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THINNING AND BURNING RESULT IN LOW-LEVEL INVASION BY NONNATIVE PLANTS BUT NEUTRAL EFFECTS ON NATIVES

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Abstract. Many historically fire-adapted forests are now highly susceptible to damage from insects, pathogens, and stand-replacing fires. As a result, managers are employing treatments to reduce fuel loadings and to restore the structure, species, and processes that characterized these forests prior to widespread fire suppression, logging, and grazing. However, the consequences of these activities for understory plant communities are not well understood. We examined the effects of thinning and prescribed fire on plant composition and diversity in *Pinus ponderosa* forests of eastern Washington (USA). Data on abundance and richness of native and nonnative plants were collected in 70 stands in the Colville, Okanogan, and Wenatchee National Forests. Stands represented one of four treatments: thinning, burning, thinning followed by burning, or control; treatments had been conducted 3–19 years before sampling. Multi-response permutation procedures revealed no significant effect of thinning or burning on understory plant composition. Similarly, there were no significant differences among treatments in cover or richness of native plants. In contrast, nonnative plants showed small, but highly significant, increases in cover and richness in response to both thinning and burning. In the combined treatment, cover of nonnative plants averaged 2% (5% of total plant cover) but did not exceed 7% (16% of total cover) at any site. Cover and richness of nonnative herbs showed small increases with intensity of disturbance and time since treatment. Nonnative plants were significantly less abundant in treated stands than on adjacent roadsides or skid trails, and cover within these potential source areas explained little of the variation in abundance within treated stands. Although thinning and burning may promote invasion of nonnative plants in these forests, our data suggest that their abundance is limited and relatively stable on most sites.

Key words: alien plants; exotic plants; forest management; forest restoration; fuel-hazard reduction; native plants; nonnative plants; Pinus ponderosa; *prescribed fire; thinning; underburning; understory plant composition.*

Introduction

Fire-adapted forest ecosystems in many regions of the world are increasingly susceptible to damage from insects, pathogens, and stand-replacing fires (Martin et al. 1976, Whelan and Muston 1991, Agee 1993, Schimmel and Granström 1997). In many systems, resource managers are attempting to reduce fuel loadings and manipulate forest structure, thus redirecting future stand development (Whelan 1995, Friederici 2003). In the United States, forest restoration treatments are being implemented at an accelerated rate: the Healthy Forests Restoration Act of 2003 (United States Senate 2004) authorized federal agencies to spend 750 million dollars to mechanically remove small-diameter trees (thinning) and to conduct low-intensity surface fires (prescribed burning) on up to eight million hectares of forestland to reduce fuel levels and wildfire hazard. To date, research assessing the efficacy of these treatments has primarily focused on changes to forest structure (e.g., Feeney et al. 1998, Kalabokidis and Omi 1998, Agee 2003, Waltz et al. 2003); however, limited attention has been devoted to the consequences for native plant communities or to the potential for introducing nonnative species (Korb and Springer 2003). Basic information on vegetation responses to restoration treatments is essential for resource managers to develop sound management strategies for ecosystem restoration.

Effects of thinning on understory plant communities can be complex, with responses varying by species or functional group (Moore et al. 2006). On the one hand, thinning may lead to increases in resource availability including light, water, and nutrients (Gundale et al. 2005), resulting in greater understory production and diversity (e.g., Grant and Loneragan 2001, Griffis et al. 2001). However, increases in resource availability can exacerbate competitive interactions possibly leading to reductions in plant diversity (Huston 1979, Tilman 1984). At the same time, soil disturbance associated with thinning is likely to enhance establishment of ruderal species (e.g., Bailey et al. 1998), thus increasing the diversity of native and nonnative plants.

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The effects of prescribed fire can be equally complex, contingent on initial species composition, intensity or timing of burning, soil properties, soil seed banks, and disturbance history (Korb and Springer 2003). Burning can cause dramatic reductions in plant biomass and diversity, but in fire-dependent systems community recovery is often rapid (e.g., Harris and Covington 1983) as resident species have evolved numerous strategies to persist through disturbance (Rowe 1981). Burning may promote native, fire-adapted species by increasing fecundity (Hartnett and Richardson 1989), stimulating seed germination (Jacobs 1993, Harrod and Halpern 2005), or providing sites for seedling establishment (Barker and Williamson 1988). However, burning can also result in loss of soil organic matter, exposure of mineral soil, and loss or redistribution of soil nitrogen (Johnson and Curtis 2001, Wan et al. 2001)—changes that facilitate the establishment of ruderal species or that alter competitive relationships among native and nonnative plants (Melgoza et al. 1990, Huenneke and Thomson 1994, D'Antonio 2000). Effects of burning can also vary with the timing of treatment as influenced by plant phenology or fuel moisture, or the history of recent disturbance (Hobbs et al. 1984, D'Antonio 2000). For example, fire severity should be greater in recently thinned stands where surface fuels may be more abundant.

In addition to the direct effects of thinning or burning, landscape context can also influence vegetation response to restoration treatments. In particular, roads and skid trails as sources and corridors for dispersal (Trombulak and Frissell 2000, Gelbard and Belnap 2003) can play an important role in the introduction of nonnative plants into adjacent stands (Milberg and Lamont 1995, D'Antonio et al. 1999). If propagule pressure is a primary determinant of invasion, the abundance of nonnative plants in treated stands should be related to local abundance and proximity of seed sources (Rouget and Richardson 2003, Lockwood et al. 2005).

The effects of forest thinning or burning on understory plants may change with time since disturbance (Moore et al. 2006). Initial increases in ruderal species may be transient, and residual forest species that initially decline may show gradual recovery (Halpern 1989). Nonnative plants, which often exhibit dramatic increases in abundance after disturbance (Rejmanek 1989, Hobbs and Huenneke 1992), may increase with time if introductions are successful, or decline as effects of disturbance diminish or competition from native species increases (Halpern et al. 1997, D'Antonio 2000).

Previous studies of vegetation responses to thinning and burning have been limited in geographic scope, temporal scale, or level of replication. Here we examine longer term (3–19 yr) responses over a large geographic region (70 sites in eastern Washington, USA); our results should be applicable to *Pinus ponderosa* forests across much of the inland Northwest. We address the following questions: (1) Do thinning, burning, or the combination of these treatments affect understory species composition? (2) Do these treatments affect plant abundance or diversity? If so, do responses differ among herbaceous and woody plants or between native and nonnative species? (3) Do responses vary with the characteristics of thinning and burning (residual stand basal area or proportion of ground surface burned) or with time since treatment? (4) Are nonnative plants more abundant along adjacent roadsides or skid trails than in treated stands, and does abundance in these potential source areas explain variation in abundance within treated stands?

Methods

Study sites

Our study was conducted in *Pinus ponderosa* forests on the Colville, Okanogan, and Wenatchee National Forests in eastern Washington, USA. The climate in this region is characterized by a short growing season with hot summer days (average maximum temperatures $\sim$31°C) and cool nights (average minimum temperatures of $\sim$14°C). Average annual precipitation ranges from 355 to 760 mm, but $<$25 mm falls between July and September (Franklin and Dyrness 1973). Forest stands occupy a diversity of soil types; most are well-drained sandy loams to loams, with glacial till as the most abundant parent material (Franklin and Dyrness 1973). Historically, disturbance regimes were characterized by low-intensity surface fire occurring at 3- to 35-yr intervals (see Agee 1993 for a review). However, fire suppression, logging, and grazing since the early 1900s have disrupted natural fire cycles, resulting in less frequent but higher intensity fire than had occurred historically (Gast et al. 1991, Agee 1993). As a consequence, many formerly open, park-like stands have developed dense understories of small-diameter trees and fuels, increasing susceptibility to damage from insects, pathogens, and stand-replacing fire (Hessburg et al. 1994). Thinning and underburning are now commonly used to reduce fuel loadings in overstocked stands.

Through database queries and interviews with U.S. Forest Service employees, we identified all management units (henceforth stands) on these forests that met the following criteria: (1) dominated by mature *Pinus ponderosa* with *Pseudotsuga menziesii* as a common associate; (2) aspect of southeast to southwest; (3) elevation of $<$1200 m; (4) area $\geq$8 ha; and (5) no fire or commercial harvest since 1960, except for the study treatment. Stands that met the first four criteria, but had no fire or management activity since 1960, were identified as potential controls. We visited each stand to eliminate those that did not meet all criteria. From those remaining, we randomly selected 70 representing one of three fuel-hazard-reduction treatments (thin only, burn only, or thin followed by burn) or a control (no treatment) (Table 1). Treatments had been completed three to 19 years before sampling. Timber harvest

TABLE 1. Number of replicates in treatments within each national forest.

National Forest	Treatment				Total
	Control	Thin only	Burn only	Thin + burn	
Colville	7	10	10	9	36
Okanogan	3	4	4	6	17
Wenatchee	5	6	2	4	17
Total	15	20	16	19	70

typically was done using ground-based logging systems during the dry season. Controls were ecologically similar to treated stands, but had no recent history of disturbance; many were scheduled to be treated as part of future fuel-hazard reduction projects. Most sites experienced some cattle grazing, which began as early as the 1870s on these forests and peaked in the 1920s (Carter and Holstine 1994). Although grazing still occurs, it was not possible to quantify local intensity or severity in our stands.

Field sampling

Vegetation sampling was conducted from 5 June to 7 August 2003. Within each stand, eight 20-m transects were randomly located, with the following constraints: transects were established parallel with the slope contour, were separated by at least 30 m, and were >50 m from all stand edges. In addition, if a stand was bordered by a road, we established a 20-m transect parallel to the road, ~5 m from the edge of the roadbed. If a stand contained a skid trail, we established a similar transect at a random location on the skid trail. Each transect consisted of 20 microplots (0.2×0.5 m) spaced at 1-m intervals. Within each microplot, we estimated cover of all vascular plant species; cover was recorded to the nearest 0.1% for values <1%, to the nearest 1% for values between 1 and 10%, and to the nearest 5% for values >10%. Species that could not be identified at the time of sampling were collected and identified in the lab. Nomenclature follows Hitchcock and Cronquist (1973) except for the genus *Festuca* which follows Wilson (1999).

To quantify variation in posttreatment stand structure, basal area of each stand was estimated from variable radius plots using a prism (basal area factor of 5–20, English units [Avery and Burkhart 2002]), with the center of each plot located at the midpoint of each transect. To quantify extent of burning, we estimated the proportion of each microplot that showed evidence of fire (e.g., charred organic material, black or white ash, or scorched soil [Agee 1993]).

Statistical analyses

Prior to analysis, stand-level means were computed for basal area and proportion of ground surface burned. Cover of individual plant species was averaged for each transect and stand. In addition, each species was classified by (1) growth form, i.e., graminoid, herb (ferns, forbs, and sub-shrubs), low shrub (typically <1 m tall at maturity), or tall shrub (>1 m tall at maturity); and (2) origin (native or nonnative) based on the USDA Plants Database (*available online*).[2] The Appendix presents data on mean frequency and cover of species.

Multi-response permutation procedures (MRPP; Biondini et al. 1988) were used to test for differences in species composition among treatments. Euclidean distance was used as the measure of dissimilarity. Separate tests were conducted for all treatments (control, thin only, burn only, thin + burn) and for main effects (thinned vs. unthinned, burned vs. unburned treatments). Groups were weighted as follows:

$$n - 1/\sum(n - 1) \quad (1)$$

where n is the number of samples in each group. This results in a test statistic equivalent to a one-way analysis of variance (ANOVA) F test (Mielke and Iyer 1982).

Effects of thinning, burning, and their interaction on understory plant abundance and richness were assessed using two-factor ANOVA (Sokal and Rohlf 1981). Abundance was expressed as the summed cover of species, and richness as the average number of species per transect. Separate tests were performed on native and nonnative plants, and on growth forms within each group. To correct for heteroscedasticity, values of cover and richness of nonnative plants were square-root or log-transformed. An initial set of models employed national forest and plant association as blocking factors; neither proved significant, thus a completely randomized design was used.

We used Spearman rank correlations (Sokal and Rohlf 1981) to assess relationships between vegetation responses (cover and richness by growth form and origin) and basal area, area burned, and time since treatment. For stands that were thinned and burned, we considered time since treatment to be time since burning. To test whether evidence of burning could have declined with time, we ran a Spearman rank correlation between mean area burned and time since treatment ($n = 35$ burned stands).

Finally, to examine the relationship between abundance of nonnative plants in source areas (roadsides and skid trails) and treated stands, we conducted two types of analyses. Paired t tests were used to compare mean cover of nonnatives within stands to cover along roadsides and skid trails. Cover of nonnatives within treated stands was then regressed on cover along roadsides and skid trails. For these analyses, we applied natural logarithmic transformations to dependent and independent variables to meet the assumptions of linear regression. Analyses included only those stands bordered by a road ($n = 45$ stands: 6 control, 17 thin only, 7

[2] ⟨http://plants.usda.gov⟩

TABLE 2. Summary statistics by treatment type for basal area, area burned, and time since treatment.

Variable	Control (n = 15)				Thin only (n = 20)				Burn only (n = 16)				Thin + burn (n = 19)			
	Mean	SE	Min.	Max.	Mean	SE	Min.	Max.	Mean	SE	Min.	Max.	Mean	SE	Min.	Max.
Basal area (m^2/ha)	23.8	2.0	10.8	35.6	13.3	1.4	7.0	28.1	21.8	2.2	8.3	34.2	12.3	0.8	6.9	20.2
Area burned (%)	0.1	0.0	0.0	0.2	0.6	0.5	0.0	10.2	21.3	3.4	0.1	44.9	27.3	3.4	5.9	53.9
Time since treatment (yr)					9.7	1.0	3.0	19.0	8.9	1.2	4.0	18.0	7.9	0.8	5.0	17.0

Note: Sample sizes are the number of replicates.

burn only, and 15 thin + burn) or those with a skid trail (n = 19 stands: 8 thin only and 11 thin + burn).

Multivariate tests (MRPP) were performed with PC-Ord version 4.0 (McCune and Mefford 1999); univariate analyses were conducted with Systat version 10.0 (SPSS 2001) using an alpha level of 0.05 as the criterion for reporting statistical significance; adjustments for multiple comparisons were not made.

RESULTS

Stand structure and area burned

As expected, basal area varied greatly among treatments (Table 2). The effect of thinning was significant, but burning was not ($P < 0.001$ and $P = 0.3$, respectively, from two-factor ANOVA). Percentage of area burned was comparable in burned treatments (burn only and thin + burn; Table 2). For the burn treatments overall, 67% of microplots had <20% area burned, and 25% experienced >60% area burned. Area burned was negatively correlated with time since treatment (Spearman rank correlation: $r = -0.371$, $P = 0.031$), thus it is possible that we underestimated area burned in older treatments. As expected, we observed little or no evidence of burning in control or thin-only stands (Table 2).

Floristics

Among the 70 stands, we identified 288 vascular plant taxa representing 162 genera and 52 families (Appendix). Of these, 246 (85%) were native (37 graminoids, 174 herbs, 27 low shrubs, and 8 tall shrubs). The most common (present in >75% of stands) included *Calamagrostis rubescens*, *Carex rossii*, *Agropyron spicatum*, *Achillea millefolium*, *Collinsia parviflora*, *Lupinus* spp., *Arnica cordifolia*, *Hieracium scouleri*, *Cryptantha torreyana*, and *Amelanchier alnifolia*. We observed 42 nonnative species (16 graminoids and 26 herbs); the most common (present in >20% of stands) included *Bromus tectorum*, *Poa pratensis*, *Taraxacum officinale*, *Cirsium vulgare*, *Tragopogon dubius*, *Filago arvensis*, *Lactuca serriola*, and *Verbascum thapsus*.

Understory responses to treatments

Plant species composition was only minimally affected by treatment: there was no effect of thinning and only a marginally significant effect of burning, with a very low effect size (MRPP A statistic; Table 3). Thinning and burning also had little effect on total cover and richness of native plants (Fig. 1). Of ten variables tested, only one (richness of native graminoids) showed a significant response to thinning (small positive effect of thinning; Fig. 1). Effects of burning were nonsignificant. In contrast, among nonnative plants, we observed significant positive effects of thinning and/or burning or significant treatment interactions (Fig. 2). For most variables, differences were relatively small in unburned treatments (control vs. thin only), but comparatively large in burned treatments (thin + burn vs. burn only; Fig. 2). This interaction was significant for total cover of nonnatives and richness of nonnative herbs.

Despite significant responses to treatment, mean cover of nonnatives in thin + burn stands was only 2.0% (4.9% of total plant cover) and mean richness only 2.3 species per transect (9.4% of all species). Maximum cover of nonnatives reached 6.7% (15.8% of total cover) and maximum richness, 7.0 species (20% of species).

Effects of basal area, area burned, and time since treatment

Correlation analyses suggest negative influences of basal area on richness of native graminoids and on cover and richness of nonnative plants and graminoids (Table 4). Percent area burned was negatively correlated with cover of low shrubs, but positively correlated with cover and richness of nonnative plants and herbs. Native plants as a group showed positive correlations with time

TABLE 3. Results of multi-response permutation procedures (MRPP) to detect differences in species composition among (1) all treatments and (2) main effects (thinned vs. unthinned and burned vs. unburned treatments).

Test	Treatments compared	T	A	P
All treatments	Control, thin only, burn only, thin + burn	−1.244	0.008	0.113
Thinning	Thin only and thin + burn vs. control and burn only	0.018	<0.001	0.415
Burning	Burn only and thin + burn vs. control and thin only	−1.812	0.007	0.057

Note: T is the test statistic, A is an estimate of effect size, and P is the probability of significant differences among groups.

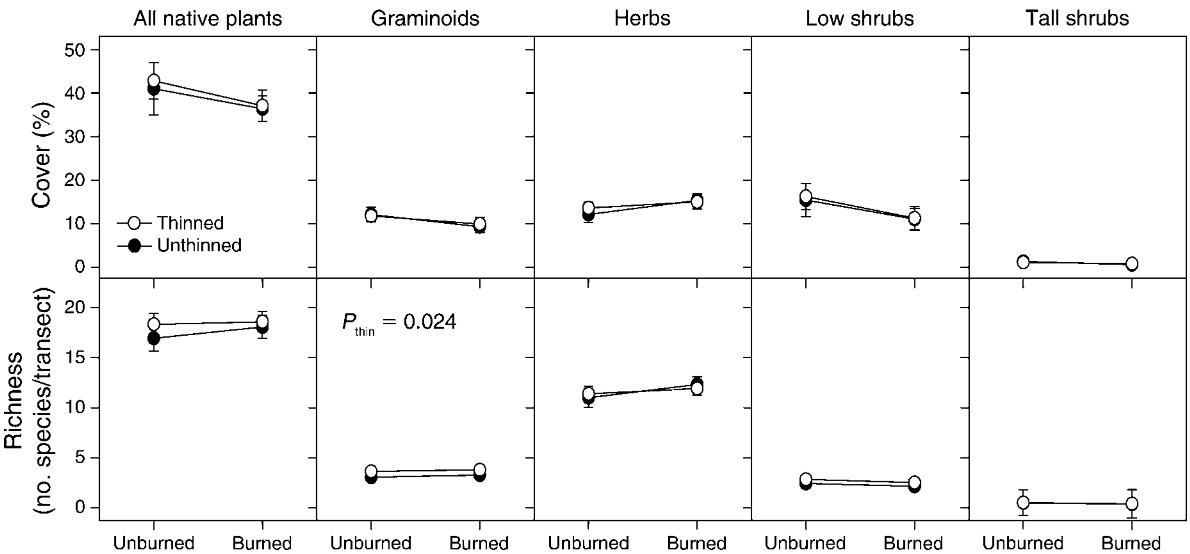

FIG. 1. Total (summed) cover and richness of native growth forms among stands with different thinning and burning histories. Values are means ± SE. P values are reported for significant ($P \leq 0.05$) main effects or interactions resulting from two-factor ANOVAs. Open circles represent thinned stands; solid circles represent unthinned stands.

since treatment (Table 4). Cover and richness of nonnative herbs also increased significantly over time.

Role of roadside and skid-trail populations of nonnatives

Nonnative plants were significantly less abundant within stands (mean cover of 1.5%) than along roadsides (mean cover of 5.5%; $t = 4.384$, $P < 0.001$) or skid trails (mean cover of 5.0%, $t = 2.381$, $P < 0.03$). In relative terms, these represent 3.5 vs. 19.0 and 27.0% of total plant cover, respectively. Cover of nonnatives along roadsides explained little of the variation in cover in adjacent stands although the relationship was significant ($R^2 = 0.243$, $P = 0.003$); cover along skid trails showed no relationship to that within unskidded areas of stands ($R^2 = 0.023$, $P = 0.6$).

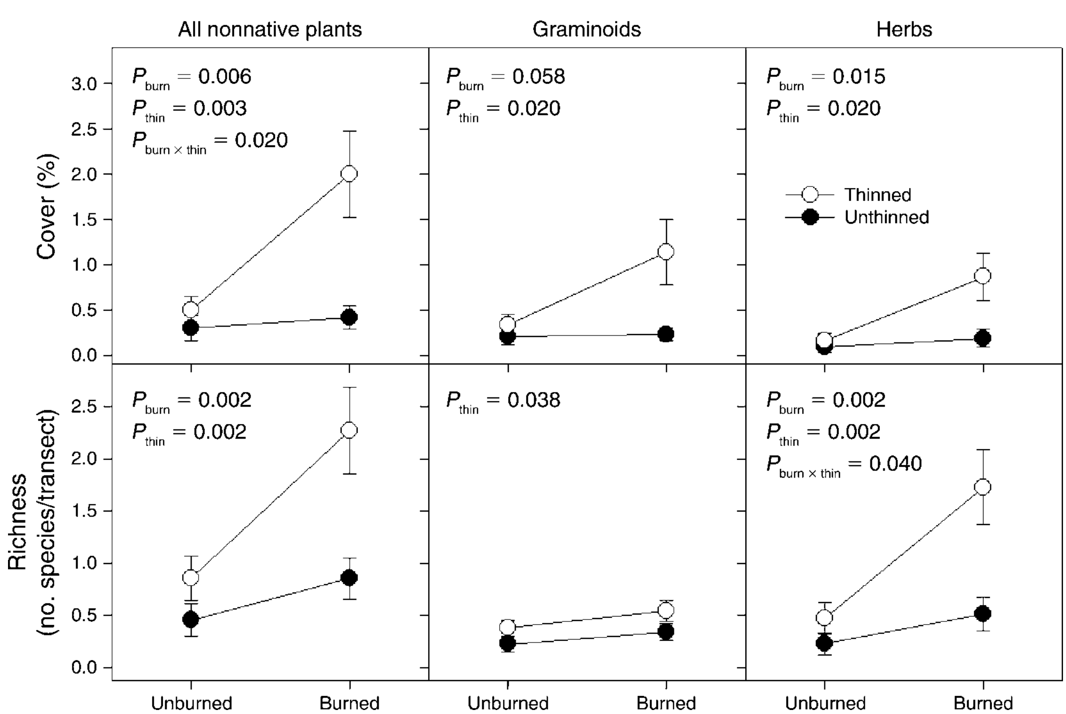

FIG. 2. Total (summed) cover and richness of nonnative growth forms among stands with different thinning and burning histories. Low and tall shrubs were absent from the pool of nonnative species. See Fig. 1 for other details.

Table 4. Spearman rank correlation coefficients (r) and significance (P) between vegetation response variables and basal area, percentage of area burned, and time since treatment.

Vegetation response variable	Basal area		Percentage of area burned		Time since treatment	
	r	P	r	P	r	P
Native plant cover						
All plants					0.34	0.022
Low shrubs			−0.28	0.039	0.51	<0.001
Native plant richness						
All plants					0.36	0.018
Graminoids	−0.37	0.002				
Herbs					0.29	0.050
Low shrubs					0.44	0.003
Nonnative plant cover						
All plants	−0.25	0.035	0.26	0.013		
Graminoids	−0.28	0.018				
Herbs			0.23	0.020	0.36	0.035
Nonnative plant richness						
All plants	−0.27	0.024	0.29	0.017		
Graminoids	−0.30	0.010				
Herbs			0.29	0.007	0.37	0.035

Notes: Only variables with significant relationships ($P \leq 0.05$) are shown. Tests for percentage of area burned included only burned stands, and tests for time since treatment excluded controls.

Discussion

Although thinning and prescribed burning are widely applied in many fire-adapted ecosystems to reduce fuel loads, their effects on understory plant communities have not been sufficiently addressed. Our study, which examines long-term responses over large areas of the inland Northwest, suggests that thinning and burning have had surprisingly small effects on the composition, cover, and diversity of forest understory plants. Previous studies of vegetation response to similar types of treatments have yielded widely varying results ranging from increases in total plant abundance (Ffolliott et al. 1977, Busse et al. 2000, Moore et al. 2006) to no response or to decreases in abundance (Gruell et al. 1982, Ayers et al. 1999, Metlen et al. 2004). These differences may reflect variation in initial composition or disturbance intensity—factors that are difficult to compare retrospectively (D'Antonio 2000), but may have large influences on outcomes, particularly where replication is limited.

The general absence of response to thinning and burning on our sites reflects, in large part, the performance of native plants that dominate these forests: neither cover nor richness was significantly affected by thinning or burning. In contrast, nonnative plants showed positive responses to thinning and burning that were magnified by the combination of treatments, with values five to seven times greater in thin + burn than in control stands. These differences in response among natives and nonnatives are consistent with patterns observed after similar treatments in forests of northern Arizona (Griffis et al. 2001) and western Montana (Dodson and Fiedler 2006, Metlen and Fiedler 2006). Although the interaction of thinning and burning may have a large effect in relative terms, nonnative plants nevertheless remained a minor component of the vegetation in all treatments, averaging only 2% cover and 2.3 species per transect. Similarly low abundance has been observed after burning in Kings Canyon National Park, California (Keeley et al. 2003), after thinning and burning in western Montana (Dodson and Fiedler 2006), and along fuel breaks in coniferous forest in California (Merriam et al. 2006). However, Griffis et al. (2001) found considerably greater cover of nonnatives (~7%) in thinned and burned forests of northern Arizona 3–11 yr after treatment. This average is comparable to our most invaded site (6.7% cover), but represents many fewer stands ($n = 4$ stands) from a limited geographic area; thus, it may reflect local conditions or particular disturbance histories that favor invasion. On the other hand, invasion of nonnatives may be greater in forests that have had multiple management entries (Hobbs and Huenneke 1992, D'Antonio 2000); our sites were chosen to have had only a single management-related disturbance within the last two decades.

The combination of thinning and burning may facilitate recruitment of nonnative species through various mechanisms. First, multiple entries into a stand allow for repeated introduction of propagules. Second, the combination of treatments can lead to greater intensity of disturbance, thus increasing the potential for invasion (Hobbs and Huenneke 1992, Dodson and Fiedler 2006). The additional ground fuels and greater canopy openness that result from thinning may increase the intensity of prescribed surface fires. Third, nonnative plants respond positively to increases in light (Parendes and Jones 2000) and soil nitrogen availability (Huenneke et al. 1990, D'Antonio 2000) which, at least in the short term, are likely to be greater in stands that are thinned and burned than in stands subjected to a single treatment (Gundale et al. 2005).

Forest management treatments, such as thinning and underburning, generally require roads or skid trails to transport equipment associated with logging or fire control; accordingly, the majority of our treated stands were bordered by roads that potentially served as conduits for invasion of nonnative plants (Trombulak and Frissell 2000, Gelbard and Belnap 2003). Despite the abundance and proximity of seed sources along these corridors, cover and richness of nonnatives were low within treated stands, even in our most highly invaded sites. That cover was substantially lower within treated stands and at most, only weakly related to cover along roadsides, suggests that establishment of nonnatives is limited more by environmental conditions (biotic or abiotic) than by availability of seed.

Most studies of nonnative plant responses to disturbance are of short duration (but see Moore et al. 2006), thus little is known about longer term dynamics (D'Antonio 2000). In our sites, correlations with time since treatment suggest that populations of nonnative graminoids have been relatively stable and that herbs have increased very slowly over the two decades since treatment. However, to infer temporal trends retrospectively requires that nonnatives have been present in the landscape since the earliest treatments—a reasonable assumption for many of our naturalized species (e.g., *Bromus tectorum*, *Taraxacum officinale*, and *Cirsium vulgare* [White 1980, Knapp 1996, Young and Allen 1997]). Long-term trends suggest that further increases in abundance are likely to be small, although changes in climate, disturbance, or land use can potentially alter competitive interactions among native and nonnative plants (D'Antonio 2000).

A small percentage of sites (6% of all stands; 21% of thin + burn stands) exhibited substantially higher than average invasion by nonnatives (>10% relative cover). Basal area explained some of this variation: we found higher abundance of non natives as basal area decreased. We also found a positive relationship between area burned and abundance of nonnatives; in fact, this relationship may have been stronger if the observed negative relationship between area burned and time since treatment implies an underestimate of area burned in older treatments. Identifying additional factors or events that have contributed to localized proliferation of nonnatives in these forests could inform future efforts to minimize invasions.

Management considerations

Forest restoration treatments are being conducted at an accelerated rate in many western forests. In the United States, the focus of these treatments has largely been in stands in which *Pinus ponderosa* is dominant or codominant. These are the types of forests that are most adapted to frequent, low-intensity fire, and that have experienced the most substantial changes in fire regime over the last century. Selective logging of the largest, most fire-tolerant trees, grazing, and fire exclusion characterize the vast majority of these stands (Henjum et al. 1994). Establishment of shade-tolerant tree species, increases in density of subcanopy trees, and development of a multilayered forest structure have resulted (Hessburg et al. 2000), and many of these stands are now at high risk for stand-replacing wildfire. In addition to excessive mortality of canopy trees, stand-replacing fire can also lead to degradation of native understory communities either through the direct effects of intensive burning or the indirect effects of rehabilitation treatments, such as seeding of nonnative grasses (Schoennagel and Waller 1999, Keeley 2004). The high ecological value of remnant stands (Noss et al. 1995) suggests the need for ecological restoration to reduce risk of stand-replacing wildfire, but the potential for management activities to have unintended adverse ecological consequences, including introducing nonnative plants, remains a concern.

The management treatments imposed at our sites are representative of the types of thinning (50% reduction in basal area [Agee and Skinner 2005]) and prescribed burning that have been conducted throughout the interior Northwest to reduce hazardous fuel conditions. Thus, the small increase in nonnative plants observed in our geographically extensive sample can be viewed as a result that is regionally applicable to stands that have been managed in similar ways (i.e., single management entries using ground-based systems during the dry season). Invasions may be even smaller when logs are yarded in winter over snow or by helicopter, practices that result in less soil disturbance. However multiple management entries are often employed to maintain fire-safe conditions in dry forests. Although nonnatives exhibited only slight increases in abundance and diversity in response to initial treatments, exposure to repeated entries may increase the magnitude of invasion (Hobbs and Huenneke 1992, D'Antonio 2000). Managers concerned with the spread of nonnatives should monitor future responses to subsequent fuel-hazard reduction activities.

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APPENDIX

Mean frequency (percentage of sites) and cover of species ordered by growth form and origin (*Ecological Archives* A018-025-A1).