PLANT SUCCESSION AND COMMUNITY RESTORATION FOLLOWING FELLING AND BURNING IN THE SOUTHERN APPALACHIAN MOUNTAINS

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ABSTRACT

Recent declines in the yellow pine component of pine-hardwood stands in the southern Appalachian Mountains has prompted managers to increase the use of fire as a silvicultural tool. The fell and burn treatment is designed to remove competing vegetation (hardwoods and mountain laurel [*Kalmia latifolia*]) to ensure successful establishment of planted eastern white pine (*Pinus strobus*). Two years after burning, mountain laurel had accumulated more biomass than any other species and accounted for 43% of total biomass in year 1 and 20% in year 2. By year 4, mountain laurel ranked fifth (8.9% of total) in total biomass among hardwood species behind Allegheny serviceberry (*Amalanchier arborea,* 14.3%), chestnut oak (*Quercus prinus,* 13.7%), red maple (*Acer rubrum,* 12.4%), and scarlet oak (*Q. coccinea,* 9.3%). Across sites, woody species richness ranged from 19–24 in year 1 and 14–22 in year 4. Species richness varied across sites and years, and there were substantial changes in the distribution of biomass among species.

The introduction of fire allowed the once dominant pitch pine (*P. rigida*) to successfully reestablish. On our sites, pine accounted for 25% of pretreatment stem density, but <1% and 2% in the first and fourth growing seasons after burning, respectively. However, in year 1, pines had increased in density 20-fold compared to pretreatment levels, and by year 4, had maintained a 17-fold increase compared to pretreatment. The use of fire in forest management has been the subject of considerable criticism. In light of current public concerns over the loss of critical or unique habitats, fire may gain public support for use as a restoration tool.

keywords: fell and burn, forest succession, prescribed fire, restoration, site preparation, southern Appalachians.

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INTRODUCTION

The oak-pine forest type, which occurs on xeric midslopes and ridges, is an important component of southern Appalachian forest ecosystems. The oak-pine type is characterized by scattered low-quality hardwoods and various yellow pine species (primarily pitch pine) in the overstory, a dense evergreen understory (mountain laurel and various *Vaccinium* species) (Vose and Swank 1993). In recent years, the pine component has been substantially reduced by drought and associated southern pine beetle (*Dendroctonus frontalis*) infestations (Smith 1991).

Historically, these stands regenerated following intense, stand-replacement wildfires (Vose et al. 1997). High-intensity fires are necessary for the establishment of yellow pine species, whose microsite requirements for seed germination include exposed conditions. Williams and Johnson (1992) attributed the lack of Table Mountain pine recruitment on oak-pine sites in southwestern Virginia to the absence of fire. Pitch pine and Table Mountain pine (*P. pungens*), both important components of this community, have evolved to be at least partially serotinous in their fruiting habits. Serotiny enables rapid colonization of burned-over sites, maintaining the importance of fire-dependent species in the community. Fire suppression has limited the role of anthropogenic and natural fires in perpetuating these stands, and in the absence of fire, they do not regenerate to commercially productive tree species (Barden and Woods 1973, 1976, Van Lear and Waldrop 1988) or regain the mix of pine and hardwood historically found on these sites. In the southern Appalachians, many of these stands are slowly degrading into poorly stocked stands with dense understories dominated by mountain laurel (Clinton et al. 1993, Swift et al. 1993, Vose and Swank 1993).

Over the past 10–20 years, some of these stands have been chainsaw felled, burned, and planted to eastern white pine in an attempt to convert these shrub-dominated ecosystems to more productive stands of mixed hardwood and white pine. The "fell and burn" treatment was developed by Abercrombie and Sims (1986) for pine-hardwood ecosystems in South Carolina. Its original intent was to serve as a low-cost, post-harvest site-preparation tool. However, many of the stands designated for this treatment have very low volumes of merchantable material (or none at all). In these cases, all standing material is felled and left on site. In most applications, the primary objective of the fell and burn treatment is to reduce the competitive influence of mountain laurel sufficiently to allow planted white pine and hardwood sprouts to become established. Prescribed fire rarely eliminates mountain laurel, so competition with the planted pines for light and perhaps other resources may intensify over time.

Moreover, in addition to restoring the commercial viability of these stands, this treatment may also be an effective means of restoring pre-existing (i.e., before fire suppression) levels of species composition and productivity (Clinton et al. 1993).

Our objectives were to: (1) investigate the successional sequence of xeric oak-pine sites following the fell and burn prescription and (2) to examine the success or failure of this treatment in restoring pre-existing community composition.

METHODS

Study Site Description

In 1990, 3 sites approximately 4 hectares in size were located in the Wayah Ranger District of the Nantahala National Forest in western North Carolina, denoted as Jacobs East (JE), Jacobs West (JW), and Devils Den (DD). Sites were selected as replicates based on pretreatment vegetation structure, topographic position, aspect, and soil type (Swift et al. 1993). All sites were cut in early summer and burned in early fall. No material was removed from any of the sites prior to burning. Pretreatment stand age was approximately 80 years (Swift et al. 1993), and average basal area and density (for stems >10 centimeters diameter at breast height [DBH]) were 14.8 square meters per hectare and 461 stems per hectare, respectively (Vose and Swank 1993). All vegetation was cut by chainsaw during the summer of 1990. Cut vegetation cured for 44, 50, and 55–89 days on DD, JW, and JE, respectively. Average fuel moisture (for all size classes) at the time of burning varied across sites (JE = 33%, JW = 28%, DD = 37%) (Swift et al. 1993, Vose and Swank 1993). All 3 sites were burned in September 1990 with fires of high intensity and low severity (Ottmar and Vihnanek 1991). Fire intensity relates to the upward heat pulse produced by the fire (Ryan and Noste 1985). Fire severity incorporates the downward heat pulse and is often a measure of forest floor consumption (Van Lear and Waldrop 1988). On these study sites, temperature in the forest floor and mineral soil (i.e., severity) ranged from 45°–60°C at 2.5–5 centimeters deep, and peak flame temperature (i.e., intensity) ranged from 625°–803°C (Ottmar and Vihnanek 1991). For further site description and burning characterization, see Swift et al. (1993).

Pretreatment Measurements

Woody Vegetation

Estimates of aboveground woody biomass for each site were made on 5 permanently marked 15 × 33-meter plots, distributed evenly across the sites to ensure representative sampling. Standing vegetation was grouped into 2 diameter classes (≤10.2 and >10.2 centimeters). Diameter was measured on stems >10.2 centimeters to the nearest 0.1 centimeter at DBH. For hardwoods, total-tree mass (stem, branch, and bark) was estimated from DBH using allometric equations of Clark and Schroeder (1986). For pines, total-tree mass was estimated from DBH with the allometric equation for loblolly pine (*Pinus taeda*) (Van Lear et al. 1986). Stems ≤10.2 centimeters DBH were measured on a 7.5 × 7.5-meter plot located in a randomly selected corner of each 15 × 33-meter plot. For stems with clear boles (no branching below DBH), DBH was measured with calipers or diameter tape. For all other stems ≤10.2 centimeters DBH, basal diameter (approximately 3 centimeters above ground level) was measured with calipers, and mass was estimated using allometric equations of Phillips (1981). Mountain laurel was measured at the base, and mass was estimated using the equations of Boring and Swank (1986). Additionally, mass of all woody plants <1 centimeter basal diameter was estimated using the equations for oak seedlings (Boring 1979). Mass for woody plants ≤1 centimeter basal diameter was estimated with equations from Boring (1982).

Herbaceous Vegetation

Herbaceous vegetation was inventoried in 4 1 × 1-meter plots randomly located within each 15 × 33-meter plot. Individuals were identified to species, and percent cover was estimated. In some cases it was difficult to identify individuals to species or even to genus, so the next highest order in the classification scheme was used.

Posttreatment Measurements

Plot size was reduced to 3 × 3 meters for woody vegetation in the posttreatment measurements, and an additional 3 3 × 3-meter plots were randomly located within each 15 × 33-meter plot. Plots were reinventoried in the summers of 1991, 1992, and 1994 (1, 2, and 4 years after treatment). Because all stems were ≤10 centimeters, only the 3 × 3-meter plots were used for wood biomass. On these plots, basal diameter was measured on each stem with calipers. Biomass (total aboveground dry weight of wood only) was estimated with site-specific allometric equations (Elliott and Clinton 1993), and with allometric equations developed by others for southern Appalachian species (Swank and Schreuder 1974, Phillips 1981, Boring and Swank 1986). In the 1994 measurement, it was noted whether an individual stem was part of a clump or was solitary. Due to the clumped nature and prolific sprouting exhibited by mountain laurel, clump area (square meters) estimates were determined based on the average of 2 diameter measurements and were converted to biomass using allometric relationships developed specifically for this study (B.D. Clinton, unpublished data). Herbaceous vegetation was inventoried using preburn methods. All species nomenclature follows Radford et al. (1968) and Little (1979).

Statistical Analysis

Plant community descriptors used were species richness (number of species), species evenness, and species diversity. The diversity index was estimated using the Shannon formula (Magurran 1988):

$$H' = -\sum p_i \ln p_i,$$

where,

H' is the index of diversity and
p_i is the probability of occurrence of the ith species.

Density was used as the measure of abundance in the index calculation. Estimates of species evenness were determined with the equation (Magurran 1988):

$$E' = H'/H_{max}$$

where,

E' is the estimate of evenness of species distributions,
H' is the estimate of species diversity, and
H_{max} is the maximum level of diversity possible within a given population and equals the ln (number of species).

Differences in biomass and basal diameter of species originating from sprouts versus non-sprout origin were determined by analyses of variance (SAS Institute Inc. 1987), and Duncan's multiple range test (SAS Institute Inc. 1987) was used to test for site differences. Significance levels for all tests were $\alpha = 0.05$.

Quantifying species diversity requires 2 important assumptions: (1) all individuals assigned to a specific class are equal, and (2) all species or classes are equally different (Peet 1974). These assumptions are often false. The first assumption ignores differences due to genetic heterogeneity, and the second ignores the varying functional attributes of the classes or species (e.g., nitrogen fixers, evergreens, root parasites, saprophytes). Although these assumptions may be violated, the indices can reveal subtle differences in composition, structure, and function between communities (Taylor 1978), which, as Magurran (1988) points out, is a critical attribute of any index of diversity if one is examining the impacts of environmental stresses on community diversity.

RESULTS AND DISCUSSION

Pretreatment Stand Structure and Composition

Prior to burning, these pine-hardwood stands had been highgraded and had incurred substantial overstory mortality, which resulted in low-quality residual stems of mixed age classes. The oldest overstory stems were >80 years (Swift et al. 1993). Even though these were relatively mature stands, mean DBH of the overstory was ≤20 centimeters, and basal area ranged from 9.5–18.8 square meters per hectare (Vose and Swank 1993), a reflection of the inherently low productivity of these sites.

Preburn overstory species composition was dominated by red oaks (scarlet oak, northern red oak [*Q. rubra*], black oak [*Q. velutina*], and blackjack oak [*Q. marilandica*]), white oaks (post oak [*Q. stellata*], chestnut oak, white oak [*Q. alba*]), and yellow pines (pitch pine, shortleaf pine [*Pinus echinata*], Virginia pine [*P. virginiana*]). The understories were dominated by mountain laurel, which accounted for 94% (22.3 metric tons per hectare) of the biomass and 48% (15,211 stems per hectare) of the density for stems ≤10 centimeters DBH, and contained a minor component of rosebay rhododendron (*Rhododendron maximum*) (Vose and Swank 1993). The red oaks had the highest proportion of total preburn woody biomass (32%), followed by mountain laurel (21%), the white oaks (19%), and yellow pines (15%). The remaining biomass was distributed primarily among red maple, sourwood (*Oxydendron arboreum*), Allegheny chinquapin (*Castanea pumila*), and *Vaccinium* species. For a complete characterization of pretreatment stand conditions, see Vose and Swank (1993).

Posttreatment Composition

Substantial changes in vegetation occurred over the 3 sampling years in this study, and biomass recovery was rapid for woody species. In terms of density, year 1 woody vegetation was dominated by sassafras (*Sassafras albidum*) (21%), followed by black gum (*Nyssa sylvatica*) (15%), and mountain laurel (13%). Sassafras continued to have the highest number of stems through year 2 but experienced high mortality prior to year 4, resulting in a 45% reduction in the number of stems of that species between years 2 and 4. By year 4, black gum had the highest density (22%). Most species experienced some reduction in density over time due to density-dependent mortality. This reduction may have also been a consequence of an extremely dry growing season in year 3 (30% rainfall deficit, unpublished data for Coweeta), which could have accelerated rates of mortality. For the period that spanned this dry growing season (years 2–4), overall density declined by 26%.

Black locust (*Robinia pseudoacacia*), an important early successional, nitrogen-fixing species in the southern Appalachians (Boring and Swank 1984), occurred predominantly as ramets (solitary root sprouts) on all sites. On sites comparable to ours, black locust is capable of attaining heights of up to 8 meters and basal diameters of 7.5 centimeters within a few years after clear-cutting (without burning) (Boring and Swank 1984). By year 4 in our study, black locust ranked ninth in woody biomass and had an average basal diameter of 2.2 centimeters, exceeded only by Allegheny chinquapin in basal diameter (2.5 centimeters). While the clear-cut sites studied by Boring and Swank (1984) were similar to ours in terms of pretreatment conditions, the use of fire on our sites was intended to delay the sprouting response of hardwoods. This treatment difference may have contributed to the weaker response of black locust to disturbance on our study sites compared to Boring and Swank (1984).

Species Richness and Diversity

Eighty-nine herbaceous species were identified over the 4-year sampling period; however, numbers of species varied by site and year. Table 1 lists 33 her-

Table 1. Summary of herbaceous species density (stems per square meter) and percent cover ranked in descending order based on density in 1994. Species listed represent 90% of total density.

Species	Preburn		1991		1992		1994	
	Density	Cover (%)	Density	Cover (%)	Density	Cover (%)	Density	Cover (%)
Vaccinium vacilans	3.0	2.0	3.9	1.1	3.8	1.0	12.5	2.2
Panicum commutatum	0.1	<0.1	0.2	<0.1	0.2	<0.1	4.5	1.3
Potentilla canadensis	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	4.2	0.4
Smilax glauca	2.2	0.6	2.5	0.5	2.2	0.6	3.6	1.4
Erigeron pulchellus	0	0	0	0	0	0	2.3	0.1
Panicum dichotomum	0.1	0.1	0.2	0.1	0.3	0.1	1.9	1.0
Uvularia pudica	0.7	0.2	0.7	0.2	0.8	0.2	1.8	0.3
Gnaphalium obtusifolium	0	0	0	0	0	0	1.5	0.1
Pteridium aquilinum	0.1	0.2	0.1	0.2	0.1	0.3	1.4	1.5
Scleria triglomerata	0	0	0	0	0	0	1.4	0.2
Gaylussacia ursina	0.5	0.4	0.6	0.5	0.6	0.4	1.3	0.4
Lespedeza intermedia	0	0	0	0	0	0	1.2	0.3
Carex spp.	0	0	0	0	0	0	1.2	0.4
Rubus occidentalis	0	0	0	0	0	0	1.2	1.0
Lysimachia quadrifolia	0.2	0.1	0.2	0.1	0.1	0.1	1.2	0.4
Scleria pauciflora	0	0	0	0	0	0	1.1	0.1
Galax aphylla	4.0	1.1	4.5	1.3	4.8	1.2	1.0	0.1
Lespedeza hirta	0	0	0	0	0	0	1	0.7
Viola palmata	0	0	0	0	0	0	1	0.1
Helianthus microcephalus	0	0	0	0	0	0	1	0.4
Polygala curtissii	0	0	0	0	0	0	0.9	<0.1
Coreopsis major	0.1	0.1	0.2	0.1	0.2	0.1	0.9	0.3
Lespedeza repens	0	0	0	0	<0.1	<0.1	0.8	0.4
Baptisia tinctoria	0.1	0.1	0.1	0.1	0.1	<0.1	0.7	0.7
Solidago odora	0	0	0	0	0	0	0.7	0.2
Pyrularia pubera	1.8	2.6	2.3	2.4	2.2	2.2	0.7	0.4
Sorghastrum nutans	0	0	0	0	0	0	0.7	0.2
Iris verna	0.2	<0.1	0.2	<0.1	0.3	0.1	0.6	0.2
Hypericum stragalum	0	0	0	0	0	0	0.5	<0.1
Lyonia ligustrina	0	0	0	0	0.1	<0.1	0.5	0.2
Aster divaricatus	0	0	0	0	0	0	0.4	<0.1
Smilax rotundifolia	0.3	0.1	0.2	0.1	0.3	0.2	0.4	0.6
Vaccinium stamineum	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	0.4	0.5

baceous species that represented 90% of herbaceous density in year 4. Many of the late arriving species were grasses and sedges and members of the genus *Lespedeza*. The most dominant species in the herbaceous layer was low-bush blueberry (*Vaccinium vaccilans*). Greenbrier (*Smilax glauca*) and sawbrier (*S. rotundifolia*), important early successional species on disturbed sites, were also abundant after treatment. Together they ranked fourth in density and second in percent cover in year 4 (Table 1). There was no substantial change in species diversity (H') the first and second years following treatment compared to pretreatment, but all sites had higher H' in 1994 (Figure 1a). This increase in diversity was partly due to an increase in species richness in 1994, particularly at DD (Figure 1b). Species evenness (E') remained relatively unchanged over the sampling period except for a continuous decline in E' at JW (Figure 1c). Although the magnitude of the decline in E' is not great, it reflects a disproportionate increase in density of greenbrier and low-bush blueberry on that site.

Thirty-three woody species were identified over the sampling periods. As with the herbaceous layer, the numbers varied by site and year (Figure 2a). In year 1, 28 woody species were identified; by the next year only 24 species were identified. Several species became locally absent, including 2 species of yellow pine. In year 4, 33 species were identified. Woody species diversity abruptly increased at JW and DD during the first year after treatment and was relatively unchanged at JE (Figure 2b). The posttreatment diversity estimates were unchanged for JE and JW but declined at DD in 1994. This decline in species diversity was driven by a decrease in the total number of species at DD (Figure 2b). Although total number of species declined on all sites after treatment, species evenness was higher (Figure 2c), which led to increased diversity estimates in year 1. In years 2 and 4, species evenness continued to be higher than during pretreatment. The relatively minor fluctuations in species composition resulted primarily from gains or losses of species that regenerated from seed. However, most species on these sites became reestablished through vigorous stump sprouting or by adventitious sprouts from roots. Both of these strategies are important for rapid recovery of biomass and site nutrient conservation.

Woody Vegetation Responses

We tested for differences in mean biomass and basal diameter between stems originating from stumps versus those resulting from adventitious roots or regeneration from seed (non-sprouts). Over all sites and species, we found significant differences in biomass ($P = 0.0004$) between sprouts and non-sprouts (Figure 3). On a species basis, significant differences were found

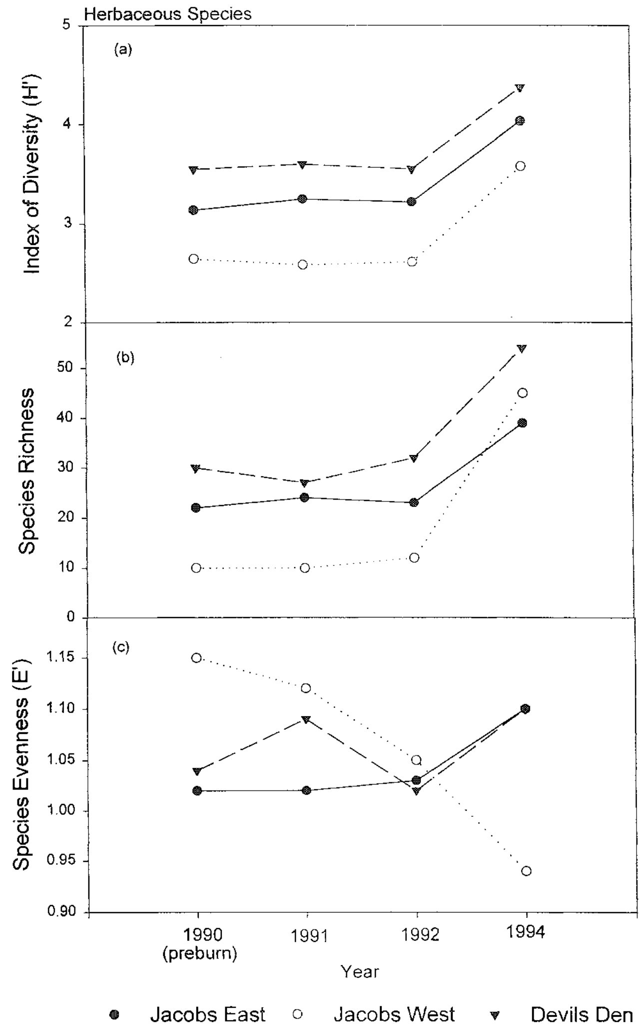

Fig. 1. Estimates of species diversity (a: Shannon Index, H'; b: species richness; c: species evenness, E') for herbaceous vegetation by site and year.

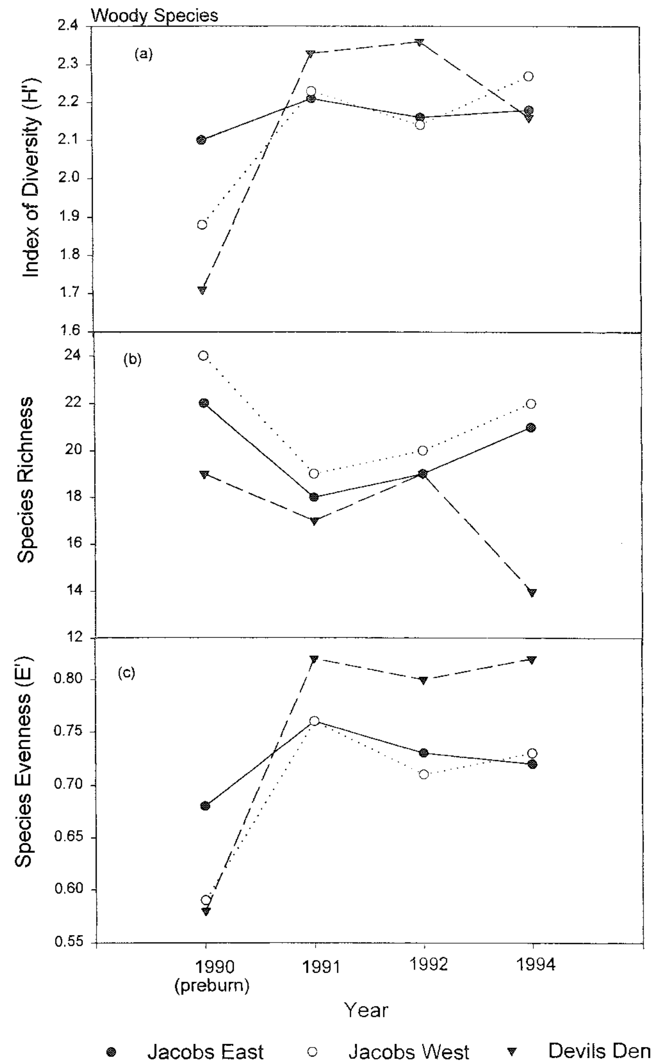

Fig. 2. Estimates of species diversity (a: Shannon Index, H'; b: species richness; c: species evenness, E') for woody vegetation by site and year.

in sprout versus non-sprout biomass and basal diameter for red maple, Allegheny serviceberry, sourwood, scarlet oak, and chestnut oak. Allegheny chinquapin had a significantly greater basal diameter of sprouts versus non-sprouts. For the oaks, the red oak group and the white oak group both had significantly greater basal diameters (P = 0.0001, F = 16.76; P = 0.0001, F = 19.14, respectively) and the red oak group had greater biomass (P = 0.03, F = 4.74) for stump sprouts than non-sprouts. On a site basis, there were no significant differences in biomass between stump sprouts and non-sprouts, but JW (P = 0.03, F = 4.76) and DD (P = 0.04, F = 4.36) had significantly greater basal diameters for stump sprouts. Across all sites, stump sprouts accounted for 78% of the biomass and 67% of the density.

Individual species' responses to treatment varied (Figure 4a–d). In the overstory, scarlet oak was the dominant species before treatment but lost its position to chestnut oak after treatment (Figure 4a). In the shrub and mid-story layer, mountain laurel was dominant before treatment and maintained its dominance

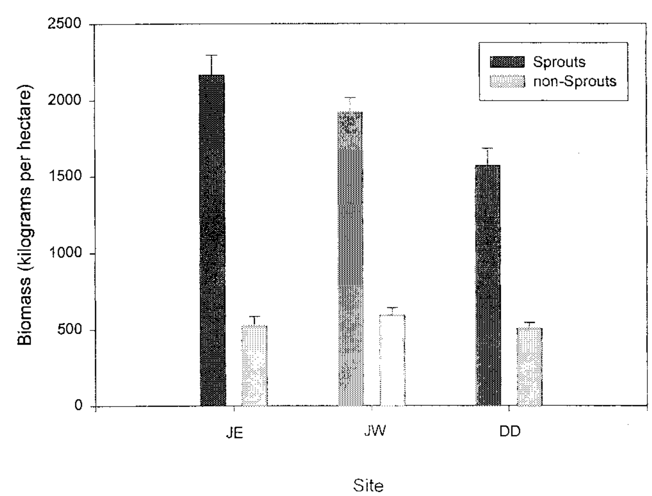

Fig. 3. Distribution of woody biomass by site (JE = Jacobs East, JW = Jacobs West, DD = Devils Den) for sprout origin and non-sprout origin stems. Error bars represent 1 standard error.

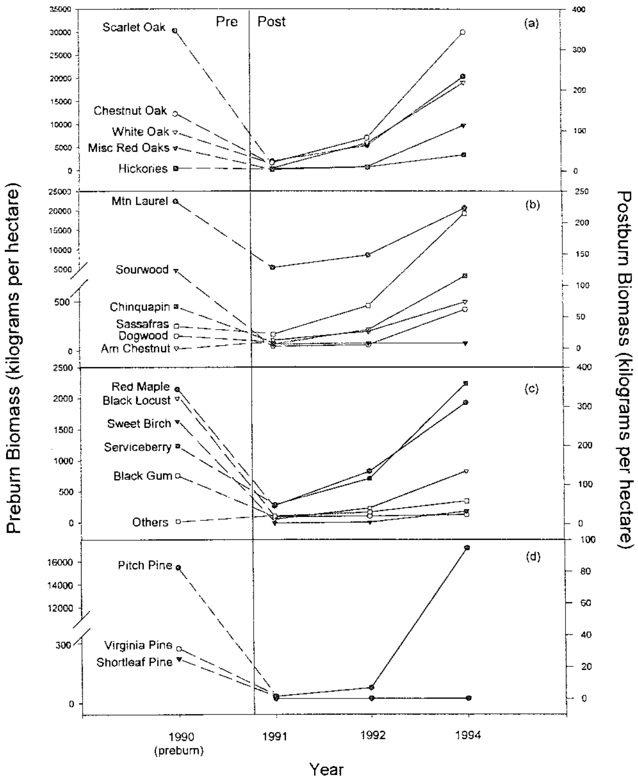

Fig. 4. Pre- and posttreatment biomass for woody species by year: (a) oaks and hickories, (b) shrub and mid-story hardwoods, (c) other overstory, and (d) yellow pines. Values for pretreatment biomass differ slightly from Vose and Swank (1993) due to the use of a general allometric equation in their study.

throughout the posttreatment period. Sassafras and American chestnut became important species following treatment. Sourwood density was lower following treatment, but was present on all sites (Figure 4b). Less dominant pretreatment overstory species (Figure 4c) exhibited different levels of importance following treatment. Serviceberry accounted for a moderate portion of pretreatment biomass but had the greatest biomass of all species and species groups by year 4. Black locust, typically an important early successional species on these sites, contributed moderately to total posttreatment biomass. Pitch pine was the most important pine species before treatment and was the only pine species to show recovery in year 4 (Figure 4d). Differential species responses were observed across all sites and were likely due to the combined effects of varying root carbohydrate reserves, species-specific sprouting potentials, and varying fire intensities across sites.

Rapid regrowth following disturbance can be an important site nutrient retention tool (Boring et al. 1988, Van Lear et al. 1990). Vitousek et al. (1979) suggested that plant accumulation of nutrients may singularly explain patterns of nitrate losses from forest ecosystems. Similarly, Marks and Bormann (1972) implicated elemental accumulation through rapid regrowth of early successional vegetation as a mechanism for minimizing nutrient losses from a northern hardwood ecosystem. Not only is vigorous sprouting an important nutrient retention tool following disturbance, but for some species, there may be an added advantage of fixing more carbon per unit leaf area. Kruger and Reich (1993) found that for northern red oak, sprout leaves had higher maximum rates of photosynthesis and maintained higher rates during midday periods when control leaves on non-sprouts exhibited depressed rates, leading to greater total daily carbon gain in sprouts. Heichel and Turner (1983) found similar patterns with red maple sprouts. This physiological response or adaptation by sprout-origin stems hastens vegetative recovery on disturbed sites and enhances the mechanism through which site nutrient retention is accomplished in 2 ways; not only do sprout stems have a vast carbohydrate reserve in existing root systems to utilize, but photosynthetic capacity may be higher, as well.

Additionally, differential photosynthetic rates in sprouts may favor some genera or species over others during early succession, increasing their competitive advantage.

Community Restoration

Information on premanagement herbaceous species abundances and distributions on these dry sites is generally lacking. Highgrade logging and fire control, which led to degraded stand conditions on these sites, began to make their mark in the distant past (>50 years ago), while efforts to characterize the herbaceous layer have been relatively recent. Hence, the only measure of response to treatment in this layer in terms of restoration to premanagement levels, comes from degraded stands whose species abundances and distributions are likely very different from historical conditions. We do know, however, that the fell and burn treatment increases diversity on these sites over the mid-term (13 years; Clinton et al. 1993) and in the short-term (1–4 years). In order to fully evaluate the potential of this treatment to restore the herbaceous community, a more complete characterization of this layer over a range of stand conditions would be required.

In the overstory, the pretreatment yellow pine component (15% of total biomass and 25% of total density) had been substantially depleted by drought-related southern pine beetle infestations, and pine seedlings and saplings (36 stems per hectare) were scattered and suppressed. Following treatment, pine seedling density had increased 20-fold after year 1 (740 stems per hectare) and 17-fold in the fourth year (630 stems per hectare). Although this represents only 2% of total density in year 4, the likelihood of yellow pine survival is high considering the competitive nature of yellow pines early in succession. By the fourth year, pines of all species were producing cones (B.D. Clinton, personal observation), which could further enhance yellow pine importance on these sites. Vose et al. (1997) compared stand responses of yellow pine to various fire treatments (e.g., wildfire, fell and burn, stand replacement). In every case, pine comprised >70% of overstory density 13 and 25 years following treatment. This response in pine density illustrates the importance of periodic fire for maintenance of this community type.

MANAGEMENT IMPLICATIONS

Mountain laurel is thought to have a major negative impact on white pine and hardwood species establishment; hence, control of this species during site preparation is a necessary practice in the southern Appalachians. Mountain laurel sprouts quickly following fire treatment and in time, regains its dominance in the understory. Before mountain laurel begins to dominate the understory, however, planted pine and other species can become well established in the mid- and overstory (Clinton et al. 1993). The fell and burn treatment appears to be an effective tool for accomplishing this goal.

Although the use of fire in forest management has been the subject of considerable criticism, forest management at the community level (e.g., for beta diversity) has drawn increasing attention in light of current public concerns over the loss of critical or unique habitats (Southern Appalachian Man and the Biosphere 1996). Results from this study and from others (Barden and Woods 1976, Van Lear and Waldrop 1988, Williams and Johnson 1992, Clinton et al. 1993, Vose et al. 1997) illustrate the potential role of prescribed fire in restoring pine-hardwood ecosystems.

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LITERATURE CITED

Abercrombie, J.A., Jr., and D.H. Sims. 1986. Fell and burn for low cost site preparation. Forest Farmer 46:14–17.

Barden, L.S., and F.W. Woods. 1973. Characteristics of lightning fires in southern Appalachian forests. Tall Timbers Fire Ecology Conference Proceedings 13:345–361.

Barden, L.S., and F.W. Woods. 1976. Effects of fire on pine and pine-hardwood forests in the southern Appalachians. Forest Science 22:399–403.

Boring, L.R. 1979. Early forest regeneration and nutrient conservation on a clearcut southern Appalachian watershed. M.S. Thesis, University of Georgia, Athens.

Boring, L.R. 1982. The role of black locust (*Robinia pseudoacacia* L.) in forest regeneration and nitrogen fixation in the southern Appalachians. Ph.D. Dissertation, University of Georgia, Athens.

Boring, L.R., and W.T. Swank. 1984. The role of black locust (*Robinia pseudoacacia*) in forest succession. Journal of Ecology 72:749–766.

Boring, L.R., and W.T. Swank. 1986. Hardwood biomass and net primary production following clearcutting in the Coweeta basin. Pages 43–50 *in* R.T. Brooks, Jr. (ed.). Proceedings of the 1986 Southern forest biomass workshop. Knoxville, TN.

Boring, L.R., W.T. Swank, and C.D. Monk. 1988. Dynamics of early successional forest structure and processes in the Coweeta Basin. Pages 161–180 *in* W.T. Swank and D.A. Crossley (eds.). Forest hydrology and ecology at Coweeta. Springer-Verlag, New York.

Clark, A., and J.G. Schroeder. 1986. Weight, volume, and physical properties of major hardwood species in the southern Appalachian mountains. Research Paper SE-253, U.S. Department of Agriculture, Forest Service, Southeastern Forest Experiment Station, Asheville, NC.

Clinton, B.D., J.M. Vose, and W.T. Swank. 1993. Site preparation burning to improve southern Appalachian pine-hardwood stands: vegetation composition and diversity of 13-year-old stands. Canadian Journal of Forest Research 23: 2271–2277.

Elliott, K.J., and B.D. Clinton. 1993. Equations for estimating biomass of herbaceous and woody vegetation in early-successional southern Appalachian pine-hardwood forests. Research Note SE-365, U.S. Department of Agriculture, Forest Service, Southeastern Forest Experiment Station, Asheville, NC.

Heichel, G.H., and N.C. Turner. 1983. CO_2 assimilation of primary and regrowth foliage of red maple (*Acer rubrum* L.) and red oak (*Quercus rubra* L.): response to defoliation. Oecologia 57:14–19.

Kruger, E.L., and P.B. Reich. 1993. Coppicing alters ecophysiology of *Quercus rubra* saplings in Wisconsin forest openings. Physiologia Plantarum 89:741–750.

Little, E.L. 1979. Checklist of United States trees. Handbook No. 541, U.S. Department of Agriculture, Forest Service.

Magurran, A.E. 1988. Choosing and interpreting diversity measures. Pages 62–79 *in* A.E. Magurran (ed.). Ecological diversity and its measure. Princeton University Press, Princeton, NJ.

Marks, P.L., and F.H. Bormann. 1972. Revegetation following forest cutting: mechanisms for return of steady-state nutrient cycling. Science 176:914–915.

Ottmar, R.D., and R.E. Vihnanek. 1991. Characterization of fuel consumption and heat pulse into the mineral soil on the Jacob Branch and Devil Den units in North Carolina. Final report submitted to the U.S. Environmental Protection Agency, Research Triangle Park, NC.

Peet, R.K. 1974. The measurement of species diversity. Annual Review of Ecological Systems 5:285–307.

Phillips, D. 1981. Predicted total-tree biomass of understory hardwoods. Research Paper SE-223, U.S. Department of Agriculture, Forest Service, Southeastern Forest Experiment Station, Asheville, NC.

Radford, A.E., H.E. Ahles, and C.R. Bell. 1968. Manual of the vascular flora of the Carolinas. University of North Carolina Press, Chapel Hill.

Ryan, K.C., and N.V. Noste. 1985. Evaluating prescribed fires. Pages 230–238 *in* Proceedings—symposium and workshop on wilderness fires. General Technical Report INT-182, U.S. Department of Agriculture, Forest Service, Intermountain Research Station, Ogden, UT.

SAS Institute Inc. 1987. SAS user's guide: statistics, 1987 edition. SAS Institute Inc., Cary, NC.

Smith, R.N. 1991. Species composition, stand structure, and woody detrital dynamics associated with pine mortality in the southern Appalachians. M.S. Thesis, University of Georgia, Athens.

Southern Appalachian Man and the Biosphere (SAMAB). 1996. Communities and human influences in southern Appalachian ecosystems: the human dimensions. Pages 17–85 *in* The southern Appalachian assessment social/cultural/economic technical report, No. 4 of 5.

Swank, W.T., and H.T. Schreuder. 1974. Comparison of three methods of estimating surface area and biomass for a forest of young white pine. Forest Science 20:91–100.

Swift, L.W., K.J. Elliott, R.D. Ottmar, and R.E. Vihnanek. 1993. Site preparation to improve southern Appalachian pine-hardwood stands: fire characteristics, and soil erosion, temperature and moisture. Canadian Journal of Forest Research 23:2242–2254.

Taylor, L.R. 1978. Bates, Williams, Hutchinson—a variety of diversities. Pages 1–18 *in* L.A. Mound and N. Warloff (eds.). Diversity of insect faunas: 9th symposium of the Royal Entomological Society. Blackwell Scientific Publications, Oxford, UK.

Van Lear, D.H., M.A. Taras, J.B. Waide, and M.K. Augspurger. 1986. Comparison of biomass equations for planted vs. natural loblolly pine stands of sawtimber size. Forest Ecology and Management 14:205–210.

Van Lear, D.H., and T.A. Waldrop. 1988. Effects of fire on natural regeneration in the Appalachian mountains. Pages 56–70 *in* H.C. Smith, A.W. Perkey, and W.E. Kidd, Jr. (eds.). Proceedings: guidelines for regenerating Appalachian hardwood stands. Publication 88-03, Society of American Foresters, Morgantown, WV.

Van Lear, D.H., P.R. Kapeluck, and J.B. Waide. 1990. Nitrogen pools and processes during natural regeneration of loblolly pine. Pages 234–252 *in* S.P. Gessel, D.S. Lacate, G.F. Weetman, and R.F. Powers (eds.). Proceedings 7th North American forest soils conference, Vancouver, BC.

Vitousek, P.M., J.R. Gosz, C.C. Grier, J.M. Melillo, W.A. Reiners, and R.L. Todd. 1979. Nitrate losses from disturbed ecosystems. Science 204:469–474.

Vose, J.M., and W.T. Swank. 1993. Site preparation burning to improve southern Appalachian pine-hardwood stands: aboveground biomass, forest floor mass, and nitrogen and carbon pools. Canadian Journal of Forest Research 23: 2255–2262.

Vose, J.M., W.T. Swank, B.D. Clinton, R.L. Hendrick, and A.E. Major. 1997. Using fire to restore pine/hardwood ecosystems in the southern Appalachians of North Carolina. Pages 149–154 *in* J.M. Greenlee (ed.). Proceedings of the first conference on fire effects on rare and endangered species and habitats. International Association of Fire Research, Coeur d'Alene, ID.

Williams, C.E., and W.C. Johnson. 1992. Factors affecting the recruitment of *Pinus pungens* in the southern Appalachian Mountains. Canadian Journal of Forest Research 22:878–887.