Slash Pile Burning Effects on Soil Biotic and Chemical Properties and Plant Establishment: Recommendations for Amelioration

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Abstract

Ponderosa pine forest restoration consists of thinning trees and reintroducing prescribed fire to reduce unnaturally high tree densities and fuel loads to restore ecosystem structure and function. A current issue in ponderosa pine restoration is what to do with the large quantity of slash that is created from thinning dense forest stands. Slash piling burning is currently the preferred method of slash removal because it allows land managers to burn large quantities of slash in a more controlled environment in comparison with broadcast burning slash. However burning slash piles is known to have adverse effects such as soil sterilization and exotic species establishment. This study investigated the effects of slash pile burning on soil biotic and chemical variables and early herbaceous succession on burned slash pile areas. Slash piles were created following tree thinning in two adjacent approximately 20-ha ponderosa pine (*Pinus ponderosa*) restoration treatments in the Coconino National Forest near Flagstaff, Arizona. We selected 30 burned slash pile areas and sampled across a gradient of the burned piles for arbuscular mycorrhizal (AM) propagule densities, the soil seed bank, and soil chemical properties. In addition, we established five 1-m^2 plots in each burned pile to quantify the effect of living soil (AM inoculum) and seeding amendments on early herbaceous succession in burned slash pile areas. The five treatments consisted of a control (no treatment), living soil (AM inoculum) amendment, sterilized soil (no AM inoculum) amendment, seed amendment, and a seed/soil (AM inoculum) amendment. Slash pile burning nearly eliminated populations of viable seeds and AM propagules and altered soil chemical properties. Amending scars with native seeds increased the cover of native forbs and grasses. Furthermore adding both seed and living soil more than doubled total native plant cover and decreased ruderal and exotic plant cover. These results indicate that seed/soil amendments that increase native forbs and grasses may enhance the rate of succession in burned slash pile areas by allowing these species to outcompete exotic and ruderal species also establishing at the site through natural regeneration.

Key words: **arbuscular mycorrhizal fungi, disturbance, ecological restoration, exotic species, plant succession, ponderosa pine, slash pile burning, soil amendments, soil seed bank.**

Introduction

Efforts are currently underway in southwestern *Pinus ponderosa* Dougl. ex Laws. (ponderosa pine) stands to reverse structural and functional ecosystem changes caused by historical land management practices. Southwestern ponderosa pine forests before Euro-American settlement were characterized by large old growth trees intermixed with grassy meadows. Low-intensity fires carried by grassy understories reoccurred every 2–20 years in southwestern ponderosa pine ecosystems and played a major role in determining the structure, composition, and stability of these ecosystems (Cooper 1960). These frequent, low-intensity fires, along with grass competition, restricted ponderosa pine from regenerating and maintained the open, park-like structure of presettlement ponderosa pine stands. Heavy livestock grazing, intensive logging of old growth trees, and fire suppression by Euro-American settlement resulted in dense forests consisting of numerous small trees along with other structural and functional changes to the ecosystem (Covington et al. 1997). Ponderosa pine restoration aims to restore ecosystem diversity and function by returning ponderosa pine forests to a more open savannah-like structure and reintegrating the natural disturbance regime of fire (Covington & Moore 1994). Two major components of this restoration effort include tree thinning and prescribed burning. One current dilemma in forest restoration is the disposal of slash (tree crowns, bole tips, and small boles) because often in ponderosa pine restoration treatments, large quantities of small-diameter trees need to be removed before reintegrating fire back into the ecosystem. Land managers commonly gather unmarketable small-diameter trees and slash into large piles near roads

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and burn them under appropriate weather conditions. Land managers favor this method of slash removal because piles can be safely burned under a broad range of weather conditions (Hardy 1996).

Although slash pile scars only cover a small (approximately <1.0) percent of forest ecosystems, they are problematic because they may assist the spread and establishment of undesirable non-native and ruderal plants into the forest interior following thinning (Dickinson & Kirkpatrick 1987). Despite ample anecdotal evidence of detrimental effects of slash pile scars, very few quantitative studies have assessed these effects, and no quantitative studies have been conducted to determine the methods of ameliorating them. It is crucial to have a thorough understanding of the effects of slash pile burning within restoration treatment areas to determine whether this activity conflicts with restoration goals and what methods are most effective in ameliorating slash pile scars created during restoration treatments.

We hypothesized that severely burned slash pile scars would have significantly reduced propagule densities of plants and arbuscular mycorrhizae (AM) fungi as well as altered soil properties. Therefore we hypothesized that seed and soil amendments containing viable AM propagules would increase the establishment of native plants in severely burned slash pile scars. Mycorrhizae form a critical link between aboveground plants and the soil system, playing an important role in plant nutrition, nutrient cycling, and the development of soil structure (Allen 1991; Smith & Read 1997). Numerous researchers have suggested a relationship between the recovery time of disturbed ecosystems and the abundance of infective propagules of mycorrhizal fungi (Allen & Allen 1980; Noyd et al. 1995; Gange et al. 1999). The severity of a disturbance can greatly influence the abundance of AM fungi in the soil and therefore its role in succession. Studies have shown that in severely burned areas AM fungi colonization (Schenck et al. 1975; Pattinson et al. 1999) and spore density and richness are reduced (Dhillon et al. 1988; Gibson & Hetrick 1988). When AM propagule densities are reduced the benefits of AM fungi to host plants are not immediately available to mycotrophic seedlings or mature plants. A recent study by Korb et al. (2003) showed that AM densities increase following restoration thinning treatments in ponderosa pine forests, which suggests that historically, AM fungi may have played an important role in the herbaceous community structure and function before Euro-American settlement.

The specific objectives of this study were to: (1) determine the effect of slash pile burning on densities of viable plant and AM fungal propagules and soil properties; and (2) determine the effect of amelioration treatments on herbaceous plant establishment within burned slash pile scars. We found that burning slash eradicates plant and AM fungal propagules and alters soil properties and that amending fire scars with native seed in conjunction with living soil increases the re-establishment of native herbaceous plant species richness and cover in comparison with other amendment treatments.

Methods

Experimental Site and Design

This research was conducted in 2000 and 2001 on the Coconino National Forest near Flagstaff, AZ, U.S.A. (N 35 16′ 00″, W 111 44′ 00″) as part of a larger experimental study investigating the effects of restoration thinning and burning in ponderosa pine stands. Our study sites were at 2,250 m elevation in two adjacent approximately 16 ha units that were thinned in the winter of 1999–2000. Piles of slash created during tree thinning were burned in February 2000. Over 50 slash piles (averaging 9 m in diameter and 3–4 m tall) were created from the thinning treatments (Fig. 1). Estimated total fuel mass consumed for each slash pile averaged between 1.4 and 3.2 metric tons (Hardy 1996). For this research we chose 30 slash piles with similar total fuel mass to minimize variation in fire severity among piles.

Before thinning and prescribed burning, the vegetation at the study site consisted of pure ponderosa pine stands, with few large trees intermixed with numerous small trees (1162 ± 141.4 trees/ha). The perennial grasses *Elymus elymoides* (Rafinesque) Swezey (squirreltail), *Muhlenbergia montana* Nuttall (mountain muhly), and numerous forbs dominated the understory. Soils are moderately well-drained clays and clay loams derived from basalt and are classified as fine, montmorillonitic Typic Argiboroll (Terrestrial Ecosystems Survey of the Coconino National Forests 1995). The A horizon extends only 10 cm deep and the remaining soil profile extends 114–152 cm deep before reaching the fractured basalt bedrock (Overby 2001 USFS soil scientist, personal communication). Precipitation occurs as snowfall and rain during the winter, a pronounced drought in May and June, and frequent monsoon rains in July and August. Average summer precipitation is 20.8 cm. Daily mean temperatures range from 5 to 17° C throughout the year (Sackett 1980).

Figure 1. One of 30 slash piles (averaging approximately 9 m in diameter and 3–4 m tall) researched in this study before burning.

Fire Intensity Gradient

Boundaries of slash pile scars were clearly identified by white ash, charred, blackened soil, and no vegetation. Transects were established in a N–S direction across the boundaries of each of the 30 slash pile scars, with 3 m extending inside and 3 m outside the scar. We assumed that sampling points along these transects at 3 m inside the scar, 1.5 m inside the scar, scar edge, and 3 m outside the scar represent a fire intensity gradient, with the hottest soil temperatures 3 m inside the scar and coolest 3 m outside the scar.

Mycorrhizae Field and Laboratory Methodology. One soil sample was collected from each point along all 30 slash pile transects in May 2000 and May 2001 (3 and 15 months after burning). We were unable to collect soil samples before burning for these same points because the exact location of piled slash was unknown before slash piling. Therefore 10 random samples were taken in the general vicinity of the slash piles to provide a background AM inoculum measurement before burning for the study area using the same methodology (Korb et al. 2003). Soils were collected to a depth of 15 cm using a hand trowel and immediately placed into 4 cm diameter $\times$20 cm deep Conetainers (Stuewe and Sons, Inc., Corvallis, OR, U.S.A.). We used a *Zea mays* L. (corn) bait-plant bioassay to determine the relative amount of infective propagules of AM fungi in our soil samples. Corn is mycotrophic with many species of AM fungi and grows rapidly and uniformly; these advantages outweigh the disadvantage of not using a native host plant (Johnson et al. 1999). Bait-plant bioassays are designed to detect all types of viable mycorrhizal fungal propagules including spores, fragments of mycorrhizal roots, and extraradical hyphae and therefore more accurately quantify total densities of AM fungal propagules than direct counts of sporocarps, spores, or colonized root lengths (Brundrett & Abbott 1994). Corn seeds were sown in freshly collected soils and the plants were maintained in a greenhouse for 6 weeks. Corn roots were cut into 2.5-cm segments and subsamples were cleared in 5% KOH and stained with Trypan blue in lactoglycerin (Koske & Gemma 1990). Proportion of root length containing AM fungal structures (arbuscules, vesicles, coils, and hyphae) was measured by the gridline intersect method using a dissecting microscope which gives a relative amount of infective AM propagules (Giovannetti & Mosse 1980).

Soil Seed Bank Field and Laboratory Methodology. In May 2000, one soil sample was collected from each point along all 30 slash pile transects ($n = 4$) using a 5 cm diameter metal core to a depth of 5 cm to determine the viable soil seed bank. These sampling point locations were the same for the mycorrhizae soil cores. We assessed populations of viable seeds by placing each individual soil core on top of 2 cm of sterilized soil in a 11 $\times$ 11–cm plastic container in an unheated greenhouse (18–30 °C). Containers were watered daily using an automatic mist system. Over a 4-month period, plant seedlings that emerged were identified to species, counted, and removed. Seed bank data were analyzed by averaging both individual species and the total number of seedlings per transect and converting this number to the number of seeds/m^2. Seeds/m^2 is the common standardized unit used in seed bank studies. Containers of sterilized soil were also maintained without living soil cores to determine if there was any greenhouse contamination.

Soil Chemical Field and Laboratory Methodology. In May 2000, soil samples for chemical analyses were collected to a depth of 10 cm using a hand trowel 3 m inside the scar, at the scar edge, and 3 m outside the scar along all 30 slash pile transects. In October 2001, soil samples were collected 10 m outside the scar in addition to the other sampling locations to quantify KCl-extractable ammonium nitrogen (NH_4-N) and nitrate nitrogen (NO_3-N). Soil chemical properties were analyzed at the Bilby Research Soil Analysis Laboratory (Flagstaff, AZ, U.S.A.). Soil pH was determined in a 1 : 1 slurry using a pH meter. Total N and P were measured following Kjeldahl digestion by automated colorimetry using a Technicon auto-analyzer (Technicon Instruments Corp., Tarrytown, NY, U.S.A.) (Parkinson & Allen 1975). Available NH_4-N and NO_3-N were determined from KCl extracts of freshly collected soil by automated colorimetry using a Technicon auto-analyzer (Parkinson & Allen 1975). Organic C was determined by loss on ignition, samples were heated to 425° C for 24 hr and organic matter was estimated from mass loss.

Amelioration Experiment

Five 1-m^2 (0.5 $\times$ 2 m) plots were located horizontally within each slash pile scar (at least 1 m from the scar edge and at least 1 m from each other) so that each plot would have a similar underlying fire gradient intensity. Each plot was randomly assigned one of five treatments: (1) no treatment (control); (2) living soil amendment (containing microorganisms, AM fungi, and plant propagules); (3) sterilized soil amendment (no propagules); (4) seed amendment (11.2 g of native seeds); or (5) seed/soil amendment. No seed/sterilized soil amendment was applied and therefore inferences regarding the biotic soil component on plant establishment are limited to this study. Topsoil that had recently been stockpiled from road construction 1 km from the experimental site was used for the soil amendments. A previous study showed high densities of viable AM propagules in this soil (Korb et al. 2001). Soil taken from the same location was steam sterilized for 48 hr to create the sterilized soil amendment without living organisms but with similar chemical and physical properties as the living soil amendment. Similarities of chemical properties were verified between the sterilized soil and the living soil amendment after sterilization (Korb, unpublished data). In June 2000, second, third, and fifth treatments had 20,000 cm^3

of living or sterilized soil layered onto each 1-m^2 plot, resulting in an average amendment depth of 2 cm. When the amendment soil was collected, 20 soil cores were sampled to quantify populations of viable seeds using the procedures described previously.

In May 2001 (15 months after burning and 11 months after soil amendments were added) 15-cm deep soil samples were collected from the experimental plots in each slash pile scar to quantify AM propagule densities using bait-plant bioassays. We used bait-plant bioassays instead of harvesting individual plants to determine AM colonization levels because we were also interested in quantifying the long-term success of plant establishment in these amendment plots. In addition, these amendments are being used as restoration treatments to establish native plants and prevent the establishment of non-native species in slash pile scars. Therefore harvesting native plants to quantify AM propagule densities would be counterintuitive to this goal even though it would be a more direct assessment of AM colonization for our individual treatments. In October 2001 soil samples were also collected to a depth of 10 cm from the control, soil amendment, and seed/soil amendment to quantify KCl-extractable NH_4-N and NO_3-N (Table 1).

Seed amendments consisted of a mixture of 19 species of native grasses and forbs that were abundant in the extant vegetation surrounding the slash pile scars. Seeds were purchased from the nearest possible regional source (Colorado Native Seed in Salida, CO, U.S.A.). Purity of the seed was verified in greenhouse and field trials. Fourth and fifth treatments were sown with 8.4 g of grass seed and 2.8 g of forb seed per 1-m^2 plot. Seeds were sown in early July 2000 (5 months after burning) to coincide with the monsoon rains in late summer and mimic treatments that could be implemented by land managers (i.e., no supplemental water). In September 2000 and 2001 (7 and 19 months after the piles were burned) seedlings and mature plants in all the plots were identified and counted. In addition the cover of each plant species was determined using the ocular estimate method to the nearest 0.1%. The phenological stage (e.g., rosette, flower, or seed) and plant height was also recorded.

Statistical Analysis

Repeated measures multivariate analysis of variance (MANOVA) and analysis of variance (ANOVA) were used. The Shapiro–Wilk test was used to test data for normality, and Leven's test was used to test for homogeneity of the variance (Zar 1984). Before analysis, some data were transformed to improve normality and homoscedasticity assumptions (Zar 1984). Root colonization data were arcsine square-root transformed, soil seed bank densities and some individual plant species variables were $\log(x + 1)$ transformed, and soil properties were square-root transformed. Tukey's honestly significant difference test was used to make multiple comparisons of means following a significant result. All analyses were performed using SPSS version 8 (SPSS 1998).

Results

Fire Intensity Gradient

Three and 15 months after burning, corn bait-plants grown in soils collected from inside the slash pile scars had significantly lower AM colonization than those grown in soils collected from the scar edges and 3 m outside the scars (Fig. 2). Infective propagule densities of AM fungi were also significantly reduced at the scar edges in comparison with 3 m outside the scars. Between 3 and 15 months after burning, there was a significant increase in AM propagule densities at the scar edge and 3 m outside the scar but not in the interior of the slash pile scar (Fig. 2).

In 2000, there were significantly more viable seeds 3 m outside the scars than 3 m inside the scars or at the scar edges (Fig. 3). Most viable seeds were from exotic or ruderal species (native species that prefer disturbed, high-nutrient habitats) (Grime 1977). Three meters outside the scar, the exotic seed bank was dominated by *Verbascum thapsus* L. (mullein), an early successional species of disturbed habitats, with a mean of 368 seeds/m^2. *Linaria dalmatica* (L.) P. Mill. (dalmation toadflax) accounted for the remaining exotic seed bank with 45 seeds/m^2. *Laennecia schiedeana* (Less.) Nesom (pineland marshtail) and *Erigeron divergens* Torrey & Gray (spreading fleabane) were the dominant ruderal species, and *Carex* spp. and *Muhlenbergia montana* were the dominant native species in the seed bank.

Differences in soil chemical properties were evident across the fire intensity gradient. Soil pH was significantly higher 3 m inside the scars in comparison with the scar edges and 3 m outside the scars (Fig. 4a). In contrast, total N and organic C were significantly higher 3 m outside the scars in comparison with the scar edges and 3 m inside the scars (Fig. 4b & 4c). There was no significant difference

Table 1. KCl-extractable inorganic nitrogen 20 months after slash burning at four locations from the slash pile edges.

	3 m Inside Scar	*Scar Edge*	*3 m Outside Scar*	*10 m Outside Scar*
NH_4-N (mg/kg)	7.92 ± 1.44^a	3.94 ± 0.34^b	5.47 ± 0.39^a	4.1 ± 0.47^b
NO_3-N (mg/kg)	12.95 ± 5.73^a	0.29 ± 0.16^b	0.97 ± 0.45^b	1.02 ± 0.81^b

Data are expressed as means ± SE (n = 30). Within each row, values indexed by different letters are significantly different at the $p \leq 0.05$ level as determined by Tukey's honestly significant difference test. NH_4-N analysis was on natural transformed data and NO_3-N analysis was on square-root-transformed data.

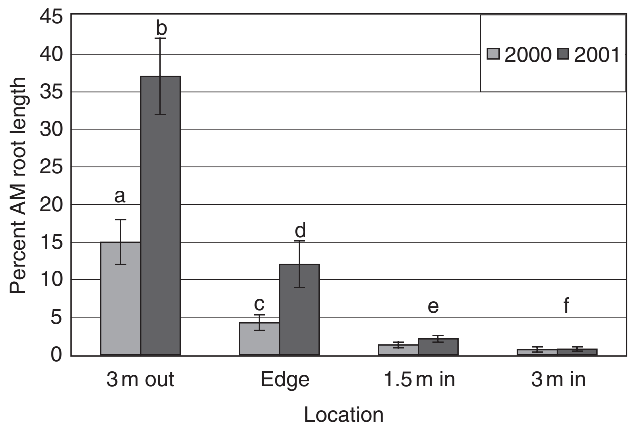

Figure 2. The relative amount of mycorrhizal propagules indicated by percent arbuscular mycorrhizal (AM) colonized root length in corn from bait-plant bioassays along the fire intensity gradient in 2000 and 2001, 7 months and 19 months after burning. Differences between the locations along the fire intensity gradient were determined by MANOVA repeated measures on arcsine square-root-transformed data. Values indexed by different letters are significantly different at the $p = 0.05$ level.

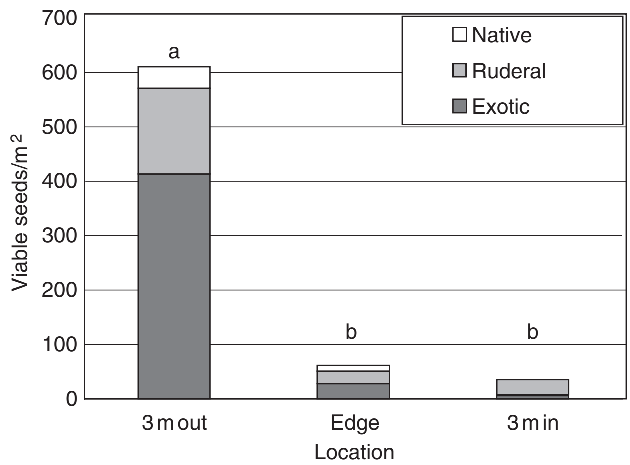

Figure 3. The density of viable seeds of native, ruderal, and exotic plant species along the slash pile gradient in 2000, 7 months after burning. Differences between the locations along the gradient were determined by ANOVA on $\log(x + 1)$-transformed data. Values indexed by different letters are significantly different at the $p = 0.05$ level.

in total P across the transects (Fig. 4d). KCl-extractable NH_4-N and NO_3-N were significantly higher 3 m inside the scar in comparison with the scar edge and 10 m outside the scar edge (Table 1).

Amelioration Experiment

Soils collected from control plots and plots amended with sterilized soil had significantly lower densities of infective AM propagules than soils from plots amended with both seed and soil 15 months after burning and 11 months after soil amendments were added to the slash pile scars (Fig. 5). There was no significant difference in KCl-extractable soil NH_4-N or NO_3-N between the control, sterilized soil amendment, and seed/soil amendment plots (mean values were 9.7 mg/kg NH_4-N and 14.1 mg/kg NO_3-N).

The live soil amendment contained an average of 256 viable seeds/m^2, with *Carex* spp. the most abundant species and *M. montana, V. thapsus*, and *L. schiedeana* also common. During the first growing season in 2000 (7 months after burning and 2 months after seeding) only seeded forbs and grass species became established, and there was no natural regeneration of ruderals or exotics. Forb and graminoid cover was significantly higher in the seed/soil amendments than the seed amendments (Fig. 6a & 6b). During the second growing season in 2001, native forb and graminoid cover was significantly higher in the seed-amended plots than in the control, sterilized soil or soil amended plots but lower than plots amended with both seed and soil (Figs. 6a & 6b, 7a & 7b). There was no significant difference in exotic plant cover between any of the amendment treatments, but the seed/soil amendments had the lowest exotic cover (Fig. 6c). The most abundant exotic species was *V. thapsus*. Other exotic species included *Cirsium vulgare* (Savi) Tenore (bull thistle), *Amaranthus albus* L. (amaranth), and *L. dalmatica*. Ruderal species had significantly higher cover in the control, sterile soil amendment, and soil amended plots than those amended with both seed and soil (Fig. 6d). The most abundant ruderal species was *L. schiedeana*, an early colonizing biennial cudweed.

Seventeen species flowered the second growing season. *Elymus elymoides* was the most abundant flowering species followed by *Heliomeris multiflora* Nuttall (showy goldeneye), *Muhlenbergia wrightii* Vasey (spike muhly), and *Bouteloua gracilis* Humboldt (blue grama). The seed/soil amendments had the highest total number of flowering individuals (41) followed by the seed amendments (27), soil amendments (17), sterile soil amendments (8), and the controls (6). Three exotic species flowered in the control, sterilized soil, or soil amendments: *V. thapsus*, *L. dalmatica*, and *A. albus*.

Five of the seeded species did not germinate in any of the plots either growing season and four species only germinated during the second growing season (Table 2). Native grass cover was significantly higher for seeded *B. gracilis*, *E. elymoides*, and *M. wrightii* in plots amended with live soil and seed than seed-only plots for 2000 and 2001 (Table 2). Similarly native forb cover was significantly higher for seeded *Ipomopsis aggregata* and *Linum lewisii* in plots amended with live soil and seed than seed-only plots for 2000 and 2001, and *Penstemon virgatus* and *Thermopsis divaricarpa* was significantly higher in plots amended with live soil and seed for 2001 (Table 2). None of the native seeded species did significantly better in the seed-only plots than the live soil and seed plots (Table 2).

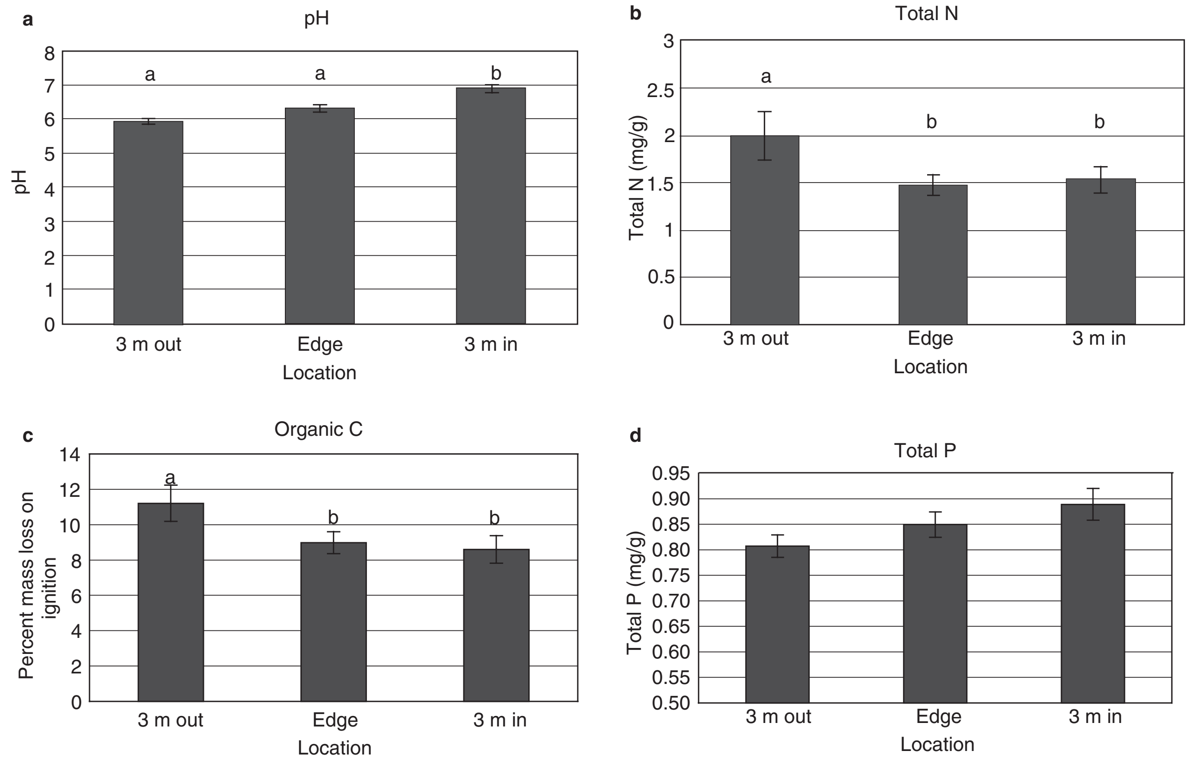

Figure 4. Soil chemical properties along the fire intensity gradient for 2000, 7 months after burning. Differences between the locations along the fire intensity gradient were determined by ANOVA on square-root-transformed data. Values indexed by different letters are significantly different at the $p = 0.05$ level.

Discussion

Slash pile burning leaves persistent scars on the land and therefore may conflict with managers' goals for *Pinus ponderosa* stands. We found that slash pile scars have significantly altered soil properties and virtually no viable plant seeds and AM propagules remain. In contrast, a preburn reference study measuring viable plant seeds and AM propagules in the same study area showed an average of 1,176 seeds/m^2 and an average AM colonization of 33.2% (Korb et al. 2003). Both of these studies corroborate that the absence of viable plant seed and AM propagules is the result of slash pile burning. Other studies also report the loss of seed banks (Hassan & West 1986; Odion & Davis 2000) and viable AM propagules (Klopatek et al. 1988; Berch et al. 1993) after intense fire. Combustion of the litter and duff horizon along with soil heating during slash burning is most likely responsible for the low number of viable seeds (Moore & Wein 1977) and AM propagules in the scars that we studied. Maximum soil surface temperatures are usually 500–700 °C during heavy slash burns in chaparral (Rundel 1983), and temperatures at 10 cm and 22 cm depth in mineral soil can reach over 250 and 100° C, respectively, in pine (Roberts 1965). Within one growing season, populations of viable AM fungal propagules were recovering at the scar edge but not in the center of the scar. This corroborates with other studies that show neutral or positive responses of AM activity after low-intensity fires where AM hyphae most likely survive in roots of nearby plants (Rashid et al. 1997) or as propagules deeper in the soil profile where soil heating is not severe (Pattinson et al. 1999; Korb et al. 2001).

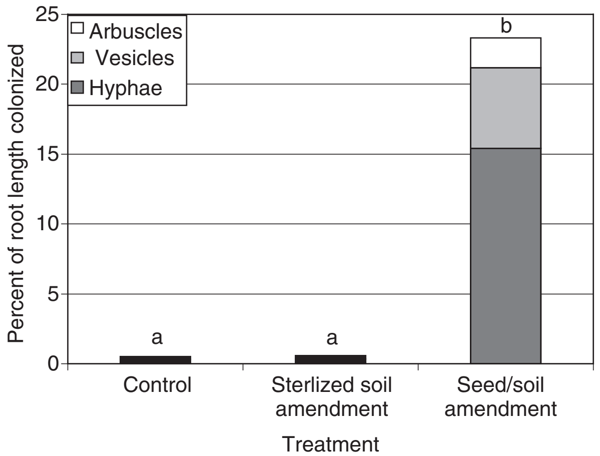

Figure 5. Relative root colonization with arbuscular mycorrhizal (AM) fungal structures (arbuscules, vesicles, and hyphae) in corn roots grown in soils collected in 2001 from the control plots and plots amended with sterilized soil (no propagules) and both seed and live soil. Differences between the AM fungal structures were determined by ANOVA on arcsine square-root-transformed data. Values indexed by different letters are significantly different at the $p = 0.05$ level.

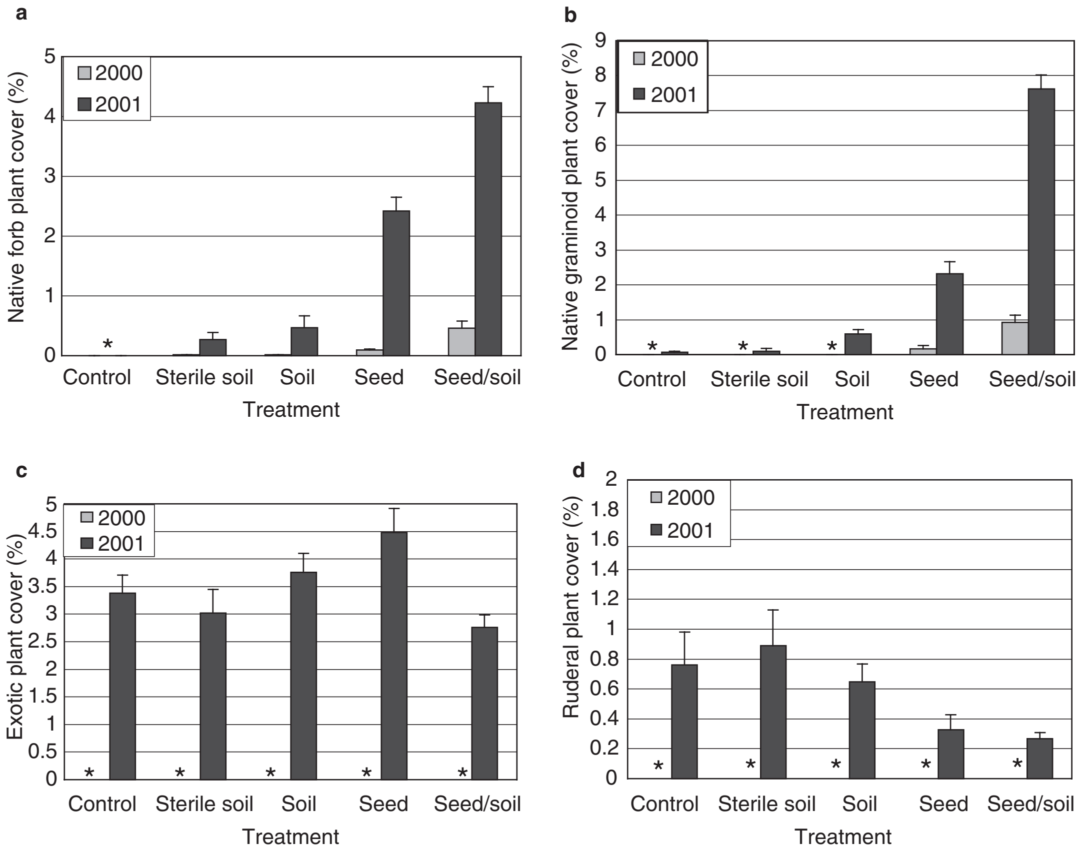

Figure 6. Average cover of (a) forbs (b) graminoids (c) exotic plants, and (d) ruderal species established under different amelioration treatments. Differences between treatments were determined by MANOVA repeated measures. The symbol * indicates that herbaceous plant cover was zero.

Slash pile burning also altered soil chemistry. Elevated soil pH within the scars was likely due to the hydrolysis of base cation oxides that are abundant in ash after severe burns (Fuller et al. 1955; Ballard 2000). Organic C and total soil N were significantly lower inside the scars, suggesting that the intense fire volatilized C and N compounds (Klemmedson 1976; Weast 1988). Studies have shown that although burning may reduce total soil N immediately after a fire, NH_4-N often increases because of pyrolysis of forest floor material, and within a year NO_3-N also increases due to nitrification of the NH_4-N pulse, which is consistent with our study findings (Covington et al. 1991; Ellingson et al. 2000). Unlike N and organic C, total soil P showed no significant response to slash pile burning. Because 94–98% of soil P is located in the mineral soil layers and not in the litter layer, P is not as sensitive to severe burning as N (Neary et al. 1999). Soil chemistry changes, which affect nutrient availability, can influence post-disturbance vegetation. In particular ruderal and non-native species can often outcompete native species in high-nutrient environments.

Slash pile construction using heavy mechanized equipment can often be very detrimental to soil physical properties. Soil physical parameters were not measured in slash pile-constructed areas; however, a study throughout the entire restoration treatment area showed no significant soil compaction associated with tree-thinning treatments (Korb, unpublished data). Areas of moderate and severe soil profile disturbance were noted in areas surrounding slash pile areas due to slash pile construction (Korb, unpublished data).

Slash pile scars are problematic because they often remain unvegetated (Covington et al. 1991) or become colonized by exotic (frequently invasive) plant species (Dickinson & Kirkpatrick 1987). Consequently we were interested in identifying restoration treatments that would encourage the establishment of native plants and discourage colonization by exotic plants. Our amelioration experiment indicates that the addition of both living soil and native seed generated the highest species richness and abundance of native plants and the lowest species richness and abundance of exotic and ruderal plants. Direct seeding on ash resulted in lower native species richness and less

Figure 7. (a) Squirreltail (*Elymus elymoides*) in a soil/seed amendment 1-m^2 plot (19 months after burning and 15 months following soil/seed amendment). (b) Numerous grass and forb species in a soil/seed amendment 1-m^2 plot (19 months after burning and 15 months after soil/seed amendment).

than half the native plant cover compared with plots amended with both seed and living soil. Also exotic plant cover was significantly lower in plots receiving dual seed and soil amendments compared with those receiving only seeds. Areas amended with living soil and seed were more closely related to adjacent areas that had low-severity burns than the other amendment treatments including seed only. On average native plant cover in our living soil and seed-amended plots in 2001 was 11.9% cover, which is similar to native plant cover outside the slash piles (11.2%) in areas that received tree thinning and a low-severity burn (restoration treatment), which represents an artificial target for native plant cover (Korb, unpublished data). Conversely areas within the slash pile scar with no soil or seed amendment (control) had on average less than 0.01% native plant cover.

Plant community composition is clearly impacted by the presence and function of AM symbioses (van der Heijden et al. 1998; Hartnett & Wilson 1999; Klironomos 2002). Consequently reintroduction of AM fungi in the living soil amendment may have facilitated the success of the seed/soil treatments. Propagule densities of AM fungi were higher in the seed/soil treatment than in any of the other treatments. However, because we did not specifically control for AM fungi in this study, this conclusion needs further verification through future research. Other studies have found that AM fungi can promote the establishment of native plant cover in disturbed habitats (Gange et al. 1990; Smith et al. 1998) and that AM fungal propagule densities remain low in severely disturbed soil environments unless efforts are made to restore them (Rives et al. 1980; Gould et al. 1996). Other biotic soil components such as bacteria and actinomycetes may have played a role in facilitating plant establishment.

Abiotic properties of the soil amendment may have also facilitated successful establishment of native plants by generating an environment that is more conducive to seed germination and recruitment. In our experiment adding 2 cm of topsoil probably ameliorated water infiltration, soil–gas exchange, microclimate, and microtopography compared with bare ash (Chambers et al. 1990; Chambers 2000).

Management Recommendations and Conclusion

Biotic and abiotic changes that occurred in the burned slash pile areas for our study are consistent with research in different forest types that have quantified changes in severely burned areas (DeBano et al. 1979; Dickinson & Kirkpatrick 1987; Moreno & Oechel 1991). This consistency in biotic and abiotic changes, regardless of forest type, implies that our findings are relevant to other forest community types where slash pile burning, which results in severely burned areas, is part of forest management. If slash pile burn areas are either qualitatively or quantitatively assessed to be severely burned, amelioration techniques to minimize non-native plant establishment need to be implemented regardless of the forest community type. At a minimum, severely burned slash pile areas need to have seed amendments. If AM fungi play an important role in plant establishment for a particular forest community type, then soil amendments are also recommended. These recommendations are not to be confused with severely burned areas from prescribed or natural fire. Some forest and woodland types in the western U.S. have evolved with high-intensity fires (e.g., lodgepole pine and chaparral), and these areas generally revegetate on their own because their native species are highly adapted to severe fires and many depend on severe fires for regeneration.

Our results show that burning slash piles generate scars with increased susceptibility to invasion by exotic plant species. Generally the most opportune time to control or reduce exotic species is during early spread before large densities or large areas are occupied (Byers et al. 2002). Because anecdotal evidence suggests that slash pile scars have long-term negative effects, we recommend that land managers minimize burning large piles of slash in any forest treatments. Management decisions about slash removal need to weigh the feasibility of alternative slash

Table 2. Seeded species' average cover for the seed and seed/soil amendment plots in 2000 and 2001.

Species	Average Cover (2000) Seed	Average Cover (2000) Seed/Soil	Average Cover (2001) Seed	Average Cover (2001) Seed/Soil
Grasses				
Bouteloua gracilis	0.03 ± 0.01^a	0.35 ± 0.06^b	0.07 ± 0.03^a	0.26 ± 0.06^b
Elymus elymoides	0.10 ± 0.03^a	0.41 ± 0.07^a	2.02 ± 0.96^a	6.12 ± 1.3^b
Festuca arizonica	0.01 ± 0.01^a	0.02 ± 0.01^a	0.07 ± 0.03^a	0.14 ± 0.07^a
Koeleria macrantha	*	*	*	*
Muhlenbergia wrightii	0.02 ± 0.02^a	0.13 ± 0.02^b	0.17 ± 0.09^a	0.82 ± 0.30^b
Poa fendleriana	*	*	*	*
Schizachyrium scoparium	*	*	*	*
Forbs				
Artemisia ludoviciana	0 ± 0^a	0.06 ± 0.04^a	0.02 ± 0.01^a	0.14 ± 0.04^b
Castilleja linariifolia	*	*	*	*
Geum triflorum	*	*	*	*
Heliomeris multiflora	0.01 ± 0.01^a	0.04 ± 0.02^a	0.48 ± 0.21^a	0.36 ± 0.13^a
Ipomopsis aggregata	0.02 ± 0.01^a	0.12 ± 0.02^b	1.07 ± 0.25^a	2.16 ± 0.49^b
Iris missouriensis	0 ± 0.0^a	0 ± 0^a	0 ± 0^a	0.01 ± 0.01^a
Linum lewisii	0.01 ± 0.01^a	0.11 ± 0.04^b	0 ± 0^a	0.13 ± 0.07^b
Lupinus argenteus	0.02 ± 0.01^a	0.01 ± 0.01^a	0.02 ± 0.01^a	0.01 ± 0.01^a
Oxytropis lambertii	0.03 ± 0.01^a	0.06 ± 0.02^a	0.22 ± 0.21^a	0.28 ± 0.22^a
Penstemon barbatus	0 ± 0^a	0 ± 0^a	0.51 ± 0.21^a	0.59 ± 0.13^a
Penstemon virgatus	0 ± 0^a	0 ± 0^a	0.02 ± 0.01^a	0.18 ± 0.08^b
Thermopsis divaricarpa	0 ± 0^a	0 ± 0^a	0 ± 0.01^a	0.12 ± 0.02^b

Data are expressed as means ± SE ($n = 30$). Values indexed by different letters are significantly different at the $p \leq 0.05$ level as determined by Tukey's honestly significant difference test for each sampling year within a row. Species that did not germinate or did not establish as seedlings during the 2000 or 2001 growing season are denoted by an asterisk.

removal techniques against financial constraints and recognize that trade-offs exist between alternative restoration decisions (e.g., local soil effects versus economic impacts). If slash pile burning is the only realistic management option for safely reducing large amounts of fuel in restoration treatments, we recommend that slash be piled and burned on existing roads to minimize ecological impacts. We also recommend that severely burned slash pile scars are treated with seed and soil amendments within the first growing season after burning to increase the rate of native plant establishment and discourage the invasion of exotic species. In addition native seeds of species common in the area should be used from a local source to minimize genetic contamination from seeding amendments, and soil should also be taken from the local vicinity.

Some ecological restoration treatments, particularly a few forest restoration activities, result in severe environmental disturbances that need restoration treatments themselves (e.g., slash piles, landing areas, and roads). Because these forest restoration treatments are ecologically based, the first priority of these treatments should be to minimize or eliminate the need to restore areas within the treatment area because of restoration activities. If restoration-caused disturbance is deemed necessary to complete the treatment, restoration researchers, however, need to understand how to rehabilitate or restore these areas so that by trying to restore forest health we do not create new restoration problems (e.g., the establishment of exotic species).

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