

Effects of Ecological Restoration Alternative Treatments on Nonnative Plant Species Establishment



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Abstract—Disturbances generated by forest restoration treatments have the potential for enhancing the establishment of nonnative species thereby impeding long-term native plant recovery. In a ponderosa pine forest next to the Fort Valley Experimental Forest, Arizona, we examined the establishment of nonnative species after three alternative treatments with different intensities of tree thinning, coupled with prescribed burning and an untreated control, in relation to total species abundance and richness. Pretreatment data were collected in 1998 and posttreatment responses were measured from 2001 through 2006. Total herbaceous cover and richness were significantly higher in the two more intensely thinned areas compared to the control over the entire post-treatment period. Native species were the most prevalent in terms of cover (92%) and richness (90%) across all treated units, though greater understory plant responses were linked to heavier amounts of tree thinning. Nonnative species abundance and richness also increased significantly in response to restoration treatments, particularly in the two more intense treatments. The proportion of nonnative abundance to the total abundance within the two heavily treated areas decreased through time and began to converge back towards the undisturbed control unit. One year following treatments, 15% of the total cover (27%) was composed of nonnative species in the heaviest treated unit. This proportion dropped almost 50% by the fifth year following treatment. Our results suggest that disturbances associated with restoration treatments can facilitate establishment of nonnative plants, however the post-treatment plant community was increasingly dominated by native species.

Introduction

Altered forest structure, functional processes and past land management practices have led to many critical conservation problems in southwestern ponderosa pine ecosystems, including loss of native biological diversity, declining herbaceous productivity and increased severity of disturbances such as wildfires (Bakker and

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Moore 2007, Covington and Moore 1994). Currently, efforts are underway to restore the ecological integrity and biodiversity of these ecosystems. Ecological restoration treatments, using thinning and prescribed fire are an effective approach for reversing the loss of habitat and biodiversity in ponderosa pine ecosystems (Landres et al. 1999, Moore et al. 1999). Both overstory thinning and prescribed burning can have mixed results on understory recovery, depending upon thinning level, burning frequency and severity, the community composition prior to the disturbance, past land use and climatic conditions during ecosystem recovery (Swetnam and Betancourt 1998; Wienk et al. 2004). While a primary goal of ecological restoration is to promote a self-sustaining indigenous plant community possessing all functional groups necessary to maintain the ecosystem (SER 2004), disturbances can shift the system into an alternate stable state (Laycock 1991). Proliferation of nonnative, disturbance-loving plant species can alter the successional trajectory of an ecosystem, leading to undesired results from the restoration project (Allen et al. 2002, Westoby et al. 1989).

Here, our objectives were to: (1) evaluate nonnative plant responses to different intensities of restoration treatments and (2) track changes in understory vegetation cover and richness over time. We hypothesized that total plant cover and richness, including nonnative species, would increase with increasing treatment intensity. However, we also hypothesized that nonnative species would eventually decline over time and contribute relatively little to the overall understory composition.

Methods

Study Site

We implemented restoration treatments on a ~56-ha (140-acre) site adjacent to the Fort Valley Experimental Forest, on the Coconino National Forest, northwest of Flagstaff, Arizona (N 35° 16', W 111° 44'). Prior to treatment, stands were close-canopied, even-aged "blackjack" *Pinus ponderosa* that averaged 726 trees/ha (1793 trees/acre) with occasional patches of presettlement "yellowpine." Stands were previously thinned but remained close-canopied. For a detailed site description reference Korb et al. (2007). Mean annual precipitation is 56 cm (22 in), although precipitation varied extremely throughout the duration of the study (Figure 1).

Restoration Treatments

All treatments focused on restoring site-specific overstory density and spatial arrangement consistent with presettlement forest patterns (Covington and Moore 1994, Fulé et al. 1997, Mast et al. 1999). Restoration treatments retained all living presettlement trees (described in: Covington and Moore 1994, White 1985). In addition, we retained postsettlement trees as replacements for remnant presettlement materials (e.g snags, logs, stumps). The three treatments differed in the numbers of postsettlement trees selected to replace dead presettlement evidence as described in Fulé et al. (2001). Trees were whole tree harvested, creating large slash piles. Slash piles were then burned prior to broadcast burning. Broadcast burning was conducted in spring 2000.

Treatments were randomly assigned to each unit and included: (a) 1.5-3 tree replacement (high-intensity), (b) 2-4 tree replacement (medium-intensity), (c) 3-6 tree replacement (low-intensity), and (d) no thinning, no burning (Control).

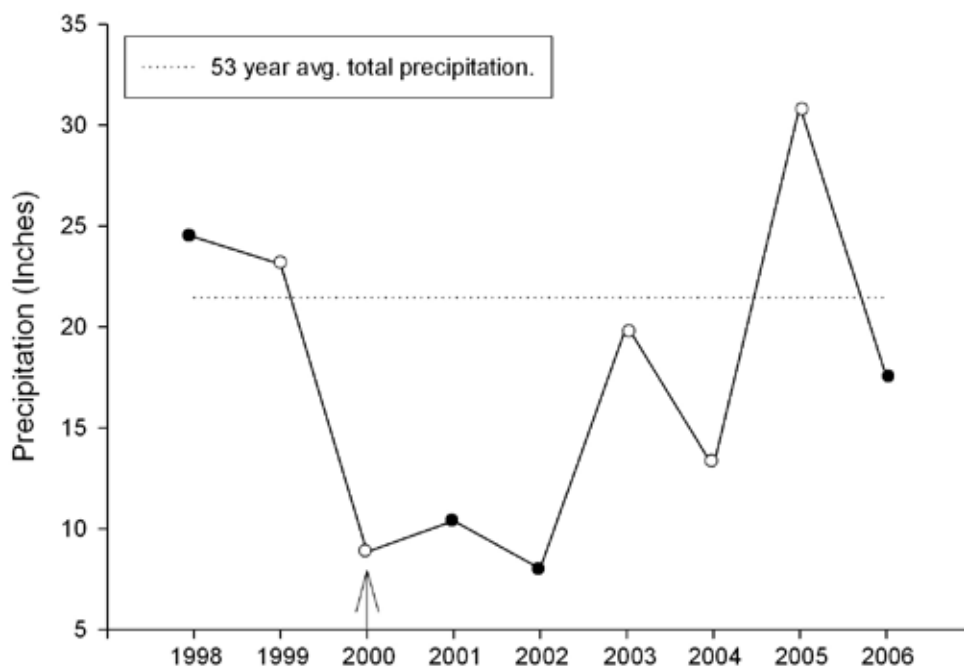


Figure 1. Mean annual precipitation during the study (1998-2006) versus the long-term 53 year average. The arrow denotes prescribed burn year. Dark symbols indicate years in which vegetation was sampled. Weather data were obtained from the Fort Valley Experimental Station weather records (USDA, Forest Service Rocky Mountain Research Station 2006).

Field methods and analysis

Each treatment was applied on a 14-ha (35-acre) unit. We established twenty subplots in each of the four treatment units. Understory data were collected in 1998 (pre-treatment), and re-measured in 2001, 2002, and 2006. A 50-m (164-ft) point line transect was used to quantify plant foliar cover and a belt transect 500m² (5382ft²) was used to quantify species richness on each plot (modified from USDI NPS 1992).

Statistical comparisons between the treatment units were carried out using 20 pseudoreplicated subplots in each treatment, since only one instance of each experimental treatment was implemented to each experimental unit. Understory total cover and richness within treatment were analyzed with repeated measures MANOVA. To account for significant differences in pretreatment richness, 1998 data were included as part of the effects model and analyzed with repeated measures MANCOVA. Total cover was transformed (square-root) in order to meet ANOVA assumptions. Following a significant treatment x time result, we compared treatment differences within year with Tukey's post hoc tests. Treatment effects on nonnative cover and richness were analyzed using non-parametric Kruskal-Wallis tests because these data strongly violated the assumption of normality. When results were significant, Mann-Whitney tests were used to make pairwise treatment comparisons. For all analyses, $\alpha = 0.05$. Where appropriate, alpha levels were adjusted using a Bonferroni correction (Kuehl 1994).

Results

We detected no pretreatment differences for any parameter except total richness (Figure 2). Total cover and richness averaged 12.3% and 18.4 species, respectively across all experimental units. Nonnative species contributed <1% of the cover and richness prior to treatment.

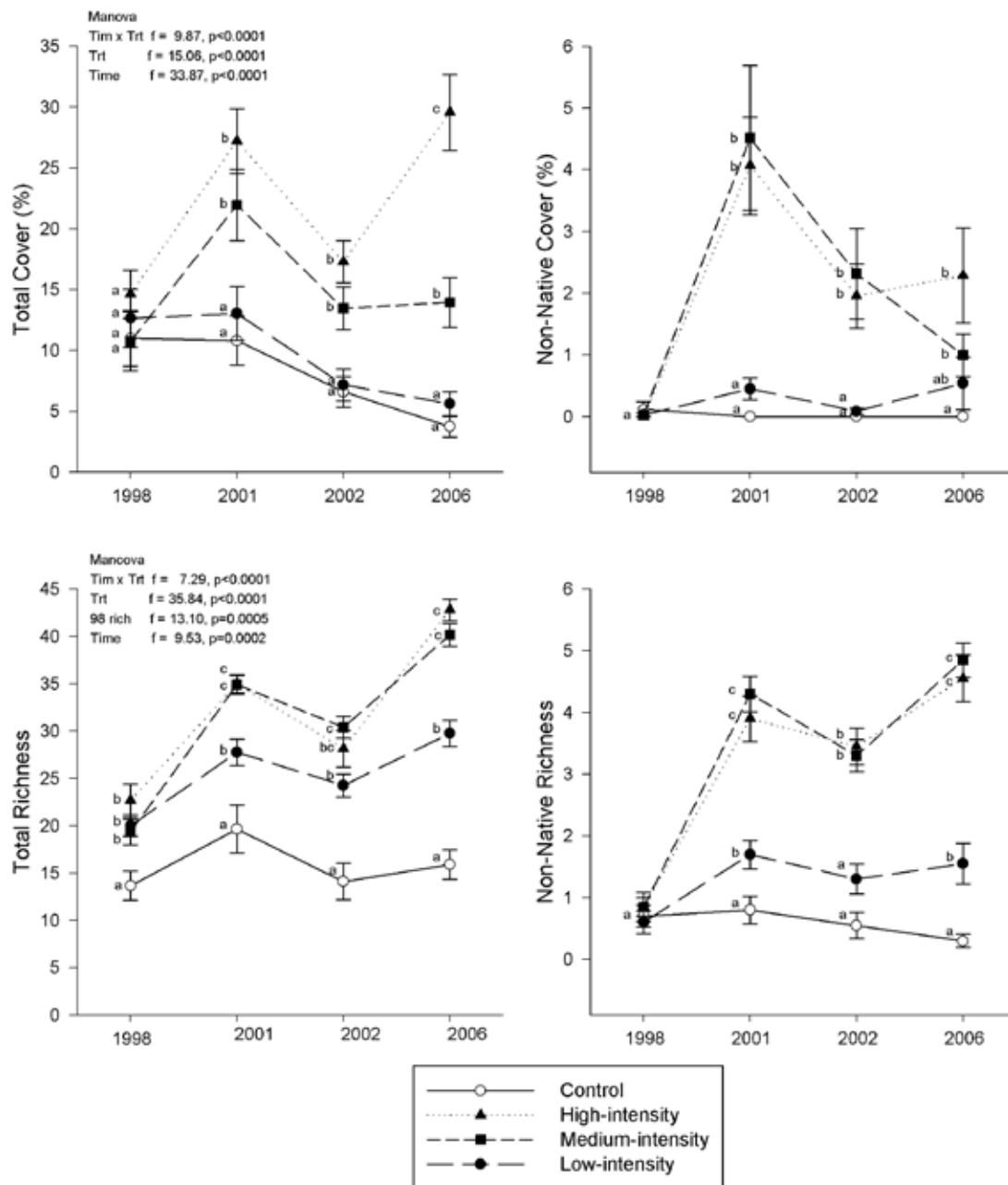


Figure 2. Average percent cover and species richness for total and nonnative species under experimental treatments in 1998, 2001, 2002, and 2006. Values indexed within each year by a different letter are significantly different at $\alpha = 0.05$. Bars represent ± 1 standard error of the mean ($n = 20$).

Table 1. Mean cover (S.E.) of dominant native and nonnative herbaceous species for each treatment unit and each sampling year. Nonnative species are denoted with an *. Graminoid species are denoted with an **. T signifies cover values less than 0.1%.

	Control			High			Medium			Low		
	1998	2001	2002	1998	2001	2002	1998	2001	2002	1998	2001	2002
<i>Carex</i> sp**	1.9 (0.5)	2.0 (0.7)	1.1 (0.3)	0.8 (0.3)	2.2 (0.5)	1.3 (0.3)	1.9 (0.3)	1.5 (0.4)	0.9 (0.2)	0.9 (0.2)	3.0 (0.7)	1.4 (0.4)
<i>Cirsium wheeleri</i>	0	0.2 (0.1)	T	T	0.4 (0.2)	0.3 (0.1)	1.5 (0.3)	0.8 (0.3)	T	0.6 (0.2)	0.2 (0.1)	T
<i>Elymus elymoides</i> **	0.8 (0.2)	1.5 (0.4)	0.7 (0.2)	0.4 (0.1)	2.9 (0.5)	3.6 (0.6)	6.9 (0.9)	6.8 (0.9)	3.6 (0.5)	4.3 (0.8)	2.4 (0.6)	1.1 (0.3)
<i>Festuca arizonica</i> **	2.7 (0.9)	3.0 (1.0)	1.9 (0.6)	0.9 (0.3)	3.0 (0.7)	5.8 (1.2)	6.7 (1.5)	3.9 (1.2)	2.8 (0.7)	2.4 (0.7)	1.3 (0.5)	1.0 (0.4)
<i>Muhlenbergia Montana</i> **	3.7 (1.2)	2.5 (1.1)	2.3 (0.8)	1.2 (0.7)	5.5 (1.4)	3.6 (0.8)	7.6 (1.7)	2.4 (1.2)	2.3 (1.0)	2.3 (0.9)	2.3 (0.5)	2.6 (0.7)
<i>Poa fendleriana</i> **	T	0.2 (0.2)	T	T	0.5 (0.1)	0.4 (0.2)	0.5 (0.2)	0.3 (0.1)	0.4 (0.1)	0.5 (0.2)	T	T
<i>Cirsium arvense</i> *	0	0	0	0	0	0	0.3 (0.2)	T	T	0.4 (0.2)	0	0
<i>Linaria dalmatica</i> *	T	0	0	0	0	0.6 (0.2)	1.4 (0.5)	T	T	0.4 (0.2)	T	T
<i>Verbascum thapsus</i> *	0	0	0	0	0	3.7 (0.7)	0.4 (0.2)	4.9 (1.2)	2.3 (0.7)	0.2 (0.1)	0.5 (0.2)	0.1 (0.1)

After treatment, total plant cover and richness differed significantly among treatments, time and treatment x time interaction (Figure 2). Plant cover and species richness were greatest in the high- and medium-intensity units following every posttreatment year. There were no differences in plant cover between the control and low-intensity treatments, although species richness was significantly greater in the low-intensity unit when compared to the control unit. Graminoids dominated the posttreatment understory (Table 1). Though the majority of increases in total cover and richness were due to native plants, there was a significant increase in nonnative species richness on all three treatments and nonnative cover in the high- and medium-intensity plots.

In 2001 (one year following burning), the understory in the high- and medium-intensity treatments showed significant increases in nonnative species cover and richness when compared to the low-intensity treatment and untreated control unit (Figure 2). Nonnative species comprised of 17% and 12% of the total understory cover (24.6%) and species richness (35.0), respectively across the two higher intensity treatments. In 2006, nonnative species cover continued to be significantly greater in the high- and medium-intensity treatments though decreased when compared to initial responses (Figure 2). Species richness continued to be significantly different between treatment and control units, however did not differ among the treatment intensities (Figure 2). The most common nonnative species across all treated units included: *Verbascum thapsus*, *Linaria dalmatica*, *Cirsium vulgare* (Table 1).

Discussion

Plant cover and species richness increased with thinning and prescribed burning treatments with the greatest responses occurring in the areas most heavily thinned. Our results are consistent with several other overstory-understory studies in ponderosa pine forest that demonstrated increases in understory productivity through the reduction of overstory density (Moore et al. 2006, Wienk et al. 2004, Moore and Deiter 1992). Research has also shown that understory production and diversity increases following fire, though increases are often species specific and highly dependent on the fire severity (Harris and Covington 1983, Wayman et al. 2006).

Disturbances have highly variable impacts on understory communities and often promote the establishment of nonnative species (Griffis et al. 2001, Keeley 2005). In our study the post-disturbance flora was comprised of mostly native species, although significant increases in nonnative species were found in the high- and medium-intensity treatments. While several of the nonnative species are of management concern, the total average cover of nonnative species did not exceed 5.0% in any of the treated units. Only *Verbascum thapsus* had foliar cover greater than 3%, immediately following treatment. After five years, however, cover had reduced to less than 0.5%.

Disturbance severity is often an important predictor in the spread of nonnative species (Crawford et al. 2001, Hunter et al. 2006, Keeley 2005). For example, Crawford et al. 2001 found high values of nonnative species establishment following severe wildfires, whereas Laughlin et al. (2004) found few nonnative species following a low-intensity wildfire. In the present study, prescribed fire severities were relatively low; though burning of slash piles resulted in high fire severity on a local scale that may have promoted the establishment of nonnative species. Our results suggest that varying levels of thinning intensity may influence the establishment of nonnative species, though thinning trees in general has the potential to promote the establishment of nonnative plants (Hunter et al. 2006). Different harvesting techniques may also produce different levels of soil disturbance that can facilitate the establishment of nonnative species (Battles et al. 2001, Korb et al. 2007). The present study was whole-tree harvested which can produce high levels of soil disturbance (Korb et al. 2007), thereby facilitating the initial establishment of nonnative species.

Disturbance is inevitable in ecological restoration treatments, thereby providing an opportunity for nonnative species to establish (D'Antonio and Meyerson 2002). What is not exactly clear is whether this invasion is short-lived or whether such disturbances provide an opportunity for the long-term persistence of nonnative species. Our results suggest disturbances associated with restoration treatments can facilitate the establishment of nonnative plants. Encroachment by nonnative species does not mean that these species will dominate the system (Figure 3). Time since disturbance should be considered an important factor when evaluating restoration targets within southwestern ponderosa pine forests. While our results are encouraging, more research is clearly needed as ecosystems are dynamic and further changes in community composition and structure are to be expected. The continued presence of aggressive nonnative species suggests continued monitoring of the site and potentially, further maintenance of the understory.

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Figure 3. Time-series photographs of a high-intensity plot prior to treatment (1998, top photo), 1 year after prescribed burn (2000, middle photo), and 5 years after prescribed burn (2006, bottom photo). The arrows highlight the same tree with a reference tag.

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