SOIL AND PLANT RESPONSE TO SLASH PILE BURNING IN A PONDEROSA PINE FOREST

by

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APPROVAL

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This thesis has been read by each member of the thesis committee and has been found to be satisfactory regarding content, English usage, format, citation, bibliographic style, and consistency, and is ready for submission to the Division of Graduate Education.

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ABSTRACT

Slash pile burning is the most common method of forest residue disposal following ponderosa pine restoration harvests, which are intended to reduce the risk of catastrophic fire and restore the historical structure and function of forests in western Montana. The impact of high-intensity, long-duration fire (pile burning) on soil processes and plant community dynamics is not well understood. The objectives of this study were: (1) to characterize the influence of slash pile burning on soil nutrient availability, soil microbial activity, and arbuscular mycorrhizal (AM) infection; (2) to compare seeding and soil amendment effects on burn scars. In May 2006, slash piles were burned in a ponderosa pine stand near Florence, Montana and 45 scars were sampled. Soil samples were collected from three locations in each slash pile to a depth of 10 cm and characterized for available soil NH_4^+-N, NO_3^--N, potentially mineralizable nitrogen (PMN), and total C and N, water-soluble PO_4^{3-}-P, microbial biomass, and mycorrhizal inoculum potential (MIP). In the burned center, soil NH_4^+-N was greatest one month post-burn and remained elevated one year later. There was no observable increase in NO_3^--N until one year post-burn. Soluble PO_4^{3-}-P was not impacted by burning. Microbial biomass was reduced by burning and did not recover one year later. Pile burning greatly reduced MIP.

In October 2006, fire scars were either seeded with native graminoids or left non-seeded, divided into subplots, and assigned to one of five treatments: control, addition of local organic matter, scarification, scarification and organic matter addition, or scarification and commercial compost addition. Soils were monitored for the previously measured soil parameters and resin-sorbed inorganic N. Scarification with organic matter amendment and scarification with compost amendment both ameliorated soil properties. Seeding most effectively increased plant cover and suppressed non-native invasive species, while scarification or scarification with organic matter amendment further improved early plant establishment. Collectively, these data help characterize the impacts of slash pile burning as a management technique in ponderosa pine forests and illustrate potential treatments for restoring burn pile scars.

INTRODUCTION

Background

Many forests in the northern Rocky Mountains are composed of pure and mixed stands of ponderosa pine (*Pinus ponderosa* Dougl. ex. Laws)/Douglas-fir (*Pseudostuga menziesii* (Mirb.) Franco). Currently, there is a drive to restore forest structure and function impacted by historic land management practices, namely fire suppression and intensive cattle grazing (Metlen & Fiedler 2006). Fire is a natural disturbance in many ecosystems that has an important influence on ecosystem dynamics and function. Historically, pure and mixed stands of ponderosa pine in the western United States were maintained by low-severity fires (Gruell 1983, Veblen & Lorenz 1986, Covington & Moore 1994, Arno et al. 1995, Swetnam & Baisan 1996, Allen et al. 1998, Brown et al. 1999, Mast et al. 1999, Moore et al. 1999, Kaufmann et al. 2000, Schoennagel et al. 2004). Frequent, low-severity fires likely generated fire tolerant, park-like stands of mature ponderosa pine with a grassy understory (Covington & Moore 1994, Covington et al. 1997, Fulé et al. 1997, Laughlin et al. 2004). This fire-maintained system was likely ephemeral since regional climate shifts alter fire regime every one or more centuries (Hessburg 2007). Fire exclusion following European settlement combined with historical logging has served to increase stand density (Mackenzie et al. 2004), increase living and dead biomass (Sala et al. 2005) and alter understory composition (Covington & Moore 1994*b*, Keane et al. 2002, Mackenzie et al. 2004), as litter accumulations lower species richness and herbaceous cover (Laughlin et al. 2004). Together, these factors result in

forests that are susceptible to high-intensity, stand replacing fires, which threaten homes in the wildland-urban interface and may be outside the range of natural variability (Feeney et al. 1998).

Increasingly, fire has been used as a management tool to increase biodiversity and promote ecosystem sustainability (Frost & Robertson 1985, Covington et al. 1997). Some of the benefits of burning include short-term increases in soil N mineralization (Raison 1979, DeBano et al. 1987, DeLuca & Zouhar 2000, DeLuca & Sala 2006), reduced fuel loading that could result in stand-replacing fires (Covington et al. 1997), and greater herbaceous productivity (Barney & Frishknecht 1974). Simultaneously, burning can have negative consequences such as alteration of soil structure leading to potential soil erosion (Neary et al. 2005), alteration and reduction of soil microbial communities (Dumontet et al. 1996, Neary et al. 1999, Neary et al. 2005, Jimenéz Esquilín et al. 2007), volatilization of nutrients and loss of C (Neary 2005), changes in the soil seed bank (Clark & Wilson 1994), and increased risk of non-native invasive species invasion (Allen 1991, Haskins & Gehring 2004).

Fuel reduction in ponderosa pine forests typically involves thinning and prescribed burning to restore the historical forest structure and function (Dahms & Geils 1997). Forest harvest operations generate large amounts of non-merchantable forest residues (or slash) that are often disposed of, particularly before fire can be safely reintroduced back into the ecosystem. Slash piling and burning is the most common method of slash disposal, because it is economical and, unlike broadcast burning, it can be performed under a broad range of weather conditions (Hardy et al. 1996). As fuel

reduction treatments become more common in the western United States, the potential problems associated with slash pile burning have been more closely evaluated (Korb et al. 2004, Jimenéz Esquilín et al. 2007). Slash pile burning is distinct from wild-fire because the large quantity of fuels results in high-intensity, long-duration fire. It is therefore important to determine whether the impacts of slash pile burning are contrary to the greater ecosystem goal of restoring the historical structure and function to ponderosa pine forests. While slash pile scars make up less than one percent of forest ecosystems (Korb 2004), several fire studies have reported important effects on soil physical and biochemical properties, such as increases in soil pH (Pietikäinen & Fritze 1995), which can affect plant nutrient uptake; increases in the sand fraction of the soil (Arocena & Opio 2003); loss of total C and N (Korb et al. 2004) and a temporary increase in NH_4^+-N (Choromanska & DeLuca 2002), followed by a pulse of NO_3^--N (DeLuca & Zouhar 2000). Soil P may not be impacted by slash-pile burning (Korb 2004), but this is dependent on fire intensity (Romanya et al. 1994). Soil microbial biomass is reduced by slash pile burning (Haskins & Gehring 2004; Theodorou & Bowen 1982; Choromanska & DeLuca 2001; Villar et al. 2004) as is microbial community composition (Pietikäinen et al. 2000; Hart et al. 2005). Slash pile burning may dramatically reduce arbuscular mycorrhizal (AM) fungal propagules (Korb et al. 2004), and another study indicated no effect on AM fungi after 5 years (Haskins & Gehring 2004). Finally, non-native plant species germination, establishment, or abundance may increase in slash pile scars (Dickinson & Kirkpatrick 1987; Haskins & Gehring 2004; Korb et al. 2004).

Restoration of slash pile scars is rarely documented or systematically performed by land managers. Scarification or tillage is shown to accelerate the recovery of compacted soils and improve seedling establishment (Bulmer 2000; Cole & Spildie 2007). Alternatively, composted biosolids may increase native plant establishment following wildfire (Meyer et al. 2004). In what appears to be the only study of its type to date, Korb (2004) performed a factorial slash pile restoration experiment involving seed and soil amendments. Natural revegetation was negligible after two years and the addition of live soil and seed most effectively increased native graminoid cover, however, by only a small margin.

Research Objectives

The influence of slash pile burning in ponderosa pine forests on soil physical and chemical properties, belowground microbial and fungal communities, and aboveground plant communities has not been thoroughly explored. Furthermore, there is little documentation regarding slash pile restoration techniques. The objectives of this study were: (1) to characterize the impact of slash pile burning on soil physical and biochemical properties; (2) to compare seeding and soil amendment effects on restoration of burn scars; (3) to quantify the growth potential and AMF colonization levels of native grass species and non-native invasive forbs in burned and unburned soil. This study is unique in that slash volume in each pile was small relative to most commercial logging operations.

The overarching goal of this research was to test whether burning of small fuel-reduction piles yields significant impacts on soil and plant processes. Objective 1, to characterize the impact of burning on soil properties is addressed in Chapter 2. One month post-burn, the following properties were measured in the center and edge of slash pile scars, and an adjacent control: soil physical properties including soil texture, bulk density, and water holding capacity; and soil biochemical properties including pH, soil organic matter (SOM), total N, NH_4^+-N, NO_3^--N, resin extractable inorganic N, potentially mineralizable N (PMN), phosphorous (PO_4^{3-}-P), and microbial biomass. Soil temperature under small slash piles was not expected to be intense enough to alter the measured soil physical properties. Soil biochemical properties, however, were likely to be influenced by slash pile burning. Immediately following burning, rapid N mineralization was expected to occur along with reduced nitrification rates. Following the recovery of nitrifying bacteria a subsequent peak of NO_3^--N in the second growing season was expected. In addition, total C and N and PMN was predicted to be reduced relative to control plots. Microbial biomass was predicted to be negatively correlated with burn intensity.

In the slash pile restoration study, the aim was to facilitate the recovery of soil nutrients, microbial activity, and native vegetation. The second objective, to compare seeding and soil amendments on burned scars is addressed in Chapter 3, with a restoration experiment performed on a subset of the sampled slash pile scars. Five treatments were applied to each slash pile including a no-treatment control, surface organic matter addition, scarification, surface organic matter addition with scarification,

and commercial compost addition with scarification. Half of these scars were seeded with native grasses and half received no seed. The following spring and summer, soil physical and biochemical analyses were repeated for each treatment. Revegetation was measured by percent cover estimates 10 and 22 months following restoration treatments. It was estimated that scarification, seeding and the addition of organic matter or compost would enhance native revegetation in burned areas.

The role of arbuscular mycorhhizal (AM) fungi for native and non-native species in soil exposed or not to pile burning is also discussed in Chapter 3. In a greenhouse experiment, native grasses and non-native forbs were grown for 10 weeks in unburned, burned, and burned soils with an ash addition. Mycorrhizal inoculum potential (MIP), a measure of the relative abundance of AM fungal propagules in these soils was assessed. In addition, plant biomass and root: shoot was measured. It was predicted that non-native forb biomass would be greater than native graminoids in soils exposed to pile burning. The addition of ash to soils may further hinder plant establishment by interfering with the rooting zone. AM fungal colonization was predicted be reduced in soils exposed to pile burning.

Soil processes, such as decomposition and nutrient mineralization, are fundamentally important to ecosystem function (Setälä et al. 2000) and both of these processes are directly influenced by the occurrence or exclusion of fire. Slash pile burning concentrates impacts over a limited area, while adjacent areas are relatively undisturbed. This project provided a unique opportunity to examine high-severity fire without dramatic landscape variability between disturbed and undisturbed sites.

Specifically, understanding the consequences of slash pile burning within the context of ponderosa pine forest restoration is needed to determine whether slash pile burning is in conflict with forest restoration goals.

EFFECTS OF SLASH PILE BURNING ON SOIL PHYSICAL AND BIOCHEMICAL PROPERTIES

Introduction

The effects of burning slash piles are distinct from ground-fire or even severe wildfire due to the tremendous quantity of fuels which result in extreme fire intensity and duration. In a ponderosa pine forest ecosystem, up to 25 Mg/ha of fuels may accumulate following 100 years of fire suppression (Hungerford et al. 1991). In contrast, a slash pile is estimated to load up to 950 Mg/ha of fuel on the forest floor (Korb et al. 2004). While a forest fire may heat the soil surface on the order of 200 to 300 °C, surface temperatures under a slash pile fire may range from 500 to 700 °C and periodically exceed 1500 °C (Neary et al. 1999). Soil is an excellent insulator and short duration pulses of heat will have minimal effect on the mineral soil (Van Wagner 1970). In contrast, a long-duration slash pile burn will conduct more energy and result in greater sub-surface heating, likely causing the most significant belowground damage (Certini 2005). As soil temperature rises, protein degradation and plant tissue death will begin between 40 to 70 °C. Roots desiccate around 48 to 54 °C, while the seed bank will be affected by temperatures ranging from 70 to 90 °C. The highest seed densities are found in the O horizon (Zabinski 2000), which is typically eliminated by slash burning. Mortality of soil microbes will occur between 50 to 121 °C, with bacteria being more resistant than fungi (Neary et al. 2005). At 220 °C soil organic matter may be partially oxidized, and at 460 °C complete combustion can occur (Giovannini et al. 1988). Nitrogen will volatilize when soil temperatures reach 220 °C (Giovannini et al. 1990).

While the consequences of wildfire are well-documented, there is relatively little quantitative data specific to slash pile burning. We do know that slash pile burning can alter soil physical properties through the removal of the surface duff layer and soil organic matter (SOM; Kennard & Gholz 2001). SOM is the measure of the mass of organic compounds in soil and is typically approximately 58% C (Brady & Weil 2002). While about 25 to 90% of the surface duff layer may be consumed in a natural forest fire (Neary et al. 1999; Neary et al. 2005), complete consumption of surface organics often occurs in severe fires (Czimczik et al. 2005), and under a slash pile. Surface organics include the O (humus horizon), which holds the majority of forest nutrient reserves, improves soil water holding capacity, and serves as a temperature buffer (Kennard & Gholz 2005). Surface organic matter contains important biota such as mycorrhizal fungi, and is the primary substrate for many roots systems.

Slash pile burning reduces soil organic matter most dramatically in the top 1.25 cm, with noted differences to depths of 30 cm, and only slight recovery after two (Austin & Baisinger 1955). With the loss of SOM, there is a reduction in available soil nutrients, microbes, reduced aggregation, and soil physical structure may be altered. These changes can result in soil compaction, crusting, and reduced drainage, aeration, and moisture evaporation (Austin & Baisinger 1955). Soil particle distribution may even be altered above 460 °C, at which point the sand fraction may increase when clay particles fuse (Giovannini et al. 1988). Changes in soil physical properties may inhibit plant growth and scars may become the foci of erosion initiation, particularly on steep slopes (Conacher & Sala 1998) where soil is less capable of absorbing water.

Ash deposition following burning may be greater for slash piles relative to wildfire. Ash is the result of the full combustion of organic matter and is comprised of carbonates and oxides of the alkaline earth metals, as well as silica, heavy metals, sesquioxides, phosphates, and small amounts of organic and inorganic N (Raison 1979). Composition will vary greatly depending on the organic matter origins and burn severity. When the ash is wetted, hydrolysis of cations occurs and forms an alkaline residue. This residue may have a pH exceeding 12.0 (Raison 1979). Ash provides a direct input of N and other cations through deposition, as well as indirect N inputs caused by the stimulation of plant N fixation with the rise of pH (Smithwick et al. 2005). Changes in soil pH may adversely affect plant establishment and nutrient uptake for some species, while providing a competitive advantage to others.

In northern ponderosa pine/Douglas-fir forests, N and P content of merchantable timber are 0.30 percent and 0.05 percent, respectively, while N and P content of non-merchantable timber are estimated to be 1 percent and 0.12 percent, respectively (Laiho & Prescott 2004). Therefore, forest harvest debris contains a greater proportion of forest nutrient capital and transporting all of the harvest debris off-site may effectively mine the forest of N and P. Nitrogen is the limiting nutrient in western forests and prescribed fire is known to impact plant available N (DeLuca & Zouhar 2000). The soil N pool may be directly impacted through N volatilization, pyrolysis of organic material, N translocation between mineral and organic soil layers, and lysis of microbial cell walls and fine root material (Smithwick et al. 2005). Following high-severity wildfire, N volatilization and pyrolysis rates are high and there is a rapid release of inorganic N in the form of NH_4^+-N

(Neary et al. 1999; Neary et al. 2005; DeLuca & Zouhar 2000; Choromanska & DeLuca 2001). This short term release of N may be 2 to 26 times background levels and often diminishes after only two growing seasons (Certini 2005; Smithwick et al. 2005). Thereafter, the increased concentration of NH_4^+-N, increased soil pH, decreased microbial immobilization, and recovering populations of nitrifying bacteria may result in a pulse of available NO_3^--N in the mineral soil (Smithwick et al. 2005). This increase is often two to five times greater than background concentrations NO_3^--N (Smithwick et al. 2005; Turner et al. 2007). Potentially mineralizable N (PMN) is a good measure of the N available to plants throughout the growing season. Soil PMN may increase following fire, then decrease below pre-fire conditions after one year (DeLuca & Zouhar 2000).

An abundance of charcoal in slash pile ash may also enhance nitrification rates when litter is rich in phenolic compounds (MacKenzie & DeLuca 2006). In the absence of fire, ericaceous shrubs become increasingly dominant in the ponderosa pine understory. Ericaceous shrubs produce phenolic compounds that are resistant to decomposition and may be allelopathic or toxic to nitrifying bacteria. Charcoal has a great capacity to adsorb organic compounds such as polyphenols (Berglund 2004; DeLuca 2006; Gundale & DeLuca 2007; MacKenzie & DeLuca 2006; Wardle et al. 1998; Zackrisson et al. 1996) and potentially adsorb up to 80% of the polyphenols from leaf litter leachate (MacKenzie & DeLuca 2006). The sorptive capacity of charcoal may be maintained for about one century after fire (Wardle et al. 1998) and may be reactivated following secondary heating (Pietikäinen et al. 2000). Charcoal is an indirect fire-induced mechanism that may maintain plant available N in ponderosa pine forests for up to 30 to

40 years (MacKenzie et al. 2006). Abundant charcoal or the absence of charcoal in a slash pile scar is one factor that will influence nitrification rates following burning.

Soil nutrient cycling and productivity are influenced by soil microorganisms. While microbial biomass may only represent one percent of soil organic matter, microorganisms contribute greatly to plant available nutrients through mineralization and nitrification (Villar et al. 2004; Pietikäinen & Fritze 1995) and enhance soil aggregation (Oades 1993). Fire may reduce soil microbial biomass through heating and moisture changes, particularly in the top few centimeters of soil (Haskins & Gehring 2004) and recovery may take as long as 13 years (Villar et al. 2004). Microbial mortality may be greatest in moist soils where thermal conductivity may heat soils up to 90 °C at depth (Smithwick et al. 2005). Conversely, microbial activity may be stimulated by fire and increase N immobilization. This may occur because of the increases in available C and P following fire (Smithwick et al. 2005). Fungal hyphae are particularly sensitive to higher surface temperatures and are more vulnerable than bacteria during severe fire (Dahlberg 2002).

Burn severity will also influence charcoal formation, which is important to soil microorganisms. While charcoal does not influence total microbial biomass, it may contribute to a shift from the fungal-based to the bacterial-based energy channel (Pietikäinen et al. 2000). The high adsorption capacity of charcoal increases bacterial biomass since organic compounds easily adsorb to the surface and may provide an accessible C source for bacteria (Wardle et al. 2008). In addition, the porous structure may provide protection for bacteria from soil faunal predators (Berglund 2004).

Slash pile burning may increase or have no effect on soil P. Soil P is volatilized above 360 °C and is transferred with ash particulates in smoke. Additional losses may occur when wind or rain erosion exports ash (Romanya et al. 1993). One study measured an increase in labile inorganic P and a decrease in total organic P with fire (Romanya et al. 1993), while another study concluded that total P is not greatly altered by slash pile burning (Korb 2004). Since the majority of soil P is in the mineral soil rather than the litter (Neary et al. 2005), burn intensity and proximity to the surface are important factors affecting P dynamics. In addition, P availability may be indirectly influenced by soil symbionts such as arbuscular mycorrhizal fungi. Even where fire has a neutral effect on total soil P, loss of AM fungal propagules could reduce plant P uptake by facultative AM plants.

The purpose of the work reported was to evaluate the influence of slash pile burning on nutrient capital and availability over the two years following burning. Specifically, the effect of slash pile burning on N, P, and C availability along with microbial biomass and soil physical properties was investigated. It was predicted that rapid N mineralization would occur along with reduced nitrification rates immediately following slash pile burning. Following the recovery of nitrifying bacteria there would be a subsequent peak of NO_3^--N in the second growing season. In addition, total C, N, and PMN would likely be reduced relative to control plots.

Methods

Site Description

The research site is on private land adjacent to the Bitterroot National Forest (BNF) near Florence, Montana at an elevation of 1,103 m with a mean temperature of 7 °C. The site receives an average of 33 to 40 cm annual precipitation and with an east-facing aspect and a slope ranging from a 2 to 25%. Soils have a mean pH of 5.2 and a C:N ratio of 24:1. Soils are part of the Haplocryepts and Eutrocryepts composed of stony sandy loam formed on granite residuum and composed of 30% sand and 37% clay. The O horizon was approximately 7 cm deep and was composed of 4 cm of Oi, 1 cm of Oe, and approximately 2 cm of Oa. The mineral horizon has an ochric epipedon with an average thickness of 14 cm, and the mean depth to decomposed granite parent material of 18 cm. Ponderosa pine and Douglas-fir dominate the canopy, and the habitat type is Douglas-fir – ninebark (*Physocarpus malvaceus* (Green) Kuntz). A fuels-reduction thinning occurred in March to April of 2006 and logging residues were tractor-piled and burned from late April to May of the same year. Scar diameter ranged from 2.75 m to 7.56 m and averaged 4.48 m. Post-thinning canopy cover averaged 58%.

Soil Analysis

In May 2006, soil thermocouples were buried at three locations at four different depths ranging from the surface ash to 20 cm deep. Slash was moved over the top of the buried thermocouples. Another set of thermocouples were buried at the edge of the three

slash piles. Slash was ignited and temperatures monitored until temperatures normalized or the data logger filled.

In June 2006, 45 of the 67 slash pile scars were randomly selected for analysis. Any scars with highly irregular boundaries were excluded. Soil was collected to evaluate post-fire effects from the center, edge, and adjacent unburned area (control) from each scar. The edge was identified along the periphery of the scar where litter was charred to the mineral soil, but not completely combusted into ash. Control plots were established in a random direction 6 m from the edge of the scar. Litter and ash were removed from the surface for controls and burned soil, and composite samples (ten sub-samples) were collected to a depth of 10 cm using a 2.5 cm diameter stainless steel soil probe. Sub-samples were composited and kept in a cooler and returned to the lab where they were refrigerated overnight and extracted the following day.

Fresh soil samples were sieved to 2 mm, homogenized, and analyzed for extractable inorganic N, potentially mineralizable N (PMN), water soluble P (PO_4^{3-}-P), and microbial biomass (*n*=138). To determine extractable inorganic N content, an oven-dry equivalent of 25 g of soil was added to 50 ml of 2 M KCl and shaken for 30 minutes. The solution was then filtered with Whatman # 2 filter paper and the extract analyzed for NO_3^--N and NH_4^+-N (Mulvaney 1996) by using an Autoanalyzer III (Bran Luebbe, Chicago, IL) one month post-burn and a Lachat flow injection autoanalyzer (Milwaukee, WI) 13 months post-burn. Soluble NO_3^--N was determined using the cadmium reduction method (Mulvaney 1996) and soluble NH_4^+-N was determined using the salicilate-nitroprusside method. To determine water soluble P, 10g of soil was placed in 20 ml of

0.01 M $CaCl_2$ and shaken for 30 minutes. The solution was then filtered through Whatman # 42 filter paper and the extract analyzed for soluble PO_4^{-3}-P using the molybdate-ascorbic acid method on Autoanalyzer III (Bran Luebbe, Chicago, IL) one-month post-burn and by hand 13 months post-burn (Milwaukee, WI) 13 months post-burn. Potentially mineralizable N (PMN) was measured using the 14-day anaerobic incubation procedure (Bundy & Meisinger 1996), where a 5 g soil sample was placed into 12.5 ml of distilled water, and a stream of N_2 gas was bubbled through the suspension for 10 seconds. The sample was then placed in a constant temperature chamber at 25 °C for 14 days, after which 12.5 ml of 4 M KCl was added to the aqueous suspension to create a 2 M KCl extractant. This suspension was shaken for 30 minutes, filtered through Whatman #2 filter paper, and the extract analyzed for NH_4^+-N as described above. To calculate PMN, the concentration of NH_4^+-N in the un-incubated soil samples (time zero sample) was subtracted from the NH_4^+-N concentration at 14 days. To assess variation from the center to edge, *in situ* N mineralization was tested using ionic resin capsules (Unibest, Bozeman, Montana). Polyester capsules, 2 cm in diameter, containing approximately 1 g of mixed bed ionic resin, were carefully placed 5 cm below the soil surface using a stainless steel soil probe in the center (n=23) and the edge (n=12) of slash pile scars. Capsules were buried in October of 2006 and retrieved in June of 2007. Resins were extracted by sequential washing in 2 M KCl and analyzed for NH_4^+-N by the Berthelot method and NO_3^--N by the cadmium reduction method via flow injection analyzer (Mulvaney 1996). Microbial biomass was determined with the fumigation extraction, ninhydrin-reactive N method (Jorgenson & Brooks 1990), as

modified by DeLuca & Keeney (1993). A 25 g soil sample was placed in a 200 ml French Square bottle and exposed to chloroform for 24 hours. The fumigated samples as well as an un-fumigated control sample were extracted with 50 ml of 2 M KCl as described above and both samples analyzed for ninhydrin reactive N (Jorgenson & Brooks, 1990). Biomass N was then calculated as the difference between ninhydrin reactive N in fumigated samples relative to the control, multiplied by a factor of 3.1 (Joergensen & Brooks 1990).

Mineral soil samples were analyzed for moisture content, pH, and bulk density (*n*=138). Soils were also analyzed for total C and N (*n*=138), particle size (*n*=15), and water holding capacity (*n*=45). Soil moisture content was determined gravimetrically by drying a 30 g moist sub-sample at 105 °C for 24 hours and then reweighing the sample. Dried mineral soils were analyzed for pH in a 2:1, 0.01 M $CaC1_2$ soil suspension. Bulk density was determined with a 5 cm diameter by 5 cm long brass sleeve that was inserted into the soil using a slide hammer coring device. The core was removed, returned to lab, dried at 60 °C, analyzed for total mass and density by dividing mass by volume. Total soil C and N was analyzed by dry combustion (Leco Truspec Instrument). Particle size distribution was determined by hydrometer (Gee & Bauder 1986). Water holding capacity (WHC) was analyzed volumetrically by saturating 50 g of soil in a funnel for 30 min. and filtering through glass wool. Water was measured after 30 min. of draining. Soil WHC was calculated as the volume of water retained in the soil minus the water retained in the wool plus percent soil moisture content.

Data Analysis

Post-fire soil effects for differences between the burned center, the burned edge, and unburned soils (control) were tested with ANOVA (analysis of variance; SPSS General Linear Model Univariate ANOVA, SPSS, Inc., Chicago, IL.Version 15.0). Soil treatment was a fixed effect. The effects of fire one year post burn were assessed using one-way and two-way ANOVA. With the two-way ANOVA model, soil treatment and year were fixed effects. The Least Squares Difference (LSD) post-hoc test was used for all ANOVAs to determine which treatments differed significantly (α=0.05). All data were evaluated for conformations to the assumptions of parametric statistical analysis. Homogeneity of variance was tested using Levene's test (>0.05) and normal distribution of the dependent variable was verified with a normal curve and by observing skewness and kurtosis data. Those data not meeting these assumptions were transformed when possible. The Games-Howell post-hoc test was used when data did not meet the assumption of equal variance. The Kruskal-Wallis non-parametric technique was used when the assumptions of equal variance and normal distribution were violated. Games-Howell is a post-hoc test used to determine which treatment differed significantly in two-way ANOVA. In contrast, Kruskal-Wallis only identifies a treatment effect. As an added caution, when the Games-Howell post-hoc test was used, the results were verified using Kruskal-Wallis. The following analyses were completed for data collected one month post-burn; soil organic matter (square root transformed), total N (log transformed), NH_4^+-N (square root transformed), NO_3^--N (square root transformed), PMN (analyzed non-

parametrically using the Kruskal-Wallis test), PO_4^{3-}-P, and microbial biomass (square root transformed). For data collected one-year post burn, total C and total N were log transformed.

Results

Soil Burn Characterization, One Month Post-burn

Thermocouples were used to monitor fire intensity and duration in the center and edge of three slash piles. Controls were placed at the surface adjacent to the burns. Maximum temperatures in the burned center were 155 °C at 3 cm, 96 °C at 6 cm, and 247 °C in the ash surface. The burned edge maximum ranged from 30 °C at 4 cm, 60 °C at 4cm, and 117 °C at 2.4 cm (Figure 1). Due to rocky terrain, temperature probes could not be placed at consistent depths. These data approximate soil heating and illustrate the heterogeneity of intensity and duration.

All soils were high in clay content and classified as clay loam according to the ISSS definition. Soil texture did not vary with burn position (Table 1).

Table 1. Soil particle size analysis and texture by soil treatments one month post-burn in slash piles in Western Montana averaging 4.5 m in diameter.

Burn Position	Soil Texture	Sand (%)	Silt (%)	Clay (%)
Center	Clay Loam	30	32	38
Edge	Clay Loam	35	25	40
Control	Clay Loam	30	33	37

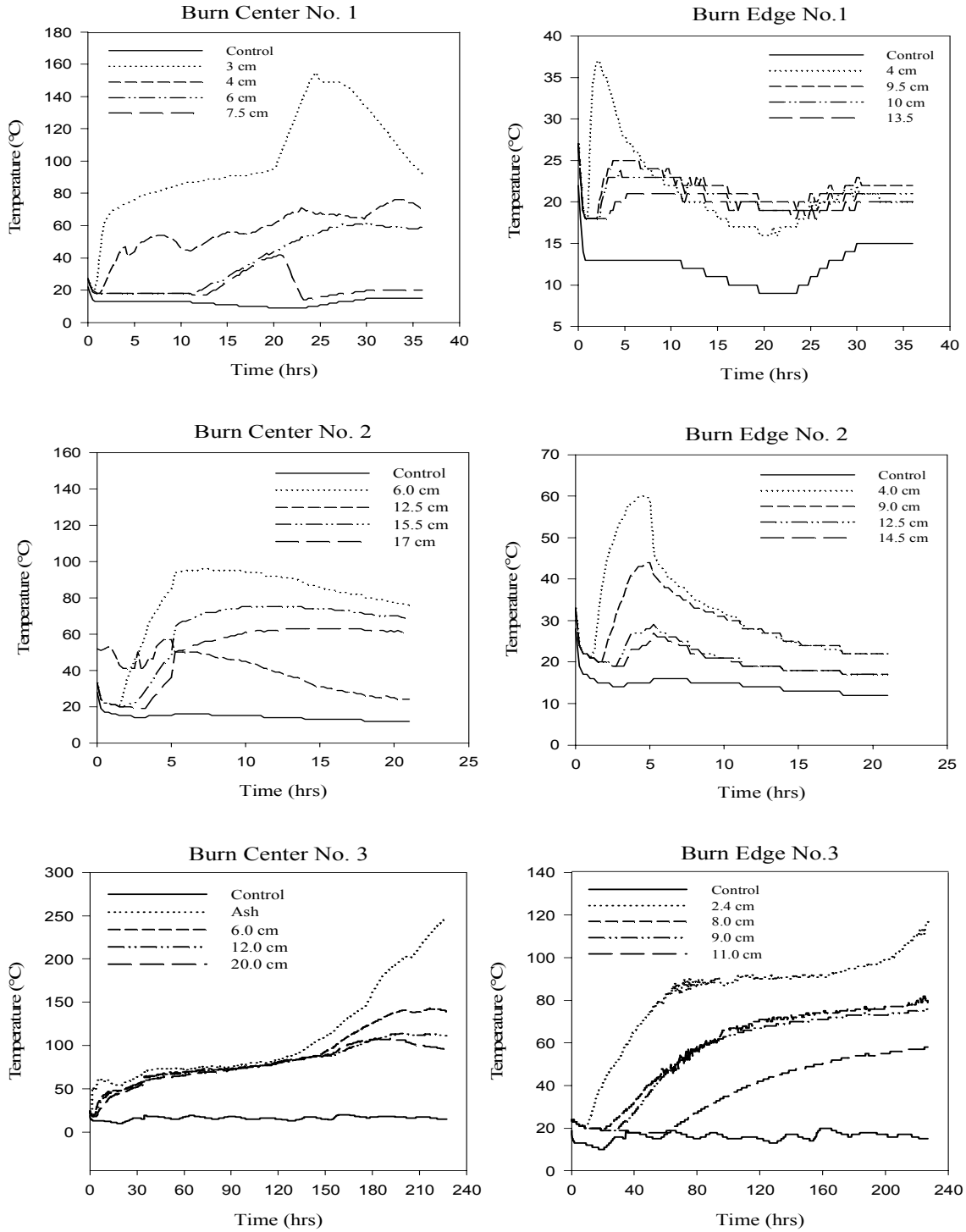

Figure 1. Soil temperature data for the burned center and edge at four depths taken from three slash pile burns in a ponderosa pine forest ecosystem in Western Montana. Due to rocky terrain, thermocouples could not be placed at consistent depths. Note differences on x- and y-axis. Control is located at the surface outside of burn pile.

Soil pH was 5.9 in the burned center, 5.7 in the burned edge, and 5.2 in the control (Table 2). A one-way ANOVA shows pH was different among positions, (p<0.01; Table 3) and the LSD post-hoc test suggests differences between the center, edge, and control.

Table 2. Fire effects on soil parameters in the slash pile scar center, edge, and control one month following burn in slash piles in Western Montana. Values indicate mean ± 1 S.E. of the mean and letters indicate significant differences between values in a column.

Burn Position	pH	Bulk Density (g/cm^3)	Water Holding Capacity (%)
Center	5.9 ± 0.1 a	0.9 ± 0.0	32.4 ± 0.0 a
Edge	5.7 ± 0.1 b	0.8 ± 0.0	46.9 ± 0.0 c
Control	5.2 ± 0.1 c	0.8 ± 0.0	39.3 ± 0.0 b

One month following slash pile burning, mean soil bulk density was 0.9 g/cm^3 in the burned center, 0.8 g/cm^3 in the burned edge, and 0.8 g/cm^3 in the control (Table 2), and did not differ between treatments (Table 4). WHC ranged from 32.2 to 47.2%. Soil WHC differed greatly by position (p<0.01; Table 4). WHC of soils in the burned center of slash piles was lower (32.4%) than the control (39.3%), but soils at the edge of slash piles had the highest WHC (46.9%; Table 2).

One month post-burn, total C differed by position (p=0.01; Table 4), and was lower in the burned center and edge of slash piles burns compared to the control plot, averaging 5.2 $\mu g\ g^{-1}$, 5.8 $\mu g\ g^{-1}$, and 7.1 $\mu g\ g^{-1}$, respectively (Table 3, Figure 2a). In contrast, total N did not change with position, (p=0.09; Table 4), with a mean of 0.3 $\mu g\ g^{-1}$ in the burned center, 0.3 $\mu g\ g^{-1}$ in the burned edge, and 0.3 $\mu g\ g^{-1}$ in the control (Table 3, Figure 2c).

Table 3. Fire effects on soil biochemical properties in the burned center, edge, and control one month post-burn in slash piles in Western Montana. Values indicate mean ± 1 S.E. of the mean and letters indicate significant differences between values in a column.

	Total C (%)	Total N (%)	NH_4^+-N (μg g^{-1})	NO_3^--N (μg g^{-1})	PMN (μg g^{-1})	PO_4^{3-}-P (μg g^{-1})	Microbial biomass (μg g^{-1})
Center	5.16 ± 0.2 a	0.25 ± <0.01	82.1 ± 4.6 a	1.5 ± 0.2	51.7 ± 2.8 a	3.8 ± 0.6	16.0 ± 2.3 a
Edge	5.77 ± 0.5 a	0.25 ± <0.01	32.6 ± 4.9 b	1.6 ± 0.3	66.2 ± 4.1 b	3.8 ± 1.0	31.0 ± 2.7 b
Control	7.09 ± 0.7 b	0.29 ± <0.01	14.0 ± 3.9 c	2.1 ± 0.3	48.0 ± 7.5 a	4.1 ± 1.1	59.4 ± 14.7 c

Plant available N in the form of NH_4^+-N differed with burning (p<0.01; Tables 3 & 4) and was highly variable ranging from 2.01 to 196.88 μg g^{-1}. NH_4^+-N concentrations were elevated in the soils from the center of the slash pile scars (82.08 μg g^{-1}), intermediate in the soils from the edge of burns (32.57 μg g^{-1}) and lowest in the control soils (13.99 μg g^{-1}) (Figure 2b). Soil NO_3^--N, in contrast, did not change with burning (p=0.07; Table 4, Figure 2b). Values were 1.51 μg g^{-1} in the center of slash pile burns, 1.55 μg g^{-1} in the edge of burns, and 2.14 μg g^{-1} in the control (Table 3). Potentially mineralizable N (PMN) was highly variable, ranging from 1.02 to 155.38 μg g^{-1}. Mean PMN in soils from the center of burns was 51.7 μg g^{-1}, 66.2 μg g^{-1} in soils from the burned edge, and 48.0 μg g^{-1} in the control (Table 3). Slash pile burning significantly increased soil PMN (p<0.01; K-W Test; Table 4) and this difference was only between the burned center and edge (p<0.01; Games-Howell post-hoc).

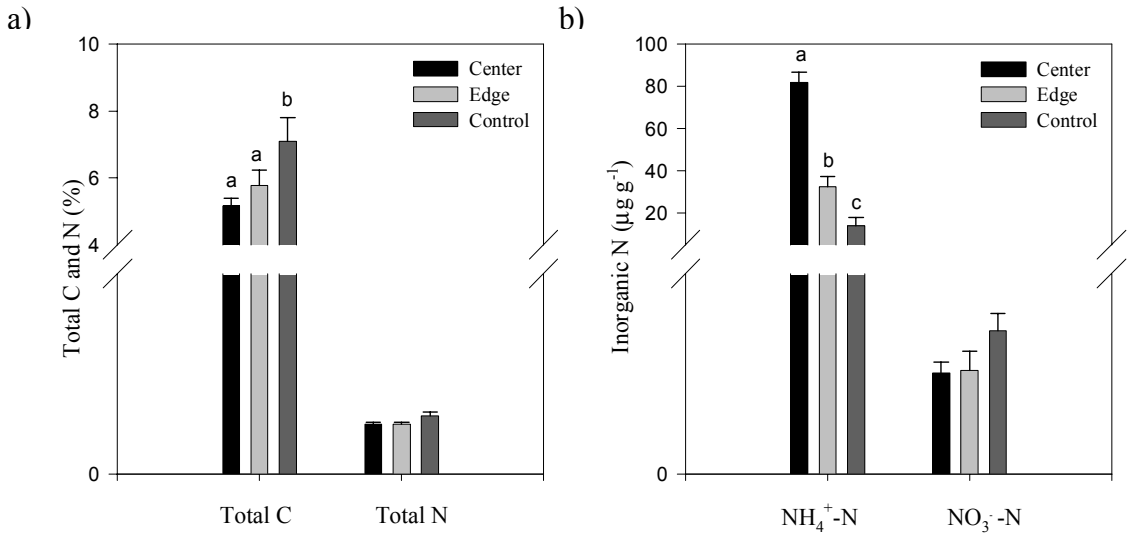

Figure 2. Response of soil C and N (a) and inorganic N (b) to fire in the burned center, edge, and control one-month after slash pile burning in Western Montana. Letters indicate significant differences between treatments (α<0.05), and error bars signify 1 S.E.

Water soluble phosphate (PO_4^{3-}-P) was not affected by burning (Table 4). Mean PO_4^{3-}-P was 4.1 µg g^{-1} in soils from the center, 3.8 µg g^{-1} in soils from the edge, and 3.8 µg g^{-1} in control soils (Table 3).

Microbial biomass, measured as fumigation extraction, differed with burning (p<0.01; Table 3 & 4). Values were highly variable ranging from 0.00 to 74.1 µg g^{-1} across all burn treatments. All positions differed with the lowest microbial biomass in soils from the center of slash pile burns (16.0 µg g^{-1}), followed by soils from the edge of burns (31.0 µg g^{-1}), and control soils (59.4 µg g^{-1}; Table 3).

Table 4. Analysis of variance burn position results including degrees of freedom, mean squares, and F and *p*-values for pH, bulk density, WHC, total C (square root transformed), total N (log transformed), NH_4^+-N(square root transformed), NO_3^--N (square root transformed), PMN (K-W Test) , PO_4^{3-}-P, and microbial biomass (square root transformed) for slash pile burned soils one month post-burn in Western Montana.

	pH				Bulk Density (g/cm^3)			
	d.f.	MS	F	P	d.f.	MS	F	P
Burn	2	5.18	31.11	<0.01	2	0.02	0.55	0.58
Error	132	0.17			117	0.03		
	WHC (%)				Total C (%)			
	d.f.	MS	F	P	d.f.	MS	F	P
Burn	2	2365.11	70,391.90	<0.01	2	0.72	4.86	0.01
Error	132	0.034			132	0.15		
	Total N (%)				NH_4^+-N (µg g^{-1})			
	d.f.	MS	F	P	d.f.	MS	F	P
Burn	2	0.30	2.45	0.09	2	367.45	88.39	<0.01
Error	132	0.12			132	4.16		
	NO_3^--N (µg g^{-1})				PMN (µg g^{-1})			
	d.f.	MS	F	P	d.f.	Chi-Sq	Asymp.Sig	
Burn	2	0.57	2.69	0.07	2	11.77	<0.01	
Error	132	0.21						
	PO_4^{3-}-P (µg g^{-1})				Microbial Biomass (µg g^{-1})			
	d.f.	MS	F	P	d.f.	MS	F	P
Burn	2	1.05	0.02	0.97	2	1,040,071.21	44.07	<0.01
Error	132	39.13			131	23,602.99		

Soil Burn Characterization, One Year Post-burn

Soil data analyses were repeated in 2007 (*n*=46) and did not include the burned edge. Mean soil bulk density in the burned center remained constant at 0.86 g/cm^3 (Table 6). Soil pH in the center of slash pile burns increased from 5.9 to 7.4 in 2007 ($p<0.01$; Table 5).

One year following slash pile burning, total C remained constant in the center of burns, averaging 5.1 µg g^{-1} compared to 5.2 µg g^{-1} one month after the burn (Table 5). Soil total C differed between burned and unburned soils ($p<0.01$), but there

Table 5. Summary of mean and ± 1 S.E. for soil properties in the burned center and control one-month and one-year post-burn in a small-scale slash pile burn study in Western Montana. Significance is indicated by letters in rows for two-way ANOVA and a (*) for one-way ANOVA.

	2006		2007	
	Burn	Control	Burn	Control
pH	5.9 ± 0.1 b	5.2 ± 0.1 a	7.4 ± 0.1 c	5.1 ± 0.1 a
Total C (%)	5.16 ± 0.2 a	7.09 ± 0.7 b	5.14 ± 0.3 a	6.06 ± 0.3 ab
Total N (%)	0.25 ± <0.01	0.29 ± <0.01	0.25 ± <0.01	0.30 ± <0.01
PMN (μg g^{-1})	59.10 ± 3.1*		7.45 ± 1.6*	
PO_4^{3-}-P (μg g^{-1})	4.05 ± 0.6	3.78 ± 1.1	3.44 ± 0.5	1.94 ± 0.3

was no change over time (two-way ANOVA; Table 7). Total N did not change from 2006 to 2007 (Tables 5 & 7). The C:N in burned soils was reduced from 24 to 17 in 2006.

The spike in NH_4^+-N in burned soils declined from 82.08 μg g^{-1} in 2006 to 4.79 μg g^{-1} in 2007, but remained elevated compared to the control (0.01 μg g^{-1}; Figure 3a). There was a significant burn x year interaction on soil NH_4^+-N (p<0.01; Table 7). In contrast, soil NO_3^--N in the burned soils increased from 1.51 μg g^{-1} one-month after the burn to 9.27 μg g^{-1} the following year (Figure 3b). There was a burn position and time effect (Kruskal-Wallis, p<0.01) for soil NO_3^--N (Table 8). This difference was found in the center of slash pile burns between 2006 and 2007 and between the center and control in 2007 (Games-Howell post-hoc, p<0.01; Table 7).

Resin-sorbed inorganic N was collected over the winter of 2006 through the summer of 2007 using ionic resin capsules to test the spatial heterogeneity within the

slash piles from the center to the edge. NH_4^+-N in soils from the center of the burns ranged from 0.01 to 5.57 µg g^{-1} and 0.05 to 1.92 µg g^{-1} in soils from the edge of the burns. Extractable NO_3^--N in the center of burns ranged from 0.02 to 3.22 µg g^{-1} and 0.04

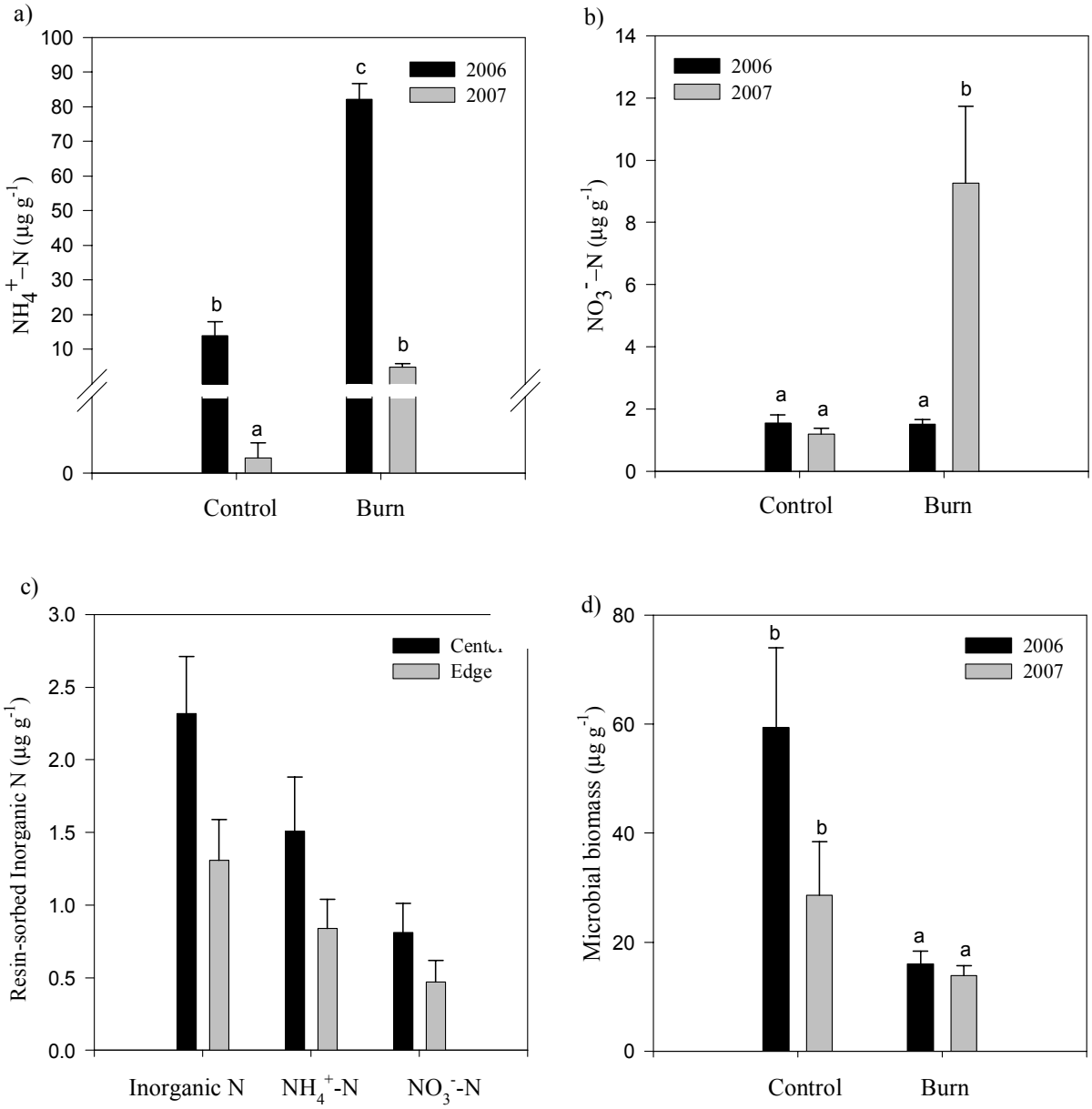

Figure 3. Comparison of 2006 and 2007 NH_4^+-N (a), NO_3^--N (b), resin-sorbed inorganic N (c), and microbial biomass (d) following small-scale slash pile burning in Western Montana. Letters indicate differences between burn location and year. Error bars signify 1 S.E.

to 1.5 µg g^{-1} in the edge of burns. There were no differences between the center and edge of the burns for either NH_4^+-N and NO_3^--N (Figure 3c). When inorganic N values are combined, there is still no difference from the center and edge of slash pile scars (p=0.10).

Soil PMN varied widely from 2006 to 2007, and was significantly higher one month post-burn than the following year (Table 6), ranging from 15.67 to 101.06 µg g^{-1} in 2006 and only 0.24 to 26.13 µg g^{-1} one year later. Mean PMN was 59.10 µg g^{-1} in 2006 and decreased to 7.45 µg g^{-1} in 2007 (Table 5).

Table 6. ANOVA by year for pH, bulk density, and PMN (square root transformed) for the burned center in 2006 to 2007 and ANOVA for resin-sorbed inorganic N for the burned center and edge of small-scale slash pile burns in Western Montana.

	pH				Bulk Density			
	d.f.	MS	F	P	d.f.	MS	F	P
Year	1	3.79	26.27	<0.01	1	6.89E-006	0.00	0.99
Error	66	0.14			61	0.03		
	Resin-sorbed NH_4^+-N (µg g^{-1})				Resin-sorbed NO_3^--N (µg g^{-1})			
	d.f.	MS	F	P	d.f.	MS	F	P
Year	1	2.97	1.47	0.24	1	0.78	1.32	0.26
Error	28	2.01			28	0.59		
	PMN							
	d.f.	MS	F	P				
Year	1	299.96	180.96	<0.01				
Error	53	1.67						

Water soluble PO_4^{3-}-P concentrations did not change from 2006 to 2007 and no difference was observed between the center of the burn and control for both years (Table 7). In the center of the burn, mean PO_4^{3-}-P was 4.05 µg g^{-1} in 2006 and 3.44 µg g^{-1} in 2007. Mean PO_4^{3-}-P for the control was 3.78 µg g^{-1} in 2006 and 1.94 µg g^{-1} in 2007 (Table 5).

In 2007, microbial biomass was again significantly lower in soils from the center of burn piles (13.86 µg g^{-1}) relative to control soils (28.55 µg g^{-1}; Figure 2d). A treatment effect was found using the non-parametric Kruskal-Wallis test ($p<0.01$; Table 8). The Games-Howell post-hoc test indicates that in 2006 and 2007, the center of the burn and control soils were different ($p<0.01$), but microbial biomass in the center of the burn did not change from 2006 to 2007 ($p=0.89$; Table 7; Figure 3d).

Table 7. ANOVA for SOM (log transformed), total N (log transformed), NH_4^+-N, NO_3^--N, PO_4^{3-}-P, and microbial biomass for burned and the control in 2006 and 2007 for small-scale slash pile burns in Western Montana.

	Total C (%)				Total N (%)			
	d.f.	MS	F	P	d.f.	MS	F	P
Burn (B)	1	1.36	11.11	<0.01	1	0.73	6.77	0.10
Year (Y)	1	0.03	0.24	0.63	1	0.07	0.66	0.42
Y * B	1	0.04	0.29	0.59	1	0.01	0.07	0.79
Error	132	0.12			132	222.76		
	NH_4^+-N (µg g^{-1})				NO_3^--N (µg g^{-1})			
	d.f.	MS	F	P	d.f.	MS	F	P
Burn (B)	1	40418.12	73.38	<0.01	1	491.14	19.79	<0.01
Year (Y)	1	63381.50	115.07	<0.01	1	418.47	16.86	<0.01
Y * B	1	30502.58	55.38	<0.01	1	500.10	20.15	<0.01
Error	132	550.82			132	24.82		
	PO_4^{3-}-P (µg g^{-1})				Microbial biomass (µg g^{-1})			
	d.f.	MS	F	P	d.f.	MS	F	P
Burn (B)	1	23.52	0.96	0.33	1	25672.54	7.70	<0.01
Year (Y)	1	45.27	1.85	0.18	1	8267.40	2.48	0.12
Y * B	1	11.25	0.46	0.50	1	6268.62	1.88	0.17
Error	131	24.51			132	3332.48		

Table 8. Kruskal-Wallis non-parametric test for NH_4^+-N, NO_3^--N, and Microbial biomass for burned and control soils in replicated small-scale slash pile burns in Western Montana.

	NH_4^+-N ($\mu g\ g^{-1}$)			NO_3^--N ($\mu g\ g^{-1}$)		
	d.f.	Chi-square	Asym.Sig	d.f.	Chi-square	Asym.Sig
Burn	1	41.12	<0.01	1	9.82	<0.01
Year	1	68.67	<0.01	1	9.30	<0.01
	Microbial biomass ($\mu g\ g^{-1}$)					
	d.f.	Chi-square	Asym.Sig			
Burn	1	49.97	<0.01			
Year	1	1.37	0.24			

Discussion

The effects of long duration, high intensity slash pile burning may diverge from those of forest fire due to the following: 1) slash pile burning consumes most of the forest litter and surface soil organic carbon, while low-severity fire may only cause minor distillation or charring; 2) the duration of slash pile burning causes sub-surface soil temperatures to exceed those of typical forest fire; and 3) a thick layer of ash is deposited on the soil surface, whereas ash is more evenly distributed during a forest fire. In this study, these effects influenced pH, WHC, total C, N dynamics, and microbial biomass.

Total soil C and water holding capacity both decreased with burn intensity and did not recover after 13 months. The high volume of ash may have contributed to the reduction of soil water holding capacity, as ash may seal or clog soil pores (Certini 2005). More importantly, loss of total C can directly reduce water holding capacity, soil fertility and productivity. Soil C helps create space for moisture infiltration and root growth. Soil C serves as a food source for microbes, which may partially explain why microbial biomass did not recover 13 months post-burn.

A reduction in total N and PMN 13 month post-burn was expected, since total ecosystem N generally declines with severe fire (Smithwick et al. 2005). Total N did not, however, differ between burned and unburned soils. Since SOM does not partially oxidize until reaching 220 °C (Giovannini et al. 1988) and soil temperature data from this study indicates 160 °C was the maximum temperature reached at a depth of 3 cm, it is likely temperatures stayed below the critical threshold for total N volatilization. Mineral soil PMN, in contrast, was highest at the edge of the slash pile burn. This suggests that the organic N fraction of the soil was mostly consumed in the center of the slash pile where fire severity was highest**.** In contrast, the edge was moderately heated, which distilled inorganic N that would otherwise have remained stable as amine groups such as proteins or humic compounds (Neary et al. 2005; Certini 2005). As expected, soil C and N accumulation did not increase after 13 months in the burned center.

Other soil physical properties were similar across burn treatments with respect to bulk density and particle size distribution. This indicates that soil structure beneath the slash pile was not greatly altered following burning.

In this study, several soil biochemical properties responded to slash pile burning. Soil pH increased at the center and edge of the slash pile with the addition of ash. Often the increase in soil pH is eliminated after one year because burning denatures organic acids and subsequently releases K and Na oxides, hydroxides, and carbonates, which are not stable through wetting and drying cycles (Certini 2005). After 13 months, however, soil pH increased in the burned center, suggesting that ash material may have more thoroughly mixed within the soil sub-surface and alkalinity may have been maintained by

Ca^+, which may take two or more years to normalize (Certini 2005). Low soil pH can slow plant growth by influencing factors such as plant uptake of P and micronutrients (Kennard & Gholz 2001). An increase in soil pH can shift belowground biota from the fungal energy channel to the bacterial energy channel (Wardle 2002), which could influence decomposition and indirectly influence nutrient cycling.

Slash pile burning caused a net release of NH_4^+-N, measurable one-month after the fire event. Since soil temperatures exceeded 121 °C in the upper 4 cm of soil, bacteria was likely reduced. Mortality of nitrifying bacteria slowed nitrification and no additional NO_3^--N was recorded. Microbial biomass results support this conclusion with biomass concentration negatively correlated with burn intensity.

The increase in soil pH and NH_4^+-N ions may have had a balancing effect from the perspective of plant nutrient uptake in the rhizosphere. Positively charged NH_4^+-N ions are exchanged at the root surface with hydrogen ions, thereby lowering the pH (Brady &Weil 2002). Any pH reduction caused by NH_4^+-N ions may have been neutralized by the addition of ash, which typically increases pH. This interaction could influence the uptake of other ions, such as P.

Thirteen months post-burn, NH_4^+-N decreased dramatically from its post-fire peak in both soils from the center of slash pile burns and control soils. The relatively high background NH_4^+-N concentration one month post-burn and the subsequent reduction 13 months post-burn may suggest N deposition occurred in unburned areas following slash pile burning via ash or NH_3. The higher level of microbial biomass one month post-burn relative to 13 months post-burn in control soils suggests nitrifying bacteria responded to

N release and deposition. The decrease in NH_4^+-N in the burned center indicates the bank of potentially mineralizable N was depleted and the NH_4^+-N was either converted to NO_3^--N following the recovery of nitrifying bacteria, was adsorbed onto negatively charged surfaces of minerals and organics in the soil (Certini 2005), or taken up by plants. Charcoal deposition may have also influenced nitrification by stimulating nitrifying bacteria. Charcoal will sorb inhibitory compounds, such as phenols and terpenes that can have an allelopathic effect on nitrifying bacteria (Berglund 2004; DeLuca 2006; Gundale & DeLuca 2007; MacKenzie & DeLuca 2006; Wardle et al. 1998; Zackrisson et al. 1996). Total microbial biomass did not increase in the burned center from 2006 to 2007. The concentration of NO_3^--N 13 months post-burn was not proportional to the flush of NH_4^+-N occurring immediately following the burn, indicating that microbes could have been substrate limited. Losses may also be due to leaching of NO_3^--N from the system or due to plant uptake of inorganic N where plots were seeded or natural plant establishment occurred. This will be further discussed in Chapter Three.

Resin-sorbed inorganic N was measured *in situ* in the center and edge of slash piles from fall 2006 to spring 2007. The resins suggest no difference between the center and edge for either NH_4^+-N or NO_3^--N. Resins were likely placed in the soil after the fire induced peak of NH_4^+-N; however, the NO_3^--N peak should have been captured if it occurred. When the two N fractions are combined into inorganic N, there is still no difference between the center and edge. This also may suggest that nitrifying bacteria did not fully recover in the center or edge of slash pile burns.

Soil pH increased dramatically from 2006 to 2007 in the burned center in spite of the increase in NO_3^--N that should have increased the concentration of H^+ ions. This is likely because the thick ash layer, which is unique to slash pile burns, mixed into the subsurface soil after one year and increased pH throughout the core sample.

Soil PO_4^{3-}-P did not change with burn location. Water-soluble PO_4^{3-}-P only provides information on the mobile P fraction, which is relatively fleeting. Any increase in PO_4^{3-}-P may not have been detected one month post-burn. Labile P may have increased, but rain and wind could have exported ash and surface soil off-site. This is supported since PO_4^{3-}-P in the control was greater in 2006 than 2007, indicating P deposition by ash transport may have occurred immediately after fire. Since PO_4^{3-}-P is more soluble in somewhat alkaline conditions, it is possible the concentration of ions were too low to detect differences.

Slash pile burning may be a greater detriment to soil properties than a typical forest fire. In this study, slash pile burning depleted total C. The large quantity of ash had a lasting effect on pH and soil water holding capacity. Soil N dynamics confirmed some prior expectations in that soil heating resulted in a temporal flush of NH_4^+-N and a subsequent increase in NO_3^--N (Smithwick et al. 2005). Finally, fire severity was great enough to reduce microbial biomass even after 13 months. Alteration of soil properties such as pH, WHC, and microbial biomass may influence plant growth. Following fire, pH changes may influence P-sorption reactions as well as the production of organic acids (Blank et al. 1994). Reduced WHC may favor more drought-tolerant species. Changes in the volume and composition of microbes may slow nutrient cycling. On the other hand,

these factors may be offset by the fire-induced increase in N availability. Understanding these dynamics helps evaluate the impact of slash pile burning on site productivity and may advise revegetation efforts.

SLASH PILE RESTORATION EFFECTS ON SOIL BIOCHEMICAL PROPERTIES AND VEGETATIVE COVER

Introduction

Slash pile burning is the most common method of handling forest harvest residues across the west. On an annual basis 25 million tons of logging debris is produced in the U.S., however, proper management of these residuals is still debated. Approximately 15 to 50 percent of forest biomass is left behind following harvest. The quantity of harvest residues will continue to increase over the next decade with fuels reduction as an explicit goal of the U.S. Forest Service and reinforced as a priority with the Healthy Forest Initiative and Healthy Forests Restoration Act of 2003. Since 2003, Montana alone has treated 200,750 ha of federal land for fuels reduction and nationally more than eight million ha of federal land is eligible for fuels reduction treatment (www.forestsandrangelands.gov). Many of these fuels reduction operations are performed in low elevation ponderosa pine (*Pinus ponderosa* Dougl. ex. Laws) and Douglas-fir (*Pseudostuga menziesii* (Mirb.) Franco) forests, and slash piling and burning is frequently used to dispose of non-merchantable forest debris. While the scar left from pile burning makes up only a small fraction of the forest ecosystem, these sites may become a source of soil erosion (Neary et al. 2005; Austin & Baisinger 1995), cause a shift in plant community dynamics, and possibly favor non-native invasive species (Dickinson & Kirkpatrick 1987; Haskins & Gehring 2004; Korb et al. 2004).

Quantitative data on slash pile burn restoration are limited (Korb et al. 2004) and mid- to long-term monitoring of restoration work is poorly documented. Given the strong

directive to perform fuel reduction thinnings on public lands and slash pile burning as one of the primary associated disturbances, there is need for a more complete assessment of ecosystems impacts and to understand the potential for non-native invasive colonization of burned soils resulting from common slash pile restoration techniques.

To examine the effects of various slash pile restoration techniques, soil characteristics important to plant growth and overall ecosystem sustainability were measured, including soil physical properties, the N pool, P, microbial biomass, and mycorrhizal inoculum potential (MIP). This research was conducted to determine how seeding and soil amendments alter the cover and composition of plant groups, as well as to identify the post-disturbance role of mycorrhizae in several early successional species. Small-scale slash pile burn scars in a forest recently treated for fuels reduction were selected for restoration work. Successful amelioration of the scars may be identified through the relationships between soil physical and biochemical factors, as well as vegetative cover.

Soil Properties

While there is a great deal of data documenting the impacts of wildfire on soil properties, there is a dearth of data quantifying the effects of slash pile burning. After a slash pile burn, a change in soil physical and biochemical properties may contribute to a shift in plant community composition. Initially, fire removes vegetative cover from the surface resulting in greater soil temperature and moisture fluctuations (Pietikäinen & Fritze 2005). Depending on fire severity, there may be a loss of soil organic matter

(SOM), which will potentially reduce soil water holding capacity (WHC) and nutrient capital (Kennard & Gholz 2001). With a reduction in soil water holding capacity, drought tolerant species may gain the competitive advantage. High temperatures could cause mortality of biota within surface soils including mycorrhizal fungi, which are found throughout the soil matrix. When SOM is lost, available soil nutrients are reduced, and aggregation and physical structure can be altered. In turn, soil compaction, crusting, and reduced drainage occur, which may lead to soil erosion (Austin & Baisinger 1995).

Soil N tends to be limiting in western forests and fire dramatically alters N dynamics (DeLuca & Zouhar 2000). The pool of N is directly altered by fire through volatilization, pyrolysis of organic material, translocations, and lysis of microbial cell walls (Smithwick et al. 2005). Depending on fire intensity, there is a rapid release of N in the form of NH_4^+-N (Neary et al. 1991; Neary et al. 2005; DeLuca & Zouhar 2000; Choromanska & DeLuca 2001) that can be observed for one to two years (Certini 2005), followed by a peak of NO_3^--N with the recovery of nitrifying bacteria. When N mineralization is greater than immobilization and plant uptake, the mobile NO_3^--N fraction can leach and potentially influence the biochemistry of nearby bodies of water (Turner et al. 2007).

The temporal abundance of NH_4^+-N followed by NO_3^--N in the system may influence revegetation of burned scars. After severe fire, net primary productivity is low initially, but an herbaceous community dominated by graminoids and forbs is often first to establish (Turner et al. 2007). These ruderal species benefit from reduced competition and are capable of rapidly accessing mineralized N, and often prefer NO_3^--N to NH_4^+-N

or organic N (Wardle 2002; Turner et al. 2007). In several slash pile burn studies, non-native species were often the first to colonize scars (Dickinson & Kirkpatrick 1987; Haskins & Gehring 2004; Korb et al. 2004).

Ash deposition may be greater following slash pile burning due to the large volume of burned biomass in a concentrated area. In some instances, heavy ash deposition may stimulate plant productivity within a year after fire (Raison 1979). Increased plant productivity may be the result of deposition of nutrients through ash, but may also be due to heaping of soil onto the slash during tractor piling, reduced plant competition, increased soil temperatures, or a shift in the composition of microbial communities (Raison 1979).

Microbial biomass may be positively or negatively impacted by a loss of vegetative cover (Pietikäinen & Fritze 2005). Fire may reduce microbial biomass through direct heating, but some propagules will survive or recolonize from adjacent areas. Generally, fire reduces soil microbial biomass in the top few centimeters of soil (Haskins & Gehring 2004) and it may contribute to a shift from the fungal-based energy channel to the bacteria-based (Pietikäinen et al. 2000). Changes in the soil environment may yield a long-term reduction in soil microbial biomass. For instance, lower moisture content and altered structure or quantity of soil organic matter will inhibit recolonization of a burned area (Pietikäinen & Fritze 1995).

Slash Pile Scar Restoration

Where forest restoration is the goal, it is imperative that the impact of slash pile burning on site productivity and plant community dynamics be more thoroughly evaluated. Land managers often attempt to ameliorate slash pile scars, but the success of various strategies has not been well-documented or quantitatively evaluated. There is a need to identify practical slash pile scar restoration techniques that increase vegetative cover to reduce soil erosion, reduce the risk of nutrient losses, and suppress non-native plant invasion.

Ideally, restoration would emphasize the use of on-site materials without impairing the productivity of adjacent areas, thus minimizing costs and enhancing the potential of re-introduction of native seed and biota. Simple measures such as soil scarification may assist in plant recovery by incorporating the thick ash layer into the mineral soil, thereby homogenizing the plant rooting zone and aerating the soil. Scarification or tillage is shown to accelerate the recovery of compacted soils and improve seedling establishment (Bulmer 2000; Cole & Spildie 2007). The addition of organic matter may accelerate the recovery of slash pile scars by compensating for the vegetation, humus, and SOM removed during fire. Spreading or incorporating locally collected forest litter over the scar would provide a local seed source, reintroduce native soil biota, and a source of soil nutrients. Alternatively, the use of composted biosolids is shown to aid in native plant establishment by improving soil stability, decreasing soil C:N following wildfire (Meyer et al. 2004) and results in the slow release of macronutrients and micronutrients while also improving WHC (Zabinski et al. 2002). In

what appears to be the only study of its type to date, Korb (2004) performed a factorial slash pile restoration experiment involving seed and soil amendments. Natural revegetation was negligible after two years and the addition of live soil and seed most effectively increased native graminoid cover; however, by only a small margin.

Mycorrhizal Inoculum Potential (MIP)

Plant mutualisms often involve mycorrhizal fungi (Brundrett 1991). Nutrient uptake by most plants in natural ecosystems is strongly influenced by associations with mycorrhizal fungi (Grogan et al. 2000). The fungus colonizes plant roots and acquires C from the host plant, and in exchange increases access to nutrients such as P. Mycorrhizae may increase water uptake by effectively increasing root surface area, and in some cases, protection from pathogens. Mycorrhizae are also considered to be important in soil formation, increasing soil stability, and aggregation (Smith & Read 1997). Yet, little is known in regard to how mycorrhizal fungi are impacted by high-severity fires, such as slash pile burning, and how the presence or absence of mycorrhizal fungi may favor certain species.

Prescribed fire may decrease, have no effect, or increase arbuscular mycorrhizal (AM) fungal propagule density (Korb et al. 2003). Fire severity likely determines the effects on AM fungi. Heating above 80 °C has been known to almost completely eliminate AM propagule densities (Korb et al. 2003) and after only 15 minutes, a slash pile may reach 600 °C at a depth of 5 cm (Neary et al. 2005). Korb et al. (2004) indicates that slash pile burning almost eliminated AM fungal propagules and the site remained

negatively impacted for 15 months. After five years, AM fungi may recover from slash pile burning (Haskins & Gehring 2004). Relevant factors were highlighted by Klopatek et al. (1988), who found that the effect of burning on AM fungi depended on the quantity of litter and soil moisture. Dry burned soils had the greatest decrease in AM colonization. The quantity of litter was correlated with surface temperature, and higher temperatures resulted in a greater loss of AM colonization.

AM fungi are an important symbiont for many herbaceous plants in ponderosa pine forest ecosystems (Korb et al. 2003), and AM fungal colonization rates following fire are correlated with establishment of native plant cover (Korb et al. 2004). In a study following wildfire, decreased propagule densities of AM fungi resulted in lower germination rates of the most dominant pre-fire species, thus impacting the recovery of understory species (Vilarino & Arines 1991). Competitive interactions may also be affected by mycorrhizae, either directly by differentially increasing some species growth (Wilson & Hartnett 1998) or indirectly via transfer of photosynthetically fixed C among plants (Selosse et al. 2006).

The importance of AM fungi in fire-suppressed ponderosa pine ecosystems may be diminished where the forest canopy is dense and there is a reduced abundance of facultative or dependent AM fungal species. Korb et al. (2003) found the quantity of infective propagules of AM fungi was positively correlated with graminoid cover and herbaceous understory species richness and negatively correlated with overstory tree canopy cover and litter. Due to prior shading, recently thinned ponderosa pine forests may not have a high degree of infective AM fungal propagules, therefore slash pile

burning may not greatly alter that portion of the soil biota. Other mechanisms, such as smoke, may also influence spore germination or mycelial growth of some fungi by reducing their susceptibility to infection (Raison 1979).

Successful revegetation and non-native species management in slash-pile burned areas requires the use of species which can establish and grow in the post-fire soil environment. An additional objective of this research is to quantify the growth potential and AM fungal colonization levels of native grass species and non-native species found when growing in the greenhouse in burned and unburned soils. Soil physical and biochemical properties are dramatically altered by slash pile burning, and the changes are likely to shift the growth potential of both native and non-native plant species. Rapid nutrient mineralization may favor certain ruderal species, while a thick ash layer may have an inhibitory effect on plant establishment where high mineral cation concentrations may be toxic to plants (Giovannini et al. 1990; Kennard & Gholz 2001). Mortality of AM fungal propagules following heating may adversely impact colonization levels and therefore reduce the ability of some species to acquire nutrients. The purpose of this portion of the study is to assist with revegetation efforts by comparing the growth potential of graminoid and non-native forb species to determine whether certain species favor or tolerate fire-induced change in the soil environment. The secondary objective is to identify the role of AM fungi in ponderosa pine forest ecosystems recently treated for fuels reduction. Documenting the response of individual species to slash pile burning will assist revegetation efforts.

Methods

Site Description

The research site is on private land adjacent to the Bitterroot National Forest (BNF) near Florence, Montana at an elevation of 1,103 m and with a mean annual temperature of 7 °C. The site receives an average of 33 to 40 cm annual precipitation and with an east-facing aspect, and a slope ranging from a 2 to 25%. Soils have a mean pH of 5.2 and a C:N of 24:1 (Chapter 2). Soils are Haplocryepts and Eutrocryepts composed of stony sandy loam formed on granite residuum and composed of 30% sand and 37% clay. The O horizon is approximately 7 cm deep and composed of 4 cm of Oi, 1 cm of Oe, and approximately 2 cm of Oa. The mineral horizon has an ochric epipedon with an average thickness of 14 cm, and the mean depth to decomposed granite parent material of 18 cm. Ponderosa pine and Douglas-fir dominate the canopy, and the habitat type is Douglas-fir – ninebark (*Physocarpus malvaceus* (Green) Kuntz). Total vegetative cover in the unburned forest understory averages 42% (n=45). The site contains a mix of shrubs (24%), graminoids (8%), forbs (7%), non-native species (2%), and tree seedlings (1%). A fuels-reduction thinning occurred in March to April of 2006 and logging residues were tractor-piled and burned from late April to May of the same year. Scar diameter ranged from 2.8 m to 7.6 m and averaged 4.5 m. In October of 2006, 23 scars were randomly selected for the restoration experiment.

Restoration Treatment

A total of 23 scars were selected at random from the 45 slash pile scars previously assessed for post-fire soil biochemical impacts. Scars with highly irregular boundaries or large stumps that would impede restoration work were excluded. The edge was identified along the periphery of the scar where litter was charred to the mineral soil, but not completely combusted into ash. Control plots were established in a random direction 6 m from the edge of the scar. Scars were randomly assigned to one of two blocks--seeded with native grasses or no seed addition. Scars were divided into five radial sections to account for the fire temperature gradient from center to edge, and the following treatments were assigned to each wedge: 1) control (no amendment); 2) addition to the surface of locally collected organic matter; 3) scarification 4) scarification with addition of locally collected organic matter; 5) scarification with addition of commercial compost.

Control plots received no soil treatment, and those that were seeded had seed sprinkled on the ash surface. Local organics included material from the Oa/Oe horizons (duff layer) with a C:N ratio of 32:1 (n=42) and were added to the surface as a 2.5 cm layer or incorporated into the soil to a depth of 10 cm. In the scarification-only treatment, the upper 10 cm of the mineral soil was hand-scarified using Pulaskis and rakes. Compost was acquired from Eko Compost©, Missoula, MT. Eko Compost© is composed of log-yard waste and Missoula Water Reclamation Facility sewage sludge with a C:N of 20:1 (DeLuca & Lynch 1997). The mix was applied to the surface as a 5 cm layer and scarified into the ash and soil to a depth of 10 cm.

Seeded slash pile scars were broadcast treated by seed with the following species and application rates: bluebunch wheatgrass (*Pseudoroegneria spicata* (Pursh) A. Löve*)* 8.4 kg/ha (7.5 lbs/ac), Pryor slender wheatgrass (*Elymus trachycaulus* (Link) Gould ex Shiners) 8.9 kg/ha (8.0 lbs/ac), Idaho fescue (*Festuca idahoensis* E.) 2.8 kg/ha (2.5 lbs/ac), and Sandberg's bluegrass (*Poa sandbergii* V.) 2.2 kg/ha (2.0 lbs/ac). The mix was selected to incorporate graminoids that emerge from early spring through mid-summer. Certified weed-free native seed was acquired from the Coeur d'Alene Forest Service Nursery, Idaho. Seeds were raked into the upper one to two cm following treatments.

Soil Parameters

Composited mineral soil samples (10 sub-samples; 0 to 10 cm) were collected in June of 2007 from the center of each restoration treatment point 13 months post-burn. Litter, ash, and charcoal were removed from the surface, and composite samples were collected to a depth of 10 cm using a 2.5 cm diameter stainless steel soil probe. Composite samples were kept in a cooler and returned to the lab where they were refrigerated overnight and extracted the following day.

Fresh soil samples were sieved to 2 mm, homogenized, and analyzed for extractable inorganic N, potentially mineralizable N (PMN), water soluble P (PO_4^{3-}), and microbial biomass (n=138). To determine extractable inorganic N content, an oven-dry equivalent of 25 g of soil was added to 50 ml of 2 M KCl and shaken for 30 minutes. The solution was then filtered with Whatman # 2 filter paper and the extract analyzed for

NO_3^--N and NH_4^+-N (Mulvaney 1996) by using Lachat flow injection autoanalyzer (Milwaukee, WI) 13 months post-burn. Soluble NO_3^--N was determined using the cadmium reduction method and soluble NH_4^+-N was determined using the salicilate-nitroprusside method (Mulvaney 1996). To determine water-soluble P, 10 g of soil was placed in 20 ml of 0.01 M $CaCl_2$ and shaken for 30 minutes (Murphy & Riley 1962). The solution was then filtered through Whatman # 42 filter paper and the extract analyzed for soluble PO_4^{-3}-P using the molybdate-ascorbic acid method on Lachat flow injection autoanalyzer (Milwaukee, WI). Potentially mineralizable N (PMN) was measured using the 14-day anaerobic incubation procedure (Bundy and Meisinger 1996), where a 5 g soil sample was placed into 12.5 ml of distilled water, and a stream of N_2 gas was bubbled through the suspension for 10 seconds. The sample was then placed in a constant temperature chamber at 25 °C for 14 days after which 12.5 ml of 4 M KCl was added to the aqueous suspension to create a 2 M KCl extractant. This suspension was shaken for 30 minutes, filtered through Whatman #2 filter paper, and the extract analyzed for NH_4^+-N as described above. To calculate PMN, the concentration of NH_4^+-N in the un-incubated soil samples (time zero sample) was subtracted from the NH_4^+-N concentration at 14 days. To assess *in situ* post-thaw mineralization and nitrification, spherical nylon capsules were installed containing 1 g of mixed bed ionic resin (Unibest, Bozeman, Montana). Total N mineralization and nitrification is based on sorption of NO_3^--N and NH_4^+-N to the ionic resins. The capsules, 2 cm in diameter, were carefully placed 5 cm below the soil surface using a stainless steel soil probe in the center (*n*=80) of slash pile scars. Capsules were left in place for eight months and retrieved in June of 2008. Resins

were extracted by sequential washing in 2 M KCl and analyzed for NH_4^+-N by the Berthelot method and NO_3^--N by the cadmium reduction method via flow injection analyzer (Mulvaney 1996). Microbial biomass was determined with the fumigation extraction, ninhydrin-reactive N method (Jorgenson and Brooks 1990), as modified by DeLuca & Keeney (1993). A 25 g soil sample was placed in a 200 ml French Square bottle and exposed to chloroform for 24 hours. The fumigated samples as well as an un-fumigated control sample were extracted with 50 ml of 2 M KCl as described above and both samples analyzed for ninhydrin reactive N (Jorgenson and Brooks, 1990). Biomass N was then calculated as the difference between ninhydrin reactive N in fumigated samples relative to the control, multiplied by a factor of 3.1 (Joergensen & Brooks 1990).

Mineral soil samples were analyzed for moisture content, pH, and bulk density (*n*=138). Soils were also analyzed for total C and N (*n*=138) by dry combustion (Leco Truspec Instrument). Soil moisture content was determined gravimetrically by drying a 30 g moist sub-sample at 105 °C for 24 hours and then reweighing the sample. Dried mineral soils were analyzed for pH in a 2:1, 0.01 M $CaC1_2$ soil suspension. Bulk density was determined with a 5 cm diameter by 5 cm long brass sleeve that was inserted into the soil using a slide hammer coring device. The core was removed, returned to lab, dried at 60 °C, analyzed for total mass and density by dividing mass by volume (*n*=75).

Vegetation

Slash pile scar plant establishment was monitored with a post-restoration vegetation survey in August 2007 and again in August 2008. Each plot was assessed for

percent cover using a 0.5 m^2 frame placed near the center and edge in each pie section treatment area. At each sampling point (*n*=115), total vegetation, shrub, grass, sedge, forb, non-native invasive species, and tree seedling percent cover was estimated. Shrubs, sedges, and tree seedlings were only present in trace amounts, and were later eliminated from analysis. Four slash piles were adjacent to a road where non-native species, namely leafy spurge (*Euphorbia esula* L.) were abundant. The percent cover of leafy spurge 1, 3, and 6 m outside of each treatment section was recorded (*n*=60). A vegetation survey of unburned soil was also performed in July 2007 (*n*=23) where percent cover and density were recorded to the species level using a 0.5 m^2 frame placed 6 m in a random direction from the slash pile scars. Percent forest canopy cover was estimated using a hand-held densiometer (*n*=23). Percent slope was estimated using a clinometer (*n*=23).

Greenhouse Experiment

Composite soil samples were collected in August of 2006 from the center of the slash pile scars and adjacent control plots to a depth of 10 cm. Ash samples were also collected to the surface of the mineral soil. Soils from scar centers were composited and homogenized, as were soils from control plots and, along with ash samples, were placed in cool storage until planting in January 2007.

Three native bunchgrass species from the revegetation mix and three non-native invasive forb species were selected for the greenhouse experiment. Graminoids included: bluebunch wheatgrass (*Pseudoroegneria spicata* (Pursh) A. Löve), Idaho fescue (*Festuca idahoensis* E.), and Sandberg's bluegrass (*Poa sandbergii* V.). Non- native invasive seed

was obtained from the USDA Agricultural Research Service, Miles City, MT and Montana State University Agricultural Extension, Corvallis, MT and included orange hawkweed (*Hieracium aurantiacum* L.), leafy spurge (*Euphorbia esula* L.), and spotted knapweed (*Centaurea maculosa* Lam.) All species are mycorrhizal. The grasses are common in seeding mixes and are native to the ponderosa pine understory. The forbs are common category one or two noxious weeds in Southwest Montana.

The experiment was a complete factorial design with three soil treatments, six plant species and 10 replicates of each treatment. Soil treatments include an unburned control soil, a burned soil, and a burned soil with a 2.5 cm surface layer of ash. Four seeds were sown in 4 cm diameter x 13 cm depth conetainers filled with soil, and conetainers were placed in a greenhouse and watered daily. Leafy spurge seed was germinated prior to seeding. Conetainers were spaced by one tray cell rotated to random positions within trays each week. Photoperiod was not regulated.

Seedlings were thinned to one plant per conetainer, and seedlings were transplanted into containers if necessary from plants sown simultaneously with the experimental plants in all three substrates. After two weeks, transplanting no longer occurred.

Plants were harvested at ten weeks in the greenhouse. Shoots and roots were cleaned, dried at 60 °C for 48 hours or until a constant mass was reached, and weighed. Root: shoot ratio and biomass was calculated and compared between soil treatments.

Mycorrhizal inoculum levels were measured in roots of each plant species grown in the three soil substrates. Dry roots were cut into 2.5 cm segments, cleared in 5% KOH

and stained with trypan blue in lactoglycerin (Koske & Gemma 1990). Counts of AM fungal structures included arbuscules, vesicles, and internal mycorrhizal hyphae. The gridline intersect method using a microscope was used to quantify colonization levels for each treatment combination.

Data Analysis

Differences in restoration treatments on soil properties of slash pile scars were tested with ANOVA (analysis of variance; SPSS General Linear Model Univariate ANOVA, SPSS, Inc., Chicago, IL.Version 15.0). The effects of restoration treatments were assessed using either one-way or two-way ANOVA, with soil treatment and seeding as fixed effects. The Least Squares Difference (LSD) post-hoc test was used to determine which treatments differed significantly ($\alpha<0.05$). All data were evaluated for conformations to the assumptions of parametric statistical analysis. Homogeneity of variance was tested using Levene's test ($\alpha<0.05$) and normal distribution of the dependent variable was verified with a normal curve and by observing skewness and kurtosis data. When data could not be transformed to meet these assumptions, the Kruskal-Wallis (K-W) non-parametric technique was used. The following analyses were completed for slash pile restoration soil data: pH (K-W), bulk density, total C and N, NH_4^+-N (log transformed), NO_3^--N (log transformed), resin-sorbed NH_4^+-N (square root transformed), resin-sorbed NO_3^--N (log transformed), PMN, PO_4^{3-}-P (square root transformed), and microbial biomass.

Vegetation survey data were analyzed using one-way and two-way ANOVA. For one-way ANOVA, soil treatment was the fixed effect. Total cover of non-native species in seeded plots was square root transformed for both 2007 and 2008. The K-W non-parametric test was used for forb data in seeded plots in 2008 and graminoid and forb data in non-seeded plots in 2007. All other ANOVAs used the LSD post-hoc test to assess differences between treatments ($\alpha<0.05$). In two-way ANOVAs, seeding and treatment were the fixed effects. Total vegetative cover and vegetative cover without non-native species were square root transformed for 2007 and 2008 data.

In the greenhouse experiment, one-way ANOVA was used with soil treatment as the fixed effect. Shoot biomass data for orange hawkweed was log transformed and the K-W test was used for Idaho fescue. Root: shoot data required the K-W test for orange hawkweed. Mycorrhizal infectivity was also analyzed non-parametrically with K-W to detect differences between soil substrate.

Results

Restoration Treatment Effects on Soil Parameters

Slash pile restoration treatments altered several soil properties. Elevated soil pH within the scars fell from 7.4 in the control (un-treated burn) to 6.4 in the scarified soils; however, all treatments were elevated above the unburned soil pH of 5.1 (Table 9). Soil pH changed with restoration treatments ($p<0.01$), but not with seeding ($p=0.91$; Table 14). Mean soil bulk density differed by treatment ($p=0.04$), but none of the treatments differed from the unburned bulk density value of 0.82 g/cm^3 (Table 9).

Total C declined from 6.06% in unburned soils to 5.14% in burned soil. No treatments increased total C, but scarification was significantly lower than the unburned soil (Table 9). For total N, scarification was lowest in the scarification treatment and scarification and compost treatment most effectively increased total N.

Table 9. Mean ± 1 S.E. (n=138) for soil pH, bulk density, and total C and N following slash pile restoration treatments in Western Montana. Letters in columns indicate treatment differences (α<0.05).

	pH	Bulk Density (g/cm^3)	Total C (%)	Total N (%)
Control	7.4 ± 0.1 d	0.86 ± <0.01 ab	5.14 ± 0.3 ab	0.25 ± <0.01 abc
Organic Matter	7.1 ± 0.1 c	0.93 ± <0.01 b	5.14 ± 0.3 ab	0.23 ± <0.01 ab
Scarification	6.4 ± 0.1 b	0.82 ± <0.01 ab	4.84 ± 0.2 a	0.23 ± <0.01 a
Scarification & Organics	7.3 ± 0.1 d	0.78 ± <0.01 a	5.37 ± 0.3 ab	0.24 ± <0.01 ab
Scarification & Compost	7.4 ± 0.1 d	0.80 ± <0.01 ab	6.00 ± 0.4 ab	0.29 ± <0.01 bc
Unburned	5.1 ± 0.1 a	0.82 ± <0.01 ab	6.06 ± 0.3 b	0.30 ± <0.01 c

Seeding did not influence the recovery of total C or N (p=0.17 and p=0.58; Table 14) one year post-burn.

Soil inorganic N in all soil amendment plots was higher than the unburned control for both NH_4^+-N and NO_3^--N. The concentration of soil NH_4^+-N was greatest in the un-treated burned control and where surface organic matter was applied (Table 11). No differences were found between restoration treatments for soil NO_3^--N (Table 11), but a seeding effect was observed (p<0.01; Table 13). Overall, soil NO_3^--N was 9.3 $\mu g\ g^{-1}$ in seeded plots and 13.5 $\mu g\ g^{-1}$ in plots that received no seed. Within the non-seeded plots,

the scarification treatment had a higher concentration of NO_3^--N than the un-treated burn (Table 10).

Table 10. Treatment mean ± 1 S.E. (n=138) for NO_3^--N for the overall mean (total), seeded plots, and non-seeded plots in a slash-pile burn study in Western Montana. Asterisks indicate a seed effect and letters indicate a treatment effect within a column (α<0.0.5).

	NO_3^--N (µg g^{-1})	
	Seed	No Seed
Control	7.4 ± 3.5 b	11.3 ± 3.5 b
Organic Matter	9.7 ± 2.4 b	14.9 ± 4.3 bc
Scarification	12.1 ± 4.8 b	22.1 ± 2.9 c
Scarification & Organics	15.0 ± 5.7 b	16.0 ± 3.4 bc
Scarification & Compost	10.1 ± 3.7 b	15.9 ± 4.3 bc
Unburned	1.2 ± 0.2 a	1.2 ± 0.2 a
Total	9.3 ± 1.6*	13.5 ± 1.6

Total N mineralization, measured as resin-sorbed NH_4^+-N, did not differ between treatments, but total nitrification was greatest in the scarification with compost treatment at 3.93 µg g^{-1} (Table 11). No seeding effect was observed (Table 14). There was no treatment effect (p=0.07; Table 14) for PMN and no seeding effect was found (p=0.07; Table 14).

Table 11. Means for KCl extractable inorganic N ± 1 S.E. (n=138) and resin-sorbed inorganic N (n=80) following restoration treatment of slash pile burn scars in Western Montana. Letters indicate significant differences among treatments (α<0.05)

	NH_4^+-N (µg g^{-1})	NO_3^--N (µg g^{-1})	Resin-sorbed NH_4^+-N (µg g^{-1})	Resin-sorbed NO_3^--N (µg g^{-1})
Control	4.79 ± 1.0 c	9.27 ± 2.5 b	1.51 ± 0.4	0.81 ± 0.2 a
Organic Matter	3.84 ± 0.8 c	11.96 ± 2.3 b	1.81 ± 0.6	1.30 ± 0.3 a
Scarification	1.00 ± 0.3 b	16.89 ± 3.0 b	1.88 ± 0.4	0.57 ± 0.1 a
Scarification & Organics	1.12 ± 0.5 b	15.47 ± 3.3 b	1.18 ± 0.3	1.27 ± 0.3 a
Scarification & Compost	0.56 ± 0.2 b	12.87 ± 2.8 b	0.64 ± 0.13	3.93 ± 0.7 b
Unburned	0.01 ± <0.01 a	1.20 ± 0.2 a	-	-

Differences in water-soluble P were found among restoration treatments (p<0.01; Table 14). The control, surface organic matter, and scarification with organic matter contained the lowest concentration of PO_4^{3-}-P. The unburned plots were intermediate and the scarification with compost had the highest concentration with 5.65 µg g^{-1} (Table 12). All treatments, except scarification, exceeded the unburned plots. Microbial biomass was greatest in the unburned plots at 28.55 µg g^{-1}. Among the restoration treatments, scarification with organic matter and scarification with compost had more microbial activity than the control, surface organic matter, and scarification treatments (Tables 12 and 14).

Table 12. Mean ± 1 S.E. for PMN, water-soluble P, and microbial biomass ± 1 S.E. following restoration treatment of slash pile burn scars in Western Montana. Letters indicate significant differences between treatments (α<0.05)

	PMN ($\mu g\ g^{-1}$)	PO_4^{3-}-P ($\mu g\ g^{-1}$)	Microbial biomass ($\mu g\ g^{-1}$)
Control	7.45 ± 1.6	3.44 ± 0.5 b	13.86 ± 1.8 a
Organic Matter	10.51 ± 1.7	3.57 ± 0.5 b	9.84 ±1.4 a
Scarification	7.34 ± 1.0	2.79 ± 0.5 ab	14.25 ± 1.4 a
Scarification & Organics	11.85 ± 1.3	2.91 ± 0.3 b	21.13 ± 1.8 b
Scarification & Compost	10.0 ± 1.2	5.65 ± 0.6 c	21.45 ± 2.0 b
Unburned	-	1.94 ± 0.3 a	28.55 ± 1.6 c

Table 13. ANOVA for NO_3^--N (log transformed) and PMN by treatment, and PMN by seed application following slash pile scar restoration treatments in Western Montana.

	NO_3^--N ($\mu g\ g^{-1}$)				PMN ($\mu g\ g^{-1}$)			
	d.f.	MS	F	P	d.f.	MS	F	P
Seed	5	16.73	10.82	<0.01	4	74.16	2.21	0.07
Error	132	1.55			89	33.49		
	PMN-Seed ($\mu g\ g^{-1}$)							
	d.f.	MS	F	P				
Seed	1	118.89	3.46	0.07				
Error	92	34.33						

Table 1. ANOVA by treatment and seeding for pH, bulk density, total C and N, C:N, NH_4^+-N (log), NO_3^--N (K-W), resin-sorbed NH_4^+-N (square root), resin-sorbed NO_3^--N (log), PMN (K-W), PO_4^{3-}-P, and microbial biomass following slash pile restoration treatments in Western Montana.

	pH				Bulk Density (g/cm^3)			
	d.f.	MS	F	P	d.f.	MS	F	P
Treatment	5	18.13	168.51	<0.01	5	0.07	2.48	0.04
Seed	1	0.00	0.01	0.91	1	0.07	2.42	0.12
T x S	5	0.15	1.41	0.22	5	0.04	1.45	0.21
Error	127	0.11			125	0.03		
	Total C (%)				Total N (%)			
	d.f.	MS	F	P	d.f.	MS	F	P
Treatment	5	5.97	2.98	0.01	5	0.02	4.94	<0.01
Seed	1	3.78	1.89	0.17	1	0.00	0.32	0.58
T x S	5	2.12	1.06	0.39	5	0.00	0.80	0.55
Error	126	2.00			126	0.00		
	Microbial biomass ($\mu g\ g^{-1}$)				NH_4^+-N ($\mu g\ g^{-1}$)			
	d.f.	MS	F	P	d.f.	MS	F	P
Treatment	5	1065.5	16.24	<0.01	5	10.20	22.79	<0.01
Seed	1	90.78	1.38	0.24	1	0.19	0.43	0.51
T x S	5	39.07	0.60	0.70	5	0.18	0.40	0.85
Error	126	65.60			112	0.45		
	NO_3^--N ($\mu g\ g^{-1}$)				Resin-sorbed NH_4^+-N ($\mu g\ g^{-1}$)			
	d.f.	Chi-Square	Asymp. Sig.		d.f.	MS	F	P
Treatment	5	39.08	<0.01		4	0.67	1.87	0.12
Seed	1	6.96	0.01		1	0.76	2.11	0.15
T x S					4	0.35	0.98	0.42
Error					80	0.36		
	Resin-sorbed NO_3^--N ($\mu g\ g^{-1}$)				PO_4^{3-}-P ($\mu g\ g^{-1}$)			
	d.f.	MS	F	P	d.f.	MS	F	P
Treatment	4	10.38	6.32	<0.01	5	2.67	7.33	<0.01
Seed	1	0.63	0.38	0.54	1	0.73	2.01	0.16
T x S	4	0.45	0.27	0.89	5	0.39	1.06	0.39
Error	79	1.64			120	0.36		

Restoration Treatment Effects on Vegetation

Total cover was higher in seeded and non-seeded plots. The mean of seeded plots ranged from 28 to 68% cover, while non-seeded plots ranged from 2.5 to 25% cover

(Figure 4). Among seeded plots in 2007, only one treatment differed from the control, but within the treatments plant cover was lower on the organic matter plots than the scarification plots. In 2008, two treatments resulted in higher plant cover than the control, but between the different treatments there was no significant difference. In non-seeded plots there were treatment differences in 2007, but not in 2008. Scarification with organic matter yielded the highest total cover in 2007, but all differences disappeared one year later (Figure 4b). In 2007, there is a seeding, treatment, as well as a seeding and treatment interaction (Table 15), indicating the effects of seeding on percent cover depended on soil treatment. By 2008, there is still a seeding and treatment effect, but no longer an interaction. Scarification and compost yielded the highest percent cover in both 2007 and 2008. The control was consistently lowest (Figure 4a).

Total vegetative cover was primarily composed of graminoids, forbs, and non-natives. Seeding influenced the composition of vegetative cover in 2007 and 2008 ($p<0.01$). Graminoids dominate cover in seeded plots for both years. Restoration treatments affected graminoid cover in 2007 ($p<0.01$) where scarification and compost plots have the highest cover and the control has the lowest. There is a treatment effect ($p=0.04$; Table 15) in 2008 where both scarification and scarification with compost amendment have higher graminoid cover than the control. In contrast, graminoid cover in non-seeded plots did not differ between treatments in 2007 or 2008 (Figures 5c and 5d). Forbs (native and non-native) were not influenced by treatment. Among the non-seeded plots, no plant group varied by restoration treatment.

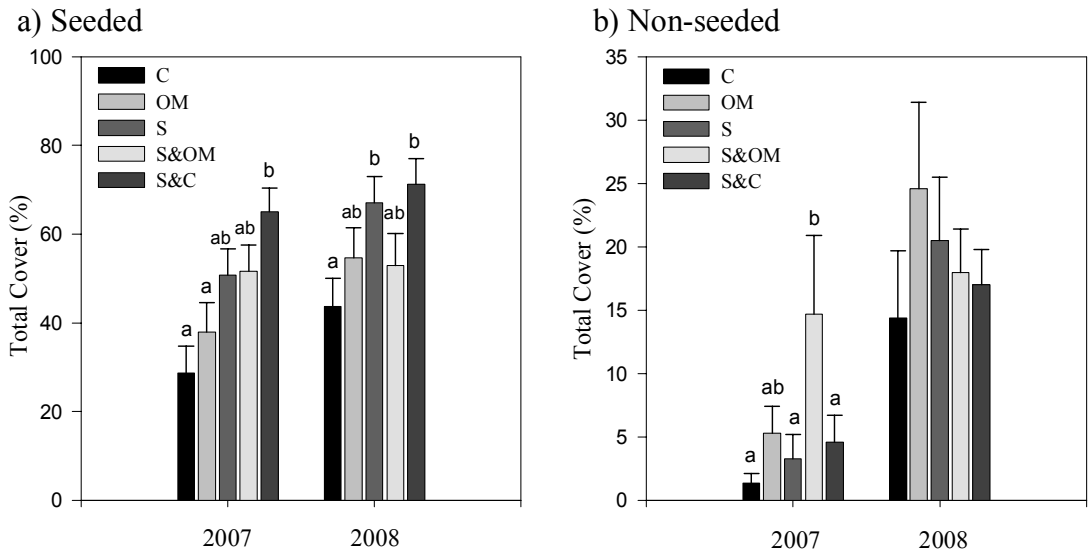

Figure 4. Total vegetative cover (%) by soil amendment for seeded (a) and non-seeded (b) slash pile scars in Western Montana one and two years post-burn. Letters denote significant differences between treatments by year (α<0.05; n=138), and error bars are 1 S.E. Note the scale difference for the y-axis.

There is a trend for non-natives to make up the highest proportion of total cover in non-seeded plots (Figures 5c and 5d). To determine whether seeding or treatments help suppress non-native species, a non-parametric test was used to assess seeding application and restoration treatment in 2007 and 2008. This indicates that seeding affected non-native cover in 2007, but not 2008. Soil amendments had no influence over non-native forb cover for both years (Table 16). In contrast, both amendments and seeding are significant for total cover minus non-native species in 2007 and 2008; however, there is no interaction for either year (Table 15).

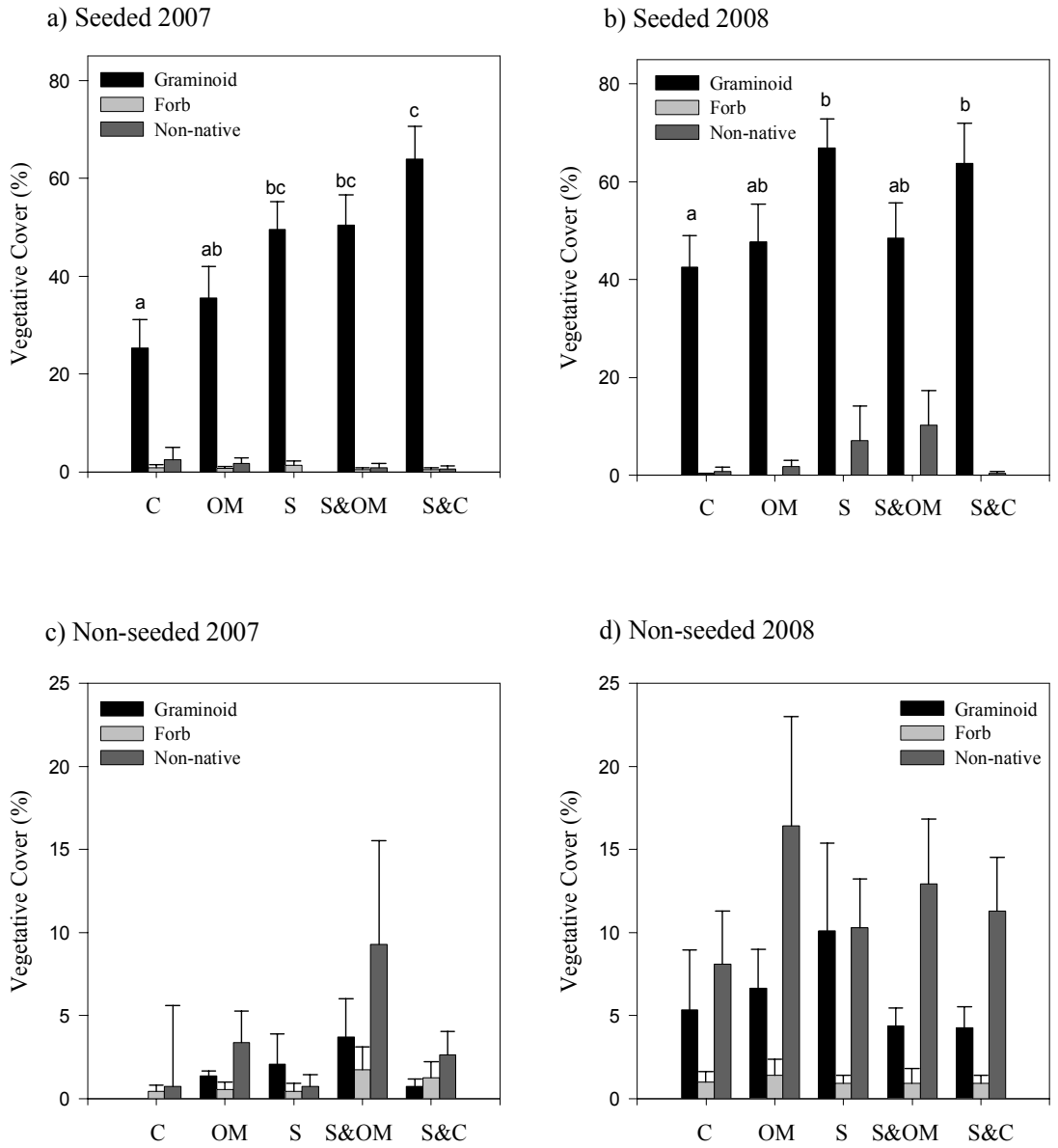

Figure 5. Plant cover (%) of graminoids and native and non-native forbs by soil treatment for seeded plots in 2007 (a) and 2008 (b) and non-seeded plots in 2007 (c) and 2008 (d) for slash pile scars in Western Montana. Letters denote significant differences between treatments by year (α<0.05; n=138) and error bars are 1 S.E. Note the y-axis scale difference between seeded and non-seeded plots.

Table 15. Two-way ANOVA for total vegetative cover (square root transformed) and vegetative cover without non-native species (square root transformed) for seed application and restoration treatments in 2007 and 2008 for small-scale slash pile burns in Western Montana.

	2007				2008			
Total Cover (%)	d.f.	MS	F	P	d.f.	MS	F	P
Treatment	4	16.7	5.9	<0.01	4	7.26	2.67	0.04
Seed	1	729.54	257.86	<0.01	1	329.24	121.32	<0.01
T x S	4	7.99	2.83	0.03	4	2.90	1.07	0.38
Error	105	2.83			105	2.71		
Cover w/o non-natives	d.f.	MS	F	P	d.f.	MS	F	P
Treatment	4	10.19	3.62	0.01	4	6.03	2.30	0.06
Seed	1	867.77	308.27	<0.01	1	774.62	296.08	<0.01
T x S	4	4.10	1.46	0.22	4	2.93	1.12	0.35
Error	105	2.82			103	2.62		

Table 16. Kruskal-Wallis non-parametric test of non-native cover for seed application and restoration treatment in 2007 and 2008 for small-scale slash piles burns in Western Montana.

Seed	Chi-Square	d.f.	Asymp. Sig.
2007	5.73	1	0.02
2008	0.30	1	0.59
Treatment			
2007	7.28	4	0.12
2008	1.13	4	0.89

Table 17. ANOVA for vegetative cover (%) of graminoids and forbs (native and non-native) for slash pile restoration treatments by seed application in Western Montana one and two years post-burn. Data for non-natives in seeded plots were square root transformed for 2007 and 2008. The Kruskal-Wallis test was used for forb data in seeded plots in 2008 and graminoid and forb in non-seeded plots in 2007.

	2007				2008			
SEEDED	d.f.	MS	F	P	d.f.	MS	F	P
Graminoid								
Treatment	4	2,658.11	6.24	<0.01	4	1,378.88	2.25	0.08
Error	55	425.70			55	612.53		
Forb								
Treatment	4	1.73	0.44	0.78	4	8.14	0.09	
Error	55	3.94				Chi-square	Asymp. sig	
Non-native								
Treatment	4	0.52	0.48	0.75	4	4.72	1.34	0.27
Error	55	1.09			55	3.52		
NO SEED	d.f.	MS	F	P	d.f.	MS	F	P
Graminoid								
Treatment	4	6.85	0.14		4	63.53	0.59	0.67
Error		Chi-square	Asymp.sig		50	107.78		
Forb								
Treatment	4	3.7	0.51	0.75	4	0.43	0.07	0.99
Error	50	7.33			50	5.99		
Non-native								
Treatment	4	5.28	0.26		4	105.66	0.55	0.70
Error		Chi-square	Asymp. sig		50	191.4		

The Effects of Burning on Mycorrhizal Inoculum Potential

In a controlled greenhouse experiment, six species were grown in three substrates including an unburned control, burned soil, and burned soil with a layer of ash. Shoot and root biomass were affected by the soil substrate for all species. In the unburned control, all the non-native forbs had lower total biomass. Among the native grasses in the control, bluebunch wheatgrass had the lowest biomass, while Idaho fescue increased biomass in

both the control and burned soil with ash. In the burned soil, Sandberg's bluegrass and Idaho fescue had lower biomass and only orange hawkweed had the maximum biomass in the burned soil. Two species increased biomass in the burned with ash soil (Sandberg's bluegrass and spotted knapweed), while orange hawkweed responded negatively to the ash addition (Tables 18 and 19).

Table 18. Summary of mean ± 1 S.E. for total biomass by species grown in a controlled greenhouse experiment with soil from a slash pile burn, burned soil with a layer of ash, or control soil. Letters in rows indicate differences between treatments (α<0.05). Spotted knapweed and orange hawkweed were square root transformed.

	Control	Burned	Burned with Ash
PSSP	0.50 ± 0.01 a	0.61 ± 0.02 b	0.59 ± 0.02 b
FEID	0.32 ± 0.01 b	0.12 ± 0.17 a	0.32 ± 0.03 b
POSA	0.24 ± 0.02 b	0.14 ± 0.01 a	0.32 ± 0.03 c
CEMA	0.37 ± 0.02 a	0.56 ± 0.02 b	0.73 ± 0.06 c
EUES	0.22 ± 0.05 a	0.49 ± 0.03 b	0.44 ± 0.04 b
HIAU	0.39 ± 0.04 ab	0.46 ± 0.13 b	0.24 ± 0.07 a

Table 19. ANOVA for total biomass of species grown in a controlled greenhouse experiment with soil from a slash pile burn, burned soil with a layer of ash, or control soil.

	PSSP				FEID			
	d.f.	MS	F	P	d.f.	MS	F	P
Treatment	2	0.07	16.57	<0.01	2	0.13	27.70	<0.01
Error	27	0.00			27	0.01		
	POSA				CEMA			
	d.f.	MS	F	P	d.f.	MS	F	P
Treatment	2	0.84	15.20	<0.01	2	0.14	14.42	<0.01
Error	27	0.01			27	0.10		
	EUES				HIAU			
	d.f.	MS	F	P	d.f.	MS	F	P
Treatment	2	0.20	5.88	0.01	2	0.04	3.4	0.05
Error	27	0.04			27	0.01		

In contrast, the root: shoot was only occasionally influenced by soil substrate. Idaho fescue and Sandberg's bluegrass both had a lower ratio in burned soil relative to the other

treatments. The burned soil with ash resulted in the highest ratio for Sandberg's bluegrass. The burned treatment reduced root: shoot in Idaho fescue but were similar in the control and burned with ash treatments (Tables 20 and 21).

Table 20. Summary of mean ± 1 S.E. for root: shoot biomass ratio by species grown in a controlled greenhouse experiment with soil from a slash pile burn, burned soil with a layer of ash, or control soil. Letters indicate differences between treatments for each species (α<0.05).

	Control	Burned	Burned with Ash
PSSP	1.18 ± 0.7	1.0 ± 0.1	1.09 ± 0.1
FEID	0.71 ± 0.6 b	0.4 ± 0.1 a	0.83 ± 0.0 b
POSA	1.30 ± 0.1 b	0.7 ± 0.0 a	1.5 ± 0.1 c
CEMA	1.11 ± 0.1	1.1 ± 0.2	1.15 ± 0.1
EUES	1.35 ± 0.2	1.8 ± 0.2	1.81 ± 0.2
HIAU	1.48 ± 0.1	1.5 ± 0.1	1.13 ± 0.3

Table 21. ANOVA for root: shoot for species grown in a controlled greenhouse experiment with soil from a slash pile burn, burned soil with a layer of ash, or control soil. The Kruskal-Wallis non-parametric test was used for orange hawkweed

	PSSP				FEID			
	d.f.	MS	F	P	d.f.	MS	F	P
Treatment	2	0.09	1.24	0.28	2	0.40	14.70	<0.01
Error	27	0.07			27			
	POSA				CEMA			
	d.f.	MS	F	P	d.f.	MS	F	P
Treatment	2	2.02	54.77	<0.01	2	0.00	0.04	0.96
Error	27	0.04			27	0.11		
	EUES				HIAU			
	d.f.	MS	F	P	d.f.	Chi-square	Asymp.Sig	
Treatment	2	0.74	2.76	0.81	2	0.03	0.98	
Error	27	0.27						

Plant MIP was clearly influenced by soil substrate. MIP was negligible in both burned soil treatments. Conversely, when grown in unburned soil, non-native forb species

had a colonization level of 42.2 to 51.0%, while the graminoids ranged from 0.0 to 3.6% (Table 22).

Table 22. Summary of ± 1 S.E. for MIP by species grown in a controlled greenhouse experiment with soil from a slash pile burn, burned soil with a layer of ash, or control soil. Differences are indicated with an asterisk (Kruskal-Wallis).

	Control (%)	Burned (%)	Burned with Ash (%)
PSSP*	3.6 ± 1.3	0.0 ± 0.0	0.0 ± 0.0
FEID	0.0 ± 0.0	0.0 ± 0.0	0.0 ± 0.0
POSA	0.0 ± 0.0	0.6 ± 0.6	0.0 ± 0.0
CEMA*	51.0 ± 3.8	0.6 ± 0.3	0.0 ± 0.0
EUES*	45.6 ± 4.7	0.4 ± 0.3	0.7 ± 0.3
HIAU*	42.2 ± 5.4	0.4 ± 0.4	2.3 ± 0.0

Discussion

Restoration Effects on Soil Parameters

A key component of overall ecosystem sustainability occurs belowground; therefore recovery following fire is tied to the soil's physical, chemical, and biological functions and processes (Nearly et al. 1999). The high temperatures reached during slash pile burning altered soil biochemistry, and restoration treatments had an impact on several measured soil parameters. Soil temperatures were high enough (Chapter 2) to oxidize organic matter leaving a residual concentration of N, S, and basic cations, causing soil pH to increase. While addition of forest litter or commercial compost was expected to most effectively ameliorate soil pH, scarification alone was most effective. The organic matter used in the other two treatments may have had a buffering effect on the incorporated ash. Total C and N began to return to background levels where scarification with compost was applied. While C:N did not change with treatment, seeding lowered

C:N under the control and surface organic matter treatments. A lower C:N can result in higher quality litter and more rapid nutrient turnover (Smithwick et al. 2005), which may favor faster growing, shade intolerant early-successional species such as grasses and forbs over slow growing, shade tolerant late-successional species such as shrubs and trees.

Nitrogen cycling was moderately influenced by soil restoration treatments. Following fires, soil NH_4^+-N typically increases due to pyrolysis of forest floor material and subsequently nitrification increases, which results in a significant pulse of NO_3^--N (nearly ten times background levels) approximately one-year after the burn (Neary et al. 1999; Neary et al. 2005; DeLuca & Zouhar 2000; Choromanska & DeLuca 2001). While soil NH_4^+-N was moderated by three treatments (scarification, scarification with organics, and scarification with compost) as a result of NH_4^+-N immobilization, all treatments yielded greater NH_4^+-N than unburned plots. Soil PMN was greater in the scarification and organic matter treatment. N mineralization depends on many variables, but is closely tied to the quality of organic matter and soil moisture (Smithwick 2005). This indicates that the added organic matter directly increased N mineralization or it aided in the retention of soil moisture. Soil NO_3^--N was not altered by restoration treatments, but resin-sorbed NO_3^--N was greatest in the scarification and compost treatment, indicating that compost may have increased the abundance or activity of nitrifying bacteria. In this treatment nitrification exceeded that of unburned soils.

Seeding influenced NO_3^--N in the un-treated burn and organic matter treatments, thus plant uptake of NO_3^--N likely occurred. Seeding may be considered more than an

erosion control strategy, but also be a means of preventing NO_3^--N, the mobile N fraction, from leaching into the soil and possibly polluting waterways. The other primary consideration in water-quality protection is P. Water-soluble P was elevated in all treated soils except scarification, but only scarification with compost was greater than the un-treated burn. Compost may result in an artificially high P level, which could affect waterways, revegetation patterns, and even suppress AM fungal colonization where plants would exchange fixed C for P from AM fungi.

Reduced microbial biomass is common in soils exposed to fire as prescribed fire or wildfire (Pietikäinen & Fritze 1993; Vasquez et al. 2004; Hart et al. 2005). The microbial pool in soil is critical in litter decomposition, thus important in nutrient transformations and cycling, and maintaining site fertility (Pietikäinen & Fritze 1995; Hart et al. 2005). In this study, microbial biomass did not appear to recover after two growing seasons, indicating longer term effects on this soil parameter. None of the soil treatments recovered microbial biomass; however, scarification with organic matter or compost both increased the concentration above the untreated burn. Changes in the soil environment following burning may maintain long-term reductions in soil microbial biomass. Lower moisture contents, increased pH, altered soil structure, or reduced SOM as a food source may inhibit recolonization. Given that severe fire can impact the microbial community for as many as 12 years (Pietikäinen & Fritze 1995), any recovery is relevant from the perspective of slash pile scar restoration. Longer-term studies are needed to determine whether reduced microbial biomass results in a persistent change in plant communities.

Impact of Restoration on Vegetation

Burning slash as a management strategy may result in a persistent scar (Korb et al. 2004) or a plant community dominated by non-native species (Dickinson & Kirkpatrick 1987). The aim of this study was to identify restoration treatments that increase vegetative cover and suppress non-native invasive species. The results indicate that seeding a slash pile burned area may effectively increase cover over time. When seeded, the scarification or scarification with compost treatments further increased cover; however, without seeding there was no treatment effect after two years. Mean cover of the unburned areas was 42%. After one growing season, seeded plots receiving scarification, scarification with organic matter, and scarification with compost all exceeded the mean vegetative cover of unburned areas. After two growing seasons, all seeded treatments exceeded this benchmark. Since plots were adjacent to one another in a radial shape without a physical barrier, the treatment effect may have become diluted after two growing seasons via rhizomatous root encroachment into neighboring plots. Without the seed addition, no treatments reached the mean cover of unburned areas. Lack of viable seed was likely the limiting factor for plant establishment. Given that roots desiccate around 48 to 54 °C and the seed bank will be affected by temperatures ranging from 70 to 90 °C (Giovannini et al. 1988), the seed bank was likely impacted by the loss of litter and the humus horizon during soil heating, and seed density in the O horizon may be over four times greater than the A horizon in a forest ecosystem (Zabinski 2000). These results are corroborated by another slash pile burning study where viable plant seed was eliminated (Korb et al. 2004).

The colonization and establishment of non-native invasive species after disturbance may alter community structure and ecosystem function (George et al. 1995). Non-native invasive species may affect soil surfaces by changing germinations sites, affecting surface microclimates, and changing the amount and quality of litter resources, and root exudates made available to soil organisms (Pritekel et al. 2006). In this present study, non-native species were effectively suppressed by seeding; however, the impacts of restoration treatments were marginal by the second growing season. A subset of slash piles (n=4) were in close proximity to patches of leafy spurge, which sprouts from the root crown and roots after top-kill by fire. Furthermore, fire may increase leafy spurge density by promoting sprouting of previously dormant buds along the extensive rhizome and root system (Cole 1991; Masters 1995). Leafy spurge also has an extensive root system, has potential allelopathic properties, and the plant has high-starch latex that seals wounds (Pritekel et al. 2006). In this study, mean leafy spurge colonization of scars was 5.2% in seeded scars and 10.5% in non-seeded scars by the second growing season. The increase in N availability stimulated by fire coupled with an abundant viable seed source likely contributed to the establishment of native graminoids and aided in suppression of leafy spurge.

The surveys of percent cover by graminoids, forbs, and non-native forbs did not indicate that forbs or other groups were greatly suppressed by graminoid seeding. In general, forbs were slow to recover following slash pile burning. A survey of plant density was not likely to yield different results since forb percent cover was typically composed of one to two plants. Seeding with graminoids is an attempt at establishing

early successional species, with a restoration trajectory that moves from bunchgrasses to forbs to a woody shrub cover typical of inland northwest ponderosa pine forest understory (MacKenzie et al. 2004). Additional monitoring is necessary to ascertain whether species diversity is adversely impacted by slash pile restoration seeded with graminoids only, and to determine whether non-native invasive species continue to increase cover in seeded plots.

Mychorrhizal Inoculum Potential

Mycorrhizae may enhance the competitive ability of native bunchgrasses and thus play a role in plant invasion resistance (Goodwin 1992). Slash pile burning may dramatically alter the soil environment and reduce AM fungi, which can affect revegetation patterns (Reeves et al. 1979). A reduction in AM fungi following burning is normally thought to favor non-native species that are non-mycorrhizal (Haskins & Gehring 2004). In this study, all species grown were mycotrophic. The non-native species had the greatest inoculum potential in unburned soil, while graminoid inoculum potential was extremely low or absent (Table 21). Lower colonization levels in graminoids are common, but mycorrhizae may still enhance the competitive ability of bunchgrasses (Goodwin 1992). Higher rates of AM fungal colonization in non-native species compared to native grasses is consistent with other studies (Haskins & Gehring 2004). While mycorrhizae may enhance the competitive ability of native bunchgrasses, it may have a comparable or greater effect on non-native species (Goodwin 1992). For instance, Marler et al. (1999) found that when Idaho fescue was grown together with spotted knapweed,

the presence of AM fungi increased the competitive effects of spotted knapweed. Selosse et al. (2006) also note the potential for C transfer amongst plants via mycorrhizae.

Greater root biomass might be expected in species with high AM colonization (Haskins & Gehring 2004) or reduced root biomass may be found with higher AM colonization (Smith & Read 1997). This study shows that species with higher AM colonization (forbs in unburned soil) were associated with lower total biomass. AM fungi can be parasitic during the early life stages of a plant prior to developing an extensive hyphal network capable of accumulating resources. Limited light availability in the greenhouse may also be a factor. The root: shoot data did not correlate with MIP. Since AM colonization was very low in all burned treatments, other soil properties, such as nutrients or soil physical properties, likely influenced the observed differences. Thus, there is no evidence suggesting that the absence of AM propagules might provide a competitive advantage to non-native species, the mycorrhizae present on the site may, however, improve survivorship of facultative non-native species as AMF begins to recover in slash pile scars. In this study, all the non-native invasive forbs increased in biomass when grown in burned soil, while the native grasses each responded differently to burned soil. The non-natives may indeed have benefited from the flush of mineralized nutrients following burnings. However, since AM colonization of the non-natives was very low in burned soils, the increased biomass of plants in burned soil may suggest that the non-natives were released from a parasitic relationship with AM fungi.

Future studies should examine the role of AM fungi in a recently thinned ponderosa pine forest to determine whether understory vegetation depends on AM

associations for growth. The reduction in AM propagules following slash pile burning may be important. However, it should also be recognized that canopy and soil disturbance prior to the slash pile burn may shift the role of AM propagules. There is insufficient evidence to suggest that the post-forest thinning herbaceous community is strongly mycorrhizal.

CONCLUSIONS

Forest management activities such as fire exclusion have modified the structure and ecological processes in ponderosa pine (*Pinus ponderosa*) and Douglas-fir (*Pseudostuga menziesii*) forests in western USA. Restoration and fuel reduction treatments employing thinning and slash pile burning are common as managers attempt to move forest structure and density toward a more stable, fire-tolerant condition. Ponderosa pine tree ring records suggest low severity fire was most common in dry forests and more severe fire occurred in moist forests, though environment and regional climate dictated variability in fire severity and frequency (Hessburg et al. 2007). Many ponderosa pine forests were characterized by a shorter fire return interval (Arno et al. 1995; Choromanska and DeLuca 2001; Sala 2005), with localized weather patterns driving fire frequency and variability (Heyerdahl 2008). Frequent fire is thought to have maintained open stands dominated by clumps of mature ponderosa pine (Laughlin et al. 2004). Douglas-fir has become more prevalent where fire is excluded (Heyerdahl et al. 2006); and today, dense patches of Douglas-fir in the understory create ladder fuels that increase the risk of stand-replacing wildfire. High-intensity fire can have negative impacts on dry ponderosa pine forests, which are adapted to low-intensity fire regimes (Certini 2005). Land managers are now considering reducing fuel loading in ponderosa pine forests to mitigate the risk of high-intensity wildfire that threatens forest ecological function as well as endangering homes located in the wildland-urban interface.

With fuels reduction as the primary directive of the U.S. Forest Service under the Healthy Forests Restoration Act of 2003 and 8 million ha of forest land eligible for fuels

reduction, proper management of harvest residue is imperative. Slash piling and burning is still the primary method of disposal of forest harvest residue. Where forest restoration is the goal, studying its impacts on soil nutrient capital and biochemical properties is requisite.

No reported studies have examined the impacts of small-scale slash pile burning on vegetative composition and soil biochemical properties, and there are no slash pile burn studies to date in Inland Northwest ponderosa pine ecosystems. Several soil parameters corroborate with other fire studies, such as the increases in inorganic N (Neary et al. 1999; Neary et al. 2005; DeLuca & Zouhar 2000; Choromanska & DeLuca 2001); soil PO_4^{3-}-P results were similar to another slash pile burn study (Korb et al. 2004) and did not vary with burning, while microbial biomass was dramatically lower after one-month and did not recover within a year, which is supported by Haskins & Gehring (2004). In contrast to past fire studies, particle size distribution was not affected by burning (Arocena & Opio 2003) and total N was unchanged (Korb et al. 2004). In addition, PMN was lower in the burned center, but greatest near the edge of the burn. This indicates N cycling was stimulated by fire, but the total pool of N was not completely volatilized or mineralized. Given the small size of the slash pile scars and the spatial variability of natural low-intensity fire, there is limited indication that the slash pile burns resulted in a disturbance outside of the range of natural variability.

Documentation of slash pile scar restoration is rare, particularly for small-scale burns. Scarification with on-site organic matter and scarification with commercial compost both mitigated soil parameters. Seeding was the most important factor in

establishing plant cover and suppressing non-native invasive species. Where there is a perceived risk of soil erosion, nutrient loss, or non-native plant invasion, scarification or scarification with commercial compost may assist with rapid plant establishment in the first growing season. After two years, none of the soil restoration treatments had a great impact on plant cover composition; however, long-term monitoring of species composition would provide more insight on the impact of restoration on plant community dynamics. In the greenhouse experiment, all the non-native invasive forbs increased in biomass when grown in burned soil as a result of the flush of mineralized nutrients or the release from AM fungal colonization. While only three graminoids were examined, the low AM colonization of bunchgrass species may suggest that the role of AM fungi is diminished in fire suppressed, shaded ponderosa-pine/Douglas-fir forests. Forest restoration thinning may, over time, promote AM fungal activity following the increase in light availability. This question deserves further investigation.

Where fire is used to manage forest harvest residues, knowledge of its effects on total nutrient capital and on nutrient dynamics is desirable. The complex nature of fire, particularly in the heterogeneous soil environment makes interpretation of the effects on fire soil nutrients challenging. Many of the findings of this study, however, are supported by previous studies involving high-severity fire or slash pile burning, thus may have application where slash piles are of comparable fuel loading and located in similar forest ecosystems.

When the goal of a forest restoration thinning is to restore ecosystem structure and function, there is a need for a more complete assessment of ecosystem impacts

related to the logging operation and disposal of forest harvest residue, which may be necessary prior to reintegrating prescribed fire. Burning residues on a landing or road would help concentrate impacts to a more confined area, though hauling the residues to a road for burning or off-site transport may require increased activity of heavy equipment in the forest stand and result in widespread soil compaction. Several large slash piles within the stand would also require transporting residue a greater distance, possibly causing compaction over a greater area and more severe burning impacts including larger scars. In this study, forest residues were piled by hand and tractor into numerous, small-scale slash piles, which mitigated burn intensity and reduced the need to haul slash residue over long distances.

There were many indicators of ecosystem function measured in this study and only some were impacted by the disturbance of slash pile burning, some of these recovered with restoration and other did not. The stated concern over slash pile burning often relates to soil erosion, nutrient losses, or non-native plant establishment. In this study, seeding resulted in high a rate of vegetative cover, thus ameliorating the commonly stated indicators of disturbance following slash pile burning. This is accomplished because vegetative cover will reduce soil erosion, reduce nutrient losses through plant uptake (particularly NO_3^--N), and help suppress non-native species.

Burning on-site using numerous, small-scale slash piles may allow for some of the ecological benefits of fire. Fire in ponderosa pine forests is a mosaic of intensity and spatial variability (Hessburg et al. 2007). In this study, fire intensity was high in the center of slash piles and lower over the perimeter of the burn where the Oi/Oa horizons

were charred, which may mimic this mosaic pattern. In northern ponderosa pine/Douglas-fir forests, forest harvest debris contains a greater proportion of forest nutrient capital than the merchantable wood (Laiho & Prescott 2004) and transporting all of the harvest debris off-site may effectively mine the forest of nutrients, particularly N and P. Removal of non-merchantable debris without fire may increase C:N and slow decomposition rates over time and extensive periods of fire exclusion may result in N-limiting conditions (Kimmins 1996; Wardle et al. 1997). In addition, the stand would not experience the soil nutrient releases stimulated by fire such as micronutrients, increased cation exchange capacity, charring of organics (Certini 2005), and the production of charcoal (Berglund et al. 2004). The slash pile burns in this study presumably stimulated nutrient cycling as evidenced by increases in inorganic N. Even unburned areas may have benefited from N deposition as a result of slash pile burning in adjacent areas. In addition, charcoal was deposited within the slash pile burns, which is likely to be partially dispersed throughout the site by wind and water erosion.

In the case of forest restoration or thinning, numerous, small-scale slash piles may represent a compromise between the ecological ideal of prescribed burning and the need for land managers to control fire for human safety in the wildland-urban interface. This investigation suggests that with careful pre- and post-harvest management, the impacts of burning slash piles may be minimized and even provide ecological benefits.

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