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Spatial variability in soil nitrogen dynamics after prescribed burning in Ohio mixed-oak forests

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Abstract

This study describes the results of the application of a single dormant season prescribed fire to two southern Ohio forest sites for the purposes of restoring the ecosystem functional properties that existed in these sites prior to major human intervention (clearcutting, fire suppression, and atmospheric deposition). Each forest site was composed of three contiguous watershed units, two of which were burned in April of 1996. The forest sites differed in soil pH and available litter mass prior to the fires, and in both sites pH and available inorganic N varied among landscape positions such that inorganic C increased with increasing longterm soil moisture potential (measured as the GIS-derived Integrated Moisture Index [IMI] developed for this region). The fire temperatures at 10 cm above the litter surface were generally 150–300 °C, and 29–80% of the litter was consumed, depending on site and landscape position. Soil solution total inorganic N (TIN) present one month after the fires did not differ significantly from that present prior to the fires in either burned or unburned watersheds, but was consistently greater in mesic landscape positions than in more xeric ones. N mineralization potential and organic C content varied both among fires and landscape positions. At the site which burned at higher intensity, soil N mineralization and TIN were both decreased by fire. At the less intensely burned site, fire resulted in increased TIN in the soils from the more xeric landscape position, and greater soil organic C in soils from the intermediate soil moisture areas. Path analysis produced models for fire-induced changes in C and N dynamics capable of explaining 26–69% of the observed variation using combinations of landscape and fire behavior. Losses of N to volatilization from these single fires were generally < 1 kg N/ha, and thus could not be expected to ameliorate the effects of atmospheric N deposition in these sites.

Introduction

Many accounts of early explorers, trappers, and botanists include mention of the common occurrence of fire in the eastern deciduous forest landscape prior to widespread Euro-american settlement (Day 1953; Williams 1989; Whitney 1994). Although large, catastrophic crown fires and their impacts were more likely to be noticed by such observers than might be smaller, cooler surface fires, recent and on-going studies indicate that dormant season fires occurred frequently in this region (Sutherland 1997) at least over the last century. In oak-hickory (*Quercus-Carya*) forests on the unglaciated Allegheny Plateau of southern Ohio, such surface fires may have occurred at intervals of 3–4 years over the last century, with few forested areas being fire free for more than 24 years (Sutherland 1997). Thus, the forested landscape that was settled and cleared by the Euroamerican settlers in the early 1800's was one which had been shaped not only by its geomorphology and climate, but also by the direct and indirect effects of human activity mediated through fire.

During the mid- and late-1800's, the iron industry dominated southern Ohio. To supply the charcoal needed for the blast furnaces of the era, the oak-hickory forests of southern Ohio were cut repeatedly (Stout 1933). This industry and its associated cutting pattern ceased around 1890–1900 as richer iron ore deposits to the north and west were opened for exploitation. Since the cessation of clearcutting for charcoal production, much of the landscape of the unglaciated Allegheny plateau has experienced almost a century of secondary succession. However, this succession has occurred during a century of chronic atmospheric deposition and a half century of effective fire suppression. Through their effects on soil properties and vegetation, we feel both of these factors have the potential to affect the pattern and pace of succession.

Over the last three decades, the abundance of red oaks, white oaks, and hickories have declined by 41%, 31%, and 22%, respectively, whereas the abundance of red maple (*Acer rubrum*), sugar maple (*A. saccharum*), yellow-poplar (*Liriodendron tulipifera*), and black cherry (*Prunus serotina*) have increased by 70%, 44%, 38% and 129% (Iverson et al.1997). As these changes have occurred in unmanaged, uncut plots resampled periodically by the USDA Forest Service, they cannot be the result of harvesting practices. Our overarching hypothesis is that these changes in vegetation dynamics and composition are, at least to some degree, the consequences of the long history of fire suppression and atmospheric deposition to which this landscape has been subjected.

Prescribed fire has been used for fuel and understory management, especially in conifer plantations, for at least 75 years (Riebold 1971). In recent years, the use of prescribed fire for restoration of forest ecosystems that have been altered by fire suppression, insect outbreaks, or management has grown (Hardy and Arno 1996). As the judicious use of prescribed fire in the oak-hickory forests of our region has the potential to both restore the fire regime present prior to this century and partially ameliorate the impact of atmospheric deposition (by volitalizing N and S), we began in 1994 a longterm study of the use of prescribed fire in the restoration of Appalachian oak-hickory forest ecosystems.

The overall objective of our larger, multiyear study is to evaluate the efficacy of prescribed fire at differing intervals on the restoration of Appalachian oak-hickory forest ecosystems through modification of the understory, subcanopy, and forest floor subsystems. The specific objectives of the research reported here were to (1) quantify the pattern and degree of spatial variation in prefire fuel conditions and fire behavior in this heterogeneous landscape, (2) determine the immediate (i.e., one month) and short term (i.e., one growing season) temporal effects of fire on soil organic N and C dynamics and inorganic N availability, and (3) to clarify the interactions between spatial and temporal sources of variability so that we may parameterize a spatially-explicit model of the effects of prescribed fire on C and N dynamics in this landscape.

Based on the considerable literature on prescribed burning in coniferous forests, we hypothesized that a single dormant season fire would increase soil organic C content, N mineralization and nitrification rates, and total inorganic N available for plant and microbial growth. However, we also hypothesized that these effects would vary with landscape position and fire characteristics. Finally, we hypothesized that the quantity of N volatilized in such a fire would be small compared to the annual deposition of N in this region.

Methods

Study sites and sampling design

The two sites chosen for this study were located in Vinton and Lawrence Counties on the unglaciated Allegheny Plateau of southern Ohio. The two sites were each contiguous blocks of 75–90 ha occupied by mixed oak forests which had developed following clearcutting for charcoal production 100–150 yr ago. The Vinton County study site, Arch Rock (latitude 39°11′ N, longitude 82°22′ W and the Lawrence County study site, Young's Branch (latitude 38°43′ N, longitude 82°41′ N) were separated by approximately 40 km. These two sites were a subset of a larger group of sites being used for the longterm prescribed fire/restoration study (see Sutherland 2000 for a complete description of the study and study sites).

The parent materials underlying the study sites were sandstones and shales of Pennsylvanian age. The soils of all four sites were silt loams formed from colluvium and residuum, and were predominantly alfisols (Boerner and Sutherland 2000). The climate of the region is cool, temperate and continental with mean annual temperature and precipitation of 11.3 °C and

1024 mm for the Vinton County site and 12.9 °C and 1059 mm for the Lawrence County site (Sutherland and Yaussy 2000). Microclimatic gradients generated by the steep, dissected topography of the region included the tendency for S, SW and W facing slopes to be drier and warmer than NW, N and E facing slopes due to the strong relief in this region (Wolfe et al. 1949; Hutchins et al. 1976).

The study areas were chosen from a larger pool of candidate study sites on the basis of the following criteria: (1) they met the age and land use history criteria listed above, (2) the three watershed units within each study area were as similar as possible in topography and geology, and (3) there were no indications of significant disturbance since the clearcutting in the mid-to-late 1800's.

Each watershed was stratified using a GIS-based integrated moisture index (IMI) developed by Iverson and Prasad (2000) for this region. The IMI stratification was achieved through integration of aspect, hill shade profile, solar radiation potential, flow accumulation, water holding capacity of the soil, and curvature profile of the landscape (Iverson et al. 1997). Each component was weighted and standardized on a 0–100 scale, and three IMI classes were delimited: xeric, intermediate, and mesic. As this metric delimited areas of differing microclimates it was ideal for representing the 'topographic scale' within each of the watersheds. Within each of the watersheds, three sample plots of 0.125 ha were established in each of the three IMI classes, for a total of nine sample plots per watershed unit and 27 sample plots per study site.

The positions of the sample plots were determined from a digital elevation model overlain with an IMI class map in an ARC/INFO environment, and the overall experiment was designed to be a balanced, randomized block design with study areas as blocks (Iverson and Prasad 2000).

Analysis of the soils of these study areas indicated that soil chemical properties varied significantly among sites and among IMI classes, but not among watersheds within sites (Morris and Boerner 1998a; Boerner et al. 2000). The soils at Young's Branch had significantly greater inorganic N pool size, relative nitrification rate, organic C content, pH, Ca, Mg, and molar Ca:Al ratio than did the soils at Arch Rock. However, N mineralization potentials were greater in soils from Arch Rock than in those from Young's Branch. Similarly, soils from plots located in the xeric IMI class had lower inorganic N pool sizes, N mineralization potential, relative nitrification rate, extractable PO_4, Ca, and Ca:Al ratio than soils in the mesic IMI class plots (Morris and Boerner 1998a; Boerner et al. 2000).

In April 1996, two of the three watersheds in each study area (designated FREQ and INFR) were burned. The designations FREQ and INFR refer to long-term fire regimes planned for these watersheds, with the FREQ watersheds being burned annually and INFR being burned every fourth year. As the results we present here cover only the first set of fires in each watershed, these designations serve here only to differentiate the watersheds. Arch Rock INFR and part of of Young's Branch FREQ were burned on April 18, 1996, a warm, breezy, dry (relative humidity <30%) day. No rain had fallen for 48 hours prior to these fires. In contrast, Arch Rock FREQ, Young's Branch INFR and the remainder of Young's Branch FREQ were then burned on April 19, a cooler and moister day in which approximately 3 mm of rain fell during the early morning.

After the fire lines were cleared and inspected, areas inside the firelines and facing into the wind were burned with backfires. Strip headfires were then started and allowed to burn into the areas blackened by the initial backfires. Because the backfires burned slowly, most of the areas were burned by headfires. In general, flame heights were 20–50 cm. Bark char recorded on saplings ranged from 3–36 cm, but was usually < 15 cm.

Fire temperatures were recorded using temperature-sensitive Tempilac® paints applied to aluminum tags (Cole et al. 1992). One-to-several days prior to a fire, a live-cut tree sapling (<3 cm diameter) was placed in the ground adjacent to each of the four corner posts of each sample plot and at the midpoint of each long axis of each sample plot, yielding N=6 per sample plot. An aluminum tag with paint strips designed to melt at 38, 79, 121, 163, 204, 316, and 427 °C were fastened on the sapling, 10 cm above the forest floor. The maximum temperature melted for each tag was recorded following the fire. In cases where none of the paints melted, 21 °C, the approximate air temperature during the fires, was recorded.

Field methods

To determined the degree to which the fires consumed the unconsolidated litter, litter mass was determined in late March/early April, prior to the fire, and in mid-May, several weeks after the fire. Six samples were collected at each sample plot using a square steel frame (0.0225 m^2) with a sharpened cutting edge. Samples were taken adjacent to each of the four corners of the sample plot and at the mid-point of the long axes. The O_i and O_e layers were collected, including all woody material <0.5 cm diameter. The 0_a layer was not collected as we did not expect this layer to burn under the prescribed fire conditions we utilized (cf. Boerner 1983). The litter samples were dried at 70 °C for 48 h, then weighed.

To determine soil N and C pools and turnover rates, approximately 400 g fresh mass samples of the $O_a + A$ horizon were taken from opposite corners of each of the 27 sample plots in each study site in May of both 1995 (prefire) and 1996 (one month postfire), yielding N=6 per watershed-by-IMI class combination and N=54 for each study area on each date. The 1996 samples were all taken from points within 30 cm of the 1995 sample points. All samples were returned to the laboratory under refrigeration. The results from the 1995 prefire sampling are described by Morris and Boerner (1998a) and are not repeated here.

Laboratory methods

Each soil sample was air dried and sieved to remove roots and large cobbles. A subsample of approximately 15 g of soil was extracted with 2M KCl, and analyzed for NH_4^+ and NO_3^- using colorometric methods on a Lachat QuikChem autoanalyzer. Total inorganic N (TIN) was calculated as the sum of NH_4^+ and NO_3^-. Soil pH was determined in a 1:5 soil slurry of 0.01 M $CaCl_2$ (Hendershot et al. 1993), and organic C was determined using the Walkley–Black method (Allison 1965).

A second subsample of approximately 50 g was placed in an incubation chamber and water added to bring the soil up to 70% of field capacity. The soil samples were incubated for 27–29 days at 24–28 °C. Every third day each soil sample was weighed and sufficient water added to bring the moisture content back to randomly chosen level within the range of 50–70% of field capacity (Morris and Boerner 1998a). Laboratory incubations were chosen for use in this study because the manner in which the moisture regime of the incubating samples were maintained recreated the frequent fluctuations in soil moisture that are characteristic of the growing season in the mixed oak ecosystems of southern Ohio reasonably well (Morris and Boerner, 1998a,b).

At the end of the incubation period, a subsample of 15 g of the incubated soil was extracted and analyzed for NH_4^+ and NO_3^- as above. Net N mineralization was determined by subtracting the NH_4^+ and NO_3^- content in the initial samples from that in the incubated samples. Net nitrification was estimated as the difference between initial and final NO_3^-. Proportional nitrification was calculated by determining the amount of NO_3^- produced from the total NH_4^+ available for nitrification. Total Inorganic N (TIN) was estimated as the sum of NH_4^+-N + NO_3^--N.

Estimating the net effect of fire on TIN, organic C, N mineralization, and nitrification was difficult because of potentially confounding spatial and temporal variations unrelated to fire. We first determined the temporal changes in these variables in each watershed in each study by subtracting the 1995 prefire rates or concentrations from the 1996 postfire rates or concentrations for each pair of corresponding samples (i.e., each IMI class by sample plot by sample position combination). This gave us an estimate of the spatial and temporal variation at each sample point in each watershed independent of the effect of fire. Then, to estimate the actual net fire effect, we subtracted the 1995–1996 changes observed in samples from the unburned CONT watershed in each study site from the corresponding changes in the samples from the burned watersheds.

Data analysis.

All response variables were found to be normally distributed (PROC UNIVARIATE; SAS 1985), and were analyzed by analysis of variance using a nested, unbalanced design (PROC MIXED; SAS 1995). Where main effects were significant, least squares means were used to test differences among sites, watersheds within sites, and IMI classes. In addition, pairwise t-tests were used to determine if net changes observed in the burned plots were significantly different from zero. Exploratory path analysis (Amos 3.51: Smallwaters Corporation, Chicago, IL)., a form of structural modelling, was used to develop multiple regression models for the effects of the fires on C and N dynamics. We chose path analysis over more traditional multiple regression because the putatively-independent variables for our model (i.e. the soil, forest floor, and fire behavior parameters we measured) lacked true independence and covaried significantly, and because we felt both direct and indirect/interactive effects of soil and fire variables may have been important in determining the effects of fire on nutrient dynamics. Exploratory path analysis handles such data sets

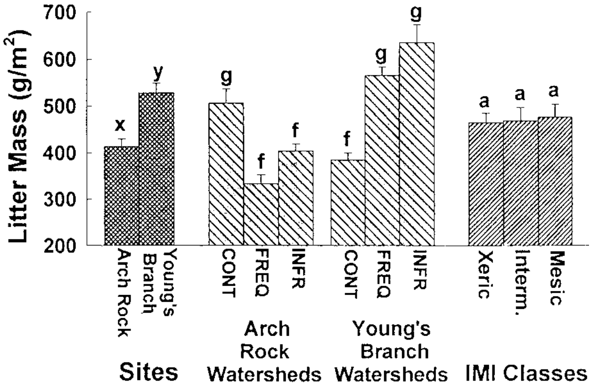

Figure 1. Prefire litter mass, soil pH, and total inorganic N in two southern Ohio forest sites. Means and standard errors of the means are given, and within a group (sites, watersheds withing sites, IMI classes), histogram bars with the same lower case letter were not significantly different at $p < 0.05$.

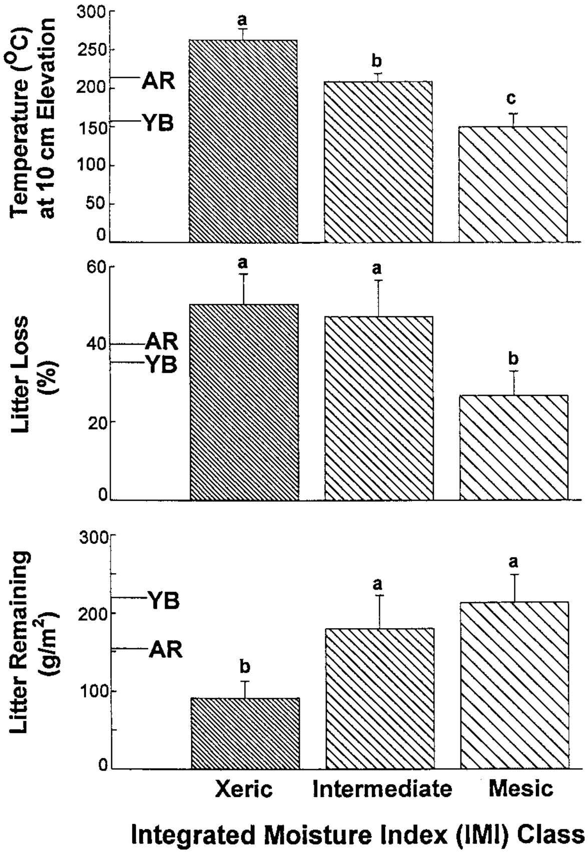

Figure 2. Fire temperature, proportion of litter consumed, and litter mass remaining after fire in two southern Ohio forest sites. The points indicated by initials and lines are the means for each site with IMI classes pooled: AR=Arch Rock, YB=Young's Branch. Histogram bars with the same lower case letter were not significantly different at $p < 0.05$.

better than do standard multiple regression methods (Arbuckle 1995).

Results

Prefire fuel conditions

Prior to the fires, litter mass was significantly greater at Young's Branch than at Arch Rock (Figure 1, Table 1). Prefire litter mass also varied significantly among watersheds in both sites (Figure 1; Table 1). However, there were no significant differences in prefire litter mass among IMI classes (Table 1). There was also a significant interactive effect of forest site and IMI class on prefire litter mass (Table 1). At Arch Rock, prefire litter mass was significantly greater in xeric sites than in intermediate or mesic plots, whereas at Young's Branch the opposite was the case (data not shown).

Fire behavior

The full analysis of variance of fire temperature indicated a significant difference in fire temperature and mass of litter remaining after the fire between the Arch Rock fires and the Young's Branch fires (Table 1). In addition there were significant ($p < 0.05$) or marginally significant ($p < 0.10$) interactive effects of forest site and IMI class on fire temperature and the proportion of litter consumed. This led us to conclude that the fires at the two sites were significantly different in their behavior, and we decided to further analyze the behavior and impact of fire at each forest site separately.

Most of Young's Branch was burned on a cool, damp day. As a consequence, it was a low-intensity fire, with a mean temperature at 10 cm above the forest floor of 157 °C (±12). An average of 35.3% (±4.0) of the litter was consumed by the fire, leaving an average of 220 g/m^2 (±29) on the site. There was no significant variation due to IMI class for any of these parameters, nor were there any differences in fire behavior between the two burned watersheds at Young's Branch. Linear regression revealed no significant relationship between sample plot IMI score and fire temperature ($p < 0.667$), the proportion of litter consumed ($p < 0.232$), or the mass of litter remaining after the fire ($p < 0.818$).

Table 1. Analysis of variance in prefire litter mass and fire behavior variables. All full models were significant at $p < 0.001$, N=108.

Variance component	Prefire litter mass		Temperature at 10 cm elevation		Proportion of litter lost		Remaining litter mass	
	F	*p<*	*F*	*p<*	*F*	*p<*	*F*	*p<*
Site	33.70	0.001	9.86	0.003	2.57	0.114	9.64	0.003
IMI Class	0.23	0.797	7.13	0.002	7.31	0.112	1.95	0.151
Watersheds (Site)	21.20	0.001	1.26	0.291	22.42	0.001	7.44	0.002
Site X IMI class interaction	4.81	0.011	4.25	0.019	2.68	0.077	2.07	0.134

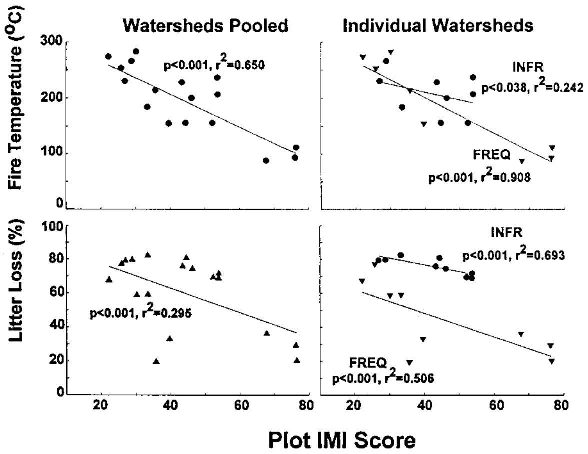

Figure 3. Linear regression of fire temperature and the proportion of litter consumed on the individual sample plot Integrated Moisture Index (IMI) scores for Arch Rock watersheds pooled (left) and for each Arch Rock watershed separately (right).

In contrast, the fires at Arch Rock were both more intense and more complex in behavior. The mean temperature of the Arch Rock fire at 10 cm above the forest floor of 210 °C (±11) was significantly greater than that of the Young's Branch fire, and the mean fire temperature at Arch Rock decreased with increasing IMI class (Figure 2). There was also a strong linear relationship between fire temperature and actual plot IMI score (Figure 3).

Although there was no significant difference in the proportion of litter consumed by the fire between sites (Arch Rock: 40.1% ±4.6 vs Young's Branch: 35.3% ±4.0), the proportion of litter consumed at Arch Rock was significantly less in mesic plots than in plots from the dry and intermediate IMI classes (Figure 2). There was a significant difference between forest sites in the absolute mass of litter remaining after the fire, with approximately 40% more litter mass remaining at Young's Branch than Arch Rock (Figure 2). Again, the amount of litter remaining after the fire did not vary among IMI classes at Young's Branch but did decrease with increasing IMI class at Arch Rock (Figure 2). There was a significant linear relationship between the mass of litter remaining and plot IMI score at Arch Rock ($p < 0.001$, r^2=284).

The complexity of the Arch Rock fires went farther than just the effects of topography and aspect (measured as IMI). The fire in INFR was both hotter overall and less heterogeneous (as measured by the strength of the temperature vs IMI score relationship) than was the fire in FREQ. There were also significant differences (at $p < 0.001$) between the two burned watersheds at Arch Rock in the proportion of litter consumed (FREQ: 44.5% ±4.9 vs INFR: 75.8% ±1.2) and in the mass of litter remaining after the fire (FREQ: 308 g/m^2 ±16 vs INFR: 162 g/m^2 ±25). Once again both the strength of the linear relationship between litter loss and plot IMI score (Figure 3) and the coefficient of variation in the proportion of litter consumed (FREQ: 46.7% vs INFR: 6.5%) support the view that the fire was more homogeneous at INFR than FREQ. Finally, there was a significant linear relationship between the mass of litter remaining after the fire and plot IMI score in Arch Rock FREQ but not in Arch Rock INFR (Figure 3).

In summary, based on fire behavior and the patterns of litter consumption, our prescribed burns produced two watersheds which experienced relatively cool, homogeneous fires (Young's Branch FREQ and INFR), one with intermediate temperature and a high degree of patchiness (Arch Rock FREQ) and one watershed that experienced a relatively intense but homogeneous fire (Arch Rock INFR).

Table 2. Analysis of variance results for measures of C and N dynamics one month after the 1996 fires. The full model is followed by the individual site models.

Variance component	Organic C (mg/g)		N mineralization (mgN/g soil)		Net nitrification (mgN/g soil)		Proportional Nitrification (%)	
	F	*p*<	*F*	*p*<	*F*	*p*<	*F*	*p*<
Full model (N=107)								
Site	11.79	0.001	0.46	0.500	7.18	0.009	4.40	0.039
Watersheds (Site)	1.45	0.224	0.28	0.892	0.32	0.864	0.23	0.923
IMI class	0.65	0.523	34.34	0.001	3.93	0.023	3.92	0.024
Site X IMI class	0.48	0.672	0.43	0.688	2.40	0.097	1.62	0.204
Arch rock (N=53)								
Watershed	0.75	0.480	0.30	0.744	0.82	0.448	1.53	0.228
IMI Class	5.60	0.007	17.09	0.001	1.47	0.241	1.34	0.272
Watershed X IMI Class	1.48	0.225	0.74	0.571	0.39	0.817	0.95	0.444
Young's Branch (N=54)								
Watershed	2.50	0.094	0.04	0.961	0.26	0.771	0.12	0.885
IMI class	2.28	0.114	17.75	0.001	3.24	0.049	2.67	0.081
Watershed X IMI class	0.77	0.553	0.65	0.630	2.19	0.086	2.10	0.097

C and N dynamics: One month postfire

The full, mixed model analysis of variance of postfire organic C content and N dynamics indicated strong effects of IMI class and significant differences between sites (Table 2). As a result of these site differences, we further analyzed the C and N dynamics of the two forest sites separately.

At Arch Rock, post-fire organic C content was significantly greater in soils from the xeric IMI class and least in soils from the intermediate IMI class (Table 2, Figure 4); whereas at Young's Branch there were no significant differences in postfire soil organic C among IMI classes (Table 2). Postfire N mineralization potentials increased with increasing IMI class in both sites (Table 2, Figure 4). There were no significant differences in postfire organic C content or N mineralization potential between burned and unburned watersheds in either study site (Table 2).

The full, mixed model analysis of variance in net and proportional nitrification potentials also indicated strong and significant site and IMI class effects (Table 2). Overall, the rates of net and proportional nitrification at Young's Branch were greater than those at Arch Rock (Young's Branch: 10.43 mg N/g $\pm$2.60 and 11.4% $\pm$3.0 vs Arch Rock: 2.71 mg N/g $\pm$0.92 and 0.3% $\pm$1.3).

There was also a significant effect of IMI class on net and proportional nitrification in the pooled data set (Table 2). However, there were also significant inter-

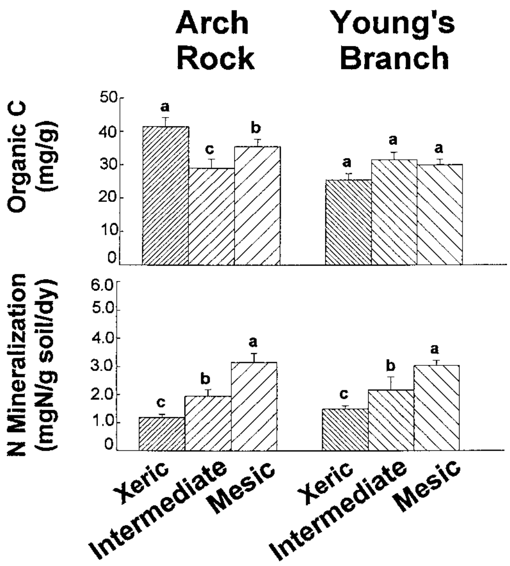

Figure 4. Organic C content and N mineralization rates one month after the 1996 fires in two southern Ohio watersheds in relation to Integrated Moisture Index (IMI) class. Within a site, histogram bars with the same lower case letter were not significantly different at $p < 0.05$.

active effects of site and IMI class, indicating that the influence of soil moisture was not consistent between the two sites. Postfire nitrification was significantly greater in soils from the mesic and intermediate IMI class plots than in soils from the xeric plots at Young's Branch, whereas no consistent pattern of postfire nitri-

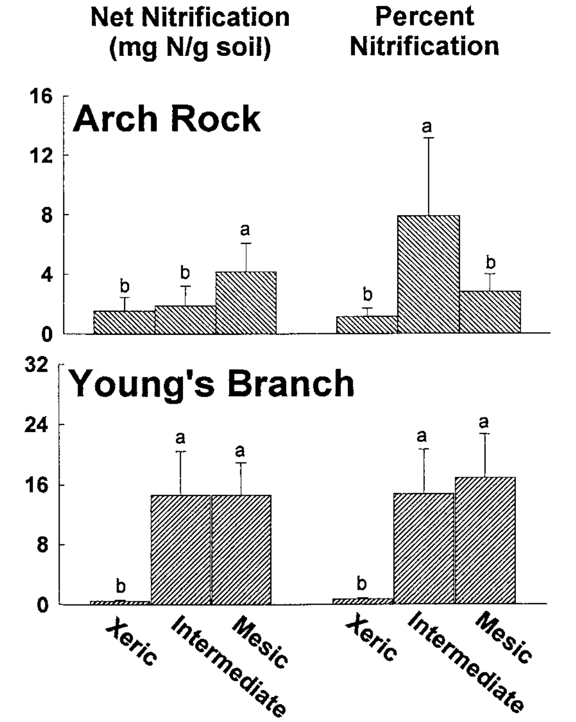

Figure 5. Net and proportional nitrification rates one month after the 1996 fires in two southern Ohio watersheds in relation to Integrated Moisture Index (IMI) class. Within a site, histogram bars with the same lower case letter were not significantly different at $p < 0.05$.

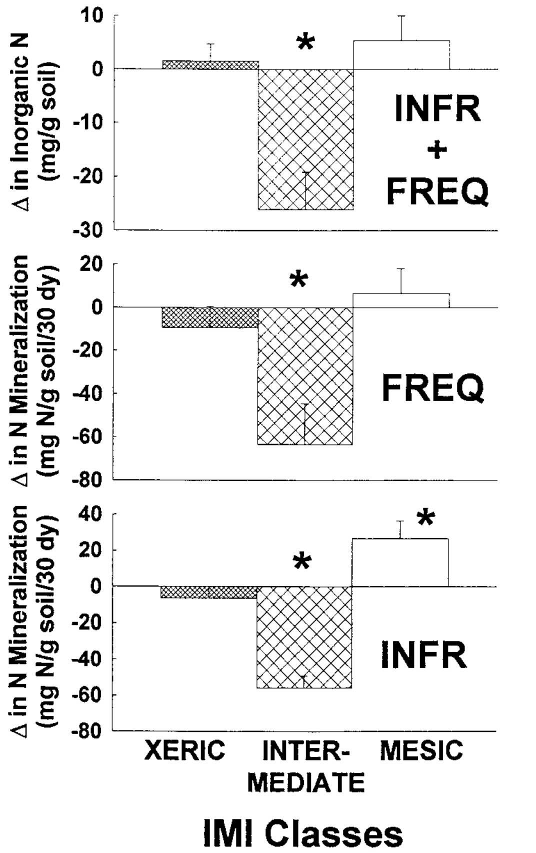

Figure 6. Net, fire-related changes in total inorganic N in the soil solution and N mineralization potential at Arch Rock in relation to Integrated Moisture Index (IMI) classes. Means which differed significantly from zero (=no net change) are indicated by *. N=17–18 for each watershed-by-IMI class combination.

fication among IMI classes was present at Arch Rock (Table 4, Figure 5).

There were no significant differences among watersheds within a site in total inorganic N in the soil solution (TIN) one month after the fire (Arch Rock: $p < 0.244$, Young's Branch, $p < 0.256$); however, there was a significant variation in TIN among IMI classes (Arch Rock: xeric 22.4 mg N/kg soil ±3.8, intermediate 29.7±4.1, mesic 41.2±7.2 and Young's Branch: xeric 15.1±1.2, intermediate 33.2±5.9, mesic 31.4±2.1).

Net fire effects

Once the temporal changes which occurred at the CONT watershed in each site were removed from those observed at the respective INFR and FREQ watersheds, the net fire effects became clear. At Arch Rock there were significant net fire-related reductions in TIN in the intermediate IMI class soils in both burned watersheds (Table 3, Figure 6). In addition, net N mineralization was significantly reduced by fire in the intermediate IMI class soils in both burned watersheds, and increased in the mesic IMI class soils from INFR (Table 3, Figure 6). There were no significant effects of fire on soil organic C content or net nitrification at Arch Rock (Table 3).

The net effects of fire at Young's Branch were quite different. Fire caused a significant increase in TIN in soils from xeric IMI class plots in both burned watersheds and in the intermediate IMI class soils from FREQ (Table 3, Figure 7). Fire also increased soil organic C content in the intermediate IMI class soils from both burned watersheds; in contrast, there was a small but statistically significant decrease in soil organic C content in mesic IMI class soils from FREQ but not INFR (Table 3, Figure 7). There were no significant, net effects of fire on N mineralization or net nitrification at Young's Branch (Table 3).

To determine which site variables or fire properties might be most responsible for the temporal and spa-

Table 3. Analysis of variance of net effects of fire on C and N dynamics.

Variance component	Total inorganic N (mg N/g soil)		Organic C (mg C/g soil)		N Mineralization (mgN/g soil)		Net Nitrification (mg N/g soil)	
	F	$p<$	F	$p<$	F	$p<$	F	$p<$
Arch Rock N=53								
Burned watershed	1.72	0.192	0.83	0.444	4.00	0.026	0.89	0.419
IMI class	6.39	0.004	1.02	0.371	20.94	0.001	1.47	0.242
Watershed X IMI class	1.48	0.226	1.74	0.159	4.85	0.003	1.10	0.371
Young's Branch N=54								
Burned watershed	1.40	0.257	1.34	0.274	0.86	0.212	0.27	0.763
IMI class	9.84	0.001	9.24	0.001	1.69	0.196	0.37	0.692
Watershed X IMI class	2.97	0.030	3.44	0.016	0.71	0.586	0.53	0.718

tial patterns of variation we observed, we proceeded to evaluate the effects of fire in relation to actual IMI value for each sample plot, fire intensity, and prefire site characteristics as a regression model. We felt this might improve the resolution of our analysis because: (1) actual IMI varied as much within an IMI class as it did between adjacent IMI classes, thus using IMI classes in an analysis of variance actually obscured the effect of slope position and microclimate, (2) the fire intensity, prefire fuel load, and organic C/N dynamics varied among watersheds, and (3) the spatial pattern of prefire site characteristics and actual fire behavior was highly patchy.

Regression modelling We utilized path analysis, a form of structural, causal modelling to clarify the roles of fire and landscape factors in C and N turnover. For this analysis, we utilized seven prefire site variables (plot IMI value, litter mass, soil pH, soil organic C content, TIN, N mineralization rate and nitrification rate) and two fire behavior variables (fire temperature and proportional litter loss) as independent variables and four postfire response variables (change in N mineralization, nitrification, organic C and TIN) as dependent variables. The initial phase of exploratory path analysis involves the assessment of the pattern and degree of covariance of the putatively independent variables. Of the 36 possible combinations of the site and fire variables to be used in the path model, 25 covaried significantly (at $p < 0.05$, CR>1.97). Thus, had we relied on traditional multiple regression models, we would have had to add 25 additional interaction terms to the model, thus consuming most of the available degrees of freedom. We then constructed a path model which included all 25 covariance links as well as putative causative links from the independent variables to the fire response variables.

The composite path model for all four fire effect parameters produced models whose strength ranged from r^2=0.256 for the change in N mineralization potential to r^2=0.684 for the change in net nitrification (Figure 8). The fire-related change in N mineralization was affected significantly only by site characteristics (prefire litter mass, $p < 0.05$ and IMI $p < 0.10$), although those site variables did covary with proximate fire variables (fire temperature and percent litter loss). In contrast, the fire-related change in soil organic C content was affected directly by both site factors (prefire organic C and soil pH) and fire behavior (percent litter loss) (Figure 8).

The changes in net nitrification due to fire effects were strongly affected by both site factors (prefire litter mass, prefire TIN, and prefire net nitrification) and by both prefire and net, fire-related changes in N mineralization potential (Figure 8). Finally, the net change in TIN was determined by both fire behavior and the effects of fire on other soil processes. The change in TIN was directly affected by percent litter loss and indirectly affected by fire temperature through its effect on litter loss. Similarly, the change in TIN was directly affected by prefire TIN and the fire-related changes in organic C content, N mineralization potential, and net nitrification. The fire-related change in TIN was also indirectly affected by prefire littermass, prefire TIN, prefire nitrification, and the postfire change in N mineralization through their effects on the change in nitrification rate (Figure 8).

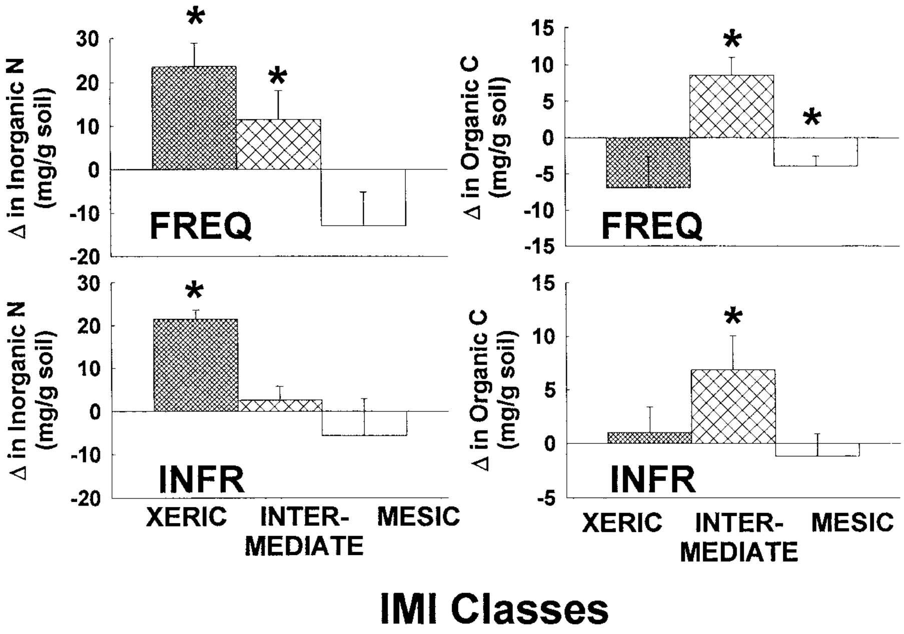

Figure 7. Net, fire-related changes in total inorganic N in the soil solution and soil organic C at Young's Branch in relation to Integrated Moisture Index (IMI) classes. Means which differed significantly from zero (=no net change) are indicated by *. N=18 for each watershed-by-IMI class combination.

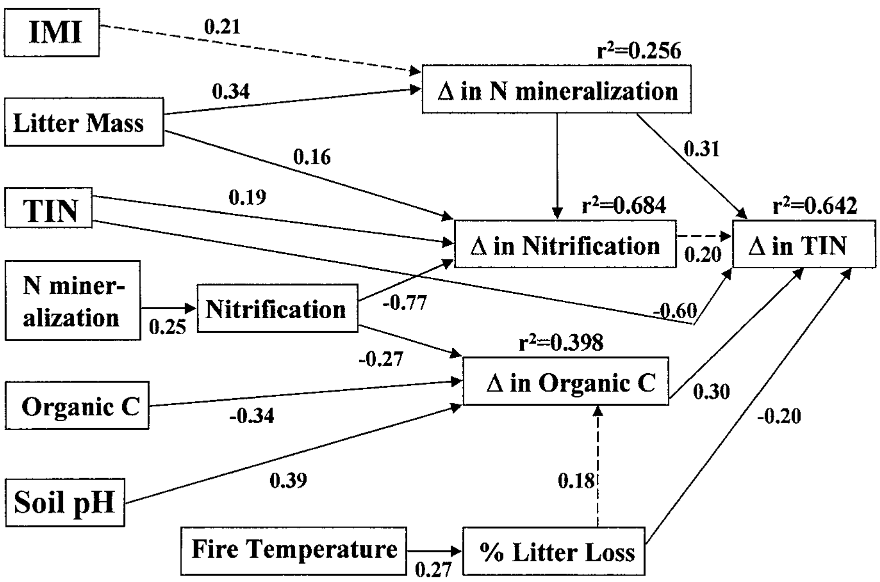

Figure 8. Path model for fire-induced changes in C and N dynamics in two Ohio forested landscapes. Regression linkages significant at $p < 0.05$ are indicated by solid lines and linkages significant at $0.05<p<0.10$ are indicated by dashed lines. Standardized regression coeffcients for regression linkages and squared multiple correlations for effect variables are indicated. Based on regression of implied covariances on actual sample covariances, the goodness of fit for the model was r^2=0.813, $p < 0.0001$.

Discussion

Although landscape-scale studies of fire frequency and dispersion are becoming more common (e.g., Turner and Romme 1994; Barrett et al. 1997) and our understanding of how landscape structure affects functional processes such as biogeochemical cycling is rapidly increasing (e.g., Schimel et al. 1985; Benning and Seastedt 1995), we believe the study presented here is one of the first to combine landscape dynamics, fire, and biogeochemical cycling in an integrated fashion and with an applied goal. To maximize resolution and minimize pseudoreplication, most long term studies of prescribed burning have been performed on sets of sites that are: (1) as uniform as possible within sites and (2) as well matched as possible among sites (e.g., Wells 1971; Boerner 1983; Vance and Henderson 1984; DeSelm et al. 1991). Although we also applied the latter criterion to our experimental design, the geomorphology and physiography of our study areas necessitated our utilizing sample 'units' that encompassed significant spatial variability at several scales. As the data presented here and in prior studies (Morris and Boerner 1998a; Decker et al. 1999) demonstrate, soil nutrient availability, rates and patterns of N cycling, and fuel mass all vary between forested study sites (Young's Branch > Arch Rock), along topographic gradients within study sites (represented in our studies by the Integrated Moisture Index [IMI] of Iverson et al. 1997), and in some cases, among experimental watershed units within study sites. Although attempting to resolve effects of low-intensity fires in such a landscape presents problems of resolution, variability, and precision, the goal of longterm ecosystem restoration in this landscape context left us no alternative. Fortunately, the power of the IMI as a synthetic variable has proven to be a powerful tool for integrating spatial and temporal effects across scales (Iverson et al. 1997; Morns and Boerner 1988a; Decker et al. 1999).

Dormant season prescribed burns, such as the ones we conducted, are typically less intense and have lesser impacts on the forest floor than fires which occur during the growing season (Wells 1971). In a survey of 44 prescribed burns in the southeast US, Hayward (1938) noted average flame heights of $\leq$1 m, surface soil temperatures which rarely and briefly exceeded 100–120 °C, and little indication of significant soil heating below 2.5 cm depth. More recent observations in oak-dominated forests (Barnes and Van Lear 1998; Brose and Van Lear 1998; Franklin et al. 1997; Grabner et al. 1997) reported comparable flame heights (Barnes and Van Lear 1998; Brose and Van Lear 1998; Franklin et al. 1997; Grabner et al. 1997) and temperatures (Franklin et al. 1997), although Cole reported considerably higher (ca. 40 °C) surface temperatures during a prescribed fire in oak woods in Indiana. In contrast, moderate-to-intense late summer fires, such as those that occur in chaparral, may have flame heights >3 m and temperatures between 600 °C and 800 °C (Debano et al. 1979). Thus, the prescribed burns we conducted, with mean temperatures of 150–300 °C at 10 cm above the litter surface, were consistent with observed dormant season, low-intensity prescribed burns in our region.

Although it is generally assumed that fire will increase spatial heterogeneity in soils properties (Raison 1979), there are few examples of this being quantified following prescribed burns in forest settings. The four fires we conducted were heterogeneous in temperature and fuel consumption, both between and within sites. Overall, the fire at Arch Rock INFR was the most intense and most homogeneous. In this watershed, 12 of the 18 sample plots experienced mean temperatures >200 °C (the temperature at which direct volatilization of N becomes significant), including the plot with the greatest mean fire temperature (371 °C). The fire at Arch Rock FREQ was, on the average, less intense but more heterogeneous. Although 16 of the 18 plots in this watershed experienced mean temperatures >200 °C, the differences in mean and maximum temperatures among plots from the three IMI classes were greater than those at Arch Rock FREQ. The two Young's Branch fires were less intense and more heterogeneous than those at Arch Rock. Mean fire temperatures were considerably lower, and only six of the 36 sample plots in the two Young's Branch fires experienced mean temperatures >200 °C. One indication of the spatial heterogeneity of the fires at Young's Branch was that four of the six plots that did experience >200 °C were in the intermediate IMI class and only two were in the xeric class.

The proportion of the litter layer consumed by these fires ranged from 28.5% to 80.4% and generally decreased with increasing moisture potential. Fuel moisture is apparently more important in explaining spatial variability of fire behavior of typical eastern hardwood forests than fuel loading (Fonteyn et al. 1984; Franklin et al. 1997). The proportion of the litter consumed in the dry IMI class plots was similar to that reported for a single prescribed burn in an oak-pine (*Quercus alba*, *Q. prinus*, *Q. stellata*, *Pinus rigida*) stand on well-drained, sandy soil in southern New Jersey (Boerner 1983) and to the losses of organic matter from the forest floor of pine plantations in South Carolina following periodic winter burning (Wells 1971). Cole et al. (1992) found a 75% reduction in litter of a (compositionally) mesic woodland dominated by *Quercus alba*, *Acer rubrum* and *Prunus serotina*, similar to our dry plots but far higher than our mesic plots. On an area basis, our fires consumed a range of 9.1–41.6 kg/ha of leaf litter fuel, again with striking differences between fires (e.g., Arch Rock INFR: 30.8 kg/ha, Arch Rock FREQ: 16.1 kg/ha). Differences among IMI classes within a fire site were small, as the prelitter mass and the proportion consumed varied with IMI in opposite directions.

Using litter N concentrations from multiyear studies of a nearby watershed with similar vegetation and geomorphology (Boerner 1984), we estimated the direct losses of N from the litter layer during the fire. Overall, N losses ranged from 0.13 kgN/ha in the Arch Rock FREQ fire to 0.32 kgN/ha in the Young's Branch INFR fire. At Arch Rock, N loss varied with IMI class; in the Arch Rock FREQ fire, N loss ranged from 0.19 kgN/ha in the xeric IMI class plots to 0.08 kgN/ha in the mesic plots. In contrast, there was no clear pattern of N loss with IMI class at Young's Branch. It is difficult to draw comparisons between the rates of mass and N loss we estimate for our fires and those estimated in other studies because most other studies examine changes during and over some time period after the fires, and thereby confound immediate losses due to fuel consumption and changes in mineralization that occur later in the season. However, these results were consistent with our hypothesis that a single, dormant season fire would not result in sufficient volatilization of N to offset the 15–20 kg N/ha/yr this region receives in atmospheric deposition (Morris and Boerner 1998b).

Although we observed significant changes in N availability, measured as total inorganic N (TIN), following our fires, the direction of change differed between forest sites and among IMI classes. At Arch Rock TIN decreased significantly in soils from intermediate IMI class plots, whereas at Young's Branch TIN increased in soils from xeric and some intermediate IMI class plots. In some ecosystems, N availability has been shown to increase after fire despite direct losses of organic N to volatilization and ash convection (Raison 1979; Boerner 1982). In sand plain grasslands (Dudley and Lajtha 1993), Norway spruce (*Picea abies*) plantations in Finland (Pietikainen and Fritze 1993), California chaparral (DeBano et al. 1979), Texas pine (*Pinus echinata, P. taeda*) forests (Webb et al. 1991) and ponderosa pine (*Pinus ponderosa*) forests in the western US (Wagle and Kitchen 1972), inorganic N availability is greater in burned than unburned soils, although the duration of the difference varies from months to several years, depending on ecosystem. In contrast, studies of both single and multiple prescribed burns in dry New Jersey pine (*Pinus rigida*) forests have failed to demonstrate any fire-induced change in soil N availability (Burns 1952; Boerner et al. 1986). Finally, in a study of 30 yr of annual and periodic burning of oak-hickory forests in Missouri, Vance and Henderson (1984) demonstrated a long term decrease in available N following both annual and periodic fires. Thus, the pattern and magnitude of change in soil inorganic N may be determined by complex interactions among site (landscape) factors and fire behavior, and may not, therefore, be predictable on an *a priori* basis in all cases.

The rates of N mineralization (or ammonification) and nitrification are commonly considered superior to instantaneous measures of N concentrations as a measure of N availability, because the former measure the rates at which the inorganic N pool is supplied with NH_4-N and NO_3-N, rather than just the magnitude of the pool at any one moment. At Arch Rock, the net effect of the fires was to reduce N mineralization without significant change in total soil organic C, suggesting that organic matter quality had decreased or that fire altered the microclimate in such a way as to reduce the rate at which N was mineralized from the organic matter. Some support for the latter explanation comes from two sources: (1) the strongest regression components for the net change in N mineralization we observed were local microclimate (as indicated by IMI) and prefire litter mass, and (2) prior studies of the effect of prescribed fire have suggested that changes in litter volume and microclimate have greater initial impact on N mineralization than do fire behavior measures (Webb et al. 1991). At Young's Branch where the fires were less intense, there was no significant short term change in N mineralization that could be attributed to the effects of fire, and soil organic C content increased significantly in some plots.

Using field and laboratory methods similar to ours, Vance and Henderson (1986) report N mineralization rates were greater in soils from control sites (2.44 mgN/kg soil/dy) than in soils from oak-hickory stands subjected to 30 yr of annual (1.68 mgN/kg soil/dy) or periodic (2.03 mgN/kg soil/dy) burns. They emphasized that the differences they observed were due more to long term changes in organic matter quality than to the quantity of organic C present, and that such differences might not be observed after a single burn, but were, rather, the cumulative effect of a lengthy burn regime.

The fires described here are the first of a planned series of 30 years of fires, with the FREQ watersheds being burned annually and the INFR watersheds every four years for the first 5 years for restoration purposes. This fire regime will then be followed by 25 years of fires every five years in both FREQ and INFR watersheds for maintenance purposes (further details in Sutherland 2000). One component of our overall restoration goal is to shift the longterm successional pathway away from those species which have been increasing dramatically over the last three decades (especially *Liriodendron tulipifera*, *Acer rubrum*, *A. saccharum*, and *Prunus serotina*) and towards those species which dominated these ecosystems prior to Euro-american settlement (especially *Quercus* spp. and *Carya* spp.). The former are all dependent on arbuscular mycorrhizae and are most dominant in ecosystems with large inorganic N pool sizes; in contrast, the latter are dependent on ectomycorrhizae and are typically dominant in ecosystems with low inorganic N availability and N mineralization rates (Vogt et al. 1991). If the results of these first fires at Arch Rock continue or increase after the subsequent burns we plan, the reductions in TIN and N mineralization rates we observed could facilitate such a shift in successional momentum. However, if the cumulative effects of multiple fires differs significantly from that of the first Arch Rock fires (as has been suggested by Vance and Henderson 1984) or if effects of subsequent fires are more consistent with what we observed after the first fires at Young's Branch, then prescribed fire may not promote *Quercus* and *Carya* species.

The use of prescribed fire as a tool in forest ecosystem restoration is increasing rapidly (Hardy and Arno 1996). From its inception in Louisiana in the 1920s to the present day, prescribed burning for ecological and silvicultural management has grown from a curious experiment to a tool used over 48,000 times a year (Riebold 1971). As we expand our attempts to restore ecosystems from single stands to the landscape level, and as we continue to recognize the key linkages among light, water, soil characteristics, and biotic responses, by necessity our goals, our analyses, and our interpretations will have to become more complex in order to capture the dynamics of heterogeneous landscapes. As the longer term changes in these ecosystems manifest themselves, we hope to be able to objectively assess the use of prescribed fire as a restoration tool in the Appalachian oak-hickory region.

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