Vegetation response to midstorey mulching and prescribed burning for wildfire hazard reduction and longleaf pine (*Pinus palustris* Mill.) ecosystem restoration

DALE G. BROCKWAY[1]*, KENNETH W. OUTCALT[2], BECKY L. ESTES[3] AND ROBERT B. RUMMER[1]

[1] Southern Research Station, USDA Forest Service, 520 Devall Drive, Auburn, AL 36849, USA
[2] Southern Research Station, USDA Forest Service, 320 Green Street, Athens, GA 30602, USA
[3] Pacific Southwest Research Station, USDA Forest Service, 2400 Washington Avenue, Redding, CA 96001, USA
* Corresponding author. E-mail: dbrockway@fs.fed.us

Summary

Dense midstorey vegetation, developed during fire exclusion, not only reduces understorey plant diversity and increases the risk of damaging wildfire but also impedes efforts to safely restore prescribed burning in longleaf pine (*Pinus palustris* Mill.) ecosystems. Our study examined the effects of midstorey reduction on stand structure and plant diversity in a forest treated by mulching alone and also when followed by prescribed fire during the winter, spring or summer. For trees ≥5 cm diameter at breast height (d.b.h.), mulching reduced stand density (1220–258 trees ha^{-1}) and basal area (24–17.7 m^2 ha^{-1}) and increased mean d.b.h. (12.8–29.2 cm), with the largest reductions in loblolly pine (*Pinus taeda* L.), sweetgum (*Liquidambar styraciflua* L.) and oaks (*Quercus* spp. L.). Removing hardwoods and smaller pines resulted in a decline in tree species richness (8.9–4.4). Despite a modest increase in evenness (0.72–0.79), tree species diversity ($H' = 1.32$–0.84) dynamics were largely driven by changes in richness. While the cover of tree seedlings initially declined from 32.4 to 16.9 per cent, rapid regrowth of hardwoods led to recovery by end of the second growing season. This, along with gains by shrubs, vines, grasses and forbs, resulted in a near doubling of understorey plant cover. Although tree seedling increases were not related to fire season, peak responses occurred for shrubs and vines after winter fire and spring fire, grasses following winter fire and forbs after summer fire. An increase in species richness (18.7–24.5) and decline in species evenness (0.86–0.70) produced only a small increase in understorey species diversity ($H' = 2.31$–2.45). The greater number of understorey species following treatment were less equitably distributed as a result of differential rates of plant growth. While mulching led to a short-term increase in woody and herbaceous understorey plants, prescribed fire is needed to curtail redevelopment of the woody midstorey and further increase grasses and forbs.

Published by Oxford University Press on behalf of the Institute of Chartered Foresters, 2009

Forestry, Vol. 82, No. 3, 2009. doi:10.1093/forestry/cpp010
Advance Access publication date 7 April 2009

Introduction

In fire-dependent ecosystems, where short-interval and low- to moderate-intensity fires were historically prevalent, forests now occur at greater densities, are more spatially uniform, have many more small trees and fewer large trees and contain greater quantities of fuel than in the presettlement past (Waldrop *et al.* 1989; Brockway and Lewis 1997; Cowell 1998; Taylor and Skinner 1998; Harrod *et al.* 1999). Fire exclusion, livestock grazing, timber harvest, farm abandonment and climate change are among the factors leading to this condition and a corresponding increase in the probability of large, high-intensity wildfires (Arno *et al.* 1997; Dahms and Geils 1997; Stephens 1998; Outcalt and Wade 2004). While treatments to reduce excessive fuel accumulations and establish sustainable structures and processes are needed, what is less clear is the suitability of methods and the appropriate balance among techniques for tree harvest, mechanical fuel treatments and prescribed burning to achieve restoration and wildfire hazard abatement objectives (Hardy and Arno 1996; Stephens 1998; Brockway *et al.* 2005a, b). It is therefore important to examine how forest ecosystems will be altered if mechanical fuel reduction treatments are used instead of or in combination with prescribed fire. High-density forests, prone to catastrophic damage by wildfire, represent a substantial challenge to forest management in the southeastern US.

Longleaf pine (*Pinus palustris* Mill.) ecosystems once dominated 38 million ha, extending along the Coastal Plain from Texas to Florida and northward to Virginia and into the Piedmont and mountains of Alabama and Georgia (Stout and Marion 1993). Longleaf pine is a species of great ecological amplitude, occurring in forests, woodlands and savannas on sites ranging from wet, poorly drained flatwoods to mesic uplands, xeric sandhills and rocky mountainous ridges (Boyer 1990). Naturally occurring longleaf pine forests are typically an uneven-aged mosaic of even-aged patches distributed across the landscape, which vary in size, shape, structure, composition and density and contain numerous embedded special habitats such as stream bottoms, wetlands and seeps (Platt and Rathbun 1993; Brockway and Outcalt 1998; Hilton 1999). The natural variability of these ecosystems makes them excellent habitat for a variety of game animals and numerous non-game and rare species (Engstrom 1993; Crofton 2001; Brockway and Lewis 2003). Frequent, low- to moderate-intensity surface fires aid in development and maintenance of the open, park-like stand structure of longleaf pine forests, making them much less prone to catastrophic damage from wildfire (Schwarz 1907; Wahlenberg 1946; Landers *et al.* 1995; Brockway and Lewis 1997; Outcalt 2000). Historically, the understories of longleaf pine forests contained few shrubs and were dominated by bluestem grasses (*Andropogon* spp. L. and *Schizachyrium scoparium* (Michx.) Nash) and wiregrass (*Aristida beyrichiana* Trin. & Rupr. and *Aristida stricta* Michx.), which together with fallen pine needles form a fine-fuel matrix that facilitates the ignition and rapid spread of fire (Abrahamson and Hartnett 1990; Landers 1991). Prior to landscape fragmentation, natural fires occurred every 2–8 years throughout much of the region and longleaf pine maintained dominance over large areas primarily because it is more tolerant of fire than competing tree species with thinner barked seedlings (Christensen 1981; Abrahamson and Hartnett 1990; Landers *et al.* 1995).

As a result of land clearing for agriculture, conversion to industrial plantations and interruption of natural fire regimes, longleaf pine forests now cover 1.02 million ha (Frost 2006). This is <3 per cent of their original extent. Much of the land previously occupied by longleaf pine has now, as a result of fire exclusion, become dominated by fire-sensitive loblolly pine (*Pinus taeda* L.) and a variety of hardwood tree species (Outcalt and Sheffield 1996). In the absence of fire, many sites support high-density stands that contain midstorey layers of trees and shrubs that can function as 'fuel ladders', allowing surface fires to spread to the forest canopy. Such conditions represent a danger of total stand loss during wildfire and complicate efforts to safely reintroduce burning and restore the natural disturbance regime of periodic surface fire. Mechanical treatment that alters stand structure, by eliminating or reducing the hazardous midstorey, should facilitate safer implementation of prescribed burning.

With wider recognition of the increased wildfire hazard posed by high-density forests, very often comprised of unmerchantable trees, has come a need for machines that supplement or replace traditional technologies like chainsaws, feller-bunchers and herbicides (Provencher *et al.* 2001b). Mulching machines, originally designed for clearing utility corridors, have been recently adapted to this task and their use in the forest has progressively expanded (Bolding *et al.* 2003; Stanturf *et al.* 2003; Fight *et al.* 2004). Mulching in this context refers to the grinding of small, medium and sometimes large trees by machines having front-mounted rotary drums with cutting teeth that can rapidly reduce entire trees to wood chips that are then left scattered upon the forest floor as a mulch. Thus, arose the need to evaluate, alone and in combination with prescribed fire, the effects of mulching technology applied in southern pine ecosystems.

A former longleaf pine forest, that had in the absence of fire become invaded and occupied by loblolly pine and hardwood trees, was treated with mulching machines to reduce its midstorey and restore a structure suitable for reintroducing periodic surface fire. The primary objective of this study was to quantify the effects of mulching alone and mulching followed by fire during the winter, spring or summer on plant community characteristics, testing the hypotheses: (1) as stand density is reduced, (A) ladder fuels will decline, (B) herbaceous plant cover will increase and (C) understorey plant diversity will rise and (2) prescribed burning will be needed to maintain the initial gains achieved with mulching. A secondary objective was to observe machine performance in the forest and document the capabilities and limitations of applying mulching technology for wildfire hazard reduction and ecological restoration. Analysis of changes in tree density, basal area, species composition, foliar cover and diversity should provide insights concerning its impacts and help managers determine whether mulching can be used as a substitute for or complement to prescribed burning in wildland/urban interface areas.

Methods

Study site and management history

This study was conducted at Fort Benning Military Base, located in west central Georgia ~300 km north of the Gulf of Mexico (32° 25′ N, 84° 55′ W) along the Fall-Line Sandhills, a transition zone between the Piedmont and Upper Coastal Plain. The climate is temperate (humid subtropical), with a 200-day growing season from April to October. Average monthly temperatures range from 18 to 27°C for the April to October period and from 7 to 13°C for November to March. Annual precipitation is abundant, averaging 1320 mm, with half of this arriving during the growing season (Johnson 1982; Green 1997).

The study site is ~90 m above sea level in a sandhills landscape, with surface topography ranging from gently inclined (2–5 per cent) to moderately sloping (5–15 per cent). Soils developed from residuum of the Tuscaloosa Formation, with parent materials on slopes and ridgetops giving rise to the Orangeburg (Typic Paleudult, thermic), Esto (Typic Paleudult, thermic), Dothan (Plinthic Paleudult, thermic), Troup (Grossarenic Paleudult, thermic) and Nankin (Typic Kanhapludult, thermic) series, which are deep, well-drained and sandy soils that are low in organic matter and nutrients and low to moderate in water holding capacity. Soils along streams are dominated by Bibb (Typic Fluvaquents, thermic) sandy loam which is poorly drained, low in natural fertility and medium in organic matter content and water holding capacity (Johnson 1982; Green 1997).

Vegetation in the area consisted of an overstorey dominated by a mixture of loblolly pine and hardwoods that developed in the presence of a smaller number of large, older longleaf pine. Loblolly pine and hardwoods also formed a high-density midstorey. Oak (*Quercus* spp. L.), sweetgum (*Liquidambar styraciflua* L.) and black cherry (*Prunus serotina* Ehrh.) were the predominant tree seedlings in the understorey. Blueberries (*Vaccinium* spp. L.), blackberries (*Rubus* spp. L.), wax myrtle (*Myrica cerifera* L.) and juneberry (*Amelanchier* spp. Medik.) were the most prominent shrubs, but were often surpassed by the ubiquitous presence of vines such as grape (*Vitis* spp. L.), greenbriar (*Smilax* spp. L.), yellow jessamine (*Gelsemium sempervirens* (L.) W.T. Aiton), poison ivy (*Toxicodendron radicans* (L.) Kuntze) and honeysuckle (*Lonicera japonica* Thunb.). Herbaceous plants were poorly represented in the area, with broomsedge bluestem (*Andropogon virginicus* L.), the most common grass, and partridge berry (*Mitchella repens* L.), the most frequently observed forb.

The study site was occupied by second-growth forest that naturally regenerated following harvest of the original longleaf pine. Since 1918, the locale has been used for various types of military training and received varying degrees of prescribed fire use and restriction (Dale *et al.* 2002). Located in the wildland–urban interface of the densely populated Main Post, the practices of exclusion and suppression impaired the natural regeneration of longleaf pine and allowed an overstorey and high-density midstorey of loblolly pine and hardwoods to develop.

Figure 1. Rubber-tired Magnum mulcher onsite September 2000.

Study design and experimental treatments

In March 2000, a randomized complete block experimental design was established, with four replications of five experimental treatments distributed across the 24-ha study area. The four blocks (i.e. replications) were topographically positioned across the moisture gradient from wet to mesic to dry conditions. Treatments were randomly assigned within blocks and consisted of (1) mulching without fire; (2) mulching followed by winter fire; (3) mulching followed by spring fire; (4) mulching followed by summer fire and (5) no mulching and no fire, serving as the control. Each treatment plot was 1.2 ha in area, having dimensions of 110 × 110 m. All measurement subplots were located at least 25 m from the treatment plot boundary to minimize the influence of edge effects.

Figure 2. Track-equipped Delta 953C mulcher onsite August 2001.

Since all longleaf pine remaining on the site were large in diameter, the treatment prescription specified mulching of all hardwood trees (regardless of size) and all pines <20 cm at diameter at breast height (d.b.h.). During September 2000, the midstorey was treated by a rubber-tired 500 hp Magnum mulcher (Figure 1). Problems related to machine design (rubber tires bogged down in moist areas) and operation (inability to traverse slopes) resulted in an incomplete initial treatment. Therefore, during August 2001, the site was treated by a tracked 500 hp Delta 953C mulcher (Figure 2), which did not exhibit the problems identified above. Although the second treatment more closely corresponded to our prescription, results in both cases did not meet expectations since visual inspection revealed that many large hardwoods and some smaller trees remained onsite. Thus in addressing our secondary objective, we observed that while tracked mulching machines demonstrated distinct advantages when operating across slopes and on wet soils, operator skill and compliance were vitally important to achieving desired results. Prescribed fire treatments were then applied during the winter (December 2001), spring (May 2002) and summer (July 2002) seasons. These treatments followed the initial mulching treatment by 15, 20 and 22 months, respectively, and second mulching treatment by 4, 9 and 11 months, respectively.

Measurements and analysis

In May 2000, overstorey and midstorey stand structure and understorey plant cover were measured to establish pre-treatment forest conditions. Repeated post-treatment measurements were completed in September 2002 and 2003 to assess the changes resulting from mulching and prescribed fire. Overstorey and midstorey composition and structure were measured by recording the species and d.b.h. for all trees within five subplots (10 × 10 m) systematically located within each 1.2-ha treatment plot. Total foliar cover (vertical projection of canopy) of all understorey plant species was measured by the line-intercept method along five permanent 15-m line transects within each treatment plot (Bonham 1989). Plant identification and nomenclature were according to the taxonomic authorities (Godfrey 1988; Wunderlin 1998; Duncan and Duncan 1999; Miller and Miller 1999).

Vegetation data were summarized as estimates of the mean for each plot and analysed by treatment and change through time. Stand density, mean stand diameter and stand basal area were computed from tree diameter data. Data for tree basal area and understorey foliar cover were used as importance values for computing several diversity indices (Ludwig and Reynolds 1988). Species richness (N_0) was characterized on each plot by counting the number of species present, evenness was calculated using the modified Hill ratio (E_5) (Alatalo 1981) and diversity was estimated using the Shannon diversity index (H') (Shannon and Weaver 1949).

The means of the dependent variables for each plot were used to estimate the means and variances for the treatment units. A repeated measures analysis of variance, using initial conditions as covariates, was used to evaluate time and treatment effects and interactions (Hintze 1995). Responses of mulching treatments and prescribed fire season were compared using a set of four pairwise contrasts. The trend through time after treatment was analysed using orthogonal polynomials. Statistical analysis of the time and treatment interaction for computed diversity indices was completed using the bootstrap technique PROC MULTTEST in SAS (Efron and Tibshirani 1993; Westfall and Young 1993; SAS Institute 1996). Adjusted *P*-values, which maintain a constant Type I error across the full range of comparisons, were used to determine significant differences among means (10 000 bootstrap iterations). A probability level of 0.05 was used to discern significant differences.

Results

Stand structure and composition

For trees ≥5 cm d.b.h., mulching treatment reduced the overstorey and midstorey density from 1220 to 258 trees ha^{-1} (Table 1). This 79 per cent decline in tree density was significantly different from the relatively unchanged control (Figure 3). Mulching also reduced stand basal area by 26 per cent, from 24 to 17.7 m^2 ha^{-1}. The differential degree of reduction between tree density and basal area resulted from targeting smaller diameter trees for removal while preferentially retaining larger trees (>20 cm at d.b.h.). The greatest reductions were observed for loblolly pine, sweetgum, water oak (*Quercus nigra* L.) and laurel oak (*Quercus hemisphaerica* Bartram ex Willd.). Removal of so many smaller trees correspondingly increased the mean stand diameter by 128 per cent, from 12.8 to 29.2 cm at d.b.h.

Few short-term stand structural differences were discernible among the fire treatments. Although smaller reductions in basal area appear following winter and spring fires than summer fire, these differences were not statistically significant. While fire temperatures increased as air temperature increased through the seasons, peak flame temperatures remained ≤185°C and mulch generated by mechanical treatment burned little if at all. The long-time delay between mulching and prescribed burning during the summer allowed dense patches of hardwood tree seedlings and vines to develop from surviving root systems.

Foliar cover of understorey plants

During the 13 months following mulching, the foliar cover of understorey tree seedlings declined from 32.4 to 16.9 per cent overall (Table 2). The cover of shrubs and vines increased from 23.6 to 33.4 per cent, that for grasses increased from 4.9 to 14.4 per cent and forb cover increased from 2.7 to 5.8 per cent. Within 2 years of mulching and 1 year of prescribed fire, total foliar cover of all vascular plants in the understorey nearly doubled, with significant gains in sweetgum, blackberry, beautyberry (*Callicarpa americana* L.), grape, Virginia creeper (*Parthenocissus quinquefolia* (L.) Planch.), spikegrass (*Chasmanthium*

Table 1: Mean overstorey and midstorey tree responses to mulching and prescribed fire for pre-treatment (2000) and post-treatment (2002) years

	Control	Mulching only	Mulching and winter fire	Mulching and spring fire	Mulching and summer fire	Mean of non-control treatments
Mean density (trees ha^{-1})						
2000	807	1220	935	1175	1550	1220
2002	793	212*†	215*†	305*†	300*†	258*†
Density change (%)						
2002	−2	−83	−77	−74	−81	−79
Mean d.b.h. (cm)						
2000	13.4	12.6	13.7	13.1	11.6	12.8
2002	13.5	24.1*†	31.6*†	28.2*†	32.9*†	29.2*†
Mean diameter change (%)						
2002	1	91	131	115	184	128
Basal area (m^2 ha^{-1})						
2000	17.7	22.7	25.7	25.2	22.5	24.0
2002	16.4	13.1	21.5	21.0	15.1	17.7
Basal area change (%)						
2002	−7	−42	−16	−17	−33	−26

* Significantly different from control, $P < 0.05$.
† Significant change through time from pre-treatment condition, $P < 0.05$.

spp. Link), panicgrass (*Panicum* spp. L.) and fireweed (*Erechtites hieracifolia* (L.) Raf. ex DC.). This trend appears driven by the post-treatment recovery of trees (as sprouts or new seedlings) and progressive expansion of shrubs, vines, grasses and forbs. Only the foliar cover of laurel oak and highbush blueberry (*Vaccinium corymbosum* L.) was significantly reduced by treatment. On the undisturbed control plots, overall plant cover remained largely stable; however, significant declines were noted for tree seedlings and increases observed for beautyberry, spikegrass and ferns.

No response differences related to fire season were noted for tree seedlings. Although the cover of shrubs and vines was significantly greater on plots burned during the winter or spring, blackberry, grape and beautyberry also responded significantly following summer fire. While significant grass cover and forb cover increases occurred from burning during all three seasons, the peak graminoid response was observed from winter fire and that for forbs resulted from summer fire. The enlarged cover of graminoids was principally driven by the appearance and expansion of spikegrass across all fire seasons in the post-treatment environment and increases in panicgrass following mulching and summer burning. Among forbs, the season of fire also had significant differential effects, with winter fire increasing creeping wood sorrel (*Oxalis corniculata* L.), spring fire augmenting elephantsfoot (*Elephantopus tomentosus* L.) and summer fire expanding fireweed and yankeeweed (*Eupatorium compositifolium* Walter).

Plant species diversity

Our site supported a moderately rich plant community of 108 vascular plant species. The 46 species of grasses, forbs and ferns were poorly represented, reflecting the impoverished initial condition of an herbaceous plant community dominated by woody plants in the understorey, midstorey and overstorey. While 31 species were forbs, their occurrence, except for partridge berry, was initially low (<30 per cent). Increases following treatment were noted for fireweed, yankeeweed, elephantsfoot, wood sorrel, milk-pea (*Galactia volubilis* (L.) Britton), sweet goldenrod (*Solidago odora* Aiton), pineweed (*Hypericum gentianoides* (L.) Britton *et al.*), pencilflower (*Stylosanthes biflora* (L.) Britton *et al.*) and bush-clover (*Lespedeza* spp. Michx.). Except

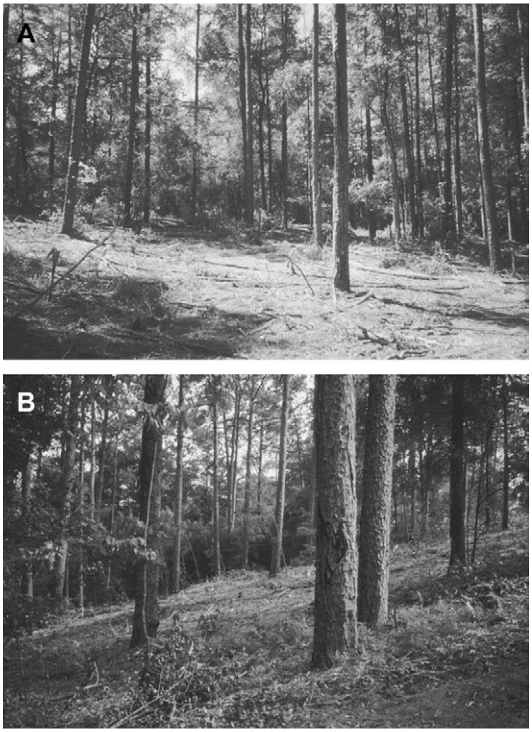

Figure 3. Open stand structure following mulching at foreground as contrasted with high-density midstorey of untreated control plot in background, (a) ridgetop treated by rubber-tired Magnum mulcher in September 2000 and (b) slope treated by track-equipped Delta 953C mulcher in August 2001.

for broomsedge bluestem, occurrence of the 10 graminoid taxa was also initially low. Increases in the frequency of grasses resulted from expansion of spikegrass and panicgrass. The 31 tree species, 18 shrub species and 8 vine species represented the dominant understorey plant groups. The occurrence of blackberry, beautyberry, Virginia creeper, wax myrtle, greenbriar poison ivy, peppervine (*Ampelopsis arborea* (L.) Koehne), winged sumac (*Rhus copallinum* L.), partridge pea (*Chamaecrista fasciculata* (Michx.) Greene), deerberry (*Vaccinium stamineum* L.), yellow-poplar (*Liriodendron tulipifera* L.), southern red oak (*Quercus falcata* Michx.), water oak and sassafras (*Sassafras albidum* (Nutt.) Nees) increased following treatment. Of the 26 tree species in the overstorey and midstorey, longleaf pine, loblolly pine, laurel oak, sweetgum, water oak, yellow-poplar, white oak (*Quercus alba* L.), American beech (*Fagus grandifolia* Ehrh.) and flowering dogwood (*Cornus florida* L.) most frequently occurred in the post-treatment overstorey.

Mulching caused a significant overall decline in species richness (N_0) in the overstorey and midstorey, decreasing the mean number of tree species from 8.9 to 4.4 (Table 3). Most of the trees lost were of small to moderate stature, such as red maple (*Acer rubrum* L.), mimosa-tree (*Albizia julibrissin* Durazz.), river birch (*Betula nigra* L.), American hornbeam (*Carpinus carolina* Walter), black tupelo (*Nyssa sylvatica* Marshall), eastern cottonwood (*Populus deltoides* W.Bartram ex Marshall), American plum (*Prunus americana* Marshall), black cherry, southern red oak, Carolina buckthorn (*Rhamnus caroliniana* Walter), winged elm (*Ulmus alata* Michx.) and common prickly-ash (*Zanthoxylum americanum* Mill.). Plant species diversity for these layers, though not consistently significant, follows a similar declining trend. Values for the Shannon diversity index (H') decreased on mulched plots from 1.32 to 0.84 during the period of study. Values for species evenness were indicative of moderate equity in the distribution of tree species present on the site. A subsequent increase in the modified Hill ratio (E_5) was noted across most treatments, rising overall from 0.72 to 0.79; however, this trend was not statistically significant. Therefore, diversity changes in these layers were driven primarily by variations in species richness.

Species richness among understorey plants increased significantly following treatment, from an average of 18.7–24.5 species (Table 4). Although some variation resulted from time differences between fire application and understorey measurement, overall responses were reasonably similar across all fire seasons and comparable to plots mulched and not burned. Plant species diversity followed a similar overall trend, with H' increasing on average from 2.31 to 2.45 after the second post-treatment growing season. However, this upward trend for species diversity was not statistically significant and was moderated by a significant decline in species evenness among understorey plants, from 0.86 to 0.70. Decreased evenness resulted from the expansion of sweetgum, blackberry and grape relative to other species. Although species richness increased in the understorey following treatment, the plant species

Table 2: Mean understorey foliar cover (%) response to mulching and prescribed fire for pre-treatment (2000) and post-treatment (2002 and 2003) years

	Control	Mulching only	Mulching and winter fire	Mulching and spring fire	Mulching and summer fire	Mean of non-control treatments
All plants						
2000	67.1	64.7	58.7	69.4	62.0	63.7
2002	60.3	76.5	82.7	68.7	56.4	71.1
2003	71.5	112.2*†	130.1*†	113.1*†	114.3*†	117.4*†
Woody plants						
2000	58.4	52.3	51.5	61.4	58.6	56.0
2002	42.1	45.8	63.0	50.1	42.1	50.3
2003	52.2	69.4*†	92.7*†	85.9*†	68.3*†	79.1*†
Trees						
2000	31.3	31.8	30.3	32.7	34.8	32.4
2002	10.0†	19.2*†	19.4*†	17.9*†	11.2†	16.9*†
2003	14.1†	28.5*	28.8*	28.3*	24.3*	27.5*
Liquidambar styraciflua						
2000	4.9	8.1	5.0	10.2	5.2	7.1
2002	4.5	9.2	7.7	8.3	4.4	7.4
2003	4.3	14.8*†	14.9*†	16.4*†	12.6*†	14.7*†
Quercus hemisphaerica						
2000	17.9	9.1	10.2	10.9	12.4	10.7
2002	2.3†	0.2*†	0.1*†	0.0*†	0.0*†	0.1*†
2003	5.5†	1.0*†	1.2*†	0.1*†	0.8*†	0.8*†
Shrubs and vines						
2000	27.1	20.5	21.2	28.8	23.8	23.6
2002	32.1	26.7	43.7†	32.2	31.0	33.4
2003	38.1	40.9†	63.9*†	57.6*†	44.1†	51.6*†
Callicarpa americana						
2000	0.0	0.0	0.0	0.0	0.0	0.0
2002	3.7	1.2	4.0	3.2	5.7†	3.5
2003	6.7†	3.0	4.7†	3.9	5.6†	4.3†
Parthenocissus quinquefolia						
2000	0.0	0.2	0.1	0.0	0.1	0.1
2002	0.3	1.1	2.3	0.6	0.6	1.2
2003	1.1	4.1*†	4.6*†	5.5*†	0.4	3.7*†
Rubus spp.						
2000	0.6	0.1	0.3	0.1	0.2	0.2
2002	2.7	1.8	6.3†	5.0†	5.4†	4.6†
2003	3.9	5.7†	7.7†	15.2*†	10.9*†	9.9*†
Vaccinium corymbosum						
2000	7.0	5.4	6.7	5.8	1.0*	4.7
2002	4.3	1.0*†	2.8†	0.6*†	0.8*	1.3*†
2003	7.2	2.8*	2.9*†	1.0*†	0.6*	1.8*†
Vitis spp.						
2000	1.9	4.7	3.2	2.7	3.0	3.4
2002	2.1	10.8*†	14.3*†	8.6*†	5.3	9.8*†
2003	3.1	13.6*†	32.8*†	18.4*†	13.3*†	19.5*†
Graminoids						
2000	3.5	8.6	4.4	3.9	2.6	4.9
2002	7.8	22.8*†	12.8†	10.7†	11.3†	14.4†
2003	5.9	33.4*†	24.5*†	16.6*†	18.7*†	23.3*†

Table 2: Continued

	Control	Mulching only	Mulching and winter fire	Mulching and spring fire	Mulching and summer fire	Mean of non-control treatments
Chasmanthium spp.						
2000	0.0	0.0	0.0	0.0	0.0	0.0
2002	6.8†	12.9*†	6.2†	4.9†	5.0†	7.3†
2003	4.7†	21.2*†	20.9*†	9.9*†	8.9*†	15.2*†
Panicum spp.						
2000	0.0	0.2	0.0	0.1	0.2	0.1
2002	0.0	6.0*†	5.6*†	3.8*†	5.3*†	5.2*†
2003	1.1	6.7*†	2.1	3.8†	7.8*†	5.1*†
Forbs						
2000	3.2	3.6	2.5	4.1	0.7	2.7
2002	3.8	6.8†	6.2†	7.5†	2.8	5.8†
2003	1.2	8.3*†	11.1*†	10.3*†	27.4*†	14.3*†
Elephantopus tomentosus						
2000	0.0	0.0	0.0	0.0	0.0	0.0
2002	0.1	0.7	0.3	1.3	0.2	0.6
2003	0.3	1.0	0.1	3.5*†	0.2	1.2
Erechtites hieracifolia						
2000	0.0	0.0	0.0	0.0	0.0	0.0
2002	0.0	1.3	1.0	1.8	0.1	1.1
2003	0.0	2.1	0.8	2.3	14.5*†	4.9*†
Eupatorium compositifolium						
2000	0.0	0.0	0.0	0.0	0.0	0.0
2002	0.0	0.8	1.1	1.4	0.4	0.9
2003	0.0	0.8	1.9	1.2	4.1*†	2.0
Oxalis corniculata						
2000	0.0	0.0	0.0	0.0	0.0	0.0
2002	0.0	0.0	0.1	0.0	0.0	0.0
2003	0.0	0.0	5.5*†	0.0	0.0	1.4
Ferns						
2000	2.1	0.2	0.4	0.0	0.2	0.2
2002	6.6†	1.0*	0.7*	0.5*	0.3*	0.6*
2003	12.1†	1.1*	1.8*	0.3*	0.0*	0.8*

* Significantly different from control, $P < 0.05$.
† Significant change through time from pre-treatment condition, $P < 0.05$.

present were less equitably distributed across the site.

Discussion

Stand structure and fire

Mulching changed stand structure by converting the midstorey into wood chips distributed across the forest floor and leaving an overstorey of fewer, larger and more widely spaced longleaf pine and loblolly pine trees. Fuel ladders were largely eliminated, thereby altering the behaviour and severity of subsequent fires and diminishing the risk of destruction by catastrophic wildfire (Graham *et al.* 2004). Thus, hypothesis 1A was accepted that reducing stand density through mulching decreases the presence of fuel ladders. Mulching has potential for use in wildfire hazard reduction and ecological restoration. These post-treatment conditions are consistent with the desired composition and structure of stands and fuelbeds in forests having regimes of frequent

Table 3: Overstorey and midstorey tree species richness, diversity and evenness response to mulching and prescribed fire for pre-treatment (2000) and post-treatment (2002) years

	Control	Mulching only	Mulching and winter fire	Mulching and spring fire	Mulching and summer fire	Mean of non-control treatments
Number of species						
2000	8.7	8.4	8.8	10.0	8.3	8.9
2002	8.7	4.0*†	4.3*†	6.0*†	3.3*†	4.4*†
Shannon index						
2000	1.15	1.34	1.22	1.49	1.24	1.32
2002	1.13	0.78*†	1.01	1.17†	0.40*†	0.84*†
Modified Hill ratio						
2000	0.72	0.73	0.71	0.73	0.70	0.72
2002	0.67	0.83	0.87	0.79	0.68	0.79

* Significantly different from control, $P < 0.05$.

† Significant change through time from pre-treatment condition, $P < 0.05$.

surface fires (Fule' *et al.* 2001; Outcalt 2003; Romme *et al.* 2003; Agee and Skinner 2005; Stephens and Moghaddas 2005).

Mechanical treatment has been suggested as a possible alternative to prescribed fire in densely populated wildland–urban interface zones (Behm and Duryea 2003; Stanturf *et al.* 2003). However, sprouting was vigorous on plots remaining unburned for a year or more after mulching and such areas became occupied by dense thickets of hardwood seedlings interlaced with vines. Therefore, repeated burning is likely to be needed to prevent redevelopment of another high-density midstorey following mulching. This finding supports acceptance of hypothesis 2 that follow-up burning will be needed to maintain the initial gains achieved through mulching. It may be best to initially burn such sites within 18 months or less of mulching, during a period when hardwood seedlings are most vulnerable to fire-induced mortality. Although implementation is complex in the wildland–urban interface, prescribed burning has successfully reintroduced fire into many fire-adapted plant communities, including small remnants of longleaf pine on sandhills in suburban neighborhoods (Heuberger and Putz 2003; Miller and Wade 2003).

Sustainable longleaf pine ecosystems require frequent surface fires to control competing woody plants, provide favourable seedbed conditions for natural regeneration and maintain herbaceous plant diversity (Brockway and Outcalt 2000; Wade *et al.* 2000; Van Lear *et al.* 2005; Brockway *et al.* 2006; Hiers *et al.* 2007). Frequent prescribed burning also reduces longleaf pine mortality in natural and planted stands, even when wildfires occur during severe drought (Outcalt and Wade 2004). Yet, many fire-excluded pine forests across the southeastern US remain in degraded condition, where a surface fire could severely scorch or rapidly ascend to the canopy. As a remedy, Outcalt (2003) demonstrated how fire can be restored in long-unburned pine forests, through overstorey and midstorey thinning followed by prescribed burning in the spring. Prescribed fire during winter and spring also appears to be least harmful to pine root processes (Sword Sayer and Haywood 2006). While mulching may serve as a short-term component of wildfire hazard mitigation in long-unburned southern pine forests, prescribed fire is a necessary follow-up measure for ecosystem restoration and sustainability.

Foliar cover changes

The cover of tree seedlings, shrubs, vines, grasses and forbs was increased by a likely rise in solar radiation and available soil moisture following mulching. Since recruitment of new and expansion of existing species often follow episodes of disturbance in longleaf pine forests, this response was not unexpected. Not only may dominant species, like pines and grasses, respond (Gilliam and Platt 1999; Cox *et al.* 2004) but blackberries and

Table 4: Understorey plant species richness, diversity and evenness response to mulching and prescribed fire for pre-treatment (2000) and post-treatment (2002 and 2003) years

	Control	Mulching only	Mulching and winter fire	Mulching and spring fire	Mulching and summer fire	Mean of non-control treatments
Number of species						
2000	18.0	19.8	16.5	19.8	18.5	18.7
2002	20.3	27.2†	26.3†	24.8†	20.0	24.6†
2003	20.7	26.0†	21.5	25.3†	24.0†	24.5†
Shannon index						
2000	2.11	2.42	2.28	2.36	2.16	2.31
2002	2.48	2.58	2.52	2.71	2.37	2.55
2003	2.33	2.57	2.22	2.55	2.46	2.45
Modified Hill ratio						
2000	0.63	0.89*	0.91*	0.80	0.84	0.86
2002	0.91†	0.71†	0.77†	0.92	0.90	0.83
2003	0.81	0.72†	0.66†	0.71†	0.70†	0.70†

* Significantly different from control, $P < 0.05$.
† Significant change through time from pre-treatment condition, $P < 0.05$.

other species can also increase rapidly because of their ability to quickly overrun disturbed sites (Cain and Shelton 2003). With the fundamental components of this forest still largely intact, the understorey plant community became increasingly dominated by shrubs and vines (23.6–51.6 per cent cover) along with substantial increases in grasses (4.9–23.3 per cent cover) and forbs (2.7–14.3 per cent cover). This significant post-treatment increase in herbaceous plant cover supports acceptance of hypothesis 1B.

Although grass and forb increases are commonly observed after thinning (Harrington and Edwards 1999), these may be short lived as further stand development diminishes sunlight and soil moisture availability and increases needle fall and forest litter accumulation. Burning at our site reinforced the initial trend of understorey plant increases and will be needed for continuing development of the herbaceous plant community. In some boreal forests, localized disturbance from lightning-caused fire is the principal determinant of understorey vegetation (Nilsson and Wardle 2005). Similarly in longleaf pine forests, a well-developed understorey of pyrogenic species is essential for long-term ecosystem maintenance. In the absence of periodic fire, hardwood species are poised to assert their dominance.

While season of burning had no effect on tree seedlings and the expansion of shrubs and vines was mostly favoured following winter and spring fires, blackberry, grape and beautyberry also increased after summer fire. Although grasses and forbs expanded from burning in all three seasons, greatest response for spikegrass appeared following winter fire and those for fireweed and yankeeweed arose after summer fire. Southern pineland was historically burned during winter to encourage forage production for cattle grazing. Following a period of prolonged fire exclusion in the mid-twentieth century, interest in burning during the growing season increased. The positive response of a broad range of plant groups to burning during any season is evidence that this longleaf pine ecosystem was in poor condition (i.e. dense midstorey clearly suppressed the understorey plant community). Fire during any season is decidedly better than no burning at all.

The decline in tree seedlings (31.3–14.1 per cent cover) was not surprising on control plots since this is often a normal result of the shading of intolerant species during stand development. However, the greatest decrease (17.9–5.5 per cent cover) was noted for laurel oak, a shade-tolerant species. The corresponding increases in beautyberry (0–6.7 per cent cover), spikegrass (0–4.7 per cent cover) and ferns (2.1–12.1 per cent cover) are likely an outcome of their progressive expansion. Perhaps, moisture stress precipitated a replacement of laurel oak by shrubs, grasses or ferns.

Diversity dynamics

Since mulching removed a number of hardwoods and smaller pines, reductions in overstorey and midstorey species richness (51 per cent) and diversity (36 per cent) were not surprising. Mulching can modify the composition and structure of a stand, adjusting its successional trajectory to more rapidly achieve restoration and sustainability. With tree species evenness unchanged, mulching contributed to these goals without altering the equitability of tree species distribution. Thus, the residual overstorey is positioned to make gains towards developing the desired open structure dominated by longleaf pine, following application of recurrent fire (Wade *et al.* 2000).

Significant gains of 30 per cent were noted for understorey plant species richness following mulching and prescribed fire. These resulted largely from the increasing frequency and/or expansion of tree seedlings (sweetgum, yellow-poplar, southern red oak, water oak, sassafras), shrubs (blackberry, beautyberry, wax myrtle, winged sumac, deerberry, partridge pea), vines (Virginia creeper, grape, peppervine, greenbriar, poison ivy), grasses (spikegrass, panicgrass) and forbs (fireweed, yankeeweed, elephantsfoot, wood sorrel, milk-pea, sweet goldenrod, pineweed, pencilflower, bush-clover). These findings support acceptance of hypothesis 1C that reducing stand density through mulching will increase understorey plant diversity. However, prominently responding herbaceous species (spikegrass, panicgrass, fireweed, yankeeweed) were fire-followers and ruderals, capitalizing on an opportunity to occupy a disturbed site, and grasses (*Aristida* spp. L., *Sporobolus* spp. R. Br., *Sorghastrum* spp. Nash) and forbs (*Pityopsis* spp. Nutt., *Aster* spp. L., *Liatris* spp. Schreb., *Tephrosia* spp. Pers.), indicative of healthy longleaf pine ecosystems, were either absent or responded poorly. Also, a significant decline in evenness indicated that the overall equitability among understorey plant species had diminished, likely as a result of the rapid emergence of hardwood seedlings, shrubs and vines following mulching. This underscores the importance of fire for curtailing woody plants and benefiting herbaceous plants, through accentuating the spatial heterogeneity that creates microsites favourable for establishment (Provencher *et al.* 2003; Thaxton and Platt 2006). Given the close linkage between understorey plant diversity and surface fuel accumulation, prescribed fire should be applied as frequently as fuel levels allow and focus on slowly reducing the forest floor (Hiers *et al.* 2007).

The number of native taxa was indicative of the high richness typical in longleaf pine ecosystems. Species richness is largely determined by interspecific competition interacting with site productivity, microsite heterogeneity and disturbance regimes. Disturbance has a major influence on diversity in many forest types and especially in longleaf pine ecosystems (Brockway 1998; Brockway and Outcalt 2000; Dilustro *et al.* 2002). Interruption of disturbance regimes has frequently resulted in negative consequences for diversity (Hooper *et al.* 2005).

Ecosystem restoration

Restoring longleaf pine ecosystems requires establishing appropriate composition within suitable structures that are maintained by ecological processes (Brockway *et al.* 2005a; Van Lear *et al.* 2005). Long-term success demands that all techniques be compatible with recurrent surface fire as a key ecological process, so that an open forest structure having a highly diverse groundcover, dominated by native grasses and forbs, can be developed and maintained (Stanturf *et al.* 2004; Brockway *et al.* 2005b). Frequent fire that favours longleaf pine dominance may also benefit hurricane-prone areas, by enhancing ecosystem resistance to and recovery from large-scale atmospheric disturbance (Provencher *et al.* 2001a; Stanturf *et al.* 2007). In the extensive areas degraded by the presence of highly dense midstories, the initial restoration step is often to reduce overstorey hardwoods by thinning and eliminate the midstorey through mechanical means (Provencher *et al.* 2001b). Once the structure is adjusted and hazardous fuels redistributed to the forest floor, prescribed fire can be used to thwart redevelopment of a high-density midstorey. Mulching is but a short-term option and programmes relying it will be more expensive than those emphasizing prescribed fire. Therefore, we recommend that longleaf pine ecosystems having high-density midstories first be treated to establish a 'fire-safe' structure and then be promptly burned with surface fire at times optimal for inducing mortality among hardwoods, expanding desirable herbaceous species and establishing longleaf pine seedlings.

Funding

US Department of Agriculture and US Department of Interior Joint Fire Sciences Program (99-1-3-06).

Acknowledgements

Special thanks to Robert Larimore and James Parker of the Fort Benning Division of Natural Resources for prescribed fire treatments and administrative cooperation, Heather MacCulloch and Mike Allen for data collection assistance and Dave Haywood, Katherine Elliott and two anonymous reviewers for helpful comments on this manuscript. Mention of trade names does not constitute endorsement by the USDA Forest Service.

Conflict of Interest Statement

None declared.

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Received 12 September 2008