SPATIAL, TEMPORAL, AND RESTORATION TREATMENT EFFECTS ON SOIL RESOURCES IN MIXED-OAK FORESTS OF SOUTHEASTERN OHIO

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ABSTRACT. —As part of a larger study of the use of fire and thinning to restore ecosystem function in eastern forests, we quantified spatial, temporal, and treatment-related variations in soil pH, available P, and N mineralization over two years in two southern Ohio mixed-oak forests (Zaleski State Forest and Raccoon Ecological Management Area/REMA). In each site, two watershed-scale treatment units of ~25 ha were sampled on a 50m grid for analysis of spatial autocorrelation and for assessment of temporal variability and treatment effects. Sampling occurred in summer 2000 (prior to treatment) and summer 2001 (after one unit in each site had been thinned+burned). Nutrient status differed more between sites than between treatment units within sites. Semivariance analysis of pretreatment samples demonstrated that pH, available P, and N mineralization were strongly structured spatially in all four treatment units (spatial structure $\geq$67%). There were no significant temporal differences in pH or N mineralization (2000 vs 2001) in control areas of the two sites; however, available P did decrease between 2000 and 2001. Neither patch size nor spatial structure changed significantly over this time. Analysis of covariance of post-treatment soil status indicated that the thinning+burning treatment resulted in significant increases in available P and soil pH at REMA but not Zaleski. Semivariance analysis indicated that the thinning+burning treatment increased patchiness (decreased patch size) in pH, but decreased patchiness in available P. Ecosystem restoration treatments affected both overall nutrient status and spatial structure in ways that should influence how plants and communities respond to such treatments.

Introduction

Forests dominated by oak (*Quercus* spp.) and hickory (*Carya* spp.) once covered much of the eastern United States. Prior to extensive alterations of this landscape by Euro-americans in the 19th century, these forests were subject to frequent, low intensity fires (Guyette and Cutter 1991, Sutherland 1997). These fires generally occurred during the dormant season and most were caused by human activities. Analysis of pollen and charcoal profiles suggests that fire return intervals were relatively constant over the last three millenia despite a shift from fewer large, regional fires to a larger number of smaller fires as Native American populations changed (Delcourt and Delcourt 1997). During the 1920's and 1930's fire suppression became both widespread and effective in this region, thus altering the fire regime under which these forests had developed. In addition, these ecosystems have been subjected to a variety of stresses and disturbances which operate on longer time scales, such as removal of a large proportion of standing biomass (e.g. periodic cutting, clearing for agriculture), weather-induced mortality (ice storms, drought), and chronic atmospheric deposition, especially of N. Thus, the mixed-oak and oak-hickory forest ecosystems that exist in this region today have been shaped by a combination of human and natural processes quite distinct from those that these ecosystems experienced for millenia prior to Euro-american settlement.

Restoration of oak forest ecosystems to conditions more similar to those that existed prior to widespread landscape alteration has begun over the last decade in a number of states. Approaches to restoration have utilized passive restoration (restricting public access), functional restoration (reintroduction of dormant season fire), structural restoration (modification of size/age-frequency distributions and species abundances by mechanical means), and, more recently, by a combination of structural and functional restoration.

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When prescribed burning is used for fuel and understory control in commercial conifer plantations, the plots to be burned are often modest in size and relatively homogeneous. Results obtained from burning one part of a larger plantation can then be extrapolated to estimate effects on larger areas by simple, linear scaling. In contrast, the forested landscapes of the Appalachian Mountains and surrounding plateaus which are the focus of current mixed-oak forest restoration efforts are often highly dissected, rugged, and heterogeneous in geomorphology, soils, and microclimate (e.g. Wolfe et al. 1949). In such complex landscapes, a specific understanding of the scale-dependency of various ecological processes and properties is necessary before results from small study plots can be scaled-up to the landscape level.

In earlier studies of the efficacy of single and multiple prescribed burns for restoration of mixed-oak forests in four sites in southern Ohio we demonstrated that fires effects on forest floor and soil C and nutrient relations exhibited strong scale dependencies ranging 10's of km (intersite differences) to 100's of m (aspect and elevation-related differences) (Boerner et al. 2000, Boerner et al. 2004). We have also demonstrated significant spatial heterogeneity and patterning at scales of 0.02-0.2 m in microbial biomass, soil organic matter and soil chemistry in unburned watersheds of this region (Morris and Boerner 1999, Decker et al. 1999); however, to date no data exist with which to determine if the effects of fire or other restoration treatments have significant spatial dependency at ranges between 2 m and 100 m, even though it is within this scale range that most routine sampling takes place. This led us to conduct a set of experiments to determine the degree to which spatial structure at ranges from a few m to a few hundred m was present in these study sites, how that spatial structure might be affected by a combined functional+structural restoration treatment, and how an understanding of spatial structure might affect the determination of the effect/efficacy of such treatments and the design of future restoration efforts.

The specific questions we sought to answer were:

- Do two study sites ~8 km apart vary more than contiguous watershed-scale treatment units within those sites?
- Are contiguous treatment units valid pseudoreplicates for large scale experiments?
- Do soil nutrient properties vary significantly between years in the absence of treatment or disturbance?
- How much do spatially-explicit factors contribute to between-unit and between-year variance?
- To what degree do ecosystem restoration treatments (i.e. reintroduction of low intensity fire and thinning to presettlement tree density) affect soil nutrient properties?
- What proportion of the treatment effect is the result of spatially-explicit factors?

Methods

Study Sites

The two study sites were located in Vinton County on the unglaciated Allegheny Plateau of southern Ohio. Each site was a block of 100-150 ha occupied by mixed-oak forests that developed following cutting for charcoal production 100-150 yr ago. One study site was located in the Raccoon Ecological Management Area (hereafter REMA) (39°11'N, 82°22'W), a research area managed cooperatively by MeadWestvaco Corporation and the U.S.D.A. Forest Service Northeastern Research Station. The second was located in Zaleski State Forest (39°21'N, 82°22'W), a site managed by the Ohio Department of Natural Resources, Division of Forestry. The two study sites were approximately 8 km apart.

The parent materials underlying the study sites were sandstones and shales of Pennsylvanian age. The soils were silt loams formed from colluvium and residuum, and were predominantly Alfisols (Boerner and Sutherland 2003). The climate of the region is cool, temperate and continental with mean annual temperature and precipitation of 11.3 C and 1024 mm (Sutherland et al. 2003). Microclimatic gradients generated by the steep, dissected topography of the region cause S, SW and W facing slopes to be drier and warmer than NW, N and E facing slopes (Wolfe et al. 1949).

A 50 m grid was established within each treatment unit using a random starting point. All grid points were GPS-located and permanently marked. For this study only two of the treatment units within each study area were used. One of the two units within each study area was randomly assigned to be a control and the other to be given restoration treatment consisting of thinning from below to presettlement tree basal area (~16 m^2/ha from typical current basal area of 25-30 m^2/ha) in November-December 2000 and reintroduction of low intensity, dormant season fire (burned in April 2001). Details of the thinning are given by Yaussy (2001) and fire behavior is described by Iverson and Hutchinson (2002).

Field Methods

Soil samples of approximately 400 g fresh mass were taken of the top 15 cm (O_a+A horizon) at a random point within 1 m of each of the grid points in July of 2000 and 2001. This sampling intensity yielded N=63 and 69 per year for the two REMA treatment units and N=43 and 52 per year for the two Zaleski treatment units. 2001 samples were taken within 50 cm of the 2000 samples, and all samples were returned to the laboratory under refrigeration.

Laboratory Methods

Each soil sample was air dried and sieved to remove roots and particulate material >2mm. A subsample of approximately 15 g of soil was extracted with 0.5M K_2SO_4, and analyzed for NH_4^+ and NO_3^- using microtiter colorimetry (Hamilton and Sims 1995). Soil pH was determined in a 1:5 soil slurry of 0.01M $CaCl_2$ (Hendershot et al. 1993), and available P by the ascorbic acid method (Watanabe and Olsen 1965).

A second subsample of approximately 50 g was placed in an incubation chamber and artificial rainwater added to bring the soil up to 70% of field capacity. The soil samples were incubated for 27-29 days at 22-28 C. Every third day each soil sample was weighed and sufficient artificial rainwater (Lee and Weber 1979) added to bring the moisture content back to randomly chosen level within the range of 50-70% of field capacity (Morris and Boerner 1998). Laboratory incubations were chosen for use in this study because the manner in which the moisture regime of the incubating samples was maintained recreated the frequent fluctuations in soil moisture characteristic of the growing season in our ecosystems reasonably well (Morris and Boerner, 1998). 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 of the initial samples from that of the incubated samples.

Data Analysis

All response variables could be transformed to normality using a log transformation (PROC UNIVARIATE; SAS 1995). Analyses of variance and covariance (using pretreatment conditions as the covariate) were designed for each of the specific experimental questions listed above, and were accomplished using the GLM procedure of SAS (SAS 1995). Semivariance analysis was accomplished using GS+ (Gamma Design Software, Plainwell, MI 49080) using untransformed data, lag distances 0.8 of maximum, and both isotropic and anisotropic models. Spatial structure is reported as the proportion of nugget variance (C_o) + structural variance (C) accounted for by structural variance. The range estimate reported is the direct range estimate (A_o) for linear/sill and spherical models and 1/3 of the estimated range ($3A_o$) for exponential models. Only best fit models are reported here, and point kriging based on the best fit model was used to interpolate between grid points and visualize spatial pattern.

Results

Partitioning of variance within and between sites indicated that potentially limiting soil properties varied significantly between sites (i.e. REMA vs Zaleski) but not between treatment units within each of the sites (table 1). During the pretreatment year (2000) soil properties varied little between the two treatment units within each study site (table 1).

Table 1. —Analysis of variance comparing variations between study sites ~8 km apart to those between treatment units within a site. Overall N=227, N=132 for REMA, and N=95 for Zaleski.

	Between sites	Between units within sites
Soil pH	F=16.11, p<0.001	F=0.88, p<0.417
Available P	F=7.85, p<0.006	F=1.65, p<0.195
N mineralization	F=7.38, p<0.008	F=0.35, p<0.704
	Between units at REMA	Between units at Zaleski
Soil pH	F=1.23, p<0.270	F=0.37, p<0.544
Available P	F=1.14, p<0.289	F=3.80, p<0.066
N mineralization	F=0.61, p<0.437	F=0.01, p<0.907

Semivariance analysis indicated that 73-80% of the variation in soil pH, available P, and N mineralization rate in REMA soils in 2000 was attributable to spatial structure, and this differed little between units within the REMA study site (table 2). At REMA, pH, available P, and N mineralization reached their maximum in the northern portion of the treatment unit (fig. 1); however the actual positions of maxima in space differed among soil parameters within a unit. This was a common result across all treatment units and years we sampled. Patch size (defined by the maximum range of significant spatial autocorrelation) was greater in the REMA control unit (120->369m) than in the REMA thin+burn unit (54-98m) (table 2) and these differences in patch size were clearly apparent in the kriged maps (fig. 1).

At Zaleski, >67% of the variance in soil pH, available P, and N mineralization rate were attributable to spatial structure (table 2). The pattern in space and maximum patch size of soil pH and available P in pretreatment soils differed little between the control and the unit to be thinned and burned (fig. 2). Small, distinct patches of low and high N mineralization rate soils were present in the kriged maps of the Zaleski control but not the Zaleski thin+burn, and as a result patch size was smaller in the former than the latter (fig. 2, table 2).

In the absence of treatment and/or disturbance, soil pH and N mineralization rate did not differ significantly between 2000 and 2001 samplings in control units in the two sites (REMA pH/2000: 3.89 $\pm$ 0.08 [std. error] vs pH/2001: 3.77 $\pm$ 0.06, p<0.161; N mineralization/2000: 12.51 $\pm$ 0.91 mgN/kg soil/dy vs N mineralization/2001: 11.33 $\pm$ 0.86, p<0.346, and Zaleski pH/2000: 3.63 $\pm$ 0.06 vs pH/2001 3.73 $\pm$ 0.05, p<0.220; N mineralization/2000: 10.46 $\pm$ 1.07 mgN/kg soil/dy vs 10.65 $\pm$ 1.03, p<0.899). In contrast, there was significantly less available P in 2001 than in 2000 in the control units in both sites. Available P decreased from 2000 to 2001 by 33% at Zaleski (2000: 231.4 $\pm$ 21.3 µgP/kg soil vs 2001: 154.3 $\pm$ 19.3, p<0.009) and 69% at REMA (2000: 348.1 $\pm$ 23.0 µgP/kg soil vs 2001: 109.0 $\pm$ 11.3, p<0.001).

The proportion of total variance attributable to spatial structure and the maximum patch size of soil pH and N mineralization differed little from 2000 to 2001 in the control units at the two sites, with the sole exception of pH patch size at REMA (figs.1 and 3, table 2). The latter difference in patch size was the result of the few, scattered patches of relatively high or low pH scattered within the larger matrix of intermediate pH in that unit. The decrease in available P from 2000 to 2001 noted earlier was apparent in the kriged maps, but neither the degree of spatial structure nor the patch size changed over that time in either site (table 2).

Analysis of covariance of the effect of the thinning+burning treatments at the two sites (using pretreatment conditions as covariates) indicated that only soil pH was affected by treatment alone (table 3). There were significant interactive effects of treatment and study site on both soil pH and available P, whereas N mineralization rate was not affected significantly by the thin+burn treatment

Table 2.—Semivariance analysis of soil parameters in samples taken in 2000 (pre-treatment) and 2001 (posttreatment) in two treatment units within each of two forested sites in Ohio. N=63,69 for REMA and N=43,52 for Zaleski. Spatial structure is the percent of nugget semivariance + structural semivariance represented by structural semivariance only. Range is an estimate of patch size as defined by the maximum distance at which samples are spatially autocorrelated.

Site/Unit	Year	Model fit (r^2)	Spatial structure	Range (m)	Model form
Soil pH					
REMA/Control	2000	0.783	81.4%	>369	exponential
REMA/Thin+Burn	2000	0.462	85.7%	98	spherical
Zaleski/Control	2000	0.424	67.4%	183	exponential
Zaleski/Thin+Burn	2000	0.872	68.0%	181	spherical
REMA/Control	2001	0.070	79.5%	>369	linear
Zaleski/Control	2001	0.773	99.9%	195	spherical
REMA/Thin+Burn	2001	0.472	93.5%	131	exponential
Zaleski/Thin+Burn	2001	0.279	78.5%	41	exponential
Available P					
REMA/Control	2000	0.254	76.6%	120	linear/sill
REMA/Thin+Burn	2000	0.086	80.0%	54	exponential
Zaleski/Control	2000	0.820	99.9%	68	spherical
Zaleski/Thin+Burn	2000	0.239	99.9%	54	linear/sill
REMA/Control	2001	0.503	83.1%	116	exponential
Zaleski/Control	2001	0.313	92.5%	61	spherical
REMA/Thin+Burn	2001	0.967	65.6%	261	linear
Zaleski/Thin+Burn	2001	0.022	84.0%	50	spherical
N mineralization					
REMA/Control	2000	0.884	73.8%	>369	exponential
REMA/Thin+Burn	2000	0.621	73.3%	62	exponential
Zaleski/Control	2000	0.805	99.8%	147	spherical
Zaleski/Thin+Burn	2000	0.995	66.5%	357	spherical
REMA/Control	2001	0.920	79.9%	294	spherical
Zaleski/Control	2001	0.701	97.8%	138	spherical
REMA/Thin+Burn	2001	0.781	53.0%	374	spherical
Zaleski/Thin+Burn	2001	0.360	79.4%	69	exponential

Table 3.—Analysis of covariance of soil parameters in relation to study site, restoration treatment, and the interaction between the two, using pretreatment conditions as the covariates. N=227.

	Treatment	Site-by-treatment interaction
Soil pH	$F=4.56,\ p<0.034$	$F=5.61,\ p<0.019$
Available P	$F=1.87,\ p<0.174$	$F=12.12,\ p<0.001$
N mineralization	$F=1.99,\ p<0.166$	$F=0.66,\ p<0.419$

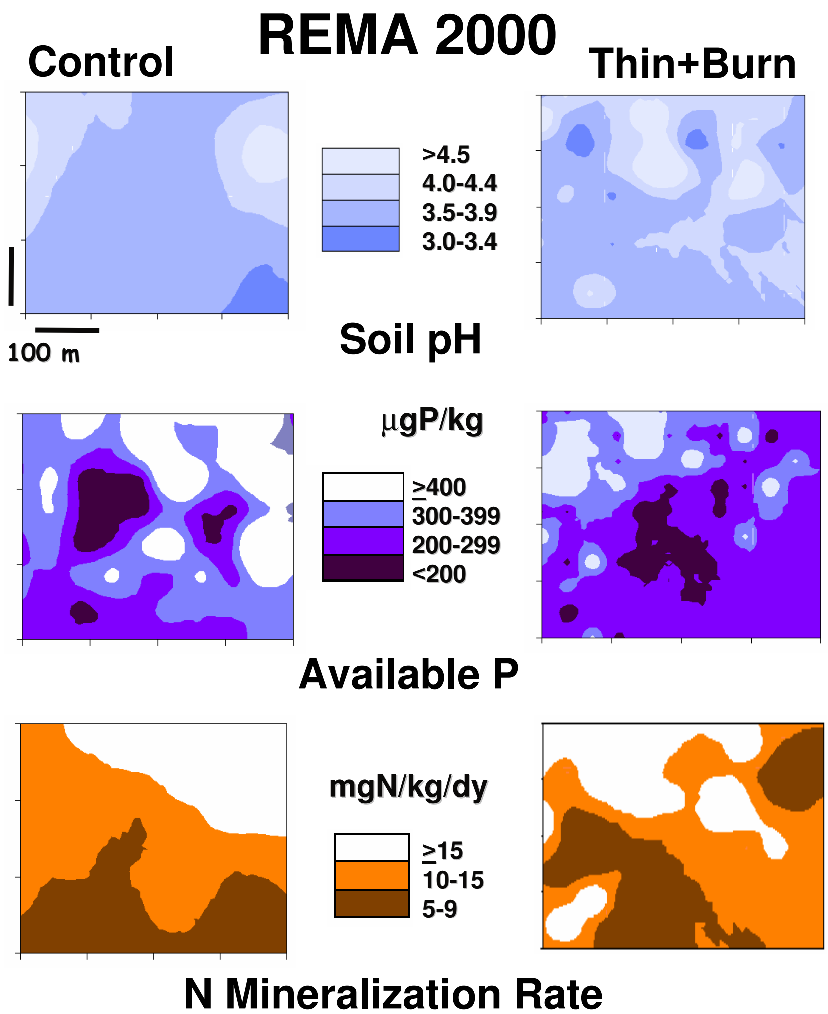

Figure 1.—Interpolated maps of soil parameters in the REMA Control and Thin+Burn units during the pretreatment year (2000). See Table 2 for spatial statistics associated with these kriged maps.

either overall or in relation to site (table 3). Overall, soil pH in the thin+burn units (3.85 ± 0.03) exceeded that of the control units (3.75 ± 0.01) significantly, and at REMA the difference of 0.24 pH units between the control (3.77 ± 0.06) and thin+burn (4.01 ± 0.07) units was significant at $p<0.001$. At REMA, available P in the treated unit (179.9 ± 20.3 µgP/kg soil) was 65% greater than that of the control unit (109.0 ± 11.3), whereas at Zaleski available P in the treated unit (110.0 ± 11.1) was 29% lower than that of the control (154.3 ± 19.3); the difference at REMA was significant ($p<0.006$) whereas the one at Zaleski was not ($p<0.195$).

Maximum pH and available P patch sizes increased from 2000 to 2001 by 25% and 475%, and the proportion of variance in available P attributable to spatial structure decreased by approximately 1/3 over that time (table 2). The increase in available P from 2000 to 2001 in the REMA treatment unit was apparent in the kriged maps as patches of moderate and high P appeared after treatment in what was a matrix of low soil P prior to treatment (figs. 1 and 3). Although the structural variance in soil pH and available P changed little from 2000 to 2001 in the treated unit at Zaleski, maximum patch size for pH appeared to decrease by approximately 77% (table 2). At Zaleski, smaller patches of relatively high soil P that were present prior to treatment were absent after treatment (figs. 2 and 4). The spatial structure of N mineralization changed little from 2000 to 2001 in either study site (table 2, figs. 2 and 4).

Discussion

The assessment of the success of ecosystem restoration prescriptions often focuses on determining whether predetermined, average conditions have been achieved. It is our premise that ecosystem

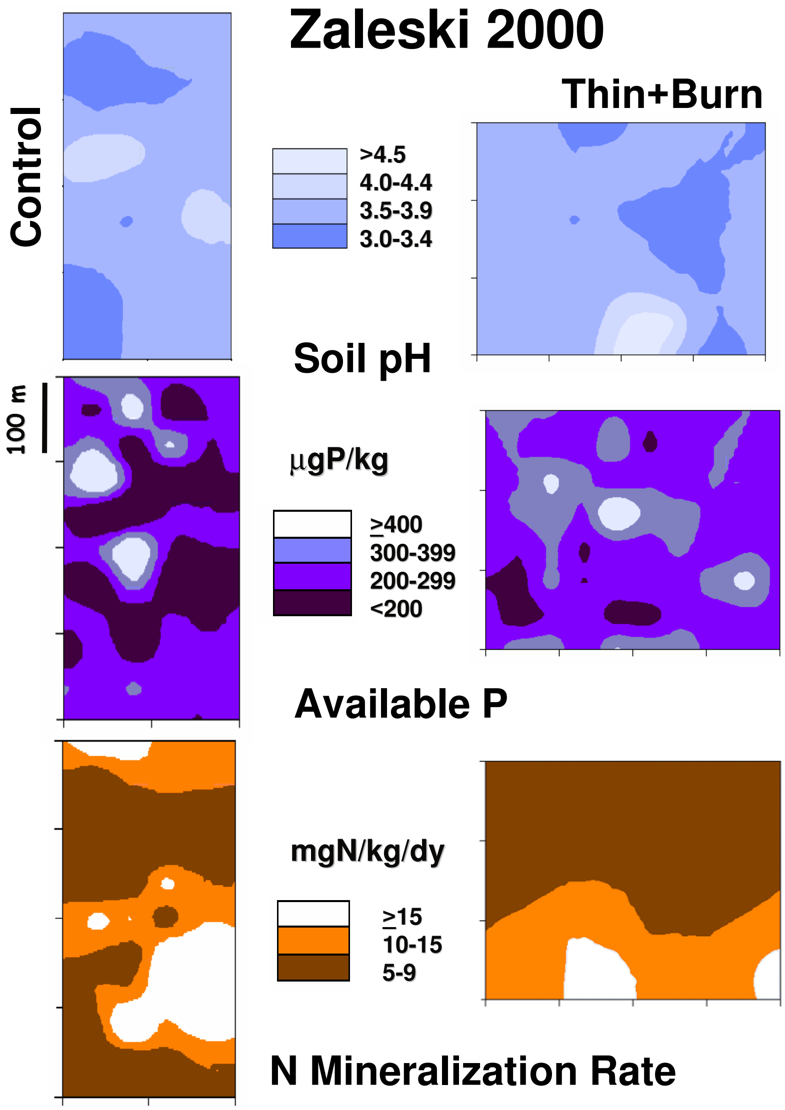

Figure 2.—Interpolated maps of soil parameters in the Zaleski Control and Thin+Burn units during the pretreatment year (2000). See Table 2 for spatial statistics associated with these kriged maps.

restoration at the landscape scale requires and additional understanding of, and an ability to measure, spatial heterogeneity at multiple scales. Even in regions where tree diversity and topographic heterogeneity are less than in our Ohio landscape, distinct spatial structure in tree stem and age distributions were common prior to Euro-american intervention (Selmants et al. 2003), and based on the well-documented effects of individual trees on soil properties near their bases (Boerner and Koslowsky 1989; Decker et al. 1999), it is reasonable to postulate that such spatial structuring existed in the forest floor and soil as well.

The first part of this study was designed to determine what the spatial pattern of potentially-limiting soil resources was in this heterogeneous landscape prior to the onset of restoration treatments, and to determine how stable this spatial pattern was in time and space. As has been the case in our prior studies in this region (e.g. Boerner et al. 2003, 2004, Boerner and Brinkman 2003), we found that mean soil properties varied much more between study sites separated by km than between neighboring watersheds. Similarities between/among neighboring watershed-scale treatment units in this region (this study, Boerner et al. 2003, Boerner and Sutherland 2003) lend strong validity to the use of such large land units as replicates (or pseudoreplicates) for experimentation, while the significant differences between sites just a few km apart (Boerner et al. 2003) argue strongly for approaching such experiments using complete block or Latin Square designs.

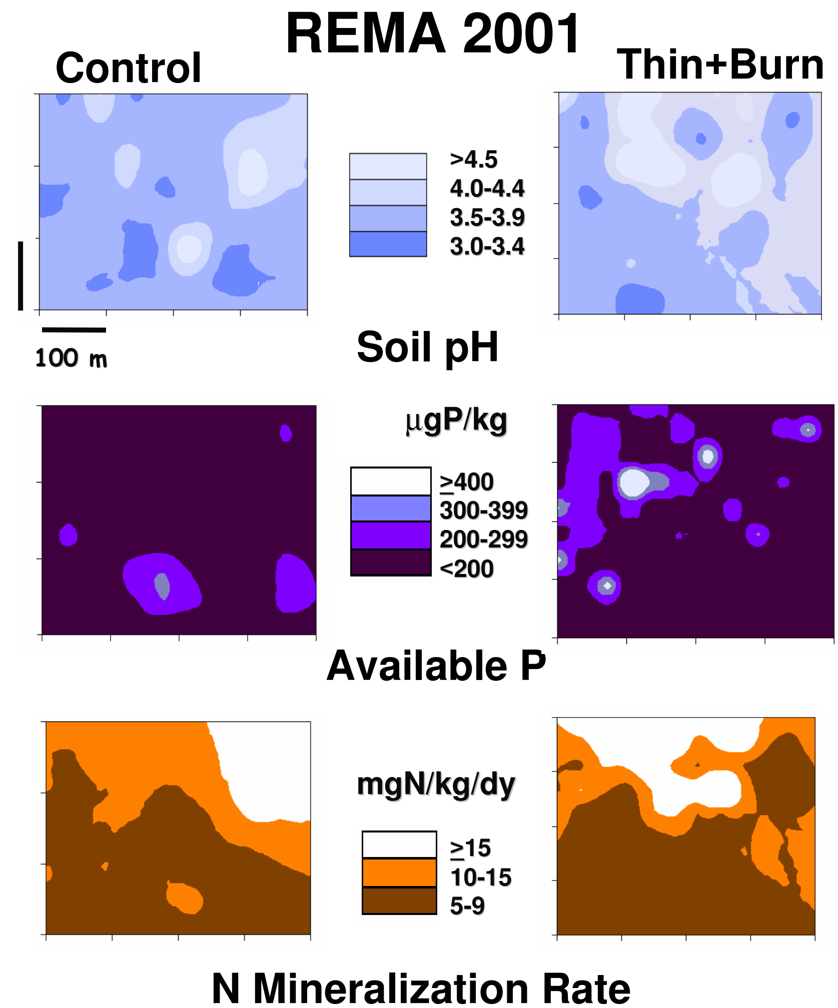

Figure 3.—Interpolated maps of soil parameters in REMA Control and Thin+Burn units during 2001. See Table 2 for spatial statistics associated with these kriged maps.

The soil properties we chose for measurement all exhibited strong spatial structure at ranges of 15-360 m across the landscape, with at least two-thirds of the total variance among samples taken in each treatment unit of ~25 ha being attributable to spatial structure. The patch size at which this spatial structure occurred (defined as the maximum range of significant spatial autocorrelation among samples) varied by a factor of 2-6 among soil parameters between the two treatment units at REMA, and by a factor of approximately 2.4 for N mineralization at Zaleski. At REMA, apparent patch size was always greater in the control unit than in the unit to be thinned and burned, and this difference correlates well with the geomorphology of the two units. Approximately half the control unit is occupied by a relatively flat area classified as xeric in the Integrated Moisture Index (IMI) system of Iverson et al. (1997), whereas the unit to be thinned and burned exhibits a more equitable distribution of relatively xeric, intermediate, and relatively mesic IMI areas. Thus the geomorphology of the control is characterized by fewer and larger patches than is the treatment unit, and this carried over into the soil properties we quantified. In contrast, the difference in patch size in N mineralization at Zaleski was due to the presence of a few very small but distinct patches of relatively higher or low activity nested within larger areas of intermediate activity being present in the control unit but not the treatment unit. These small, unique patches do not correlate well with geomorphological features and are more likely the result of finer-scale variations in organic matter deposition or content, as has also been documented in other forest types (e.g. Boerner and Koslowsky 1989, Bruckner et al. 1999).

The relatively low range in soil pH (3.6-4.4) we observed was notable, as alfisols and inceptisols formed on these parent materials are often typified as having pH of 4.0-5.0. To some degree, the

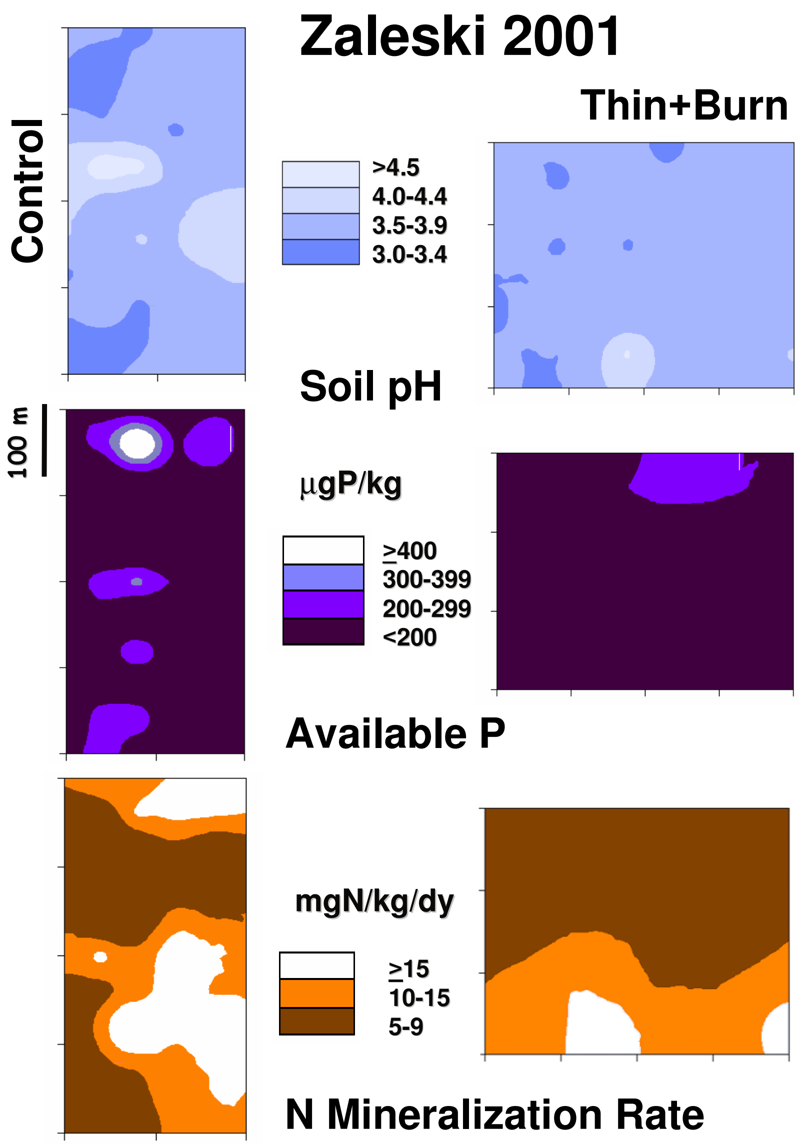

Figure 4.—Interpolated maps of soil parameters in Zaleski Control and Thin+Burn units during 2001. See Table 2 for spatial statistics associated with these kriged maps.

relatively low pH we observed may have been a byproduct of differences in methodology, as pH values in $CaCl_2$ (as we used) can be 0.5-0.7 pH unit lower than those recorded in water. However, enhanced acidification induced by chronic deposition of N in precipitation and resultant enhanced rates of nitrification may also be a contributing factor. For example, Boerner and Brinkman (2004) report similar soil pH, soluble Al concentrations of 100-400 mg/kg soil, and molar Ca:Al ratios of <1 in soils of sites located contiguous to REMA in which nitrification rates have increased 4-10 fold over the last decade.

We observed little interannual variation in soil pH or N mineralization in the control treatment units, though available P avail did decrease in both control units from 2000 to 2001. Although the magnitude of the year-to-year variation in available P was statistically significant and appeared relatively large on a proportional basis, it represented an average change of less than 0.10-0.25 mgP/kg soil, a degree of temporal variability that was small compared to spatial variability within a treatment unit. In addition, spatial structure and patch size of soil pH, available P, and N mineralization differed little between years. If anything, we were surprised by how little interannual variation in soil nutrient status we observed given how variable our summer weather often is from year to year.

Studies of spatial dependency in soil biological and chemical properties have reported a broad range of patch sizes/spatial autocorrelation ranges. Among those soil properties reported to exhibit spatial dependency at scales <1 m are soil pH in a Norway Spruce (*Picea abies*) forest (Bruckner et al. 1999), soil pH in Norway spruce, sugar maple (*Acer saccharum*), and red pine (*Pinus resinosa*) forests (Riha et al. 1986), soil pH, N, and K in a sugar maple forest (Lechowicz and Bell), soil organic C in Ohio beech-maple (*Fagus grandifolia-Acer saccharum*) forests and successional old fields (Boerner et al. 1998), and both nitrification and denitrification in a white fir (*Abies alba*) forest (Lensi et al. 1991). Spatial patterning at scales of 1-10 m are reported for inorganic N and available P in Ohio beech-maple forests and successional old fields (Boerner et al. 1998), inorganic N and organic C in a Douglas-fir (*Pseudotsuga menziesii*) forest (Antos et al. 2003), and C mineralization and soil moisture a red maple (*Acer rubrum*) forest and successional old field in Rhode Island (Görres et al. 1998). Finally, coherent spatial structure at coarser spatial scales have also been reported, including ranges of 10-20 m for P, Ca, Mg, and CEC in North Carolina forests (Palmer 1990), 10-20^{+} m for soil pH, P, and organic matter in an old field in Italy (Castrignano et al. 2000), >20 m for N mineralization in a Norway Spruce forest (Bruckner et al. 1999), 8-62 m for soil moisture in a slash pine (*Pinus elliotti*) plantation in South Carolina (Guo et al. 2002), up to 20 m for inorganic N in a Michigan successional field (Robertson 1987), and 13-42 m for soil pH N mineralization, and available P in a tropical dry forest in the West Indies (Gonzalez and Zak 1994). One must be cautious in attempting to synthesize these studies into a general model for hierarchic spatial dependency in soil resources as the diversity of study sites and, perhaps most importantly, differences in the sampling schemes and scales of possible spatial resolution differ so much among studies.

Although nested or hierarchic spatial scales are commonly postulated as being common in the field (e.g. Palmer 1990, Ettema and Wardle 2002, Franklin and Mills 2003), the detection of multiple scales of variation requires complex, hierarchic sampling designs that are uncommon in the literature. In most cases, the range at which spatial dependency can be and is reported is severely constrained by the sampling design to a single, relatively narrow range. In this study, we report observations of spatial dependency at ranges of 40-200 m for soil pH, 50-260 m for available P, and 14-375 m for N mineralization. The limitations of our sampling design limited us to detection of ranges from 10-400 m, and would thus miss entirely fine-scale spatial dependency based on single tree influences (e.g. Boerner and Koslowsky 1989, Bruckner et al. 1999) or patterns of coarse woody debris on the forest floor (e.g. Morris 1999) and coarse-scale variations based on differences in parent material among sites separated by km (Boerner et al. 2003, 2004). However, combining this study with those done earlier in our study sites at coarser (Decker et al. 1999, Boerner et al. 2000, Boerner et al. 2003, 2004, Boerner and Brinkman 2003) and finer scales (Decker et al. 1999, Morris and Boerner 1999, Morris 1999) allow us to establish the hierarchy of spatial scaling of soil properties we feel is necessary to fully evaluate the efficacy of broad-scale ecosystem restoration and management efforts.

The second portion of the study was designed to determine the initial effects of a combined structural and functional restoration treatment on soil resources, both overall and in relation to spatial structure. Overall, we found that the combined restoration treatment resulted in increased soil pH and available P at REMA, but not at Zaleski. Fire-induced increases in soil pH, available P, and base saturation have been reported commonly in eastern forests (review by Boerner 2000), and a similar increase in pH was observed following fires in studies in nearby watersheds (Boerner et al. 2004). Variations in this response among studies in the literature are the result of differences in initial base saturation, fire intensity, and the length of time since fire (Boerner 2000). Our intersite differences seem to be predominantly the result of differences in fire behavior, with the fire at the REMA site having been somewhat more intense than that at the Zaleski site (Iverson, unpublished data).

We observed no change in N mineralization rates after thinning+fire in these two sites. These results were consistent with those following fire only in neighboring study sites (Boerner et al. 2004), but clearly not with studies in a variety of other ecosystems which show increase in N availability after fire or fire+cutting. For example, fire-induced increases in TIN and/or N mineralization have been reported in *Pinus ponderosa* forests in western North America (Wagle and Kitchen 1972), mixed pine (*P. echinata* *and P. taeda*) forests in east Texas (Webb et al. 1991) and California chaparral (Debano et al. 1979). Knoepp and Swank (1993) reported increases in N mineralization that persisted at least two yr after cutting and burning, though the fires employed in that study were considerably more intense than those in our study sites. In a Douglas-fir forest in the Pacific Northwest, cutting and burning resulted in an initial increase in N availability, but it lasted less than one yr (Antos et al. 2003), and Phillips and Goh (1985) reported that the increases in NH_4 production were greater in a southern beech (*Nothofagus* spp.) site in New Zealand that was cut and burned than in a site that was only cut. One must use caution in extrapolating the effects of fire in one ecosystem to those in far different ones. For example, Vance and Henderson (1984) found that annual burning over 30 yrs in a Missouri oak forest reduced N mineralization and TIN to a greater extent than did periodic burns over the same period, and the sites studied by Vance and Henderson (1984) are more similar to our study sites than are the others cited as experiencing an increase in N mineralization after fire.

Our thinning and burning treatment tended to homogenize the soil nutrient distribution. We noted decreases in spatial structure, increases in patch size, and the loss of very small, distinct patches of differing nutrient availability. This result is consistent with those of Guo et al. (2002) who observed a loss of spatial structure in soil characteristics following cutting in a South Carolina slash pine stand. If the short term results we present here persist over a longer period, there may be consequences for community structure in these forests. Studies in southern forests have demonstrated that human-induced change in spatial structure of forests can affect ecological processes such as the spread of fungal diseases and insect pests (Perkins and Matlack 2002). We will continue to evaluate the effects of these treatments on spatial structure of soil resources through a second fire cycle in expectations of determining whether the changes we documented in this study are persistent or ephemeral.

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