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Stand restoration burning in oak–pine forests in the southern Appalachians: effects on aboveground biomass and carbon and nitrogen cycling

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Abstract

Understory prescribed burning is being suggested as a viable management tool for restoring degraded oak–pine forest communities in the southern Appalachians yet information is lacking on how this will affect ecosystem processes. Our objectives in this study were to evaluate the watershed scale effects of understory burning on total aboveground biomass, and the carbon and nitrogen pools in coarse woody debris (CWD), forest floor and soils. We also evaluated the effects of burning on three key biogeochemical fluxes; litterfall, soil CO_2 flux and soil net nitrogen mineralization. We found burning significantly reduced understory biomass as well as the carbon and nitrogen pools in CWD, small wood and litter. There was no significant loss of carbon and nitrogen from the fermentation, humus and soil layer probably as the result of low fire intensity. Burning resulted in a total net loss of 55 kg ha^{-1} nitrogen from the wood and litter layers, which should be easily replaced by future atmospheric deposition. We found a small reduction in soil CO_2 flux immediately following the burn but litterfall and net nitrogen mineralization were not significantly different from controls throughout the growing season following the burn. Overall, the effects of burning on the ecosystem processes we measured were small, suggesting that prescribed burning may be an effective management tool for restoring oak–pine ecosystems in the southern Appalachians.
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Keywords: Prescribed fire; Soil CO_2; Coarse woody debris; Litter

1. Introduction

Fire has played an important historical role as a disturbance agent in shaping the structure and function of southern forest ecosystems (Taylor, 1973; Van Lear and Waldrop, 1989). A large body of literature has emerged suggesting that fire was a significant disturbance agent prior to fire suppression policies instituted in the early 1900s. Abrams et al. (1996) suggest *Pinus* and *Quercus* species were maintained in presettlement southern Appalachian forests by periodic fire disturbance, and Native American use of fire likely played a dominant role in shaping the structure and function of Appalachian forests (e.g. Harmon, 1982). More broadly, Native American use of fire created distinct landscape patterns in the eastern United States from the 16th to 18th centuries (Hammett, 1992).

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The interaction of poor logging practices, land clearing, fire suppression and insect outbreaks have resulted in degraded conditions in pine–hardwood forest ecosystems throughout much of the southern Appalachians. For example, in oak–shortleaf pine ecosystems, high-grading and fire suppression have resulted in a successional trajectory that includes invasion by shade tolerant white pine (*Pinus strobus* L.) which may lead to a white pine overstory rather than the historical mix of shortleaf pine (*Pinus echinata* Miller) and oak species. Fire is being suggested as a possible tool to restore these ecosystems to a shortleaf pine–oak community type.

Recognizing that fire has been an important disturbance agent in forests, ecologists and land managers have championed the use of prescribed fire to mimick natural disturbance regimes, improve biodiversity of forest stands (Mitchell et al., 2002; Palik et al., 2002) or to return degraded ecosystems to presettlement conditions. Fire has also been shown to improve wildlife habitat (Lanham et al., 2002), reduce competition from undesirable species (Clinton and Vose, 2000), and may serve as an effective tool for oak regeneration (Arthur et al., 1998; Brose and van Lear, 1998). What is lacking is comprehensive analyses of the effects of prescribed fire on ecosystem pools and cycling rates. Because these processes are critical to our understanding of sustainable productivity, knowledge of fire effects on ecosystem processes is key for sound silvicultural management that includes the use of prescribed fire (Tiedemann et al., 2000; Vose, 2000).

Fire has the capacity to significantly alter carbon (C) cycling in forest ecosystems (e.g. Johnson and Curtis (2001), Harden et al. (2000)). Fire consumes C stored in forest ecosystems and can alter C cycling rates through impacts on leaf biomass and decomposition. Litterfall replenishes forest floor, soil C and nutrient stocks consumed by fire. Soil C represents the largest terrestrial C pool (Schlesinger, 1997) suggesting that small changes in soil CO_2 flux may have potentially large impacts on C storage. Few studies have examined fire effects on soil CO_2 flux and none to our knowledge in the southern Appalachians.

Nitrogen (N) commonly limits productivity in temperate forests (Vitousek et al., 1982) and fire can affect N pools and cycling rates in forest floor and soils depending on fire severity and intensity. N availability may be influenced by ash deposition of N from burned vegetation, alteration of soil microclimate through vegetation and/or forest floor consumption, or changes in microbial dynamics (Raison, 1979). N has low volatilization temperatures (200–375 °C) such that even low intensity fires may remove significant amounts of N from the system (Boerner, 1982). In addition, NO_3^--N, often produced following disturbance, is mobile and may be depleted from forest soils via leaching especially when fire significantly reduces plant biomass and uptake capacity.

Because the use of prescribed fire is expected to increase over the next decade (Haines et al., 2001), information is needed on the effects of fire on C and nutrient cycling. Although there is some evidence suggesting prescribed fire can aid in regenerating hardwoods in xeric oak–pitch pine communities without significantly degrading nutrient capital (Knoepp and Swank, 1993; Vose et al., 1999), there is little information on how fire affects ecosystem properties in shortleaf, oak–pine community types.

The objectives of our study were to quantify the effects of an understory prescribed burn on several key components of the C and N cycle in a degraded oak–shortleaf pine ecosystem. We measured changes in overstory and understory biomass, as well as C and N pools, in forest floor, small wood and coarse woody debris. We also quantified changes in three key biogeochemical fluxes; litterfall, soil CO_2 flux and soil N availability in response to the burn treatment.

2. Methods

2.1. Site description

Our study sites are located in the Conasauga River Watershed in southeastern Tennessee and northern Georgia. Within the Conasauga watershed, six small watersheds (average 11.5 ha) of similar aspect, soils, and vegetation were selected for study, three (one control and two treatment) each in Georgia and Tennessee. The Georgia sites are located in the Chattahoochee National Forest, Murray County, GA (34°49′N, 84°41′W) and the Tennessee sites are located in the Cherokee National Forest, Polka County, Tennesse (35°00′N, 84°39′W). All sites have soils classified as Junaluska and Junaluska complexes. Georgia sites are located on Junaluska soils of the fine sandy loam, thermic phase. The Junaluska is classified; fine-loamy, mixed, mesic Typic Hapludults. Sites in Tennessee are mapped in the Junaluska–Citico complex in lower slope positions and Junaluska–Brasstown in ridge and upper slope areas. The Citico soil series is a fine-loamy, mixed, mesic Typic Dystrochrept while the Brasstown is a fine-loamy, mixed, mesic Typic Hapludult. Burn and control treatments were randomly assigned within each state. On each site, we established five 10 m × 20 m plots from the ridge top to the riparian zone with the long axis of each plot parallel to the slope contour. Overstory composition was dominated by oak species (*Quercus falcat* Michaux, *Q. alba* L., *Q. coccinea* Muenchh, *Q. prinus* L.), as well as sourwood (*Oxydendron arboreum* (L.) DC), Virginia pine (*P. virginiana* Miller) and shortleaf pine. Understory composition consisted primarily of mountain laurel (*Kalmia latifolia* L.) and white pine (*P. strobus* L.). Mean annual temperature is 14 °C and yearly precipitation averages 1350 mm measured at a weather station near the sites (Cleveland, TN, National Climatic Database).

2.2. Fire characterization

Treatment sites were burned on 28 March 2000 with backing fires, using drip torches. Firebreaks were installed along the edge of each small treatment watershed prior to burning. Fire crews from the Ocoee and Cohutta ranger districts began burning the sites from the top of the watershed and proceeded to the riparian zone by burning strips at 10–25 m intervals depending on slope steepness.

We used heat sensitive chalk and paint (Omega Engineering, Inc.) coated tiles to characterize the temperature of the burn. Two days prior to burning, we suspended four 10 cm × 20 cm temperature tiles 30 cm above the forest floor in random locations within each 10 m × 20 m plot ($n = 20$ per site). Chalk temperature sensitivity ranged from 52 to 427 °C in approximately 14 °C increments. Paint temperature sensitivity was 500, 550, 732, 804 and 899 °C. We estimated temperature during the burn to be intermediate between melted and intact chalk or paint. Using a similar technique, we also monitored heat penetration into the forest floor. In each 10 m × 20 m plot, two long narrow tiles painted with heat sensitive paint were inserted 15 cm into the soil with the top edge being flush with the top of the Oe layer. Threshold temperature sensitivity of the paints was 45 and 59 °C, a range that brackets the thermal lethal point for most plants (Hare, 1961).

2.3. Aboveground biomass

We measured woody vegetation by layer before and after the prescribed fire: understory layer (woody stems >0.5 m height and <5.0 cm dbh); and overstory layer (all trees $\geq$5.0 cm dbh) in each of the five 10 m × 20 m plots per site. We sampled the understory layer in a nested 5 m × 5 m plot within each 10 m × 20 m plot. We measured understory stems to the nearest 0.1 cm at 3 cm from ground level and overstory stems at 1.37 m from ground level. To estimate aboveground biomass we used allometric equations from Martin et al. (1998) for overstory and understory hardwood species, equations from Boring and Swank (1986) for understory evergreen species, and equations from Van Lear et al. (1984) for pine species.

2.4. Forest floor mass, C and N pools

We randomly arrayed four pairs of 0.09 m^2 subplots within each 10 m × 20 m plot and measured forest floor and small wood mass 1 week prior to burning. A 0.3 m × 0.3 m wooden frame defined the sampling area and pre-burn samples were collected one week prior to the burn treatments. Pre-burn sample locations were marked using metal stakes. On each 0.09 m^2 subplot, we removed all small wood samples ($\leq$5 cm) and then cut along the edge of the wooden frame with a sharp knife to remove the forest floor. We determined the initial mass by sampling two forest floor components: litter, defined as unconsolidated material on top of the fermentation layer and the combined fermentation (Oe) and humus (Oa) layer. All samples were transported to the laboratory, dried for 72 h at 60 °C, and weighed. One week following the burn we returned to the 0.09 m^2 subplots and sampled an adjacent subplot. We re-sampled the litter layer and assessed changes by the difference between pre-burn and post-burn values. Visual observation and low fire temperatures indicated that all forest floor consumption occurred in the litter layer. Changes in post-burn mass loss in the Oe + Oa layer were estimated indirectly with a humus consumption model that relates Oe + Oa mass loss to fire temperature (Clinton et al., 1998). A subsample of the Oe + Oa layer was collected for C and N analysis. Samples from the subplots were composited, ground to <1 mm and analyzed for total C and N content using a Perkin Elmer 2400 CHN analyzer.

All coarse woody debris (CWD) within each $10\,m \times 20\,m$ plot was measured before and after the burn treatments. We defined CWD as any woody material greater than 5 cm in diameter and categorized it into five decay classes (one being freshly fallen material and five as advanced decay). Prior to burning, volume of all CWD was estimated by measuring total length and diameter at 1 m intervals, or at the plot edge if the log extended beyond the plot boundary. At each measurement point, we wrapped wire bands around the log to mark our measurement point. Volume for each segment was estimated as a truncated cone using the length of the segment and the diameter at each end. For decay class 5 material, we measured the width and depth of the decayed log at 1 m intervals and calculated volume as a truncated pyramidal frustum. Prior to the burn, we estimated mass for each decay class using specific gravity ($g\,cm^{-3}$) estimates based on subsamples from each decay class (and species, where identification was possible) at each site (five subsamples per site). We determined the percent C and N in ground subsamples for each decay class to estimate the total C and N content of CWD for each plot. Following the burn, we remeasured all remaining CWD at the wire band locations and estimated the change in mass, and C and N content by subtracting post-burn from pre-burn values.

2.5. Litterfall

We installed five $1.55\,m^2$ plastic litter baskets at 4 m intervals along a transect down the center of the long axis of the $10\,m \times 20\,m$ plots. We installed the litter baskets during the last week of September 2000 and collected litterfall (leaves and small wood) from October 2000 through February 2001, when we removed the litter baskets prior to burning. After burning in late March, we re-installed the litter baskets and resumed collections until completion of litterfall (November 2001). During October and November, when litterfall was greatest, we collected litter every 2 weeks; otherwise, litter was collected monthly. After collection, samples were transported to the laboratory, dried for 72 h at 60 °C and weighed for mass determination.

2.6. Soil CO_2 flux

Immediately after the burn, we began measuring soil CO_2 flux ($\mu mol\,m^{-2}\,s^{-1}$) in two control and two treatment sites using a LiCor 6400 gas exchange analyzer with soil respiration chamber attachment (LiCor Inc., Lincoln, NE, USA). The 6400 is a closed system infrared gas analyzer that measures soil respiration (microbial and root). We installed five PVC collars ($0.008\,m^2$) randomly located in each $10\,m \times 20\,m$ plot 2 days after the burn (25 per site, 100 total). Collars were imbedded approximately 2 cm into the soil and left in place throughout the measurement period. We did not begin measurements until 1 week after installation to minimize the effects of disturbance from collar installation. Along with soil CO_2 flux, we also measured soil temperature (LI-6400 soil Temperature Probe, LiCor Inc., Lincoln, NE, USA) and moisture (CS620, Campbell Scientific Inc., Logan, UT, USA) at 10 cm for each measurement point. We repeated these measurements approximately monthly until November 2001.

2.7. Soil nitrogen pool and nitrogen mineralization

On each plot, two transects were established parallel to the contour prior to burning. Sampling points on each transect were chosen randomly to minimize effects of soil variation within plots. In late February 2001, we measured pre-treatment soil NO_3 and NH_4 concentrations. Post-treatment measurements of NO_3 and NH_4 concentrations plus N transformation rates were made immediately following the burn, in April and again in June and July 2001.

We used a modified in situ closed core method (Adams and Attiwill, 1986) to measure net rates of N transformations. Two 15 cm long, 4.3 cm inside diameter, PVC cores were driven 10 cm into the mineral soil within 25 cm of each randomly selected sample point. One PVC core was removed immediately and returned to the laboratory for time zero determination of soil NH_4^- and NO_3-N concentrations. The second core was capped and incubated in the field for 28 days before retrieval. In the field; soils were removed from PVC cores and placed in a plastic bag. Soils were mixed thoroughly and a subsample (approximately 10 g) was added to a pre-weighed 125 ml polyethylene bottle containing 50 ml 2 M KCl. The bottles plus soil were kept cool until they were returned to the laboratory and then stored in a refrigerator at 4 °C until analyzed, within 24 h. Bottles plus soil were weighed to determine the weight of soil extracted. Remaining soil samples were moist sieved to <6 mm and a subsample (~20 g) was dried (105 °C) for >12 h to obtain oven-dry weight. Soil plus KCl samples were shaken and allowed to settle overnight (refrigerated); 15 ml of the clear KCl was pipetted into a sample tube for NH_4^+- and NO_3^--N analysis. Supernatant NH_4- and NO_3-N concentrations were determined on an autoanalyzer using alkaline phenol (Technicon, 1971) and cadmium reduction (USEPA, 1983) techniques, respectively. All soil N data are reported on an oven dry weight basis. Net N mineralization rates were calculated as soil $NH_4 + NO_3$-N concentrations at 28 days minus $NH_4 + NO_3$-N concentrations at time zero. Net nitrification rates equaled soil NO_3^--N concentrations at 28 days minus NO_3^--N concentrations at time zero.

2.8. Statistical analysis

For all analyses, we assessed differences using a significance level of $\alpha = 0.1$. We analyzed for post-fire differences in soil N content and mineralization, as well as mass and C and N pools of above-ground live woody biomass, small and coarse woody debris, and forest floor components using a paired *t*-test (SPSS, Inc.). To assess the effects of prescribed fire on litterfall and soil CO_2 flux, we measured post-fire differences in monthly litterfall and soil CO_2 flux using analysis of variance (Neter et al., 1990) to compare control and treatment sites.

3. Results

During the burn, flame length varied for 30–150 cm and the rate of spread ranged from 3 to 30 cm s^{-1}. Temperatures at 30 cm above the forest floor averaged 96 °C over all burned sites. Burn temperatures varied from the ridge to the riparian zone in each site but, overall, the differences were not significant (Fig. 1, $P > 0.1$). The mean maximum temperature for all burned plots was 135 °C. Heat penetration into the soil was minimal with temperatures of 45 and 59 °C penetrating an average of 2.0 and 1.0 cm, respectively.

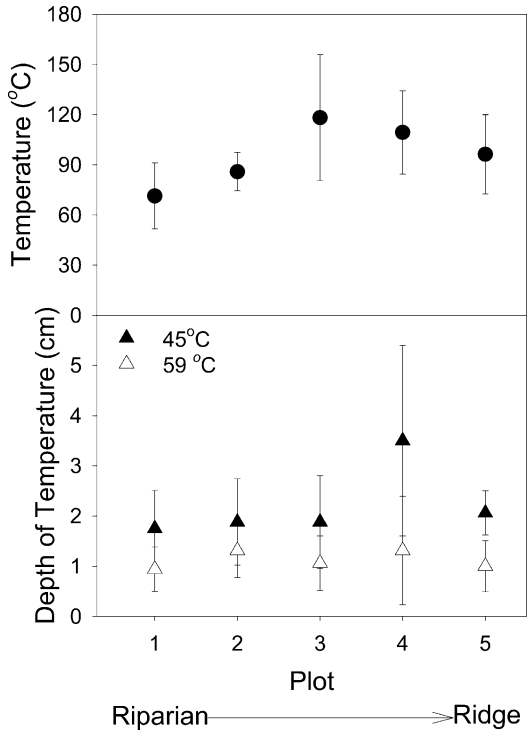

Fig. 1. Mean burn temperatures and depth of heat penetration for understory prescribed burn by plot locations. Burn temperatures and depth of heat penetration were measured with heat sensitive paint on ceramic tiles. There was no significant difference between plot location and burn temperature ($P = 0.70$) or heat penetration ($P = 0.20$ at 45 °C, and $P = 0.24$ at 59 °C). Error bars represent variation between plot locations ($n = 4$) and are ±1 S.E.

Burning had no effect on total aboveground live woody biomass (overstory and understory layers combined) ($P = 0.32$, Table 1); however, understory biomass alone was reduced by approximately 50% ($P = 0.07$). Overstory biomass (which comprises approximately 97% of total aboveground biomass) was 11% lower than pre-burn biomass, but the difference was not significant ($P = 0.5$).

Burning significantly reduced CWD and forest litter mass. Fire consumed approximately 12% of the CWD

Table 1
Vegetation and forest floor mass prior to and following understory burning in a mixed oak–pine forest in the southern Appalachians (kg ha^{-1})[a]

	Mass (kg ha^{-1})		
	Pre-burn	Post-burn	Loss
Aboveground understory biomass	424 (89)	197 (89)	227*
Aboveground overstory biomass	61,526 (6201)	54,698 (6397)	6,828
Total aboveground biomass	61,951 (6179)	54,896 (6328)	7,055
Coarse woody debris (CWD)	7,611 (1678)	6,696 (1343)	915*
Small wood	6,906 (979)	4,425 (730)	2,481*
Litter (Oi)	6,028 (1407)	1,833 (233)	4,195*
Humus (Oa, Oe)	11,435 (1043)	10,837[b] (1152)	598
Total	93,931	78,687	15,244*

[a] Values in parentheses are standard errors.
[b] Estimated using Clinton et al. (1998) model.
* Loss was significant at $P < 0.1$.

on the sites ($P < 0.1$, Table 1), while small wood and the Oi layer were reduced by approximately 36 and 70%, respectively ($P < 0.1$, Table 1). There was no significant difference in mass of the Oa + Oe layer between pre-burn and post-burn values.

Coincident with changes in wood and forest litter mass, burning significantly reduced C and N pools in CWD, small wood and forest litter. Total C content in CWD, small wood and litter layer were reduced by 12, 40 and 70%, respectively; while N contents in these same pools were reduced by 14, 49 and 74% (Table 2). A small amount of N was lost from the Oe + Oa layer (~5%), but the change was not significantly different from pre-burn values (Table 2).

There were no significant changes in monthly pre-burn and post-burn litterfall. However, there was a shift in the peak litterfall date for both control and treatment sites during the second year of measurements. During the fall preceding the burn treatment, litterfall on control and treatment sites peaked in November 2000 averaging 940 kg ha^{-1}. In the year following the burn, litterfall in control and treatment sites peaked in October 2001, averaging 1290 kg ha^{-1} (Fig. 2).

Average soil CO_2 flux on burned sites decreased relative to controls during the first 2 months (17 and 26% lower, respectively) after burning. Burn sites were not significantly different from controls for the remainder of the measurement period (Fig. 3). Abiotic soil variables (i.e., soil moisture and temperature) varied seasonally; however, only soil moisture differed between burn and control. For example, soil moisture was generally higher in burn compared to control sites, but differences were only significant during June, July

Table 2
Forest floor carbon and nitrogen content prior to and following understory burning of a mixed pine–oak forest in the southern Appalachians (kg ha^{-1})[a]

	C (kg ha^{-1})			N (kg ha^{-1})		
	Pre-burn	Post-burn	Loss	Pre-burn	Post-burn	Loss
Coarse woody debris (CWD)	3,864 (812)	3,405 (645)	459*	14.2 (4.92)	12.2 (4.07)	2.0*
Small wood	3,558 (535)	2,167 (356)	1391*	24.2 (1.6)	12.4 (2.1)	11.8*
Litter (Oi)	3,105 (736)	901 (115)	2204*	58.1 (15.8)	15.1 (1.6)	43*
Humus (Oa, Oe)	3,905 (334)	3,912 (385)	0	104.8 (8.7)	99.2 (7.3)	5.6
Total	14,432	10,385	4054	201.3	138.9	62.4*

[a] Values in parentheses are standard errors.
* Loss was significant $P < 0.1$.

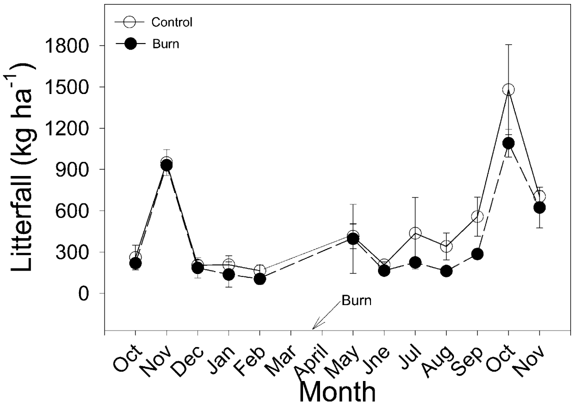

Fig. 2. Mean monthly litterfall for control and burn treatments. There were no significant monthly differences ($P > 0.1$) between control and treatment plots. Error bars represent variability between sites ($n = 2$) and are ±1 S.E.

and September (Fig. 3, $P < 0.1$). Soil temperature was not significantly different between control and treatment sites during the measurement period.

Available soil N content and N mineralization remained unchanged after burning. Soil NO_3^--N and NH_4-N remained almost constant over the measurement period (Fig. 4, $P > 0.1$). N mineralization exhibited slightly higher values later in the measurement period, but control and treatment were not significantly different over the course of the study (Fig. 4, $P > 0.1$).

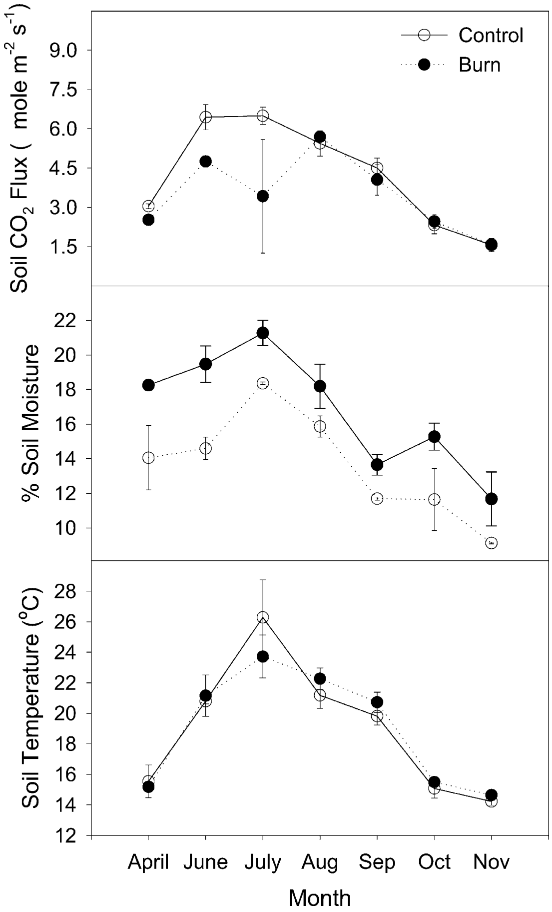

Fig. 3. Mean monthly soil respiration, soil moisture and soil temperature for control and burn treatments. Soil respiration in burn treatments was significantly lower than controls during April and June ($P = 0.03$ and $P = 0.08$, respectively) but not during any other measurement period ($P > 0.1$). Soil moisture was higher in burn treatments during June, July and September ($P < 0.1$) while soil temperature did not change after burning ($P > 0.1$). Errors bars represent variability between sites ($n = 2$) and are ±1 S.E.

4. Discussion

Fire intensity is defined by the upward heat pulse produced by the fire (Ryan and Noste, 1985), while fire severity is defined by depth of penetration (Wells et al., 1979) The prescribed burn in this study was of low intensity and severity as defined by our temperature tile measurements (Fig. 1). Temperatures were low compared to other understory burns in the southern Appalachians. Elliott et al. (in press) found temperatures averaged almost twice those measured in this study (95 °C versus 188 °C) in a mixed mesophytic cove forest and Clinton et al. (1998) measured temperatures averaging 197 °C across treatment sites in a mixed pitch pine–oak forest. Maximum flame temperatures in these other studies (800 and 700 °C, respectively) were also substantially higher than we measured (344 °C). Cooler weather conditions, higher fuel moisture and wider spacing between drip lines in this study may explain the differences.

The management prescription for our study site was implementation of a low intensity burn aimed at significantly reducing understory white pine and preparing a more favorable seedbed for shortleaf pine (Shelton and Cain, 2000) and Virginia pine (Carter and Snow, 1990) establishment. Because our fire temperatures were relatively low, total live biomass consumption was not large, however there was a significant

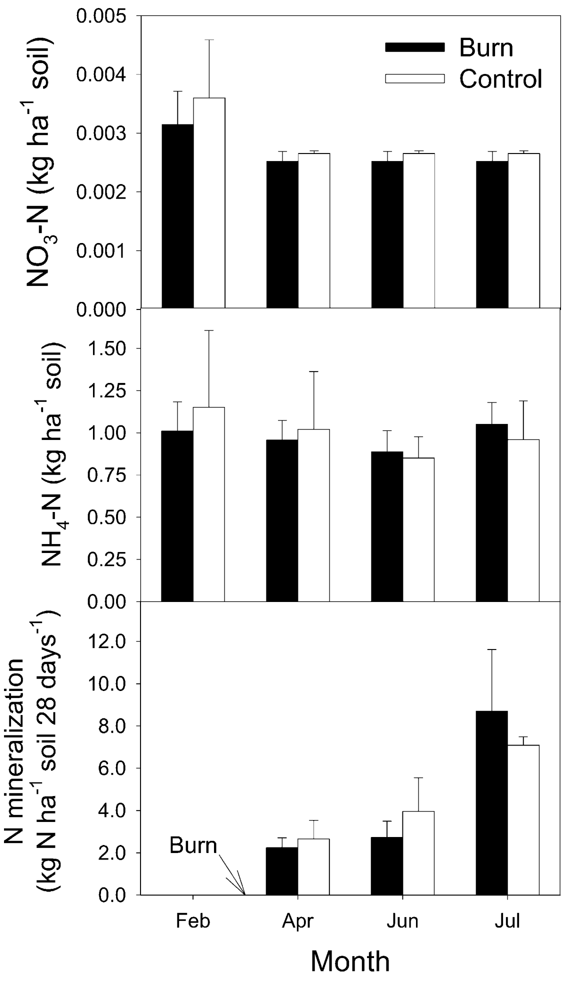

Fig. 4. Total soil NO_3^- and NH_4 and net nitrogen mineralization in the upper 15 cm of soils in control and burn treatments. There was no difference in nitrogen content or mineralization with treatment ($P > 0.1$). Errors bars represent variability between sites ($n = 2$) and are ±1 S.E.

reduction (53%) in understory biomass suggesting competition pressures on future shortleaf and Virginia pine establishment may be lessened.

CWD is an important component of forest ecosystems because it provides habitat for various forest organisms (Batzer and Braccia, 1999; Loeb, 1999) and can be a significant reservoir for C and nutrients (Arthur et al., 1993; Harmon et al., 1986; Vogt et al., 1995). In our study, approximately 12% of the CWD mass, C and N was consumed by understory burning, a loss unlikely to have a large effect on CWD-dependent invertebrate and mammalian species or C and N reservoirs. Vose and Swank (1993) found large reductions in woody biomass after a fell and burn treatment owing to large fuel loads and intense fire; but Vose et al. (1999) found no reduction in CWD mass or nutrients after a stand replacement burn in a southern Appalachian pine–hardwood ecosystem. Taken together, these studies and our data suggest that restoration burning on sites with low to moderate fuel loads is unlikely to significantly affect CWD structure and nutrient capital.

Effects of prescribed burning on forest floor C and nutrient content are critical to understanding fire impacts on C and nutrient cycling. In southern Appalachian ecosystems, up to 50% of the total plant available N is provided by the forest floor (Monk and Day, 1988). Although the understory burn in this study was of low severity and intensity, the fire consumed significant amounts of mass, C and N from small wood and litter pools. Litter mass, C and N, consumption by the fire was large (70, 71 and 74% respectively); but because monthly litterfall was unaffected (Fig. 3), it is likely that these pools will be replaced over the next several years. We calculated that the low severity burn in this study had no effect on mass, C and N content in the combined Oe + Oa layer. Results from other burn studies in the southern Appalachians are mixed regarding mass and nutrient loss from the small wood and forest floor pools. In contrast to this study, Elliott et al. (in press) found larger reductions in C (93, 53.7 and 32%, respectively) and N (93, 48 and 37%) content of small wood and Oi and Oe + Oa layers. However, even in studies with much larger fuel loads resulting from fell and burn treatments or slash burning, consumption of the Oe and/or Oa layers may not be significant. For example, Vose et al. (1999) found similar reductions in the small wood and Oi layer C and nutrient content while observing no change in the combined Oe + Oa layers. Kodama and Van Lear (1980) reported that prescribed burning in *Pinus taeda* plantations caused significant reductions in the Oi layer but no change in the Oe + Oa layer. The wide range of responses are likely due to variations in fuel conditions, fire intensity and severity as well as the magnitude of the initial pool sizes.

Annual inputs of litterfall represent the primary above ground inputs of C and nutrients to the forest floor and represent a potentially large mineralizeable pool of N. Because even low intensity fire consumes large proportions of the aboveground litter layer as in this study, fire effects on subsequent litterfall inputs may be important. Monthly litterfall on treatment sites was similar to controls before and after burning suggesting that fire did not adversely affect the likelihood that overstory canopy production will be adequate to replace the Oi layer. If fire damage causes some delayed mortality in subsequent years, increased litterfall may result in a faster recharge of the Oi layer. In both control and treatment plots, there was a shift in peak litterfall from 2000 to 2001, however we suspect that this was not caused by the burn treatments. The shift may have resulted from natural annual differences in phenology, or by the slight increase in southern pine beetle infestation on several of our control and treatment plots. We are unaware of any other studies that have examined the effects of understory burning on litterfall, but our data suggest that low intensity/severity understory burning will have little impact on annual litterfall inputs.

Soil respiration is a major C flux in forest ecosystems and can be equal to or greater than net primary production (Raich, 1998). Because soil CO_2 flux is comprised of both heterotrophic and autotrophic respiration, fire can influence soil CO_2 flux through impacts on soil temperature and moisture or by damage to plant root systems. We found little impact of understory burning on soil CO_2 flux, suggesting that there will be no significant fire effect on soil C storage in these systems. During the first two measurement periods following burning, soil CO_2 flux was lower in the burn relative to control sites (Fig. 3), but differences were small. Soil CO_2 flux is often positively correlated with soil moisture and temperature but we found higher soil moisture on burn sites with no difference in soil temperature. This suggests that decreases in soil CO_2 flux were driven by a decrease in root respiration resulting from the significant decrease in understory biomass. We are not aware of any other studies in the southern Appalachians that have examined effects of understory burning on soil respiration but restoration burning in a ponderosa pine–bunchgrass system in Arizona had little impact on soil CO_2 flux (Kaye and Hart, 1998).

We found that understory burning had no effect on net soil N mineralization or available soil N (NO_3^- and NH_4^+) (Fig. 4). Recent syntheses of fire studies suggest that in most cases fire promotes both short and long term increases in soil NO_3^- and NH_4^+ (Johnson and Curtis, 2001; Wan et al., 2001); however, the magnitude and duration of the response depends on fire intensity and severity. Increases in NH_4^+ have been attributed to volatilization of organic N from the soil surface with subsequent condensation as it moves down into cooler soil layers (DeBano, 1990; Knoepp and Swank, 1993). Fire induced increases in NH_4^+ and changes in soil physical and microbial activity may lead to higher soil NO_3^- through increased soil nitrifying rates (Wan et al., 2001). The lack of a soil N response in our study suggests that low fire temperatures resulted in little or no effect on factors influencing soil N availability.

Understory burning at these sites resulted in a total N loss of approximately 55 kg ha^{-1} (Table 1) (not including aboveground live vegetation). In a companion study at our site, Ellliot et al. (submitted for publication) found no significant nitrate leaching losses as measured by lysimeters in our riparian plots and by comparing stream chemistry in control and burn treatments. Although total N losses from fire can be large (Wan et al., 2001), our values are similar to other studies in the southern Appalachians. For example, Vose et al. (1999) found N loss from heavily burned ridge tops averaged 75 kg ha^{-1} while Elliott et al. (in press) report slightly higher losses of 91 kg ha^{-1} in a mesic mixed oak–pine forest. Because losses observed in this study were relatively low, we suspect these losses will be quickly replaced by annual N deposition (approximately 11 kg ha^{-1} per year (NADP, 2000)). Some studies have shown that N fixation can add in recovery of N losses following fire (e.g. Hendricks and Boring, 1999), however currently less than 1% of the vegetation cover is comprised of N fixing species. Either free N fixation in soils or establishment of N fixing species during the second growing season following the burn would be required to substantially affect N capital at this site.

5. Conclusions

The impacts of understory burning on C and nutrient cycling measured in our study were minimal. There was little or no effect on C and N fluxes measured by litterfall, soil respiration and N mineralization. N losses in this study were small such that they should be rapidly replaced by atmospheric inputs. Assuming that the understory burn met the silvicultural objectives of reducing the white pine understory and enhancing shortleaf and Virginia pine regeneration, we conclude that prescribed understory burns in these mixed oak–pine ecosystems is an effective management tool that will not adversely affect nutrient and C cycling.

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