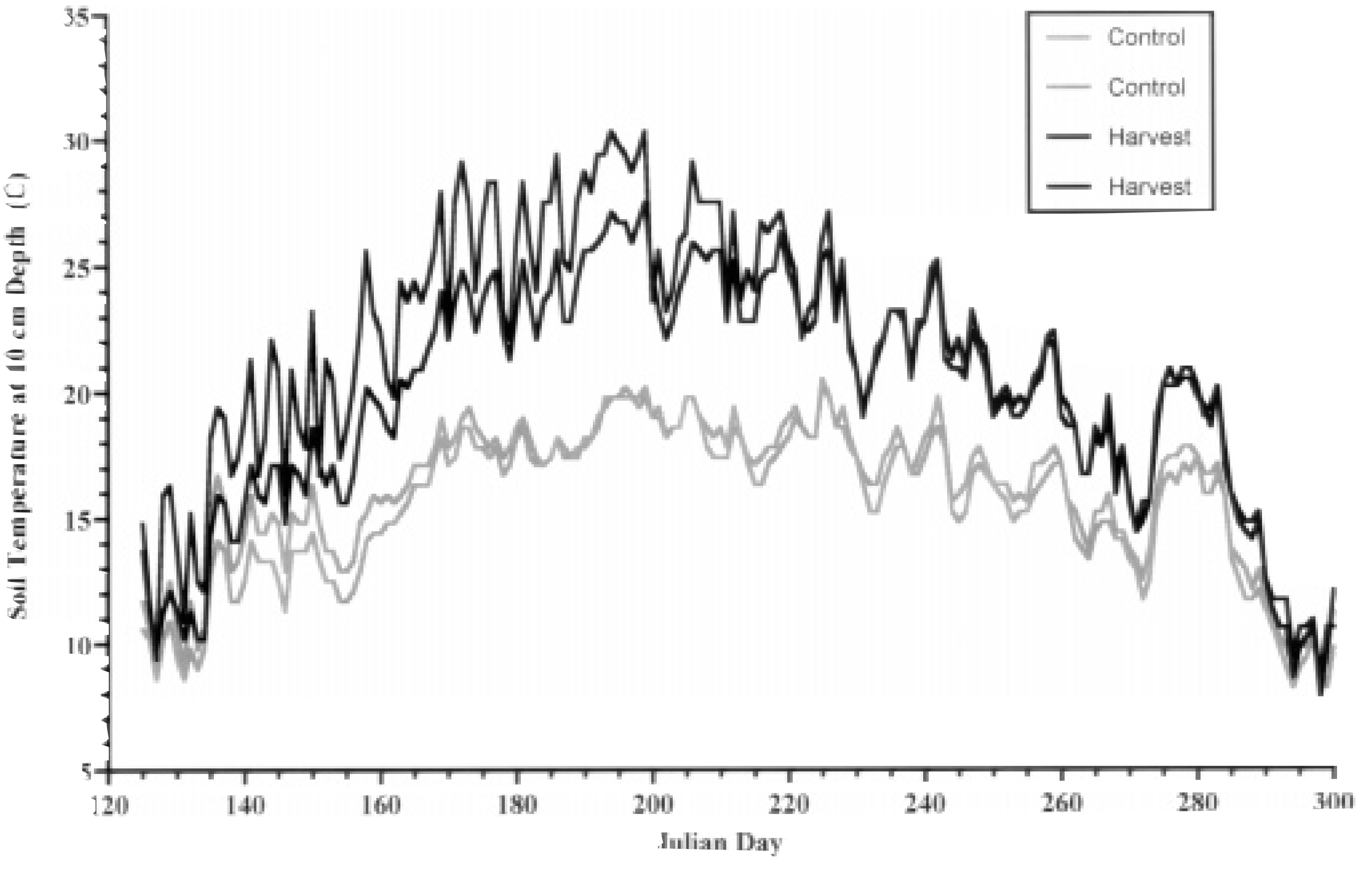

Figure 1—Soil temperature in two control and two harvest plots at Fernow Experimental Forest, WV. Data are daily maximum temperatures monitored continuously at a 10-cm depth.

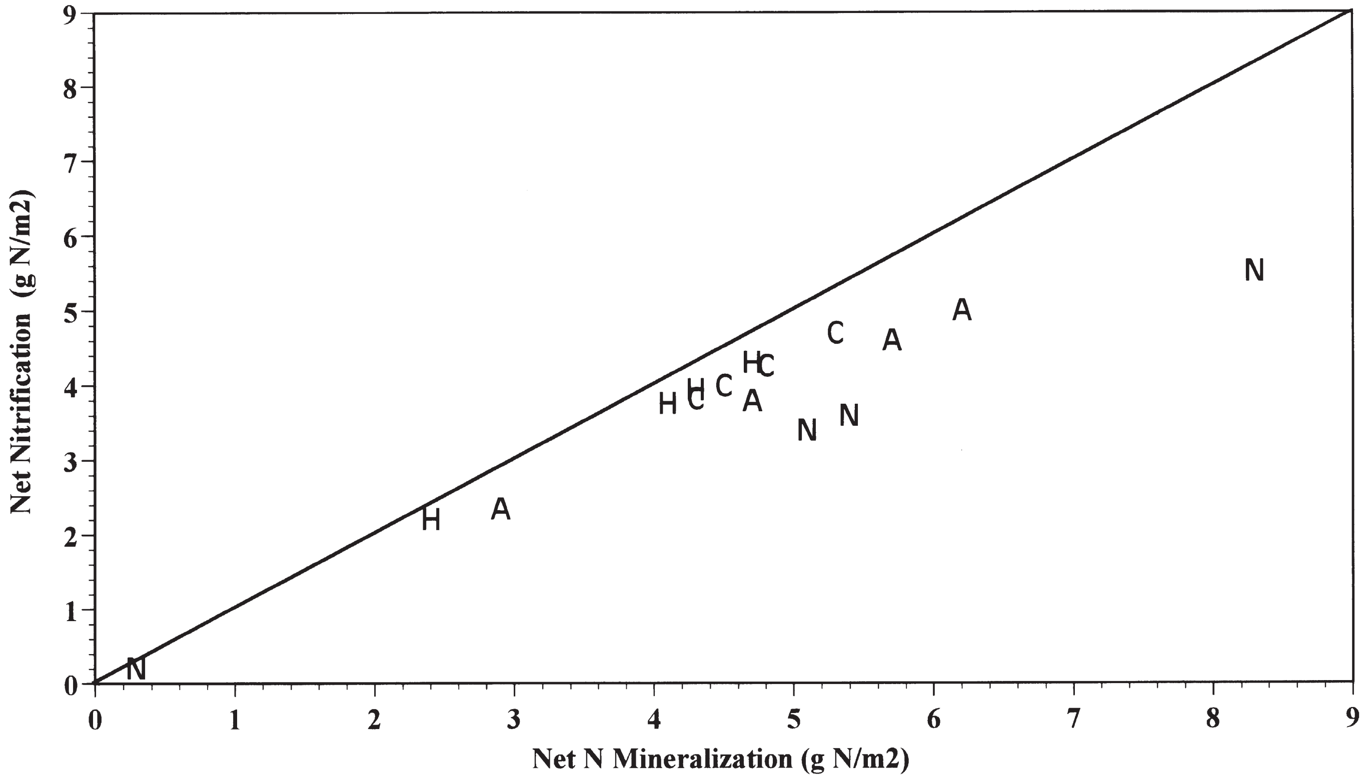

Figure 2—Nitrification versus N mineralization for individual plots during the pre-treatment period. Data are mean values for each of four plots for four treatments as follows: C = control; H = harvest; N = harvest + N; A = harvest + N + cations (i.e., A = all). Correlation was significant at P<0.01, r = 0.93. Line given is a 1:1 line representing a condition wherein N mineralization is 100 percent nitrification.

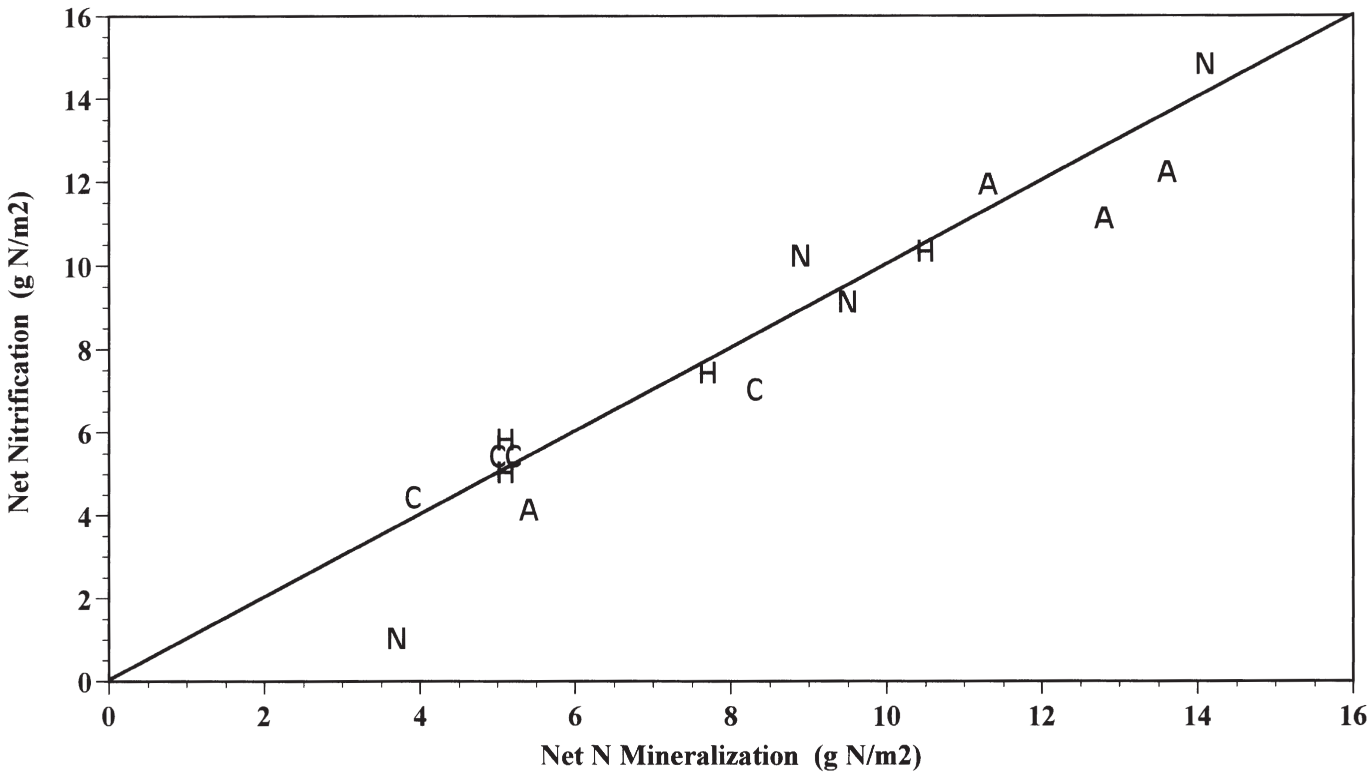

Figure 3—Nitrification versus N mineralization for individual plots during the post-treatment period. Data are mean values for each of four plots for four treatments; see fig. 2 for meaning of symbols. Correlation was significant at P<0.01, r = 0.95. Line given is a 1:1 line representing a condition wherein N mineralization is 100 percent nitrification.

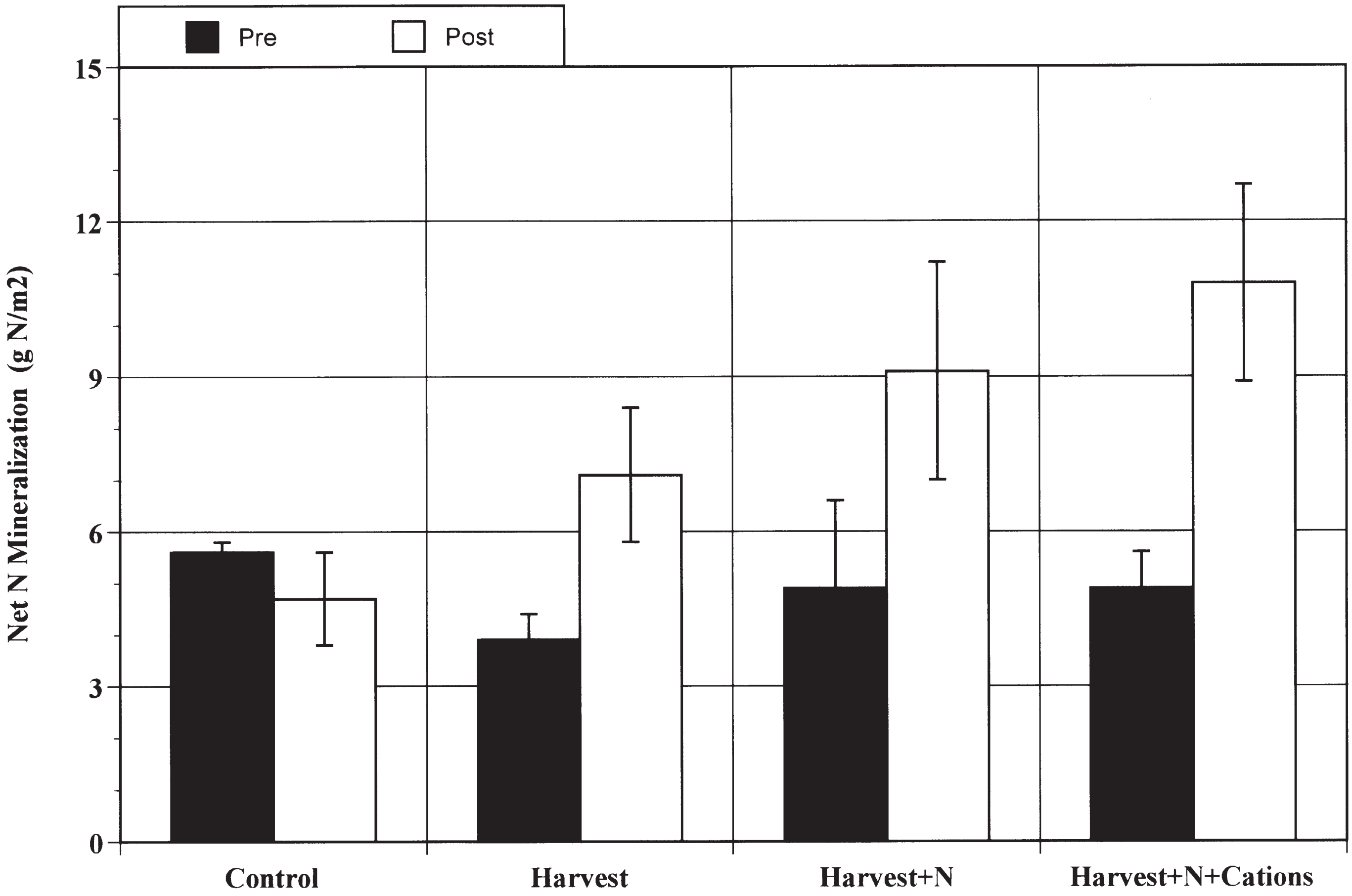

Figure 4—Net N mineralization for pre- and post-treatment periods for control and harvest treatment plots at Fernow Experimental Forest, WV. Data are means plus and minus one standard error of the mean. Analysis of variance revealed no significant differences among pre-treatment means. Pre- versus post-treatment differences were significant ($P<0.05$) for all harvest plots, but not for control plots.

however, it was important in this study to quantify harvest-mediated changes in soil temperature. Our data confirm that post-harvest increases in soil temperature can be prolonged and substantial (fig. 1). These increases likely arose from changes in the relationship between soil heat flux (S) and net radiation (Rn)(Rosenberg and others 1983). Idso and others (1975) showed that S (increases of which lead to increases in soil temperature) is highly positively correlated to Rn under a variety of soil conditions. The removal of the tree canopy during harvesting decreases interception of solar radiation and increases Rn, which increases S and results in the increased soil temperatures seen in fig. 1. These patterns have important implications for N dynamics and their responses to harvesting.

Not only was net nitrification highly correlated to net N mineralization, nitrification was also an extremely high proportion of N mineralization in both the pre- and post-treatment periods, regardless of treatment (figs. 2 and 3). Linear regressions of each of the sample periods closely approximated the theoretical 1:1 line that indicates a condition wherein nitrification is 100 percent of mineralization. These high rates of nitrification (relative to mineralization) are indicative of N saturation (Aber 1992), supporting findings and conclusions of Gilliam and others (1996) for three watersheds of FEF. This was especially the case during the post-treatment period for the harvest plots, suggesting that harvesting may enhance the relative predominance of nitrification in the N dynamics of N-saturated soils.

Patterns of soil temperature (fig. 1) combined with the relationships shown in figs. 2 and 3 are helpful in interpreting pre- versus post-treatment comparisons for N mineralization and nitrification rates (figs. 4 and 5). Koopmans and others (1995) have shown that increases in temperature greatly enhance N mineralization in N-saturated soils in Europe. Thus, the substantial increases in soil temperature caused by harvesting in our study, especially from early June to around mid September, caused significant increases in N mineralization in the soil. Furthermore, virtually all this enhancement was related to increases in nitrification, i.e., temperature-mediated increases in the activity of nitrifying bacteria.

Data from our study show also that there may be further increases in N mineralization/nitrification ratios under simulated conditions of additional N inputs and additions of

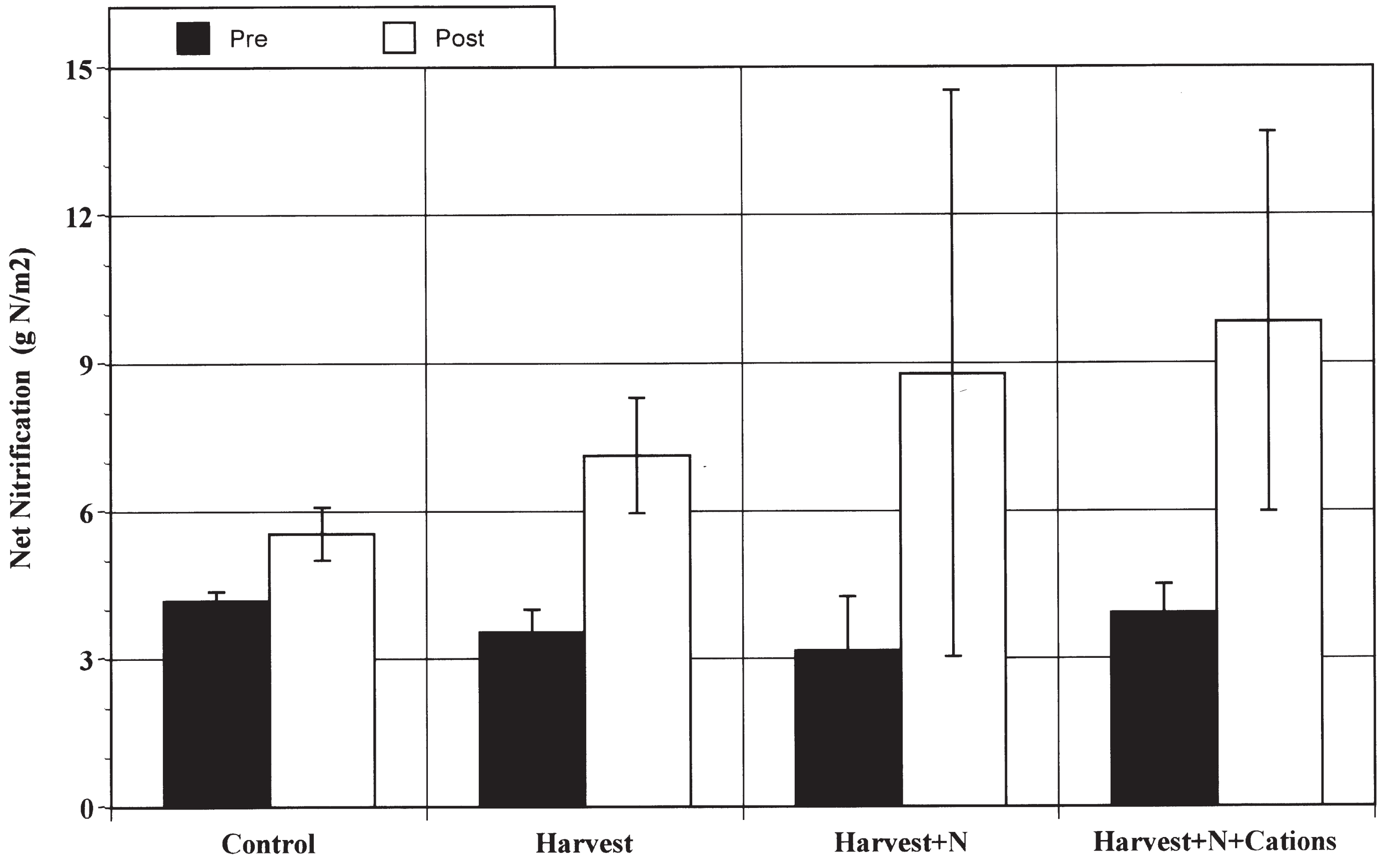

Figure 5—Net nitrification for pre- and post-treatment periods for control and harvest treatment plots at Fernow Experimental Forest, WV. Data are means plus and minus one standard error of the mean. Analysis of variance revealed no significant differences among pre-treatment means. Pre- versus post-treatment differences were significant for control plots at $P<0.10$ and for harvest and harvest+N+cations at $P<0.05$; difference for harvest+N plots was not statistically significant.

base cation designed to mitigate cation loss. Although differences were not significant among harvest treatments, there was a pattern of increases in N mineralization and nitrification related to N additions, and even greater increases with additions of base cations. This latter pattern suggests that mineralization and nitrification might be more pH limited than previously thought (Gilliam and others 1994, 1996). Future work at our site will test this hypothesis. Certainly, it suggests that managing central hardwood forests in a sustainable manner in the context of N saturation represents a complex challenge.

Implications for N Saturation in Central Hardwood Forests

Existing data from FEF strongly suggest a threat of base cation depletion and its relationship to N saturation in central Appalachian forests (Adams and others 1997, Gilliam and others 1996). Indeed, much evidence exists to suggest that soils at FEF already have become N saturated. For example, Gilliam and others (1996) found equally high rates of N mineralization based on one year's data from one N-treated (at rates identical to those in this study) and two untreated watersheds. Furthermore, nearly 100 percent of N mineralized was nitrified. On-going data from this study suggest a pattern of increasing rates through time for all three watersheds, regardless of treatment. Furthermore, available pools of N (extractable NH_4 and NO_3) were significantly higher on the treatment watersheds, suggesting that further additions of N to the treatment watershed are no longer resulting in increased uptake by plants or immobilization by microbes (Vitousek and Matson 1985).

Stoddard (1994) concluded that Watershed 4 (WS4—mature, second-growth forest) at FEF was one of the better examples of watersheds in the eastern United States which are at the later stages of N saturation. It is notable that WS4 supports a mature, mixed-age, second-growth hardwood stand and serves as the long-term control watershed at FEF, receiving no experimental treatments and, thus, it may be considered typical of many second-growth forests of the region (Gilliam and others 1996.

Using long-term (1971-1987) stream chemistry data from WS4 at FEF Edwards and Helvey (1991) found patterns of significant increases in specific conductivity with time. Using correlation analysis they concluded that most of this increase was related to increases in stream NO_3. Based on data from WS4 for an even longer time period (1969-1991), Peterjohn and others (1996) concluded that the patterns found by Edwards and Helvey (1991) were part of a larger scenario indicating N saturation for FEF.

The relative sensitivity of forest soils to accelerated cation leaching, pH depression, aluminum mobilization, and mineral weathering is largely unknown (Cronan and others 1990). Forest soils of the central Appalachians range from very resistant to very sensitive to such changes. Based on criteria established by Turner and others (1986), sensitive soils are those with combinations of low cation exchange capacity (CEC), intermediate base saturation, low pH, low levels of bases released via weathering, low sulfate adsorption capacities, and shallow soils. Given that a large percentage of soils in the Appalachian region falls within these criteria, data from this study strongly suggest that expected increases in timber harvesting, continued high levels of N deposition in the region, base cation depletion, and complications from N saturation all represent major threats to sustainability of central Appalachian forest ecosystems.

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N DYNAMICS ACROSS A CHRONOSEQUENCE OF UPLAND OAK-HICKORY FORESTS

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Changes in soil physical, chemical, and biological properties due to harvesting practices may have a significant impact upon N availability and cycling, thus future stand productivity and composition. Our study was designed to assess monthly rates of N min, nitrification, and N uptake by forest vegetation during the growing season. We selected five oak-hickory stands in Dubois County, Indiana for this study. We installed and collected PVC cores on a monthly basis from April to December. Soils were then analyzed for ammonium and nitrate.

Results show that seasonal N cycling rates are variable in all stands (figure 1). On an annual basis, the mature stand showed the highest rates of N min and N uptake and the lowest percentage of mineralized N converted to nitrate (29 percent) (table 1). The youngest stand had the lowest N min rate and the highest percentage of mineralized N converted to nitrate (77 percent). It was the only stand in which N min rates were appreciably lower than N uptake rates. The soil core method underestimates true N min rates within this stand because N min from logging slash is not reflected in the cores. For the other stands, there appears to be a tight internal cycling of N, preventing leaching losses.

Table 1—N cycling in a chronosequence of hardwood forests

Stand age	N mineralization	Nitrification	N uptake
	-----------------*g N/hectare/day*-----------------		
1	87.46	60.45	154.98
6	182.95	64.08	196.20
12	161.46	56.37	212.81
31	186.18	93.96	246.34
80-100	392.01	115.57	357.38

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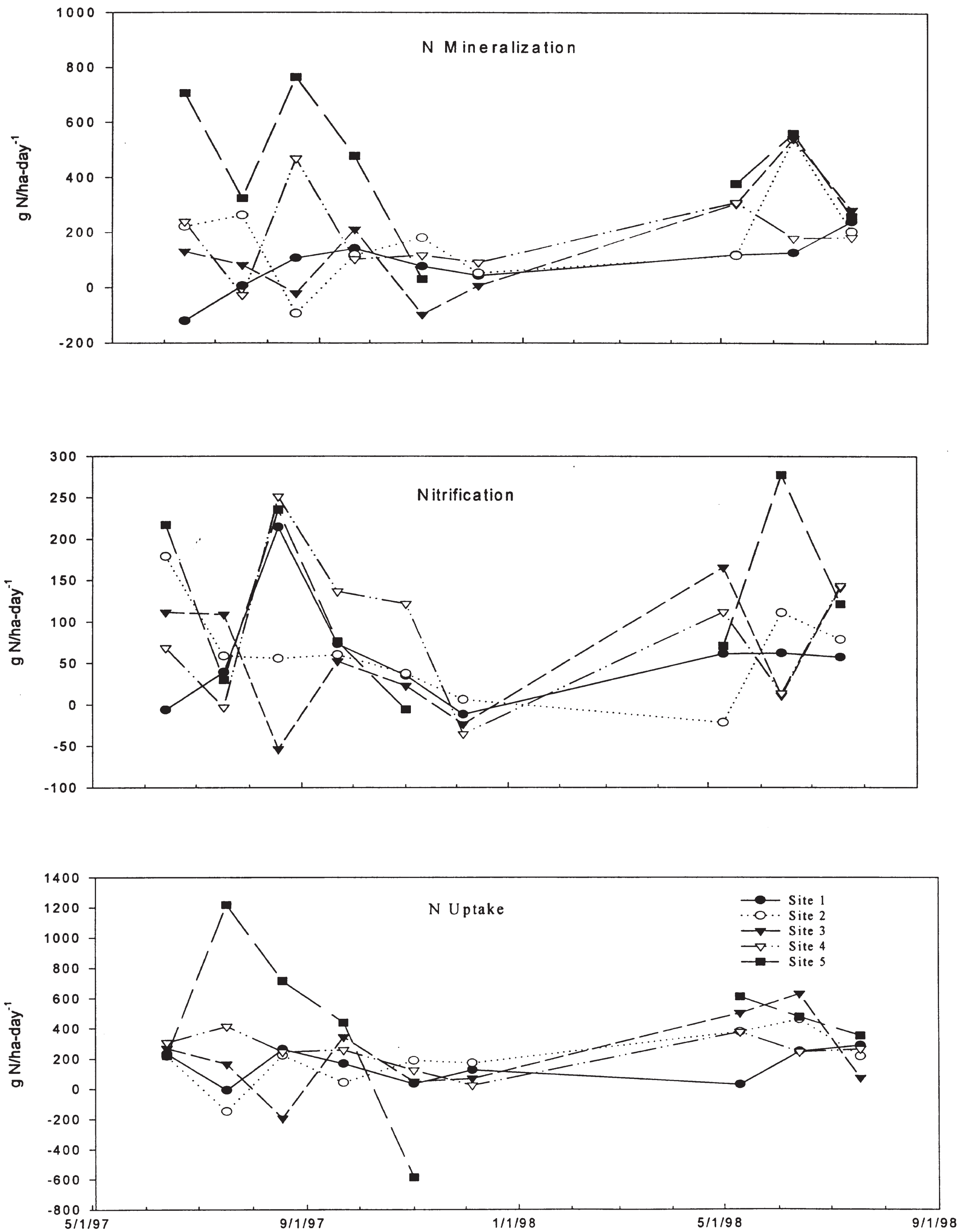

Figure 1. Nitrogen cycling across a chronosequence of hardwood forests. Stands cut in 1996, 1991, 1985, 1966 and 1900-1920, respectively.

SOIL NUTRIENT AND MICROBIAL RESPONSE TO PRESCRIBED FIRE IN AN OAK-PINE ECOSYSTEM IN EASTERN KENTUCKY

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Abstract—On the Cumberland Plateau, periodic fire may be necessary to maintain oak-dominated stands on xeric ridgetops. However, thin organic horizons and nutrient poor soils may limit the beneficial effects of fire by further reducing nutrient pools. The USDA Forest Service has reintroduced fire to oak-pine ridgetops in the Daniel Boone National Forest in the form of single, late winter prescribed burns. Objectives of this study were to document effects of single prescribed fires conducted in March 1995 and March 1996 on soil nutrients and microbial biomass. In 1996, forest floor mass was determined before and after fire. In 1995 and 1996, total and available nitrogen, total carbon, pH, extractable cations, and microbial biomass were measured pre-burn and throughout the year after burning on burned and unburned sites. Thirty-two percent of the litter layer (Oi) was combusted in 1996, while no loss of the Oe and Oa horizons was found. A transitory increase in available nitrogen was detected in burned mineral soils. Increases in pH by 0.2-0.3 units were measured in the burned organic horizons. Lower concentrations of extractable cations were measured in burned soils than in unburned soils. Fire had a positive effect on active bacterial biomass, but no effect on fungal biomass. Our study suggests that single, late winter prescribed fire had minimal effects on belowground resources in these ridgetop ecosystems. Since repeated burning might be necessary to promote oak regeneration, future research must address the effects of repeated burning on soil resources.

INTRODUCTION

Fire has played an integral role in maintaining oak-pine forest communities in the southern Appalachians for as long as 3000 years BP (Delcourt and Delcourt 1997). Aboriginal use of fire was common and settlers of European descent continued fire use in the area; however, as forest use altered, fire regimes changed (Pyne 1982, Pyne and others 1996). This resulted in shifting forest composition, most noticeably as an increasing dominance of red maple (*Acer rubrum* L.) and decreasing regeneration of oak species (*Quercus* sp.) (Arends and McCormick 1987, Lorimer 1993). Ecologists and managers are increasingly promoting the use of fire to address problems of oak decline and flourishing fire-sensitive competitors like red maple.

In the Daniel Boone National Forest (DBNF) in eastern Kentucky, active fire suppression since the 1940s has successfully excluded fire on most oak-pine ridgetops (Martin 1989). Increasing regeneration of white pine (*Pinus strobus* L.) has been documented and ascribed to fire suppression (Wehner 1991), while charcoal and pollen analyses have demonstrated a growing presence of red maple and blackgum (*Nyssa sylvatica* Marshall) over the past 100 years (Delcourt and Delcourt 1997). Because of such changes in forest composition, the USDA Forest Service on the Stanton Ranger District has begun conducting late winter prescribed fires to restore fire to these ridgetop ecosystems (Richardson 1995). We have documented the effectiveness of late winter prescribed fire in reducing competition by fire-sensitive competitors (Arthur and others 1998, Blankenship [In Press]), and studies from other areas have recommended prescribed fire to improve oak regeneration (Barnes and Van Lear 1998, Brose and Van Lear 1998, Nyland and others 1983, Thor and Nichols 1974, Van Lear and Waldrop 1989).

Complicating the use of prescribed fire in eastern Kentucky is the problem of accidental and incendiary fires burning large acreage during the spring and fall seasons each year (Environmental Quality Commission 1992). Consequently, the use of prescribed fire on public lands is perceived by some to send a misleading message about fire to the general public. Because of thin organic soil horizons and low soil nutrient concentrations on ridgetops in the Red River Gorge, fire potentially could, if too hot, contribute to a loss of nutrients already in low supply. The effects of fire on soil microbial biomass may also be important because of the role fungi and bacteria play in mediating nutrient mineralization and availability. An improved understanding of the soil nutrient and biological response, as well as the plant community response, to prescribed fire is necessary to elucidate the ecological differences between the effects of prescribed fire and the potentially hotter and more frequent incendiary fires. This type of information is essential to the public debate regarding the role of prescribed fire in the management of public forestlands.

The objectives of this study were to document effects of a single, late-winter prescribed fire on soil nutrients and microbial biomass as a first step in addressing impacts of different fire regimes in these ridgetop ecosystems. Despite the plethora of studies regarding effects of fire on soil nutrients and microorganisms in forests throughout the United States and elsewhere, few studies have examined effects of burning on soil resources in oak-pine forests in the southern Appalachians (Clinton and others 1996, Vose and others [In Press], Vose and Swank 1993) and none on the Cumberland Plateau in eastern Kentucky.

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METHODS

Site Description

Permanent study plots are located on three noncontiguous ridgetops in the Stanton Ranger District of the DBNF in the Red River Gorge Geological Area. The Red River Gorge is located in the Cliff Section of the Cumberland Plateau (Braun 1950). The geological substrate is composed of shales and siltstones of the Upper and Lower members of the Breathitt Formation and the Corbin Sandstone of the Lee Formation (Weir and Richards 1974). The ridgetops, Pinch-Em-Tight Ridge, Whittleton Ridge and Klaber Ridge, are located in Powell, Wolfe and Menifee counties of eastern Kentucky. Study plots are located in stands mostly dominated by scarlet oak (*Q. coccinea* Muenchh.) and chestnut oak (*Q. prinus* L.), with some white (*Q. alba* L.) and black oak (*Q. velutina* Lam.). Pines (*P. rigida* Mill., *P. virginiana* Mill., and *P. echinata* Mill.) dominate the most xeric areas. Red maple is abundant in mid- to overstory positions. Sourwood (*Oxydendrum arboreum* (L.)DC), blackgum and dogwood (*Cornus florida* L.) are found in the understory. Eastern white pine seedlings and saplings are plentiful and a few large white pines occur in the overstory. Heath shrubs such as mountain laurel (*Kalmia latifolia* L.) and blueberry (*Vaccinium* spp.) are common throughout.

Soils at Whittleton Ridge are composed of Gilpin silt loam, a moderately deep, well drained soil with a lower subsoil of silty clay loam of the subgroup Typic Hapludult (Hayes 1993). Soils at Klaber Ridge are similar and classified as Latham-Shelocta silt loam, moderately deep, moderately well drained, slowly permeable clayey soils and of the subgroups Typic and Aquic Hapludults (Avers and others 1974). Soils at Pinch-Em-Tight are of Alticrest-Ramsey-Rock outcrop complex, moderately deep and shallow, well drained with surface layer and subsoil of sandy loam, and are Typic and Lithic Dystrochrepts (Hayes 1993, SCS 1975). The region has a temperate, humid and continental climate with an evenly distributed average annual precipitation of about 113 centimeters. Mean daily maximum and minimum temperatures in January are 6°C and –6°C, and in July are 30°C and 17°C. Mean annual temperature is 12°C (Hill 1976).

Experimental Design

Each ridge (Klaber, Pinch-Em-Tight and Whittleton) was divided into three areas: no burning, burned in March 1995, and burned in March 1996. This yielded two treatments (burned and unburned) per year of study (1995 and 1996). Study plots were located at random, with the condition that any location within 15 meters of the slope break, a trail, or any human disturbance was rejected. We installed eight 0.04 hectare plots per treatment except at Pinch-Em-Tight Ridge. That site is topographically narrow and traversed with trails and campsites, and we could fit only 6 plots per treatment. The 1995 burn treatment on Pinch-Em-Tight was especially narrow and these plots were laid out systematically in a line, 30 meters apart along the top of the ridge. Centers of all plots were permanently marked, and two trees in each plot were tagged and located (azimuth and distance to plot center) for future reference. While plots were located at random within each treatment area, treatments were applied nonrandomly to treatment areas because of the necessity of burning a contiguous finger ridge not in contact with privately owned land.

Fire Prescription

USDA Forest Service personnel of the Stanton Ranger District conducted the 1995 and 1996 prescribed fires. The fires were ignited with drip torch by firing line from the highest point and ridges first. From the ridges, strips were pulled downslope into the wind. If backing and flanking fires were not of sufficient intensity (>0.3 m flamelength), point-source and strip firing were used to increase intensity to acceptable levels (Richardson 1995). Wind speeds during the 1995 fires were from 0 to 3.2 km/h, and in 1996 were around 1.6-4.8 km/h. Flame heights for all burns were 0.3-0.9 meters (Richardson 1995, Richardson 1996). In 1996, the prescribed burning planned for Whittleton Ridge could not be conducted due to unsuitable wind direction that would have allowed smoke to blow over an adjacent highway.

Temperatures of prescribed fires were measured using 6 different Tempilac® temperature-sensitive paints chosen to represent a temperature range of 93°C to 576°C (Cole and others 1992). A stripe of each paint was applied on aluminum tags and vertically down strips of mica. The aluminum tags were stapled to stakes at heights of 15, 45 and 75 centimeters. The stakes and the mica sheets were placed at 4 meters from each plot center on north, south, east, and west bearings. The mica sheets were inserted into the ground to a depth of 9 centimeters below the litter surface.

Forest Floor Sampling

Forest floor was collected on all sites in February 1996 prior to burning, and in May 1996 following prescribed burning. One 27.5 x 27.5 centimeter sample was collected along three randomly selected azimuths in each plot (3 samples per plot, mean of the three used for analysis). Litter (Oi) was separated from the Oe and Oa horizons in the field and measured separately.

Soil Sampling

Soil samples for nutrient and microbial biomass analyses were taken within a 0.04 hectare plot by sampling along two randomly selected 11 meter radii originating at the center point of each permanently marked plot. Soil samples were stored at 4°C until processed. Nutrients (C, Ca, K, Mg, N, and P) were analyzed on soil samples from a composite of two 4.0 centimeter cores taken at 6.0 meters along each radius. The organic horizon (Oea) was separated from the mineral horizon in the field with mineral soils sampled to a depth of 5 centimeters. Samples for nutrient analyses were taken within 30 days pre-burn, after the first rainfall post-burn (1 week post-burn in 1995 and 2 weeks post-burn 1996), and in September.

For microbial biomass, two soil cores of 2.0 centimeter diameter were taken at 1.5 meter intervals along each radius beginning at 3.0 meters from the center for a total of 24 cores collected at 12 sampling points. Mineral soils were sampled to a depth of 5 centimeters. Organic and mineral horizons were separated and soil cores composited into a single sample for analysis from each plot. Microbial biomass samples were collected more frequently than nutrient samples because of the expectation of an immediate (1 day) response to fire as well as an attenuation of the effect with time. In 1995, samples for microbial biomass were collected within 30 days pre-burn, the day after the burn, 1 week post-burn (followed first rainfall post-burn), and every 4 weeks until July, then every 6 weeks thereafter through November. Sampling frequency was reduced after July 1995 following examination of results from earlier in 1995. In 1996, after sampling 2 weeks post-burn (following first rainfall post-burn), microbial biomass samples were collected post-burn only every 6 weeks through September 1996.

Available nitrogen (ammonium- and nitrate-nitrogen) was measured on mineral soil samples collected for microbial biomass estimates. In 1995 no pre-burn measurements of available nitrogen were made; both pre- and post-burn measurements of available nitrogen were made in 1996.

Laboratory Analysis

Forest floor samples were oven-dried at 60°C and then weighed. Mehlich III-extractable P, Ca, Mg, and K were determined with the Mehlich III reagent method (Tran and Simard 1993), and analysis by an inductive coupled plasma spectrophotometer. Organic horizon pH was measured in a 1:10 soil:$CaCl_2$ mixture, and a subset of these measured in a 1:2 soil:water mixture. Statistical analysis was conducted on the organic pH values from the $CaCl_2$ method. The mineral soil pH was measured in a 1:2 soil:water mixture (Hendershot and Lalande 1993). Total C and N were analyzed using a Leco C/N analyzer. Determination of ammonium- and nitrate-nitrogen in the mineral soil was made using a Technicon II auto-analyzer following KCl extraction (Maynard and Kalra 1993).

Active and total fungal and bacterial biomass were calculated from direct counts of hyphal length (fungi) and number of organisms (bacteria) using direct count epifluorescent microscopy (Ingham and Klein 1984, Ingham and others 1991, Lodge and Ingham 1991). Fluorescein diacetate was used to stain metabolically active fungi and bacteria; total fungal hyphae (metabolically active and inactive) were counted using direct light source. Direct counts of fungi and bacteria were carried out within one week of sample collection. Based upon the assumption that hyphal tissue density averages 410 mg cm^{-3} and bacterial tissue density averages 330 mg cm^{-3} (Ingham and others 1991), and using estimates of hyphal and bacterial diameters, active fungal and bacterial biomass were calculated from the volume of fungi or bacteria in 1 gram of dry forest floor.

Statistical Analysis

We analyzed the data in two ways: (1) by comparing pre-burn to post-burn within each treatment (burned or unburned) using a t-test, and (2) by comparing the burned treatment to the unburned by analysis of variance. A covariate analysis using pre-burn parameters as the covariate was used for the nutrient data to compare burned to unburned. Statistical analysis of pH was performed on $[H^+]$ and results were converted to pH values for presentation. On microbial biomass and available nitrogen, Levene's (1960) homogeneity of variance test revealed that often group variances within a treatment were significantly different, so the Welch ANOVA for means was used (Brown and Forsyth 1974, Welch 1951).

Initially, the three ridgetops were selected as replicates for treatment for each year with treatment area as the experimental unit (n=6; 3 burned and 3 not burned), but in 1996 one ridgetop was not burned as planned, so analysis of the 1996 data had only 2 replicates (n=4; 2 burned, 2 not burned). As the study progressed we became aware of inherent site differences. Therefore, statistical analyses were also conducted on a pseudo-replicated design (Hurlburt 1984) using the 8 (or 6) plots per treatment area as the experimental unit with ridge as block and significance determined at $p<0.10$ (Table 1). If these analyses revealed significant site-by-treatment interactions in addition to significant treatment effects, subsequent analyses were conducted on reduced models comparing the burned treatment to the unburned reference at each ridgetop location (Klaber and Whittleton: n=16; Pinch-Em-Tight: n=12). All statistical

Table 1—Experimental design for study of prescribed burns conducted in March 1995 and March 1996 in the Daniel Boone National Forest, Kentucky

1995			1996		
Ridge/block[a]	Plots	Treatments	Ridge/block	Plots	Treatments
Klaber Ridge	8	Burned/unburned	Klaber Ridge	8	Burned/unburned
Pinch-Em-Tight Ridge	6	Burned/unburned	Pinch-Em-Tight Ridge	6	Burned/unburned
Whittleton Ridge	8	Burned/unburned			
Total: 3	22	2	2	14	2

[a] Experimental unit was the ridge (n=3 in 1995, n=2 in 1996) or the plot with ridge as block (n=22 in 1995, n=14 in 1996).

tests were conducted with the JMP® statistical package (SAS Institute 1995).

RESULTS

Pre-Burn Soil Chemistry

Thin, acidic, nutrient-poor organic horizons characterized these ridges prior to burning. Organic horizons (Oea) ranged from <1 to 4-5 centimeters in thickness with a mean pH of 3.70. Total nitrogen averaged 0.66 percent and total carbon 54 percent. Mineral soil pH was slightly higher at 4.24, with total nitrogen averaging 0.14 percent and carbon 4.9 percent.

Fire Temperatures

Surface temperatures ranged from 316°C to 398°C in 1995 and from 204°C to 315°C in 1996. Temperatures within the Oea horizons at 0.5 centimeters below the surface ranged from 204-315°C in 1995 and from 93-203°C in 1996.

Forest Floor Mass

Pre-burn forest floor mass averaged 98.9 grams/meter^2 for the Oi layer and 184 grams/meter^2 for Oea layers. The prescribed fires of 1996 combusted an average of 32 percent of the litter layer (Oi) without combusting the underlying Oe and Oa layers. The decrease in Oi because of the fires was highly significant ($p<0.01$). Forest floor mass measurements were not made in 1995.

Soil Nitrogen

The inorganic nitrogen in the mineral soils primarily took the form of NH_4-N. Little, if any, NO_3-N was measured. In 1995, available nitrogen concentration was significantly higher in the burned treatments than in the unburned treatments 1 day after the burns on Klaber Ridge (12.5 vs. 1.89 µg/g dry wt soil; $p<0.05$) and Whittleton Ridge (8.43 vs. 4.89 µg/g dry wt soil; $p<0.10$), but not on Pinch-Em-Tight Ridge (Table 2). One week later there were no significant differences.

In 1996, available nitrogen concentration declined from the pre-burn sampling to the post-burn sampling dates, significantly so in the unburned areas (3.23 to 1.25 µg/g dry wt soil; $p<0.0001$), but not significantly in the burned treatments based on the pre- vs. post-burn t-tests. Two weeks later, available nitrogen concentrations were significantly lower in the burned treatments (1.28 µg/g dry wt soil) than in the unburned (2.43 µg/g dry wt soil; $p<0.10$) using covariate analysis. In September 1996, total nitrogen in the mineral horizon was significantly lower in the burned treatments (0.09 pct. vs. 0.15 pct; $p<0.01$). No consistent effect of fire on total carbon was found.

pH

O horizon pH was significantly higher in the 1995 burned treatments one week post-burn ($p<0.05$) and in September 1995 ($p<0.10$), but there were significant site by treatment interactions ($p<0.05$). Analysis on the reduced models

Table 2—Mean available N (NH_4-N + NO_3-N; µg/g dry wt soil) in mineral horizon of burned and unburned treatments on Klaber, Whittleton, and Pinch-Em-Tight Ridges, Daniel Boone National Forest, Kentucky, following 1995 prescribed fires and prior to and following 1996 prescribed fires

	Klaber		Whittleton		Pinch-Em-Tight	
Date	Burned	Unburned	Burned	Unburned	Burned	Unburned
1 day post-burn 1995	12.5[a] (3.1)[d]	1.89 (0.66)	8.43[a] (1.7)	4.89 (0.92)	3.59 (0.99)	4.17 (2.0)
1 wk post-burn 1995	3.72 (0.44)	3.37 (0.22)	4.90 (0.39)	4.86 (0.14)	4.14 (0.57)	3.46 (0.13)
Pre-burn 1996	1.95 (0.38)	2.37 (0.14)			2.79 (0.16)	4.10 (0.43)
1 day post-burn 1996[b]	1.60 (0.62)	1.24 (0.23)			2.21 (0.11)	1.26 (0.13)
2 wk post-burn 1996	1.49[c] (0.57)	3.01 (0.48)			1.40c (0.34)	1.64 (0.33)

[a] Burned treatment significantly different from unburned treatment of same ridge at $p<0.10$.
[b] Pre-burn vs. post-burn t-tests showed burned treatments not significantly different but unburned treatments significantly different at $p<0.0001$.
[c] Burned treatments significantly different from unburned treatments by covariate analysis at $p<0.10$.
[d] Standard errors (in parentheses) are based upon mean within a treatment.

revealed that burned treatments had significantly higher pH 1 week post-burn at Whittleton Ridge (3.64 vs. 3.18; p<0.001) and in September at Klaber Ridge (3.64 vs. 3.23; p<0.01) (Table 3). The 1995 fires did not affect pH in the upper mineral soils.

The 1996 fires significantly increased organic horizon pH from pre-burn values to 2 weeks post-burn (3.01 to 3.24; p<0.10), even though the pH of the burned treatments did not significantly differ from the unburned treatments 2 weeks post-burn (p=0.79). In September 1996, we found significant differences in pH between burned and unburned treatments in both the organic and mineral horizons with significant site by treatment interactions. The reduced model analyses showed significantly lower pH in Klaber burned organic horizon (3.20 vs. 3.53; p<0.01), while Pinch-Em-Tight burned treatments had significantly higher pH in organic (3.17 vs. 3.08; p<0.001) and mineral horizons (4.29 vs. 3.89; p<0.05).

Cations

Burned treatments contained significantly lower extractable cation concentrations than unburned treatments later in the season. Extractable potassium concentrations were significantly lower in burned organic horizons in September 1995 (362 vs. 439 µg/g dry wt soil; p<0.10) and September 1996 (362 vs. 490 µg/g dry wt soil; p<0.10) (Table 4). In September 1996, magnesium and calcium concentrations were also lower in burned organic horizons (164 vs. 246 µg/g dry wt soil; p<0.01; 1240 vs. 2500 µg/g dry wt soil; p<0.01, respectively).

Extractable cations in the burned upper mineral soil increased or were higher soon after the fires, but by September tended to be significantly lower than in unburned soils. In 1995, significantly higher potassium concentrations were found in the burned treatments one week post-burn (86.7 vs. 84.0 µg/g dry wt soil; p<0.05). In the 1996 burned treatments, significant increases of magnesium (19.4 to 25.2 µg/g dry wt soil; p<0.05) and calcium (100 to 135 µg/g dry wt soil; p<0.05) concentrations from pre-burn to 2 weeks post-burn were found. In September 1995, potassium was significantly lower in burned treatments (62.4 vs. 71.7 µg/g dry wt soil; p<0.10), and in September 1996 lower in the burned treatment of Klaber Ridge (64.6 vs. 89.6 µg/g dry wt soil; p<0.10). In September 1995 and 1996, burned treatments had significantly less magnesium than unburned treatments (1995: 21.6 vs. 25.0 µg/g dry wt soil; p<0.05; 1996: 16.9 vs. 28.7 µg/g dry wt soil; p<0.10). Calcium concentrations were also significantly lower in burned treatments than in unburned in September 1996 (84.5 vs. 169 µg/g dry wt soil; p<0.05).

Microbial Biomass

In the O horizon, microbial biomass was dominated by fungi (32 µg/g dry wt soil) rather than bacteria (14 µg/g dry wt soil), whereas in the mineral horizon, microbial biomass of bacteria (11.3 µg/g dry wt soil) and fungi (11.4 µg/g dry wt soil) were similar.

Table 3—Mean pH values for O horizon of burned and unburned treatments of Klaber, Whittleton, and Pinch-Em-Tight Ridges, Daniel Boone National Forest, Kentucky, prior to and following March 1995 and March 1996 prescribed fires

	Site					
	Klaber		Whittleton		Pinch-Em-Tight	
Date	Burned	Unburned	Burned	Unburned	Burned	Unburned
Pre-burn 1995	3.77	3.69	3.20	3.31	3.09	3.05
1 wk post-burn 1995	3.93	3.58	3.64	3.18[a]	3.21	3.16
September 1995	3.64	3.23[a]	3.05	2.93	2.88	2.85
Pre-burn 1996	3.10	3.30			2.94	3.10
2 wk post-burn 1996	3.32[b]	3.63			3.25[b]	3.14
September 1996	3.20	3.53			3.17	3.08

[a] Covariate analysis of $[H^+]$ shows burned treatment significantly different from unburned within ridge at p<0.01 using pre-burn $[H^+]$ as covariate.
[b] Analysis of pre-burn and post-burn $[H^+]$ shows post-burn significantly different from pre-burn at p<0.10.

Table 4—Mean extractable cations (µg/g dry wt soil) in organic and mineral horizons of Burned and Unburned treatments of three ridgetops in Daniel Boone National Forest, Kentucky, prior to and following March 1995 and March 1996 prescribed fires

	O horizon				Mineral soil			
	1995		1996		1995		1996	
	Burned	Unburned	Burned	Unburned	Burned	Unburned	Burned	Unburned
K								
Pre-burn	495	469	467	510	80.5	85.9	75.5	84.3
	(30)[a]	(29)	(39)	(39)	(6.7)	(6.3)	(5.3)	(4.9)
Post-burn	513	558	473	433	86.7	84.0	95.8	102
	(35)	(34)	(47)	(47)	(5.9)	(5.9)	(6.3)	(6.0)
September	362	439	362	490	62.4	71.7	61.1	75.3
	(25)	(24)	(22)	(23)	(5.0)	(4.7)	(4.7)	(4.4)
Mg								
Pre-burn	198	340	190	226	25.3	29.1	19.4	22.2
	(93)	(90)	(20)	(20)	(2.7)	(2.0)	(3.0)	(6.0)
Post-burn	228	216	209	204	24.7	25.6	25.2	29.5
	(12)	(12)	(22)	(22)	(1.8)	(1.3)	(1.5)	1.4)
September	165	190	164	246	21.6	25.0	16.9	28.7
	(13)	(12)	(13)	(14)	(1.4)	(1.7)	(2.5)	(2.4)
Ca								
Pre-burn	2010	2040	1440	2460	148	172	100	147
	(168)	(163)	(281)	(133)	(12)	(14)	(7.8)	(22)
Post-burn	2310	2060	1560	1800	144	130	135	159
	(182)	(173)	(263)	(262)	(11)	(7.5)	(11)	(11)
September	1900	1560	1240	2500	134	127	84.5	169
	(129)	(123)	(180)	(188)	(14)	(8.0)	(23)	(23)

[a] Standard errors (in parentheses) are based upon mean within a treatment.

The prescribed fires of 1995 and 1996 did not consistently affect active fungal biomass, but active bacterial biomass in the organic horizon appeared to be positively affected by both the 1995 and 1996 burns. In the 1995 burned treatments of Whittleton and Pinch-Em-Tight Ridges, active bacterial biomass was higher than the unburned for almost every sampling date post-burn. Burned treatment active bacterial biomass was significantly higher in July 1995 (13 vs. 10 µg/g dry wt soil; $p<0.05$), in September 1995 (15 vs. 9 µg/g dry wt soil; $p<0.01$), and in November 1995 (20 vs. 16 µg/g dry wt soil; $p<0.10$) (Table 5). In May 1995, active bacterial biomass was significantly lower in the burned treatments (17 vs. 20 µg/g dry wt soil; $p<0.10$). In the 1996 burned treatments, active bacterial biomass was significantly higher 2 weeks post-burn (17 vs. 12 µg/g dry wt soil; $p<0.05$) and in August 1996 (7.3 vs. 5.2 µg/g dry wt soil; $p<0.05$). No significant effect on active bacterial biomass was found in the mineral soils.

DISCUSSION

Increases in nutrient availability following fire have been found in some systems and fire regimes and not in others. More intense fires of logging slash in the southern Appalachians have combusted some of the organic layer without significant loss of C or N from the Oea horizon (Vose and Swank 1993), and have also increased available soil nitrogen (Knoepp and Swank 1993). A stand replacement fire in this region resulted in no detectable change in soil chemistry (Vose and others [In Press]). Other fire studies in other regions show significant increases in soil nutrient concentrations following burning (Austin and Baisinger 1955, Prieto-Fernandez and others 1993).

The oak-pine ridgetops in the Red River Gorge are characterized by a thin litter layer (Oi) atop a very acidic, thin, nutrient-poor organic horizon (Oea). As none of the organic layer (Oea) combusted in these prescribed fires, the fires caused no immediate nutrient loss from the Oea horizon, and with only a fraction of the litter layer consumed, the increase of nutrient concentrations was very small. Also on such a nutrient-poor site, any addition of nutrients may have been quickly taken up by plants or immobilized by microbes (Hungerford and others 1990). While cations in the O horizon did not significantly change soon after burning, significant increases of some cations in the mineral horizon after burning suggest increased mineralization and possibly leaching of nutrients from the

Table 5—Mean active microbial biomass (µg/g dry wt soil) in the O horizon of Burned and Unburned treatments of three ridgetops in Daniel Boone National Forest, Kentucky, following March 1995 and March 1996 prescribed fires

	O horizon			
	1995		1996	
	Burned	Unburned	Burned	Unburned
1 d post-burn	15.5 (0.99)[a]	14.9 (0.97)	10.4 (2.1)	9.36 (1.1)
1 wk post-burn	15.9 (1.7)	13.2 (1.3)	—	—
2 wk post-burn	—	—	17.4 (0.90)	11.8 (1.2)
April	25.2 (2.1)	20.5 (2.0)	—	—
May	16.5 (0.93)	19.8 (1.6)	13.0 (1.0)	11.7 (2.9)
June	5.53 (1.2)	11.7 (1.3)	11.4 (0.59)	10.4 (0.63)
July/August	12.8 (1.1)	10.1 (0.73)	7.33 (0.45)	5.16 (0.90)
September	14.7 (1.7)	8.66 (0.69)	8.27 (0.53)	7.84 (2.2)
November	20.3 (2.0)	15.6 (1.3)	—	—

[a] Standard errors (in parentheses) are based upon mean within a treatment.

ash through the thin organic horizon to the mineral soil. We detected a transient increase in available nitrogen, and we know that some increased nutrient availability occurs as evidenced in increased seedling foliar N, P, and K concentrations following burning (Gilbert and others 1998, Reich and others 1990). In another acidic, nutrient-poor ecosystem, the southeastern Coastal Plain pine flatwoods, herb layer vegetation had significant increases in tissue nutrient concentrations for N, P and K following winter fire (Gilliam 1991).

Availability of nutrients for plant uptake depends in part on soil pH, which often changes after burning as oxides in ash react with hydrogen ions to raise pH (Agee 1993, Ahlgren and Ahlgren 1960). We found significantly higher pH in burned soils both years. Higher pH may also have favored bacterial populations (Ahlgren 1974) which increased following burning. While many studies show decreased microbial populations after fire (Ahlgren 1974, Borchers and Perry 1990, Pietikäinen and Fritze 1993), other studies have found bacterial populations to increase following burning (Ahlgren and Ahlgren 1965, Jurgensen and others 1981). Lower concentrations of available cations measured late in the season may reflect increased immobilization by increasing bacterial biomass (Fritze and others 1993).

Fire is being reintroduced to these ridgetops in an attempt to maintain a community diversity that has been decreasing since fire suppression. Successful regeneration of oak species is integral to this diversity. Our data (Arthur and others 1998) as well as other studies (Barnes and Van Lear 1998, Thor and Nichols 1974, Van Lear and Waldrop 1989) suggest that multiple fires will be necessary to realize this objective. This study suggests that low intensity single fires where little, if any, organic matter is lost have little effect on soil nutrient status. Future research should examine whether repeated fire on these sites has similar effects on soil nutrients and microbial biomass or whether impacts of repeated burning on soil resources further degrade already nutrient-poor sites.

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