RESEARCH ARTICLE

Initial Vegetation Response to Prescribed Fire in Some Oak-Hickory Forests of the South Carolina Piedmont

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ABSTRACT: Three upland hardwood stands on first-order stream drainages in the Clemson University Experimental Forest near Clemson, South Carolina were burned in March 1999 and sampled for vegetation in May and June 1999. Comparison of burned and unburned plots showed significantly greater tree seedling abundance and an interactive effect between slope position and prescribed fire where the effect of fire on groundcover and tree seedlings differed by slope position. The increase in seedling abundance on burned plots was driven almost entirely by an 84-fold increase in yellow-poplar (*Liriodendron tulipifera* L.) seedlings. Red oak (*Quercus* spp.) abundance was greater on mid- and upper-slope positions on burned plots, but white oak (*Q. alba* L.) abundance was lower in burned plots at every slope position. Abundance of herbs, shrubs, and vines was nearly 10% greater on burned plots. Prescribed fire caused community-level changes for seedlings and groundcover vegetation. Seedling species diversity and evenness were relatively low at lower slope positions because of the copious germination of yellow-polar, but increased at higher slope positions. Conversely, groundcover diversity and evenness was relatively low on upper slope positions due to the abundant sprouting of lowbush blueberry (*Vaccinium vacillans* Torrey). Overall, groundcover diversity and evenness were higher for lower slope positions. Groundcover diversity was significantly altered by the interaction of prescribed fire and slope position. Results of this study suggest that one low-intensity prescribed fire in upland hardwood stands can alter tree regeneration patterns and composition of the groundcover layer at least temporarily, but its potential for the long-term promotion of oak-hickory regeneration and suppression of mesic hardwoods on these sites remains unclear. A program of periodic burns will be necessary to restore oak-hickory forests in fire-suppressed upland hardwood stands.

Index terms: groundcover composition, prescribed burns, tree regeneration, upland hardwoods

INTRODUCTION

Prescribed burning has been used for decades to achieve a variety of management objectives including seedbed preparation, control of competing vegetation, fuel reduction, mineral soil exposure, wildlife habitat improvement, and control of certain diseases and insects (Smith 1986). In recent years, prescribed fire has been used to facilitate oak regeneration (Brose et al. 1998, Van Lear and Brose 2002), reduce understory shrubs such as mountain-laurel (*Kalmia latifolia* L.) (Elliott et al. 1999), and mimic historic fire regimes for the restoration of both forest and grassland communities and ecosystems (Dickmann and Rollinger 1998).

Fire is recognized as a historically important part of the disturbance regime in the southern Appalachian Mountains (Van Lear and Waldrop 1989), Piedmont pine-grasslands (Barden 1997), and Coastal Plain (Garren 1943); but its historic importance in Piedmont hardwoods is relatively unknown, possibly because of the rapidity and thoroughness of European settlement in this region (Cowell 1995). Native American use of fire throughout the southeast was ubiquitous (Pyne et al. 1996), and was the primary cause of presettlement patches of prairie landscape across the Piedmont of North and South Carolina (Barden 1997).

In the eastern deciduous forest of North America, the dominance of oak forests can be attributed to many factors, but fire is central to the development of oak forests on upland sites (Abrams 1992). Most of today's mature Piedmont second-growth hardwood (oak-hickory) stands developed 80-100 years ago during a time when fire was a frequent visitor on the landscape. However, younger second-growth oak-hickory forests have developed during a period of fire exclusion (White and Wilds 1998), and thus with little or no fire influence. In classic southeastern Piedmont old-field succession (Oosting 1942), oaks and hickories replace pioneering pines in the absence of fires frequent enough to suppress them. Relatively few studies document successional processes in southern Piedmont oak forests (Abrams 1992). Unanswered questions remain: are Piedmont oak-hickory forests self-sustaining, do they represent a fire subclimax or, are they a late-successional seral stage ultimately replaced by more shade-tolerant species? Additionally, will the introduction of prescribed fire to hardwood forests of the Piedmont region perpetuate the dominance of an oak-hickory forest type?

It has been suggested that extant Piedmont oak-hickory forests are not self-sustaining and that beech-maple is the climatic climax forest type for the region (Haig 1938). Evidence for this can be seen in the abundance of relatively young shade-tolerant understory trees in current oak-hickory forests, but not enough data are available to verify this trend (White and White 1996). Oak-hickory forests may be self-sustaining when viewed from the appropriate spatial and temporal scales. On dry sites where mesic hardwoods invade less successfully, frequent fire may not be necessary for the perpetuation of an oak-hickory forest, but on mesic sites, lack of fire may allow shade-tolerant hardwoods to invade and eventually dominate the site (Abrams 1992). The result of fire exclusion is a contraction of dry-adapted fire-tolerant species with an expansion of shade-tolerant mesic species (White et al. 1998). Empirical surveys of pre-settlement Piedmont forests are rare but in the central Georgia Piedmont, Cowell (1995) reports forest composition was over 50% oaks (*Quercus* spp.), 27% pines (*Pinus* spp.), and 10% hickories (*Carya* spp). By comparison, mesophytes such as maples (*Acer* spp.), American beech (*Fagus grandifolia* Ehrhart), yellow-poplar, and blackgum (*Nyssa sylvatica* Marshall) together comprised less than 4% of all trees.

Prescribed fire is frequently used to suppress hardwood competition in pine stands but has been little used in hardwood silviculture because of concerns over bole damage and uncertainty of its benefits (Van Lear and Waldrop 1989). Recently there has been increased awareness of the potential benefits of prescribed fire to meet both management and ecological goals (Sharitz et al. 1992; Dickmann and Rollinger 1998). Reintroduction of fire into Piedmont hardwood stands may meet both management and ecological goals by promoting the recruitment of oaks and hickories, increasing groundcover diversity, and creating open woodland structure.

Our objective was to document short-term ground cover response in mature, oak-dominated stands in the Piedmont to an initial late winter prescribed fire. We hypothesized low-intensity late winter (early growing season) fire would: (1) improve recruitment of fire-tolerant oaks and hickories, (2) increase germination of yellow-poplar seeds stored in the duff, and (3) increase the diversity and abundance of the herbaceous layer in Piedmont oak-hickory stands.

DESCRIPTION OF THE STUDY AREA

Three oak-hickory stands, each bisected by a first-order stream, were selected for this study. Stands were located in the Issaqueena Lake Division of the Clemson University Experimental Forest (CUEF). CUEF is located in the lower foothills forest habitat region of the Piedmont physiographic province (Myers et al. 1986) adjacent to Clemson, South Carolina. The topography is moderately steep hills and slopes formed from mica schist, gneiss, and hornblende gneiss. Soils are of the Pacolet-Madison-Wilkes association and are acidic, clayey, kaolinitic, thermic typic Hapludults, with medium permeability (Smith and Hallbick 1979).

Similar to other portions of the Piedmont, CUEF land was farmed intensively until around 1900. Extensive soil erosion resulted from early Piedmont agricultural practices (Trimble 1974). The hardwood forests used in this study are natural secondary growth arising after field abandonment in the 1920s and 1930s. Forest structure is multilayered and dominated by oaks, hickories, yellow-poplar, and few relict pines in the overstory. Site 2 had a strong component of mountain-laurel (*Kalmia latifolia*) confined to streamside habitats, while on Site 3 this species was present roughly 30 m up the slope. Streams ran west to east on Site 1, and east to west on Sites 2 and 3. Aspect of study stands was either north or south.

METHODS

In each stand, four contiguous 100 m x 100 m (1 ha) plots were situated using a latin-square design, two on each side of a stream. The northwest (south facing) and southeast (north facing) plots were burned on Sites 1 and 2, and the southwest plot was burned on Site 3, where only the north aspect plots were sampled. Stands used in this study were burned on February 16, February 25, and March 23, 1999. Burns were of low intensity, with flame heights reaching an average of < 0.3 m at each site. Backing fires were ignited off the upper slopes to widen control lines, and strip head fires (maximum velocity of 2 m/minute) were lit at 10 m intervals to burn slopes. Backing fires burned more sheltered areas on lower slope positions. Due to light fuel loading (mostly hardwood leaf litter) and short residence time, little mineral soil was exposed. Vegetation sampling was not conducted prior to burning and the results of this study are based on the assumption that within-site conditions were homogenous prior to treatment. Post-burn sampling was conducted during late May and June of 1999.

Trees and saplings were sampled on each 1-ha treatment plot, using four 0.04 ha circular relevés, two randomly located on the upper half of the slope and two on the lower half. Basal area was calculated for each species from dbh measurements. Woody vegetation ≥ 10 cm dbh was considered a tree; saplings included all woody vegetation ≥ 0.5 m in height to < 10 cm dbh and seedlings (which includes sprouts) were defined as single-stemmed woody plants < 0.5 m in height. Groundcover (herbs, shrubs, and vines) was sampled by establishing three 100 m transects located 25 m (lower slope), 50 m (mid-slope), and 75 m (upper slope) from and parallel to streambeds. Species abundance was determined using stem counts of individual species in 25 1-m^2 quadrats located randomly along each of three 100-m transects within the 1-ha treatment plot. Nomenclature follows Radford et al. (1968).

Importance values (IV) for trees and saplings were calculated using the formula: IV = (relative density + relative basal area)/2. Groundcover importance values were calculated by converting stem counts into relative densities (number of individual stems of a species per plot/total stems per plot). Comparisons between burned and unburned plots for groundcover plants were made using various expressions of diversity: species richness (S), Shannon-Weiner's diversity index ($H' = -\Sigma p_i \ln(p_i)$) where p_i is the proportion of total abundance attributable to species i, and Pielou's evenness index ($J'=H'/H'_{max}$) where H'_{max} is the maximum level of diversity possible within the population (ln(S)). Factorial statistical tests were performed using analysis of variance (ANOVA) (SAS 1987) to examine differences in species diversity and abundance between burned and unburned plots by site and slope position.

RESULTS

The overstory stratum of the three upland hardwood forests was similar among sites. For all sites combined, 23 tree species were identified. Stand density averaged 292 trees/ha, with an average basal area of 15.2 m^2/ha. Composition of upslope and lower slope forests was similar, although there was variation in importance value ranks (Table 1). No post-fire mortality of tree-size woody vegetation was observed on any plot. Low-intensity prescribed burns consumed the litter layer from the forest floor and most of the aboveground biomass of vegetation in the herb layer, and top-killed most seedling-size woody vegetation.

Strip-head fires burning upslope caused leaf scorch on many saplings but did not appear to kill or permanently damage them. While sapling abundance on burned sites was just 61% of that on unburned sites,

Table 1. Relative Density and Relative Basal Area of the ten most abundant tree species on three Piedmont upland hardwood stands ranked by Importance Value (IV).

Species	Rel. Density	Rel. Basal Area	IV	Species	Rel. Density	Rel. Basal Area	IV
Upslope plots				Downslope plots			
Quercus coccinea	0.18	0.25	0.22	*Quercus alba*	0.38	0.43	0.41
Quercus alba	0.21	0.23	0.22	*Liriodendron tulipifera*	0.19	0.2	0.19
Quercus velutina	0.13	0.12	0.12	*Quercus coccinea*	0.07	0.08	0.08
Carya tomentosa	0.09	0.08	0.09	*Pinus echinata*	0.04	0.05	0.05
Liriodendron tulipifera	0.06	0.08	0.07	*Carya tomentosa*	0.04	0.04	0.04
Oxydendrum arboreum	0.07	0.04	0.05	*Carya glabra*	0.03	0.04	0.04
Pinus taeda	0.05	0.04	0.04	*Quercus falcata*	0.04	0.02	0.03
Quercus falcata	0.04	0.03	0.04	*Fagus grandifolia*	0.03	0.03	0.03
Quercus stellata	0.03	0.03	0.03	*Oxydendrum arboreum*	0.04	0.02	0.03
Prunus serotina	0.03	0.02	0.02	*Cornus florida*	0.04	0.02	0.03

Table 2. Differences in abundance of yellow-poplar, white oak (*Q. alba*), and red oak (includes *Q. coccinea*, *Q. falcata*, *Q. marilandica*, *Q. rubra*, and Q. velutina) seedlings by slope position; lower, middle, and upper on three upland hardwood stands.

	Yellow-poplar		White oak		Red oak	
	Unburned	Burned	Unburned	Burned	Unburned	Burned
Lower slope	7	970	184	118	45	46
SE	(0.8)	(68)	(22)	(15)	(5)	(1)
Middle slope	5	294	192	111	40	84
SE	(0.6)	(35)	(14)	(12)	(2)	(9)
Upper slope	4	83	128	71	55	100
SE	(0.6)	(3.7)	(15)	(8)	(5)	(5)

Table 3. Species diversity (H´), evenness (J´), abundance (N), richness (S), and dominance (Ds) for tree seedlings and groundcover on lower, middle, and upper slope positions in unburned and burned plots on three upland hardwood stands.

	Slope position														
	Lower					Middle					Upper				
Seedlings	H´	J´	N	S	Ds	H´	J´	N	S	Ds	H´	J´	N	S	Ds
Unburned	1.69	0.61	324	15	0.59	1.71	0.6	340	17	0.41	1.73	0.62	241	16	0.14
SE			(22)					(14)					(18)		
Burned	1.05	0.37	1288	17	0.34	1.7	0.6	567	17	0.34	2.22	0.82	393	15	0.31
SE			(68)					(29)					(12)		
Groundcover															
Burned	3.08	0.74	2300	64	0.07	2.5	0.66	1297	44	0.15	1.09	0.31	2537	35	0.61
SE			(108)					(35)					(126)		
Unburned	2.45	0.6	2871	60	0.2	2.18	0.57	1402	47	0.24	1.98	0.52	1342	45	0.3
SE			(115)					(30)					(45)		

this difference is not necessarily a result of burning. Relatively few dead saplings were noted in burned plots a few months after burning, and an above average density of yellow-poplar saplings was observed in the unburned, northeast plot on Site 2. Thus, while the prescribed fire caused some mortality of saplings in burned plots, the disparity in sapling abundance between burned and unburned plots was not solely attributable to fire mortality.

A total of 27 species of tree seedlings was recorded, with 24 species and 905 individuals in unburned plots and 21 species and 2,248 individuals in burned plots. Tree seedling diversity indices (H´) did not differ between burned and unburned treatments (1.554 vs. 1.826, respectively), nor did evenness (J´) (0.511 vs. 0.575); however, seedling abundance (N) did differ significantly ($P < 0.05$).

Burning increased abundance of yellow-poplar seedlings by over 80 fold (1347 on burned plots vs. 16 on unburned plots). Most yellow-poplar seedlings on burned plots were in the cotyledon stage. Oak seedlings/sprouts were much fewer in number on both burned and unburned plots but were taller with more leaf area than most yellow-poplar seedlings. Yellow-poplar regeneration occurred mostly on the lower slope positions, while oak seedlings were found at every slope position in each stand. Abundance of red oak seedlings was greater on burned sites (210 vs. 140 stems) but white oak seedling abundance was lower (300 vs. 504). Greater abundances of red oak seedlings were found on mid-slope and upslope positions on burned plots while red oak abundance on the lower slope position remained constant (Table 2).

Abundance of all seedling-size trees was greater in burned plots at every slope position (Table 3). The factorial effects model indicated a significant effect of slope position and an interactive effect between slope position and prescribed burning for seedling N. Although N varied only slightly by slope position on unburned sites, it differed significantly between burned and unburned plots and by slope position on burned plots. On unburned sites seedling H´ and J´ were notably constant among slope positions. Due primarily to the high rate of germination of yellow-poplar in burned plots, H´ and J´ were relatively low on mesic plots and increased with elevation on burned sites. On lower slope positions H´ and J´ were significantly lower in burned plots than analogous unburned plots. Middle slope positions had similar H´ and J´ values between treatments while

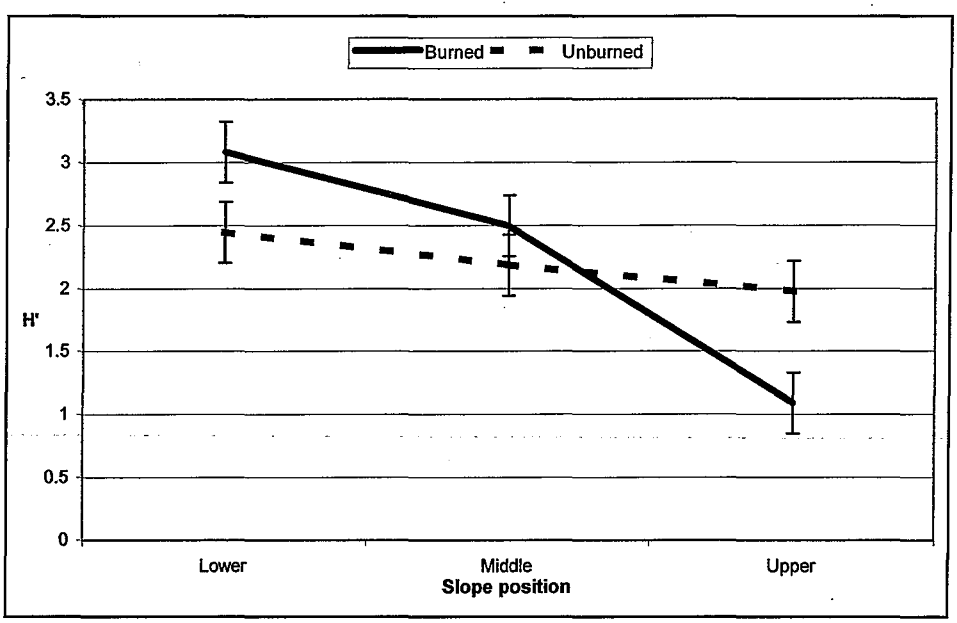

Figure 1. Interactive effect of slope position and prescribed burning on herb species diversity. Upper and lower slope differences were significant at ($P < 0.05$).

on more xeric upper slope positions H´ and J´ on burned plots were significantly higher than on unburned plots.

A total of 91 groundcover species were identified in this study, with 77 species and 5,615 stems on unburned plots and 72 species and 6,170 stems on burned plots (Table 4). Prescribed fire did not significantly affect groundcover N. Estimates of H´ and J´ were slightly, but not significantly, higher on unburned plots. The factorial effects model indicated groundcover H´ was significantly influenced by slope position and the interaction between slope position and prescribed fire (Table 3 and Figure 1). On burned sites, these parameters were significantly higher at lower slope positions and significantly lower at higher slope positions. However, there were no significant differences at middle slope positions between burned and unburned sites.

Many of the groundcover plants appeared to respond to fire with higher abundances in burned plots (Table 5). Plants with an apparent positive response were yellow star grass (*Hypoxis hirsute* (L.) Coville), lowbush blueberry (*Vaccinium vaccillans*), and violets (*Viola* spp.). Abundances of many other species in burned plots equaled their abundances in the unburned control plots indicating they had probably fully recovered and were not permanently removed by the prescribed burn.

Notably, many other species did not appear to respond positively to burning. Partridge-berry (*Mitchella repens* L.) abundance in burned plots was only 25% of its abundance in unburned plots; New York fern (*Thelypteris noveboracensis* L.) responded similarly (Table 5). Abundance of other mesophytic herbs was lower in burned plots, including Jack-in-the-pulpit (*Arisaema triphyllum* (L.) Schott) and rattlesnake plantain (*Goodyera pubescens* (Willd.) R. Brown) which were not found in burned plots despite their relative commonness in unburned plots.

Table 4. Abundance (N), species richness (S), diversity index (H´), evenness (J´), and density (D_s) of groundcover on burned and unburned treatments in Piedmont hardwood forests.

	Unburned	Burned
N	5615a	6170a
SE	(55)	(60)
S	77	72
H′	2.68a	2.55a
J′	0.613	0.598
D_s	0.128	0.185

Mountain-laurel appeared to have been top-killed by the fire in the burned plots. Backing fires burning through streamside

Table 5. Stem count abundance (N) of the 15 most common groundcover species in burned and unburned plots on three upland hardwood stands ranked by Importance Value (IV).

Species	N	IV	Species	N	IV
Unburned plots			Burned plots		
Vaccinium. vacillans	1337	0.24	*Vaccinium vacillans*	2432	0.39
Thelypteris noveboracensis	1150	0.2	*Vitis rotundifolia*	805	0.13
Vitis rotundifolia	804	0.14	*Polystichum acrostichoides*	331	0.05
Smilax glauca	245	0.04	*Thelypteris noveboracensis*	306	0.05
Mitchella repens	239	0.04	*Hypoxis hirsuta*	216	0.04
Panicum boscii	215	0.04	*Smilax glauca*	198	0.03
Polystichum acrostichoides	200	0.04	*Viola rostrata*	193	0.03
Viola rostrata	142	0.03	*Viola papilionacea*	174	0.03
Panicum commutatum	115	0.02	*Panicum commutatum*	165	0.03
Danthonia sericea	89	0.02	*Panicum boscii*	115	0.02
Hypoxis hirsuta	88	0.02	*Viola hastata*	113	0.02
Chimaphila maculata	82	0.01	*Viola pedata*	103	0.02
Stipa avenicea	79	0.01	*Stipa avenicea*	97	0.02
Arundinaria gigantea	77	0.01	*Festuca myuros*	90	0.01
Viola hastata	76	0.01	*Galium circaezans*	62	0.01

mountain-laurel on sites 2 and 3 scorched an estimated 5-10% of the crowns. Strip head fires burning through mountain-laurel on Site 3 caused leaf scorch on about 90% of the thicket canopy.

DISCUSSION

Prescribed fire in hardwood stands has traditionally received little study, although the literature on this topic is growing (i.e. Van Lear and Watt 1992, Barnes and Van Lear 1998, Brose et al. 1998, Kuddes-Fischer and Arthur 2002). Shelterwood cutting followed by burning has been used to regenerate oaks (Brose et al. 1998) and may facilitate ecosystem restoration (Dickmann and Rollinger 1998), but definitive evidence of the utility of fire can only be acquired over time. Results of the current study suggest prescribed fire can cause an immediate, short-term change in tree seedling emergence and composition of groundcover species in Piedmont hardwood forests. Whether or not the late winter 1999 burns in this study caused lasting changes to the structure and composition of the hardwood stands remains to be seen. In the absence of future prescribed burns, vegetation in these stands may revert to pre-burn assemblages.

Despite concerns that even low intensity spring burns can cause bole damage in hardwood stands (Wendel and Smith 1986), large high-quality trees on burned plots in this study appear unaffected. Each stand contained a relatively high percentage of large (> 35 cm dbh) thick-barked oaks in the overstory, which likely reduced bole damage from low-intensity surface fires used in this study.

Effects of prescribed fires on saplings were inconclusive; the greater density of saplings on unburned sites was due to an unusually high number of yellow-poplar saplings on one of the unburned sites. Low sapling mortality resulted following the low intensity and short residence time burns. Both factors are major determinants of mortality to woody vegetation during a burn (Van Lear and Waldrop 1989). Sapling-sized individuals surviving in burned plots included species normally sensitive to fire (i.e., 24 red maple (*A. rubrum*) saplings survived the burn as did several American beech, yellow-poplar, and American holly (*Ilex opaca* Aiton) saplings). However, post-burn mortality rates of fire-sensitive species often increase in the years following a fire (Barnes and Van Lear 1998). Fire-return intervals as long as 10 years may favor fire-resistant tree species and suggest episodic fire control regeneration of non-oak species (Harmon 1984; Arthur et al. 1998). Subsequent, recurring fires in these stands may do more to eliminate the non-oak, shade-tolerant species than did the single fires in winter of 1999.

The forest stratum most impacted by fire was the groundcover layer, especially for tree seedlings. The immediate effect of the burn was to shift tree seedling domination by oaks, especially white oak, to overwhelming domination by yellow-poplar, especially on lower slope positions. Seed of yellow-poplar is known to persist in the soil for many years (Clark and Boyce 1964) and germinate following fires (Shearin et al. 1972). Barnes and Van Lear (1998) noted that increased numbers of yellow-poplar following a single winter burn decreased dramatically with a successive winter burn. Although exogenous disturbance can increase yellow-poplar abundance in old-growth forests, Busing (1995) found this tree, like most shade-intolerant species, requires a canopy gap of at least 0.04 ha to become established from seed. Fire did not affect the overstory canopy on burned sites in this study. Therefore, the high yellow-poplar seedling abundance observed in burned plots in this study seems unlikely to result in increased abundance of this species as a canopy tree on these sites.

Single prescribed fires have often failed to promote increased abundance of oak advance regeneration (Johnson 1974; Teuke and Van Lear 1982; Wendel and Smith 1986) and are expected to have little effect on composition of other understory vegetation in southern Appalachian and Piedmont forests (Van Lear and Waldrop 1989). Higher abundance of red oak seedlings (which include seedling-sized sprouts) on burned plots in this study suggests prescribed fire may foster sprouting or germination and, consequently, the establishment of this species, especially on upslope sites.

The apparent decrease in abundance of white oak may be explained by its propensity to germinate in the fall (Harlow et al. 1991). Even low intensity burns can kill recently germinated oak seedlings (Barnes and Van Lear 1998; Brose et al. 1998). Abundance of white oak seedlings may increase following germination in the fall of 1999.

Density of groundcover plants is expected to be greater after fire (Ahlgren and Ahlgren 1960), and results of this study generally support that expectation. Species richness in this stratum was slightly lower on burned plots. Total density of groundcover plants was greater on burned plots and some species had greater abundances on burned sites. Plants demonstrating the greatest ability to increase following burning were those with the ability to sprout from roots or tubers (e.g., low-bush blueberry).

Mortality of mountain-laurel should temporarily increase abundance of herbaceous species and tree seedlings (Clinton et al. 1994) until it sprouts and reestablishes its dominance. It was noted that more groundcover appeared to be growing in burned mountain-laurel thickets in this study than is usually seen in unburned mountain-laurel stands. Other studies have noted temporary increases in abundance and diversity of herb layer vegetation after burning mountain-laurel thickets (Elliott et al. 1999).

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