3-2009

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Recommended Citation

Schwilk, Dylan W.; Keeley, Jon E.; Knapp, Eric E.; McIver, James; Bailey, John D.; Fettig, Christopher J.; Fiedler, Carl; Harrod, Richy J.; Moghaddas, Jason J.; Outcalt, Kenneth W.; Skinner, Carl N.; Stephens, Scott L.; Waldrop, Thomas A.; Yaussy, Daniel A.; and Youngblood, Andrew, "The National Fire and Fire Surrogate Study: Effects of Fuel Reduction Methods on Forest Vegetation Structure and Fuels" (2009). *Forest Management Faculty Publications*. 15.
https://scholarworks.umt.edu/forest_pubs/15

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Ecological Applications, 19(2), 2009, pp. 285–304

The national Fire and Fire Surrogate study: effects of fuel reduction methods on forest vegetation structure and fuels

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Abstract. Changes in vegetation and fuels were evaluated from measurements taken before and after fuel reduction treatments (prescribed fire, mechanical treatments, and the combination of the two) at 12 Fire and Fire Surrogate (FFS) sites located in forests with a surface fire regime across the conterminous United States. To test the relative effectiveness of fuel reduction treatments and their effect on ecological parameters we used an information-theoretic approach on a suite of 12 variables representing the overstory (basal area and live tree, sapling, and snag density), the understory (seedling density, shrub cover, and native and alien herbaceous species richness), and the most relevant fuel parameters for wildfire damage (height to live crown, total fuel bed mass, forest floor mass, and woody fuel mass).

In the short term (one year after treatment), mechanical treatments were more effective at reducing overstory tree density and basal area and at increasing quadratic mean tree diameter. Prescribed fire treatments were more effective at creating snags, killing seedlings, elevating height to live crown, and reducing surface woody fuels. Overall, the response to fuel reduction treatments of the ecological variables presented in this paper was generally maximized by the combined mechanical plus burning treatment. If the management goal is to quickly produce stands with fewer and larger diameter trees, less surface fuel mass, and greater herbaceous species richness, the combined treatment gave the most desirable results. However, because mechanical plus burning treatments also favored alien species invasion at some sites, monitoring and control need to be part of the prescription when using this treatment.

Key words: delayed mortality; dry forests; forest management; hazard reduction; prescribed burning; species richness; thinning.

Introduction

Many North American forests that historically experienced frequent low-intensity surface fires have undergone extensive alterations over the past century due to changes in land management. Prominent among them are the loss of Native American burning, increasingly effective fire suppression, timber harvesting, and livestock grazing (Parsons and DeBenedetti 1979, Agee 1993). These and other factors have led to greatly increased forest tree density, a higher proportion of saplings and sub-canopy trees, altered community compositions that favor more shade-tolerant and fire-intolerant tree species, fewer and smaller canopy gaps, elevated surface fuel loads, and/or altered habitats for

Manuscript received 26 October 2007; revised 26 August 2008; accepted 5 September 2008. Corresponding Editor: D. McKenzie. For reprints of this Invited Feature, see footnote 1, p. 283.
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numerous plant and animal species (Leopold et al. 1963, Kilgore 1973, Parker 1984, Covington and Moore 1994, Skinner 1995, Cowell 1998, Taylor 2000, Hessburg and Agee 2003, Frost 2006).

Our increased understanding of forest ecosystems over the past several decades has revealed the vital role natural fires play in the functioning of these ecosystems (Biswell 1973, Van Lear and Waldrop 1989, Stephenson et al. 1991, Agee 1993, Ware et al. 1993, Arno et al. 1997). Fire-induced tree mortality is recognized as an important ecosystem process that varies among tree species (Ryan and Reinhardt 1988) and is influenced by patterns of fire severity (Glitzenstein et al. 1995, Kobziar et al. 2006) and fuel consumption (Stephens and Finney 2002) as well as postfire bark beetle dynamics (McHugh and Kolb 2003, Parker et al. 2006, Fettig et al. 2007). Fire-killed trees are important habitat for wildlife (Farris and Zack 2005) and the resulting gaps in the canopy result in accelerated growth of remaining trees and provide sites for tree regeneration and the establishment of a diverse understory of grasses, forbs, and shrubs (Cooper 1960, Brockway and Lewis 1997, Keeley and Stephenson 2000, Agee and Lolley 2006, Moghaddas et al. 2008).

The ecological health and persistence of many forest types has historically been dependent on natural fires to thin stands and reduce the buildup of surface fuels in order to make forests less susceptible to stand-replacing crown fires (Agee et al. 1977, Parsons and DeBenedetti 1979, Knapp et al. 2005). Although past frequencies of stand-replacing crown fires in landscapes typified by low-severity fire regimes are generally unknown, there is a belief that the probability and spatial extent of such fires now greatly exceeds historical levels (Arno and Brown 1991, Skinner and Chang 1996).

In recent years, unusually large stand-replacing wildfires have heightened public concern and increased recognition of the need for fuel treatments to mitigate fire hazard. Prescription burning and the use of wildland fires have been advocated as management tools for restoring forest structure and reducing fuels (Biswell 1973, Pyne 1982, Stephens and Moghaddas 2005*a*). Legislation such as the Healthy Forests Restoration Act of 2003 (U.S. Public Law 108-148) specifies that the majority of fuel reduction activities will occur within the wildland urban interface. However, prescribed burning is difficult to implement in many of these areas due to concerns regarding aesthetics, air quality, and structural protection (Berry and Hesseln 2004, Liu et al. 2005). In the eastern United States, prescribed fire is more commonly used in wildland urban interface zones, but conditions make it much more complex and limit the areas where it can be applied (Miller and Wade 2003). In areas with limited opportunities for prescribed fire, mechanical thinning treatments are being used as a surrogate for the stand-thinning actions of fire. This is sometimes termed "emulation silviculture" (McRae et al. 2001) or more broadly "emulating natural disturbances" (Crow and Perera 2004).

It is unlikely that the varied ecological roles of wildland fire can ever be entirely replaced by mechanical thinning. However, in today's fuel-rich environments, even prescribed fire may lead to ecological outcomes that differ from historical wildfires. Mechanical harvesting may help to create conditions that allow subsequent prescribed burning (and perhaps wildland fire use) to accomplish fire-related objectives more precisely and rapidly than burning alone, but mechanical treatments may not be able to mimic ecological effects of fire such as soil heating.

Little comparative scientific information on the ecological implications of different fuel treatment options is available to guide management decisions. To address this knowledge gap, a team of federal, state, university, and private scientists designed the Fire and Fire Surrogate (FFS) study, an integrated national network of long-term multidisciplinary experiments to evaluate the ecological effects associated with mechanical thinning and prescribed burning for reducing forest fuels (Weatherspoon 2000, McIver et al. 2009). The FFS study currently consists of a network of 13 sites throughout the United States, representing ecosystems that historically experienced frequent, low-intensity fires. At each site a common experimental design was installed that included three treatments (prescribed fire, mechanical thinning, and mechanical thinning followed by prescribed fire) plus an untreated control. A common sampling protocol was used across sites to study the response of a broad array of variables including forest and fuel structure, bark beetle and pathogen dynamics, wildlife trends, soil properties, and fire behavior. Further details about the FFS network are described in Youngblood et al. (2005) and can be found at the Fires Research and Management Exchange Systems web site.[15]

The FFS network is the largest operational-scale experiment ever funded to study ecological responses to silvicultural treatments designed to reduce fire hazard. Although many studies of ecological responses at individual sites have been published, the FFS experimental design allows unprecedented comparisons to be made at the scale of a national network of sites. One of the central questions being addressed is, for which ecological variables can mechanical treatments act as a surrogate for fire, and for which variables does fire have unique effects? Also, for what variables can generalizations across a broad array of sites be made? In light of these questions, this research investigates treatments from the perspective of competing hypotheses for each response variable under consideration:

1) Treatments show no effect relative to the controls (null hypothesis).

[15] ⟨http://frames.nbii.gov⟩

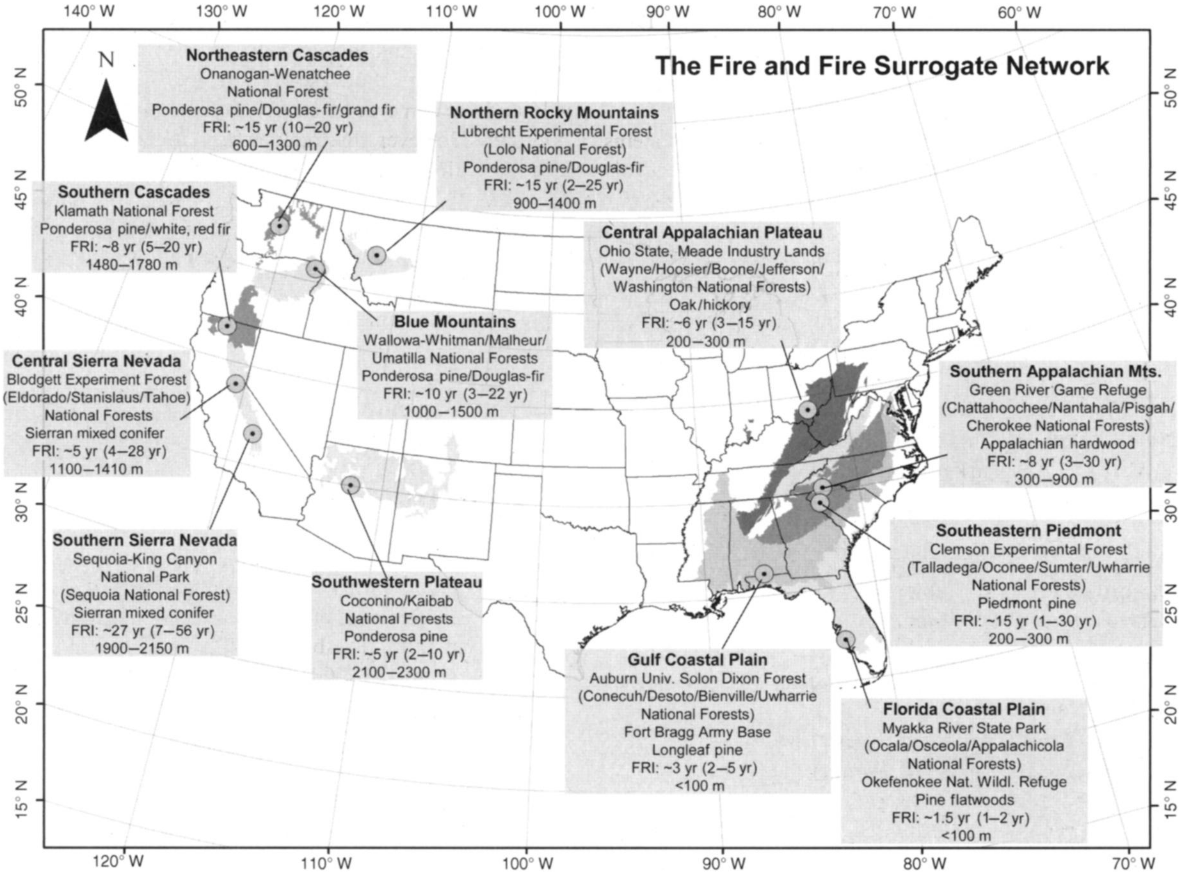

FIG. 1. Name and location of the 12 national Fire and Fire Surrogate (FFS) sites, showing forest type, fire return interval (FRI), and elevational range (m). Shading indicates "representative land base," or the area to which FFS results can be most directly applied for each site. Representative land bases are derived from EPA Type III Ecoregions ⟨www.epa.gov/wed/pages/ecoregions/level_iii.htm⟩.

2) Treatments differ from controls but show few differences among one another.

3) Effects are controlled primarily by burning.

4) Effects are controlled primarily by mechanical treatment.

5) Burning and mechanical treatments have distinct effects.

METHODS

The FFS study was implemented on forests administered by the USDA Forest Service, National Park Service, State Parks, universities, and private industry at 13 sites across the United States (Fig. 1). This paper reports results from 12 of these sites. We report statistical results on the 10 sites that have had a full complement of treatments and with pre- and posttreatment data available for analyses, plus an additional site (South Cascades), where no pretreatment data were collected and effects were instead calculated in relation to the control. The Southern Sierra site lacked mechanical treatments and data were therefore only included in the summary tables (Table 1). No data were available from the 13th site (New Mexico), because of difficulties with treatment implementation.

Four treatments were implemented at 11 of the 12 sites: control (C), untreated control; burn (B), prescribed burning only; mechanical (M), initial and/or periodic mechanical treatment, such as thinning; and mechanical plus burn (MB), mechanical treatment followed by prescribed burning (see Plate 1). At the Southern Sierra Nevada site located in Sequoia-Kings Canyon National Park (Fig. 1), two burn treatments were implemented, early season burn and late season burn, per National Park Service policy. At all sites, each of these treatments was replicated three to four times. These replicates are referred to as "experimental units" and it is at the level of these units that the statistical analyses were conducted. Each experimental unit was at least 10 ha in size and surrounded by a buffer of at least 50 m that received like treatment. All pre- and posttreatment measurements were referenced to a set of fixed points established 40–60 m apart on a grid in the interior of each experimental unit. Vegetation data was collected from multiple subplots within each experimental unit plot (>10 per experimental unit).

TABLE 1. Fire and Fire Surrogate Study site descriptions, treatment methods, plot type, and data collection years used for this analysis.

Site name and location	Mechanical methods	Burn methods
Northeastern Cascades, Okanogan-Wenatchee National Forest, central WA (Harrod et al. 2007)	2001—fell, limb, and buck with chainsaws; yard with helicopter; residue on site	2004—spring under-burn using combination of backing and strip head-fires
Blue Mountains, Wallowa-Whitman National Forest, northeastern OR (Youngblood et al. 2006)	1998—fell, limb, and buck with tracked single-grip harvesters; yard with forwarders; residue left on site	2000—fall under-burn, strip head-fire
Northern Rocky Mountains, University of Montana, Lubrecht Experimental Forest, western MT (Metlen and Fiedler 2006)	2001—fell, limb, and buck with tracked single-grip harvesters; yard with forwarders; residue left on site	2002—spring under-burn, strip head-fire
Southern Cascades, Klamath National Forest, northeastern CA (Ritchie 2005)	2001—fell with feller-buncher; yard whole trees with rubber-tired or tracked skidders	2001—mechanical plus burn, fall under-burn, strip head-fire; 2002—burn only
Central Sierra Nevada, University of California, Blodgett Forest Experimental Station, central CA (Stephens and Moghaddas 2005*a*)	2002—fell, limb, and buck trees >25 cm dbh with chainsaws; lop and scatter tops and limbs; yard with skidders; post-harvest masticate 90% of trees <25 cm dbh	2002—fall under-burn using a combination of backing and strip head-fires
Southern Sierra Nevada, Sequoia National Park, south-central CA (Knapp et al. 2005)	none	2002, 2003—fall and spring under-burn, using strip head-fires
Southwestern Plateau, Kaibab and Coconino National Forests, northern AZ (Converse et al. 2006)	2003—fell, limb, and buck trees > 13 cm dbh with chainsaws; fell and lop trees < 13 cm to waste with chainsaws	2003—fall under-burns conducted as both backing and strip head-fires
Central Appalachian Plateau, Mead Corporation, Ohio State Lands, southern OH (Waldrop et al. 2008)	2001—fell, limb, buck trees > 15 cm dbh with chainsaws	2001—spring under-burns conducted as strip head-fires
Southern Appalachian Mountains, Green River Wildlife Conservation Lands, western NC (Waldrop et al. 2008)	late 2001–early 2002—chainsaw felling all tree stems >1.8 m height and <10.2 cm diameter at breast height (dbh) as well as all shrubs, regardless of size	2003, 2006—winter ground fires were ignited by hand and by helicopter using the strip head-fire and spot fire techniques
Southeastern Piedmont Univ. Clemson Exp. Forest, western SC (Phillips and Waldrop 2008)	late 2000–early 2001—fell with feller-buncher, yard whole trees with rubber-tire skidders, slash distributed across the site	2001, 2004—burn only, winter ground fires ignited by hand using the strip head-fire technique; 2002, 2005—mechanical plus burn
Gulf Coastal Plain Auburn Univ. Solon Dixon Exp. Forest, southern AL (Outcalt 2005)	2002—fell with feller-buncher; chainsaw limb, tree yarded with rubber-tired skidders	2002—spring under-burn, strip head-fire
Florida Coastal Plain Myakka River State Park, west-central FL (Outcalt and Foltz 2004)	2002—chop with marden aerator pulled by four-wheel drive rubber-tired tractor	2000, 2001—spring under-burn, strip head-fire

Notes: Vegetation data were collected from subplots within experimental units (~10 subplots per experimental unit). Abbreviations are: WA, Washington; OR, Oregon; MT, Montana; CA, California; AZ, Arizona; OH, Ohio; NC, North Carolina; SC, South Carolina; AL, Alabama; FL, Florida.

The implementation of prescribed fire and mechanical treatments were different at each site, but the minimum goal in designing all treatments was to achieve stand and fuel conditions such that, if subjected to a head fire under 80th percentile weather conditions, at least 80% of the basal area of the dominant and codominant trees would survive. Stricter requirements of fire hazard reduction (i.e., >80% survival under 80th percentile weather conditions) were used where they were common practice at the local site. For mechanical treatments, each site used a biomass and/or saw-log removal system that was locally applicable to that site. Burning was conducted in the fall or spring based on common local practices, and in both seasons at the southern Sierra Nevada site. The combined treatment (MB) required waiting a full season for fuels to cure before burning at western U.S. sites (Blue Mountains, Northern Rocky Mountains, Northeastern Cascades, Southern Cascades and Southwestern Plateau; Central Sierra Nevada waited 12 months after harvest and five months after mastication). The methods used at each site are summarized in Table 1. Although the application of prescribed fire was fairly uniform among the 12 sites, prescriptions for mechanical treatments varied considerably (Table 1). In particular, trees smaller than 25 cm at the Central Sierra site were masticated to further break down the surface fuels, and the saw palmetto understory at the Florida Coastal Plain site was masticated, leaving the sparse overstory untouched. All other sites used machines to alter the overstory.

Twelve distinct response variables were considered for this paper, with sites having different subsets of data available for use in among-site analyses (see Appendix A). The tree survival data were generally collected within

TABLE 1. Extended.

	Data collection years		
Subplot size and type	Pretreat	First	Final
0.1 ha, Whittaker	2000	2004	NA
0.04 ha, circular	1998	2001	2004
0.1 ha, Whittaker	2001	2002	2005
0.1 ha, Whittaker	NA	2004	NA
0.04 ha, circular	2001	2003	NA
0.1 ha, Whittaker	2001	2002	2004
0.1 ha, Whittaker	2000	2004	NA
0.1 ha, Whittaker	2000	2002	2004
0.1 ha, Whittaker	2001, 2002	2004	2006
0.1 ha, Whittaker	2000, 2001	2002	2003
0.1 ha, Whittaker	2001	2002	2003
0.1 ha, Whittaker	2000, 2001	2001	2003

20×50 m (0.1 ha) modified Whittaker plots (Keeley and Fotheringham 2005), 10 of which were established per experimental unit; with two sites (Central Sierra Nevada and Blue Mountains) using a systematic grid of 0.04 ha circular plots (Table 1). Within plots, all trees >10 cm dbh were labeled with a uniquely numbered tag. Saplings were considered to be the smaller diameter trees (>1.37 m tall but <10 cm dbh) and these were not permanently tagged. Species, status (alive, standing dead, dead and down) and dbh were recorded for all trees and saplings. Total height and height to the base of live crown were measured for each tree. Cover was estimated for grasses, forbs, and shrubs, at multiple subplots in each plot. Mass of woody surface fuel was estimated both prior and following treatment using either Brown's planar intercept method (Brown 1974), with transects established in reference to the network of grid-points within each experimental unit, or a destructive sampling method. Although surface fuels are often defined to include living understory vegetation, we do not have mass estimates or particle size distributions for all species in this study. We evaluate effects on understory vegetation separately from that on dead woody fuels and forest floor mass. For the Brown's method, the number of intersecting downed woody stems in different time-lag size classes (1-hour fuel, 0–6 mm; 10-hour fuel, >6–25 mm; 100-hour fuel, >25–76 mm; and 1000-hour+ fuel, >76 mm) were recorded along each transect. For the 1000-hour+ fuel size category, the diameter and decay class (sound or rotten) of each log were recorded. At 11 of 12 sites, data were taken for all vegetation and fuels variables prior to treatment and one year posttreatment (at Southern Cascades, no pretreatment data were taken, and the first full set of posttreatment data were taken at year two). Most sites also collected a final set of data between two and four years after treatment (Table 1; Appendix A).

Although treatments and measurements were conducted in different calendar years at different sites, we used year of treatment as a point of reference to place measurements into a common temporal scale. Response variables were expressed as the difference between pre- and posttreatment experimental unit means. Multiple measurements within an experimental unit were averaged to provide values for each replicate plot. Differences pre- to posttreatment were used to control some of the spatial variation among experimental units within sites. An alternative method for dealing with this variation would be to use pretreatment values as covariates in the models. Our exploration of a subset of the results shows that both methods gave similar results. We have presented results based on pretreatment vs. posttreatment differences, rather than percentage change, because the former method preserves the original measurement units. Data for some response variables were not collected at all sites, and these sites were therefore dropped from those analyses (Appendix A). To allow graphical presentation and meet assumptions of linear models, some data were log-transformed. Because pre- to posttreatment differences could be negative, when posttreatment − pretreatment values were $\log_{10}$-transformed, we have presented the changes as positive or negative according to the sign of the original difference. In other words, for a post − pre difference, Δx, the transformed differences = $\text{sign}(\Delta x) \log_{10}|\Delta x|$. Zero difference before transformation was set as zero.

Data on tree size structure were analyzed in two ways: (1) using the actual dbh of every tree within the vegetation plots at each site and (2) by grouping trees into site-specific size classes. To compare treatment effects on various size classes of trees across the network, each site categorized its trees (>10 cm dbh) into four relative size classes (Appendix B). In this way, we could compare treatment effects on large vs. small trees across the network despite the large differences in average tree size across sites.

To test the relative importance of the treatment factors on our response variables, we used an information-theoretic approach. In addition to treating each treatment as a completely separate effect, we created three additional two-level factors from treatment combinations: MECH (presence/absence of mechanical treatment), BURN (presence/absence of burning), and FUEL (presence/absence of any treatment other than control). Using these additional factors we produced five models to test: four one-factor models and one two-factor model. In addition, each model included study site as a random nesting effect. Each tested model corresponds to alternative hypotheses listed in the introduction. The tested models are listed below with Y representing the response variable (change in value pre treatment to post treatment).

1) $Y = \text{SITE}$: Response depends only on the random site effect and there are no consistent treatment effects (null model).

2) $Y = \text{FUEL} + (\text{SITE})$: Response depends upon presence or absence of fuel treatment: two levels (control vs. all other treatments). Model implies that burning and mechanical treatment effects are not distinguishable from one another.

3) $Y = \text{BURN} + (\text{SITE})$: Response depends upon presence or absence of burning. Model implies that mechanical treatments had little effect.

4) $Y = \text{MECH} + (\text{SITE})$: Response depends upon the presence of mechanical treatment. Model implies that burning had little effect.

5) $Y = \text{BURN} + \text{MECH} + (\text{SITE})$. Burning and mechanical treatments having separate, additive effects. This is the two-factor model and implies that effects were of different magnitude and potentially of different sign.

All analyses were carried out with the R software package (R Development Core Team 2005). The models were fitted using the linear modeling ("lmer") function of the R Matrix package by Douglas Bates. This procedure allowed us to use likelihood based information theoretic methods to evaluate this set of competing models. We used Akaike's information criterion adjusted for small sample size (AIC_c) to evaluate models (Burnham and Anderson 2002). Models were fitted with maximum likelihood procedures, but fitting with restricted maximum likelihood procedures produced nearly identical results and did not change model rankings. For each response variable, the model was selected from the five competing models based on relative AIC weights. By explicitly testing the null model (model 1) and the model which groups all fuel treatments together (model 2) we can distinguish these two patterns from one another as well as from the case where a lack of consistent pattern across sites results in little ability to distinguish competing models.

In an effort to condense the considerable amount of information represented in this paper, patterns of change are presented for clusters of variables representing each stratum of the ecosystem analyzed (i.e., overstory plants, understory plants, and fuels). Although all vegetation may act as fuel, here we include downed woody fuels and forest floor mass as surface fuel variables and height to live tree crown as a measure of ladder fuels. We used box plots to exhibit among-treatment patterns, directional change tables to show among-site patterns, and a summary table showing the best AIC_c model fit for each response variable.

Results

Fuel treatments had a substantial effect on all ecological variables presented in this paper. The null model with among-site variation as the only factor had the least or close to the least explanatory power of nearly all models tested. Results from the first year posttreatment are presented first, followed by results from posttreatment years two through four. Analyses of change in fuel variables are only given for the first posttreatment measurement, because most sites did not collect fuels data twice after treatment, and change between the first and second posttreatment remeasurement at sites that did was relatively minor.

First posttreatment measure

Trees and saplings.—Not surprisingly, density of trees in all size classes was generally lower in fuel treatment units than in the controls (Fig. 2A). The model with the strongest support included separate and additive effects for burning and mechanical treatments (Table 2). Mechanical treatments (M and MB) had greater effects on tree density than burning alone, particularly for the medium and large tree size classes (Fig. 2A). Only the Southern Sierra site experienced a sharp decline in tree density with B. The M and MB significantly reduced tree densities at most sites except Florida Coastal Plain (Table 3), where mechanical treatments were not used to influence the overstory.

Burning was much more effective at reducing the numbers of saplings than at reducing the number of larger trees (Fig. 2B). For this sapling size class (see Appendix B), the burning and mechanical treatments had effects of similar magnitude and the model with strongest support did not distinguish among fuel treatments (Table 2): all fuel treatments reduced sapling numbers similarly. Of the 10 sites that compared mechanical and burning treatments in the first year posttreatment, only two (Blue Mountains, Southwest Plateau) showed no change in sapling density with either B or M treatments (Table 3), but both of these sites reported lower sapling density after the combined treatment.

Basal area followed a pattern similar to tree density, with mechanical treatments causing a greater reduction in basal area than did burning alone (Fig. 2C), but with MB having the greatest overall effect (Table 2). This result is not surprising, because at most sites some medium and large trees, which contribute dispropor-

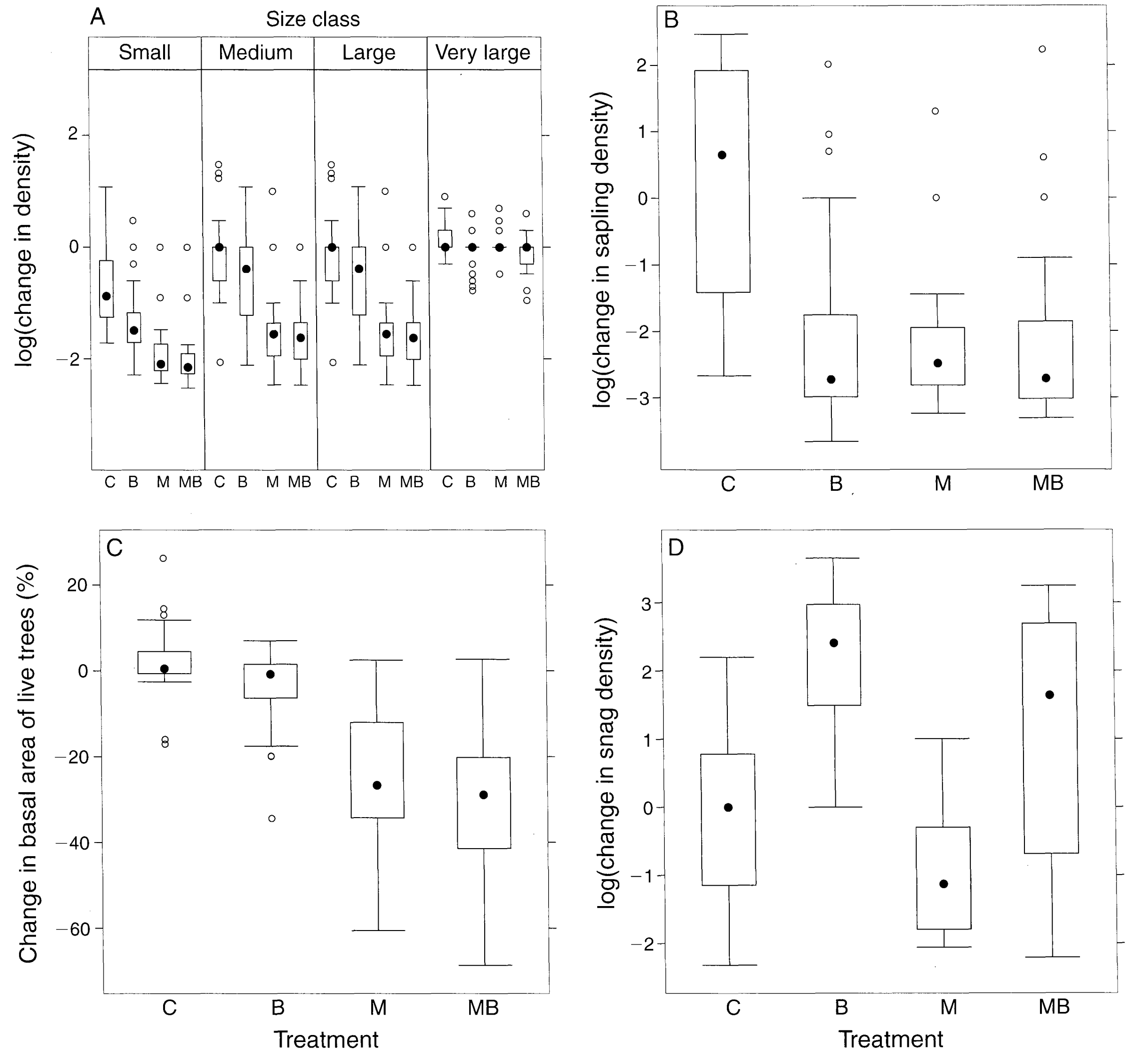

FIG. 2. Overstory changes from pretreatment to the first year posttreatment for (A) tree density (no./ha, log-transformed); (B) sapling density (no./ha, log-transformed); (C) basal area (m^2/ha); and (D) snag density (no./ha, log-transformed). Treatment abbreviations are: C, control; B, burn; M, mechanical; and MB, mechanical + burn. Plots show the median (solid circle), the second to third quartiles (box), and the minimum and maximum values (whiskers) excluding outliers which are shown as individual points (open circles). Outliers were defined as $\geq 3 \times$ IQR (interquartile range) above the third quartile or below the first quartile.

tionately to basal area, were targeted for removal. The M treatment resulted in lower basal area at seven of 10 sites for which posttreatment data were available (Table 3), with three sites in the southeast United States (Southern Appalachian Mountains, Southeast Piedmont, Florida Coastal Plain) showing no change. At the Southeast Piedmont site, basal area was reduced by MB, but surprisingly at the Florida Coastal Plain site, where no overstory trees were removed, this treatment slightly increased basal area.

Density of snags (standing dead saplings and trees) generally followed a different trajectory for mechanical and burning treatments (Fig. 2D). Snag density increased with burning, especially for B, while it either declined or was unchanged at most sites after M. Model results thus showed additive effects of opposite sign for burning and mechanical treatments (Table 2). MB had variable effects among sites (Table 3).

Understory vegetation.—There were few trends in the effect of fuel treatments on tree seedling density (Table 4). There was a tendency towards slightly higher seedling density after all fuel treatments, with the model including fuel treatment as the effect having the greatest support (Fig. 3A). However, there was little ability to distinguish among models due to very high variability in seedling density among sites (Tables 2 and 4).

There was no treatment effect on total understory cover in the first posttreatment year (data not shown). All three active fuel treatments led to small decreases in percent shrub cover at most sites (Fig. 3B; Table 4). The model allowing separate effects of burning and mechanical treatments showed the strongest support, but was

TABLE 2. Summary of AIC model selection results for all response variables for change between pretreatment and first posttreatment measurement year.

Response variable	Superior model†	AIC_c relative weight of best model
Change in tree density (by size class)	Y = SIZE_CLASS + BURN (−0.29) + MECH (−0.90)	1.0
Change in basal area	Y = BURN (−0.07) + MECH (−0.29)	1.0
Change in quadratic mean tree diameter	Y = BURN (2.04) + MECH (3.06)	1.0
Change in snag density	Y = BURN (2.02) + MECH (−1.06)	0.99
Change in sapling density	Y = FUEL (−2.44)	1.0
Change in seedling density	Y = BURN	0.46
Change in shrub cover	Y = BURN (−4.86) + MECH(−2.76)	0.70
Change in herbaceous species richness	Y = MECH	0.33
Change in alien species richness	Y = MECH	0.33
Change in total surface fuels	Y = BURN (−0.97) + MECH (0.34)	0.89
Change in height to base of live crown	Y = BURN (1.36) + MECH (0.96)	0.99

Notes: The superior model was selected from the group of five competing models listed in the *Methods* section. Results are grouped according to which competing model was judged superior for that response variable. Site was included in all models as a random effect but was omitted from the table for conciseness. Low weights relative to the best model generally indicate little consistent pattern and therefore little ability to distinguish among competing models.

† Estimated coefficients of fixed effects are in parentheses. Coefficients are omitted when best model weight < 0.7.

only marginally stronger than the fuel treatment model (i.e., fuels treatments behaved similarly with respect to decreasing shrub cover, Table 2).

Herbaceous species richness showed no clear trend in response to fuel reduction treatments one year posttreatment (Fig. 3C; Tables 2 and 4). In fact, herbaceous species richness at two sites (Northern Rockies, Florida Coastal Plain) tended to increase in the controls between measurement periods. Although alien herbaceous species richness sometimes increased in the year after fuel treatment (Fig. 3D), effects were subtle and sites varied in response (Table 4). Hence, there was little ability to distinguish among competing models other than to reject the null model (Table 2, null model had AIC_c relative weight approaching zero, result not shown).

Fuels (woody surface fuels and crown height).—Height to base of live crown, a measure that provides some indication of the effectiveness of treatments to reduce ladder fuels (Agee and Skinner 2005), tended to increase with fuel treatments (Fig. 4A) (higher height to base of live crown). Changes in height to live crown were similar to those for tree density as both B and M increased height to live crown, with B having an effect of slightly larger magnitude (Table 2). As a consequence, MB had the largest overall effect, increasing height to live crown more than either of the other individual treatments (Fig. 4A). Site differences were variable, but showed that MB had the most consistent effect (Table 5).

There was an overall reduction in total surface fuel load immediately after the burning treatments, with B having the greatest effect overall (Fig. 4B), and the most consistent effect among sites (Table 5). M tended to increase surface fuel load (Fig. 4B), because of the production of slash fuels (<7.6 cm diameter woody material). When mechanically-treated stands were burned (MB), total fuel loads declined but not as much as with B (Table 5). Overall, the changes in woody fuel mass obscures a major difference between western and eastern U.S. sites (Fig. 1) as western sites contained less live understory biomass and proportionally more sur-

TABLE 3. Change in trend for live tree density, sapling density, basal area, and snag density between pretreatment and first year posttreatment means for 12 Fire and Fire Surrogate (FFS) sites, for control (C), burn (B), mechanical (M), and mechanical + burn (MB) treatments.

	Live tree density (no./ha)				Sapling density (no./ha)				Basal area (m^2/ha)				Snag density (no./ha)			
Site	C	B	M	MB	C	B	M	MB	C	B	M	MB	C	B	M	MB
Northeast Cascades	0	0	↓	↓	0	0	↓	↓	0	0	↓	↓	0	0	0	0
Blue Mountains	0	0	↓	↓	0	0	0	↓	0	0	↓	↓	0	↑	0	0
Northern Rockies	0	0	↓	↓	0	0	↓	↓	0	0	↓	↓	0	↑	0	↑
Southern Cascades	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Central Sierra	0	0	↓	↓	0	↓	↓	↓	↑	0	↓	↓	0	↑	↓	↓
Southern Sierra	0	↓	NA	NA	0	↓	NA	NA	0	↓	NA	NA	0	↑	NA	NA
Southwest Plateau	0	0	↓	↓	0	0	0	↓	0	0	↓	↓	0	↑	↓	0
Central Appalachian Plateau	0	0	↓	↓	0	↓	↓	↓	0	0	↓	↓	↓	0	↓	↓
Southern Appalachian Mountains	NA	0	0	↓	0	↓	↓	↓	NA	0	0	0	NA	↑	0	↑
SE Piedmont	0	0	↓	↓	0	↓	0	↓	0	0	0	↓	0	↑	0	↑
Gulf Coastal Plain	↓	0	↓	↓	0	↓	↓	↓	0	0	↓	↓	↓	↑	↓	↑
Florida Coast Plain	0	0	0	0	↑	↑	↓	0	0	0	0	↑	0	↑	0	0

Notes: Key to symbols: ↑, increase; ↓, decrease; 0, no trend change for indicated variable, with trend indicated by nonoverlapping standard errors. NA indicates that data were not available.

TABLE 4. Trend for seedling density, shrub cover, native herbaceous species richness, and alien species richness between pretreatment and first year posttreatment means for 12 FFS sites, for control (C), burn (B), mechanical (M), and mechanical + burn (MB) treatments.

	Seedling density (no./ha)				Shrub cover (%)				Native herbaceous species richness				Alien herbaceous species richness			
Site	C	B	M	MB	C	B	M	MB	C	B	M	MB	C	B	M	MB
Northeast Cascades	NA	NA	NA	NA	0	0	0	↓	NA	NA	NA	NA	NA	NA	NA	NA
Blue Mountains	↑	↑	↑	0	↑	0	↑	0	NA	NA	NA	NA	NA	NA	NA	NA
Northern Rockies	0	0	0	↓	↓	↓	↓	↓	↑	0	↑	0	0	0	0	0
Southern Cascades	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Central Sierra	0	↓	0	↓	0	0	0	↓	NA	NA	NA	NA	NA	NA	NA	NA
Southern Sierra	↑	↑	NA	NA	0	↓	NA	NA	0	0	NA	NA	0	0	NA	NA
Southwest Plateau	NA	NA	NA	NA	0	0	0	0	0	0	0	0	0	0	↑	↑
Central Appalachian Plateau	0	↑	0	↑	0	↓	↓	↓	0	0	0	↑	NA	NA	NA	NA
Southern Appalachian Mountains	0	↑	0	↑	0	↓	0	↓	0	↑	0	0	NA	NA	NA	NA
Southeast Piedmont	0	0	0	↑	0	0	↓	0	0	↑	0	↑	0	0	0	0
Gulf Coastal Plain	0	0	0	0	0	↓	↓	↓	0	0	0	0	0	0	↑	0
Florida Coast Plain	0	0	↑	↓	↓	↓	↓	↓	↑	↑	0	↑	0	0	0	0

Notes: Key to symbols: ↑, increase; ↓, decrease; 0, no change for indicated variable, with trend indicated by nonoverlapping standard errors. NA indicates that data were not available.

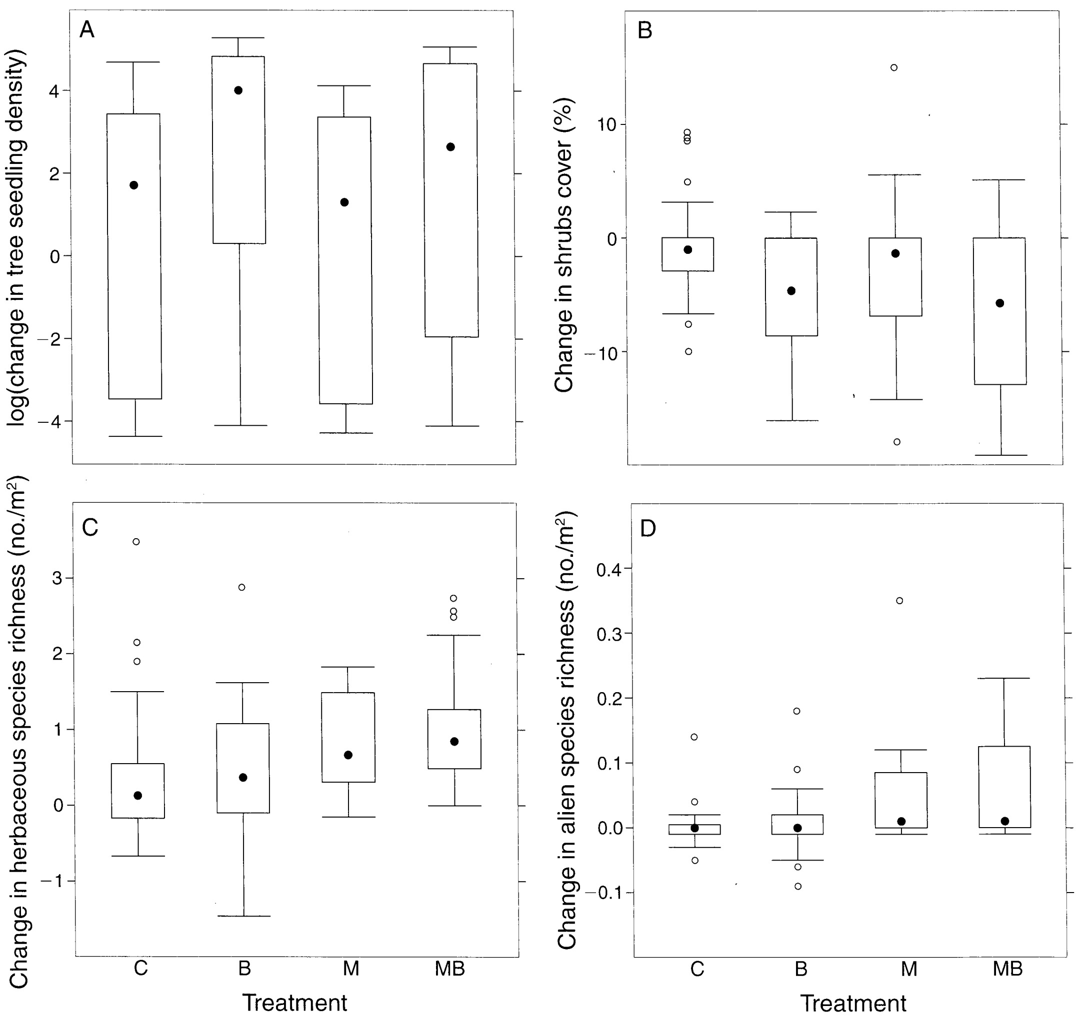

FIG. 3. Understory changes from pretreatment to the first year posttreatment for (A) seedling density (no./ha, log-transformed); (B) shrub cover (%); (C) native herbaceous species richness (no./m^2); and (D) alien species richness (no./m^2).

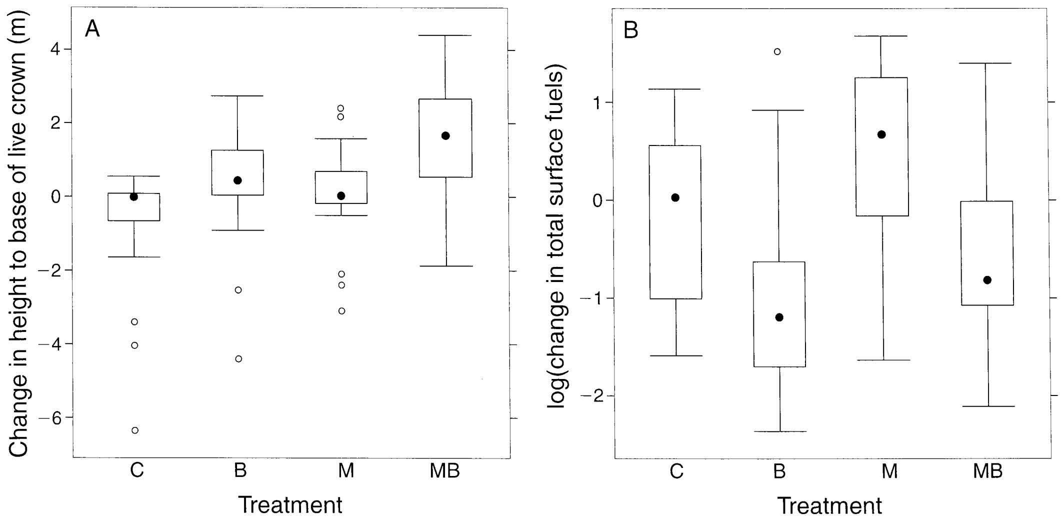

FIG. 4. Changes from pretreatment to the first year posttreatment for (A) height to live crown (m) and (B) woody surface fuel mass (Mg/ha, log-transformed).

face fuel woody biomass than eastern sites and burning consumed this woody biomass. In fact, B did not result in decreases in woody fuel mass at any of the eastern sites, yet was the most influential treatment at all of the western sites (Table 5). Model results thus showed additive effects of opposite sign for burning and mechanical treatments (Table 2).

Final posttreatment measure

Final year of measurement was 2–4 years after treatment and varied by site and by response variable (Table 1; Appendix A). For overstory variables, the means for some variables changed between the first and final posttreatment remeasure, but the relative ranking of the different models did not. For understory variables, there were several changes in the pattern observed between subsequent assessments.

Trees and saplings.—Tree density continued to decline in B and MB two to four years posttreatment, with little additional mortality in M (Table 7) and with burning and mechanical treatments having cumulative effects of similar magnitude by the final measurement year (Table 6). Tree density at the Southeast Piedmont sites declined with all treatments, including C, due to a southern pine beetle (*Dendroctonus frontalis*) outbreak in this and adjacent areas (Boyle 2002). Due to additional mortality of small trees in burning treatments (Fig. 5A), tree density changes became more similar among treatments by the final year posttreatment despite burning leading to lower initial mortality than did mechanical treatments. Changes in quadratic mean diameter also become more similar across all treatments by

TABLE 5. Trend in height to live crown, total fuel mass (sum of forest floor mass and woody fuel mass), forest floor fuel mass, and woody fuel mass between pretreatment and first year posttreatment means for 12 FFS sites, for control (C), burn (B), mechanical (M), and mechanical + burn (MB) treatments.

	Height to live crown (m)				Total surface fuel mass (Mg/ha)				Forest floor mass (Mg/ha)				Woody fuel mass (Mg/ha)			
Site	C	B	M	MB	C	B	M	MB	C	B	M	MB	C	B	M	MB
Northeast Cascades	0	↑	0	↑	0	↓	↑	0	0	↓	↑	0	0	↓	↑	0
Blue Mountains	↓	0	↓	0	NA	↓	0	↓	NA	0	0	0	NA	↓	↑	↓
Northern Rockies	0	0	0	↑	NA	↓	↑	↓	NA	↓	0	↓	NA	↓	↑	0
Southern Cascades†	NA	NA	NA	NA	NA	↓	0	↓	NA	↓	0	↓	NA	↓	↑	↓
Central Sierra	↓	0	0	↑	0	↓	0	↓	↓	↓	↓	↓	0	↓	0	↓
Southern Sierra	0	↑	NA	NA	0	↓	NA	NA	0	↓	NA	NA	0	↓	NA	NA
Southwest Plateau	0	0	0	↑	NA	↓	0	↓	NA	↓	0	↓	NA	↓	↑	0
Central Appalachian Plateau	0	0	0	0	0	0	↑	0	0	↓	0	0	0	0	↑	↑
Southern Appalachian Mountains	NA	0	NA	0	NA	↓	NA	0	NA	0	NA	0	0	0	0	0
SE Piedmont	0	0	0	0	0	↓	0	0	0	↓	0	0	0	0	0	0
Gulf Coastal Plain	0	0	0	↑	0	0	0	↓	0	↓	↓	↓	↓	0	↑	0
Florida Coast Plain	0	0	0	↑	0	0	0	0	0	0	0	0	0	0	0	0

Notes: Key to symbols: ↑, increase; ↓, decrease; 0, no trend change for indicated variable, with trend indicated by nonoverlapping standard errors. NA indicates that data were not available.

† Change estimate for active treatments at Southern Cascades site substitutes control for pretreatment.

Table 6. Summary of AIC model selection results for all response variables for change between pretreatment and second posttreatment measurement year (2–4 years posttreatment).

Response variable	Superior model†	AIC_c relative weight of best model
Change in tree density (by size class)	Y = SIZE_CLASS + BURN (−0.25) + MECH (−0.21)	1.0
Change in basal area	Y = BURN (−0.09) + MECH (−0.23)	0.96
Change in quadratic mean tree diameter	Y = BURN (2.1) + MECH (2.8)	0.98
Change in snag density	Y = BURN	0.52
Change in sapling density	Y = FUEL	0.37
Change in seedling density	Y = (site)	0.37
Change in shrub cover	Y = BURN	0.38
Change in herbaceous species richness	Y = BURN (0.82) + MECH (0.73)	0.99
Change in alien species richness	Y = BURN + MECH	0.56
Change in height to base of live crown	Y = BURN (1.36) + MECH (0.96)	0.99

Notes: The superior model was selected from the group of five competing models listed in the *Methods* section. Results are grouped according to which competing model was judged superior for that response variable. Site was included in all models as a random effect but omitted from the table for conciseness. Low weights relative to the best model generally indicate little consistent pattern and therefore little ability to distinguish among competing models.

† Estimated coefficients of fixed effects are in parentheses. Coefficients are omitted when best model weight < 0.7.

the final posttreatment year (Fig. 6). Comparison of quadratic mean diameter demonstrated that tree size distributions shifted with time in all fuel treatments, with stands becoming increasingly dominated by larger trees. Although M and MB still produced stands with distinctly larger quadratic mean diameter, the difference between these two treatments and B diminished with time (Fig. 6).

Sapling density continued to decline in burning treatments at western U.S. sites, but the response was opposite at eastern sites, with large increases noted across all treatments at four of five sites by the final year of measurement (Table 7). Increases in sapling density at the eastern sites were particularly marked for M and MB (Fig. 2; Fig. 5). Consequently, there was little ability to distinguish among models (Table 6).

Basal area changes mirrored those observed for quadratic mean diameter, with B approaching M and MB over time (Fig. 5C). Trajectories for basal area remained similar between measurement times, with primarily minor additional declines in B at some of the eastern sites (Table 7). Model results for basal area, comparing immediate posttreatment results, and the response measured two to four years after treatment, were nearly identical (Table 6).

Patterns of change for snag density remained similar for the final posttreatment measurements, with snag density highest in the burning treatments (Fig. 5D). However, increases in the number of snags in M and C at the Southeast Piedmont and Southern Appalachian sites led to a slight shift in the overall direction of change in these treatments (Table 7). In the analysis of change by final measurement year the ability to distinguish among competing models decreased with support shifting from a model that showed additive effects of opposite sign for burning and mechanical treatments (Table 4), to a model that estimated an effect of burning alone (Table 6).

Understory vegetation.—By the final year, tree seedling density had increased in all treatments, especially C

Table 7. Trend in live tree density, sapling density, basal area, and snag density between pretreatment and second to fourth year posttreatment means for 12 FFS sites, for control (C), burn (B), mechanical (M), and mechanical + burn (MB) treatments.

	Live tree density (no./ha)				Sapling density (no./ha)				Basal area (m^2/ha)				Snag density (no./ha)			
Site	C	B	M	MB	C	B	M	MB	C	B	M	MB	C	B	M	MB
Northeast Cascade	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Blue Mountains	0	0	↓	↓	↑	0	0	↓	0	0	↓	↓	0	↑	0	0
Northern Rockies	0	0	↓	↓	0	0	0	↓	0	0	↓	↓	0	↑	0	↑
Southern Cascades†	NA	0	↓	↓	NA	↓	↓	↓	NA	0	↓	↓	NA	↑	↓	↓
Central Sierra	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Southern Sierra	0	↓	NA	NA	0	↓	NA	NA	0	↓	NA	NA	0	↑	NA	NA
Southwest Plateau	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Central Appalachian Plateau	0	0	↓	↓	0	0	↑	↑	0	0	↓	↓	↓	↓	↓	↓
Southern Appalachian Mountains	0	0	0	↓	↑	↑	↓	↑	0	0	0	↓	↑	↑	↑	↑
Southeast Piedmont	↓	↓	↓	↓	0	0	0	↑	↓	↓	↓	↓	↑	↑	↑	↑
Gulf Coastal Plain	0	0	↓	↓	↑	↓	↑	0	0	0	↓	↓	↓	↑	↓	0
Florida Coast Plain	0	0	0	0	0	↑	↓	0	0	0	0	0	↓	0	0	0

Notes: Key to symbols: ↑, increase; ↓, decrease; 0, no change for indicated variable, with trend indicated by nonoverlapping standard errors. NA indicates that data were not available.

† Change estimate for active treatments at Southern Cascades site substitutes control of second year posttreatment for all pretreatment.

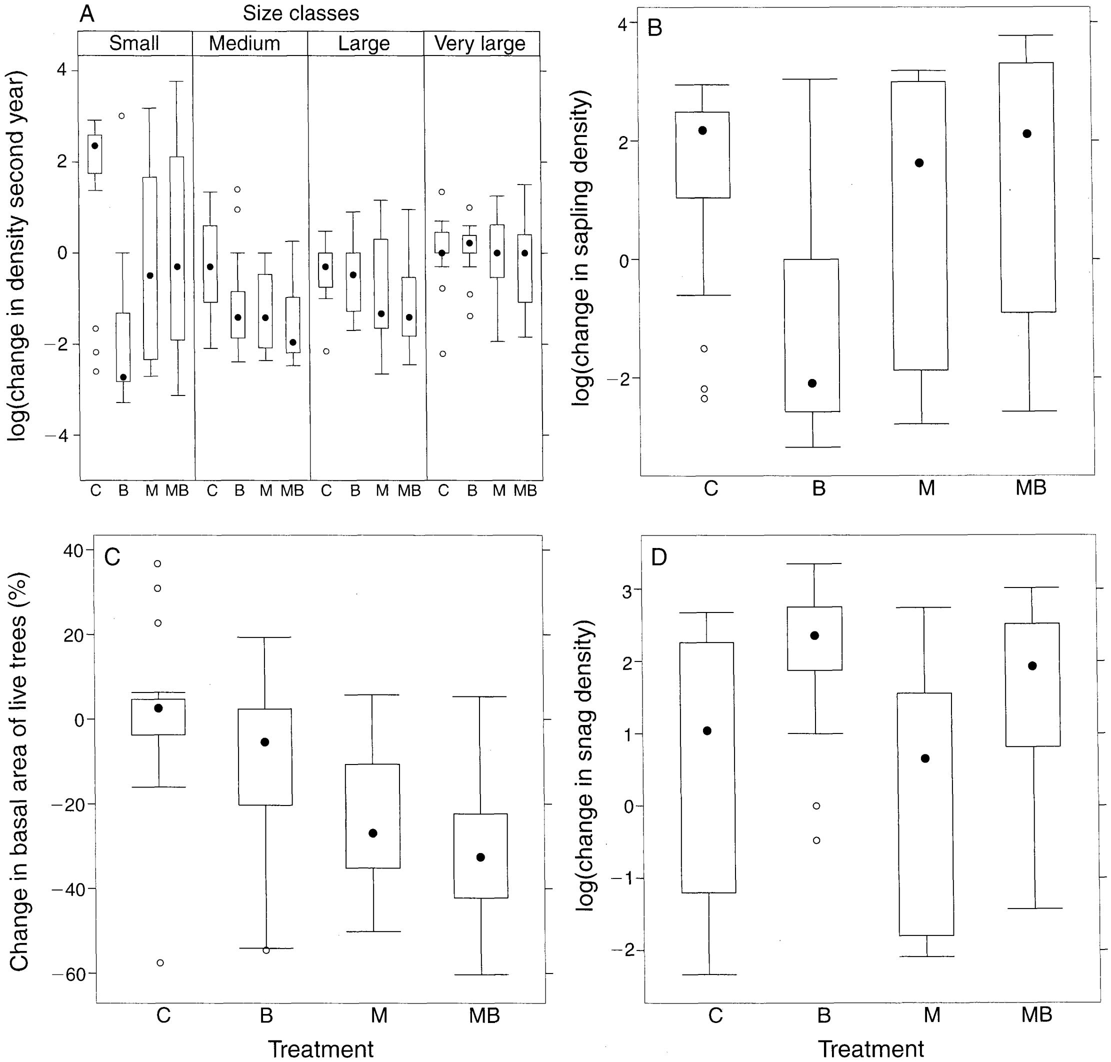

FIG. 5. Overstory changes from pretreatment to the second, third, or fourth year posttreatment for (A) tree density (no./ha, log-transformed); (B) sapling density (no./ha, log-transformed); (C) basal area (m^2/ha); and (D) snag density (no./ha, log-transformed). See Table 1 for details on site-specific posttreatment measurement years.

and M (Fig. 7A; Table 8). Model results showed that response was so variable among treatments and across the network, that there was little ability to generalize treatment effects (Table 6).

Shrub cover had begun to recover at most sites, compared to first year results (Table 8), and the recovery was generally strongest in mechanical treatments, which tended to differ little from C by the final measure (Fig. 7B). This pattern is reflected in the model-selection results that show burning as the best predictor of shrub cover by the final measurement (Table 6).

Although sites continued to vary in overall response, most showed increases in species richness between pretreatment and final posttreatment measurements (Table 8). By the final year posttreatment, overall native herbaceous species richness had increased in all fuel treatments, with B and M showing effects of nearly equal magnitude, and MB showing the greatest relative increase (Fig. 7C) compared to first-year results. Although alien herbaceous species richness remained low by the final posttreatment year, response was variable across sites. The greatest increases occurred in MB (Fig. 7D). The favored model reflected this observation, with the model allowing separate effects of burning and mechanical treatments being the best fit (Table 6).

DISCUSSION

The Fire and Fire Surrogate study was designed to provide information for both fire and resource managers about ecological responses to different fuel treatment options. Specifically, we were interested in determining if

PLATE 1. Treatments in the national Fire and Fire Surrogate study involved under-burning and some form of mechanical fuel reduction, such as this forwarder working with a single-grip harvester to remove small-diameter ponderosa pine at the northeastern Oregon study site. Photo credit: A. Youngblood.

mechanical treatments were capable of emulating important ecosystem processes historically associated with relatively frequent fires and the extent to which prescribed burning alone could produce stand structures more resilient to disturbance. The purpose of this paper was to evaluate whether broad generalizations could be made about responses of certain ecological variables to fire and mechanical treatments across a network of 12 forest sites with surface fire regimes spanning the United States. While site-level information is important, statistically valid generalizations based on sound scientific data from a broad network of research sites is exceptionally useful to managers and policy makers.

Trees and saplings

Network-wide response of stand structure variables such as tree density, basal area, and quadratic mean diameter to treatments showed that greater change was produced by mechanical treatments than by burning. While mechanical treatments were generally a thinning from below, focused mainly on smaller trees, more and larger trees were removed than were killed by prescribed burning. Prescribed burns were quite effective at reducing the density of saplings, but generally did not kill as many of the moderate or larger trees as were removed in the mechanical thinning operations, partially because prescribed burns are typically conducted under mild conditions when the risk of extreme fire behavior and escape are low, and partially because many decades of fire suppression have allowed trees to grow to a size where they are less susceptible to mortality under these burning prescriptions (Miller and Urban 2000). Therefore, mechanical treatments may not be a surrogate for a single prescribed burn in today's fuel and forest stand

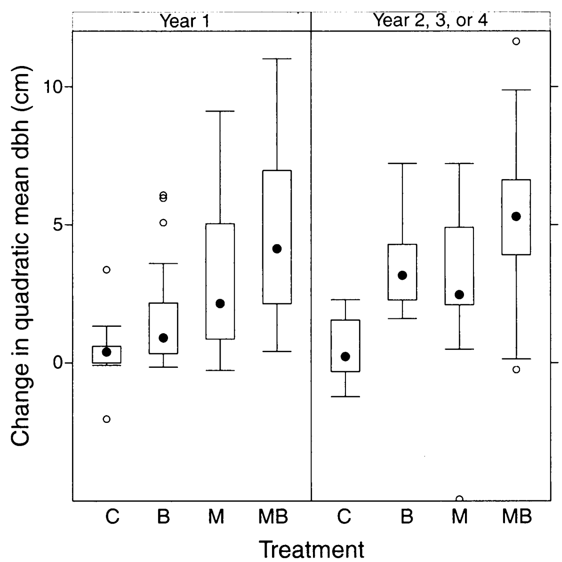

FIG. 6. Quadratic mean tree diameter change between pretreatment and posttreatment year 1 and pretreatment and posttreatment years 2, 3, or 4. See Table 1 for details on site-specific posttreatment measurement years.

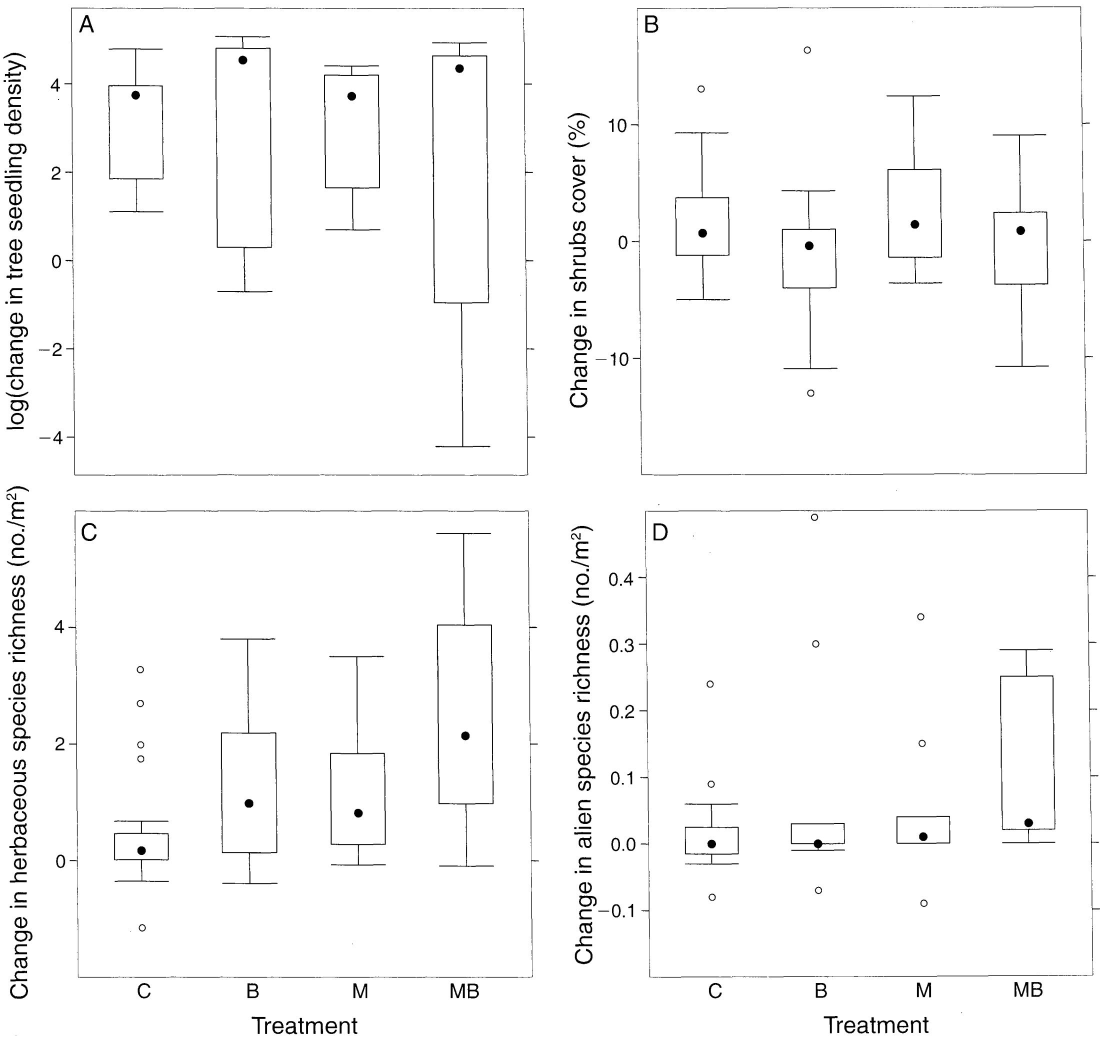

FIG. 7. Understory changes from pretreatment to the second, third, or fourth year posttreatment for (A) seedling density (no./ha, log-transformed); (B) shrub cover (%); (C) native herbaceous species richness (no./m^2); and (D) alien species richness (no./m^2). See Table 1 for details on site-specific posttreatment measurement years.

conditions. However, it is critical to compare the resulting stand structure variables to desired future conditions that are resilient to perturbations such as wildfire, insect outbreaks or climate change. Although tree density continued to decline over time in B as a result of secondary mortality associated with cambial damage and insect attacks, many of the experimental units still contained much higher stand densities than occurred pre-settlement. For example, reconstructions show that stand density was generally less than 100 trees/ha in many seasonally dry forest types of the western United States (Covington and Moore 1994, Harrod et al. 1999, Stephens 2000, Taylor 2004, Youngblood et al. 2004), less than the stand density produced by mechanical treatments and much less than produced by B in this study. Following mechanical treatments with burning (MB) led to some additional tree mortality at all sites, producing a tree density closest to, but still higher than historical numbers. Similarly, for longleaf pine sites in the Southeast, posttreatment tree densities were generally still higher than historical descriptions (Schwarz 1907). It is important to note that burning treatments may lead to additional tree mortality not captured by these short-term experiments. Bark beetles are an important mortality agent following fires in coniferous forests (Fettig et al. 2007), because sublethal heating of plant tissues can increase the susceptibility of insect attack. While bark beetle mortality is generally greatest in the first year after a fire (Schwilk et al. 2006), delayed mortality may continue for several years (Mutch and Parsons 1998, Parker et al. 2006). If burning were the only treatment option, it is possible that multiple burns, each leading to some additional tree mortality may eventually produce a stand structure closer to historical norms, but the size of trees that have grown during an era of fire suppression

TABLE 8. Trend in seedling density, shrub cover, herbaceous species richness, and alien species richness between pretreatment and second to fourth year posttreatment means for 12 FFS sites, for control (C), burn (B), mechanical (M), and mechanical + burn (MB) treatments.

	Seedling density (no./ha)				Shrub cover (%)				Native herbaceous species richness				Alien herbaceous species richness			
Site	C	B	M	MB	C	B	M	MB	C	B	M	MB	C	B	M	MB
Northeast Cascades	NA	NA	NA	NA	NA	NA	NA	NA	0	0	↑	↑	0	0	0	↑
Blue Mountains	↑	0	↑	0	↑	0	↑	0	NA	NA	NA	NA	NA	NA	NA	NA
Northern Rockies	0	0	0	↓	↑	↑	0	0	↑	0	↑	↑	0	0	0	↑
Southern Cascades†	NA	↑	0	↓	NA	0	0	0	NA	0	0	↑	NA	0	0	↑
Central Sierra	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Southern Sierra	0	↓	NA	NA	↓	↓	NA	NA	0	0	NA	NA	0	0	NA	NA
Southwest Plateau	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Central Appalachian Plateau	↑	↑	↑	0	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Southern Appalachian Mountains	0	↑	↑	↑	0	0	0	0	0	↑	0	↑	NA	NA	NA	NA
Southeast Piedmont	↑	0	↑	0	↓	↑	0	↑	↓	↑	↑	↑	0	0	↑	↑
Gulf Coastal Plain	0	0	0	↓	0	0	↑	0	0	0	0	↑	↓	↓	0	0
Florida Coast Plain	0	0	0	0	0	0	0	↓	0	↑	0	↑	0	0	0	0

Notes: Key to symbols: ↑, increase; ↓, decrease; 0, no change for indicated variable, with trend indicated by nonoverlapping standard errors. NA indicates that data were not available.

† Change estimate for active treatments at Southern Cascades site substitutes control of second year posttreatment for pretreatment.

may limit the ability of burning to produce the desired results under all but the most aggressive burning prescriptions (Miller and Urban 2000, Schmidt et al. 2006).

Burning was also less effective than mechanical treatments at the eastern sites for reducing tree density and basal area and increasing quadratic mean diameter, but in these sites management emphasis is focused more on shifting tree species composition than on historical overstory density and size structure. Management here aims to shift composition away from tree species with low fire tolerance, mid-story hardwoods, and shrubs that have increased since fire suppression (Brockway et al. 2005). Because mechanical treatment prescriptions can be very selective about what species are removed or retained, these treatments have a greater potential to achieve management goals than burning treatments. In general, mechanical treatments were successful at reducing the density of mid-story hardwood species, while burning treatments were more effective at controlling shrub species. The MB treatments have the advantage of operator selectivity for the larger material, while also reducing shrub cover, leading to a more rapid restoration of a resilient stand structure.

Understory vegetation

The understory community responded differently to treatments than did the overstory: for some variables burning had effects not emulated by mechanical treatment and for others effects showed little consistent pattern across sites. No clear trends emerged in how treatments affected tree seedling density. At some sites, all active treatments reduced tree seedling density, indicating that what was gained through germination was less than that lost through fire and mechanical treatments. At other sites, burning treatments led to a large increase in seedling density, suggesting that removal of the duff layer and exposure to mineral soil may have been important (Moghaddas et al. 2008).

Determining the effect of fuel treatments on tree regeneration is limited by the year-to-year variability in seed production among tree species, sprouting vigor, and weather factors. In many forest types, the timing of burning and thinning treatments in relation to these seed production cycles can greatly affect composition of the future stand. For example, white fir (*Abies concolor*) typically produces a mast year every 2–3 years whereas ponderosa pine (*Pinus ponderosa*) every 3–4 years (Fowells 1965) and longleaf pine every 4–5 years (Brockway et al. 2006). In the absence of adverse weather, members of the white oak group (*Quercus* section *Quercus*) produce acorns every other year with good crops about once in 4 years (Johnson et al. 2002). The red oak group (*Quercus* section *Lobatae*) appears to be less synchronous which results in lower year-to-year variation (Johnson et al. 2002). Burning followed immediately by a mast year of seed production provides more resources for seedling recruitment than a mast year delayed two or more years after burning (Keeley and van Mantgem 2008). The timing of burning for regeneration of eastern pines varies by species and geographic location. Loblolly (*Pinus taeda* L.) and longleaf (*P. palustris* Mill.) pines have abundant seed crops in most years along the Atlantic Coastal Plain (Burns and Honkala 1990) but can have seed crop failures frequently in other regions. Table Mountain pine (*P. pungens* Lamb.) has serotinous cones and can store viable seed up to 10 years (Barden 1979) thus allowing germination after fires of any season. A close neighbor to Table Mountain pine is pitch pine (*P. rigida* Mill.), which does not have serotinous cones in the southern end of its range and is reported to have good seed crops every 3 to 9 years (Burns and Honkala 1990). Climate and other land management uses (e.g., grazing) also play large roles, particularly for the seasonally dry western sites, with only certain years having conditions suitable for germination and survival (Oliver and Larson 1996), while late frosts inhibit acorn production in the East (Johnson et al. 2002). Hardwood regeneration is generally from resprouts rather than seedlings. At eastern sites with abundant hardwoods, resprouting can be a major component of forest regeneration. It may take 3–5 years after acorn germination for the roots to store a sufficient amount of starches for oak sprouts to successfully outgrow competing tree species after being topkilled (Brose and Van Lear 1998).

The observed treatment effects on seedling density may not be particularly meaningful at this stage. Numerous mortality factors typically lead to steep declines in the years after germination. Also, factors that allow seeds to successfully germinate, such as bare mineral soil, may be quite different from factors that allow these seedlings to persist and become saplings and trees over time, such as the lack of competition for light (Stark 1965). Additionally, these treatments are being applied with the objectives of fuel reduction and ecosystem restoration and appropriate prescriptions for successful regeneration can be applied after this has been accomplished.

Although some of the sites had very low shrub cover prior to treatment (data not shown), burning generally reduced cover across sites. Burning also was as effective at reducing shrub cover as the combination of mechanical treatment and burning. Those sites with final (second–fourth) year posttreatment data on shrub cover, however, show that shrubs tend to recover with a rapid increase following the initial decreases associated with treatment. Many shrubs are vigorous resprouters, fire stimulates seeds of other species to germinate (Knapp et al. 2007), and opening the overstory canopy may in general favor shrub growth.

The initial response of native herbaceous understory species richness was positively influenced by fuel treatments with little difference found between mechanical and burning treatments. Understory vegetation often responds to light (Riegel et al. 1995, Naumburg et al. 2001, Wayman and North 2007) and both burning and mechanical treatments increase the light available at ground level. In addition, both treatments expose bare mineral soil on which understory species may establish (except at the Central Sierra where mastication residues covered the soil with heavy slash [Moghaddas and Stephens 2007]), burning by removing the duff layer, and mechanical treatments through skid trails, tire tracks, and other disturbances. This increase in understory species richness was expected based on the intermediate disturbance hypotheses (Connell 1978) and patch dynamics (Pickett 1980) with disturbance creating new patches, i.e., microsites that species can colonize.

Alien herbaceous species richness increased with fuel treatments and greatest increases tended to be in the mechanical plus burning treatment. Alien species often respond to the severity of disturbance (Keeley et al. 2003, Dodson and Fiedler 2006, Kerns et al. 2006, Collins et al. 2007). Because of the added activity fuels, more fuel mass was consumed by burning in this treatment than in the burning-alone treatment. The deeper soil disturbance caused in the mechanical operations may have also promoted alien herbaceous species On the other hand, soil exposure and disturbance is ephemeral, and the numbers of these species may decline over time as the soil becomes covered with litter and duff (Keeley and McGinnis 2007). However, should certain alien herbaceous species present a management issue locally, the trend toward greater invasion with combined mechanical plus burning treatments may need to be considered in choices about treatment type.

Fuels

A reduction in surface fuel loading, at least in the western U.S. sites, was most strongly associated with treatments involving burning. Conversely, mechanical treatments alone substantially increased the surface fuel loading at some sites and caused little change at others, with much of the variation likely due to harvesting method (Table 1). For example, all three sites showing increased total surface fuel mass after mechanical treatment employed systems (helicopter, single-grip harvester, chainsaw fell-limb-buck) that left slash in the forest. Conversely, the two sites (Southern Cascades, Southeastern Piedmont) that used whole-tree harvesting methods showed no change in surface fuel mass (Table 5).

Although the immediate effect of burning alone at western sites was to reduce surface fuels, this drop in fuel loading will be temporary. The burning only treatment also led to large numbers of snags (saplings and trees) that will fall over the next several years to decades, increasing the amount of fuel loading once again (Skinner 2005, Stephens and Moghaddas 2005*b*). In order to maintain low fire hazard conditions, it is therefore critical to maintain a program of frequent burning to consume this material as it dies and falls. Multiple sequential burns may be required before the fuel loading and the rate of accumulation of fuels are maintained at lower levels (Keifer et al. 2006). At the western sites, the combined mechanical plus burning treatment generally produced stand structures with fewer ladder fuels (saplings) and lower rates of fuel accumulation (i.e., fewer snags that remain to fall and less twig and litter fall from live trees due to reduced basal area), leading to more rapid development of conditions resilient to wildfire (Stephens et al. 2009). Without burning to treat the surface fuels, many of these mechanically thinned stands might resist crown fire initiation and spread, but could still be lost as a result of excessive heating and crown scorch in a wildfire (Agee and Skinner 2005, Ritchie et al. 2007).

Surface fuel loading was reduced proportionally less by burning at the eastern sites than at the western sites, presumably because understory vegetation is more abundant and because when killed by fire, these small woody shrub and tree stems fall to the ground more rapidly in these warm, moist environments. Thus, at least some of the vegetation killed by fire did become surface fuel prior to the posttreatment re-measurement. As with the western sites, removal of the biomass with mechanical treatment prior to burning results in a more rapid restoration of stand resilience to wildfire.

While burning increased the height to base of live crown more so than mechanical treatments, the difference between the individual active treatments in this study was not great. It is possible that because most stands were quite dense prior to treatment, live branches of the larger trees generally had self pruned due to shading. Although mechanical treatments generally do not increase the height to live crown of individual trees, removing the smaller trees resulted in an increase in the average height to live crown of the remaining trees. At some sites, this mechanical treatment effect was more than that of burning alone. The mechanical plus burning treatment had the greatest average height to base of live crown as a result of both processes and also presumably because the presence of slash led to a more intense surface fire and therefore more thermal pruning of the lower canopy.

Conclusions

Despite widely varying forest types across the network, some clear generalizations about response of ecological variables to fuel treatments are emerging. Across the network, mechanical treatment was generally more effective at manipulating overstory stand structure than was burning. If the objective of mechanical treatments is simply to act as a surrogate for prescribed fire under today's forest and fuel loading conditions, the mechanical treatments used in this study may be viewed as too aggressive. If, however, the objective is to produce a stand structure that is more resilient to disturbances such as bark beetle outbreaks, and closer to what existed historically, mechanical treatments may achieve these objectives. Mechanical treatments resulted in stand densities more in-line with our understanding of historical conditions. Many trees have established in the absence of fire and are now large enough to resist mortality under typical prescribed burning conditions (although a single prescribed fire may still reduce wildfire danger significantly as demonstrated by Stephens et al. [2009]).

Fuel variables such as total surface fuel loads and height to live crown were affected more strongly by burning than mechanical treatments. However, burning alone produced large numbers of dead saplings and small trees, which will ultimately fall and contribute to surface fuel. Mechanical treatments followed by burning produced the strongest result at most sites, with more resilient forest structures (lower density in the West, greater reduction in subcanopy hardwoods in the East), lower surface fuel loads, and reduced rate of accumulation of surface fuels. If burning alone were the only management option, additional burns might over time reduce tree densities and fuel loading, but the mechanical plus burning treatments achieved this condition more rapidly.

Mechanical treatments alone did not generally emulate fire's effects on understory vegetation: response to treatments showed no particular trend for some variables and was associated with burning (burning only and mechanical plus burning treatments) for other variables. Tree seedling density declined with treatment at some sites, increased with treatment at others, and appeared to be strongly associated with burning at some sites, such as the Southern Appalachian Mountains. Interaction of treatments with local factors, such tree seed production cycles and climate, may have overwhelmed the response to treatment, making generalizations at the network scale difficult. On the other hand, understory herbaceous species richness (both native and alien) appeared to respond positively to intensity of treatment (both amount of canopy removal and amount of soil exposed) at most sites. Increases in alien herbaceous species were particularly strongly associated with the combined mechanical plus burning treatments, presumably because this treatment resulted in the greatest increase in resources for growth and the highest amount of soil disturbance. At some sites the response of native and alien herbaceous species diversity appeared to be driven more strongly by mechanical treatments (canopy removal, or deeper soil disturbance), while at other sites, the response appeared to be more strongly associated with burning (extent of bare mineral soil exposure and possibly stimulation of germination by heat and/or compounds in smoke). Variation among sites is likely due to the differential implementation of treatments, level of disturbance, and the mix of species found at the respective sites.

Overall, the desired response of the ecological variables presented in this paper to fuel treatments involving burning and/or mechanical treatments was generally maximized by the combined mechanical plus burning treatments. These treatments produced desired changes in stand structure, while reducing surface fuel loading and rate of fuel accumulation in the near-term, and also increasing native understory herbaceous species diversity. Because mechanical plus burning treatments also appeared to favor alien herbaceous species invasion, this negative may need to be balanced against the positive attributes where alien species present particular management issues.

Results reported here profile responses for the initial few years after fuel treatments were implemented. It is vital that additional data are collected to not only verify these trends, but to investigate new trends that may not have materialized in this initial posttreatment time frame. Only after short- and longer-term responses to treatment are known, will managers have the information to fully understand the consequences of different fuel treatment options on stand resilience and forest health.

Acknowledgments

This paper is contribution #152 of the National Fire and Fire Surrogate Study, funded by the USDA/DOI Joint Fire Science Program, the National Fire Plan, the USDA National Research Initiative, and the home institutions and agencies of participating scientists. We thank numerous field crew members who collected much of the data at the 12 research sites over 5+ years of this study for their dedication and hard work. We also acknowledge the critical role played by our partners in the forest management world, who were responsible for implementing the treatments in a timely manner.

Literature Cited

Agee, J. K. 1993. Fire ecology of Pacific Northwest forests. Island Press, Washington, D.C., USA.

Agee, J. K., and M. R. Lolley. 2006. Thinning and prescribed fire effects on fuels and potential fire behavior in an eastern Cascade forest, Washington, USA. Fire Ecology 2:3–19.

Agee, J. K., and C. N. Skinner. 2005. Basic principles of forest fuel reduction treatments. Forest Ecology and Management 211:83–96.

Agee, J. K., R. H. Wakimoto, and H. H. Biswell. 1977. Fire and fuel dynamics of Sierra-Nevada conifers. Forest Ecology and Management 1:255–265.

Arno, S. F., and J. K. Brown. 1991. Overcoming the paradox in managing wildland fire. Western Wildlands 17:40–46.

Arno, S. F., H. Y. Smith, and M. A. Krebs. 1997. Old growth ponderosa pine and western larch stand structures: Influences of pre-1900 fires and fire exclusion. Technical Report 324. USDA Forest Service, Ogden, Utah, USA.

Barden, L. S. 1979. Serotiny and seed viability on *Pinus pungens* in the Southern Appalachians. Castanea 44:44–47.

Berry, A. H., and H. Hesseln. 2004. The effect of the wildland–urban interface on prescribed burning costs in the Pacific Northwestern United States. Journal of Forestry 102:33–37.

Biswell, H. 1973. Fire ecology in ponderosa pine grassland. Proceedings of the Tall Timbers Fire Ecology Conference 12: 69–73.

Boyle, J. F. 2002. Short-term response of bark beetles to fuel reduction treatments in the Upper Piedmont. Thesis. Clemson University, Clemson, South Carolina, USA.

Brockway, D. G., and C. E. Lewis. 1997. Long-term effects of dormant-season prescribed fire on plant community diversity, structure, and productivity in a longleaf pine wiregrass ecosystem. Forest Ecology and Management 96:167–183.

Brockway, D. G., K. W. Outcalt, and W. D. Boyer. 2006. Longleaf pine regeneration ecology and methods. Pages 95–134 *in* S. Jose, E. J. Jokela, and D. L. Miller, editors. The longleaf pine ecosystem. Springer, New York, New York, USA.

Brockway, D. G., K. W. Outcalt, D. J. Tomczak, and E. E. Johnson. 2005. Restoring longleaf pine forest ecosystems in the southern U.S. Pages 501–522 *in* J. A. Stanturf and P. Madsen, editors. Restoration of boreal and temperate forests. CRC Press, New York, New York, USA.

Brose, P. H., and D. H. Van Lear. 1998. Responses of hardwood advance regeneration to seasonal prescribed fires in oak-dominated shelterwood stands. Canadian Journal of Forest Research 28:331–339.

Brown, J. K. 1974. Handbook for inventorying downed woody material. USDA Forest Service, General Technical Report GTR-INT-16, Washington, D.C., USA.

Burnham, K., and D. Anderson. 2002. Model selection and multi-model inference: a practical information-theoretic approach. Second edition. Springer-Verlag, New York, New York, USA.

Burns, R. M., and B. Honkala, editors. 1990. Silvics of North America. Volume 1, conifers. Agriculture Handbook 654. USDA Forest Service, Washington, D.C., USA.

Collins, B. M., J. J. Moghaddas, and S. L. Stephens. 2007. Initial changes in forest structure and understory plant community following fuel reduction activities in a Sierra Nevada mixed conifer forest. Forest Ecology and Management 239:102–111.

Connell, J. H. 1978. Diversity in tropical rain forests and coral reefs. Science 199:1302–1310.

Converse, S., G. C. White, and W. M. Block. 2006. Small mammal responses to thinning and wildfire in ponderosa pine-dominated forests of the southwestern USA. Journal of Wildlife Management 70(6)1711–1722.

Cooper, C. F. 1960. Changes in vegetation, structure, and growth of southwestern pine forests since white settlement. Ecological Monographs 30:129–164.

Covington, W. W., and M. M. Moore. 1994. Southwestern ponderosa forest structure: changes since Euro-American settlement. Journal of Forestry 92:39–47.

Cowell, C. 1998. Historical change in vegetation and disturbance on the Georgia Piedmont. American Midland Naturalist 140:78–89.

Crow, T. R., and A. H. Perera. 2004. Emulating natural landscape disturbance in forest management: an introduction. Landscape Ecology 19:231–233.

Dodson, E. K., and C. E. Fiedler. 2006. Impacts of restoration treatments on alien plant invasion in *Pinus ponderosa* forests, Montana, USA. Journal of Applied Ecology 43:887–897.

Farris, K. L., and S. Zack. 2005. Woodpecker–snag interactions: an overview of current knowledge in ponderosa pine systems. Pages 183–195 *in* M. W. Ritchie, D. A. Maguire, and A. Youngblood, technical coordinators. Proceedings of the symposium on ponderosa pine: issues, trends, and management. General Technical Report PSW-GTR-198. USDA Forest Service, Albany, California, USA.

Fettig, C. J., K. D. Klepzig, R. F. Billings, A. S. Munson, T. E. Nebeker, J. F. Negron, and J. T. Nowak. 2007. The effectiveness of vegetation management practices for prevention and control of bark beetle outbreaks in coniferous forests of the western and southern United States. Forest Ecology and Management 238:24–53.

Fowells, H. 1965. Silvics of forest trees of the United States. Number 271 in USDA Forest Service Handbook. Government Printing Office, Washington, D.C., USA.

Frost, C. 2006. History and future of the longleaf pine ecosystem. Pages 9–42 *in* S. Jose, E. J. Jokela, and D. L. Miller, editors. The longleaf pine ecosystem. Springer, New York, New York, USA.

Glitzenstein, J. S., W. J. Platt, and D. R. Streng. 1995. Effects of fire regime and habitat on tree dynamics in North Florida longleaf pine savannas. Ecological Monographs 65:441–476.

Harrod, R. J., B. H. McRae, and W. E. Hartl. 1999. Historical stand reconstruction in ponderosa pine forests to guide silvicultural prescriptions. Forest Ecology and Management 114:433–446.

Harrod, R. J., N. A. Povak, and D. W. Peterson. 2007. Comparing the effectiveness of thinning and prescribed fire for modifying structure in dry coniferous forests. Pages 327–346 *in* B. W. Butler and W. Cook, compilers. The fire environment: innovations, management, and policy; conference proceedings, 26–30 March 2007, Destin, Florida. Proceedings RMRS-P-46CD. USDA Forest Service, Rocky Mountain Research Station, Fort Collins, Colorado, USA.

Hessburg, P., and J. Agee. 2003. An environmental narrative of inland northwest US forests, 1800–2000. Forest Ecology and Management 178:23–60.

Johnson, P. S., S. R. Shifley, and R. Rogers. 2002. The ecology and silviculture of oaks. CABI Publishing, New York, New York, USA.

Keeley, J. E., and C. J. Fotheringham. 2005. Plot shape effects on plant species diversity measurements. Journal of Vegetation Science 16:249–256.

Keeley, J. E., D. Lubin, and C. J. Fotheringham. 2003. Fire and grazing impacts on plant diversity and invasives in the southern Sierra Nevada. Ecological Applications 13:1355–1374.

Keeley, J. E., and T. W. McGinnis. 2007. Impact of prescribed fire and other factors on cheatgrass persistence in a Sierra Nevada ponderosa pine forest. International Journal of Wildland Fire 16:96–106.

Keeley, J. E., and N. L. Stephenson. 2000. Restoring natural fire regimes to the Sierra Nevada in an era of global change. Pages 255–265 *in* D. N. Cole, S. F. McCool, W. T. Borrie, and J. O'Loughlin, compilers. Wilderness science in a time of change conference: wilderness ecosystems, threats, and management; May 23–27, 1999, Missoula Montana. Proceedings RMRS-P-15, Volume 5. USDA Forest Service, Rocky Mountain Research Station, Ogden, Utah, USA.

Keeley, J. E., and P. J. van Mantgem. 2008. Community ecology of seedlings. Pages 255–273 *in* M. A. Leak, V. T. Parker, and R. L. Simpson, editors. Seedling ecology and evolution. Cambridge University Press, Cambridge, UK.

Keifer, M., J. W. van Wagtendonk, and M. Buhler. 2006. Long-term surface fuel accumulation in burned and unburned mixed conifer forests of the central and southern Sierra Nevada, CA (USA). Fire Ecology 2:53–72.

Kerns, B. K., W. G. Thies, and C. G. Niwa. 2006. Season and severity of prescribed burn in ponderosa pine forests: Implications for understory native and exotic plants. Ecoscience 13:44–55.

Kilgore, B. 1973. The ecological role of fire in Sierran conifer forests: its application to national park management. Quaternary Research 3:496–513.

Knapp, E. K., J. E. Keeley, E. A. Ballenger, and T. J. Brennan. 2005. Fuel reduction and coarse woody debris dynamics with early season and late season prescribed fire in a Sierra Nevada mixed conifer forest. Forest Ecology and Management 208:383–397.

Knapp, E. K., D. W. Schwilk, J. M. Kane, and J. E. Keeley. 2007. Role of burning season on initial understory vegetation response to prescribed fire in a mixed conifer forest. Canadian Journal of Forest Research 37:11–22.

Kobziar, L., J. J. Moghaddas, and S. L. Stephens. 2006. Tree mortality patters following prescribed fires in a mixed conifer forest. Canadian Journal of Forestry Research 36:3222–3238.

Leopold, S. A., S. A. Cain, C. A. Cottam, I. N. Gabrielson, and T. L. Kimball. 1963. Wildlife management in the national parks. American Forestry 69:32–35,61–63.

Liu, Y., J. J. Qu, X. Hao, and W. Wang. 2005. Improving fire emission estimates in the eastern United States using satellite-based fuel loading factors. Pages 1–4 *in* EastFIRE Conference Proceedings, Fairfax, Virginia, May 11–13. George Mason University, Fairfax, Virginia, USA.

McHugh, C. W., and T. E. Kolb. 2003. Ponderosa pine mortality following fire in northern Arizona. International Journal of Wildland Fire 12:7–22.

McIver, J., A. Youngblood, and S. L. Stephens. 2009. The national Fire and Fire Surrogate study: ecological consequences of fuel reduction methods in seasonally dry forests. Ecological Applications 19:283–284.

McRae, D., L. Duchesne, B. Freedman, T. Lynham, and S. Woodley. 2001. Comparisons between wildfire and forest harvesting and their implications in forest management. Environment Reviews 9:223–260.

Metlen, K.L., and C. E. Fiedler. 2006. Restoration treatment effects on the understory of ponderosa pine/Douglas-fir forests in western Montana, USA. Forest Ecology and Management 222:355–369.

Miller, C., and D. L. Urban. 2000. Modeling the effects of fire management alternatives on Sierra Nevada mixed-conifer forests. Ecological Applications 10:85–94.

Miller, S. R., and D. D. Wade. 2003. Re-introducing fire at the urban/wildland interface: planning for success. Forestry 76(2)253–260.

Moghaddas, E., and S. L. Stephens. 2007. Thinning, burning, and thin-burn fuel treatment effects on soil properties in a Sierra Nevada mixed-conifer forest. Forest Ecology and Management 250:156–166.

Moghaddas, J. J., R. A. York, and S. L. Stephens. 2008. Initial response of conifer and California black oak seedlings following fuel reduction activities in a Sierra Nevada mixed conifer forest. Forest Ecology and Management 255:3141–3150.

Mutch, L. S., and D. J. Parsons. 1998. Mixed conifer forest mortality and establishment before and after prescribed fire in Sequoia National Park, California. Forest Science 44:341–355.

Naumburg, E., E. L. DeWald, and T. E. Kolb. 2001. Shade responses of five grasses native to southwestern U.S. ponderosa pine forests. Canadian Journal of Botany 79: 1001–1009.

Oliver, C. D., and B. C. Larson. 1996. Forest stand dynamics. John Wiley and Sons, New York, New York, USA.

Outcalt, K. W. 2005. Restoring structure and composition of longleaf pine ecosystems of the Gulf Coastal Plains. Pages 97–100 *in* J. S. Kush, compiler. Proceedings of the 5th Longleaf Alliance Regional Conference, October 12–15, 2004, Hattiesburg, Mississippi. Longleaf Alliance Report No. 8. Auburn University, Auburn, Alabama, USA.

Outcalt, K. W., and J. L. Foltz. 2004. Impacts of growing-season burns in the Florida pine flatwoods type. Pages 30–34 *in* K. F. Connor, editor. Proceedings of the 12th biennial southern silvicultural research conference. General Technical Report SRS-7. USDA Forest Service, Southern Research Station, Asheville, North Carolina, USA.

Parker, A. 1984. A comparison of structural properties and compositional trends in conifer forests of Yosemite and Glacier National Parks. Northwest Science 58:131–141.

Parker, T. J., K. M. Clancy, and R. L. Mathiasen. 2006. Interactions among fire, insects, and pathogens in coniferous forests of the interior western United States and Canada. Agricultural and Forest Entomology 8:167–189.

Parsons, D. J., and S. H. DeBenedetti. 1979. Impact of fire suppression on a mixed-conifer forest. Forest Ecology and Management 2:21–33.

Phillips, R., and T. Waldrop. 2008. Changes in vegetation structure and composition in response to fuel reduction treatments in the South Carolina Piedmont. Forest Ecology and Management 235:3107–3116.

Pickett, S. T. A. 1980. Non-equilibrium coexistence of plants. Bulletin of Torrey Botanical Club 107:238–248.

Pyne, S. 1982. Fire in America. Princeton University Press, Princeton, New Jersey, USA.

R Development Core Team. 2005. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. ⟨http://www.R-project.org⟩

Riegel, G. M., R. F. Miller, and W. C. Krueger. 1995. The effects of aboveground and belowground competition on understory species composition in a *Pinus ponderosa* forest. Forest Science 41:864–889.

Ritchie, M. W. 2005. Ecological research at the Goosenest Adaptive Management Area in northeastern California. General Technical Report PSW-GTR-192. USDA Forest Service, Pacific Southwest Research Station, Albany, California, USA.

Ritchie, M. W., C. N. Skinner, and T. A. Hamilton. 2007. Probability of tree survival after wildfire in an interior pine forest of northern California: effects of thinning and prescribed fire. Forest Ecology and Management 247:200–2008.

Ryan, K. C., and E. D. Reinhardt. 1988. Predicting post-fire mortality of seven western conifers. Canadian Journal of Forest Research 18:1291–1297.

Schmidt, L., M. G. Halle, and S. L. Stephens. 2006. Restoring northern Sierra Nevada mixed conifer forest composition and structure with prescribed fires of varying intensities. Fire Ecology 2:204–217.

Schwarz, G. F. 1907. The longleaf pine in virgin forest. A silvical study. Wiley, New York, New York, USA.

Schwilk, D. W., E. E. Knapp, S. M. Ferrenberg, J. E. Keeley, and A. C. Caprio. 2006. Tree mortality from fire and bark beetles following early and late season prescribed fires in a Sierra Nevada mixed conifer forest. Forest Ecology and Management 232:36–45.

Skinner, C. N. 1995. Change in spatial characteristics of forest openings in the Klamath Mountains of northwestern California, USA. Landscape Ecology 10:219–228.

Skinner, C. N. 2005. Reintroducing fire into the Blacks Mountain Research Natural Area: effects on fire hazard. Pages 245–257 *in* M. Ritchie, D. Maguire, and A. Youngblood, editors. Proceedings of the symposium on ponderosa pine: issues, trends, and management. 2004 October 18–21, Klamath Falls, OR. USDA Forest Service, Pacific Southwest Research Station, Albany, California, USA.

Skinner, C. N., and C. Chang. 1996. Fire regimes, past and present. Pages 1041–1069 *in* Sierra Nevada Ecosystem Project: Final Report to Congress. Volume II: Assessments and scientific basis for management options, Center for Water and Wildland Resources, University of California, Davis, California, USA.

Stark, N. 1965. Natural regeneration of Sierra Nevada mixed conifers after logging. Journal of Forestry 63:456–457,460–461.

Stephens, S. L. 2000. Mixed conifer and upper montane forest structure and uses in 1899 from the central and northern Sierra Nevada, California. Madroño 47:43–52.

Stephens, S. L., and M. A. Finney. 2002. Prescribed fire mortality of Sierra Nevada mixed conifer tree species: effects of crown damage and forest floor combustion. Forest Ecology and Management 162:261–271.

Stephens, S. L., and J. J. Moghaddas. 2005*a*. Experimental fuel treatment impacts on forest structure, potential fire behavior, and predicted tree mortality in a mixed conifer forest. Forest Ecology and Management 215:21–36.

Stephens, S. L., and J. J. Moghaddas. 2005*b*. Fuel treatment effects on snags and coarse woody debris in a Sierra Nevada mixed conifer forest. Forest Ecology and Management 214: 53–64.

Stephens, S. L., J. J. Moghaddas, C. Edminster, C. E. Fiedler, S. Haase, M. Harrington, J. E. Keeley, E. E. Knapp, J. D. McIver, K. Metlen, C. N. Skinner, and A. Youngblood. 2009. Fire treatment effects on vegetation stucture, fuels, and potential fire severity in western U.S. forests. Ecological Applications 19:305–320.

Stephenson, N. L., D. J. Parsons, and T. W. Swetnam. 1991. Restoring natural fire to the sequoia–mixed conifer forest: should intense fire play a role? Pages 321–337 *in* High intensity fire in wildlands: management challenges and options. Volume 17. Tall Timbers Research Station, Tallahassee, Florida, USA.

Taylor, A. H. 2000. Fire regimes and forest changes in mid and upper montane forests of the southern Cascades, Lassen Volcanic National Park, California, U.S.A. Journal of Biogeography 27:87–104.

Taylor, A. H. 2004. Identifying forest reference conditions on cut-over lands, Lake Tahoe Basin, USA. Ecological Applications 14:1903–1920.

Van Lear, C., and T. Waldrop. 1989. History, use and effects of fire in the Appalachians. General Technical Report SE-GTR-54. USDA Forest Service, Southeastern Forest Experiment Station, Asheville, North Carolina, USA.

Waldrop, T. A., D. Yaussy, R. J. Phillips, T. A. Hutchinson, L. Brudnak, and R. A. J. Boerner. 2008. Fuel reduction treatments affect vegetation structure of hardwood Forests in western North Carolina and southern Ohio, USA. Forest Ecology and Management 255:3117–3129.

Ware, S., C. Frost, and P. D. Doerr. 1993. Southern mixed hardwood forest: the former longleaf pine forest. Pages 447–493 *in* W. H. Martin, S. G. Boyce, and A. C. Echternacht, editors. Biodiversity of the southeastern United States: lowland terrestrial communities. John Wiley and Sons, New York, New York, USA.

Wayman, R. B., and M. North. 2007. Initial response of a mixed-conifer understory plant community to burning and thinning restoration treatments. Forest Ecology and Management 239:32–44.

Weatherspoon, C. P. 2000. A proposed long-term national study of the consequences of fire and fire-surrogate treatments. Pages 117–126 *in* L. F. Neuenschwander, K. C. Ryan, and G. E. Goldberg, editors. Proceedings of the Joint Fire Science Conference and Workshop. Crossing the millennium: integrating spatial technologies and ecological principles for a new age in fire management. University of Idaho Press, Moscow, Idaho, USA.

Youngblood, A., K. T. Max, and K. Coe. 2004. Stand structure in old growth ponderosa pine forests of Oregon and northern California. Forest Ecology and Management 199:191–217.

Youngblood, A., K. L. Metlen, and K. Coe. 2006. Changes in stand structure and composition after fuel restoration treatments in low elevation dry forests of northeastern Oregon. Forest Ecology and Management 234:143–163.

Youngblood, A., K. Metlen, E. Knapp, K. Outcalt, S. Stephens, T. Waldrop, and D. Yaussy. 2005. Implementation of the Fire and Fire Surrogate Study: a national research effort to evaluate the consequences of fuel reduction treatments. Pages 315–321 *in* C. E. Peterson and D. A. Maguire, editors. Balancing ecosystem values. General Technical Report PNW-GTR-635. USDA Forest Service, Pacific Northwest Research Station, Portland, Oregon, USA.

APPENDIX A

Number of Fire and Fire Surrogate Study research sites included in analysis of each of 12 response variables at 1 and 2–4 years posttreatment (*Ecological Archives* A019-012-A1).

APPENDIX B

Definitions of tree-size classes by site (*Ecological Archives* A019-012-A2).